MINNESOTA

LAKE WATER QUALITY

ASSESSMENT REPORT:

DEVELOPING NUTRIENT CRITERIA

Third Edition

Minnesota Pollution Control Agency

September 2005

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Minnesota Pollution Control Agency

September 2005

To Whom It May Concern:

Enclosed for your information is a copy of the report Minnesota Lake Water Quality Assessment Report: Developing Nutrient Criteria. Third Edition. This report serves as the technical basis for Minnesota's proposed draft lake nutrient criteria.

Minnesota's previous Lake Water Quality Assessment (LWQA) reports (1988 and 1990) were developed as a requirement for participation in Clean Lakes Program (Section 314 of the Clean Water Act of 1987). The first two editions described regional patterns in lake water quality in Minnesota and served as a basis for developing ecoregion-based phosphorus criteria. The reports have long provided a basis for assessing the water quality of Minnesota's lakes and the criteria have been used extensively for water quality goal setting and related activities.

As a part of the Clean Water Action Plan of 1997, nutrients were identified as a significant national problem and U.S. Environmental Protection Agency was requested to develop a National Nutrient Strategy. One aspect of this strategy recommended that states develop ecoregion-based criteria for total phosphorus, total nitrogen, chlorophyll-a, and Secchi transparency.

This edition builds on the previous LWQA reports and provides a detailed description of our approach for setting lake nutrient criteria. The draft criteria presented herein have been developed based upon multiple sources of information, including: reference lake data, statewide lake data, historic reconstruction of lake water quality from fossil algae in lake sediments, lake user perceptions, fishery and macrophyte requirements and other factors. Draft criteria are presented and examples of how the criteria may be used to further lake management are included in this edition of the LWQA.

If you have any questions or need additional copies of the report, please contact Steven Heiskary, Environmental Analysis and Outcomes Division, by phone at (651) 296-7217 or (800) 657-3864 or by e-mail at *steven.heiskary@pca.state.mn.us*. Copies are also available at: *http://www.pca.state.mn.us/water/lakequality.html#reports*.

Sincerely,

Michael J. Sandusky
Division Director
Environmental Analysis and Outcomes Division

MJS:jae

Enclosure

MINNESOTA

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ASSESSMENT REPORT:

DEVELOPING NUTRIENT CRITERIA

Third Edition

Written and prepared by:

Steven A. Heiskary
Water Assessment & Environmental information Section
Environmental Analysis & Outcomes Division
and
C. Bruce Wilson
Watershed Section
Regional Division

MINNESOTA POLLUTION CONTROL AGENCY

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Minnesota Pollution Control Agency

Acknowledgments

This report is based in large part on the previous MLWQA reports from 1988 and 1990. Contributors and reviewers to the 1988 report are noted at the bottom of this page. The following persons contributed to the current edition.

Report sections:

Mark Ebbers – MDNR, Division of Fisheries Trout and Salmon consultant:
Stream Trout Lakes report section

Reviewers:

Dr. Candice Bauer – USEPA Region V, Nutrient Criteria Development coordinator

Tim Cross – MDNR Fisheries Research Biologist (report section on fisheries)

Doug Hall – MPCA, Environmental Analysis and Outcomes Division

Frank Kohlasch – MPCA, Environmental Analysis and Outcomes Division

Dr. David Maschwitz – MPCA, Environmental Analysis and Outcomes Division

Word Processing – Jan Eckart

Contributors to the 1988 edition:

MPCA – Pat Bailey, Mark Tomasek, & Jerry Winslow

Manuscript review of 1988 edition:

MPCA – Carolyn Dindorf, Marvin Hora, Gaylen Reetz, Curtis Sparks & Dr. Ed Swain

MDNR – Jack Skrypek, Ron Payer, Dave Pederson & Steve Prestin

University of Minnesota – Dr. Robert Megard, Dr. Joseph Shapiro & Dr. Patrick Brezonik

St. Anthony Falls Hydraulic Laboratory – Dr. Heinz Stefan & Dr. Mike Riley

Kent State University – Dr. Dennis Cooke

USEPA-Environmental Research Laboratories (Corvallis) – Sharon Clark, James Omernik, & Thom Whittier; (Duluth) Dr. Gary Glass

Metropolitan Council – Dick Osgood

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EXECUTIVE SUMMARY

The Clean Lakes Program was reauthorized in Section 314 of the Water Quality Act of 1987. Reauthorization added several new requirements, including a Minnesota Lake Water Quality Assessment Report (MLWQAR). The assessment report was required for state participation in the Clean Lakes Program and as part of the 305 (b) report. Minnesota met this requirement with its 1988 305 (b) report, "Minnesota Water Quality, The 1988 Report to the Congress of the United States" (MPCA, 1988) and the MLWQAR (Heiskary and Wilson, 1988).

We decided to update and expand the MLWQAR to:

- augment information presented in the 305 (b) report,
- provide new information on Minnesota lake water quality;
- provide a framework for future lake assessments,
- provide a useful applied limnological reference for Minnesota lake resource managers,
- and in this edition, provide the basis for lake nutrient (eutrophication) criteria development as required by USEPA.

This report is organized into an introduction, four chapters, and appendices. The appendices include: a) listing of ecoregion reference lakes; b) listing of sediment diatom-reconstruction lakes; c) listing of designated lake trout lakes; and d) a series of individual lake case studies. A brief description of the chapters follows.

<u>Background and Database Development</u>

This chapter describes the databases used in this report and includes sampling, analytical, and data analysis procedures. Included among the databases are: ecoregion reference lakes, overall statewide 305(b) assessed lakes (2004 assessment), EPA's nutrient criteria databases, and estimates of pre-European phosphorus concentrations based on sediment diatom reconstruction.

<u>Considerations for Lake Water Quality Criteria Development</u>

This chapter provides a general discussion of the criteria-setting process and the variables that should be considered when establishing criteria or goals for Minnesota lakes. Information in this chapter was derived from previous and ongoing analyses of Minnesota lakes and literature review. The data presented and methods used are intended to serve as a basis for evaluating lake resources in Minnesota and establishing appropriate criteria.

The ecoregion framework is used extensively in this chapter. Among the issues addressed are patterns in lake trophic status in Minnesota and role of lake morphometry and land use as they contribute to these patterns. Important links are made among phosphorus, chlorophyll-a, transparency, and hypolimnetic oxygen. Impacts of eutrophication on fisheries and drinking water supplies are also addressed. The expectations of lake users are considered and linked with quantitative measures of lake water quality. Empirical models are presented that incorporate the aforementioned information.

Lake Pollution Control Programs in Minnesota

This chapter provides an overview of current programs for protecting or improving the water quality of Minnesota lakes, including regulatory, grant-management, and education programs. Nutrient criteria can be used in a variety of ways in these programs and some examples include:

- prioritizing and selecting projects to be funded through the Clean Water Partnership (CWP) and Section 319;
- basis for Section 303(d) listing of impaired waters;
- developing water quality management plans;
- communicating what can reasonably be expected in terms of lake quality; guiding regulatory agencies;
- and interpreting nondegradation statutes.

Deriving Eutrophication Criteria by Ecoregion

This chapter summarizes the previous discussions and provides a basis for selecting appropriate criteria for water quality protection or goal setting for restoring lake water quality. Individual case studies are included for each region that demonstrate how criteria may be applied to a range of lakes and lake condition.

An important aspect of the criteria-setting process requires the definition of "most sensitive sub-uses" of lakes. In this context, we have defined a sensitive sub-use of a lake as that use (or uses) which can be affected or even lost as a result of an increase in the trophic status of the lake. Two examples of sensitive uses include coldwater fisheries and primary contact recreation (aquatic recreation use support). In the case of a coldwater fishery, increased nutrient loading will result in a reduction of oxygen in the hypolimnion, and die-offs of coldwater species may occur as these populations are driven into warmer surface waters. In the case of aquatic recreational use, excess phosphorus stimulates the production of algae growth that can lead to frequent and severe nuisance blooms and reduced transparency that will limit use of the resource. Most sensitive uses have been identified for each region, and appropriate phosphorus, chlorophyll-a and Secchi criteria are noted. These criteria are ecoregion-based and reflect several considerations, including: reference lake condition; assessed lake condition; background trophic status based on diatom reconstruction of phosphorus; interrelationships among phosphorus, chlorophyll-a, Secchi and nuisance bloom frequency; lake morphometry; lake-user perception; lake ecology (fishery composition and rooted macrophytes); and appropriateness as reflected by overall characteristics of the ecoregions and assessed trophic status for each ecoregion. Subsequent discussion describes considerations for applying the criteria and related information. Case studies, included in the Appendix, provide examples of how the criteria may be applied for goal setting, lake protection and 303(d) listing purposes.

Proposed eutrophication criteria by ecoregion and lake type.

Ecoregion	TP	Chl-a	Secchi
	ppb	ppb	meters
NLF – Lake trout (Class 2A)	< 12	< 3	> 4.8
NLF – Stream trout (Class 2A)	< 20	< 6	> 2.5
NLF – Aquatic Rec. Use (Class 2B)	< 30	< 9	> 2.0
CHF – Stream trout (Class 2a)	< 20	< 6	> 2.5
CHF – Aquatic Rec. Use (Class 2b)	< 40	< 14	> 1.4
CHF – Aquatic Rec. Use (Class 2b) Shallow lakes	< 60	< 20	> 1.0
WCP & NGP – Aquatic Rec. Use (Class 2B)	< 65	< 22	> 0.9
WCP & NGP – Aquatic Rec. Use (Class 2b) Shallow lakes	< 90	< 30	> 0.7

Minnesota Lake Water Quality Assessment Report: Developing Nutrient Criteria

I. INTRODUCTION

Excess nutrients, specifically phosphorus and nitrogen, have long been recognized as the primary pollutants that contribute to the cultural eutrophication of lakes. This fact is commonly stated in Minnesota's 305(b) Report to Congress, as well as those from other states across the nation. However, even though they are a significant pollutant, as of 2000, there were no Federal (and few State) water quality criteria for nutrients for the purposes of protecting waters from eutrophication. As a part of the Clean Water Action (CWAP) in 1997, nutrients were identified as a significant national problem and USEPA was requested to develop a National Nutrient Strategy. One aspect of this strategy recommended that states develop ecoregion-based criteria for total phosphorus, total nitrogen, chlorophyll-a and Secchi transparency. Complete details and background on this strategy can be found in several USEPA publications (USEPA 1998).

Several actions were initiated as a part of this effort. USEPA organized regional groups of representatives to serve on Regional Technical Assistance Groups (RTAG). The RTAGs provided a basis for sharing information on state approaches to nutrient criteria development, developing a more region-wide approach where possible and keeping states informed of USEPA developments on this issue. Minnesota has been involved with the Region V RTAG since its inception.

Another major task undertaken by USEPA was the development of technical guidance manuals for each waterbody type: lakes and reservoirs, rivers and streams, wetlands and estuaries. A group of experts was assembled and the documents were developed. The intent of the documents was to provide background on the issue of criteria development and specific information on approaches that have or might be used to develop nutrient criteria. Given the vast amount of information available on the impact of excess nutrients on lentic systems the first document drafted was "Nutrient Criteria Technical Guidance Manual: Lakes and Reservoirs" (USEPA, 2000a). The manual presents good background information on nutrient impacts on lakes and reservoirs, process of developing criteria, and approaches that might be used. Minnesota's ecoregion-based approach was noted prominently in this document (e.g., Heiskary and Wilson, 1989).

The next requirement of the National Nutrient Strategy was the development of "Ambient Water Quality Criteria Recommendations" by USEPA (2000b). These documents are waterbody-type and ecoregion specific data summaries. Lake and reservoir documents are available for the three "nutrient" ecoregions that encompass Minnesota. These documents provide statistical summaries of TP, TN, Secchi and chlorophyll-a data based on available data in Legacy STORET and other readily accessible sources such as National Water Quality Assessment (NAWQA) and National Stream Quality Assessment Network (NASQAN). Data collected from January 1990 to December 1999 were used in the statistical analysis. Statistics are supplied both for the "aggregated" level III ecoregions as well as the individual level III ecoregions that we have routinely referred to in Minnesota. These documents are intended as a starting point for nutrient criteria development – they describe a process for developing criteria, provide a statistical overview of the resource base, and provide one option on how reference condition might be determined based on these populations. Since Minnesota had previously developed a "reference lake" database the USEPA data served primarily as an additional basis for understanding the within region distribution of these parameters.

This document will draw heavily and in some cases directly from several previous publications on this topic (e.g., Heiskary and Wilson 1989) and is based largely on the previous Minnesota Lake Water Quality Assessment reports (LWQA) (Heiskary and Wilson 1988 and 1990). These reports were originally developed in response to a requirement of the Clean Lakes Program as reauthorized in Section 314 of the Water Quality Act of 1987. This assessment was required for state participation in the Clean Lakes Program and as a part of the 305 (b) report. Minnesota met this requirement in its 1988 305 (b) report – "Minnesota Water Quality, the 1988 Report to the Congress of the United States" (MPCA, 1988). We have updated the LWQA database on a bi-ennial basis since that time and routinely place the data summary on the MPCA Web site: http://www.pca.state.mn.us/water/lakequality.html . New information and data will be added where appropriate but, in general, our current approach to ecoregion-based criteria development (in support of USEPA's requirement that states develop nutrient criteria) is essentially unchanged from our efforts to develop P criteria in the late 1980's.

The report is organized into five chapters and an appendix, which includes case studies that demonstrate applications of nutrient criteria. A brief description of the subsequent four chapters is as follows:

<u>Background and Database Development</u> – This chapter describes the data bases used in the context of this report and includes sampling, analytical and data analysis procedures.

<u>Considerations for Lake Water Quality Criteria Development</u> – This chapter provides a general discussion of the criteria-setting process and the variables that should be considered when establishing criteria or goals for Minnesota lakes. The information in this chapter was derived from previous and ongoing analyses of Minnesota lake data. The data presented and methods used in this chapter are intended to serve as a basis for evaluating lake resources in Minnesota and establishing appropriate criteria.

<u>Lake Pollution Control Programs in Minnesota</u> – This chapter provides an overview of current programs for protecting or improving the water quality of lakes in Minnesota, including regulatory, grant-management and education programs. The use and application of lake water quality criteria relative to these programs is noted.

<u>Deriving Eutrophication Criteria by Ecoregion</u> – This chapter summarizes the previous discussions and brings together the information used for criteria development. The criteria have been developed based upon multiple sources of information including: reference data, ecoregion-wide assessed data, diatom-reconstructed phosphorus concentrations, user perception information, fishery and macrophyte considerations, and other factors. Individual case studies showing how phosphorus criteria may be derived and applied to individual lakes are included for each region.

II. BACKGROUND AND DATABASE DEVELOPMENT

Several data bases are referred to in the context of this report. Brief descriptions of the four primary databases: assessment, ecoregion reference, diatom-inferred phosphorus and USEPA criteria development follow. Each database is important to our overall assessment of Minnesota lakes and our criteria development efforts. Water quality data from all databases may be found in STORET. Relevant field and laboratory methods and quality assurance information, which applies to the three Minnesota databases, are summarized in the following section.

Field and Laboratory Methods

Water quality data were generally collected during the open water season (May-November). Sampling stations were located at mid-lake at the greatest lake depth. Surface samples were generally collected with a PVC tube 6 feet in length (~2 m) with an inside diameter of 1¼ inches (integrated samplers) that collects a sample from the upper 2 m of the lake. Samples were acid preserved at the time of collection (MPCA) or within six hours of collection (MCES). The Minnesota Department of Health (MDH) analyzed samples collected by MPCA. The MCES laboratories analyzed samples collected by the Metropolitan Council.

The most commonly employed methods of analyzing total phosphorus were colorimetric, automated block digester for MPCA and acid digestion and modified ascorbic acid reduction method for the Metropolitan Council. (Methods 365.4 and 365.3, respectively, in U.S. Environmental Protection Agency 1979.) At the time of the original reference lake work (mid 1980s) detection limits were 10 µg/1 for MPCA and Metropolitan Council (Heiskary et al. 1987). Mean precision for the MPCA data was 4.9 µg/l based on 10 percent duplicate analysis. MCES reported a mean standard deviation of 7 µg/l based on samples collected in triplicate. Accuracy, expressed as percent recovery, was 104 percent at a concentration of 20 µg/l and 101 percent at a concentration of 40 µg/l for MPCA. Accuracy was not reported for Metropolitan Council data. More recently (1990s) the detection limit for MDH TP analysis was changed to 2 µg/L.

Chlorophyll-a samples were chilled and kept in the dark immediately after collection. Samples were filtered through 4.5 µm diameter glass fiber filters within 8 hours of collection and kept frozen until analyzed. Samples for MPCA were analyzed by spectrophotometer and corrected for pheophytin, according to Standard Methods (APHA, 1980). MCES samples were not corrected for pheophytin but represent “viable chlorophyll-a” according to their methodology for data collected in the 1980s through mid 1990s. MCES began to run both corrected and uncorrected chlorophyll-a in the late 1990s. Based on comparisons of corrected chlorophyll-a and uncorrected chlorophyll-a for 4,500 data pairs from 2002-2004 corrected chlorophyll-a accounted for 88% of uncorrected chlorophyll-a, i.e., uncorrected values may be on the order 12% higher than corrected (Anhorn, 2005, personal communication). In general we recommend the spectrophotometric measurement of corrected chlorophyll-a whenever possible as this is what most of our work (data analysis) is based on.

An example of some relevant quality assurance information, as drawn from a recent report (Heiskary and Markus, 2001), is included below. While this information was assembled from a river study it is applicable to lakes as well.

Laboratory methods and precision estimates for Minnesota river-nutrient study (Heiskary and Markus, 2001).

Parameter	Reporting Limit & Units	EPA method number	Precision: [1] mean difference	Difference as Percent of observed
Total Phosphorus	10.0 µg.L-1	365.2	4.8 µg.L-1	2.7 %
Total Kjeldahl N	0.1 mg.L-1	351.2	0.05 mg.L-1	2.8 %
$NO_2 + NO_3$	0.01 mg.L-1	353.1		
Total Suspended Solids	0.5 mg.L-1	160.2	2.8 mg.L-1	9.6 %
Total Suspended Volatile Solids	0.5 mg.L-1	160.4	--	--
Turbidity	0.2 NTU	180.1	--	--
BOD5	0.5 mg.L-1	405.1	0.15 mg.L-1	6.6 %
Chlorophyll-a	0.16 µg.L-1	446.0	1.7 µg.L-1	7.4 %
Pheophytin	0.27 µg.L-1	446.0	--	--

[1] Average of individual means of 10 duplicates and expressed as a % of measured concentrations.

Assessment Database

The first is referred to as the assessment database. This database includes all lake stations in STORET (in A = 21 MINNL) with data for one or more of the trophic status variables, i.e., TP, chlorophyll-a or Secchi transparency and includes data for approximately 2,790 lakes, collected during the past 32 years (1970-2002) (Figure 1). The data were collected by the Minnesota Pollution Control Agency (MPCA), Metropolitan Council Environmental Services (MCES), Minnesota Department of Natural Resources (MDNR), citizen volunteers, and numerous other entities such as local water plan and coalition of lake association monitoring efforts. The majority of the Secchi and “user perception” data was obtained through the MPCA’s Citizen Lake-Monitoring Program (CLMP). This database was used in our most recent 305(b) lake water quality assessment (MPCA, 2004) that is available on our Web site. Much of the original data analysis (e.g., Heiskary and Wilson, 1988) was conducted in the Statistical Analysis System (SAS), while Microsoft Access and Excel were used primarily in this current edition of the LWQA.

All 305(B) assessed lakes (Fig. 1) were classified into one of two categories, either monitored or evaluated, as follows:

Monitored – Lakes with “summer” data collected between calendar years 1993 through 2002 (inclusive), with summer defined as the time period between June – September. Summer data are preferred for assessment purposes as they generally correspond to maximum productivity of the lake, yield the best agreement between trophic variables, and reflect the period of maximum use of the resource. Summer means and supporting statistics were calculated for each variable and are available on the MCPA web site at http://www.pca.state.mn.us/water/lkwqSearch.cfm .

Evaluated – Lakes not qualifying as monitored (e.g., data collected prior to 1993) but with total phosphorus, chlorophyll-a and/or Secchi transparency measurements collected between 1970-1992. Summer season is used for calculating mean chlorophyll-a and Secchi transparency. Mean total phosphorus was calculated from data collected during the "open water" season (May – November). Expanding the "season" for total phosphorus measures allowed for the inclusion of a large number of lakes in northern Minnesota that were sampled only during spring or fall turnover as a part of our acid rain lake monitoring efforts.

Trophic Status was assessed for each lake in the assessment data base using Carlson's Trophic State Index (TSI) (Carlson, 1977). This index was developed from the interrelationships of summer Secchi transparency, and epilimnetic concentrations of chlorophyll-a and total phosphorus (Fig. 3). TSI values are calculated as follows:

Secchi disk TSI (TSIS) = 60 – 14.41 ln (SD);

Total phosphorus TSI (TSIP) = 14.42 ln (TP) + 4.15;

Chlorophyll-a TSI (TSIC) = 9.81 ln (Chl-a) + 30.6;

with chlorophyll-a and total phosphorus in µg/l and Secchi disk in meters.

The resulting index values generally range between 0 and 100 with increasing values indicating more eutrophic conditions (with the exception that due to some very high TP measurements some of the TSIP values will exceed 100). The trophic states for the index are defined by using each doubling of Secchi transparency as the criterion for the division between each state, i.e., each time the transparency doubles from some base value a decrease in TSIS of 10 units occurs and a new trophic state will be recognized. Because the relationship between Secchi and total phosphorus is a simple inverse function, a doubling of total phosphorus causes TSIP to increase by 10 units.

Ecoregion Reference Lake Database

A second database is referred to as the reference lake database, which is a subset of the assessment database. The aquatic ecoregion framework (Omernik 1987) was used as a basis for selecting these lakes and analyzing the data. Though most of this work was conducted in the mid to late 1980's it is still considered relevant to our current efforts to develop nutrient criteria and has been used as one example of an approach for developing a reference lake data set in recent USEPA guidance (USEPA, 2000a).

The reference lake database is comprised of approximately 90 lakes distributed among the four ecoregions that contain 98 percent of Minnesota's lakes by number (Figure 3; Appendix I). Lakes included in this database were representative, considered to be minimally impacted, and serve as reference lakes for their respective ecoregions. Regional reference sites are sites (lakes) with watersheds characterized by regionally predominant landscapes that are minimally impacted by point and nonpoint sources of pollution (Hughes and Larsen, 1988). Factors such as maximum depth, surface area and fishery classification were considered in selecting the lakes. Lakes with known point sources, major urban areas and/or major feedlots in their watersheds were excluded. Recommendations from MDNR area fishery manager were also instrumental in selecting these lakes. Data from these lakes was initially used for developing regional goals as noted by Hughes and Larsen (1988).

Analysis of Minnesota's lakes focused on those ecoregions with the majority of the states' lake resources. These ecoregions are the Northern Lakes and Forests (NLF), North Central Hardwood Forests (CHF). Western Corn Belt Plains (WCP), and Northern Glaciated Plains (NGP). Within these ecoregions, representative lakes in minimally impacted watersheds were selected for sampling (Fig. 2). For example, in the southern ecoregions agricultural land use predominates across the ecoregion and in the reference lake watersheds. For the southern ecoregions minimally impacted simply suggests watersheds without major urban areas, known point sources or numerous feedlots. Whenever possible lakes were selected in the "most typical" portions of the ecoregions, as defined in Omernik (1987).

Following are some of the considerations used to select "representative" and "minimally impacted" (reference) lakes for each ecoregion:

1. Data in the STORET were assessed by ecoregion to ascertain ranges and distributions (similar to Table 1) for total phosphorus, Secchi disk, surface area, and maximum depth (Heiskary et al. 1987). This analysis provided information for steps 2 and 3.
2. Further review was conducted to identify lakes that may be at or near the "background" phosphorus concentration for each ecoregion. Application of the morphoedaphic index (MEI), as reported by Vighi and Chiaudani (1985) was used to assist in this process.
3. Sought to select lakes that were <u>representative</u> in terms of size, depth and mixing status, e.g., extremely shallow or extremely deep lakes were excluded based on the analysis in #1.
4. Selected a range in lake drainage patterns, e.g. closed lake basins, lakes with outlets, and lakes with inlets and outlets. Large flowage lakes or reservoirs that may drain more than one ecoregion were excluded.
5. Selected representative lakes with respect to Minnesota Department of Natural Resources (MDNR) ecological fishery classification. The following is an example:

- lake trout:	NLF
- walleye:	NLF, CHF
- bass/panfish:	NLF, CHF, WCP, NGP
- marginal fishery: waterfowl or rough fish	NLF, CHF, WCP, NGP

6. Selected a few high resource value lakes from each region. This selection was based on MDNR or MPCA staff recommendations and feedback from the public.
7. Excluded lakes from areas with ongoing data collection efforts by other entities (e.g. seven county metro area where there exists substantial data generated by the MCES).
8. Miscellaneous considerations included "clustering" of lakes to increase sampling efficiency, existence of adequate public access, etc.

The reference lakes were generally sampled three to four times each during the summers of 1985, 1986 or 1987. Some lakes were sampled in all three years and allow for characterization of year-to-year fluctuation of the trophic variables. Subsets of these lakes have been sampled since that time and this data allows for further assessment of year-to-year variability. In the case of the NGP reference lakes, a comprehensive effort to revisit these lakes in 2002 (Heiskary et al. 2003) allowed us to modify the typical ranges of water quality for that region (Table 2).

Water sampling focused on summer conditions and generally ran from mid-June to mid-September. At least two sites per lake were sampled. The following parameters/analyses were conducted at the primary site (generally over the point of maximum depth in the lake):

- Secchi disk transparency
- Dissolved oxygen (DO) and temperature profile
- Chlorophyll a (2 meter integrated samples)
- General chemistry of epilimnion (2 m integrated sample), including total suspended solids, pH, alkalinity, color, turbidity, conductivity, chloride, and nitrite + nitrate nitrogen
- Total phosphorus and total Kjeldahl nitrogen from epilimnion and hypolimnion

At a secondary site(s) Secchi disk transparency, chlorophyll a, and epilimnetic total phosphorus and total Kjeldahl nitrogen were measured. All chemistry samples were analyzed by the Minnesota Department of Health (MDH).

Lake morphometric information (surface area and mean depth) was gathered with assistance from U.S. EPA Corvallis Environmental Research Laboratory (ERL). Corvallis ERL employed the modified Omernik method (Omernik and Kinney, 1982) for determination of mean depths. Lake surface area was done by planimetry. Lake volume was calculated as follows:

Volume = mean depth x surface area.

MPCA determined lake volumes through planimetry of bathymetric maps according to Lind (1974). Where lake volume = frustrum volumes and:

$$\text{Frustrum Volume} = \frac{h}{3}\left(a_1 + a_2 + \text{sq root}(a_1\, a_2)\right)$$

h = depth of frustrum
a_1 = area of frustrum surface
a_2 = area of frustrum bottom

An analysis of the comparability of these two methods may be found in Liukkonen et al. (1985).

The number of inlets and outlets was determined by inspection of USGS quadrangles and field observation. This information was used to define the hydrologic or drainage type of each lake as adapted from Eilers et al. (1983).

A lake with no inlets or outlets was considered seepage, a lake with no inlets and an intermittent outlet was considered drained, a lake with no inlets and a permanent outlet was considered headwater, a lake with both inlet(s) and outlet(s) was considered drainage, and a lake with inlet(s) but no outlet was considered an inflow lake (Liukkonen, 1985).

Lake mixing status, or mixis as it is also referred to, was determined by inspection of dissolved oxygen and temperature profiles as follows: Lakes that were stratified all summer or on at least two consecutive (monthly) sampling dates were considered dimictic. Lakes exhibiting well mixed conditions on all dates were considered polymictic. Lakes that were intermittently stratified were classed as intermediate for the purposes of this study.

Total watershed areas were delineated on USGS quadrangles. U.S. EPA Duluth ERL assisted in this effort. Watersheds were digitized into the LMIC computer system and land use composition derived from 1968-69 LMIC 40-acre statewide land use data. This required conversion from an EPPL6 Raster file to an ARC/INFO vector file for analysis (Baker, 1987).

<u>Diatom-inferred phosphorus database</u>

A third database was developed based on recent lake-sediment core, diatom-reconstructed TP data as derived from three separate, but related studies as follows:

- "55 lakes study" included lakes from the NLF (20), urbanized portion of CHF (20), rural portion of CHF (10) and WCP (5). Surface and deep cores were taken from all lakes. Details on this study may be found in Heiskary and Swain (2002) and Ramstack et al. (2002).
- Southwest Minnesota lakes study, included about 25 shallow lakes distributed across the WCP and NGP. Surface sediments were collected on all lakes, which contributes to the diatom "training set" (model development), and deep cores were taken from seven lakes, which provides a basis for estimating pre-European phosphorus concentrations. Details on this study may be found in Heiskary et al. (2003).
- West-Central Shallow Lakes Study, as the name suggests, focused on 31 lakes in west – central Minnesota (primarily in the CHF ecoregion). Nine lakes were selected for deep cores in this study and of those, diatoms were adequately preserved in six lakes to allow for estimates of pre-European phosphorus concentrations. Details on this study may be found in Heiskary and Lindon (2005).

Data from these studies will be summarized in this report and a listing of all lakes with sediment core reconstructions from the previously noted studies is included in Appendix II.

<u>USEPA Criteria Development Database</u>

A fourth database is drawn from the USEPA (2000 b,c,d) criteria documents. Though much of this data is from Minnesota and overlaps with our assessment database it will in some cases allow for larger datasets than what we may have available for only Minnesota since they contain data from neighboring states in the same ecoregion. It also provides some data for ecoregions where we had limited data, i.e., Red River Valley, Northern Minnesota Wetlands, and Driftless Area.

Data in the criteria documents was drawn primarily from STORET for the period 1990-1999. Statistical distributions for various trophic status variables are presented by aggregated ecoregion, level III ecoregions, and by season. Data for individual sites (lakes) were reduced to seasonal medians. In our application of this data we chose to focus on compiled statistics for lakes in the level III ecoregions for the summer index period, which is consistent with the other Minnesota datasets as follows:

Aggregated ecoregion	Level III	Name	Citation
6	46	Northern Glaciated Plains	USEPA, 2000b
6	47	Western Corn Belt Plains	USEPA, 2000b
6	48	Red River Valley	USEPA, 2000b
7	51	North Central Hardwood Forests	USEPA, 2000c
7	52	Driftless (Paleozoic Plateau)	USEPA, 2000c
8	50	Northern Lakes and Forests	USEPA, 2000d
8	49	Northern Minnesota Wetlands	USEPA, 2000d

Figure 1. 2004 305(b) Assessed Lakes. Includes all lakes in "assessed" database. Monitored implies data collected in most recent 10 years (1993-2002). Evaluated implies data collected prior to 1993.

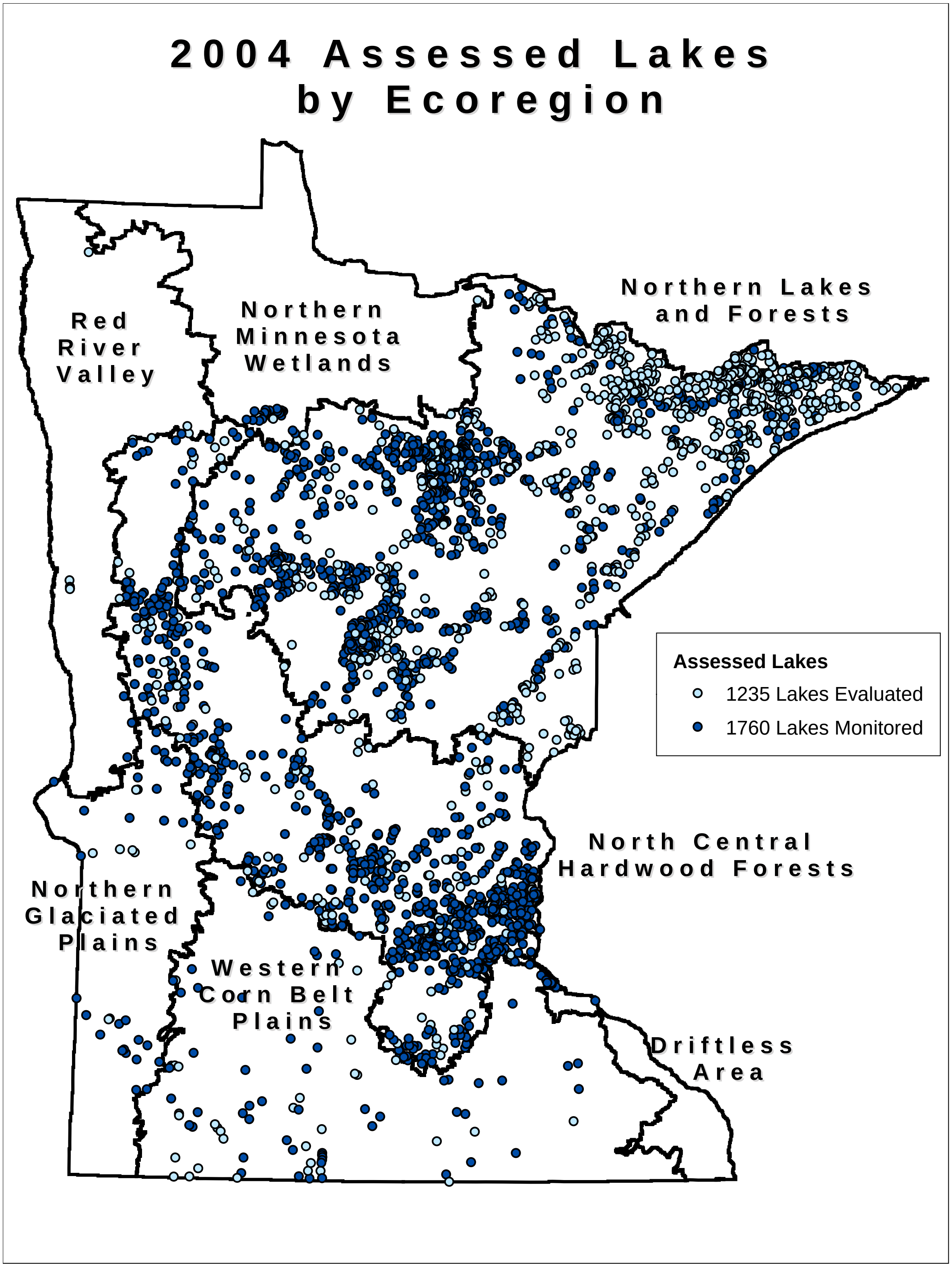
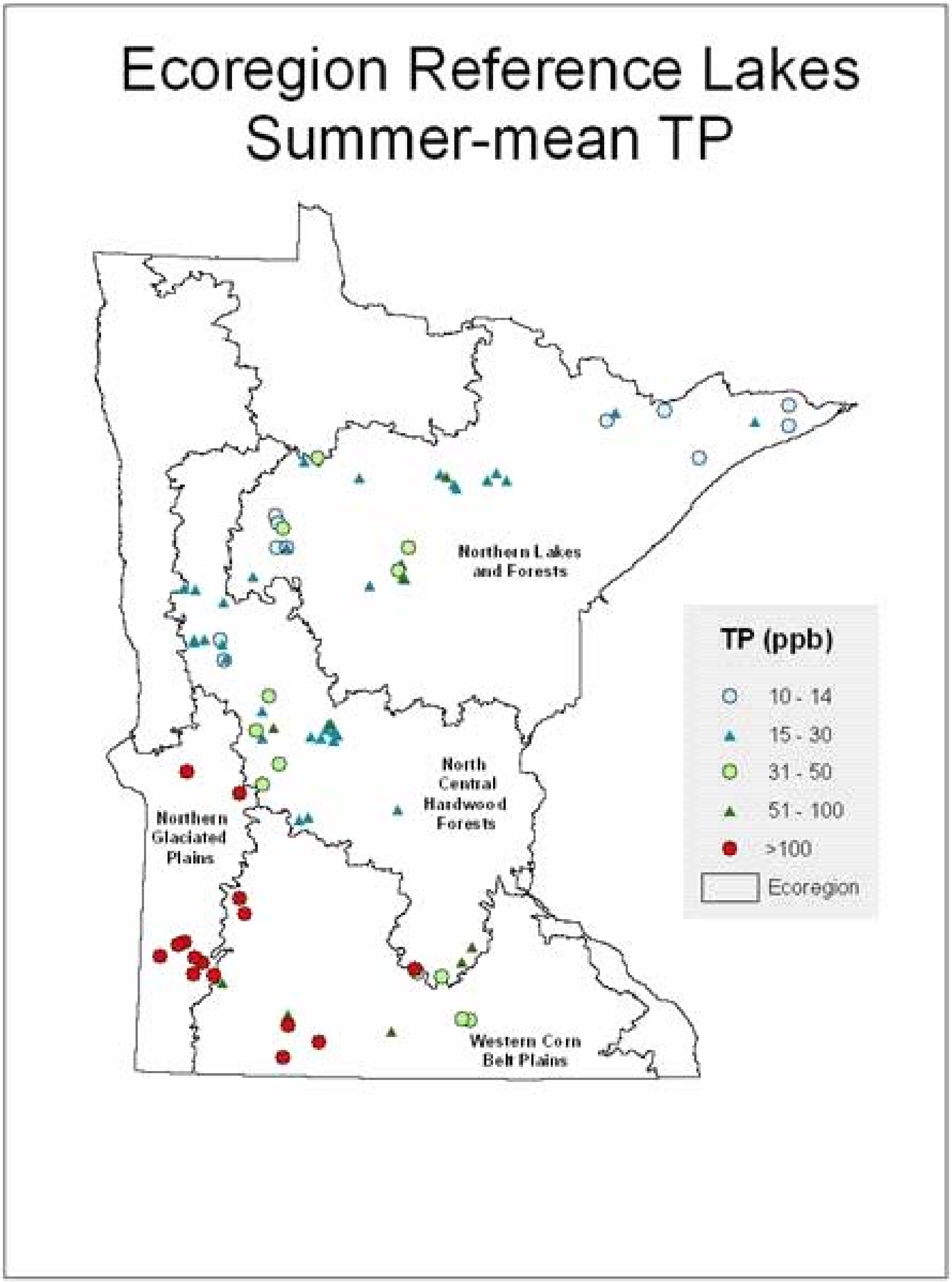


Figure 2. Location of ecoregion reference lakes. Summer-mean TP noted.



III. CONSIDERATIONS FOR LAKE EUTROPHICATION CRITERIA DEVELOPMENT

Developing Criteria

Water quality numerical criteria should reflect the latest scientific knowledge on the identifiable effects of pollutants on public health and welfare, aquatic life, and recreation. Water quality criteria are qualitative or quantitative estimates of the concentration of a water constituent which when not exceeded, will ensure water quality sufficient to protect a designated water use.

The need to establish lake water quality criteria or standards has been recognized at the state, provincial and federal levels of government. Establishment of nutrient criteria is deemed essential to the protection of the water quality of lakes. The state of Maine, for example, implemented a standard (Maine DEP, 1986) requiring “stable or declining trophic status” for its lakes. This standard, in effect, does not allow changes in the land use in the watershed of a lake that may adversely impact the trophic status of the lake. British Columbia established phosphorus criteria to protect the most sensitive uses of lakes in that province, citing in particular, drinking water or recreation and aesthetics (less than 10 µg/l P) and support of a coldwater fishery (5-15 µg/l P) as among the most sensitive uses (Nordin, 1985). At a national level, the North American Lake Management Society (NALMS) in 1987 established a task force on Lake Water Quality Standards to determine the thinking of states regarding the need for lake standards and to gather information related to existing lake standards. At the federal level, the re-authorized Clean Water Act of 1987 in Section 319 established the need to develop new or implement existing standards as a basis for submitting and evaluating projects. As a response to the continued problem of eutrophication and the general lack of numeric nutrient criteria USEPA (1998) has called for the development of numeric nutrient criteria by the states. USEPA (2000a) and NALMS (1992) present several examples of state and provincial government approaches to setting nutrient criteria or goals.

Because of regional diversity in lake and watershed characteristics, it was felt that a single total phosphorus value could not be adopted as a statewide criterion for lake protection in Minnesota (Heiskary, et al. 1987). As such, we felt a methodology was needed for developing phosphorus criteria on a regional or lake-specific basis. The methodology for establishing our original phosphorus criteria in Minnesota considered the following (Heiskary and Walker, 1988):

(1) phosphorus impacts on lake condition (as measured by chlorophyll-a, bloom frequency, transparency, and hypolimnetic oxygen depletion);
(2) impacts on lake user (aesthetics, recreation, fisheries, water supply, etc.);
(3) appropriateness (as related to watershed characteristics, regional phosphorus export values, lake morphometry, etc.).

These considerations remain valid for our current effort. The following sections identify a number of factors or “considerations” which should be taken into account when attempting to derive water quality criteria suitable for protecting uses of lake resources or for setting goals to improve or expand the use of a resource. While not all of the following are necessary for establishing criteria for protection or goal setting for restoration, the more information that is compiled the better the likelihood that a reasonable criteria has been selected. This will be particularly important as these criteria will be the basis for 303(d) "impaired waters" listing and lake improvement efforts. The following types of information can be valuable in the assessment of lake condition and can contribute to the development of nutrient criteria:

- Ecoregion – in which the lake and its watershed are located. If the lake is at or near an ecoregion boundary, assess which ecoregion is appropriate for the lake based on ancillary information (i.e., trophic status, morphometry, etc.) compiled for the lake.
- Trophic characteristics – including mean summer epilimnetic measures to total phosphorus (TP), chlorophyll-a, and transparency. Other measures such as total suspended solids, color, total nitrogen (TN), TN:TP ratios may be helpful also.
- Morphometry – including mean and/or maximum depth and surface area.
- Mixing status – whether the lake is dimictic (thermally stratified during summer), polymictic (does not stratify) or mixes intermittently (may stratify for short time periods during summer) as defined by a series of dissolved oxygen and temperature profiles or estimated based upon morphometric characteristics.
- Watershed – size and land use composition.
- Lake drainage type – drainage, seepage, inflow, reservoir, etc.
- Fishery or wildlife – MDNR classification.
- Macrophytes – composition, number of native species, extent and maximum depth of coverage of lake basin (particularly important for shallow lakes).
- Most sensitive sub-uses of the lake, e.g., domestic water supply or coldwater fishery. Documentation of other uses may be helpful too.
- Lake user expectation for the lake as derived by survey or extrapolated based on uses and regional tendencies.
- Development in the shoreland area. Intensity of development and MDNR classification, e.g., general, recreational or natural.
- History of the lake and its watershed. For example, has the lake received wastewater effluent, has extensive development occurred in the watershed, have watershed boundaries changed as a result of development or other factors, is the lake subject to severe water level fluctuations, etc.

Crockett, et al. (1989) also provide a listing of information that should be gathered prior to criteria setting. Their listing is somewhat different. However, it reflects important information for the resource they are proposing for which criteria be developed. Also, they share a similar view as to the necessity of this information, when they state: "Setting water quality criteria before taking these factors into account may mean that the criteria are inappropriate to the system being managed."

The following sections provide a means for interpreting the information that has been assembled for an individual lake or a set of lakes. The chapter on current lake pollution control programs in Minnesota provides information on existing programs to protect or restore lake water quality in Minnesota. In the final chapter, eutrophication criteria are derived by ecoregion based on the previous information and analysis. Case studies are used throughout the report and appendix to provide specific Minnesota examples of the concepts discussed.

Figure 3. Carlson's Trophic State Index

Carlson's Trophic State Index
RE Carlson

TSI < 30 Classic Oligotrophy: Clear water, oxygen throughout the year in the hypolimnion, salmonid fisheries in deep lakes.

TSI 30 - 40 Deeper lakes still exhibit classical oligotrophy, but some shallower lakes will become anoxic in the hypolimnion during the summer.

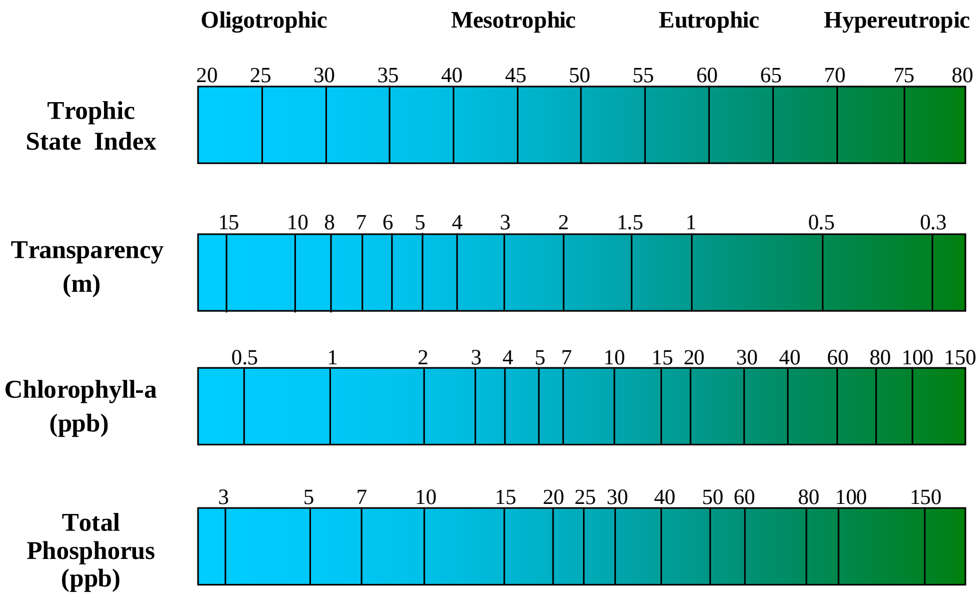
TSI 40 - 50 Water moderately clear, but increasing probability of anoxia in hypolimnion during summer.

TSI 50 - 60 Lower boundary of classical eutrophy: Decreased transparency, anoxic hypolimnia during the summer, macrophyte problems evident, warm-water fisheries only.

TSI 60 - 70 Dominance of blue-green algae, algal scums probable, extensive macrophyte problems.

TSI 70 - 80 Heavy algal blooms possible throughout the summer, dense macrophyte beds, but extent limited by light penetration. Often would be classified as hypereutrophic.

TSI > 80 Algal scums, summer fish kills, few macrophytes, dominance of rough fish.



After Moore, 1. And K. Thornton, [Ed.]1988. Lake and Reservoir Restoration Guidance Manual. USEPA>EPA 440/5-88-002.

<u>Trophic Status and Regional Water Quality Patterns for Minnesota Lakes</u>

Before realistic nutrient criteria can be developed we must understand the interrelationships among the variables most commonly used to characterize lake trophic status: total phosphorus, total nitrogen, chlorophyll-a and Secchi transparency. Further, since we seek to develop ecoregion-specific criteria we must understand the among and within region range in trophic status. Previous publications have addressed the patterns in lake trophic status in Minnesota (Moyle, 1956; Heiskary, 1985 and Heiskary, et al., 1987, Heiskary and Wilson, 1989). Others such as Gorham et al. (1983) described regional patterns in ionic chemistry for Minnesota lakes. In this edition we revisit these previous efforts and update it with data from more recent assessments and continued monitoring from the reference lakes. First, we will address some of the important variables for assessing the trophic status of lakes and secondly, we will look at regional patterns in lake trophic status and the factors that contribute to differences in lake trophic status across Minnesota.

a. Regional patterns and trophic status interrelationships

The parameters most frequently used to assess the trophic status of lakes are total phosphorus (TP), chlorophyll-a, and Secchi transparency (Figure 3). The assessed data provide an opportunity to explore regional patterns in trophic status and lake morphometry as summarized in Table 1. While distinct regional differences in the distribution of lake trophic status are evident among some regions (e.g., Northern Lakes and Forests (NLF) and Western Corn Belt Plains (WCP)) there is also substantial overlap among others (e.g., WCP and Northern Glaciated Plains (NGP)). For example based on the interquartile range (IQ; 25^{th}-75^{th} percentile) typical phosphorus-based TSI (TSIP) values range from mesotrophic to mildly eutrophic in the NLF (TSIP = 41 – 53; TP = 13 – 30 µg/L), to eutrophic to hypereutrophic in the CHF (TSIP=52 – 73; TP = 28 – 188 µg/L), to hypereutrophic in the WCP (TSIP=71 – 82; TP= 103 – 221 µg/L), and NGP (TSIP=74- 87; TP=127-313 µg/L). The regional differences are evident for chlorophyll-a and Secchi as well. If we focus on values in the interquartile range we note three rather distinct classes or regions, e.g., NLF (forested), North Central Hardwood Forests (CHF) (transitional) and the two agriculture-dominated ecoregions. For the purposes of this report, the word "typical" refers to lakes/concentrations between the 25^{th} and 75^{th} percentile for each ecoregion.

The among-region differences are also quite evident in the reference lake data (Table 2, Fig. 3). These differences have been described in various publications (e.g., Heiskary and Wilson, 1989) and only a brief overview is offered here. Lakes in the NLF ecoregion are moderately deep (Table 1) and their watersheds are dominated by forest and wetland land uses (Table 11a). Based on the NLF reference lakes, typical trophic conditions can be summarized as follows: total phosphorus concentrations range from 14 to 27 µg/l, mean chlorophyll-a concentrations 4 - 10 µg/l, maximum chlorophyll-a concentrations less than 15 µg/l, and Secchi transparency on the order of 2.5 to 4.5 meters. Total Kjeldahl nitrogen (TKN) values are low and nitrate+nitrite-N values are typically at or below detection. Resulting TN: TP ratios suggest phosphorus-limited conditions. In general, these lakes can be considered mesotrophic to mildly eutrophic. Total suspended solids (TSS) are very low in these lakes and seldom are a cause for light limitation. Water color, a reflection of humic substances (incompletely dissolved organic matter) in the water, can be quite variable across the ecoregion and may range from very low coloration (< 20 Pt-Co Units), to moderate coloration (20-50 Pt-Co Units) to very dark coloration (> 50 Pt Co Units). Color may reduce the amount of light available for algal growth at very high levels. Inspection of individual trophic status data will aid in determining whether it is an important factor or not for individual lakes. For example, Big Sandy Lake (01-0062), a reservoir that drains a very large watershed characterized by forests and wetlands and has a very high color value of 170 Pt Co units, exhibited the following trophic status values: TP= 51 µg/L (TSIP=61), chlorophyll-a= 17 µg/L (TSIC=59), and Secchi= 1.3 m (TSIS=56). In this case chlorophyll-a is only slightly lower than anticipated by the TP but Secchi is actually higher than anticipated based on TP.

The CHF ecoregion is characterized by moderately deep lakes as well but does have a high percentage of shallow lakes as well (Table 1). No single land use is dominant across the ecoregion (Table 11a). Based on the CHF reference lakes typical values are as follows: total phosphorus ranged from 23-50 µg/l, mean chlorophyll-a from 5 to 22 µg/l, maximum chlorophyll-a from 7 to 37 µg/l and Secchi transparency ranged from 1.5 to 3.2 m. TKN values are slightly higher than the NLF lakes and nitrate-N remains low (Table 2). TN: TP ratios suggest P limitation. In terms of trophic state, these lakes would generally be considered eutrophic. Again TSS is fairly low in these lakes and will seldom cause light limitation for algae. Color values are typically low in these lakes with the exception of lakes that have a high percentage of their watershed comprised of wetlands.

The lakes of the WCP ecoregion are predominately shallow (Table 1) and agricultural land use is dominant across the ecoregion (Table 11a) and in the watershed of the reference lakes (Table 11b). Based on the reference lakes typical trophic conditions can be summarized as follows: total phosphorus concentrations range between 65 to 150 µg/l, mean chlorophyll-a concentrations range from 30 to 80 µg/l, maximum chlorophyll-a from 60 to 140 µg/l, and Secchi transparency from 0.5 to 1.0 m. Those lakes with total phosphorus concentrations less than 65 µg/l were among the few which are deep enough to maintain stratification. TSS concentrations may be elevated in many of the shallow windswept lakes of this ecoregion. Inorganic suspended sediment (i.e., clay and soil particles) may constitute a significant portion of the TSS (Table 2).

Virtually all lakes in the NGP would be considered shallow and as with the WCP agriculture is the dominant land use in the watershed (Table 11a). Based on the reference lakes typical trophic conditions can be summarized as follows: total phosphorus concentrations range from 122 to 160 µg/l, mean chlorophyll-a from 36 to 61 µg/l, maximum chlorophyll-a from 66-88 µg/l, and Secchi transparency ranging from 0.4-0.8 m. No lakes in this ecoregion were found to be stratified and all could be considered hypereutrophic. Again TSS concentrations may be quite high in these lakes as reinforced by a recent study of southwest Minnesota lakes (Heiskary et al. 2003). Individual values ranging up to 110 mg/L, of which 82% was inorganic suspended solids (ISS), were noted under very windy conditions in some lakes (Loon Lake 32-0020). However in some lakes, like Lime Lake (51-0024), high TSS events were dominated by algae based on ISS of 33% and chlorophyll-a of 286 µg/L during summer 2002 (Heiskary et al. 2003).

Once again, some overlap in trophic conditions is noted among the four regions. From a lake water quality management standpoint, two very distinct classes are evident, i.e., forest dominated (NLF) and agriculture dominated (WCP and NGP). The transitional zone (CHF) between these two extremes will share characteristics of these two classes, but may present management problems unique to this ecoregion.

Table 1. Minnesota Lake Water Quality Assessment Data ("MPCA assessed") summarized by ecoregion (MPCA, 2004). Water quality values represent summer means. Central tendency (interquartile range) noted in bold.

		Percentile							
Ecoregion	**Parameter**	**5**	**10**	**25**	**50**	**75**	**90**	**95**	**N**
NLF & NMW[1]	Area (acres)	15	22	**49**	**129**	**347**	835	1,654	1,809
NLF & NMW	Depth-max. (feet)	7	10	**19**	**33**	**54**	80	100	1,519
NLF & NMW	TP ppb	7	9	**13**	**21**	**30**	45	58	863
NLF & NMW	Chl-a ppb	2	2	**3**	**5**	**8**	14	22	521
NLF & NMW	Secchi (m)	0.9	1.2	**1.8**	**2.8**	**4.0**	5.1	5.9	1,394
CHF & RRV	Area (acres)	13	22	**58**	**165**	**400**	984	1,754	976
CHF & RRV	Depth-max. (feet)	6	8	**16**	**28**	**46**	68	82	829
CHF & RRV	TP ppb	15	18	**28**	**51**	**112**	229	351	691
CHF & RRV	Chl-a ppb	3	4	**8**	**21**	**45**	89	131	622
CHF & RRV	Secchi (m)	0.4	0.5	**1.0**	**1.6**	**2.6**	3.5	4.2	968
WCP	Area (acres)	32	61	**143**	**322**	**694**	1,776	2,222	110
WCP	Depth-max. (feet)	4	6	**7**	**10**	**16**	25	33	87
WCP	TP ppb	54	62	**99**	**159**	**234**	404	609	89
WCP	Chl-a ppb	11	14	**32**	**50**	**83**	125	173	79
WCP	Secchi (m)	0.2	0.3	**0.4**	**0.6**	**1.0**	1.5	2.2	109
NGP	Area (acres)	80	108	**150**	**364**	**658**	2,091	4,700	38
NGP	Depth-max. (feet)	4	5	**8**	**10**	**15**	18	25	28
NGP	TP ppb	46	54	**104**	**148**	**194**	396	405	30
NGP	Chl-a ppb	4	9	**25**	**36**	**52**	64	66	27
NGP	Secchi (m)	0.3	0.4	**0.5**	**0.7**	**1.6**	1.9	2.1	37

[1] NLF summary includes lakes from Northern Minnesota Wetlands (NMW) and CHF includes lakes from Red River Valley (RRV).

Table 2. Reference Lake Data Base Water Quality Summary
(Summer Average Water Quality Characteristics for Lakes by Ecoregion)*

Parameter	NLF	CHF	WCP	NGP
# of lakes	**32**	**43**	**16**	**13**
Total Phosphorus (ug/l)	**14 - 27**	**23 - 50**	**65 - 150**	**122 - 160**
Chlorophyll mean (ug/l)	**4 - 10**	**5 - 22**	**30 - 80**	**36 - 61**
Chlorophyll maximum (ug/l)	**< 15**	**7 - 37**	**60 - 140**	**66 - 88**
Secchi Disk (feet) **(meters)**	**8 - 15** **(2.4 - 4.6)**	**4.9 - 10.5** **(1.5 - 3.2)**	**1.6 - 3.3** **(0.5 - 1.0)**	**1.3 – 2..6** **(0.4 – 0.8)**
Total Kjeldahl Nitrogen (mg/l)	**0.4 – 0.75**	**< 0.60 - 1.2**	**1.3 - 2.7**	**1.8 - 2.3**
Nitrite + Nitrate-N (mg/l)	**<0.01**	**<0.01**	**0.01 - 0.02**	**0.01 - 0.1**
Alkalinity (mg/l)	**40 – 140**	**75 - 150**	**125 - 165**	**160 - 260**
Color (Pt-Co Units)	**10 – 35**	**10 - 20**	**15 - 25**	**20 - 30**
pH (SU)	**7.2 - 8.3**	**8.6 - 8.8**	**8.2 - 9.0**	**8.3 - 8.6**
Chloride (mg/l)	**0.6 – 1.2**	**4 - 10**	**13 - 22**	**11 - 18**
Total Suspended Solids (mg/l)	**< 1 – 2**	**2 - 6**	**7 - 18**	**10 - 30**
Total Suspended Inorganic Solids (mg/l)	**< 1 – 2**	**1 - 2**	**3 - 9**	**5 - 15**
Turbidity (NTU)	**< 2**	**1 - 2**	**3 - 8**	**6 - 17**
Conductivity (umhos/cm)	**50 – 250**	**300 - 400**	**300 - 650**	**640 - 900**
TN:TP ratio	**25:1 - 35:1**	**25:1 - 35:1**	**17:1 - 27:1**	**13:1 - 17:1**

*Based on Interquartile range (25th - 75th percentile) for ecoregion reference lakes.
Derived in part from Heiskary, S. A. and C. B. Wilson (1990).

It is important to understand the interrelationships of total phosphorus, chlorophyll-a and Secchi transparency if phosphorus management strategies intended to improve lake condition (as measured by chlorophyll-a, transparency, fisheries or other biological measures) are to be successful. One convenient method is with Carlson's TSI scale as noted previously in Figure 3. Another means is to examine their relationships graphically. Figures 4a (linear) and 4b (log-log) depicts scatter plots of these three variables based on summer mean measurements from the reference data set (data for all regions combined).

A strong linear relationship (R^2=0.71) is evident when plotting mean total phosphorus versus chlorophyll-a (Figure 4a). Much scatter about the regression line is evident though, in particular, at high phosphorus concentrations (above about 50 µg/L TP) where phosphorus may not be the primary element limiting algal production. Log-log regression yields a higher R^2 (0.88). In contrast, a plot of mean Secchi transparency versus mean chlorophyll-a reveals a hyperbolic (log-normal) relationship (Figure 4a). The greatest change in Secchi disk transparency occurs below about 20 -30 µg/l chlorophyll-a. Beyond this point (e.g. 30-80 µg/L), there is very little change in mean transparency. Forsberg and Ryding (1980) also noted minimal changes in Secchi disk transparency at chlorophyll-a concentrations above 40 µg/l. The scatter above the curve at high chlorophyll-a concentrations may be a function of algal populations dominated by large colonial forms (e.g., Aphanizomenon), which allow deep penetration of light into the water column. In contrast, the scatter below the curve may reflect the effect of non-algal suspended material or water color. Lastly a comparison of TP and Secchi suggests significant changes in Secchi over the concentration range from <10 – 50 µg/L (Figure 4a). Minimal changes in mean Secchi disk transparency are noted at TP concentrations above 40-50 µg/l. Overall these three graphs suggest the greatest changes in summer-mean chlorophyll-a and Secchi are likely to occur over a range of TP from <10 – 50 µg/L.

Regression analysis was used on these data to generate equations for predicting changes in Secchi as a function of chlorophyll-a (Chl-a) and Secchi and chlorophyll-a as a function of mean summer total phosphorus. The equations are expressed as log-log as this serves to normalize the data and provides the best basis for describing the relationship among these variables (Fig. 5b). The equations are as follows (in m and µg/L):

$$\log_{10} \text{Chl-a} = 1.31 \log_{10} \text{TP} - 0.95 \qquad R^2 = 0.88;\ n = 108 \qquad \text{(Eq. 1)}$$
$$\log_{10} \text{Secchi} = -0.59 \log_{10} \text{Chl-a} + 0.89 \qquad R^2 = 0.85;\ n = 108 \qquad \text{(Eq. 2)}$$
$$\log_{10} \text{Secchi} = -0.81 \log_{10} \text{TP} + 1.51 \qquad R^2 = 0.81;\ n = 108 \qquad \text{(Eq. 3)}$$

These equations and R^2 values are modified slightly from the previously published equations (Heiskary and Wilson, 1988). This is a result of adding a few lake trout lakes with TP concentrations below 10 µg/L (improves resolution at that end of the scale) and updating data for some of the reference lakes based on more current and complete data sets than were available in the original work. (e.g., data for several NGP and WCP lakes were updated based on Heiskary et al. 2003). Additionally, relationships have been developed specifically for shallow lakes based on the West-central lakes study (Eq. 4; Heiskary and Lindon, 2005). Eq. 4 exhibits a slightly lesser slope than the reference lake-based equation (Eq. 1) but over the range of lakes tested the TP: chl-a relationship is not substantially different.

$$\log_{10} \text{Chl-a} = 1.08 \log_{10} \text{TP} - 0.66 \qquad R^2 = 0.80;\ n = 31 \qquad \text{(Eq. 4)}$$

In general, relationships between total phosphorus, chlorophyll-a, and Secchi tended to be stronger in phosphorus-limited or dimictic lakes than in lakes with TN:TP < 17 or in polymictic lakes. This was particularly evident in TP & chl-a regressions developed for the shallow southwest Minnesota lakes whereby the R^2 was 0.10 when all 26 lakes were included in the regression but improved to R^2=0.51 when it was limited to those lakes with TP < 200 µg/L (Heiskary et al. 2003). Other factors such as color (due to bog stain) or suspended sediments may reduce light availability thus affecting the phosphorus – chlorophyll-a – transparency relationships (Garn and Parrott, 1977). In highly colored lakes, productivity for a given nutrient concentration will be less than that observed for clear lakes. For our data, no discernible differences in the expected phosphorus – chlorophyll-a – transparency relationships were identified for lakes with color values less than 50 platinum cobalt units (Pt-Co Units). This is comparable to the findings of Garn and Parrot (1977). Too few lakes with color values greater than 50 Pt-Co Units were available in the reference data base to further examine this relationship. Colored lakes occur most frequently in the NLF ecoregion where watersheds are typically dominated by forested and wetland land use. Suspended sediments can be an interference in shallow wind-swept lakes, which cannot effectively settle out these particles. Light limitation due to suspended sediments most commonly occurs in the two agricultural ecoregions: WCP and NGP, where total suspended solids levels are markedly higher than the other two regions (Table 2). This was also quite evident during the monitoring of 24 NGP and WCP lakes in 2002 where lakes with high inorganic TSS often had lower than expected chlorophyll-a concentrations (Heiskary et al. 2003).

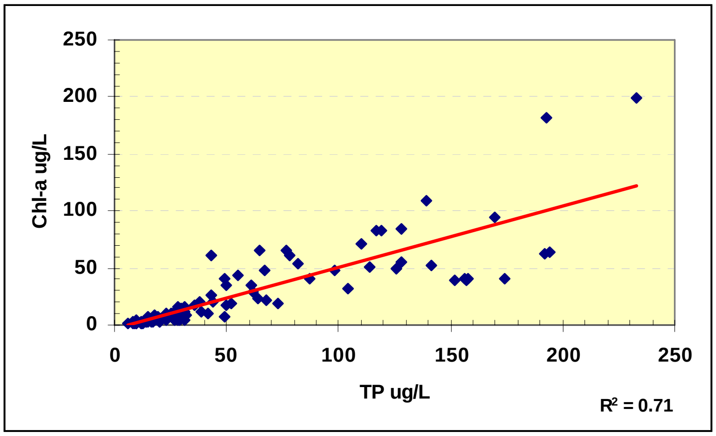
Smith (2003) notes that in general, comparative studies of freshwater eutrophication strongly suggest that efforts to control external nutrient loading to lakes will tend to achieve similar reductions in their average algal biomass. However, he notes that growing season average biomass (chlorophyll-a) is probably not consciously measured by lake users as a primary index of impairment; hence the need to focus on maximum concentrations that occur during transient blooms. The reference lake data also provide a basis for predicting extreme chlorophyll-a values as a function of the summer-mean. Summer maximum chlorophyll-a was highly correlated (R^2 = 0.89) with summer-mean chlorophyll-a (Fig. 5a).

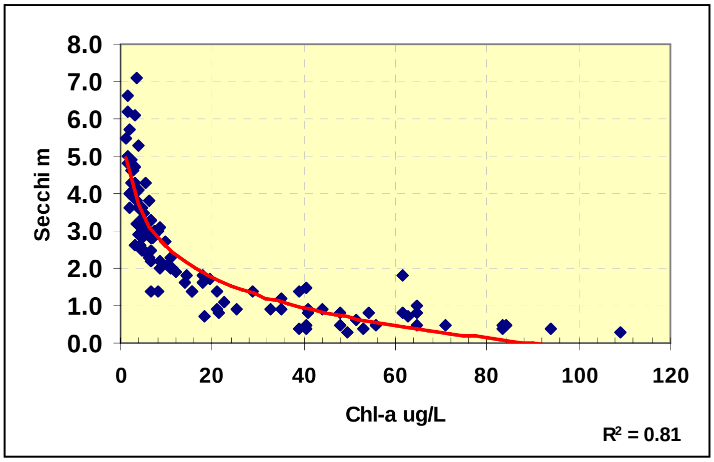
$$\text{Chl-a (max)} = 1.33\ \text{Chl-a (mean)} + 5.15 \qquad R^2 = 0.89;\ n = 108 \qquad \text{(Eq. 5)}$$

The R^2 in Eq. 5 is much higher than that reported by Jones et al. (1979) in a similar regression; however the slope of their regression equation was somewhat steeper (1.70 vs. 1.33). This relationship can provide a basis for estimating the summer-mean concentration that would be required to avoid certain extreme events. Walker (1984) took this relationship a step further by associating the mean with the frequency of various classes or levels of chlorophyll-a, referred to as “bloom frequency” (Fig. 5b). For example, if summer-maximum chlorophyll-a of 30 µg/L was to be avoided a summer-mean chlorophyll-a of about 17 µg/L or less would be required (Eq. 4). Alternately, based on Fig. 5b, there would be a 10 percent or less chance of encountering a maximum chlorophyll-a of 30 µg/L at a summer-mean of 17 µg/L.

Bachmann et al. (2003) examined bloom frequency relationships for Florida lakes relative to summer-mean chlorophyll-a and TP or TN concentrations with predictions that were stratified by TN: TP ratios (TN: TP<10, 10<TN: TP>17, and TN: TP>17). The relationships they found are quite similar to those presented by Walker (1984) in Figure 5b. Further, they acknowledge that their analysis should be applicable for northern temperate lakes as well.

Figure 4a. Summer-mean TP, chlorophyll-a and Secchi linear relationships. Based on reference lake data.





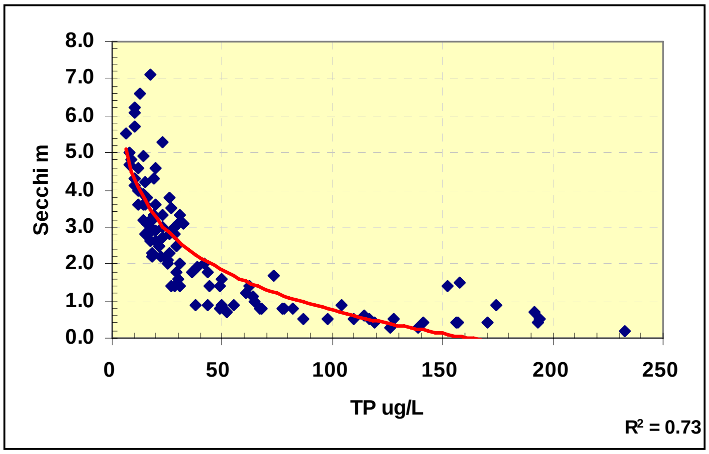
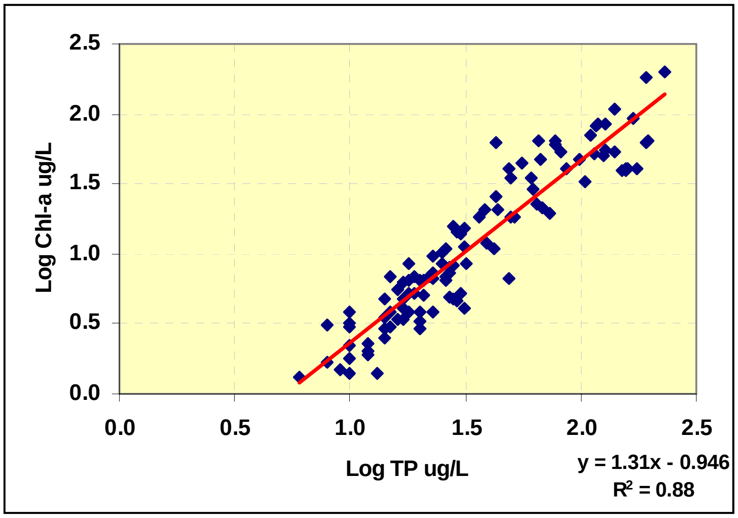
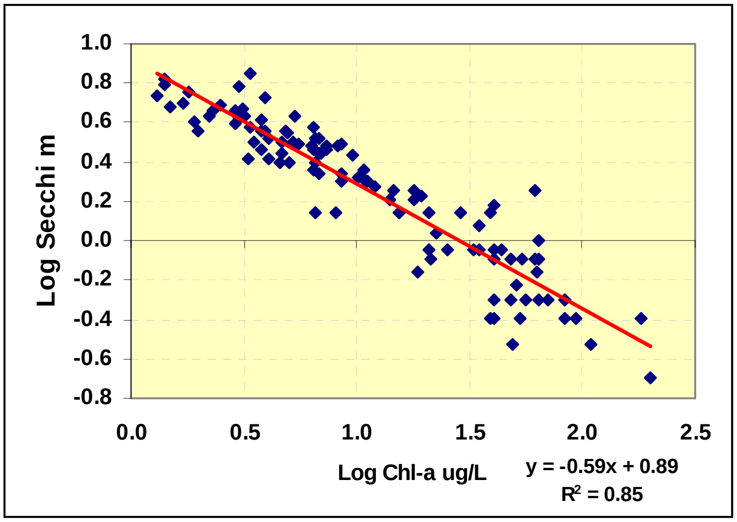


Figure 4b. Summer-mean TP, chlorophyll-a and Secchi log-log relationships. Based on reference lake data.





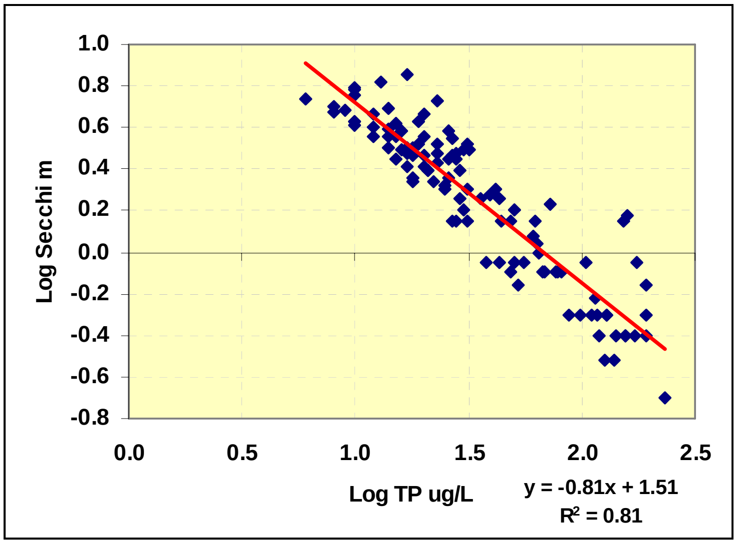


Figure 5a. Summer-maximum chlorophyll-a as a function of summer-mean chlorophyll-a. Based on reference lake data.

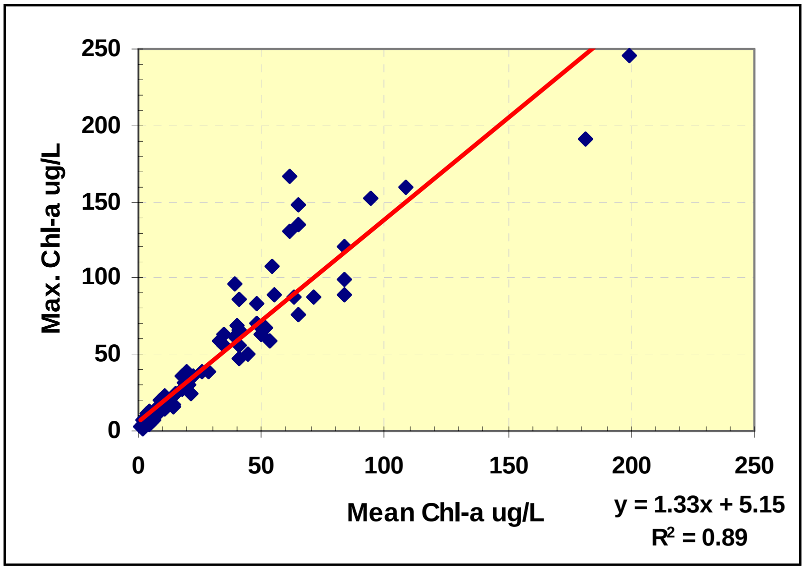
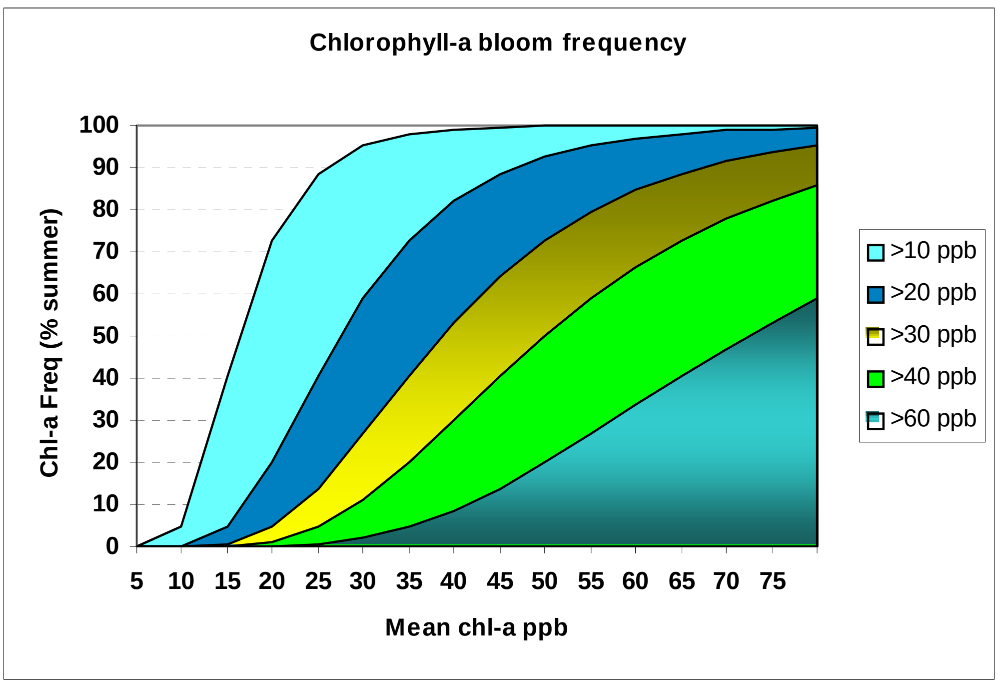


Figure 5b. Chlorophyll-a bloom frequency as a function of summer-mean. Derived from equations in Walker, 1984.



An expansion on this approach and probably a more useful means for examining the interrelationships of phosphorus, chlorophyll-a, and transparency, i.e., “lake response,” was developed using cross-tabulation. Paired phosphorus and chlorophyll measurements (n = 641) and paired phosphorus and transparency measurements (n = 630) from the reference lake data set were used to demonstrate the relationships among phosphorus concentration and nuisance-level frequencies of chlorophyll-a and transparency. Details on this procedure are noted in Heiskary and Walker (1988). Figure 6a depicts the chlorophyll-a frequency response relative to total phosphorus. This approach can be used to assess the “risk” of encountering nuisance level frequencies of chlorophyll-a and transparency.

The “risk analysis” approach was derived from a classification system developed by Walmsley (1984) for South African reservoirs. This system expresses lake conditions based upon the frequency of extreme chlorophyll-a concentrations (“blooms”), as opposed to average concentrations. User-perceived problems related to algae tend to be episodic, rather than continuous in nature. Bloom frequency more adequately reflects temporal variability in lake conditions and is thought to be a better indicator of potential use impairment. The phrase “nuisance criteria” refers to specific chlorophyll-a or transparency levels which result in perceived impairment. The State of Florida, for example, uses chlorophyll-a > 40 µg/L as an indication of an algal bloom (Bachmann et al. 2003). In Heiskary and Walker (1988) a wide range of criteria was tested for each lake response variable (>5, 10, 20, 30, 40, and 60 µg/l for chlorophyll-a and <.5, 1, 2, 3, and 5 meters for transparency).

For both chlorophyll-a and transparency, extreme value frequencies exhibit a nonlinear response to increasing phosphorus concentration (Figure 6a). For example, the observed frequency of chlorophyll-a concentrations exceeding 30 µg/l (Walmsley’s (1984) “severe nuisance” condition) is 0% at phosphorus concentrations below approximately 30 µg/l, increases steadily to approximately 70% at a phosphorus concentration of 100-120 µg/l, and levels off at higher phosphorus concentrations. This response is similar to that observed for transparencies less than 1 m. The “threshold” phosphorus concentration corresponds to the onset of detectable nuisance frequencies (in this example, 30 µg/l). The presence of a phosphorus threshold is important because it represents a logical focus for criteria development, provided that an appropriate nuisance algal level (e.g., 20, 30, 40 µg/l chlorophyll-a) can be defined.

More recent data from shallow WCP and NGP lakes allow us to examine this relationship specifically for shallow, nutrient-rich lakes (Fig. 6b) and expands on the previous analysis (Figure 6a). At TP concentrations less than 70 µg/L, severe nuisance blooms (chl-a 30-60 µg/L) occur about 15% of the summer, while very severe nuisance blooms (chl-a >60 µg/L) occur less than 5% of the summer. As TP increases to the 70-90 µg/L range the frequency of severe nuisance blooms increases to about 45% and very severe nuisance blooms increase to 10%. These frequencies are rather stable until TP increases above about 110 µg/L where a large increase in the frequency of very severe nuisance blooms was noted. The frequency of very severe nuisance blooms peaks at about 60% over the TP range 200-250 µg/L. As TP increases above this level there is an apparent decrease in frequency that may be a function of light limitation, from algal cells, high TSS, or other factors that limit algal production.

Figure 6a. Chlorophyll-a interval frequencies versus total phosphorus. Based on cross-tabulation of reference lake data from 1985-1988.

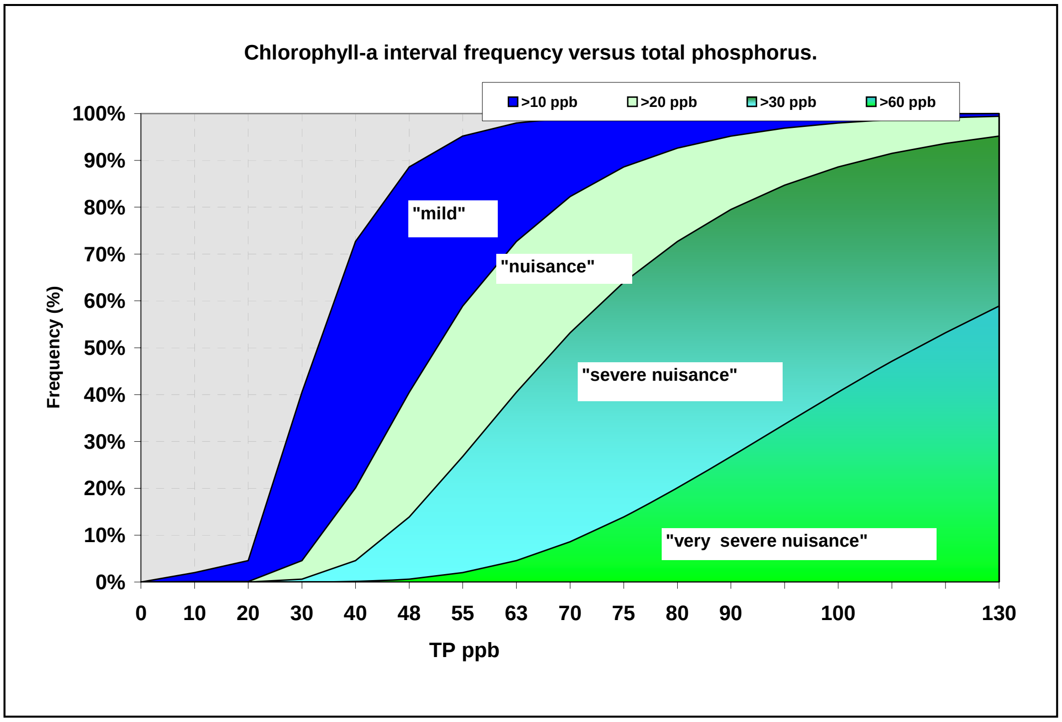
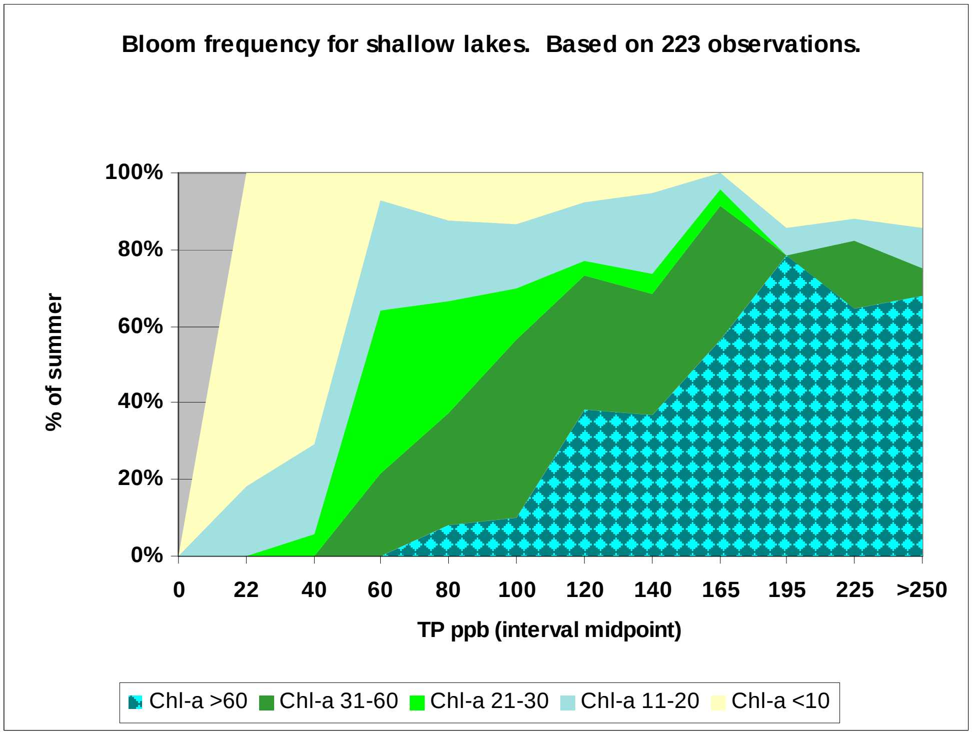


Figure 6b. Chlorophyll-a interval frequencies for shallow lakes. Based on cross-tabulation of data from southwest & west-central MN shallow lakes.



b. Regional patterns in nitrogen and TN:TP ratios

Nitrogen (N), while not considered the limiting nutrient in most cases for freshwater lakes, is nonetheless an essential nutrient for algal and rooted plant growth (Wetzel, 2001). And, as EPA has requested states to consider development of N criteria, some brief discussion of N, as related to TP, chlorophyll-a and trophic status is merited here. Total nitrogen (TN), for the purposes of discussion is essentially equal to total Kjeldahl nitrogen (TKN) as nitrite + nitrate N (and of this most is in nitrate form) is at or below detection (0.01 mg/L) for most NLF and CHF lakes (Table 2). In WCP and NGP lakes, nitrate N values typically range between detection and 0.1 mg/L. As such, nitrate N contribution to TN is rather low in most Minnesota lakes.

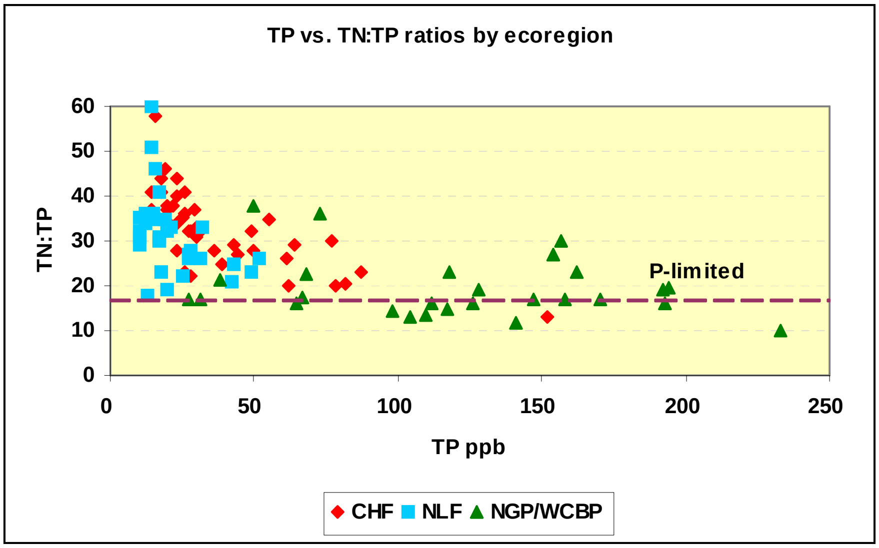
TKN is rather low and exhibits a small range in the NLF lakes (Table 2). The CHF, in contrast, is characterized by higher values and a larger range that can be attributed to the more nutrient-rich soils of this region and anthropogenic inputs from the increasingly agricultural or urban landuse that is common across the ecoregion. TKN concentrations are higher yet in the highly agricultural WCP and NGP ecoregions (Table 2).

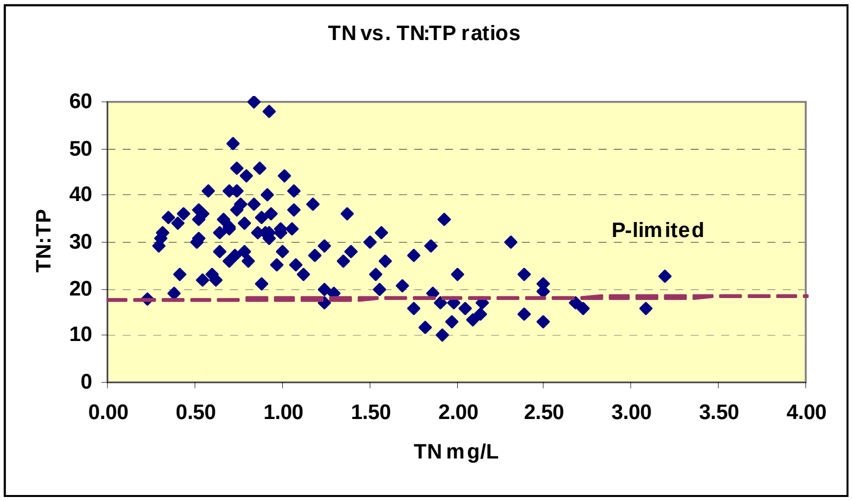
TN:TP ratios (TN calculated as sum of TKN + nitrate N) are often used as a basis for defining "P-limitation (when TN:TP >17:1)" vs. "N-limitation (when TN:TP<10:1)" with lakes in between being either P or N-limited (Smith, 1983). However, even when lakes are likely to be "N-limited" this generally means, from a lake water-quality management standpoint, that very large P reductions are required to bring the lake closer to "P limitation." One Minnesota example of this would be Lake Shaokatan (Case Study #21), a shallow NGP lake that exhibited extremely high TP (~375 µg/L in 1991) and low TN:TP ratio (~8:1) prior to attempts to rehabilitate the lake. Watershed efforts focused on P reduction and targeted feedlot runoff as a major source of nutrients (which also yielded some N reduction as well). Various measures were instituted and by 1999 in-lake P was reduced to ~125 µg/L and TN:TP ratio was increased to 15:1, which approached P limitation. Further efforts to reduce P loading to the lake continue.

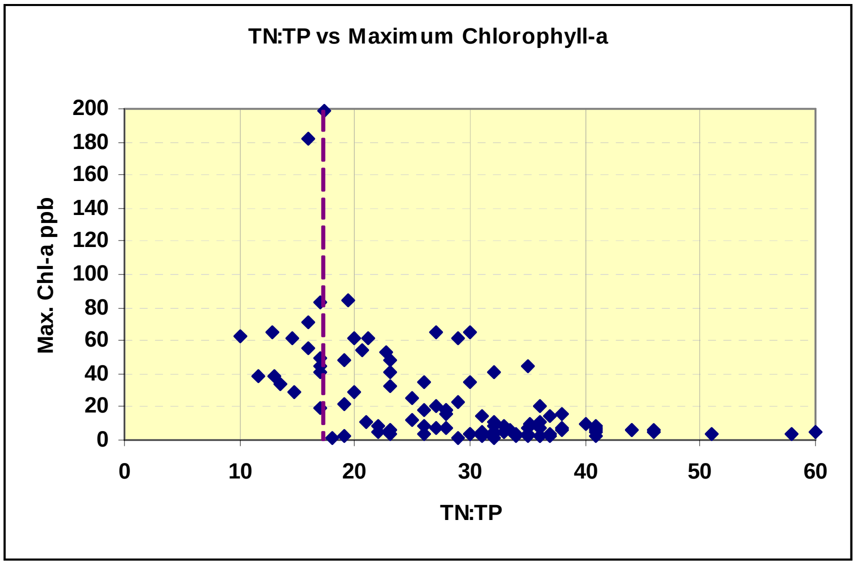
Based on the reference lake data NLF and CHF lakes will generally be considered P-limited, while WCP and NGP lakes may tend toward N-limitation (Table 2, Figure 7). TN:TP ratios (based on the reference lakes) are more highly correlated with TP ($R^2 = 0.63$) than to TN ($R^2 = 0.28$). Below TP concentrations of about 100 µg/L lakes will tend toward P limitation (Figure 7).

Based on Smith (1983) dominance by blue-green algae has been associated with TN:TP ratios <29:1. In a 2002 study of 22 NGP and WCP lakes blue-green algae were found to be dominant in most of the lakes (Heiskary et al. 2003). TN:TP was <29:1 in all lakes and <17:1 in the majority the lakes in that study. Maximum chlorophyll-a concentrations tend to increase as TN:TP declines with marked increases noted at TN:TP < 30 (Figure 7). And it is our experience that blue-greens are often the dominant alga responsible for these "blooms." Further, as TN:TP falls below 17:1 maximum chlorophyll-a is generally > 30 µg/L – a level typically considered to be a "severe nuisance bloom." These observations argue for keeping TN:TP ratios above 29:1 and P concentrations below 100 ug/L in order to minimize intense and frequent blue-green algae blooms.

Figure 7. TN:TP ratios compared to TP, TN and chlorophyll-a (reference lake data).







Lake Morphometry, Mixing, and Trophic Status

Previous investigators recognized that lake morphometry, in addition to watershed factors, plays an important role in determining lake productivity (Rawson, 1955; Ryding and Forsberg, 1980; Riley and Prepas, 1985). These factors (characteristics) must also be considered when developing lake nutrient criteria as they may: influence TP, chlorophyll-a and Secchi relationships; influence the species of fish that may be found in the lake; influence internal nutrient recycling; and/or may influence whether primary productivity is expressed primarily through rooted submerged vegetation or through phytoplankton. Table 1 summarizes the distribution of maximum depths and surface areas for the assessed lakes in each of the ecoregions. In general, lakes in the NLF and CHF tend to be deeper and smaller than those in the WCP and NGP.

In deeper lakes, as the surface temperature increases over the spring, the density differences between the surface water and bottom waters may exceed the ability of wind, eddy gradients and advective flows, etc., to mix these two layers. In shallow lakes, however, the wind may provide enough energy to destabilize the water density differences. The lake depth required for stable lake thermal stratification is a function of lake surface area, basin orientation relative to prevailing winds, lake depth-volume relations, protection by the surrounding topography and vegetation and other factors (Walker, 1985b). Osgood (1988) offered an index for classifying lakes as to their likelihood to mix based on lake morphometric factors.

The role of lake mixing status as it pertains to lake productivity has been addressed by various authors, e.g., Riley and Prepas (1985). In general, a lake that remains thermally stratified throughout the summer can be expected to exhibit stable or declining epilimnetic phosphorus (P) concentrations over the course of the summer due to algal uptake and sedimentation. This assumes that external supplies are low and littoral inputs are low. While internal release of phosphorus will occur in stratified lakes with anoxic hypolimnia, much of this P remains in the hypolimnion and typically is not available for summer algal production in the epilimnion.

In contrast, polymictic lakes or lakes which mix intermittently throughout the summer can be expected to react somewhat differently. These lakes may be characterized by increasing or widely fluctuating epilimnetic P concentrations during the summer (Fig. 9a). Summer- mean epilimnetic P may also be higher than spring (overturn) phosphorus concentrations. Various explanations include recycling of phosphorus from the sediments due to mixing of water column and general lack of effective sedimentation of phosphorus.

Internal loading of phosphorus may be a very significant source for lakes that temporarily stratify, form an anoxic layer near the sediments (allowing for phosphorus release from the sediments) and mix with the upper layers at a later date (Larsen, et al. 1981). However, polymictic lakes that remain oxic near the water-sediment interface over most of the summer may have low summer internal phosphorus loading rates (Nurnberg, 1984).

Lake mixing status was determined for the reference lakes based on summer oxygen and temperature profiles (e.g., Figures 8a & b) from 1985 and 1986. Lakes were classified as:

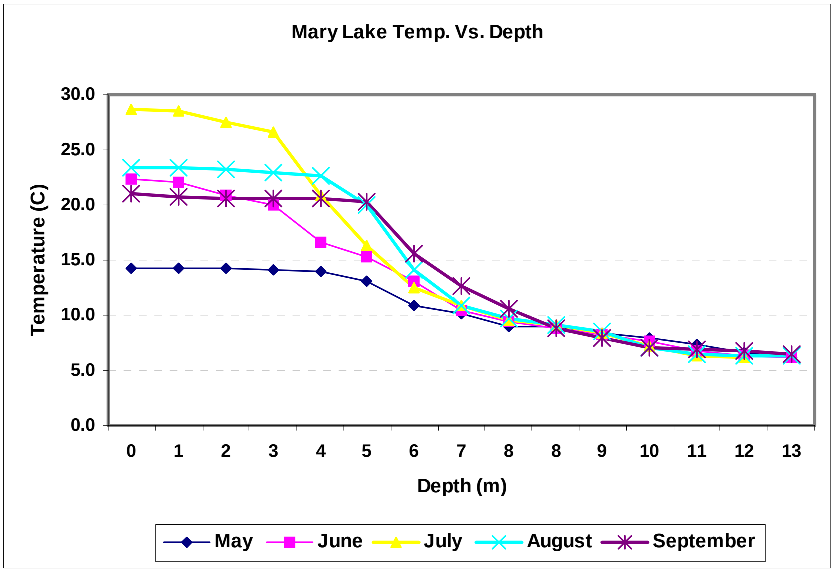
dimictic, polymictic or intermittent and their morphometric characteristics were reviewed (Table 3).

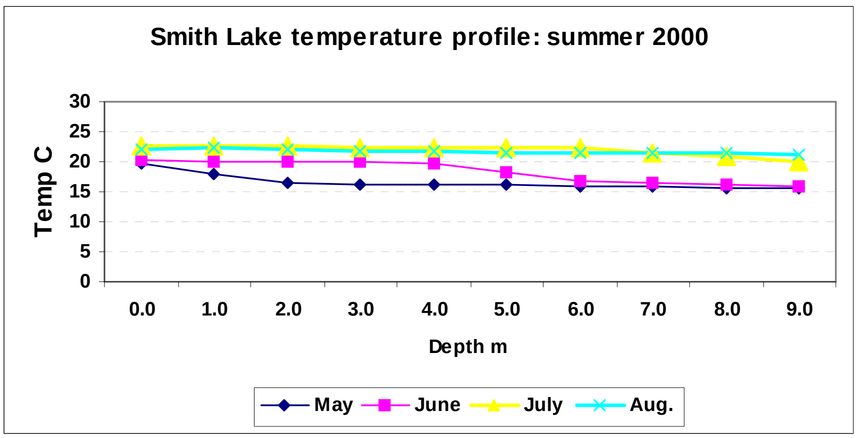
Table 3. Morphometric characteristics of dimictic, intermittent, and polymictic lakes. Based on reference lakes.

	Dimictic (N = 57)	**Intermittent** (N = 10)	**Polymictic** (N = 23)
Maximum depth	90% >10 m (33 ft)	90% ≤11 m (36 ft)	90% < 8 m (26 ft)
Mean depth	90% > 4 m (13 ft)	90% ≤ 5 m (16 ft)	90% ≤ 4 m (13 ft)
Surface area: depth ratio (ha/m)	75% <20:1	90% ≤20:1	75% >30:1

In general, this analysis suggests that most of the dimictic lakes tended to have maximum depths greater than 10 meters and surface area: maximum depth ratios less than 20:1. Whereas, the polymictic lakes generally had maximum depths less than 8 meters and surface area: maximum depth ratios greater than 30:1.

Figure 8a. Temperature profiles (May-September) for dimictic (Mary 77-0019), intermittent (Smith 21-0016)and polymictic (Budd 46-0030) lakes.





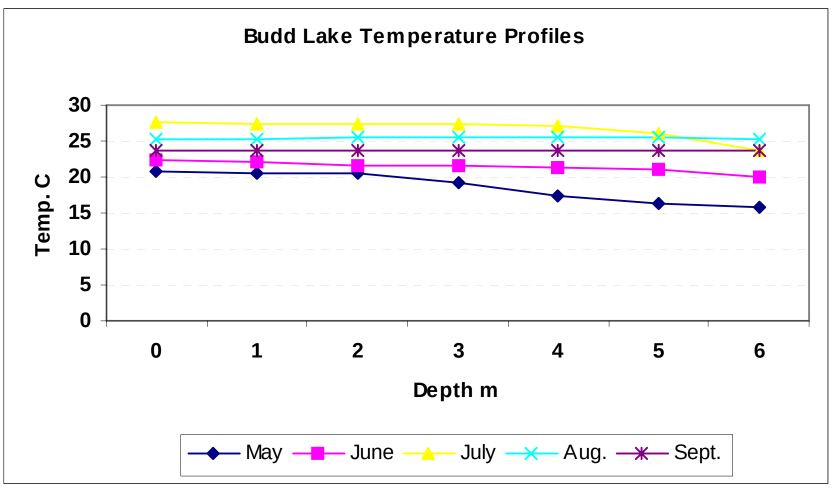
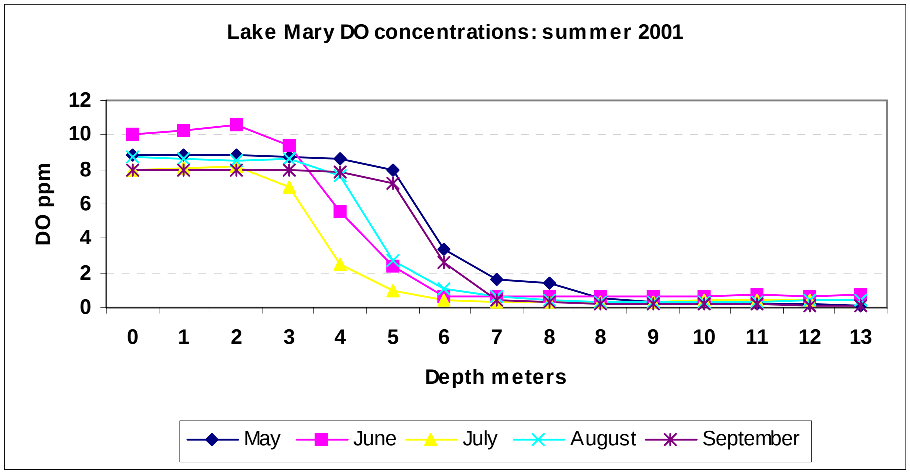
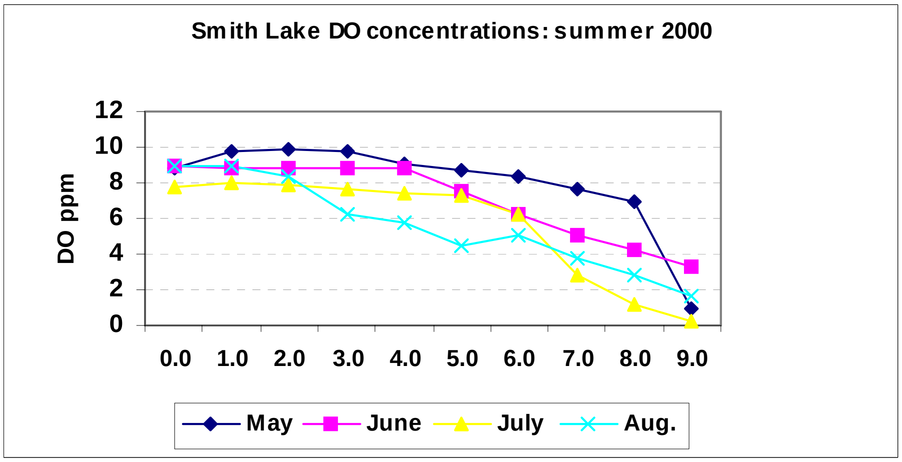
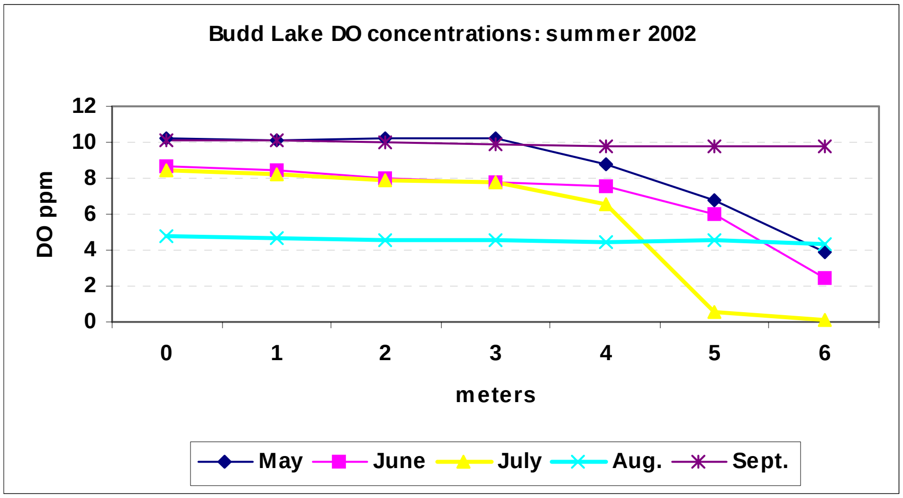


Figure 8b. Dissolved oxygen profiles for dimictic (Mary 77-0019), intermittent (Smith 21-0016)and polymictic (Budd 46-0030) lakes.







More recently Hondzo and Stefan (1996), in describing the role of lake morphometry on lake mixing, used a “lake geometry ratio” (based on Gorham and Boyce, 1989) to differentiate between stratified and polymictic lakes, whereby

lake geometry ratio (GR) = $A^{0.25}/D_{max}$
A = surface area in m^2
D_{max} = maximum depth in m

Based on this they noted that lakes with ratios > 8 were polymictic, ratios < 2 were dimictic and ratios from 2-8 were transitional. They also noted that lakes with ratios >5 did not have permanent hypolimnetic anoxia during the summer. If we apply the GR to the 90 reference lakes we found the following:

Mixing	N	GR-mean	GR - range	Dmax (m) mean	Dmax range	Dmean (m)
Dimictic	57	2.0	1.1 – 3.7	20.1	8.2 – 48.6	6.9
Intermittent	10	3.5	2.5 - 4.5	8.9	7.6 – 10.9	3.8
Polymictic	23	13.5	4.0 - 38.5	3.6	1.8 – 4.0	2.5

Based on the reference lakes dimictic lakes had GR values of < 4.0, while all polymictic lakes had values of 4.0 or greater. Intermittently mixing lakes were found over a range of values from 2.5 - 4.5. The reference lakes tend to support the above noted ranges for separating dimictic (<2) from polymictic (>8) lakes and suggest that the boundaries might be further refined.

Hondzo and Stefan (1996) created three classes of lakes: shallow, medium and deep with the respective GR and maximum depth ranges as follows:

	GR	D_{max} range (m)
shallow	5.3	1-5
medium	1.6	5-20
deep	0.9	20-45

In subsequent analysis of temperature and DO regime for the different classes they found a transition in lake behavior from dimictic lakes to polymictic lakes at a GR of 5.

Since maximum depth information was available (in contrast to mean depth) for most of the assessed lakes (Heiskary and Wilson, 1988), it was used as a basis for estimating the mixing status as follows: maximum depth $\geq$11 m (36 ft.) – dimictic, maximum depth < 8 m (26 ft.) – polymictic, and the remainder were considered intermittent.

The influence of mixing status (lake morphometry) on the trophic status of lakes in each ecoregion can be estimated from the assessed lakes. For this purpose, total phosphorus data were evaluated for each mixing status in each region (Table 4). Point source impacted lakes were excluded in this analysis. In general, dimictic lakes are quite common in the NLF and CHF ecoregions. In contrast, most lakes in the WCP are polymictic and all lakes in the NGP are polymictic.

Table 4. Distribution of Total Phosphorus (µg/l) Concentrations by Mixing Status and Ecoregion. Based on assessed lake data (Heiskary and Wilson, 1988). Dimictic = D, Intermittent = I and Polymictic = P

Ecoregion:	NLF			CHF			WCP		
Mixing Status:	**D**	**I**	**P**	**D**	**I**	**P**	**D**	**I**	**P**
Percentile value for [TP]									
90%	37	53	57	104	263	344	--	--	284
75%	29	35	39	58	100	161	101	195	211
50%	20	26	29	39	62	89	69	135	141
25%	13	19	19	25	38	50	39	58	97
10%	9	13	12	19	21	32	25	--	69
# of lakes	(257)	(87)	(199)	(152)	(71)	(145)	(4)	(3)	(38)

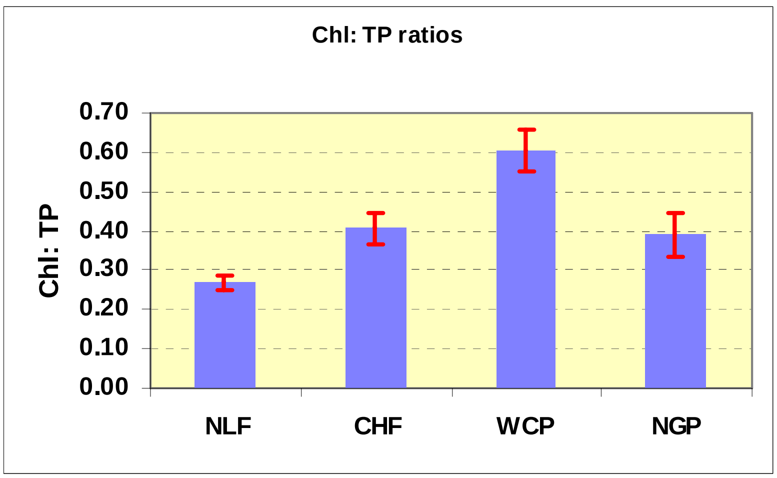
Some distinct regional patterns are evident in these data. In the NLF, differences in phosphorus concentration among the three mixing-status categories appears to be minimal up to the 75th percentile. In the CHF distinct differences in phosphorus concentrations exist among mixing-status categories over the entire range. In general, for a given percentile, e.g., 50th percentile (median), the phosphorus concentration of polymictic lakes tends to be two to three-fold higher than the dimictic lakes. The distribution of phosphorus for the intermittent mixing lakes tends to be more similar to the polymictic lakes than the dimictic lakes. Differences in phosphorus concentrations between mixing types in the WCP are also evident. However, there are too few dimictic intermittently mixing lakes in the data set to assess the significance of these differences. This analysis suggests that lake mixing status (i.e., lake depth) may be a very important factor to consider in the development of phosphorus criteria for lakes in the CHF or WCP ecoregions.

In addition to among-region differences in TP and chlorophyll-a concentration there are also some differences in the amount of chlorophyll-a produced per unit TP among regions (Fig. 9a) and across the three “mixing” classes (Fig. 9b). The polymictic (shallow) lakes tend to exhibit higher chlorophyll-a: TP ratios as compared to the dimictic or intermittent (deeper) mixing lakes based on the reference lakes and there appears to be an upward trend in the ratio as we move from dimictic to polymictic (Fig. 9b). Though sample sizes were small these patterns appeared to hold within, as well as across, ecoregions as noted in Figure 9 (with the exception of the WCP where no difference was noted).

Figure 9. Mean chlorophyll-a: TP ratios based on reference lakes by: a) ecoregion and b) mixing class. Standard error of mean noted.

	Mixing class	Dimictic (n)	Intermittent (n)	Polymictic (n)
Ecoregion	NLF	0.23 (20)	0.33 (6)	0.40 (5)
	CHF	0.36 (33)	0.47 (5)	------
	WCP	0.60 (4)	---------	0.57 (8)

a.



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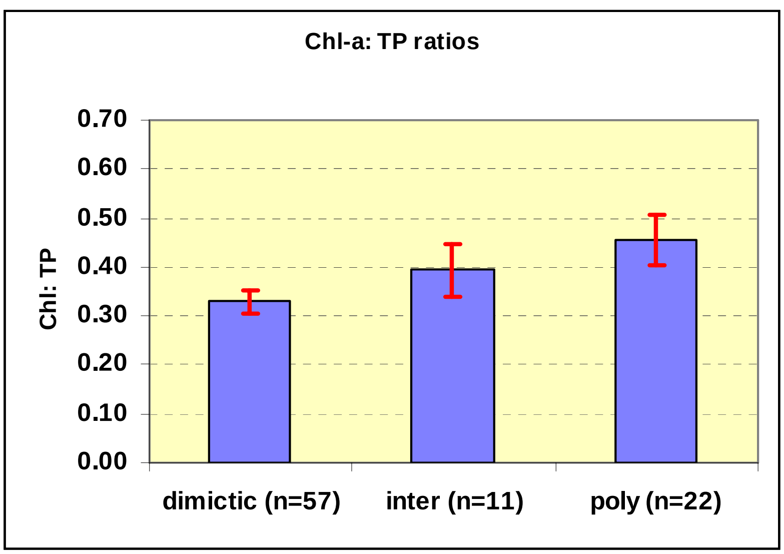

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Case Study 1.
Summer TP trends for a dimictic lake (Mary; 77-0019), intermittently mixing lake (Smith; 21-0016) and polymictic lake (Budd 46-0030)

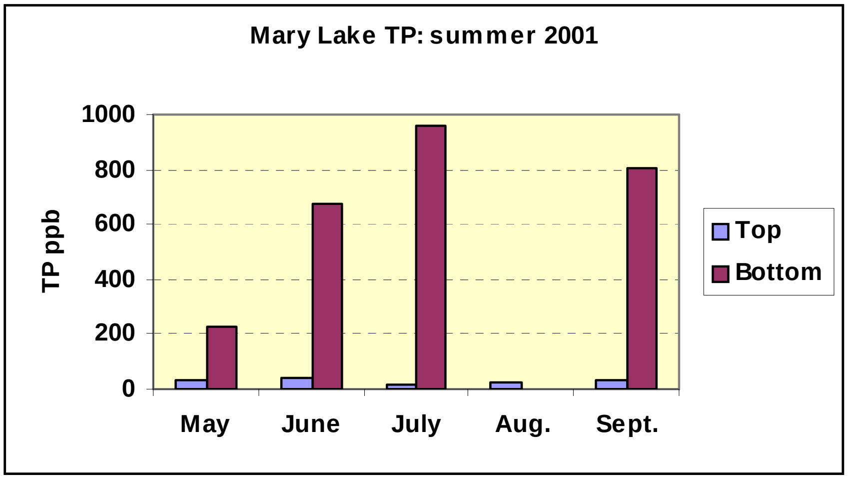
All three lakes were monitored at least one summer during the 2000-2002 time period. Mary (Todd County) and Smith (Douglas County) Lakes are among the reference lakes for the CHF ecoregion, while Budd Lake (Martin County) is in the WCP ecoregion. Mary is a 103 acre (42 ha) lake with a maximum depth of 55 feet (17 m) and area: maximum depth ratio (ha/m) of 2.5:1. Smith Lake is a 581 acre (235 ha) lake with a maximum depth of 36 feet (11 m) and area: maximum depth ratio of 22:1. Budd Lake is 207 acres (84 ha) with a maximum depth of 17 feet (5 m) and area: maximum depth ratio of 16:1.

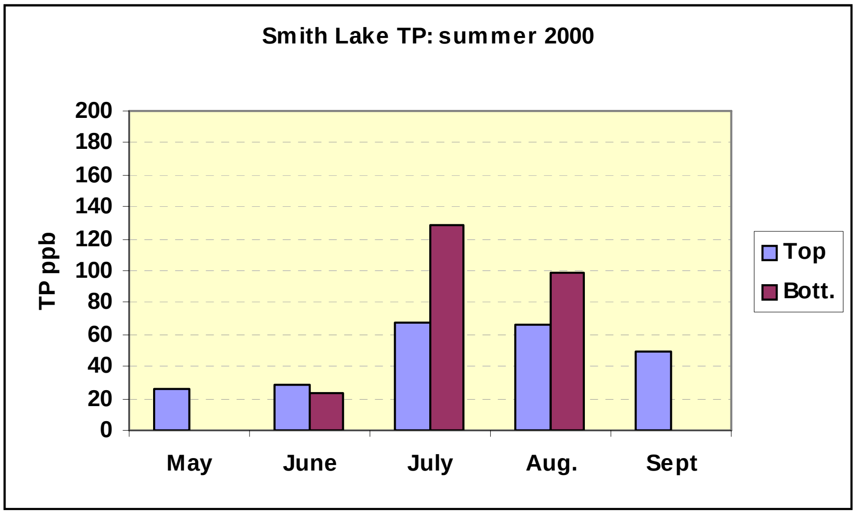
Temperature profiles for Mary (2001) reveal strong stratification from May through September (Figure 8a). Dissolved oxygen concentrations fell below 2 mg/L in the hypolimnion in May and remained at or below 2 mg/L through September (Figure 9b). Smith Lake exhibited very slight stratification in May and June but was very well-mixed on August and September sample dates (Figure 8a). DO concentrations fell below 2 mg/L just above the sediments on the July and August sample dates (Figure 8b). Budd Lake was well-mixed on all dates (Figure 8a); however DO concentrations in the lower two meters fell below 2 mg/L in late June and July presumably caused by quiescent conditions.

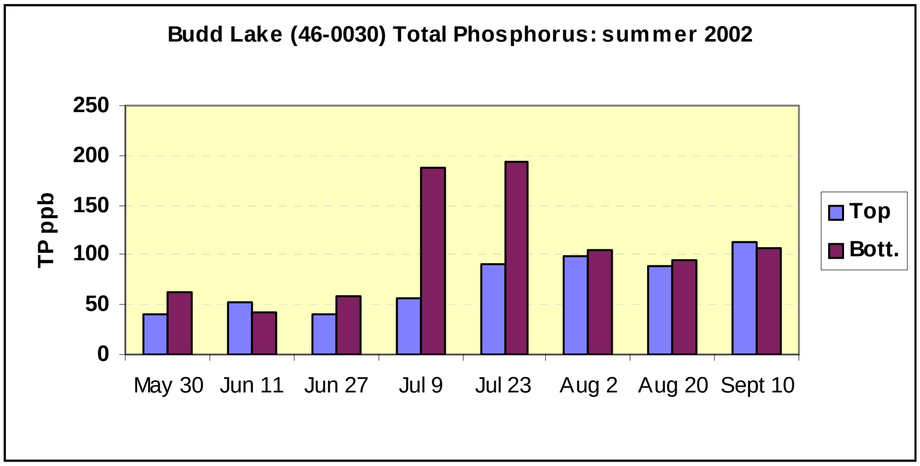
Consistently low DO concentrations in the hypolimnion of Mary Lake promoted internal recycling of phosphorus from the sediments of the lake, as evidenced by the elevated hypolimnetic TP concentrations (Fig. 10); however, epilimnetic TP concentrations remained quite low (19-38 µg/L) during the summer months and there was no evidence of mixing of the layers. Epilimnetic TP in Smith Lake tended to increase from May through August. Weak stratification resulted in low DO concentrations above the sediments, which allowed for elevated TP in the hypolimnion (Fig. 10). Because of the shallowness of the lake, stratification can be easily disrupted and the phosphorus-rich bottom waters can readily mix with the surface waters – which likely occurred in September. Epilimnetic TP concentrations tended to increase over the summer in Budd Lake (Fig. 10). There was minimal difference in epilimnetic and hypolimnetic TP until July when hypolimnetic concentrations were elevated. This can be attributed to the very low DO concentrations and very high temperatures at the water-sediment interface – both of which promote internal recycling of phosphorus. Cooling and wind-mixing in August re-introduced DO into the bottom waters and resulted in relatively uniform epilimnetic and hypolimnetic TP concentrations on the subsequent sample dates.

These three lakes provide examples of seasonal patterns in epilimnetic and hypolimnetic TP in lakes of varying depth and mixing characteristics. In shallow lakes, classified here as intermittent or continuously mixing, epilimnetic TP can be strongly influenced by internal recycling combined with wind mixing. During windy periods, TP is fairly consistent from top to bottom in shallow lakes; however under quiescent conditions DO concentrations will decline near the sediment-water interface. The combination of low DO and high temperature near the sediment-water interface helps promote internal recycling of phosphorus (Sas et al. 1989). In shallow lakes, this phosphorus is then readily mixed into the epilimnion with subsequent wind events. This is in contrast to deeper lakes that stratify firmly throughout the summer (e.g., Mary; Figure 8a and b).

Figure 10. Comparison of epilimnetic (top) and hypolimnetic (1 m off bottom) TP concentrations for Mary (dimictic), Smith (intermittent), and Budd (polymictic) Lakes.







Fisheries – Ecoregion Patterns and Effects of Eutrophication

Fishing by state residents and outstate visitors is a leisure time activity frequently enjoyed in many Minnesota lakes. In 2001, an estimated 30.1 million fishing days were spend in Minnesota waters by over 1.6 million fishermen (16 years and older; Fish and Wildlife Service, 2001). Of these, 1.3 million are Minnesota residents. Approximately $1.3 billion is spent annually for fishing and of this amount $407 million is spent in Minnesota on equipment (Fish and Wildlife Service, 2001; http://www.census.gov/prod/2003pubs/01fhw/fhw01-mn.pdf). Because Minnesota fisheries are considered such a prized natural resource, it is important in developing lake phosphorus criteria that an examination be made of how in-lake phosphorus concentrations relate to the types of Minnesota lake fisheries, how nutrient loadings may affect the type and abundance of fish, and alternately how some fish species may impact lake water quality.

Lakes in the North American temperate zone are extremely variable with regard to their biological productivity and fish community structure. This variability is due in large part to the tremendous diversity in the nature of lake watersheds and lake basin morphometry. Studies of the relationship between lake morphometry, lake water chemistry and fish yield (Rawson, 1952; Ryder, 1965; Matusek, 1978; Hanson and Leggett, 1982) have generally shown that more nutrient-rich, shallower lakes are typically more biologically productive with higher fish yields per unit area than deeper, less fertile lakes. In qualitative terms, along the productivity continuum ranging from oligotrophic through hypereutrophic, there is also a range in fish community structure. Associated with the various lake categories, a corresponding fish community exists, which has optimized its community structure to utilize the resources of the environment. In broad terms, water bodies on the eutrophic end of the spectrum are populated by sunfish (centrarchids), minnows (cyprinids) and other warm water species. Trout and whitefish (salmonids), and sculpins (cottids) inhabit coldwater oligotrophic lakes. Walleye, northern pike, and white suckers and their associates, optimally inhabited coolwater mesotrophic environs (Ryder, 1981).

Moyle (1956) examined the variety of lake types in Minnesota, by evaluating regional trends in water chemistry, fish yield, and fish species association throughout the state. He noted a relationship between several measures of lake fertility (phosphorus, nitrogen, and alkalinity) and fish yield. In general, as lake fertility increased from the northeast part of the state to the south and southwest portions, the estimated quantity of fish biomass also increased. His analysis also revealed that dominant large fish species associations varied among regional lake groups along this geographic gradient. Lake trout and tullibee were found in the northeast corner where total phosphorus (TP) concentrations averaged 20 µg/l or less based on MDNR data. Northeast and north-central lakes, having approximate mean TP of 34 µg/l, were often inhabited by walleye and its common associates, yellow perch, northern pike, and white suckers. Central Minnesota lakes with mean TP of 58 µg/l were historically bass and panfish lakes. Southern Minnesota lakes having the highest average TP, 126 µg/l, were most commonly dominated by rough fish, such as buffalo, freshwater drum and common carp.

The observed relationships between lake productivity and dominant large fish species, along with several other habitat considerations, have been used by the Minnesota Department of Natural Resources (MDNR) to classify lakes for fishery purposes. At the time of the 1990

MLWQA report there were two designations of lake classification, under the MDNR scheme, ecological and management. The ecological classification designates the fish populations that are best adapted to the physical, chemical, and biological characteristics of a lake and which the lake could be expected to support if it were left alone with no management effort (Scidmore, 1970). The MDNR fisheries management classification describes the most important species or combination of species toward which fisheries management efforts should be directed. Descriptions of each class are given in the Heiskary and Wilson (1990) and Scidmore (1970).

In an attempt to expand on Moyle's description of the geographic distribution of lake fisheries in Minnesota, 1,800 lakes were sorted by ecological and management types into the various ecoregions (Heiskary, et al., 1987). The lakes in the NLF are most typically classified as bass-panfish, bass-panfish-walleye, and soft water walleye. A small percentage of lakes in this ecoregion receive the lake trout designation. The most common ecological classification in the CHF is bass-panfish followed by winterkill – roughfish. In the WCP and NGP the most common ecological classification is winterkill-roughfish with a small percentage classified as bass-panfish, bass-panfish walleye and bullhead.

In terms of management classification, the lakes of the NLF are most commonly managed as bass-panfish-walleye. Lakes in the CHF are typically managed as centrarchid (largemouth bass) with a secondary emphasis on walleye. Lakes in the WCP in most cases are managed as centrarchid-walleye (largemouth bass with walleye). Those lakes managed for walleye in this ecoregion are usually stocked, although some natural reproduction may occur. Lakes of the NGP are typically managed for warm water game fish. However, because of the shallow and fertile nature of many lakes in the Western Corn Belt and NGP, winter aeration is often needed to meet fishery objectives.

A very general regional relationship between TP concentrations in lakes, and size and structure of the fish community was demonstrated by Moyle (1956). To further delineate the connection between TP and fisheries type, approximately 900 lakes of various ecological classification were grouped by ecoregion and the range of TP concentrations for each class was determined from the assessment data base. Figure 11 exhibits the range of TP concentrations for each ecological classification within each ecoregion.

NLF trout lakes have the lowest overall TP concentrations with an average value of 16 µg/l. This group also has the narrowest range of TP concentrations, 6-27 µg/l. Soft water walleye lakes and northern pike-sucker lakes in this region have slightly higher average TP values (22 µg/l), with a somewhat broader range of TP concentrations, 8-39 µg/l. Bass-panfish and hard water walleye lakes exhibit somewhat higher phosphorus concentrations than the other game fish lakes in the NLF, but have lower average TP when compared to the same ecologically classed lakes in the other ecoregions. Rough fish lakes in the NLF have higher average TP values relative to all other ecological classes in the region, but have comparable TP ranges to lakes classified as suitable for bass/panfish-walleye lakes in the CHF. Bass-panfish-walleye and bass-panfish lakes in the CHF have intermediate TP values in comparison to similarly classed lakes in the other ecoregions. Lakes in the WCP and NGP have the highest TP values overall.

These results reveal that within each ecoregion there is an increase in average TP from lakes ecologically typed for more oligotrophic fish species to lakes ecologically typed for more eutrophic fish species. In other words, in the NLF, average TP values increase from the lake trout group to the soft water walleye, bass-panfish, through the rough fish classed lakes. In the CHF, TP averages increase from bass-panfish walleye lakes, to bass-panfish, to roughfish lakes. In the WCP and NGP only two groups are represented in the data set, but the average TP value for bass-panfish is less than for those classified as roughfish lake type. However, the average and ranges of TP for each ecological class varies considerably between ecoregions.

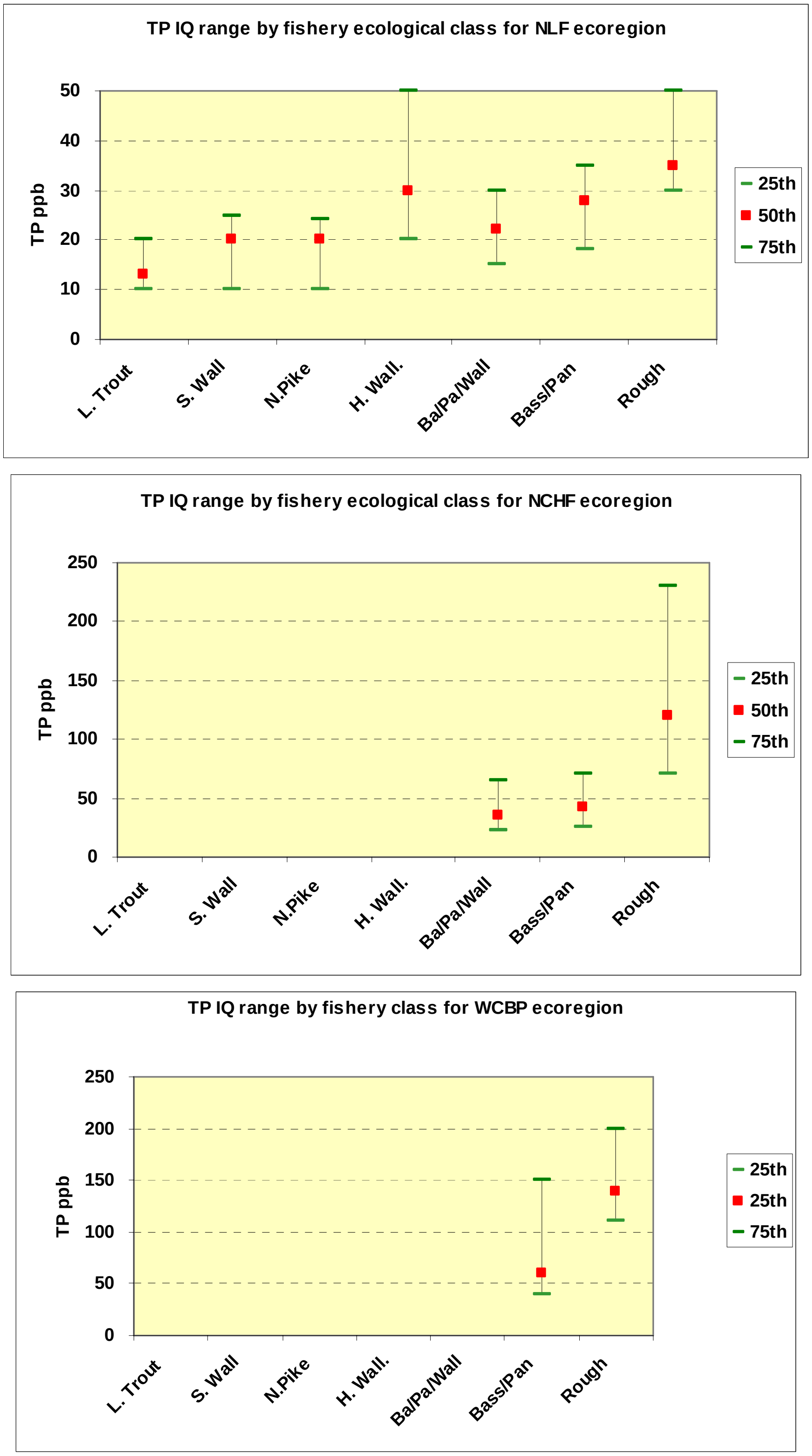
This might be expected, since suitable habitat for a given fish species is a combination of many factors, of which lake fertility, as estimated by TP, is only one variable. Other important factors include size of the water body, depth, extent of littoral zone, suitable spawning substrate and water temperature (Table 5). In addition, the ecological classification does not indicate the exact nature of the fisheries present in any lake, but rather provides an overall summation of habitat suitability.

In the time since our original analysis of fish ecological classification, relative to ecoregions and preferred ranges of TP, Schupp (1992) advanced a new MDNR ecological classification system for Minnesota lakes. This system takes many physical, chemical and morphological variables into consideration and results in classifying Minnesota's lakes into 43 types. One of the variables used was Carlson's Trophic State Index (TSI), based on Secchi transparency. In this work, lakes in northeastern Minnesota, which account for a large portion of the NLF ecoregion, fall into 19 different classes. The lakes of this area were felt to be fairly distinct from lakes elsewhere in Minnesota as many were formed mainly through the scouring of Precambrian rock by glacial ice sheets (Thiel, 1954) as opposed to deposition of glacial debris for much of the rest of the state.

A second division of the lakes was made based on an examination of the distribution of percent littoral (Schupp, 1992). Schupp (1992) found that based on a distribution of 3,029 lakes that the distribution of percent littoral was nearly normal from 0 to 80% and the number of lakes increased between 80 to 100% littoral. The subsequent division of lakes into groups < 80% littoral and ≥ 80% littoral was an approximate separation of lakes that rarely or never winter-kill from those that frequently winter-kill. This distinction (% littoral) will be used later in this report to discuss shallow lakes as opposed to deep lakes. The basic classification from Schupp's work resulted in 44 lake classes among four groups as follows:

< 80% littoral	≥80% littoral
Northeastern Minnesota	
11 classes	8 classes
Other Minnesota	
15 classes	9 classes

Figure 11. IQ range of total phosphorus concentrations by fishery ecological class and ecoregion (redrawn based on Heiskary and Wilson, 1988).



Lake classes 1 through 19 lie mainly in the three northeastern counties and most are soft-water lakes (Schupp, 1992). Included among these are abandoned iron-ore mine pits and lake trout lakes, which will be discussed in more detail later in this chapter. The remaining classes (20-44) are mainly hard-water lakes. The lake class numbers increase approximately from northeast to southwest within the state with increasing productivity, as estimated by MEI (Fig. 12).

Nutrient inputs into various lake types affect fish populations in both a qualitative and quantitative manner. Unfortunately, cultural eutrophication is typically not the only modification that may be occurring in a lake and, therefore, it is often difficult to distinguish changes in fish populations due to eutrophication, fish exploitation, species introduction, toxic effects of pollutants, habitat destruction, and the effects of disease and parasitism. However, some general trends have been observed in lakes undergoing eutrophication.

Initially, increased nutrient loading in some oligotrophic lakes may result in increased growth rates of coldwater species (Salvelinus, Salmo, Coregonus) due to increases in plants/algae/prey (Colby, et al., 1972). However, a biological surplus of any nutrient will at some point reduce the quality of the habitat (Ryder, 1981). Coldwater species require cold temperatures and ample dissolved oxygen (DO) (see Table 6). During summer lake stratification, these species may be confined to the hypolimnion where preferred combinations of temperature and DO can be found. Through continual nutrient loading and subsequent increases in organic material production, depletion of the oxygen can result in the elimination of oxygen in the hypolimnion. In such situations, the die-off of coldwater species may occur as these populations are driven into the water surface waters (Colby and Brooke, 1969).

Table 5. Water Quality and Habitat Considerations for Five Freshwater Fish.

	LAKE TROUT	SMALLMOUTH BASS	LARGEMOUTH BASS	WALLEYE	NORTHERN PIKE
DISSOLVED OXYGEN (mg/l)	• 7 mg/l – MPCA WQ standard applies • MDNR Management Criteria – To support lake trout, a lake needs to have a zone of water of at least 5 vertical feet with temp. ≤ 12.8°C and DO ≥ 5 mg/l • tended to be absent from Ontario lakes when average hypolimnetic oxygen [a] concentrations were < 6 mg/l • 4.5 mg/l – lethal to embryos at 10°C in laboratory experiments	• > 6 mg/l required for optimum growth [e] • > 6 mg/l required for embryo development in laboratory experiments • 5 mg/l MPCA standard applies • < 2.5 mg/l – no embryo survived in test study [c] • Frequently dies as DO concentrations approach 1 mg/l at 20-25°C [c]	• 5 mg/l MPCA standard applies • At < 4 mg/l growth substantially reduced in laboratory study [d] • < 1 mg/l considered lethal to adults [d]	• 5 mg/l MPCA standard applies • < 5 mg/l resulted in poor survival of stocked fry [e] • < 3.4 mg/l delayed hatching and reduction in size at hatching in laboratory study [e] • 2 mg/l tolerated for a short time by adults in tests [e] • < 1 mg/l lethal for adults [e]	• 5 mg/l MPCA standard applies • Prolonged period with DO less than 1.0 mg/l appears to cause winterkill
TEMPERATURE	• 8-15°C-preferred temp. range • 23.5°C-upper lethal temp. for adults	• 21-27°C-preferred temp. range of adult in summer [c] • > 32.3°C-upper lethal temp. for adults • 35°C-upper lethal temp. for juveniles in laboratory [c] • 38°C-upper lethal temp. for fry in laboratory [c]	• 24-30°C optimal temp. for growth of adults [d] • 26-28°C preferred temp. in field [b] • 36.4°C upper lethal temp. when acclimated at 30°C [b]	• 20-24°C preferred temp. for growth [e] • Upper lethal temp. reported as 29 to 32°C and 34 to 35°C in two studies	• young did not grow at temp. > 28°in laboratory • temp. > 32°C coincided with die-off in Missouri reservoir

Table 5. cont'd.

	LAKE TROUT	SMALLMOUTH BASS	LARGEMOUTH BASS	WALLEYE	NORTHERN PIKE
TRANSPARENCY	Mean and median for MN lakes – 4.8 m and 4.6 m respectively [g]	• Typical habitat has low turbidity ≤ 25 JTU and rarely > 75 JTU [c]	• Optimum suspended solids level are assumed to be 5-25 ppm, levels < 5 ppm indicative of low productivity	• Peak feeding at transparencies of approx. 1 to 2 m Secchi disk readings [e] • Decrease in activity at < 1 or > 5 m Secchi disk reading [e]	• Typically found Secchi disk readings of 2-4 m (in Ontario) [f]
pH	• pH 6.0-8.0-MDNR management criteria	• occur at pH 5.7-9, but optimum 7.9-8.1 [c] • lower lethal pH 3 [c]	• pH 5-10 required for successful reproduction [d] • optimal range is pH 6.5-8.5 [d]	• Typically found in pH range 6-9 [e] • pH < 6 associated with reproductive failures [e]	• Can persist in pH range 5.0-8.9
HABITAT	• Cold, oligotrophic, larger lakes [b] • Retreats to hypolimnion during summer months at southern boundary of range [b]	• Optimum lacustrine habitat characterized by large clear lakes and reservoirs with an average depth > 9 m with rock shoals • In heat of summer may retreat to greater depths. [b] However, found almost exclusively in the epilimnion during summer stratification in NE Wisconsin and Ontario. [c]	• Optimum lacustrine habitat are lakes with extensive (≥25% of surface area) shallow (≤ 6 m depth) sections to support submergent vegetation, but deep enough (3-15 m mean depth) to successfully overwinter [d]	• Most abundant in lakes or lake sections classified as mesotrophic although tolerant of a range of conditions [e] • Most abundant in moderate-to-large (> 100 ha) lakes that have extensive littoral zones and moderate turbidities [e]	• Occur in oligotrophic lakes but are more typical in mesotrophic and borderline eutrophic lakes [f] • Coolwater fish although rarely venture below thermocline [b]

Table 5. cont'd.

	LAKE TROUT	SMALLMOUTH BASS	LARGEMOUTH BASS	WALLEYE	NORTHERN PIKE
SPAWNING	• In fall over boulders and rubble bottom in water of 40' or less [b] • Prefer rubble 2.5 cm in diameter or larger	• In late spring, early summer on clean stone, rock or gravel substrate in 1-4' of water	• In late spring, early summer on a wide variety of substrates although prefer gravel in 1-4' of water [d]	• In spring during periods of rapid warming after ice-out [b] • Preferred spawning on rock-coarse gravel substrate with good water circulation [b] • Shallow shoreline areas, shoals, riffles [b]	• In spring after ice-out over vegetation in calm, shallow water [b]

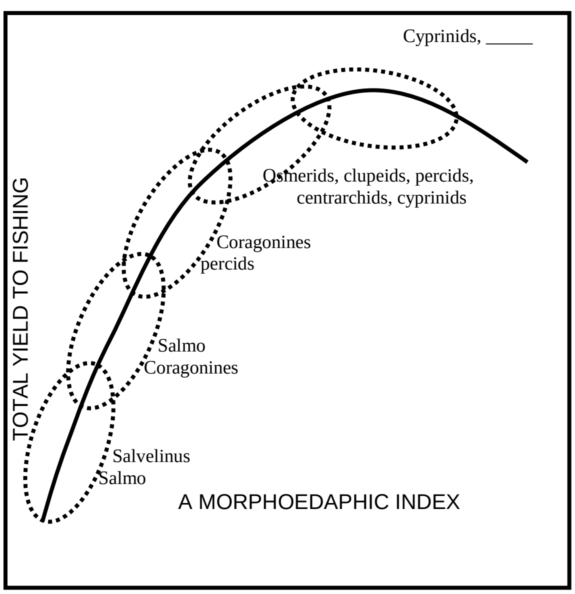
[a] Marcus et al. 1984; [b] Scott and Crossman 1973; [c] Edwards et al. 1983; [d] Stuber et al. 1982; [e] McMahan et al. 1984
[f] Inskip 1982; [g] Siesenop 2000

Less immediate impacts of eutrophication on salmonid species may include increase parasitism, possible introgression, reproductive failure and the overall development of environmental conditions favoring the growth of other species (Colby, et al., 1972). Spawning grounds may be affected through siltation of beds and/or low DO concentrations on spawning shoals. Excessive shoreline and shoal growth of Cladophora in Lake Ontario, for example, was suggested as having detrimental effects on lake trout reproduction (Christie, 1972). Environmental conditions more favorable to other species include shifts in the size, composition and relative abundance of prey as eutrophication occurs (Hayward and Margraf, 1987). Colby, et al. (1972) noted that in certain oligotrophic lakes a change in fish community from salmonids to coregonids to percids and eventually to centrarchids and cyprinids appeared to be related to changes in benthos from Mysis and Pontpereia to mayflies and eventually to chironomids and oligochaetes.

Possible changes in fish production and community structure in oligotrophic lakes undergoing eutrophication are depicted in Figure 12 where Colby, et al. (1972) used the morphoedaphic index (MEI as proposed by Ryder (1965)) as an indicator of lake productivity. Based on Figure 12, increases in total fish yield/biomass will occur as lake productivity increases, however undesirable species dominance shifts will also occur.

At some stages of cultural eutrophication, percids and other mesotrophic species may show increases in growth rates and production. However, as with the salmonids, further eutrophication will result in individual percid species responding unfavorably (Leach, et al., 1977). Conditions causing decline in percid species have been suggested to include changes in prey size structure, increases in parasitism and disease, changes in forage food organisms, decreasing DO and transparency resulting in changing distribution patterns, and alteration of spawning beds (Hayward and Margraf, 1987).

Figure 12. Suggested Relation Between Total Yield (both quantitative (kg/ha) and qualitative (taxonomic composition)) and an Index of Productivity (total dissolved solids/mean depth). Taken from Colby, et al. (1972).



The fish communities common to naturally mesotrophic and eutrophic lakes generally can be represented by the last three groupings on the MEI index curve (Fig. 12). In these lakes, anoxic or near anoxic conditions may occur in the hypolimnetic strata. Additionally, reduced oxygen concentrations may also be encountered in metalimnetic strata depending upon the degree of autochthonous and allochthonous loading rates. In general, species adapted to mesotrophic and eutrophic lakes do not have the lower temperature requirements of coldwater species and therefore may not be directly affected by the loss of oxygen in the hypolimnetic strata as they typically inhabit the warmer upper lake strata.

As lakes undergo eutrophication, there can be large fluctuations in DO over 24 hour periods and longer time cycles. These fluctuations can stress the fisheries. Under extreme eutrophic and hypereutrophic conditions, die offs of algae and macrophytes can cause severe oxygen depletion and entire fish populations can be lost to a summerkill (Barica and Mathias, 1979). High nutrient loading to these lakes may also cause winterkills (Welch, 1980). Additionally, partial kills of fisheries in areas of lakes are also potential outcomes of reduced oxygen concentrations (Shapiro, personal communication). Depending on the severity of the event, winterkill conditions may eliminate the most desirable sports fish, such as walleye and largemouth bass. Rough fish species able to tolerate short term extremely low oxygen conditions may survive in greater numbers such that in subsequent years, they can quickly increase their population densities.

Schupp (1992) expanding on some of these earlier concepts noted some distinct relationships among TSI and the presence, relative abundance, and size of several fish species. Black crappie, for example, tended to increase in abundance as TSI increased from about 55 (~35 µg/L TP) up to about 65-70 (~70 µg/L TP), however fish size tended to decrease over this range as well. Black bullhead abundance increased dramatically with increased eutrophication. Consistent with previous work, it was found that when all fish species were considered there was a distinct increase in pounds of fish as TSI increased and peaked near a TSI of about 60-65 (Fig. 13b). The number of fish species peaked at a TSI of 40 and remained fairly stable through a TSI of 65; thereafter a distinct decline in number of species was noted (Fig. 13b). However, as Bachman et al. (1996) note in their work on Florida lakes, piscivorous fish declined as a percentage of the total biomass as lakes became more productive.

Schupp and Wilson (1993) advanced these concepts further as they compared relative abundance and presence of various fish and water quality (as represented by TSI). For example the coldwater fishes: lake trout, whitefish and cisco exhibited peak abundance over a TSI range of about 30-40 (TP ~ 6 - 12 µg/L). Lake trout were generally not observed above a TSI of about 45 (TP ~ 17 µg/L), while whitefish and cisco were found at TSIs up to about 55-60 (TP ~ 34 - 48 µg/L). Species like walleye were abundant across a wide TSI range, peaking over a TSI range of ~ 40-50 (TP ~ 12 - 24 µg/L) -- typically large mesotrophic lakes, with low clarity and few rooted aquatic plants (Schupp and Wilson, 1993). Walleye relative abundance declines as TSI increases from its peak at about 40-50 (Fig. 13e). For example, at a TSI of 65-70 (TP ~ 70 - 100 µg/L) walleye relative abundance was estimated to be 25-30% (Figure 13e). Sight-feeders piscivores like northern pike and largemouth bass are most abundant at a TSI near 40 (Fig. 13f). While northern pike remain abundant over a TSI range from 40-60 (Secchi 4.0 – 1.0 m), largemouth bass decline in abundance over that same TSI range.

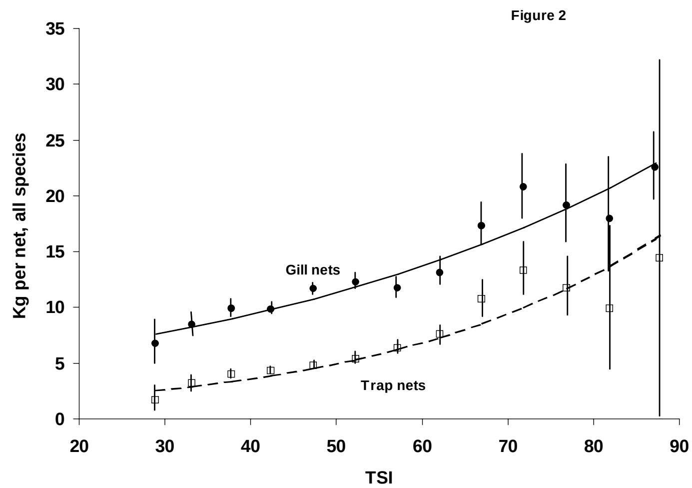
Perhaps the best fish indicators of water quality are two of the three bullhead species present in Minnesota (Schupp and Wilson, 1993). Yellow bullheads are found in the highest numbers in lakes with clear water. In contrast, black bullheads reach their highest abundance in very turbid, eutrophic waters (Fig. 13g). The ratio of blacks to yellow was proposed as an indicator of water quality (see Lake Sallie case study). Yellow bullhead relative abundance declines markedly over the TSI range from 30-65, whereas black bullhead relative abundance increases dramatically over a TSI range of about 60-70 (TP ~ 50-100 µg/L). The relative abundance of common carp increases precipitously over a TSI range of 60-80 as well, with large (step) increases noted as TSI increases from 60-65 (TP~50-70) and again from 70-75 (TP~100-140) (Fig. 13g).

Cultural eutrophication can lead to, or allow for, the dominance of common carp. And in turn, carp can further advance eutrophication via their feeding activities that may destroy rooted plant beds and promote recycling of TP from sediments – both of which favor the growth of algae. In experiments conducted by Parkos et al. (2003) adult carp were found to increase turbidity, total suspended solids, and total phosphorus presumably as a result of their feeding activities. They found further that carp negatively affected macrophyte abundance by reduction of light availability, increase of siltation rates, ingestion of plant matter or uprooting during feeding activities. They noted in particular that plant species with a weak root system, like Chara, were eliminated in the presence of common carp. They predicted that common carp would cause the largest changes in shallow systems. In a recent study of shallow lakes in southwestern Minnesota we observed that many of the lakes with highest TP and TSS concentrations often had extensive carp and black bullhead populations (Heiskary et al. 2003). And one that did not have carp (West Twin, Lyon County, inset photo) had relatively clear water and abundant growth of Chara, which was quite uncharacteristic of other NGP and WCP ecoregion lakes included in that study.

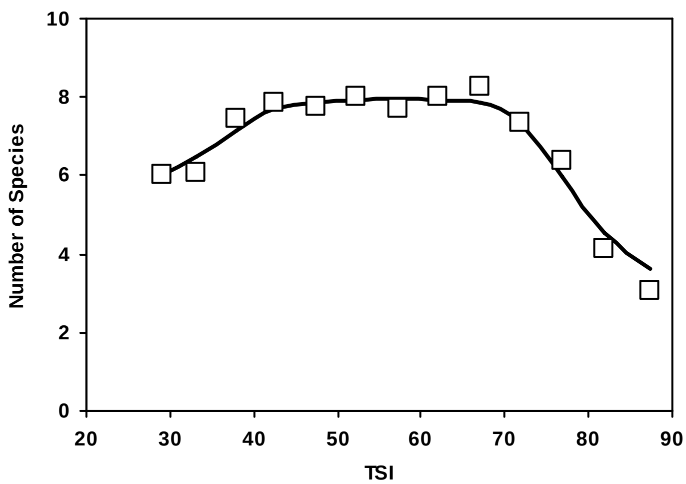
West Twin Lake, Lyon County, summer 2002 (Chara sp.)

Figure 13. Relative fish abundance as compared to lake trophic status (adapted from Schupp and Wilson, 1992 & Schupp unpublished data).

a.



b.



c.

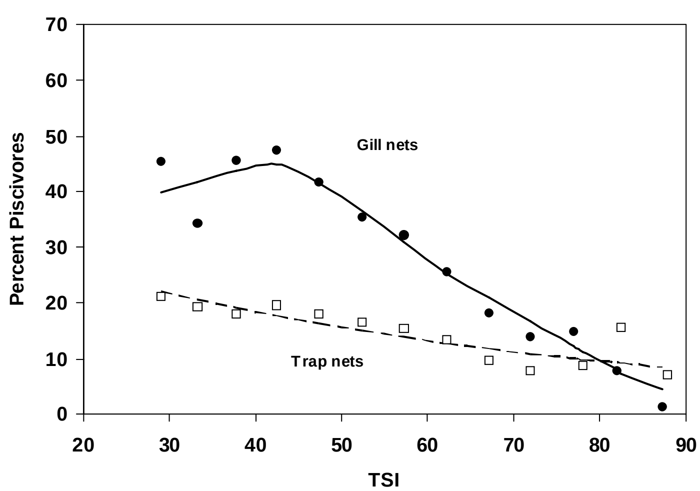
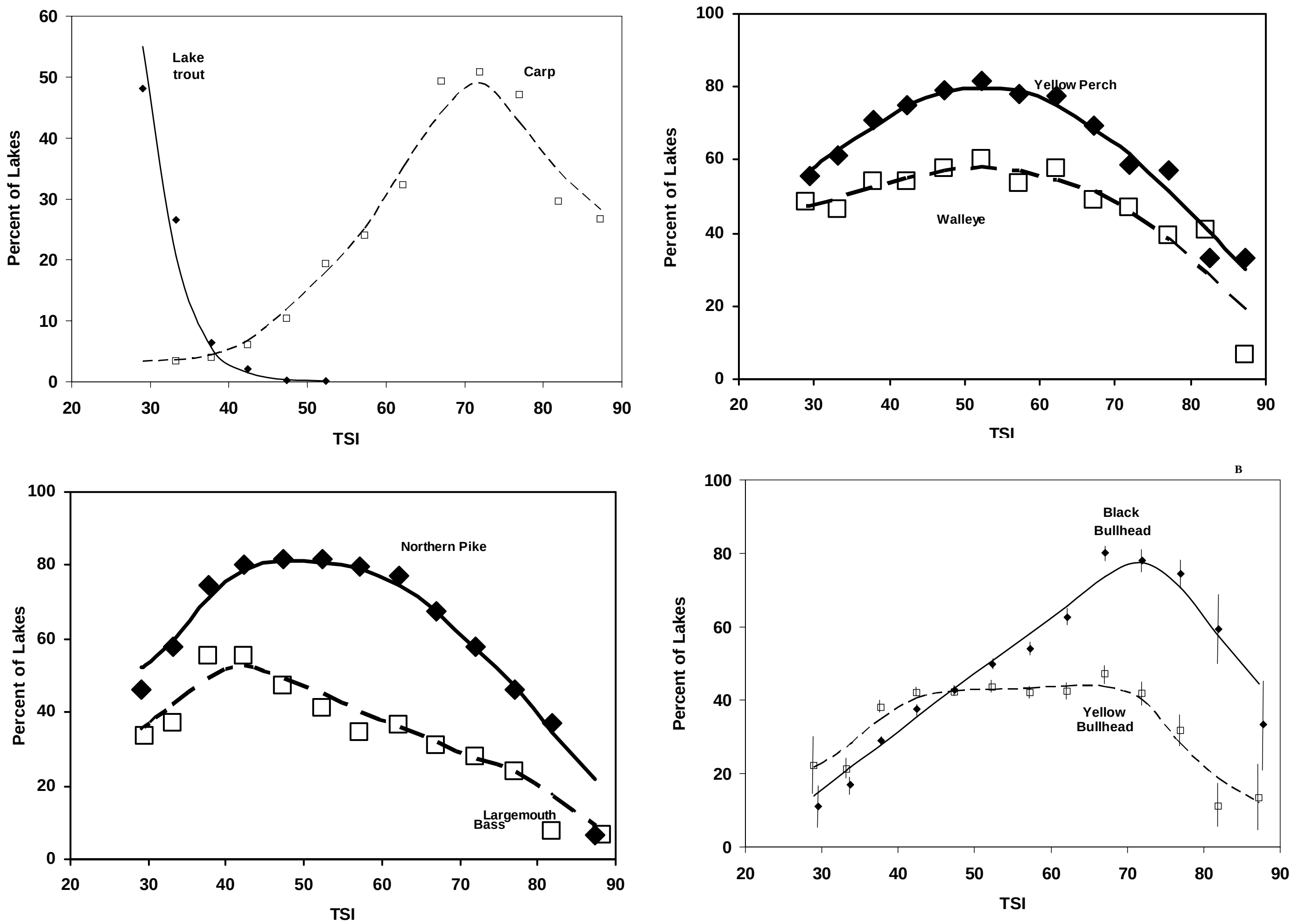


Figure 13 (cont.) d, e, f & g



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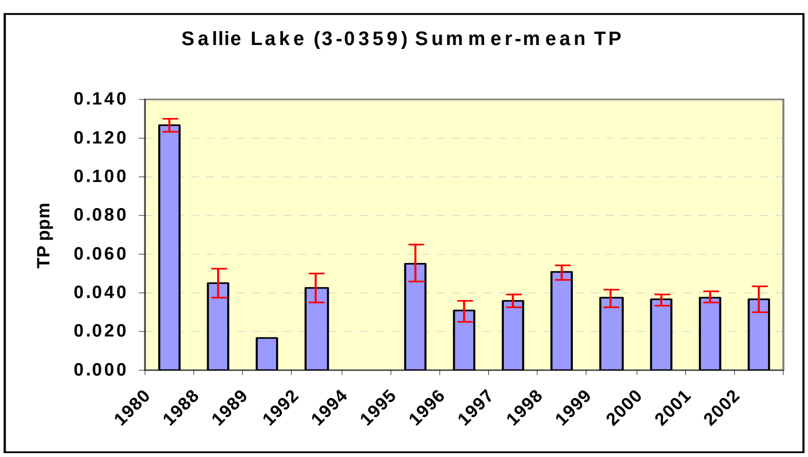
Case Study 2. Eutrophication impacts on a fishery: Lake Sallie (drawn in part from Colby, et al., 1987; Schupp and Wilson, 1993).

Lake Sallie (3-0359) in the CHF ecoregion, near Detroit Lakes, is an example of a Minnesota lake, which has undergone cultural eutrophication. The lake received sewage effluent from the Detroit Lakes wastewater treatment facility (WWTF) until diversion of the effluent in 1972. Lake surveys conducted in 1949 indicated high populations of walleye, perch, and suckers. Northern pike, bluegill, black crappie, and largemouth bass were common. The geographic area of the lake, along with lake morphometric characteristics, would strongly suggest that the lake community would be dominated by walleye and yellow perch.

As early as the 1950's, excessive growths of vegetation and nuisance blooms of algae were noted (Colby, et al., 1987). During the early 1970's, in-lake phosphorus concentration were on the order of 200-450 µg/l P (Brakke, 1977). By the 1970s, abundances of walleye, perch, and sucker were dramatically reduced (approximately 14, 34, and 20 % of peak abundances, respectively) Bullheads, present in low numbers in early surveys, were quite abundant by 1968. About 96% of the standing crop of bullheads was black bullheads, a species very tolerant of eutrophic conditions. The black to yellow bullhead ratio climbed to 1,265:1 (1971) and 3,437:1 (1972) from a ratio of 22:1 noted in a 1953 survey (Schupp and Wilson, 1993).

Subsequent to diversion of the sewage effluent from Lake Sallie in 1972, there were changes in the fisheries (Colby, et al., 1987). By 1981, walleye, perch, sucker, and bluegill populations showed evidence of recovery from the 1975 lows. Walleye abundance had tripled, perch nearly doubled, suckers increased by about 69% and bluegill abundance had nearly tripled. The abundance of bullheads declined to levels observed in the 1950's, with yellow bullheads dominating over black bullheads in 1981. Lake Sallie data indicate in-lake TP concentrations were on the order of 100 µg/l during the 1980's (Fig. 14). Concentrations have declined further with the implementation of various watershed and in-lake measures instituted in the 1990s.

Figure 14. Lake Sallie TP trends.



Dissolved Oxygen – Fishery Effects and Internal Loading Considerations

a. Dissolved oxygen, hypolimnetic oxygen depletion and internal loading

Maintaining adequate dissolved oxygen (DO) in the metalimnion and hypolimnion is a critical factor in determining whether a lake has suitable habitat for lake trout and other cold and coolwater fish species. Also the presence or absence of hypolimnetic DO and stratification stability may influence the degree and magnitude of internal recycling of phosphorus from lake sediments. Understanding these factors in relationship to lake trophic status is important to lake management and can play a role in establishing nutrient criteria.

While lake surface waters generally contain sufficient oxygen in the summertime, the input of organic materials into the hypolimnion/metalimnion may result in the depletion of oxygen. The creation of anoxic conditions in oligotrophic lakes can eliminate the coldwater fisheries or cause the fish to move to the upper oxygenated lake strata, which may be outside their temperature optima. The lack of oxygen in the bottom waters of fertile lakes, causes the sediments to release such dissolved constituents as inorganic phosphorus, ammonia, and hydrogen sulfide. Anoxic conditions in lakes may also favor the growth of blue-green algae, such as Microcystis (Reynolds and Walmsby, 1975). Also, Trimbee and Prepas (1987) found that blue-green algal biomass increased as the proportion of lake sediment covered by anoxic water increased even without increases in epilimnetic TP in some lakes. Therefore, it is desirable to define the relationship between measures of lake trophic status and the rate of the oxygen reduction. Determination of the rate of oxygen reduction will also estimate the amount of time that a stratified lake may have oxygen in its hypolimnion (or its bottom waters) and indicate the potential for internal loading of nutrients.

Cornett and Rigler (1979, 1980) found that areal hypolimnetic oxygen depletion (AHOD) is correlated with epilimnetic total phosphorus and annual ^{14}C primary production and inversely correlated with mean summer Secchi depth. Hypolimnetic oxygen depletion rates (ODR) have been used as a trophic state index by several other investigators (Hutchinson, 1938; Mortimer, 1941; OECD, 1982; Rast, et al., 1983; Welch and Perkins, 1979). In general, values of 250 mg $0_2/(m^2$ day) served as the upper limits for oligotrophy and 550 mg 0_2 (m^2 day) served as the lower limit for eutrophic lakes. Welch and Perkins (1979) found that AHOD was well correlated with phosphorus loading corrected by flushing rate (L/p). They observed that phosphorus loading above Vollenweider's (1976) critical loading rate of 200 $(Z)^{0.5}$ or approximately 630-775 mg P/(m^2 year) for lakes with mean depths ranging from 10-15 m, would likely have significant negative impacts on fisheries, particularly the coldwater species. Overall, AHOD rate was found to be correlated with epilimnetic chlorophyll-a concentrations and other surface-water measures of trophic status, including total phosphorus, transparency, and organic nitrogen.

Walker (1979) used trophic status, AHOD, mean hypolimnion depth and oxygen concentration at spring overturn to predict the effective number of days of oxygen supply present in the hypolimnion after spring turnover (in days) as follows:

T_{do} = Oi Zh/AHOD
T_{do} = the effective number of days of oxygen supply present in the spring.

where Oi = oxygen concentration at spring overturn (mg/m^3);
Zh = mean hypolimnion depth (m);
AHOD = areal hypolimnetic oxygen depletion rate (mg/m^2/day)

Zh, the mean hypolimnetic depth, may be measured directly based upon (1) dissolved oxygen (DO) profiles and lake bathymetric maps; or (2) from empirical relationships.

$$Zh = z\,(1 - Zt/Zm)$$

and

$$Zt = 1.6\,(Zm)^{0.57}$$

where Zh = mean hypolimnetic depth (m);
Zm = maximum lake depth (m);
Zt = average thermocline depth (m);
z mean depth (m).

Walker (1979) used this relationship to examine the effects of phosphorus concentration and lake morphometry in controlling the oxygen status of lakes. For this purpose anoxic conditions were defined to occur if the Tdo was less than 200 days of oxygen supply in the hypolimnion (e.g., oxygen supply through mid-October, which is typical of north temperate lakes). The relationship (Walker's Figure 11) indicated: (1) that for typical lakes, phosphorus concentrations exceeding 10-15 µg/l would produce anoxic conditions using the Tdo of 200 days; and (2) the oxygen status is most sensitive to phosphorus levels in lakes with mean depths between 5 and 10 m. For lakes with mean depths exceeding this range, the hypolimnetic oxygen status is less sensitive to depth as increasing supply of oxygen is available in the hypolimnion after stratification.

Volumetric HOD rates may be more important than areal HOD rates to water quality managers, since this rate will determine the decline of the average hypolimnetic concentrations of oxygen during the summer for stratified as well as nonstratified lakes. Walker (1985b) developed volumetric oxygen depletion relationships based upon the dependence of AHOD upon chlorophyll-a as moderated by lake morphometry (i.e., mean depth). For natural lakes, the volumetric hypolimnetic depletion rate was calculated as follows:

$$\log(HODV) = 2.34 + 0.45 \log(Bs) - \mathrm{Log}(Zh) \qquad \text{(Eq. 6)}$$

where Bs = station mean chlorophyll-a in mg/m^3 and
Zh = mean hypolimnetic depth in m.

This model explained 92% of the variance in lake HODV rates with a mean squared error of 0.020.

Observed depletion rates for reservoirs were considerably higher than predicted for lakes due to spatial variables (e.g., longitudinal gradients) in reservoirs. For the U.S. Army Corps of Engineers (USACE) data set used in the model development, HODV varied from 36 - 443 mg/m^3 day with an average value of 77 mg/m^3/day. Usually, for a HODV greater than 100, hypolimnetic oxygen supply will be depleted in less than 120 days after stratification, assuming a maximum spring oxygen concentration of 12 mg/m^3. Shallow reservoirs may have extremely high oxygen losses in excess of 500 mg/m^3 day. As a result, shallow reservoirs can very quickly lose oxygen concentrations in their bottom waters subsequent to collapse of an algal bloom.

The volumetric oxygen depletion rate occurring in the metalimnion may also be of significance to fisheries maintenance. Walker (1985b) estimated average metalimnetic oxygen depletion (MOD) by difference from HOD rates calculated at the upper and lower boundaries of the metalimnion. The best relationship between MOD and HODV was:

$$\log(\mathrm{MOD}) = 0.40 + \log(\mathrm{HOD}) + 0.38 \log(\mathrm{Zh}) \quad (R^2 = 0.86,\ SE^2 = 0.011) \qquad \text{(Eq. 7)}$$

where MOD = Metalimnetic oxygen depletion rate (mg/m^3-day)
HODV = Hypolimnetic oxygen depletion rate (mg/m^3-day)
Zh = mean hypolimnetic depth (m)

These relationships may then be used for prediction of the impacts of phosphorus loading upon oxygen concentrations during the stratified season (e.g., BATHTUB, Walker (1985b)). Comparison of the impacts may include estimation of Tdo as outlined above, or in estimating the reduction of the metalimnetic or hypolimnetic depletion rates. These results may be evaluated for consistency with fisheries management strategies. For example, coldwater fish, such as lake trout, require the cooler temperatures associated with the hypo or metalimnion of deep stratified lakes and require oxygen concentrations greater than 5 mg/l as noted in Table 6.

Using data from the ecoregion lake monitoring program, we sought to correlate measures of trophic status to various classes of lakes based on hypolimnetic oxygen concentrations. These classes of lakes may be useful for the purposes of predicting the likelihood of internal loading or likelihood of adequate oxygen for support of a coldwater fishery. Hypolimnetic Oxygen Classes (HOC) were assigned as follows, based on lake profiles obtained in late August from a subset of the ecoregion lakes (Heiskary and Wilson, 1990).

Class	Description
A.	Severe oxygen depletion resulting in hypolimnetic concentrations less than 1.0 mg/l through the entire hypolimnion. If there was only one lake station for analysis and the measured D.O. was less than 1.0 mg/l, then this was the default category.
B.	Moderate oxygen depletion resulting in concentrations less than 1.0 mg/l somewhere in the hypolimnion, but not the entire hypolimnion.
C.	Lakes with oxygen concentrations greater than 1.0 mg/l throughout the hypolimnion.
D.	Lakes with oxygen concentrations greater than or equal to 5.0 mg/l throughout the hypolimnion.

There were no dimictic lakes in the reference lake data set that had hypolimnetic oxygen concentrations exceeding 5 mg/l (Class D). However there were six lakes with hypolimnetic oxygen concentrations exceeding 1.0 mg/l (Class C) in late August and in these, the average epilimnetic TP ranged from 13-27 µg/l. It was observed that for lakes that had oxygen greater than 1.0 mg/l in portions of the hypolimnion (Class B), the average epilimnetic TP values ranged from 15-40 µg/l. Lakes showing anoxic or near anoxic hypolimnetic conditions had average epilimnetic TP values ranging from 14-160 µg/l in this data set. While these lakes are believed to be minimally impacted by point and nonpoint sources of pollution and were from the CHF and NLF ecoregions, none of the lakes had hypolimnetic oxygen concentrations greater than or equal to 5 mg/l (Class D) at the end of the summer.

Based upon the interrelatedness of measures of trophic status, such as phosphorus, chlorophyll-a and Secchi depth, to hypolimnetic oxygen concentrations, it is reasonable to define approximate ranges of trophic status that have significance to fisheries management strategies. These relationships provide links between phosphorus and dissolved oxygen criteria and existing D.O. standards. Areal and volumetric measures of hypolimnetic oxygen depletion vary directly with total phosphorus concentrations as modified by lake morphometry (Walker 1979 & 1985b). For typical lakes, total phosphorus concentrations above 10-15 µg/l will usually result in the depletion of hypolimnetic oxygen concentrations. Our analysis of 74 minimally impacted Minnesota lakes tends to confirm this observation (Heiskary and Wilson, 1990).

Nordin (1986) examined hypolimnetic oxygen depletion and phosphorus relationships from lakes in British Columbia. In this analysis, a range of total phosphorus concentrations between 5-15 µg/l was proposed for the protection of salmonid (coldwater) fisheries. It was noted that oxygen depletions generally began to occur when TP concentrations exceeded 10 µg/l, which is often used as an upper boundary for oligotrophy (Nurnberg, 1996). Expressed in other terms, an average summer chlorophyll-a of 2 µg/l and a corresponding summer-mean Secchi depth of 4.5 m and a AHOD of 0.25 g/m^2/day also generally describe the transition from oligotrophy to mesotrophy (Rast and Lee, 1983).

In summary, two considerations need to be made with respect to hypolimnetic oxygen depletion:

1. Coldwater fish require adequate oxygen in the cooler waters of the hypolimnion for survival. Without oxygen, they are required to seek depth of the lake with adequate oxygen (e.g., epilimnion). However, high temperatures stress and may kill cool or coldwater species.

2. Hypolimnetic anoxia causes increased phosphorus release from sediments. However, this may not create an algal problem unless this water reaches the photic zone during the growing season. In general, it is not expected to be a significant problem in dimictic lakes with stable stratification over the summer. However, in lakes with weak or temporary (intermittent) stratification, the phosphorus released from the sediments may enter the photic zone periodically during the summer and lead to extensive algal blooms.

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Case Study 3. Significance of HOD and internal phosphorus recycling on the lake condition: Fairmont Chain of Lakes.

A study (Stefan and Hanson, 1981) of the Fairmont Chain of Lakes, which includes Budd Lake (46-0030), was prompted due to blue-green algal problems that created nuisance conditions to lake users, and taste and odor problems for the municipal water supply. These five lakes have mean depths ranging from 2-5 m, Secchi transparency from 0.75 – 1.5 m, total phosphorus from 30-150 µg/l and chlorophyll-a concentrations above 50 µg/l in the summer. Domestic animal poisonings due to blue-green algal blooms have been reported (Stefan and Hanson, 1981). It was found that oxygen depletion varied from 600-1,650 mg/m^3-day. At these rates, there would typically be about 5-13 days of oxygen in the bottom waters based on an initial D.O. of 7 mg/l and an HOD of 600 mg/m^3/day. It was theorized that there was significant time during the growing season that anoxic conditions existed along the sediment-water interface as a result of the oxygen depletion.

Anoxic conditions were monitored over the chain of lakes and internal loading of phosphorus was found to be a substantial portion of the P load to these lakes. Surface water TP concentrations were found to increase over the course of the summer due to mixing of the hypolimnetic phosphorus into the euphotic zone (Stefan and Hanson, 1981). This was confirmed in more recent sampling as well. In 2002, the five lakes in the chain exhibited temporary stratification and low DO (< 2 mg/L) on one or more sample dates (Schlorf Von Holdt and MPCA, 2003). Hypolimnetic TP typically increased under these conditions and then decreased as the lakes became well-mixed. Epilimnetic TP increased from spring to late summer on most of the lakes. TP concentrations for these lakes were in the 40-70 µg/L range in May and peaked at about 120-140 µg/L in August (Fig. 10).

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Case Study 4. Interrelatedness of lake morphometry, stratification stability and internal phosphorus loading: Shagawa Lake (69-0069).

The interrelatedness of lake morphometry, stratification stability and internal loading of phosphorus was examined by Larsen, et al. (1981) and Stewart and Stauffer (1982). Internal phosphorus loading was shown to be enhanced by summertime mixing due to the passage of cold weather fronts. In the case of Shagawa Lake, vertical mixing of anoxic bottom waters rich in phosphorus was found to be a significant source of nutrients to the photic zone. The anoxic zone was found to include a large fraction of the mid-depth shelf sediments extending from about 6 to 10 m. Mixing of the lake was occurring through these depths during summer storms and hence introduced phosphorus from the bottom waters into the euphotic zone. This internal loading of phosphorus has slowed the recovery of Shagawa Lake, which had an 80 percent reduction in its external phosphorus load in 1973 as a result of tertiary treatment of Ely's wastewater effluent. Based on monitoring to date summer-mean TP has stabilized in the 25-35 µg/L range (Fig. 15). The influence of internal P loading is evident in monitoring data from 1998 (Fig. 16). On June 9th the lake was fairly well mixed with weak DO stratification evident. Epilimnetic and hypolimnetic TP concentrations were relatively uniform. By July 5th the lake was stratified and hypolimnetic TP began to increase consistent with the low oxygen concentrations near the sediments. Stratification continued at site 102 through the July 14th monitoring and hypolimnetic TP rose further. On August 24th the lake had mixed to a depth of about 8 m. This mixing allowed for the reintroduction of hypolimnetic TP into the epilimnion causing a rise in epilimnetic TP. Algae responded to this increase in TP as bloom conditions were evident on August 24th (Fig. 16).

Figure 15. Shagawa Lake summer-mean TP.

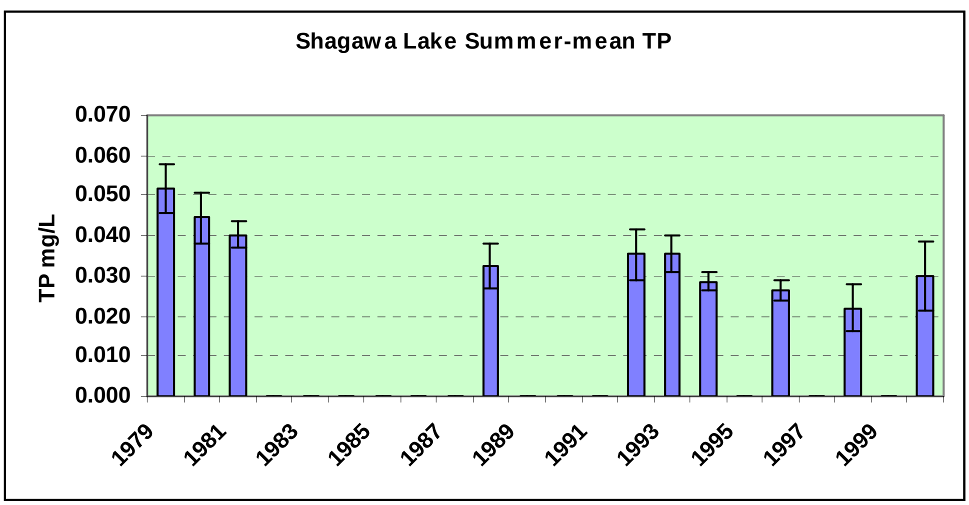
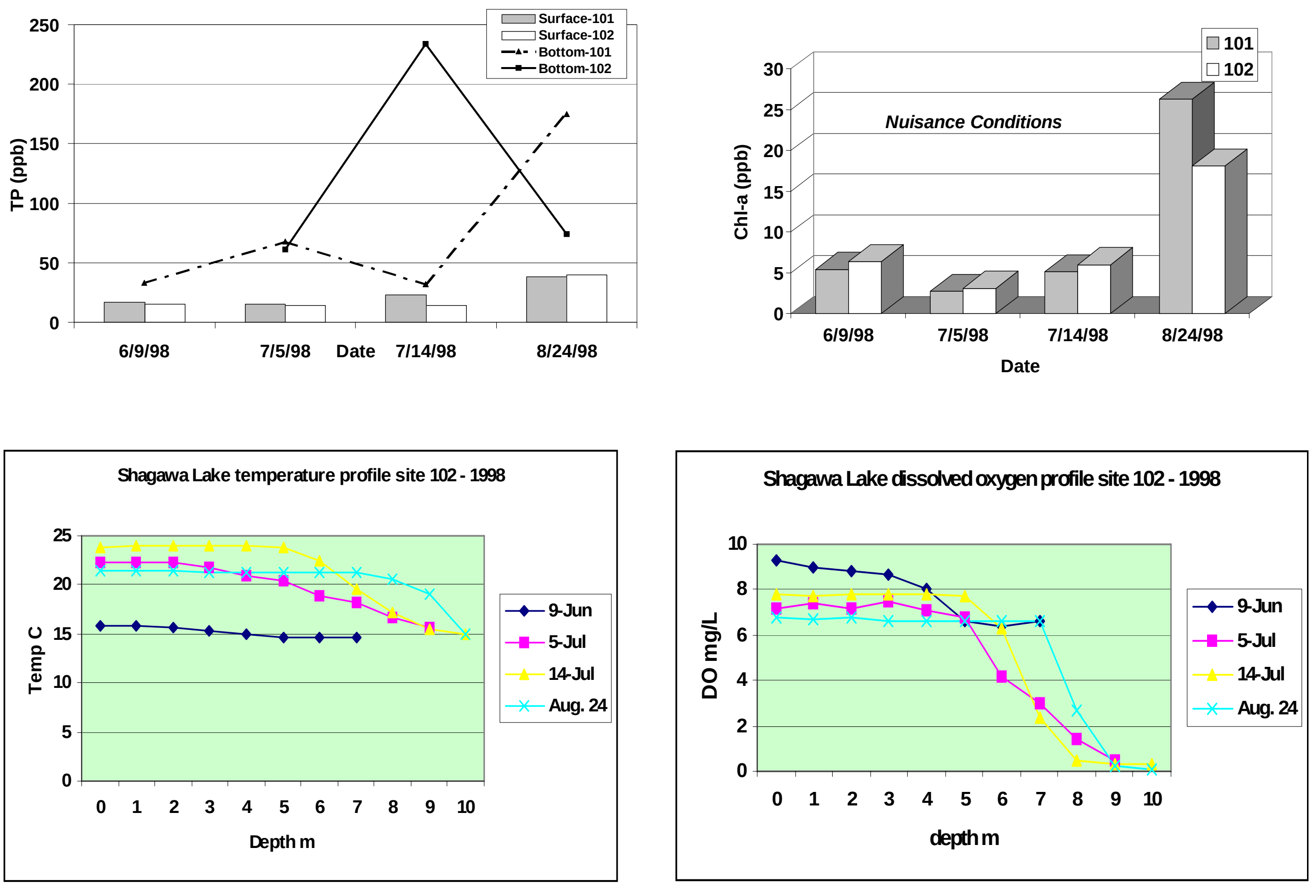


Figure 16. Shagawa Lake 1968: TP, chlorophyll-a, DO and temperature.



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b. Dissolved oxygen and fishery effects: lake trout lakes

As is evident in Table 5 the environmental requirements for support of lake trout are fairly stringent with respect to DO, temperature, and trophic status, as compared to most warm water species. For example, Siesennop (2000) in an assessment of Minnesota lake trout lakes suggests DO concentrations of 6 mg/L and temperatures of 12 C or less as bounds for suitable habitat. Minnesota lakes known to have lake rout range from 27 – 7,122 acres in size (mean = 865 acres), maximum depths range from 8 – 85 m (mean = 31 m), and mean depths range from 3.8 – 27.0 m (mean = 11.1 m) (Siesennop, 2000). Siesennop (2000) notes further that almost 50% are less than 250 acres. Minnesota's lake trout lakes represent the southern boundary of this resource in North America and are justifiably outstanding resources. Also since they are on the southern boundary they tend to be warmer than those to the north (e.g. Canada), which can potentially stress lake trout populations. Minnesota's natural lake trout lakes are specifically listed as Outstanding Resource Value Waters (ORVW) in MN Rule Ch. 7050.0470 (Appendix III). The list also includes some potential lake trout lakes that have been considered for listing by the MDNR. Review of MDNR fishery survey reports for several lake trout lakes

(http://.dnr.state.mn.us/lakefind/index.html) indicate the presence of naturally reproducing lake trout populations in lakes such as Greenwood, Hungry Jack, Gunflint, Snowbank and Trout (16-0049). Heiskary and Wilson (1988) proposed a TP criteria level of < 15 µg/L for the protection of lake trout lakes. As such we thought it relevant to review data for Minnesota's lake trout lakes and determine whether this concentration range is still appropriate and if there should be accompanying chlorophyll-a and Secchi criteria as well for this class of lakes. These lakes have been sampled periodically by MPCA staff and volunteers between the mid 1970s and 2000 through various monitoring programs including CLMP, LAP, and various regional studies that focused on status and trends in eutrophication, mercury, and acid rain impacts. Limited water quality data are also collected by MDNR as a part of their fishery surveys.

Summer-mean data (TP, chlorophyll-a, Secchi and DO and temperature profiles) were assembled for existing ORVW listed lake trout lakes, potential lake trout lakes (considered for ORVW listing), and potential lakes trout lakes (no longer considered for ORVW listing) from STORET (Table 6). This resulted in a total of 31 lakes with TP data, 19 with chlorophyll-a data and 32 with Secchi data.

The typical (IQ) range for summer-mean TP for these lakes was 9-16 µg/L with a median of 12 µg/L. Typical standard error for all lake means was about 2 µg/L, which translates to a coefficient of variation (CV) of about 17 percent of the mean. Trout Lake (31-0216), a potential trout lake (continuing to be proposed for ORVW) near Coleraine exhibited the highest TP concentration of this set of lakes. This lake previously received wastewater effluent from the cities of Bovey and Coleraine (see Case Study 5). Since the cessation of this discharge to the lake TP concentrations have declined. The lowest TP concentrations noted were found in Kabekona, Bluewater, and Little Trout Lakes. Of the 18 existing trout lakes with TP data only five exhibited TP greater than 15 µg/L. Of these five lakes the means were often based on only one or two measurements that were often over 10 years old. The lake with the highest TP (25 µg/L) among this group, Loon (16-0448), was based on only two measures of 12 and 37 µg/L respectively (possible outlier value). DNR fishery survey reports for Loon indicate an average catch of lake trout.

Table 6. Trophic status summary for ORVW lake trout lakes.
Based on lake-summers with two or more observations. [1]

Parameter	# of obs	Mini-mum	25th %tile	Median	75th %tile	Maxi-mum
TP (µg/L)	75	2.0	9.0	12.0	16.0	69.0
Chlorophyll-a (µg/L)	57	0.5	1.5	2.3	3.3	24.3
Secchi (m)	299	2.1	4.1	5.0	5.8	10.4

1. Multiple years of data for given lakes are reflected in this summary.

One concern with some of the older data in this data set is the detection limit for TP, which has been changed over time, and the high proportion of values near the detection limit, which may raise some concerns regarding precision and accuracy of measurement. For example the MDH detection limit employed at the time of a 1981 fly-in survey of lakes in the Boundary Waters Canoe Area Wilderness was 5 µg/L (Heiskary and Thornton, 1983). In recent years (1990s) the standard detection limit has been 10 µg/L, with a "low" detection limit of 2 µg/L employed as needed (when the laboratory encounters non-detects). This could be a concern if we are looking at trends over time for these lakes (or any lakes that are near the detection limit). Bearskin Lake (16-0228), for example, has one of the better data sets among the lake trout lakes (Fig. 17). Data from 1979 and 1980 suggest concentrations on the order of 25-28 µg/L, while 1981 and 1995 (LAP study) data suggest concentrations of about 12-13 µg/L. In the case of the 1979 and 1980 datasets, maximum values of 39 and 42 µg/L may have biased the summer-mean upward. In contrast, in 1981 and 1995 maximum values of only 23 and 16 µg/L were noted. Is this a trend, function of detection limits or simply some high values? At this point we are unsure and chlorophyll-a values were lacking in the earlier data set that would help corroborate the TP data.

Chlorophyll-a concentrations are quite low in these lakes with an IQ range of 1.5-3.2 µg/L. The lowest mean concentration was 0.5 µg/L for Trout Lake (16-0049) at Grand Marais and the highest was 24.4 µg/L for Trout Lake (31-0216) at Coleraine. There is very limited chlorophyll-a data on these lakes, the few with multiple years of data are displayed in Fig. 14. Chlorophyll-a is quite stable in some lakes like Burntside and Wabana where standard error of the mean is often 1 µg/L or less, which translates to a CV of 10-20 percent of the mean. In Trout Lake, at Coleraine, where summer-mean chlorophyll-a was quite variable, but declining over time, standard error of the mean ranged from 1-6 µg/L and averaged 2 µg/L, which was about 19 percent of the long-term mean chlorophyll-a (10.5 µg/L).

Secchi transparency data was more widely available for these lakes with an IQ range of 4.1 - 5.8 m based on 299 summer-mean measures for these lakes (Table 6). The median of 5 m is quite similar to the 4.6 m median that Siesennop (2000) noted in his review of Minnesota lake trout lakes. These transparencies are high as compared to the typical range for NLF ecoregion reference lakes (2.4-4.6 m). The high transparency promotes algal growth at greater depths in the lake often resulting in a metalimnetic maximum for oxygen (e.g., Fig. 21b Bearskin Lake). About 25 lakes had eight or more years of Secchi data that allowed for characterization of trends and inter-annual variability in these lakes. Of these 25 lakes 56% exhibited a slight increase in Secchi transparency over time, with the remainder generally exhibiting no long term trend. The overall average Secchi residual (long-term mean Secchi minus annual mean) was 0.5 m or 10% of the mean. Gunflint, Little Trout (31-0394), and Blue (29-0134) were among the few that exhibited a weak decline in transparency (Fig. 19). Secchi residuals ranged from 0.2 m (3%) for Little Trout to 0.5 m (14%) for Gunflint Lake. Trout (31-0216), Lower Hay and Grindstone Lakes exhibited notable increases in transparency and residuals of 1.3 m, 0.6 m, and 0.6 m (37%, 14% and 17%) respectively (Fig. 19). The range of residuals described for these lakes is consistent with that described based on trend assessments on CLMP lakes where typical residuals are on the order of 0.6 m or 20% or less of the long term mean for lakes that do not exhibit a long term trend (Heiskary and Lindbloom, 1993).

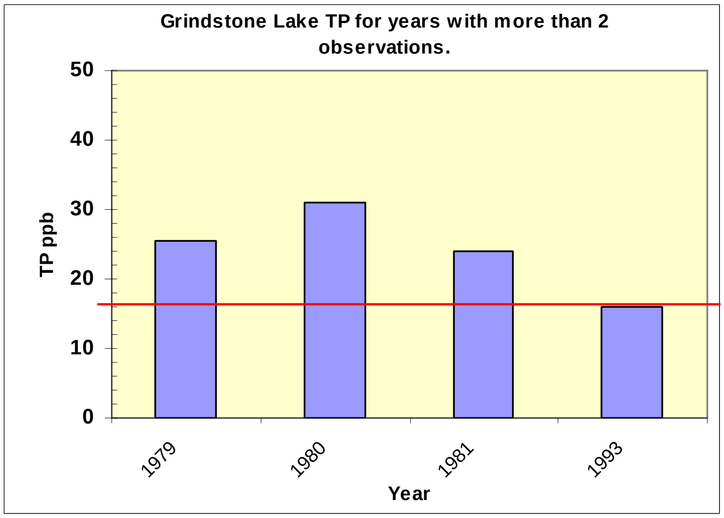
Carlson's TSI has frequently been used as a basis for interrelating TP, chlorophyll-a, and Secchi measurements. Paired measures of these three trophic status variables were converted to TSI and charted to view their correspondence for the lake trout lakes (Fig. 20). A "confidence interval" of ±5 TSI units has been added to allow for visual comparison of values. TSI_P and TSI_C agree fairly well over a TSI range from about 30-55 (TP~8-34 µg/L), though there is a slight tendency for TP based TSIs to exceed chlorophyll-a based TSI for several lakes for TSIs from 40-60. This suggests that TP is high relative to the chlorophyll-a samples. A possible explanation is that chlorophyll-a samples are collected from the upper 2 m, which would exclude algae (productivity) that may be growing at lower depths (e.g., metalimnion) because of the high transparency. Also if large Daphnia are abundant in a lake they will feed heavily on algae and partially account for low chlorophyll-a relative to TP. The poor agreement at the lower end (TSI_P=14) is likely a function of precision of measurement of TP and chlorophyll-a near the detection limits. TSI_C and TSI_S measures exhibit good agreement over the range of values, with a slight tendency for high TSI_C relative to TSI_S as TSI_C exceeds about 45 (chlorophyll-a ~ 4 µg/L) (Fig. 20). The final comparison is for TSI_S and TSI_P, which exhibit poor correspondence over the range of values in Fig. 16. Again, as with chlorophyll-a, there is a tendency for TSI_P to exceed the TSI_S, especially as TSI_P exceed about 40 TSI units. Again detection limits and other previously mentioned factors come into play.

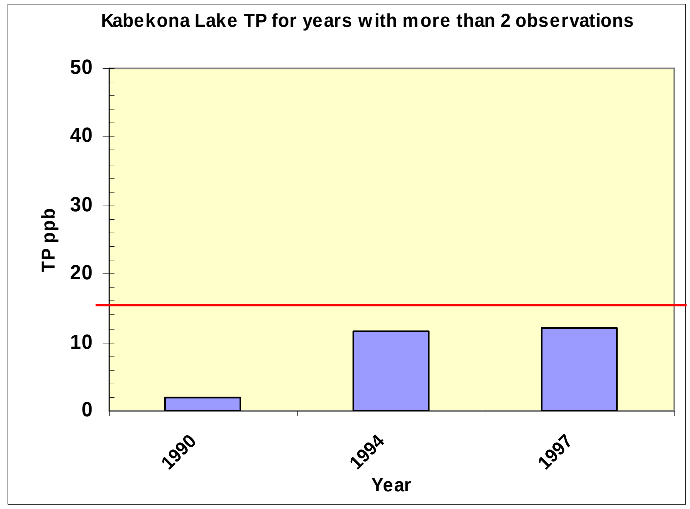
Maintaining adequate oxygen (~5-7 mg/L) in the hypolimnion is essential to maintaining a lake trout fishery as previously noted. Dillon et al. 2003 define the optimal habitat boundary for lake trout as the portion of the lake having more than 6 mg/L DO and temperature less than 10C – though they do acknowledge that some populations can be successful at higher temperatures under some circumstances. Late summer is a good index period for determining whether suitable habitat is present as stratification has been firmly established in the lake for several weeks (i.e., no new oxygen introduced into the hypolimnion) and lake temperatures will tend to increase causing an increased oxygen demand and often a reduction in suitable refuge for lake trout. MDNR (2000) note in a fishery survey for Snowbank Lake, a lake with good natural reproduction of lake trout, that "on July 18, 2000 DO was 6.1 mg/L and temperature was 43 degrees F (~8 C) at a depth of 95 feet." As such we conducted a review of late summer profiles for 15 lake trout lakes to determine if there appeared to be a "refuge" in which temperature was between 8-15 C and DO was > 5 mg/L. Of 15 lakes assessed (Fig. 17a) 73 % exhibited this refuge for the time periods assessed, 20% were periodically outside this range, and one lake (Long Lake 01-0089) did not exhibit optimal habitat for the assessed period (Fig. 17). Based on this limited sample of lakes it appears that the "refuge" for lake trout is relatively large in some lakes like Snowbank and Burntside, for example, but relatively small in some lakes like Kabekona and Long (29-0161). In lakes like Kabekona and Long DO fell below 5 mg/L in the upper part of the hypolimnion –leaving the optimal habitat (DO > 5 mg/L and temperature between 8 and 15 C) to a thin band (5 feet or less) in the metalimnion based on late summer profiles. In contrast, lakes like Greenwod, Trout (16-0049), Bearskin, and Snowbank offered optimal habitat over a depth of 20-30 feet or more. The loss of oxygen in the hypolimnion during mid to late summer stratification is evident in Fig. 17b. Based on these three examples the change from July to August in Burntside and Bearskin Lakes was much less than that observed in Long Lake (29-0161). In the case of Long Lake hypolimnetic D.O. was near 5 mg/L in July but fell to 3 mg/L or less in August.

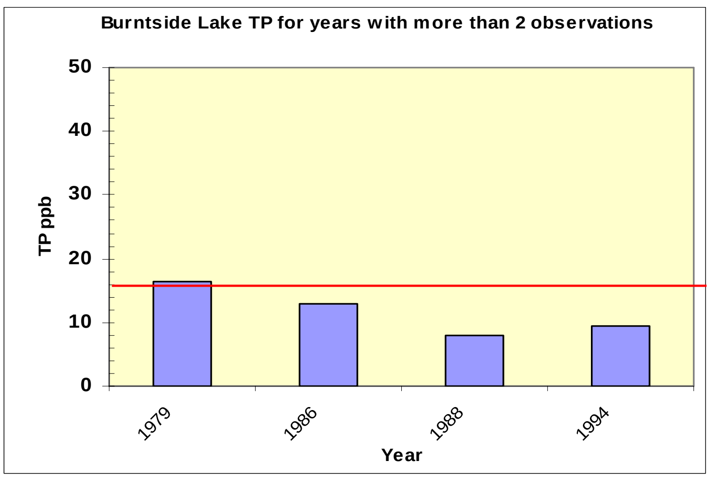
A review of TP and chlorophyll-a data for the lakes that afford optimal habitat for lake trout, based on late summer DO and temperature profiles, suggests that summer-mean TP is generally <15 mg/L and typically in the 8-10 µg/L range. Summer-mean chlorophyll-a averaged 3 µg/L and maximum values were 3-4 µg/L or less in these lakes. A notable exception was Grindstone Lake that exhibited a mean TP and chlorophyll-a of 16 and 6.3 µg/L and maximum chlorophyll-a of 7.8 µg/L. Assessed lakes that did not offer optimal habitat or a very small zone, such as Kabekona, Long, Trout (31-0216) and Wabana also had mean TP concentrations < 15 µg/L and mean chlorophyll-a on the order of 3-4 µg/L and maximum values ranging from 4.4 (Kabekona) to 10.4 µg/L (Trout). While our analysis does not reveal absolute TP or chlorophyll-a thresholds, with respect to maintenance of a lake trout fishery, it does suggest that 15 µg/L is probably the upper threshold for summer-mean TP. Likewise, this value seems to be consistent with an upper range proposed by Nordin (1986) for lake trout lakes in British Columbia. However, 15 µg/L is above the oft-recognized limit (10 µg/l) between oligotrophic and mesotrophic conditions (Nurnberg, 1996). Further based on Schupp's analysis (Fig. 13d) the percent of lakes with lake trout declined below 5% at a TSI < 40 (TP= 12 µg/L).

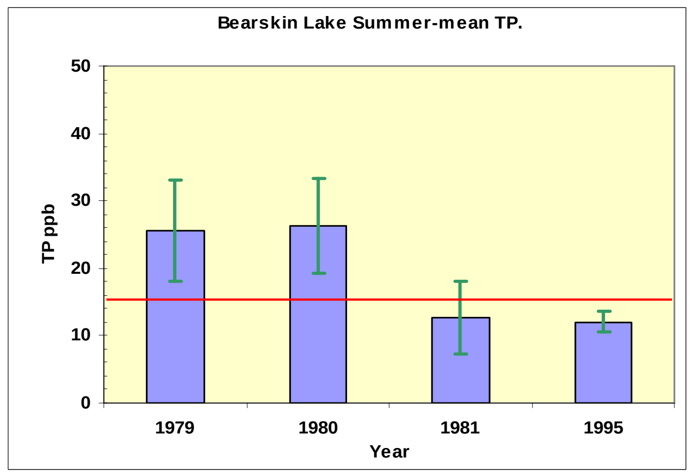
Also, it appears that summer-mean chlorophyll-a should remain below 3 µg/L and maximum values should generally remain below about 4 µg/L to minimize the impact of algae (organic matter) on hypolimnetic DO concentrations. This would be close to the oft-recognized limit between oligotrophy and mesotrophy (3.5 µg/L; Nurnberg, 1996). Based on the IQ range in Table 6 typical summer-mean transparencies for these trout lakes lies between 4.1 – 5.8 meters, which roughly corresponds to chlorophyll-a values of 2.5 and 1.5 µg/L respectively, based on Carlson's TSI (Fig. 3).

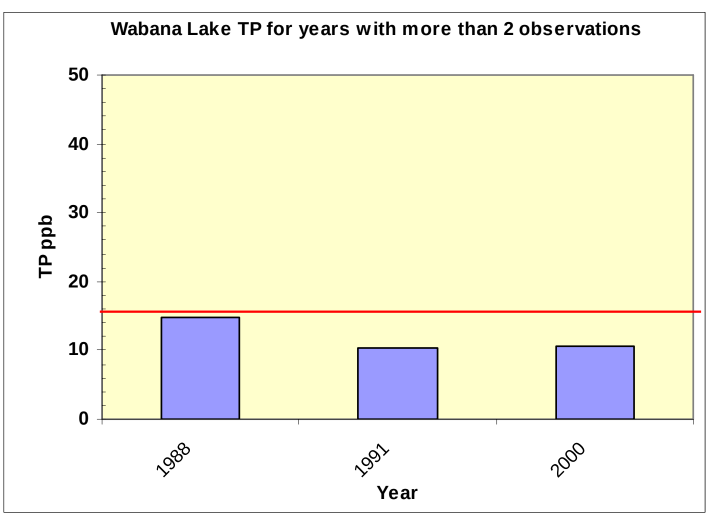
Figure 17. Summer-mean total phosphorus for select lake trout lakes. Standard error and 15 ppb criteria (Heiskary and Wilson, 1988) level noted.











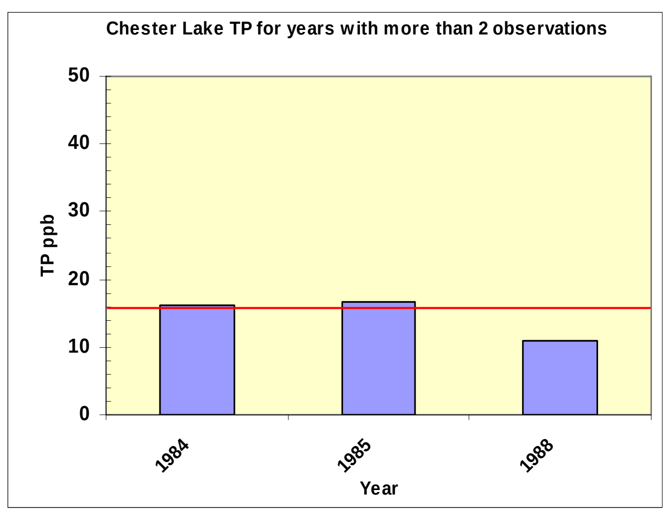
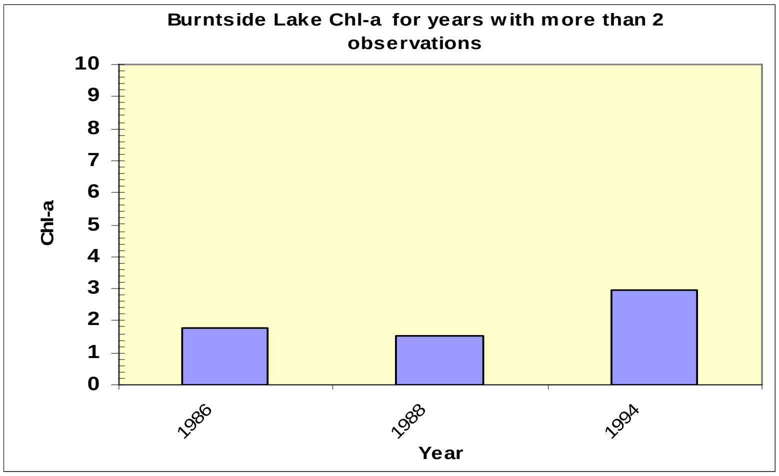
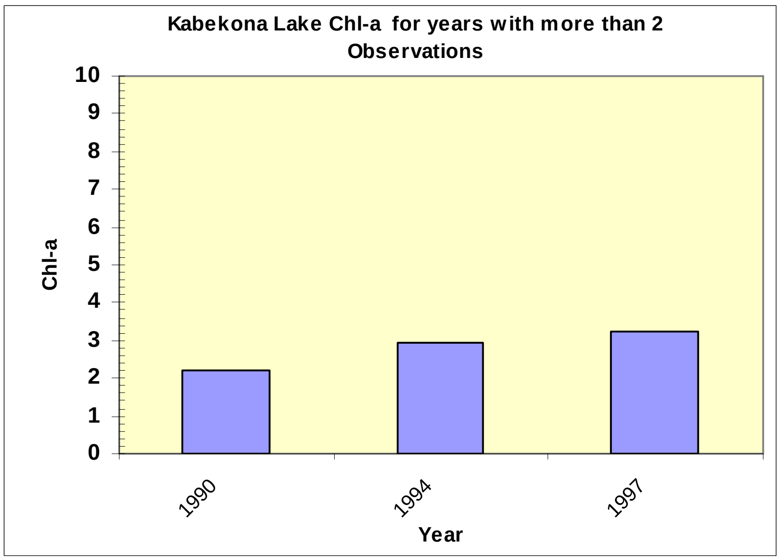


Figure 18. Summer-mean chlorophyll-a for select lake trout lakes.





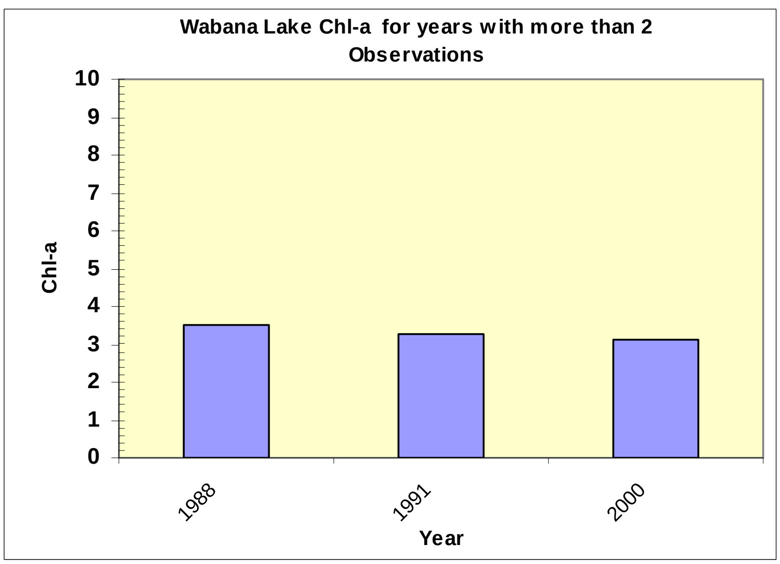


Figure 19. Summer-mean Secchi for select lake trout lakes. Long-term mean noted for each.

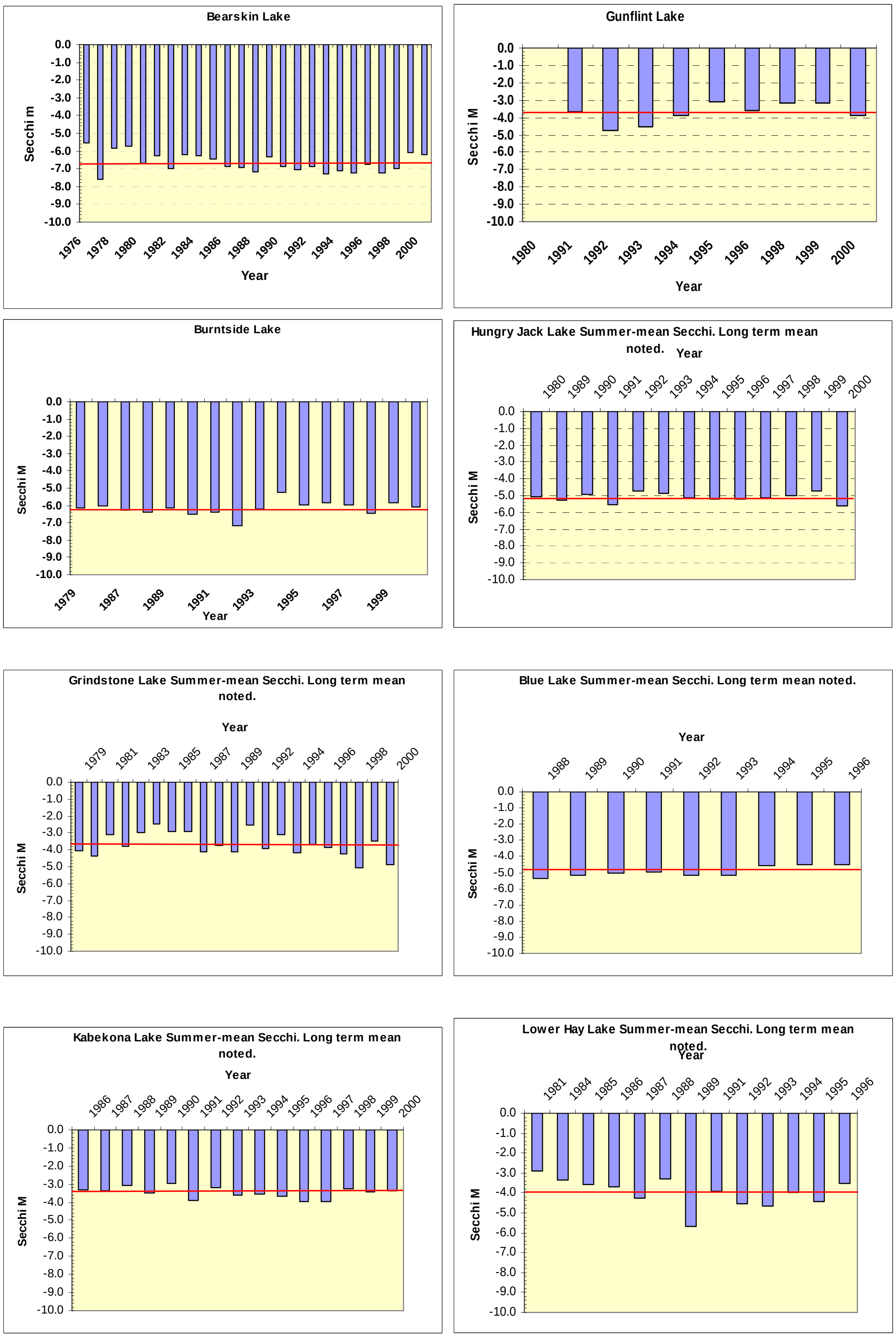
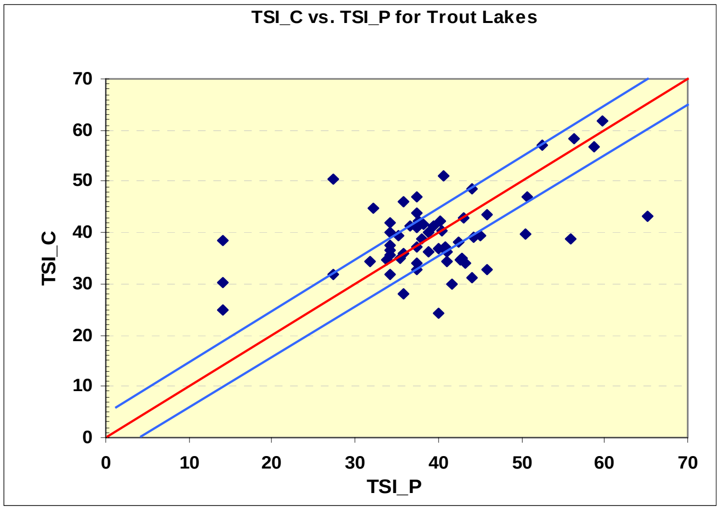
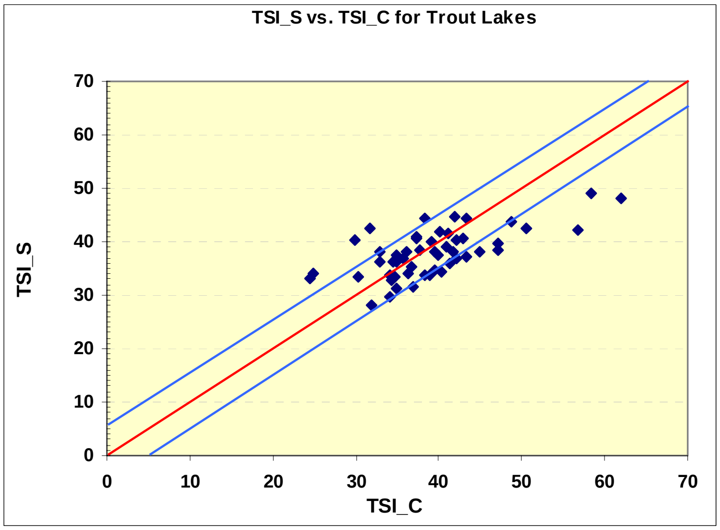


Figure 20. Trophic state index comparison for trout lakes





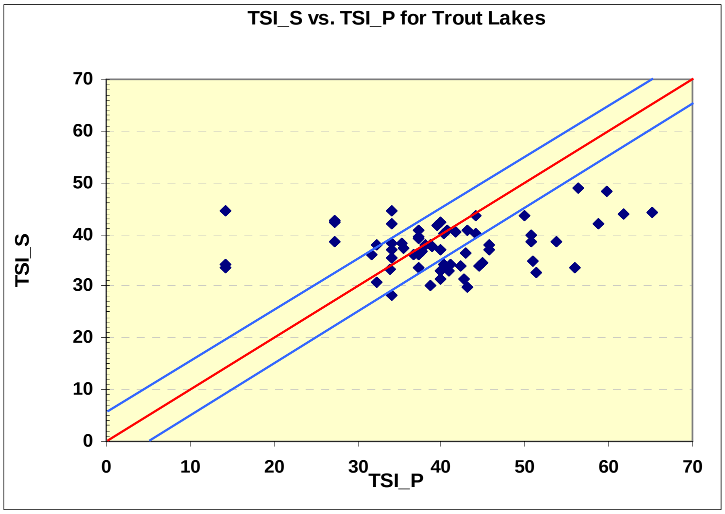
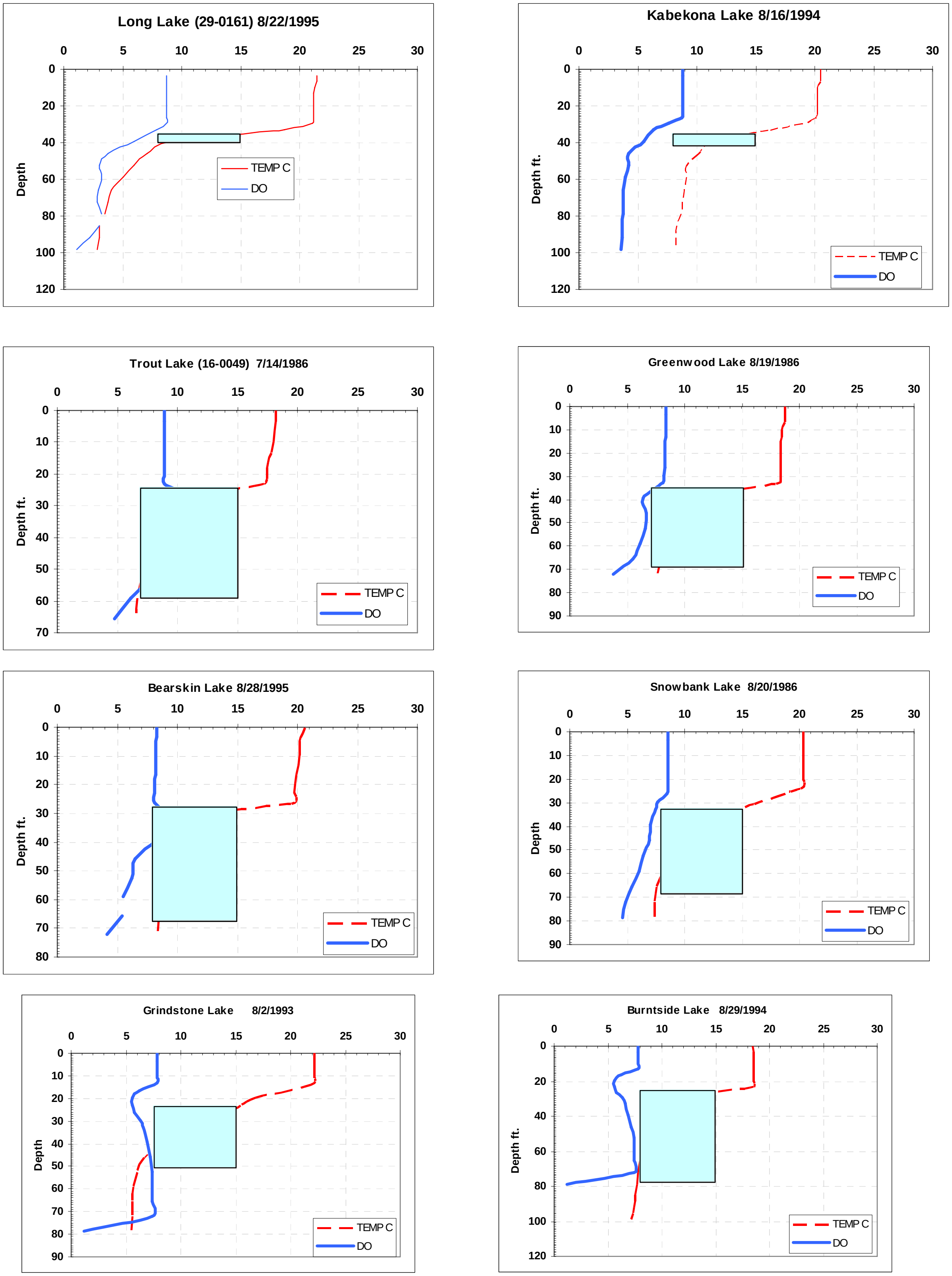
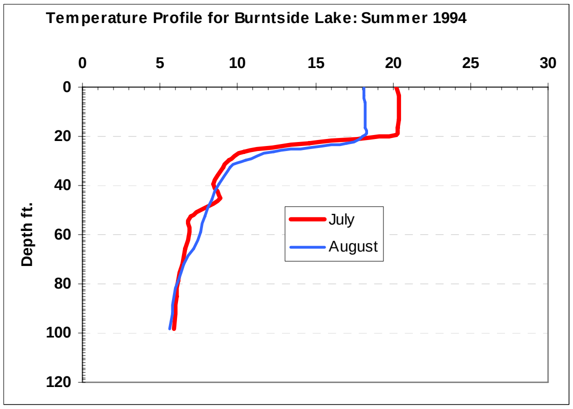
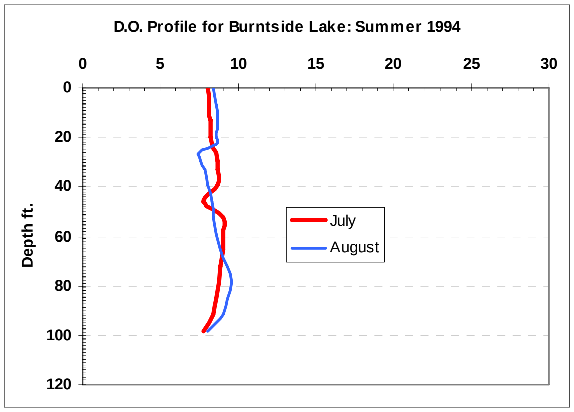


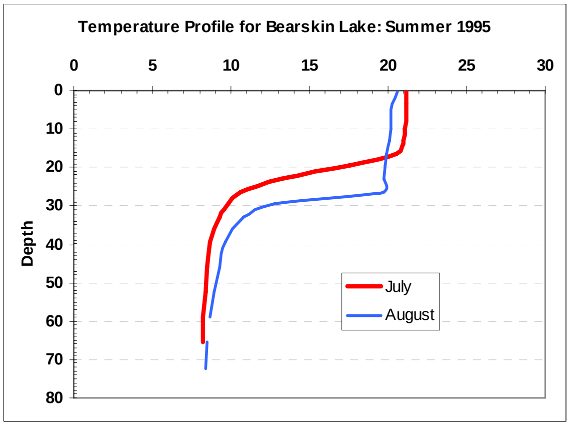
Figure 21a. Late summer DO and temperature profiles. Shaded window corresponds to optimal habitat for lake trout (depth strata with DO > 5 mg/L and temperature between 8-15 C.

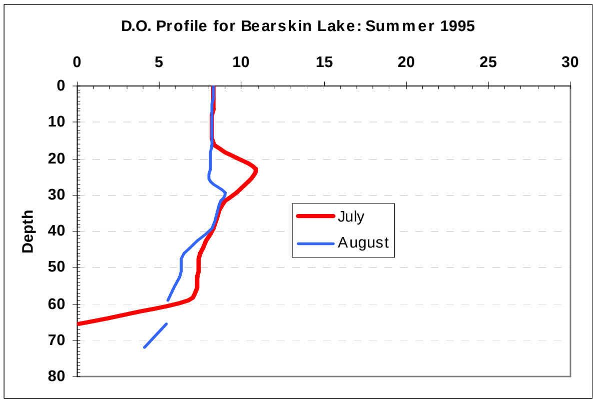


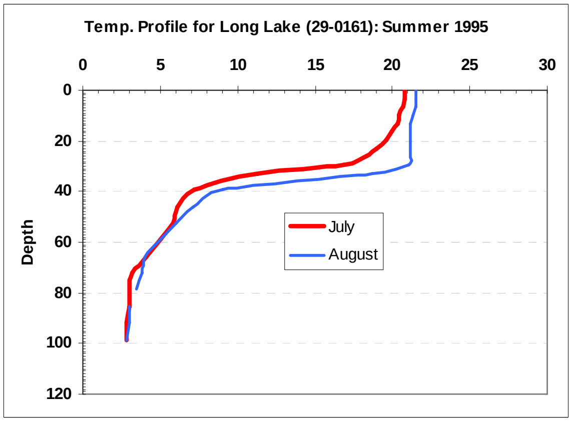
21b. Select temperature and dissolved oxygen profiles for July and August

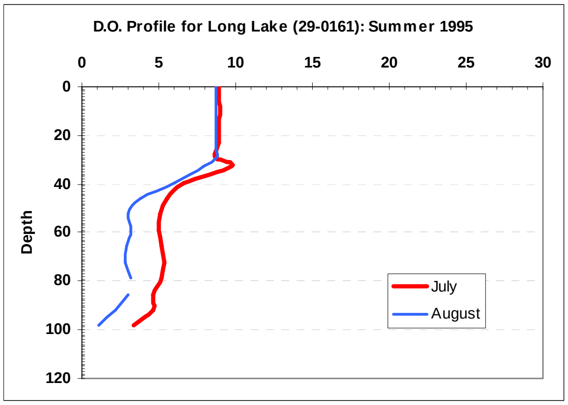












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Case study 5. Changes in P loading, trophic status and hypolimnetic oxygen: Trout Lake (31-0216).

Trout Lake (31-0216) near Coleraine is rather large at 1,890 acres and very deep with maximum depth of 135 feet (41 m). This lake received wastewater discharges from the cities of Coleraine and Bovey from 1910 through early 1987 when it was diverted out of the lake. Lake trout, present in earlier MDNR fishery surveys, had disappeared from the lake in the 1940's, which was consistent with their disappearance from other Itasca County lakes as well during that timeframe. The lake has had numerous reports of high algae or weeds impairing swimming in the 1980s' and summerkill of whitefish and other species were noted in 1969, 1971, and 1973. These water quality problems led to a request for a LAP study of the lake by concerned citizens and the City of Coleraine in 1987. This study was later followed by a CWP study in conjunction with Itasca SWCD in the early 1990s. Monitoring by MPCA, Itasca County, and citizen volunteers has continued through 2002.

In-lake conditions during the 1987 LAP study are summarized as follows: mean total phosphorus – 44 µg/l, mean chlorophyll-a – 14 µg/l, maximum chlorophyll-a – 34 µg/l and mean Secchi 3.6 meters. Respective TSI values are as follows: TSIP = 59, TSIC = 57 and TSIS = 40. These measurements were similar to those found in MPCA studies in the late 1970's (Fig. 22). Transparency was high (low TSI value) relative to the chlorophyll-s and total phosphorus concentrations because of the large colonial blue-green algae that dominate over the summer and allow for deep penetration of light into the water. This condition has been noted in other lakes dominated by blue-green algae (Osgood, 1982). These algae frequently form thick scums on the downwind shore of the lake. In addition to suspended blue-green algae, mats of filamentous algae, which start their life cycle on the bottom of the lake, were common as well in the 1970s and 1980s. By early August 1987 the hypolimnion (top ≅ 14 m) was anoxic (Fig. 23). BATHTUB modeling, done as a part of the LAP study, predicted a "steady state" TP concentration of 17 µg/L and chlorophyll-a of 5.1µg/L following removal of the wastewater discharge. The modeling also predicted a decrease in the oxygen depletion rate over time. It was estimated it would take 9-15 years to achieve "steady state."

Significant declines in TP and chlorophyll-a are evident for Trout Lake based on MPCA and Itasca County data (Fig. 1 below). TP concentrations from 1993-2000 ranged from 15-25 µg/L, while summer-mean chlorophyll-a concentrations were in the 3-5 µg/L range. The decline in chlorophyll-a (algae) has resulted in a significant increase in Secchi transparency as well with measurements in excess of the long-term mean (4.3 m) since about 1993 (Fig. 22). Anecdotal evidence suggest that filamentous algae has declined as well.

Dissolved oxygen concentrations in the hypolimnion have changed over time as well. Historical MDNR data from August 1946 revealed a total phosphorus concentration of 18 µg/l and dissolved oxygen (4.7 mg/l) down to a depth of 115 feet (35 m). However, DO profiles from mid August 1976 indicate anoxic conditions in the thermocline (~9 m) and profiles from August 1987 indicate anoxic conditions in the upper portion of the hypolimnion (top of hypolimnion ~14 m). More recent profiles, since the cessation of the wastewater discharge to the lake indicate improvements in hypolimnetic DO. An August 1993 profile indicated 1-2 mg/L DO down to about 15 m but anoxic conditions from about 18 m on. However an August 1995 profile indicated DO of 2-3 mg/L down to a depth of about 28 m and anoxic conditions below about 32 m (Fig. 23).

These results—declines in TP and chlorophyll-a, increases in Secchi transparency and hypolimnetic DO are encouraging (and somewhat consistent with model predictions). The trophic status measures are within the typical range for lakes in the NLF ecoregion and in the range of some of the earliest available data for the lake, which is a huge improvement over measurements prior to the wastewater diversion. The apparent reduction in the ODR seems encouraging as well as the metalimnion (as of 1995) had adequate oxygen to help sustain coolwater fish. And, while the DO in the upper portion of the hypolimnion was likely too low for fish (Fig. 23) it would allow refuge for vertically migrating zooplankton (Daphnia in particular) that play an important role in cropping algae. However there was not adequate refuge (in terms of DO and temperature) for support of a coldwater fishery.

Figure 22. Trout Lake summer-mean TP, chlorophyll and a and Secchi

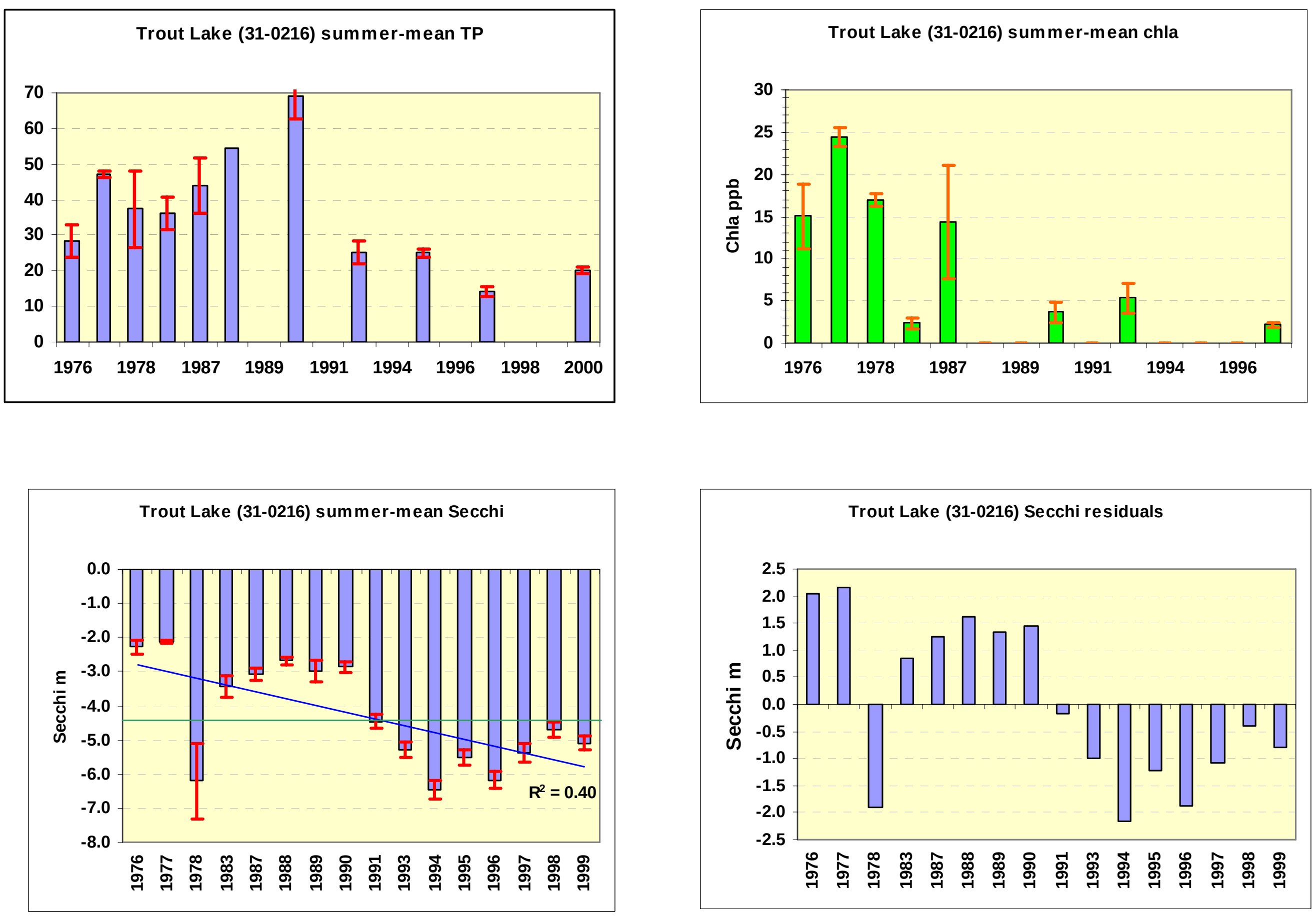
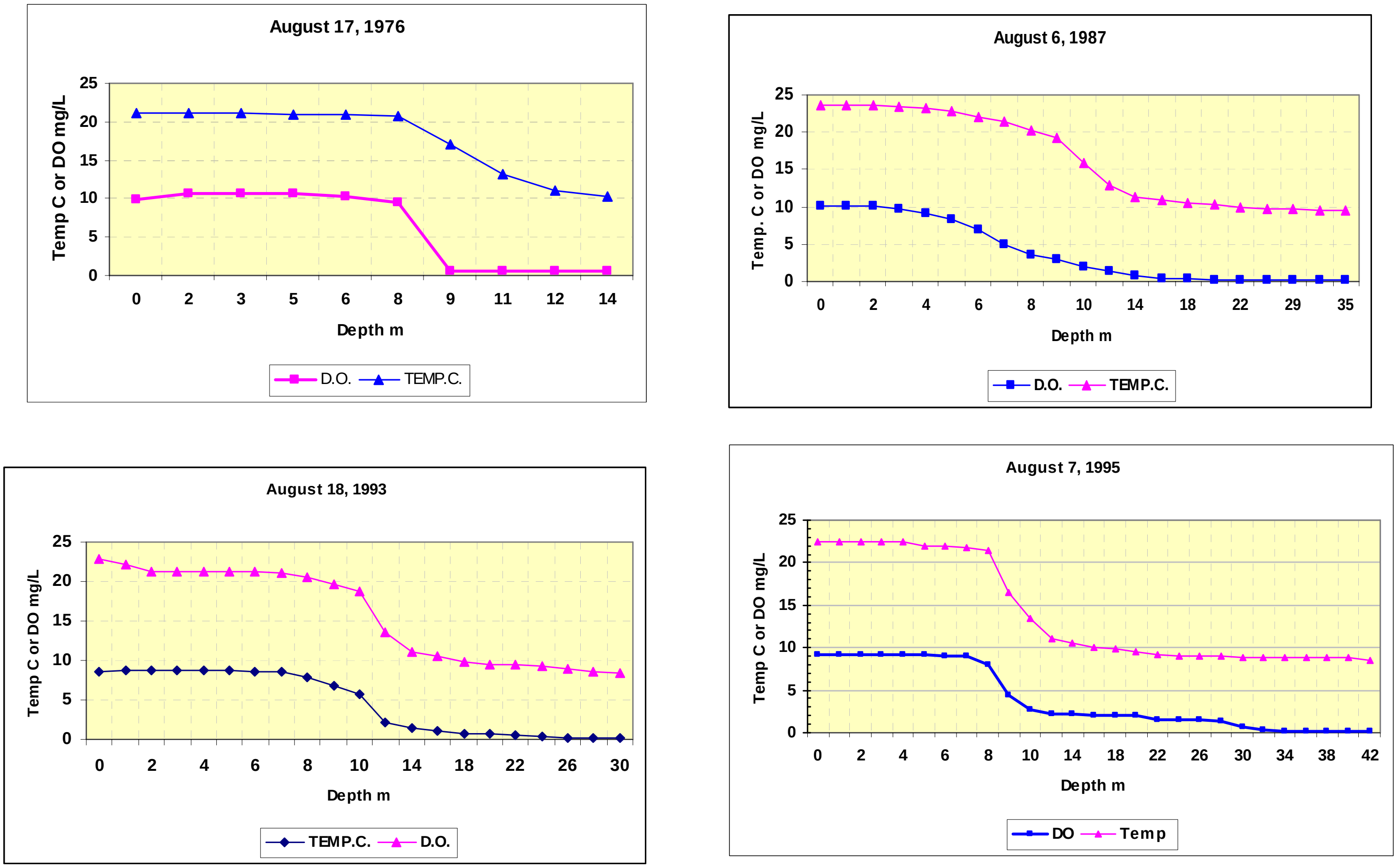


Figure 23. Trout Lake late summer dissolved oxygen and temperature profiles



Stream Trout Lakes

The Minnesota Department of Natural Resources (DNR) manages 5,400 game fish lakes. However, only 194 lakes are managed for stream trout. A stream trout is either a brook trout, brown trout, rainbow trout or a splake that is a cross between a brook trout and a lake trout. Stream trout require specific conditions to survive in lakes; however since these lakes are primarily stocked the concern is only with adult habitat requirements. The water must be cold, well oxygenated, and relatively free of pollutants. There are three types of stream trout lakes that the DNR manages: 1) designated trout lakes, 2) two-story lakes, and 3) abandoned mine pits.

Minnesota has 161 are designated trout lakes. Because trout require cold clean water, the DNR protects these lakes by designating them as trout lakes. These lakes are listed in Minnesota Rules 6264.0050. The average size of a designated trout lake is 39 acres and range in size from 0.5 to 314 acres. Twenty-seven designated trout lakes are located within the Boundary Waters Canoe Area Wilderness.

The DNR manages 16 lakes for warmwater species, such as walleye, bass, or panfish in the shallow water and for stream trout in the deeper portion of the lake. These lakes are referred to "two-story lakes". The average two-story lake is 188 acres and range in size from 10 to 800 acres. The DNR also manages abandoned mine pits for stream trout. When the mining operation is complete, these pits fill with water. The DNR manages 27 mine pits, 16 are designated stream trout lakes and two are managed as two-story lakes. The mine pits average 47 acres and range in size from 0.5 to 314 acres.

Anglers fishing for trout (stream and lake) in inland lakes spend an average of 4 days fishing in the open water season and 2 days during the winter season each year. The average one-way distance traveled to fish inland trout lakes is 160 miles, average travel party size is over three, and average trip length is three days.

Stream trout lakes are also important to Minnesota's economy. The overall annual economic impact of anglers fishing inland trout lakes is $52 million, with another $33 millions in direct income. These expenditures support over 1,000 full and part time jobs (Gartner et al. 2002).

Water temperature is the primary factor determining the distribution of these species. The preferred water temperatures for trout are between 11 and 18 ^{o} C (Coutant, 1977). The most critical time of the year is summer, when surface waters reach their highest annual temperatures. In the spring and fall, when water temperatures are cooler, trout can use the entire lake. As temperatures increase, trout will move out of the warmer shallow waters and are restricted to the deeper colder waters. As water temperatures increase, trout activity decreases, which contributes to reduced growth and increased predation risk.

In addition to coldwater temperatures, trout also require water with DO concentrations greater than 5 ppm. Oxygen levels in lakes generally decrease with increasing depth. As the upper portion of a lake warms during the summer, trout move to deeper water. At oxygen concentrations below 5 mg/L, trout show behavioral changes such as decreased swimming. Temperature and oxygen stratification during the summer limits the availability of suitable habitat and consequently the carrying capacity of a lake.

A summary of trophic status data for several stream trout lakes is presented in Table 7. This summary provides an indication of the range in water quality among lakes that are successfully managed for these species and hence provides a guide to desirable TP, chlorophyll-a and Secchi measurements for this type of lake. There is some overlap with the lakes included in Table 6 as some of these lakes may contain both lake and stream trout. Though the requirements (necessary combination of temperature and dissolved oxygen and overall low algal productivity), for sustaining stream trout are not quite as stringent as lake trout many of the same principals apply as described in the previous section; hence it is necessary to maintain relatively low TP and chlorophyll-a and high transparency in order to provide adequate refuge for the stream trout.

Table 7. Trophic status summary for stream trout lakes from assessed data (MPCA, 2004)

Parameter	# of obs (lakes)	Mini-mum	25th %tile	Median	75th %tile	Maxi-mum
TP (µg/L)	103	5	10	15	21	67
Chlorophyll-a (µg/L)	51	0.7	1.9	3.2	6.3	45
Secchi (m)	125	1.1	3.3	4.4	5.4	10.9

Water Supply Lakes

Eutrophication of drinking water supply lakes, reservoirs and ground waters can have direct and serious consequences. Eutrophied water bodies have increased production of algae and macrophytes, hypolimnetic or metalimnetic oxygen depletions, increased pH, a shift to blue-green algal dominance, and increases in particulate organic substances. All of these responses may lead to increases in dissolved organic compounds that: (a) have chelating properties; (b) impart taste and odors; (c) increase color; (d) increase potential organohalide compounds; (e) provide substances for bacterial growth in lakes, treatment plants and distribution systems; and (f) may contribute to corrosion problems. Eutrophication has also been identified as a contributor to human health problems.

In addition to taste and odor episodes, highly eutrophic lakes used as domestic water supplies may also produce algal toxics or organic compounds which may form carcinogenic compounds during water treatment (Lorenz and DiNatale, 1987). The bloom forming cyanobacteria (the blue-green algae) produce secondary chemicals that can have acute and chronic effects. The two classes of cyanobacterial toxins known to be produced include the alkaloid and peptide toxins. The alkaloid toxins (such as anatoxins and aphanotoxins) have been responsible for numerous dog/domestic animal poisonings when the animals drink/ingest sufficient quantities of the algae or infested waters. These toxins may not generally be a threat except at acute lethal doses.

The peptide hepatotoxins may cause concern at lower chronically toxic levels (Carmichael, 1986). However, no water quality standards exist for these toxins. Normal flocculation and chlorination processes do not remove the toxicity (Falconer, et al., 1989).

The use of eutrophied raw waters for domestic purposes generally requires substantial increased dosages of chlorine during treatments including: (1) raw water transportation; (2) during water treatments to control breakdown of ammonia and other substances; (3) during final disinfection; and (4) increased usage to maintain residual chlorine in the water distribution system (due to increased consumption by the organic compounds) Dorin (1980). The organic substances react readily with chlorine, forming organochlorinated compounds that are very persistent and cannot be efficiently removed. Trihalomethanes (THM's) are volatile compounds that may be produced which are a human health risk because they exhibit or are suspected of having carcinogenic and/or mutagen properties. U.S. EPA issued a standard for total THM's of 0.100 mg/l in finished water in the 1980s: however recent interim rules published in 1998 lowered this maximum to 0.080 mg/L (Cooke and Kennedy, 2001). However, THM's likely represent a small (e.g., 20 percent) proportion of all organochlorines in drinking water (Dorin, 1980). THM's were found to be correlated with total organic carbon and lake trophic state by Arruda and

Fromm (1988). These authors found that in lakes with total organic carbon concentrations greater than 4-5 mg/l and Carlson TSI values greater than 45 (summer-mean TP of about 17 µg/l), the THM standard would probably be exceeded if water treatment plant processes were not modified to minimize THM formation. Palmstrom, et al. (1988) also found that algae, as well as macrophytes were important sources of organic THM precursors in a domestic water supply lake in northern Ohio. More recently, Jack et al. (2002) demonstrated with both field and lab data that algal production, algal senescence, and photolysis may all play a role in the internal generation of THM precursors in large rivers.
Lorenz and DiNatale (1987) concluded that additional eutrophication of Standley Lake a water supply reservoir for the cities of Westminster, Thornton and Northglenn, Colorado would generate algal biomass that could cause violation of the drinking water standard for THM's. A phosphorus standard of 25 µg/l of total phosphorus was proposed in order to limit THM precursor production in the lake. Interestingly this is the same concentration that is used as a management goal for Vadnais Lake by the St. Paul Regional Water Utility (SPRWU).

Taste and odor problems occurring from lakes being used as domestic water supplies has also been traced to algae and actinomycete bacteria. In general, eutrophic lakes have more algae and actinomycetes than mesotrophic or oligotrophic lakes (Nordin, 1985). Taylor, et al. (1981) listed algal species responsible for taste and odor episodes.

Blue-green algal blooms in the supply lakes were the primary sources of taste and odor episodes experienced by the SPRWU (Walker, 1985a) and the city of Fairmont (Stefan and Hanson, 1981). Walker attributed the algal bloom potential to nutrient loadings entering the Vadnais Chain of Lakes from local watersheds and to the internal cycling of nutrients within the lakes. Expensive external and internal loading reduction measures have been instituted by the SPRWU in order to reduce in-lake nutrient and algal concentrations and hence the frequency and severity of taste and odor episodes. Historically, both St. Paul and Fairmont heavily utilized copper sulfate treatments to reduce algal blooms. In 1986, for example, the SPRWU treated Vadnais Lake with 29,500 pounds of copper sulfate. More recently though extensive watershed projects to reduce P loading to the lake, combined with iron chloride (FeCl) dosing of Mississippi River inflow water, hypolimnetic aeration and FeCl addition in the hypolimnion. This has resulted in much lower in-lake P concentrations, which averaged 29 µg/L for the period 1992 – 1999 as compared to concentrations on the order of 50 µg/L in 1984 – 1986 (Walker 2000). This has yielded a 13 percent decline in summer-mean chlorophyll-a over this same two periods: 12 µg/L vs. 14 µg/L, respectively. In their work they use chlorophyll-a concentrations exceeding 20 – 30 µg/L as an operational definition of nuisance and sever nuisance algal blooms that can be important to taste-and-odor precursors (Walker 2000). The frequency of nuisance blooms decreased from 19 percent in 1984-1988 to eight percent in 1995-1999. The frequency of severe nuisance blooms decreased from seven percent to two percent.

Blue-green algal toxicity research has indicated that application of herbicides to blooms near water intakes may not be appropriate due to lysis of algal cells which may increase the release of toxins (Berg, et al., 1987, Carmichael, personal comm.). Hanson and Stefan (1984) have also indicated that long term use for copper sulfate as an algaecide can lead to changes in fish population, diversity of benthic organisms and accumulation of copper in sediments. Prevention of the blooms is again the preferred alternative.

Communities using lakes for domestic water supply in Minnesota are summarized in Table 9, along with indications of average water quality measurements and blue-green problem, if known. Other lakes classified for use as domestic water supplies in Minnesota may be found in Minnesota Rules Chapter 7050.0460. The two largest communities utilizing lakes for all or a portion of their public water supplies are St. Paul and Fairmont, both of which have blue-green algal induced taste and odor episodes. In-lake TP was historically in the 50 µg P/l in Vadnais Lake during the summer but was in the 25-30 µg/L range in recent years as a result of watershed and in-lake efforts to reduce external P loading and internal P recycling. Water quality has historically fluctuated widely in Budd Lake (Fairmont Chain) with values usually greater than 100 µg/l. However a recent 2002 study by Martin County (Schlorf Von Holdt and MPCA, 2003) recorded summer-mean TP of 77 µg/L which, while lower than some previous years, was still high enough to allow for high chlorophyll-a (mean = 42 and maximum = 75 µg/L) and blue-green algae blooms. Budd Lake is generally well-mixed (Fig 8b), however a weak thermocline may periodically form at a depth of 4-5 meters and the "hypolimnetic" DO falls below 2 mg/L. Low hypolimnetic DO promotes internal P recycling as evidenced by elevated TP measures on those dates. As the lake becomes well-mixed again hypolimnetic values decline and epilimnetic values increase (Fig. 10).

In contrast to these two major drinking water sources available water quality data for mine pit lakes being used for domestic supplies in northern Minnesota indicate that in-lake TP values were generally less than 5-10 µg/l and very low chlorophyll-a concentrations. Secchi is often high in these lakes, with exception of highly colored lakes like Colby where bog staining from incompletely dissolved organic material limits light transparency.

Lakes that are used as domestic water supplies are valuable community resources and deserve special consideration. This priority should be recognized at both the state and local level, e.g., watershed management organization. Such consideration should include minimizing external nutrient loadings to encourage minimal algal growth, which in turn will reduce water supply treatment expenses. This will reduce taste and odor episodes and also minimize THM and THM precursor production and potential for blue-green toxin production. The supply of nutrients should also be minimized such that the hypolimnetic layers of the supply lakes are protected from low oxygen concentrations caused by excessive algae. Low hypolimnetic oxygen concentrations may also cause elevated concentrations of hydrogen sulfide, soluble iron and manganese compounds and inorganic carbon.

While specific ecoregion-based criteria will not be proposed for drinking water lakes this analysis and discussion should be useful as lake-specific goals or criteria are developed. These goals/criteria could ultimately be developed as a part of source water protection efforts. These goals will likely consider the current water quality of the lake, ecoregion-based potentials, ability to address watershed sources of nutrients and ability/costs associated with treating the water for potable use. For example where water quality is quite high and extensive treatment of the water is not required to make it ready for potable use, e.g. NLF lakes, it is desirable to set criteria at very low levels in order to minimize treatment expense and potential water quality problems. In other ecoregions of the state, maintaining or restoring lake TP concentrations as low as possible is recommended to both minimize treatment expense but also minimize the occurrence of taste and odor, production of THMs and other problems associated with excess algae in source water. For example, a value of 25 µg/l was recommended as a realistic management goal for Vadnais Lake (Walker, 1987) in order to significantly reduce the frequency of algal blooms and resulting taste and odor episodes. SPWU monitoring to date suggests that the Lake is very near that goal. For the highly eutrophic Fairmont Chain of Lakes an appropriate goal may be 70 µg/L or lower. If TP can be reduced further from 70 µg/L to 40 µg/L substantial reductions in the frequency and severity of nuisance algal blooms could be realized (Fig. 6a & b).

Table 8. Tabulation of Communities Using Lakes as Domestic Water Supplies (Excludes Lake Superior but includes mine pit lakes). Compiled from MDH (1983), Pierce and Tomcko (1989) and MPCA data (through 2002 when available).

						Available Lake Data		
Town County/ Region	**Population (1983)**	**Taste & Odor[1] Treatment (1983)**	**Lake Name**	**Blue-Green Blooms**	**TP µg/l**	**Secchi (m)**	**chl-a (µg/l)**	**Hypo. DO, color, notes**
1 Aurora St. Louis/ NLF	2,225	/F1	St. James Pit		8		1.8	
2 Chisholm St. Louis/ NLF	5,100	/F1	Fraser Pit (69-1300)		1	6.7	1.5	DO @ 82m = 6.4
3 Ely St. Louis/ NLF	3,800	Tc/Fl	Burntside (69-0118)		10	6.1	3.0	DO @ 37m < 2 LAP study
4 Eveleth St. Louis/ NLF	5,000	/Fl	St. Mary's (69-0651)		10	1.0	4.6	Color = 5 PCU
5 Hibbing St. Louis/ NLF	21,193		Well in abandoned mineshaft, adjacent to mine pit					
6 Hoyt Lakes St. Louis/ NLF	2,800	/Fl	Colby (69-0249)		19	1.5	4.0	DO <1 @ 5 m Color = 144 PCU
7 McKinley St. Louis/ NLF	250		Corsica Pit					
8 Virginia St. Louis/ NLF	14,500	/Fa	Missabe Mt. Pit (69-1292)		6	3.5	1.0	Color = 5 PCU
9 Winton St. Louis/ NLF	290	/Fl	Fall Lake (38-0811)		17	2.2	4.0	
10 Biwabik St. Louis/ NLF	1,432		Canton Pit (69-1294)		3	< 1.0	6.4	Color=5 PCU
11 St. Paul Ramsey/ CHF	385,000	TK/Fl	Vadnais	periodic	32	3.4	16	DO <1@ 7 m
12 Fergus Falls Otter Tail/ CHF	12,443		Wright Lake on Ottertail River					
13 Stephen Marshall/ RRV	958		Dam at Tamarack River					
14 Fairmont Martin/ WCP	11,506	Tc/Fl	Budd	common	77	1.0	42	2002 study

[1]TC = activated carbon, Td = chlorine dioxide, Tk = potassium permanganate, Ts = sulfur dioxide, Fl = gravity sand filtration, Fa = pressure sand, To = other (MDH, 1983).

Shallow Lakes

Based on the ecoregion reference lakes, assessment of statewide data sets for 305(b) development of guidance for listing of nutrient-impaired lakes on Minnesota's 303(d) list, and preceding analysis in this report it is apparent that there are some distinct differences in trophic status and potentials of shallow, well-mixed lakes as compared to deep, stratified lakes. Table 4, for example, demonstrates distinct differences in TP among dimictic, intermittently mixing and well-mixed lakes. These differences are in part natural – due to physical (morphometric) characteristics of the lakes and are also a reflection of cultural influences as will be noted later. For example, shallow lakes are predominately littoral (zone of the lake that is 15 feet or less) and are often dominated by rooted submergent vegetation. Shallow lakes are generally well-mixed, which may allow for resuspension of sediments and nutrients to a greater degree than would be the case for deep lakes. Deep lakes, in contrast, typically have a smaller percentage of the lake classified as littoral, tend to stratify during the summer and are much less subject to wind mixing and sediment resuspension. However, some differences (between shallow and deep lakes in Table 4) are a reflection of cultural influences as will be noted later.

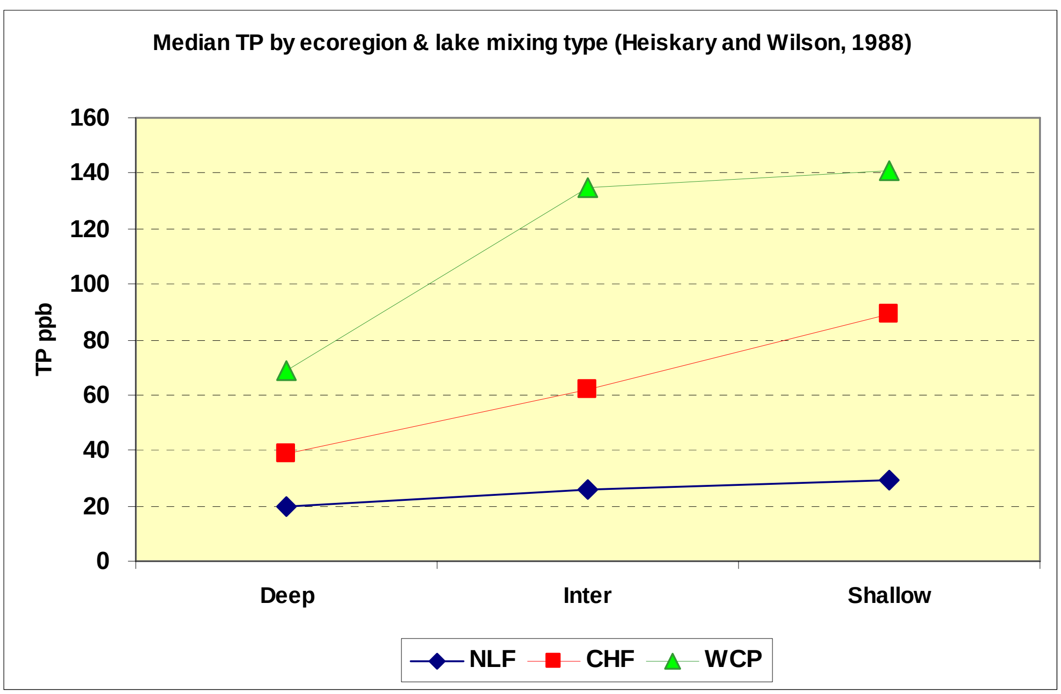
From this work it was evident that differences were particularly marked among deep and shallow lakes of the CHF and WCP ecoregions (Fig. 24). Also during public comment periods and hearings, associated with the establishment of guidance for the listing of nutrient-impaired waters, concerns were expressed by several respondents that swimming may not be the primary use in many of Minnesota's shallow lakes. Among their contentions were: the shallowness of the lakes, highly organic substrates, and often times over-abundance of rooted submergent and emergent plants. And because of these factors, it was recommended that the MPCA consider separate nutrient criteria for shallow lakes that would take these factors into account. This would hopefully result in criteria that were more closely attuned to the actual uses of these shallow lakes, which is commonly boating, fishing, aesthetics, wading and waterfowl production; rather than an emphasis on swimming (primary body contact).

Work on shallow lakes was first initiated in 2002, with the sampling of several lakes in the WCP and NGP ecoregions. The purpose of that effort was to assess trends and condition in several of the reference lakes in these regions, supply baseline data for select lakes that lacked basic water quality data and improve our understanding of the water quality and ecology (e.g., rooted plants, fish, etc.) of these shallow lakes (Heiskary et al. 2003). Since the southwest lakes were quite nutrient-rich and had depauperate rooted plant populations, further work was proposed for shallow lakes of west-central Minnesota where there was a wider range in lake trophic state and a more diverse population of rooted plants.

The 2003 study of shallow west-central Minnesota lakes was based on the concept of "alternative states" for shallow lakes as described by Moss et al. 1996; Moss, 1998 and numerous others; whereby shallow lakes may switch from relatively clear plant-dominated systems at low nutrient concentrations to cloudy, algal-dominated systems at high nutrient concentrations. While exact thresholds are not frequently noted some studies suggest inverse relationships among macrophyte coverage and phytoplankton chlorophyll-a in shallow lakes. For example Portielje and Van der Molen (1999) note that when macrophyte coverage exceeds 5% in shallow lakes there is a significant drop in chlorophyll-a. Scheffer et al. (2001) note that shallow lakes often have certain critical levels whereby the system shifts catastrophically between two stable states. In other words lake response to excess nutrient loading is not linear; rather the lake may be stable until a threshold is reached and then an alternate stable state is reached. In order to switch back one must take it back much further than it was previously to some "critical" level. Things that contribute to a loss of resilience in the system increase the likelihood that a drastic switch to an alternative state will occur.

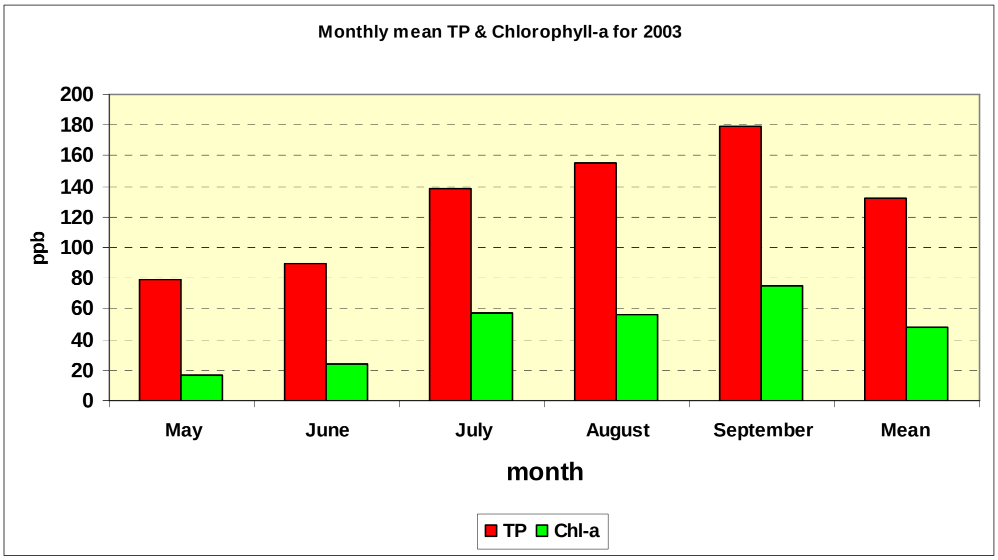
While excessive rooted macrophytes in a lake may impede recreation – macrophytes are essential to the overall ecology and stability of lakes – shallow lakes in particular. Madsen, (2001) notes that macrophytes help to slow water velocities increasing sedimentation, decreasing turbidity, thus increasing light penetration and growth of macrophytes. Sediment accumulates in areas where macrophyte beds are located. The loss of macrophytes increases the likelihood of sediment resuspension. He notes Marsh Lake (west-central Minnesota) as a good example of this; whereby, when macrophytes were absent resuspension of P occurred >30% of the time in summer. Scheffer (1998) also describes the feedback effect of macrophytes on turbidity whereby as plant biomasss increases sediment resuspension declines resulting in lower turbidity causing hysterisis (dramatic shifts) in plant mass – water turbidity relationships. Moss notes further that macrophytes store nutrients (luxury uptake) which prevents algal growth, provide zooplankton refuge from predators which increases algal grazing, may also produce allelopathic compounds which inhibit algal growth, and stabilize the sediment which lessens resuspension of nutrients. Radomski and Goeman (2001) describe multiple benefits of macrophytes to the overall ecology of the lake with a particular emphasis on fish spawning and habitat and demonstrated that shoreline areas with diverse submergent vegetation exhibited significantly more fish than shorelines that had little or no vegetation.

Figure 24. Median TP by mixing class and ecoregion . Based on 1988 MPCA assessed lake data.



The 2003 study of shallow lakes in west-central Minnesota served to reaffirm basic observations on seasonal patterns for TP and chlorophyll-a for shallow lakes (e.g. Fig. 19); as well as providing some new insights on interrelationships among TSI variables and rooted plants. Comparisons of water quality and rooted plant metrics (e.g. Floristic Quality Index (FQI) and number of submergent and floating-leaf plants) provided an opportunity to identify potential thresholds for establishing nutrient criteria for shallow lakes. Examples, as drawn from Heiskary and Lindon (2005), of these interrelationships are offered in Figure 25. This analysis provided a primary basis for recommending criteria for shallow lakes in the CHF ecoregion, which will be addressed in more detail later in this report.

Figure 25. Monthly mean TP & chlorophyll-a from 27 shallow west-central MN lakes (Heiskary & Lindon, 2005)

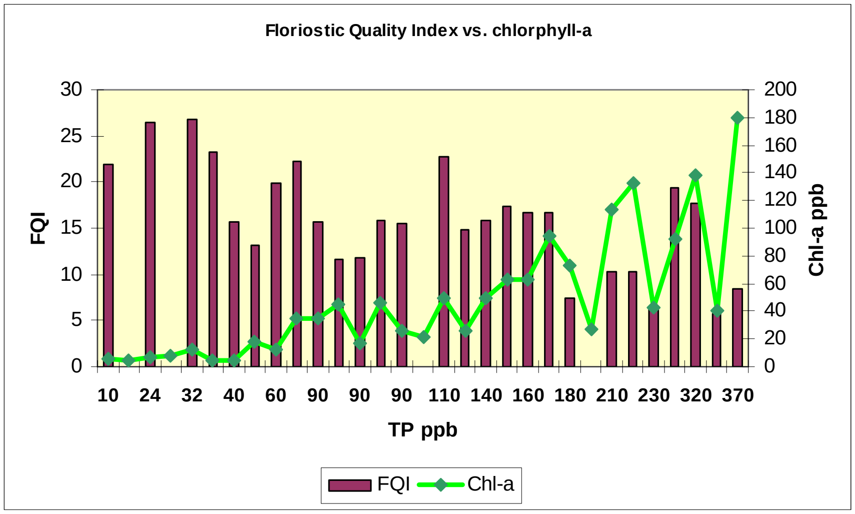


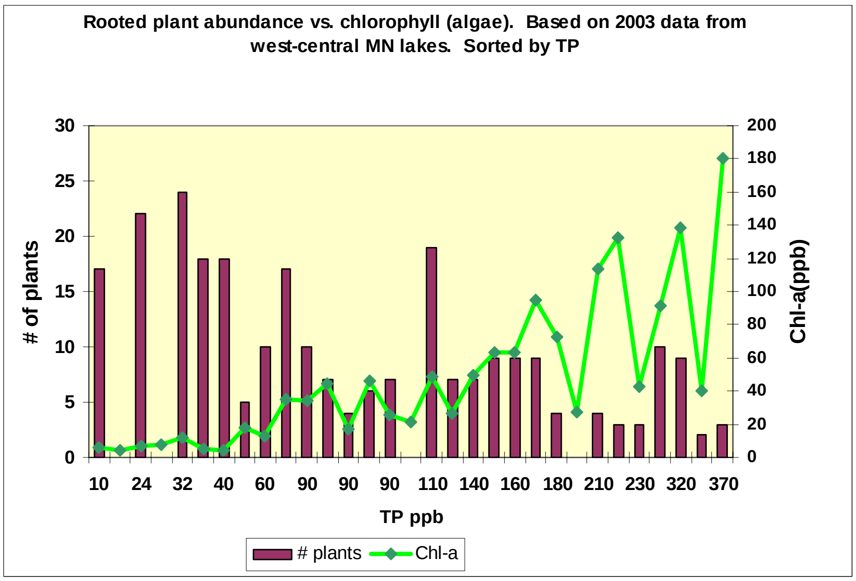
User Expectations

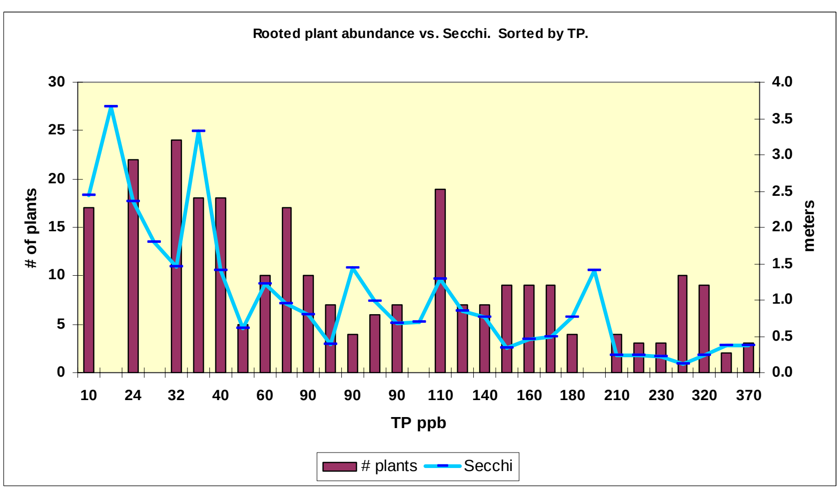
a. User expectations and economic considerations

Definitions of “acceptable” or “objectionable” lake water quality vary regionally (Heiskary and Walker, 1988). These variations may reflect observer or user acclimation to a particular range of conditions, (Lillie and Mason, 1983). A lake user in a region dominated by oligotrophic lakes would probably have much higher expectations, e.g., higher transparency and lower algal levels, as compared with a lake user in a region dominated by hypereutrophic lakes. These perceptions or expectations may influence a person’s decision regarding the selection of a lake for recreation or purchasing property. The perception of poor quality relative to other lakes in a region can lead to lost real estate sales (Realtor, personal communication), depressed real estate prices relative to lakes or locales with perceptibly better water quality (Young, 1984), a slower rate of appreciation in property values (David, 1968) and declines in tourism (Larson, 1980).

Figure 26. Comparisons among rooted plant metrics and TP, chlorophyll-a and Secchi. Based on 27 shallow west-central MN lakes (Heiskary and Lindon, 2005).







Studies in Minnesota and elsewhere (e.g. Maine) establish linkage between real estate values and water quality. Michael et al. (1996) examined the relationship between Secchi disk transparency and selling price of >900 properties on 34 lakes in Maine in during 1990-1994. They found that a decrease in Secchi disk transparency of one meter in 10 years was associated with significant declines in property values from $3,000-$9,000 per lot. Steinnes (1992), in a study conducted in the late 1980s for the Minnesota Department of Natural Resources, estimated the contribution of water clarity to lakefront property values in northern Minnesota. In that study significant correlation was demonstrated among water transparency and measures of lake lot price and he found that a one meter increase in Secchi disk transparency raised lakeshore prices by an average of $235 per lakeshore lot. Other factors tested including lake size, lake depth and accessibility did not prove to be significant. In a more recent study in Minnesota, Thompson et al. (2003) examined real estate values on several north central Minnesota lakes and demonstrated linkages between Secchi transparency and property values. Various other Minnesota-specific studies describe willingness to pay to protect or improve water quality (e.g., Lake Bemidji; Henry et al. 1988), value of lakes and good water quality to local economies in the Aitkin area (Big Sandy Lake; Dziuk, 1992), and the Itasca County/Grand Rapids area (Dziuk and Heiskary, 2003). Wilson and Carpenter (1999) provide an overview of several studies that help describe the economic valuation of freshwater ecosystems in the United States.

b. Lake water quality expectations and related considerations

Insights into lake user expectations in Minnesota regions can be derived from some previously assembled information. Cohen and Karasov (1986) in a survey of Minnesota shoreland residents found that water quality issues, in particular the presence or persistence of algal blooms or weeds, were a major concern of residents in the northern and central regions of the state. This roughly corresponds to the NLF and CHF ecoregions.

A review of the MDNR's summary of requests for aquatic nuisance control in 1986 (MDNR, 1987) suggested regional patterns in user expectations for lake quality. The majority of the permits for control of planktonic algae were issued for lakes in the CHF (46 lakes or 7,334 acres treated). In contrast, only five lakes (1,864 acres) were treated in the WCP and one lake in the NLF. Permits for treatment of filamentous algae exhibited a similar pattern with 137 lakes (1,315 acres) treated in the CHF, 2 lakes (25 acres) in WCP, and 25 lakes (14 acres) in the NLF. These patterns exist even if the seven county metropolitan area lakes (i.e., population center for this region) were omitted. Chemical control of algae is much less common in recent years based on a recent summary of permitted control work (Enger and Hanson, 2003).

Another significant concern of lake users is related to excess macrophyte growth. Citizen complaints and inquiries regarding macrophyte problems and permits for treatment are routinely dealt with through MDNR's Aquatic Plant Management Program. In a previous report on this program, MDNR (1988) noted a dramatic increase in the number of permits issued for chemical and mechanical control of macrophytes since the late 1970's. Prior to this time, macrophyte control (in terms of acres harvested or treated) was rather stable from the mid 1950's to the mid 1970's, with occasional annual peaks. The increase in permits issued for vegetation control has increased throughout the 1990s, with the most marked increases occurring in MDNR Regions: 1 (NW MN, ~CHF-NLF ecoregion transition), 3 (central MN counties and Brainerd lakes area), and 6 (seven county Metro area) (Enger and Hanson, 2003). And of all 6 regions, the most permit requests come from Region 3. Region 2 (northeast MN – Arrowhead), a region with a very high density of lakes (and lakeshore development), has relatively few permits. Likewise very few permits are issued in Region 4 (southwestern MN) and Region V (southeastern MN) which have a much lower density of lakes and much less lakeshore development. Possible explanations for the increase in number of permits include increased nutrient input to lakes, increase in the amount of marginal lakeshore being developed and increased public awareness of permitting program (MDNR, 1988).

In 1987, Minnesota Pollution Control Agency (MPCA) staff received 338 complaints and 412 technical assistance requests from citizens, lake associations, watershed districts, and other lake management groups (Munson, 1988). Analysis of 110 complaints showed the most commonly perceived problems were: (1) excessive algae and plants (42%); (2) lake water quality problems detrimental to human health (22%); and recently, degraded lake water quality (12%). A significant number of citizens responding to questions about recreation suitability stated that their lake was either greatly impaired for swimming and aesthetic enjoyment. Citizens attributed their lake problems to nonpoint sources of pollution in 88% of the complaints and to point sources in about 12%. The two most commonly reported nonpoint sources affecting lake water quality were septic tanks and feedlots (Munson, 1988).

Tabulation of citizen complaints by ecoregion showed that 74% of all complaints were from the CHF ecoregion that contains about 32% of Minnesota's lakes. The NLF ecoregion accounted for only 17% of all complaints and these frequently arose from lakes that had been or are impacted by wastewater treatment plant effluent. Very few complaints arose from the other two ecoregions. The patterns observed in 1987 were very similar to those observed in 1986, when 82% of all complaints arose from the CHF ecoregion (Munson, 1987).

c. Lake user surveys

Defining the relationship between user expectations and lake water quality measurements is an important part of the criteria setting process. Previous investigators realized the need to relate subjective measures of water quality to user perceptions (e.g., David, 1971; Nicholson and Mace, 1975). A methodology for deriving this information is presented in greater detail in Heiskary and Walker (1988). Basically, the methodology involves the use of an observer survey (Table 9) and water quality data from the ecoregion data set and Citizen Lake-Monitoring Program. Cross-tabulating the water quality measurements against observer survey categories provides a basis for calibrating nuisance criteria on a statewide and regional basis. However, this methodology does not take into consideration nuisance conditions related to excess macrophytes.

Table 9. Lake Observer Survey (Garrison and Smeltzer, 1987)

A. Please circle the one number that best describes the physical condition of the lake water today:

1. Crystal clear water.
2. Not quite crystal clear, a little algae present/visible.
3. Definite algal green, yellow, or brown color apparent.
4. High algal levels with limited clarity and/or mild odor apparent.
5. Severely high algae levels with one or more of the following: massive floating scums on lake of washed up on shore, strong foul odor, or fish kill.

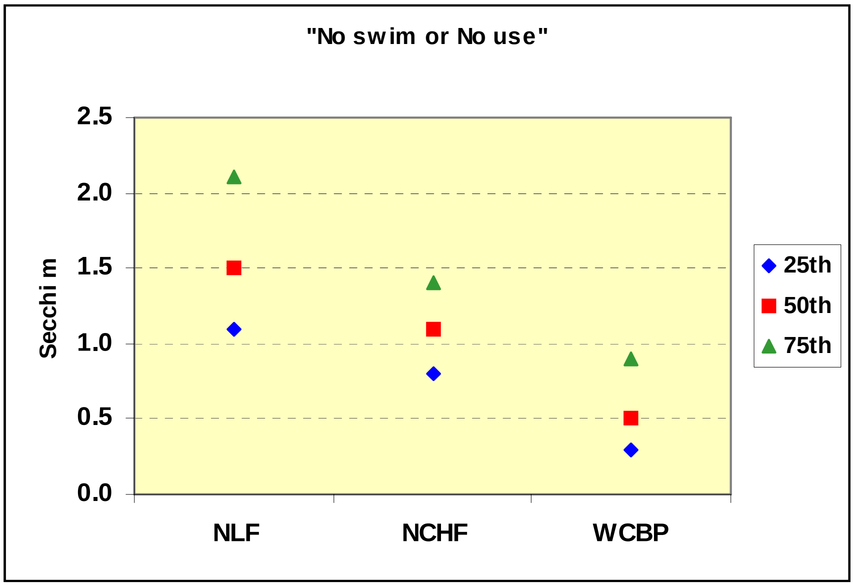
B. Please circle the one number that best describes your opinion on how suitable the lake water is for recreation and aesthetic enjoyment today:

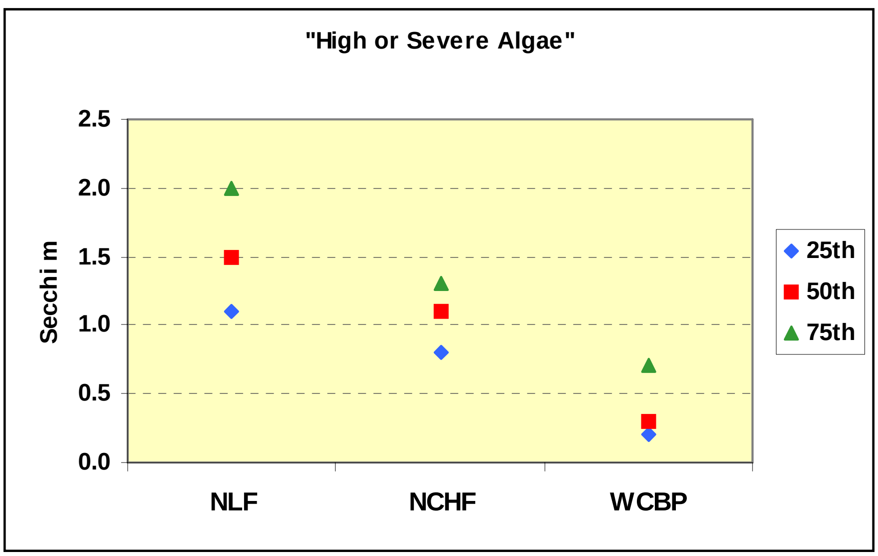
1. Beautiful, could not be any nicer.
2. Very minor aesthetic problems; excellent for swimming, boating, enjoyment.
3. Swimming and aesthetic enjoyment slightly impaired because of algae levels.
4. Desire to swim and level of enjoyment of the lake substantially reduced because of algae levels (would not swim, but boating is okay).
5. Swimming and aesthetic enjoyment of the lake nearly impossible because of algae levels.

In our earlier work (Heiskary and Wilson, 1988) we made use of user perception data derived from MPCA staff monitoring of the ecoregion lakes in early summer 1987. This included 137 samples taken from 40 lakes. Since that time user perception surveys have become a routine aspect of data collected by citizen volunteers in Minnesota's CLMP and in other states as well, e.g., Vermont and New York. Likewise several papers have been published on the use of this type of information in criteria development in Minnesota (e.g. Heiskary and Walker, 1988), comparing user perceptions across ecoregions and two states (Smeltzer and Heiskary, 1990), and a recent effort to assess user perception across several USEPA nutrient ecoregions and states (NYFLA and NYDEC, 2003). User survey data has been routinely used in Minnesota's 305(b) report to Congress as a basis for assessing support and non-support of aquatic recreational use since 1988 (MPCA, 1988). Likewise the use of user survey data for this purpose has been supported in USEPA guidance on developing nutrient criteria (USEPA, 2000a). This summary will draw from these various sources.

Regional differences in user perception are evident for Minnesota. In an analysis of CLMP user perception data from 1987 Heiskary and Wilson (1989) noted that lake users in the NLF ecoregion have high expectations regarding water quality. For example in 75% of the observations ranked as "impaired" or "no swimming" or "high or severe algae" the corresponding Secchi reading was 2.0 m or less (Figure 27). In the CHF ecoregion 75% percent of the observations in these categories were associated with Secchi readings of 1.4 m or less. And, in the WCP ecoregion 75% of the observations in these categories were associated with Secchi readings of 0.9 m or less.

Figure 27. CLMP user perceptions of recreational suitability and physical appearance based on 1987 data (Heiskary and Walker, 1988).





A much larger data set was available for an analysis and comparison of user perception data for Minnesota and Vermont (VT). In this study Smeltzer and Heiskary (1990) examined user perception responses for three Minnesota ecoregions (NLF, CHF, and WCP/NGP combined), Vermont's inland lakes and Lake Champlain. This included data from almost 500 lakes and resulted in 716 TP, 1,042 chlorophyll-a, 8,331 Secchi and about 7,500 measures of user perception. Prominent findings from that study are as follows:

- Observers in VT tended to associate respective survey response categories with higher measured water transparencies than did lake users in Minnesota.

- Within Minnesota, the Secchi values for the NLF ecoregion were generally higher for a given response than in the other regions (Table 10). The regional differences were especially pronounced at the lower end of the survey scale, i.e., responses of "crystal clear" and "beautiful, could not be any nicer." The differences between regions and survey response were statistically significant (example for physical appearance below)
- Results from this study reaffirmed regional patterns noted in earlier work (e.g. Heiskary and Wilson, 1988).

Table 10. Geometric Mean Secchi (meters) relative to user perception of physical appearance/recreational suitability. Drawn from Smeltzer and Heiskary (1990).

Region/perception (# lakes)	Clear / Beautiful	Not quite crystal clear / minor aesthetics	Definite algal green / slight impairment	High algal levels / no swim	Severely high algal levels / no swim or boat
NLF (216)	4.0 / 3.8	3.0 / 3.0	2.2 / 2.0	1.5 / 1.6	1.1 / 1.2
CHF (175)	3.0 / 2.8	2.1 / 1.9	1.4 / 1.2	1.0 / 0.8	0.9 / 0.5
WCP/NGP (23)	1.9 / ---	0.9 / 0.9	0.8 / 0.7	0.7 / 0.6	0.6 / 0.5

User perception data has long been used as a basis for defining support, partial support and non-support of aesthetic and aquatic recreational uses in MPCA 305(b) reports (e.g., MPCA, 1988). [Defining the degree of support or non-support is an EPA requirement in the 305(b) report.] The varying degrees of support are a function of the "frequency of nuisance conditions," e.g., algal blooms or reduced transparency. Impaired lakes would be characterized by a high frequency of conditions characterized as "impaired swimming or no swimming." Each of the categories is based upon user perception as it relates to lake trophic status (Carlson's TSI). General definitions for support, non-support, etc., as used in previous 305(b) reports follows:

1) Fully supporting: Lakes fully supporting should exhibit "impaired swimming" conditions less than 10% of the time and in terms of physical condition should exhibit "high algal levels" less than 10% of the time.

2) Fully supporting – threatened: These lakes may exhibit "impaired swimming" conditions 11-25% of the time and high algal levels 11-25% of the time.

3) Partial support – impaired: These lakes may exhibit "impaired swimming" 26-50% of the time and "no swimming" less than 10% of the time. In terms of physical condition, these lakes may exhibit "high algal" levels 26-50% of the time.

4) Non-support – impaired: These lakes will exhibit "no swimming" conditions greater than 25% of the time and "no recreation possible" on occasion. In terms of physical condition, these lakes will exhibit "high algal" levels greater than 50% of the time.

Using these criteria for defining support – non-support, etc., we originally related MPCA staff perceptions to lake trophic state as derived from reference lake data and later updated this with perception responses from CLMP participants. Based on this assessment, the following categories were used as a basis for determining support, non-support, etc. Lakes with an average

(average of available TSI indicators (TSI $\leq$ 50 were classified as fully supporting swimmable and aesthetic uses. From a trophic standpoint, this would correspond to oligo-mesotrophic conditions (Figure 3). Lakes with an average TSI between 51-59 were classified as supporting, but threatened. This TSI range corresponds to “mildly” eutrophic lakes. Lakes with an average TSI from 60-65 were classified as partially supporting but impaired. This range corresponds to a transition between eutrophic-hypereutrophic conditions. Lakes with an average TSI > 65 were classified as non-supporting. Lakes in this range are frequently considered hypereutrophic. The TSI level selected as a basis for determining “impaired” use, i.e., TSI > 60, appears to have support in the form of existing state beach-safety standards and CLMP user perception (Table 8). At a TSI of 60 or greater, the corresponding Secchi transparency will be 1 meter or less. The states of Massachusetts (MDPH, 1969) and New York (Effler, et al., 1984) impose a standard of > 1.2 meters for beaches. The Province of Ontario also imposes a 1.2 m transparency criteria for beaches as a safety consideration (Ontario, 1979). Thus, a transparency range of 1.0 – 1.2 m appears to be a minimum range to use as an indication of “no swimming” considering both safety and user perception.

These ranges and definitions have been modified over time to accommodate changes in USEPA reporting requirements for 305(b) and related assessments (e.g., 303(d)) but the basic principals noted herein are still applicable. For example, in more recent 305(b) assessments ecoregion-specific thresholds were established (MPCA, 1994), TP-derived TSI values are used rather than mean TSI (MPCA, 2000) and the “threatened” category is no longer used (MPCA, 2003). In the future nutrient criteria will be the basis for these assessments.

Watershed and Modeling Considerations

a. Regional patterns in land use

An understanding of regional patterns in land use, phosphorus export, and factors related to watershed – lake interactions are useful in the criteria setting process. We will discuss some of these factors in both statewide terms and by ecoregion, based on the reference lake data set. While the data used in this analysis is rather dated (c. 1967-69) we believe it adequately reflects current-day land use patterns among the regions. One possible exception may be a slight increase in the percentage of land in residential/urban uses in the CHF, in particular near the Twin Cities Metro Area. This increase in urbanized land use typically replaces agricultural and sometimes forested uses in this area.

Land use varies regionally across Minnesota (Tables 11a & b). The NLF ecoregion is dominated by forests, water and wetlands; while the NGP and WCP are primarily cultivated with some pasture and open land. The CHF ecoregion consists of a mixture of land uses. Table 11b provides a basis for comparing individual lake watersheds to those of the reference lakes.

In the NLF, forested and wetland uses dominate the overall land use composition of the region (Tables 11a & b). Based on individual watershed data, a “typical” range (25^{th} to 75^{th} percentile) for forested land use may be on the order of from 50% to 80% of the watershed (Table 12b). Water and marsh land uses typically range from 15% to 30% of the total watershed. In contrast, cultivated land use is typically less than 1% and pastured and open land use ranges from 0% -

6%. The residential and urban component is typically low (0% - 7%), however, this is biased in this sample set due to the lake selection criteria employed. This component can be expected to be quite variable among the lakes in the region.

TABLE 11a. Land use patterns by ecoregion. Percent of 40 acre parcels with land use characteristic derived from 1968-69 land use data (Planning Information Center, 1986).

	ECOREGION			
LAND USE (%)	**NLF**	**CHF**	**WCP**	**NGP**
Forest	75.2	15.9	3.5	0.7
Water & Wetland	10.6	8.1	1.7	2.9
Cultivated	4.6	49.3	82.9	83.7
Pasture & Open	7.3	21.4	10.0	11.4
Developed	1.9	5.3	1.8	1.1
Extractive	0.4	0.1	0.1	0.1

TABLE 11b. Land use composition (IQ range) for reference lake watersheds.

	ECOREGION			
LAND USE (%)	**NLF**	**CHF**	**WCP**	**NGP**
Forest	54-81%	6-25%	0-15%	0-1%
Water & Wetland	14-31%	14-30%	3-26%	8-26%
Cultivated	< 1%	22-50%	42-75%	60-82%
Pastured	0-6%	11-25%	0-7%	5-15%
Cultivated & Pastured	0-7%	36-68%	48-76%	68-90%
Developed	0-7%	2-9%	0-16%	0-2%

In the CHF (Tables 11a & b), no single land use dominates. Cultivated plus pasture and open land uses account for about 70% of the land use in the region (Table 11a). Forested land use covers about 16% of the region and is markedly lower than in the NLF. Developed land use (residential and urban) is slightly higher than in the NLF, and can be expected to increase in the Twin Cities Metropolitan Area (TCMA) and out state population centers. For example, in a data set of 24 lakes in the TCMA compiled by Osgood (unpublished data), the developed land uses ranged from 2-94% with a median of 10%.

In the WCP and NGP, agricultural land uses predominate. Cultivated land use comprises about 80%, pastured and open land use about 10% and forested land use is less than 3% (Table 11a) in each ecoregion. Similar patterns are evidenced in land use composition of the ecoregion lakes (Table 12b). However, on an individual lake basis there may be a fair amount of variation in the composition of land uses. For example, cultivated land uses in the WCP may typically range from 42-75% (Table 12a).

The watershed data reveal some patterns in land use, which when combined with lake morphometry data (Table 1), may be used to help define which ecoregion should be used for criteria purposes if a given lake is located at or near an ecoregion boundary. For example, a lake near the WCP/CHF boundary with a maximum depth of 30 feet and a watershed with 40% or more of forested and wetland land use is more likely classified as a CHF lake rather than a WCP lake based on Tables 1 and 11. The following is a summary of land use characteristics:

NLF

- forested (dominant) typically, 50% to 80%;
- cultivated (minor), typically < 1%;
- developed (low), typically 0% to 7%, but may be much higher for lakes with small, highly developed watersheds

CHF

- cultivated and pastured and open (variable), typically 35% to 70%;
- forested (low), 5% to 25%;
- developed (low in non-urbanized areas), 2% to 9%

WCP

- cultivated (dominant), typically 42% to 75%;
- pastured and open (low) typically, 1% to 7%;
- residential and urban (low), typically 0% to 16%

NGP

- cultivated (dominant), typically 68% to 90%;
- pastured and open (low), typically 5% to 15%;
- residential and urban (low), typically < 1%

The composition of land uses in the watershed of a lake in combination with the morphometry and water flow-through characteristics of a lake are primary factors in determining the quality of a lake. For example, regression analysis (Spearman rank, r_s) of the reference lake data revealed highly significant ($P < 0.001$) correlations between epilimnetic total phosphorus concentrations and cultivated land use ($r_s = 0.73$), forested land use ($r_s = -0.70$) and lake mean depth ($r_s = -0.61$). In general, this implies that in-lake TP concentration varies positively with increased percentage of land in cultivated uses and negatively with the percentage of land in forested uses. These responses are in turn modified by the mean depth of the lake. Further regression analyses were conducted on this data and can be found in Fandrei, et al. (1988).

The significance of the percentage of developed land, i.e., urban and residential is not adequately addressed by the ecoregion data due to a bias against the selection of lakes in urban areas. However, based upon previous studies, e.g., Ayers, et al. (1980), Smart, et al. (1981) and Walker (1985a), the percentage of land in developed land uses can be expected to play a very significant role in determining the water quality of a lake. Based on reported phosphorus export coefficients (Rast and Lee, 1983 and Reckhow, et al. 1980) its influence may be equivalent to or greater than that of cultivated land.

The combination of land use and other factors that account for the differing characteristics of the ecoregions strongly influence stream water quality. In a statistical assessment of water quality data from minimally impacted stream sites McCollor and Heiskary (1993) characterized the "typical" water quality of the various ecoregions. Table 13 demonstrates the distinct differences in TP and TSS concentrations among four of the ecoregions. In turn runoff from the streams will strongly influence the water quality of the lakes in each ecoregion and is among the factors considered in the modeling section that follows.

Table 12. IQ Range of Concentrations for Minimally Impacted Streams in Minnesota. Data from 1970-1992 (McCollor and Heiskary, 1993)

	Total Phosphorus (mg/L)			Total Suspended Solids (mg/L)		
Region/Percentile	**25%**	**50%**	**75%**	**25%**	**50%**	**75%**
NLF	0.02	0.04	0.05	1.8	3.3	6.0
NMW	0.04	0.06	0.09	4.8	8.6	16.0
CHF	0.06	0.09	0.15	4.8	8.8	16.0
NGP	0.09	0.16	0.25	11.0	34.0	63.0
RRV	0.11	0.19	0.30	11.0	28.0	59.0
WCP	0.16	0.24	0.33	10.0	27.0	61.0

b. Lake modeling considerations

In this section, we summarize three levels of modeling that could be employed for purposes of goal setting in the case of lake protection or restoration projects, as a means for assessing the impact of current or changing land uses in the watershed on in-lake phosphorus concentrations or as a part of a TMDL assessment. This discussion is summarized from a more comprehensive document on the application of empirical modeling techniques for Minnesota lakes (Wilson, 1988). While data input requirements will differ between the three levels of modeling, there is a basic core of information required for each and generally includes such things as lake morphometry, observed in-lake conditions, and watershed characteristics as listed at the beginning of the chapter on "Considerations for Lake Water Quality Criteria Development."

Eutrophication models are often categorized as being either mechanistic or empirical. Mechanistic approaches simulate complex physical, biological and chemical processes describing cause and effect relationships mathematically. Mechanistic models may also be used to define lake or ecosystem responses that are spatially and temporally variable. One such model, MIN-LAKE, was developed by the University of Minnesota. MIN-LAKE combines the engineering principles of conservation of mass and energy with the dominant chemical and biological kinetics affecting the dynamics of nutrient fluxes, uptake and algal growth (Stefan and Riley, undated). This model was developed to evaluate the effectiveness of lake restoration methods in a lake on a daily time scale. Treatment methods can then be modeled by modifying these components of the model, which will be directly affected by the treatment process (Riley and Stefan, 1987). Unfortunately, inclusion of MIN-LAKE in this assessment was not possible. Examples of some mechanistic models developed by or for the U. S. Army Corps of Engineers may be found at: http://www.wes.army.mil/el/elmodels/index.html#wqmodels .

Empirical approaches are based upon mass-balance and limiting nutrients, and relate average in-lake conditions to watershed loadings, hydrology and lake morphometry. Empirical models are typically derived from groups of lakes by statistical relationships. When used within their limits, empirical models offer advantages such as simplicity and limited data requirements. The following discussion will deal only with empirical lake models.

Empirical lake models generally involve the linkage of two types of models (Reckhow and Chapra, 1983):

1. Nutrient Balance Models that relate pool or discharge nutrient levels to external nutrient loadings, morphometry and hydrology.
2. Eutrophication Response Models that describe relationships among eutrophication systems such as average total phosphorus, total nitrogen, chlorophyll-a and Secchi depth.

Several researchers have developed efficient methods for lake water quality assessment based upon empirical lake modeling (Dillon and Rigler, 1974; Larsen and Mercier, 1976; Vollenweider, 1976; Canfield and Bachmann, 1981; Reckhow, et al., 1980; Reckhow and Simpson, 1980; and Walker, 1985). One important feature of the recent work (besides general utility) has been the concept of error analysis with the empirical lake modeling (Reckhow and Simpson, 1980; Walker, 1985b).

Inherent in the previously cited literature has been the significance of nutrients, particularly phosphorus, to the evaluation of lake trophic quality. Measures of the supply of phosphorus from the watershed have been related to in-lake phosphorus concentration which, in turn, has been related to algal population density as measured by chlorophyll-a concentration and water clarity (Sakamoto, 1966; Dillon and Rigler, 1974). Generally, these models have quantified in-lake total phosphorus concentrations based on annual phosphorus loading inputs, hydraulic flushing and lake morphometry (Vollenweider, 1975; Larsen and Mercier, 1976). Lake water quality, as estimated by these models, relates to watershed phosphorus management since phosphorus is generally not available from natural sources in large quantities. Most importantly, phosphorus is the only essential nutrient that can be effectively managed to limit algal growth (Golterman, 1975).

While nitrogen can also limit algal growth, it may be obtained from the atmosphere by certain algal species. Nitrogen fixation, as it is referred to, has been correlated with the presence of blue-green algae that possess heterocysts (Wetzel, 2001). Thus, attempting to control algal growth by focusing on nitrogen alone may not be too successful. However, reductions in nitrogen loading to lakes in conjunction with reductions in phosphorus loading may be beneficial for reducing algal growth and/or reducing the frequency of nuisance algal blooms.
Models have also been used to measure the impact of lakeshore development on water quality and the ability of a lake to support various levels of recreation and development. Dillon, et al. (1986) present a simulation model which measures the source of environmental impact of lakeshore cottages and their use in terms of water quality, fisheries and wildlife habitat. Hutchinson (2002) prepared an update to Ontario's Trophic Status Model for establishing lake development guidelines. The reader is referred to these publications for further information on these types of models and their application.

The following is a discussion of the models we call "Level I." Common to these models and the science of limnology are a number of abbreviations as summarized in Table 13.

In the context of this assessment, the assessment methodology may be used to define the relative changes that may occur in a given lake based upon analyses of similar lake systems. Many other factors may influence a given lake's eutrophication responses (e.g., internal loading, macrophytes, phytoplankton- zooplankton-fisheries communities). The following assessment methodologies are practical tools for analyzing lake systems. The degree of precision, or conversely uncertainty, associated with predictions from these models will depend upon the above considerations and the experience of the analyst.

TABLE 13. Commonly used limnological abbreviations

	Abbreviations	Practical Units
V	Lake Volume	m^3 or hm^3 (HM^3)
A_o	Lake Surface Area	m^2 or ha or km^2 (KM2)
A_w	Watershed Area	ha or km^2
z	Depth	m
ź	Mean Depth	m
Q_{out}	Yearly Water Discharge	m^3/year
q_s	Areal Hydraulic Load (Q_{out}/A_o = ź/Tw)	m/year
Tw	Water Residence Time	year
Pw	Flushing Rate (1/Tw)	$year^{-1}$
L_p	Total Phosphorus Supplied to Lake	kg/year
[P]	Average Lake TP Concentration	mg/m^3 or µg/l
[P_{sp}]	Average Spring TP Concentration	mg/m^3
L	Phosphorus Loading Rate (L_p/Q_y)	mg/m^2/year or gm/m^2/year
[P_{in}]	Average Inflow TP Concentration (L_p/Q_y)	mg/m^3
6_p	Phosphorus Sedimentation Rate	$\cong 0.162\ (L/z)^{0.458}$ Canfield & Bachman (1981 for natural lakes $\cong 0.114\ (L/z)^{.589}$ Canfield & Bachman (1981) for artificial lakes
R	Phosphorus Retention Coefficient	$\cong 1/(1 + Tw)$
RO	Runoff	m/yr
PT	Precipitation	m/yr
PE	Evaporation	m/yr

Level I Assessment

The first level of assessment, the Minnesota Lake Eutrophication Analysis Procedure (MINLEAP), is intended to be used as a screening tool for estimating lake conditions with minimal input data and for identifying "problem" lakes. MINLEAP is a computer program designed to predict eutrophication indicators in Minnesota lakes based upon watershed area depth and ecoregion. The program employs a network of empirical models, which have been regionally calibrated. MINLEAP is an interactive computer program originally written in BASIC (IBM-compatible personal computer) language but now available in a Windows-based format from the MPCA web site: http://www.pca.state.mn.us/water/charting.html . Input data required for each run include: watershed area, lake area, mean depth, and ecoregion. Monitored average in-lake values for TP, chlorophyll-a and Secchi depth may also be input, but are not required.

MINLEAP was developed from the reference lake data set (Wilson and Walker, 1989) as further explained below. Watershed and lake areas, mean depths and average summer total phosphorus (TP), chlorophyll-a and Secchi depths were determined for each of the study lakes. Average annual precipitation and evaporation isopleths, runoff coefficients and atmospheric loadings were assembled for the state and characterized by ecoregion in Table 14.

Table 14. Ecoregion variables for MINLEAP development..

Ecoregion	Stream P PPB	Atmos Kg/Km^2-YR	Precip (P) M	Evap (E) M	Runoff M	P-E M	Export Kg/Km^2-YR
CHF	150	30	0.8	0.7	0.1	0.04	19
NLF	50	15	0.7	0.6	0.2	0.13	12
NGP	1500	30	0.6	0.8	0.01	-0.12	76
WCP	550	30	0.8	0.7	0.1	0.06	74

Where: Atmos = Atmospheric P Export; Export = P Export in Kg/Km^2-year

The procedure is based upon simplified hydrologic and phosphorus balances described by Equations 7 and 8. The hydrologic balance is determined from the use of ecoregion average precipitation, evaporation, and direct runoff values from Table 14 as stated by Equation 7. The annual supply of phosphorus is estimated by the flux of atmospheric and stream phosphorus reaching the lake (Equation 2). Average stream phosphorus concentrations have been calibrated to given unbiased predictions of lake phosphorus concentrations in each ecoregion (Table 16).

Water Income (Q) = [Runoff x Watershed Area] + [Lake Area x (Precipitation – Evaporation)]
= [RO x Aw] + [Ao + Pr] (Eq. 7)

Phosphorus Supply Lp = [Lake Area x Atmospheric Export] + [Watershed Area x Runoff x Regional TP]
= [Ao x ATM] + [Aw x RO x TPeco] (Eq. 8)

The quantities of water and phosphorus calculated by MINLEAP are used in conjunction with the Canfield and Bachman (1981) model for natural lakes. This phosphorus retention model was employed in MINLEAP due to its general utility. The phosphorus/chlorophyll-a and chlorophyll-a/Secchi models used in MINLEAP were developed from the reference lake data (e.g. Figure 4b).

MINLEAP has been calibrated to the ecoregion data set by adjusting the stream phosphorus concentrations (i.e., Table 15) to give unbiased predictions of lake phosphorus concentration in each region. As illustrated in Table 16, calibrated stream phosphorus concentrations depend upon which model is predicting lake phosphorus retention. Since the residual errors are similar for each model tested, there is no basis for deciding which retention model is "best" for Minnesota lakes without direct loading measurements. Therefore, results presented below are based upon the Canfield and Bachman (1981) retention model (natural lake version).

Table 15. Calibrated stream phosphorus concentrations by ecoregion (in µg/L).

	Phosphorus Retention Model			
REGION	**A**	**B**	**C**	**OBS**
NLF	55	32	48	59
CHF	150	85	70	132
WCP	600	420	220	310
NGP	1500	1050	220	250
RSE	0.179	0.171	0.182	

A Canfield & Bachman (1981)
B Walker (1985) Second Order
C Vollenweider (1976)
OBS Median Measured Stream Phosphorus Conc. (McCollor and Heiskary, 1993)
RSE Residual Standard Error – Log10 (Lake P)

For the WCP and NGP, calibrated stream phosphorus concentrations exceed median measured values by factors of 1.9 and 6.0, respectively. It is unlikely that measured median stream phosphorus concentrations estimated from routine grab sample monitoring adequately reflect high-flow conditions which are responsible for the bulk of the phosphorus loading. Particularly in agricultural watersheds, stream phosphorus concentrations can increase dramatically under high-runoff conditions. Infrequent runoff events account for a large portion of the total annual loading and are not adequately reflected by median values derived from routine, periodic stream sampling. The calibrated stream phosphorus concentration for NGP is relatively uncertain because of the small number of lakes (8 in the NGP versus 11-36 in the other regions), potentially long water retention time because of low water inflow and high evaporation rates (Table 15), and high internal recycling of P via wind mixing, resuspension, and other factors.

As seen in Table 16, NLF streams have the lowest phosphorus average inflow concentrations (range 24 – 52 ppb), low estimated water residence times (average = 5 years) and greater mean depths (average = 6.3 m). In contrast, the lakes of the primarily agricultural zones, the WCP and NGP, have much greater estimated average inflow concentrations (91 – 340 µg/l), shallower mean depths (averages ranging from 1.6 to 2.5 m) and longer estimated water residence times (averages ranging from 4.8 to 36 years). Lakes of the transition zone between the NLF and the agricultural regions, the CHF, have average inflow phosphorus concentrations and water residence times also intermediate between values described previously. Net regional phosphorus export (expressed in kg/ha/year) varied from 0.14 to 0.22 in the NLF and CHF, respectively, to about 0.80 in the NGP and WCP.

As previously stated, MNLEAP was developed to assist in identifying lakes which may be in better or worse condition than they "should be" based on their location and lake basin morphometry. It is also valuable for demonstrating important differences between lakes in the four ecoregions. When applied in conjunction with the Level II assessment, it can serve as a means of corroborating the results and hopefully reducing the amount of "error" in the predictions. MINLEAP may be particularly useful for prioritizing lakes at a regional level in terms of identifying those requiring "protection" versus those requiring "restoration" (Heiskary, 1991). It will continue to be used after nutrient criteria are promulgated and is proving to be a useful diagnostic tool in 303(d) "TMDL" assessments as well.

Level II Assessment

Level II assessments typically require substantially greater amounts of data and utilize a network of empirical methods by (1) Reckhow and Simpson (1980), (2) Reckhow (1983) and (3) Walker (1984). These methods have been structured on EXCEL spread sheets to facilitate "what if" evaluations. Reckhow and Simpson (1980) have improved upon the lake carrying capacity methods of Dillon and Rigler (1975) by the inclusion of (1) error estimation procedures and (2) a lake phosphorus predictive model with general applicability for a wide range of North American lakes.

Input data include watershed areas by land use, runoff, net precipitation (precipitation minus evaporation), lake surface area, number of seasonal and permanent dwellings, and phosphorus export and point source mass loading values. The latter two variables are expressed in terms of low, most-likely and high ranges for calculation of uncertainty estimates.

This methodology is used as a means for assessing the impact of current or changing land uses in the watershed on lake eutrophication responses. Before and after land development scenarios may be evaluated by following Reckhow and Simpson's framework. When predicting the effects of changing land use upon lake water quality, inclusion of relatively limited lake data using the methods of Reckhow (1983) may substantially reduce the uncertainty of the estimates defined above. Additionally, the inclusion of lake specific chlorophyll-a and Secchi average data will allow calculation of frequency or risk of nuisance algal levels for a given lake using the methods of Walker (1984).

Table 16. Ranges of monitored average stream TP, estimated water residence time, hydraulic water loads, mean depths and TP export for MINLEAP database.

Ecoregion	Range of Typical* Stream TP (µg/l)	Water Residence (years)	Water Load (m/yr)	Mean Depth (m)	Estimated TP Export (Px) (kg/ha/year)
NLF					
Range	24 – 52	0.4 – 15	0.7 – 7.7	0.8 – 15	--
Mean	46	5	1.7	6.3	0.14
St Dev	51	3.8	1.4	4.0	--
	N = 30				
CHF					
Range	70 – 170	1.0 – 30	0.2 – 7.4	2.7 – 13	--
Mean	145	9.3	1.3	6.6	0.22
St Dev	158	6.6	1.5	2.4	--
	N = 36				
WCP					
Range	181 – 340	0.9 – 13	0.3 – 2.0	1.5 – 4.4	--
Mean	304	4.8	.8	2.5	0.78
St Dev	338	3.8	.6	1.0	--
	N = 11				
NGP					
Range	99 – 271	1.1 – 217	0.1 – 1.0	1 – 2.1	--
Mean	218	36.2	0.4	1.6	0.80
St Dev	190	69.7	0.35	0.3	--
	N = 8				

Where:
Range* = 25^{th} – 75^{th} percentiles (from Fandrei, et al., unpublished data)
Mean = Average Annual Total Phosphorus Values
St Dev = Standard Deviation
Phosphorus Export = Net Phosphorus Export Calculated from MINLEAP Equations 1 and 2 divided by the watershed area. Values represent ecoregion averages for data base.

Level III Assessment

The final level of modeling that will be referenced is the network of empirical models FLUX, PROFILE and BATHTUB that were developed by Walker (1985b) for the U.S. Army Corps of Engineers, Waterways Experimental Station (USACE-WES), Vicksburg, Mississippi. Detailed documentation of the model origins, structures and limitations may be found in Walker (1985b and 1996). A recently completed Windows-compatible version of the model is available from the USACE web site at: http://www.wes.army.mil/el/elmodels/index.html#wqmodels .

Potential applications of these models were summarized by Walker (1985b) as follows:

Diagnostic

- Formulation of water and nutrient balances, including identification and ranking of potential error sources;
- Ranking trophic state indicators in relation to user-defined data groups or the USACE reservoir data base;
- Identifying factors controlling algal production.

Predictive

- Assessing effects of changes in water and/or nutrient loadings;
- Assessing effects of changes in mean pool level or morphometry;
- Predicting long-term average conditions in a new reservoir; and
- Estimating loadings consistent with water quality objectives.

Application of FLUX and BATHTUB may require data from intensive monitoring programs, e.g., monitoring associated with diagnostic studies. Illustration of these programs is beyond the scope of this paper. In general, this software is appropriate for use in reservoir or lake restoration projects (e.g., Clean Lakes, CWP, or TMDL-related projects). Additionally, the FLUX program will be of general utility for calculations of loading estimates. Copies of the documentation and software may be obtained from the USACE-WES. A link to that site is available at: http://www.pca.state.mn.us/water/charting.html .

c. Sediment-diatom reconstruction of in-lake P

Lake-sediment cores provide a valuable archive of information on water quality and related environmental factors. One technique that was applied to 55 Minnesota lakes ("55 lake study" as described in Heiskary and Swain, 2002) involved reconstructing water quality from pre-European to modern-day based on fossil diatoms. In that study 50 of the 55 lakes sediment cores (from NLF and CHF; Table 17) had previously been collected as a part of a study to document temporal trends in mercury (hence the lakes were selected for purposes other than eutrophication trends). In general those 50 lakes were of moderate size and depth (Appendix II) and could be considered rather typical of the regions in which they were located. In contrast the five WCP lakes added to that data set were somewhat deeper than the norm for the region (Table 1).

In the 55 lakes study (Heiskary and Swain, 2002) and subsequent collections in six shallow southwest Minnesota lakes (Edlund and Kingston, 2004) and eight shallow west-central lakes

(Edlund et al. 2005) we are able to describe "background" P concentrations for each of the four previously described ecoregions. In the 55 lakes study sediment cores, obtained by a piston corer, were sectioned and dated. Core-sections corresponding to circa 1750, 1800, 1970, and 1993 time-periods were used in this analysis. Fossil diatoms were identified and enumerated in each of these samples. Predictive models were developed based on modern-day water quality (c. 1993) and diatom populations in these lakes. The models were then used to predict pre-European TP and other water quality parameters. Details on the actual reconstruction techniques and models may be found in Ramstack et al. (2002), Edlund and Kingston (2004), and Heiskary et al. (2004). A complete listing of the lakes from these three studies are included in Appendix II as modified from Edlund (2005).

It is important to note that diatom-inferences for the 55 lakes study (Table 17) were based on a model that was developed as a part of that study. In contrast the southwest and west-central Minnesota studies, which focused on shallow lakes, make use of the same model as well as data collected from the 24 shallow southwest Minnesota lakes. This results in an expansion of the model (referred to as the 79 lake model), which should improve inference of pre-European P concentrations across a wider range in trophic status. In a later phase of this work all the west-central lakes (29 lakes) and data from Itasca County lakes will be brought into the training set to yield a database on approximately 145 lakes. This may allow for the development of specific models for shallow lakes and/or some ecoregion –specific models to be developed (Edlund, 2005 personal communication). At this point we have applied the 79 lake model to all lakes (in the aforementioned studies) to provide an improved basis for among-region comparisons (Fig. 28).

Diatom reconstruction provides an opportunity to examine both temporal and spatial trends in lake trophic status and related factors (Heiskary and Swain, 2002). Ecoregion-based patterns in lake trophic status have long been recognized in Minnesota (e.g., Heiskary et al. 1987). Lakes in the forested Northern Lakes and Forests (NLF) ecoregion are moderately deep and exhibit relatively low TP while the shallow lakes in the highly agricultural Western Corn Belt Plains (WCP) and Northern Glaciated Plains (NGP) exhibit high TP concentrations. The transitional North Central Hardwood Forests (CHF), characterized by moderately deep lakes and a mosaic of land uses -- intermediate between these two extremes.

Ecoregion-based patterns are evident in the pre-European data as well (Table 17, Fig. 28). The NLF lakes were significantly lower in TP as compared to the CHF, WCP and NGP lakes based on a comparison of group-means plus or minus standard error (SE) (Fig. 28). Based on SE (typically 2-3 µg/L or less) and overall range of TP concentrations the NLF lakes were somewhat less variable as compared to lakes in the other ecoregions. Variability was slightly higher among the CHF lakes and no significant difference was evident in a comparison of rural and Metro-area CHF lakes (Table 17, Fig. 28). The shallow WCP and NGP lakes were much more variable by comparison, with a SE of 10 µg/L. However, if we express this as a percent off the mean the SE for all regions is about 10-20 percent of the corresponding means for each region.

Table 17. Comparison of diatom-inferred P and observed P by ecoregion. Drawn from : a) Heiskary and Swain (2002); b) Heiskary et al. (2003) and c) Heiskary and Lindon (2005).

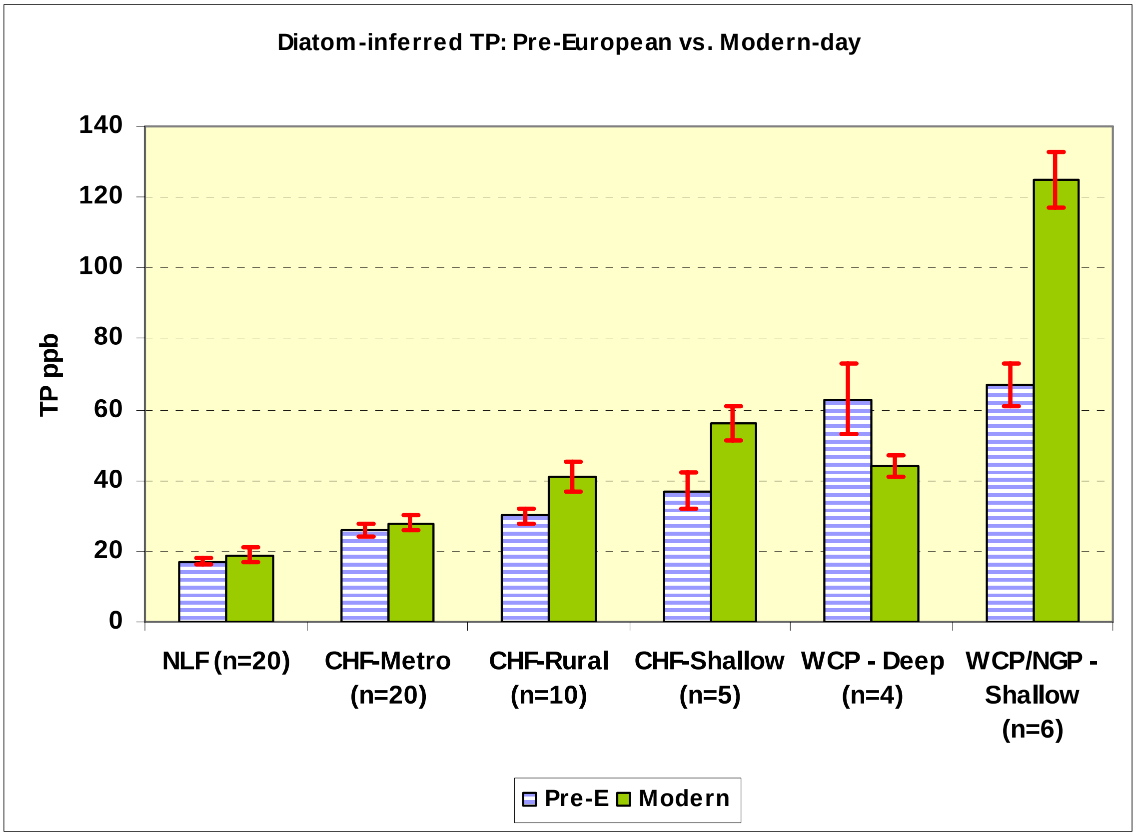
	------ Diatom-inferred -----				Observed TP
Region Subset	**Year (# of lakes)**	**1750-1800 Mean (SE)**	**1970 Mean (SE)**	**Modern Mean (SE)**	**modern Mean (SE)**
NLF (a)	20	15 (1)	15 (1)	14 (1)	15 (1)
CHF	30	24 (2)	39 (4)	37 (3)	39 (6)
Metro (a)	20	23 (2)	37 (4)	35 (4)	33 (6)
Rural (a)	10	27 (4)	44 (8)	41 (7)	49 (12)
Shallow (c)	5	37 (4)	---------	56 (8)	100 (17)
WCP					
Deep (a)	5	47 (6)	55 (10)	46 (6)	67 (16)
Shallow (b)	6	67 (11)	--------	125 (8)	108 (11)

Distinct differences among regions were evident in comparisons of pre-European and modern-day TP concentrations. For the NLF lakes, as a group, there was no significant difference in modern-day vs. pre-European TP (Fig. 28). However, distinct increases in TP were noted for the CHF lakes (Table 17) and these increases exceeded the “natural variability” noted in comparisons of pre-European (1750 and 1800) TP concentrations (Ramstack et al. 2004). They also note that the degree of change in TP among the Metro CHF lakes is significantly correlated with the percent of the watershed in urbanized landuse, while those in the rural portion exhibit a significant correlation with the percent of landuse in agricultural uses or inversely the percent in forested uses. Based on the results from the 79 lake model the increases from pre-European to modern-day for the Metro lakes was rather minimal (as a group); however the rural lakes exhibited a more distinct increase (Fig. 28).

The deeper WCP lakes also did not change significantly across the two time periods based on the “55 lakes” model and no significant associations with landuse were noted by Ramstack et al. (2004) – presumably because of the small sample size (5 lakes) and the predominately agricultural landuse in these watersheds. However, it is important to note that these five lakes had the highest modern-day P of any of the lakes in the 55 lake study and as such were at the fringe of the data used to construct the predictive model. When these four lakes (George Lake was excluded as an outlier; Edlund, 2005) were subjected to the 79 lakes model high variability among the lakes in the group was evident but little or no change from pre-European to modern-day (some lakes had lower modern-day inferred P) was evident.

Shallow CHF, WCP & NGP lakes added in the recent study of west-central (Heiskary and Lindon, 2005) and SW Minnesota lakes (Heiskary et al. 2003) exhibited a significant increase when pre-European and modern-day TP are compared (Fig. 28). Agricultural landuse predominates in all of the southwest lakes and most of the west-central lakes as well. A more detailed summary of sediment diatom reconstruction may be found in Heiskary et al. (2004).

Figure 28. Diatom inferred TP: comparison of pre-European and modern-day by ecoregion. Diatom-inferred based on 79 lake model for all data sets.



IV. LAKE POLLUTION CONTROL PROGRAMS IN MINNESOTA

Nutrient criteria will play an important role in the protection and restoration of lake water quality in Minnesota. The criteria can be used with existing regulatory, management, and educational programs. These programs are described in the following pages. Some examples of uses and application of the criteria are as follows:

1. Assessing condition of lakes for 303(d) "impaired waters" listing and 305(b) lake assessments.
2. Use in prioritizing and selecting projects to be funded through the Clean Water Partnership Program (Minn. Stat. section 115.091 to 115.103 (Supp. 1987)), and the federally funded Section 314 Clean Lakes and Section 319 Nonpoint Source Management Programs authorized by the Clean Water Act.
3. For use by resource managers in developing water quality management plans. For example, there are currently over 80 water management organizations in Minnesota preparing comprehensive local water management plans required or authorized under Minn. Stat. section 473.878 or Minn. Stat. chapter 110B.
4. Use as an educational tool for communicating what can reasonably be expected in terms of lake quality.
5. In the case of degraded lakes (e.g., 303(d) listed), the criteria can serve as reasonable targets or goals.
6. Guide enforcement and permitting decisions (e.g. effluent limitations, stormwater permits, feedlot and land application permits).
7. Particularly important for protecting the quality of lakes currently at or below criterion level.
8. Guide interpretations of nondegradation statutes.

Regulatory Programs

The Minnesota Pollution Control Agency (MPCA) has been empowered by the Minnesota Legislature "to administer and enforce all laws relating to the pollution of any of the waters of the state . . ." (Minn. Stat. § 115.03, subd. 1(a), 1986). For this purpose, several regulatory and grant programs are administered by the MPCA pertaining to point and nonpoint sources of pollution.

Point source controls relating to lakes principally limit bacteria and nutrients, especially phosphorus concentrations or phosphorus loading as specified in National Pollutant Discharge Elimination System (NPDES) or state permits. Rules pertaining to these limitations are included in Minnesota Rules (R.) Chapter (ch.) 7050 and 7065. A rule of particular importance to lakes is Minn. R. 7050.0211, Subp. 1a, which states: "Where the discharge of effluent is directly to or affects a lake or reservoir, phosphorus removal to 1 mg/l shall be required. In addition, removal of nutrients from all wastes shall be provided to the fullest practicable extent wherever sources of nutrients are considered to be actually or potentially detrimental to preservation or enhancement of the designated uses." We have been able to document improvement in lakes where major point source discharges have been removed from the watershed. Some examples include: Lake Minnetonka (Vighi and Chiaudani, 1985), Clearwater Lake in Wright County (see Case Study

16) and Lake Minnewaska in Pope County (see LAP report, MPCA web site). We have also documented improvements in lakes where dischargers were required to meet effluent limits of 1 mg P/L or lower; however in some cases (e.g. Shagawa Lake, Case Study 9) the improvements were not always as rapid or as complete as we may have hoped for.

Other rule provisions relevant to lake protection are Minn. R. 7050.0170, Subps. 3 and 5. Subpart 3 states "For all classes of fisheries and recreation waters, the aquatic habitat, which includes the waters of the state and stream bed, shall not be degraded in any material manner, there shall be no material increase in undesirable slime growths or aquatic plants, including algae, nor shall there be any significant increase in harmful pesticide or other residues in the waters, sediments and aquatic flora and fauna; the normal fishery and lower aquatic biota upon which it is dependent and the use thereof shall not be seriously impaired or endangered, the species composition shall not be altered materially, and the propagation or migration of the fish and other biota normally present shall not be prevented or hindered by the discharge of any sewage, industrial waste or other wastes to the waters."

Phosphorus limitations have also been applied to detergents as specified in Minnesota R. 7100.0210. These rules generally limit domestic detergent phosphorus content to less than 0.5% except for dishwashing detergents. Industrial and commercial detergents are generally exempted from this limitation.

Individual sewage treatment system standards have been specified by Minnesota R. ch. 7080. Administration and enforcement of these standards were specifically intended to be by the local units of government.

Standards for animal feedlots are specified by Minnesota R. ch. 7020 and Minn. R. 7050.0215 and administered in a cooperative fashion with counties choosing to participate in the program. Training is provided to county staff by MPCA staff to insure uniform administration of the feedlot rules. County programs, in many instances, represent considerable experience and sensitivity to local agricultural practices and to successful soil and water conservation. The MPCA's Division of Water Quality administers a Feedlot Permit Program to prevent pollution of surface and ground waters of the state. The owner of a proposed or existing animal feedlot for greater than ten animal units is required to apply for a permit when any of the following conditions exist:

a. a new animal feedlot is proposed;
b. a change in operation of an existing animal feedlot is proposed;
c. ownership of an existing animal feedlot is changed, or
d. an inspection by the MPCA staff or a county feedlot pollution control officer determines that the animal feedlot creates or maintains a potential pollution hazard.

Lakes in Minnesota are also protected by nondegradation provisions included as part of the state's water quality standards (Minnesota Rules 7050.0180 and 7050.0185). The degree of protection provided by the nondegradation provisions depend upon the classification and characteristics of the lake affected. The three general levels of nondegradation protection are as follows:

1. New or expanded point or nonpoint source discharges to a group of specific lakes are prohibited. The lakes receiving this highest level of protection are the lakes within the Boundary Waters Canoe Area Wilderness, Voyageurs National Park, and scientific and natural areas designated by the Minnesota Department of Natural Resources. All of these lakes are designated as outstanding resource value waters (ORVWs).

2. New or expanded point or nonpoint source discharges to other lakes designated as ORVWs are not allowed unless the discharger demonstrates to the satisfaction of the MPCA that there is no prudent and feasible alternative to the discharge. Lakes in this group include 35 existing and potential lake trout lakes (Minnesota R. 7050.0180 and 7050.0420).

3. All other lakes in Minnesota receive the general degree of nondegradation protection provided to all waters in Minn. R. 7050.0185. Under this level of protection, new or expanded point or nonpoint source dischargers that are likely to significantly lower baseline water quality, may be required to provide additional treatment beyond minimum applicable requirements in order to minimize the impact of the discharge on water quality. The rule describes what constitutes a significant lowering of baseline quality. Whether and what additional treatment requirements may be required depends upon a number of factors, including the costs of additional treatment and the relative economic and social importance of the project. In no case, may the discharge eliminate an existing beneficial use of the lake.

Other important rules pertaining to lakes have also been promulgated by several other state and local agencies, and it is beyond the scope of this effort to review these in any detail. However, the Minnesota Department of Natural Resources (MDNR) also administers a variety of programs relating to the protection, management and conservation of lake associated natural resources. A few of these programs are described as follows:

a. Water allocation: Minn. Stat. §§ 105.05 through 105.418;

b. Shoreland management: Minn. Stat. §§ 105.485

   The Shoreland Management Program is a cooperative state/local government effort to manage development near lakes and rivers. It is intended to control development in a manner that protects both the natural and economic values of lands located within 1000 feet of lakes and 300 feet of rivers throughout the state. The MDNR estimates the Program affects 93,000 miles of lake shoreline and 50,000 miles of rivers, which means about 28 percent of the state's land area is subject to local shoreland controls. The controls must meet or exceed state standards established in MDNR rules. The standards currently address primarily lot sizes, structure setbacks, and sewage system standards. They are being revised to address several additional areas such as storm water management, agricultural practices, and accelerated upgrading of sewage systems. These improvements should help local governments improve lake water quality.

c. Fisheries and wildlife management: Minn. Stat. §§ 97A, 97B, and 97C

   Fish management plans are prepared on about 400 lakes per year so that game fish production potential of the water is achieved. About 600 lake surveys are conducted per year to provide current information on the physical, chemical and biological characteristics of lakes as a basis for formulating management plans and evaluating ongoing management. Dissolved oxygen and temperature are monitored on about 600 lakes per year so that primary limiting factors for fish management are assessed. Angler and recreational use surveys are conducted to document fish harvest and pressure on individual lakes. Fish are stocked as needed to improve angling opportunities. Habitat is improved through acquisition and development of spawning areas, protection of shorelines and rough fish control. Wetlands and impoundments are acquired, protected and developed for waterfowl – furbearer production. Water levels are manipulated to promote suitable growths of aquatic vegetation for wildlife. Regulations are promulgated on the harvest and use of fish and wildlife.

d. Water surface use zoning: Minn. Stat. §§ 378 and 361.

e. Aquatic nuisance control: Minn. Stat. §§ 18B and 84.

   Regulation of aquatic plant and algae control activities on public waters through administration of a permit program, conduct of an information education program on aquatic nuisance control, licensing of commercial aquatic applicators and plant harvesters, review and authorization of use of EPA-approved aquatic herbicides, investigation of alleged misuse of aquatic herbicides, evaluation of the impact of plant control operations and control of the noxious aquatic plant, purple loosestrife.

f. Lake aeration: Minn. Stat. § 378.22

   Regulation of aeration activities on public waters through administration of a permit program, aeration of waters through MDNR cost sharing with local interest groups, conduct of an information education program on lake aeration, carrying out a winter ice safety and aeration system inspection program and evaluation of the impact of lake aeration activities.

In addition to these programs, MDNR manages a number of other programs that may directly or indirectly relate to lake management. A listing of these programs is as follows: lake level management, flood plain management, public access, forest management, lake improvement districts, waterbank, Reinvest in Minnesota (RIM) set aside and RIM critical habitat.

<u>Management – Grant Programs</u>

The MPCA has been designated as the state agency to administer grants awarded to the state from the U.S. EPA through the Section 314 Clean Lakes Program and Section 319 Nonpoint Source Program, as authorized by the Clean Water Act. The purpose of the Clean Lakes Program is to preserve and protect Minnesota's lakes to increase and enhance their public use and enjoyment. The Clean Lakes Program was an important part of the MPCA's efforts to address lake water quality problems. As of 1988, the MPCA had completed nine Clean Lakes projects and had 17 projects underway. However, Section 314 has not been formally funded since the mid 1990s. Three of the Clean Lakes projects were nonpoint source demonstration projects and the success of these demonstration projects was instrumental in establishing the state's Clean Water Partnership Program.

Recognizing the magnitude of nonpoint source pollution (NPS) and the need to establish a program for its control, the Minnesota Legislature enacted the Clean Water Partnership Program (CWP) (Minn. Stat. sections 115.091 to 115.103, Supp., 1987). The program focus is control of nonpoint sources of pollution to protect and improve surface and ground water quality in Minnesota. The program built on existing local water planning efforts established by the Comprehensive Local Water Planning Act (Minnesota Statutes, Chapter 110 B), the Metropolitan Surface Water Management Act (Minnesota Statutes, Section 473, 878) and the Watershed Management Act (Minnesota Statutes Chapter 112).
The CWP program accomplishes this through state financial and technical assistance to local units of government for water quality projects. Lake projects have been an important part of the surface water projects.

Similar to the Clean Lakes Program, CWP projects involves two phases: diagnostic studies and implementation activities. The diagnostic studies will analyze specific water quality problems, define realistic project goals and objectives and identify the management efforts necessary to meet the project goals. The implementation phase includes installation of management practices, education activities and other measures necessary to control nonpoint sources of pollution.

The MPCA is responsible for administration of this program. Administration includes project selection, award of grants, fiscal management, project coordination, review of project reports and technical assistance through development of guidance documents, assistance with monitoring design, computer modeling and technical review of products and activities. An interagency project coordination team was established to advise the agency in preparation of rules, selection of projects and coordination of the projects with other resource management programs. From 1987 to 2005 the legislature has provided the CWP Program with over $25 million for grants to local units of government. In addition, since 1995 the CWP program has awarded over $40 million in loan money. During the 1999 legislative session, the legislature significantly increased the annual appropriation to the CWP Program from slightly less than a million to approximately $2.4 million. The legislature did specify however, that the new appropriation of money over the base amount was to be used for implementation projects and continuations.

Additionally, the CWP establishes the authority and mechanism for Minnesota to implement projects on a watershed basis through the federal Nonpoint Source Management Program established by Section 319 of the Water Quality Act of 1987. These two programs are coordinated via Minnesota's Nonpoint Source Management Program Plan**.** This plan is a requirement for Minnesota to remain eligible to receive NPS grant funds from the US Environmental Protection Agency (US EPA) under Section 319 of the CWA. More information is available on the Nonpoint Source Management Program Plan page.
http://www.pca.state.mn.us/water/nonpoint/mplan.html

There are a number of similarities between the CWP and Section 319 programs, and since 2000, the application process for the programs has been administratively combined. Both programs address nonpoint sources of pollution, which are in aggregate by far the largest threat to water quality. Because the two programs are funded from two different sources, (Section 319 is federally funded and the CWP is funded with state money), there are some differences in how the grant money can be spent. For example, Section 319 funds cannot be spent on diagnostic work (other than TMDL, or total maximum daily load, development) and CWP grant money cannot be spent on in-lake treatment. In general, projects eligible to compete for available funds are those that address a nonpoint-source pollution issue. Also not eligible for Section 319 or CWP funds are projects under enforcement action, permitted wastewater treatment plants, and projects addressing feedlot NPDES or stormwater permit requirements. Over the last few years, some Section 319 funds have been dedicated to developing and implementing TMDLs (total maximum daily load; Section 303(d) of the Clean Water Act) and it appears likely that a majority of the future 319 and CWP funds will be used to address impaired waters.

In 1994, low-interest loans, made available through the State Revolving Fund (SRF), were added to the CWP program for Phase II implementation projects. These loan funds are administered through the CWP Program and are available for nonpoint-source pollution projects. The use of loans for funding Phase II projects allows for more effective leveraging of limited grant funds. Loans can be used for implementation measures and can cover the entire cost of the implementation phase or supplement a grant.

Another means to improve the management of Minnesota's lakes is a Memorandum of Agreement between MPCA, MDNR, MDH, Department of Agriculture, Board of Water and Soil Resources, Office of Strategic and Long Range Planning, and Metropolitan Council (2000). The goal of this agreement is to establish policies and administrative procedures that will provide for a more effective working relationship between the various agencies in achieving sound lake management. Nothing in the agreement alters the statutory authorities of the agencies or departments. Rather, it is intended to facilitate cooperative implementation of those statutory requirements and efforts. One feature of the agreement includes the establishment of a joint coordinating team called the Interagency Lake Coordination Committee (ILCC). The ILCC, consisting of one or two members from each department, is responsible for oversight and implementation of this agreement, including coordinating departments' activities, sharing lake information and numerous other lake coordination issues.

Monitoring and Education Programs

The MPCA has developed programs and a variety of materials pertaining to lake and watershed management. Some of these programs are valuable for both data gathering and education. Booklets and accompanying slide shows have been very useful for education. The following is a brief description of some of these programs and publications.

Citizen Lake-Monitoring Program (CLMP) was initiated in 1973 by the University of Minnesota Limnological Research Center and included 74 lakes. In 1979, administration of the program was transferred to the MPCA. In 1986, 252 lakes were monitored through this program and by 2003 the program had grown to include 928 volunteers on 909 lakes.

This program involves voluntary assistance from citizens residing on or near lakes. Participants take weekly transparency measurements of a lake during the summer months using a Secchi disk. Data from CLMP are extremely valuable to persons or groups interested in assessing and keeping records of the water quality of a lake over time. In many lakes, this represents the only monitoring data available. CLMP data are entered into STORET, U.S. EPA's national water quality data base, along with all other water quality data. This program is a very basic means for obtaining data, but more importantly is an excellent educational tool for helping citizens understand water quality interactions in lakes.

The program was expanded in 2001 to allow for the collection of water chemistry samples along with Secchi transparency on a limited number of lakes. CLMP Plus, as it is called, is conducted in cooperation with county water planners and lake association volunteers. To date, select lakes in Sherburne, Chisago, Crow Wing, Aitkin and Cook Counties have been assessed in this manner. Details on CLMP and CLMP Plus may be found on the MPCA Web site at: http://www.pca.state.mn.us/water/clmp.html .

Lake Assessment Program (LAP) was initiated in 1985 as a pilot program. During that summer, three lakes were included in the program. Since that time, over 150 lakes have been included in the program. Current work effort has ranged from 2 to 12 lakes per year. Application to this program has been rather informal with interested groups submitting a completed questionnaire, which briefly describes their organization and perceived water quality problems in the lake. Requirements for participation may change as program emphasis changes in the future.

This program takes the CLMP ideas one step further. LAP is a cooperative study of a lake involving MPCA staff and local citizens, e.g., lake association or municipality. LAP studies serve to characterize a lake's condition and provide some basic information regarding the interaction of the lake and its watershed. The format used in the LAP studies provides valuable information for the local unit, MPCA, and others interested in protecting or improving the quality of a lake. An individual report is generated for each LAP project. LAP studies will compliment the Clean Water Partnership Program and Clean Lakes Program in the future by providing information necessary for selecting projects and communicating ideas regarding lake protection and restoration. A more detailed explanation of LAP may be found in Heiskary (1989). Copies of completed LAP studies and other details on the program may be found at: http://www.pca.state.mn.us/water/lakequality.html .

In addition to these programs, MPCA has produced a number of educational publications and frequently participates in meetings and educational forums with group interested in protecting and improving lake water quality. Among the groups/organizations which MPCA routinely works with are lake associations and area wide associations, such as the Coalition of Lake Associations, watershed districts, Soil Conservation Service, and Soil and Water Conservation Districts and Minnesota Lakes Association. Some of the educational booklets MPCA has developed or collaborated on include:

- Citizens Guide to Lake Protection
- Protecting Minnesota Waters . . . The Land Use Connection
- 305(b) Report to Congress

- CLMP annual reports
- Minnesota Lake and Watershed Data Collection Manual
- Developing Lake Management Plans

These booklets and related publications have been very useful for training and educating a wide range of persons/organizations. Many of these publications may be found on our Web site at: http://www.pca.state.mn.us/water/lakequality.html .

In addition to MPCA and MDNR's activities in the area of lake management, a number of other agencies are involved. Included among these are counties (via local water plans), regional commissions, watershed districts, water management organizations and soil and water conservation districts. For example, numerous counties and the Metropolitan Council take an active role in the monitoring of lakes, actively participate in lake restoration projects and help educate lake users and local lake managers. The Metropolitan Council's Citizen Assisted Monitoring Program (CAMP), for example, is one of the larger volunteer monitoring programs in the state. This program not only generates huge amounts of data for lakes in the Metropolitan area but also provides a good basis for educating lake homeowners, associations, and water management organizations on lake water quality assessment.

V. DERIVING EUTROPHICATION CRITERIA

General Considerations

This chapter specifically addresses the setting of phosphorus, chlorophyll-a and Secchi criteria for lakes in each ecoregion. The procedures involved incorporate information from the previous discussions, e.g., lake trophic status. The approach described herein can apply as well where site-specific standards may need be developed. Individual case studies have been included for each ecoregion (Appendix IV) to provide examples of how criteria may be applied.

Minnesota's approach for developing in-lake nutrient criteria, relative to attainment of aquatic recreational use and protection of coldwater fisheries, was previously presented in earlier LWQA reports (e.g. Heiskary and Wilson, 1988) and related publications (e.g., Heiskary and Walker, 1988). In addition to the information that was originally compiled and considered, e.g. reference lake data (Table 2), user perceptions (Fig. 27), and fishery requirements (Table 5) we now have additional data sets that can be considered and include: our most recent 305(b) assessment (MPCA 2004), the various EPA criteria documents, pre-European diatom-inferred P concentrations (Heiskary and Swain, 2002), and our study of shallow southwest and west-central Minnesota lakes (Heiskary et al. 2003; Heiskary and Lindon, 2005). Data summaries from several of these data sets are compiled by ecoregion in Figures 29-32. In this fashion we can make comparisons among the various data sets, in addition to considering the interrelationships of TP, TN, chlorophyll-a and Secchi and many of the aforementioned factors (e.g., user perception). This will also allow for the selection of corresponding chlorophyll-a and Secchi thresholds (criteria) that are now required as a part of this effort.

Although we frequently view our lakes as having multiple "sub-uses" within the overall umbrella of the aquatic life and recreation use class, certain sub-uses may or may not be attainable in all lakes. Typically, the most sensitive of these sub-uses plays an important role in the selection of appropriate total phosphorus, chlorophyll-a and Secchi criteria, given that the sub-use is realistic considering the lake's morphometry and watershed characteristics. In this context, we have defined "most sensitive sub-use" (or uses) of a lake as that use which can be impacted or possibly lost as a result of a change (increase) in the trophic status of the lake.

The aquatic life and recreation use class (Class 2 waters) is defined in Minn. R. 7050.0200, subp. 3, and designated Class 2 subclasses are described in Minn. R. 7050.0222. The Class 2 subclasses are listed below. These subclasses correspond loosely to the sensitive sub-uses upon which the nutrient criteria are based.

- Class 2A – coldwater fishery, including lake trout lakes and stream trout lakes; and water recreation of all types including swimming. Trout lakes are individually listed in Minn. R. 7050.0470. In addition, lake trout lakes have all been designated as Outstanding Resource Value Waters that receive added nondegradation protection (Minn. R. 7050.0180).
- Class 2B – cool and warm water fishery, and water recreation of types including swimming (includes the vast majority of Minnesota lakes).
- Class 2C – indigenous aquatic community, and water recreation of all types where usable (could potentially include many shallow lakes).
- Class 2D – wetlands, nutrient criteria are under development for wetlands; they are not discussed in this report.

While the nutrient criteria discussed in this report are based on protecting sub-uses of the aquatic life and recreation use, some lakes are also protected as a source of drinking water. Protection of drinking water supply lakes from eutrophication may be as sensitive a use as certain sub-uses of aquatic life and recreation. For example, eutrophication can increase water treatment costs, contribute to taste and odor problems (Walker, 1985a), and increase production of trihalomethanes during the treatment process (Dorin, 1980; and Palmstrom, et al., 1988). If necessary, the MPCA can develop site-specific nutrient criteria for lakes protected for drinking in conjunction with the management authority for the particular source water and the MDH Source Water Protection program. For some communities, such as SPRWU, eutrophication concerns are already part of their overall management plan. The criteria and approaches described in this report should aid in that process.

The most sensitive sub-uses are noted for each ecoregion. Lakes corresponding to some of these categories have been specifically identified in Minnesota R. 7050.0470, subps. 1-8, and include designations for the following:

- Domestic consumption (as defined in Minn. R. 7050.0221).

- Fisheries and recreation (as defined in Minn. R. 7050.0222) whereby Class 2A specifically refers to waters designated for the propagation and maintenance of coldwater fish, with lake trout lakes specifically identified in Minn. R. 7050.0470 and included in Appendix III.

In addition to considerations of the most sensitive sub-use we have also come to recognize that primary contact recreation (includes swimming, diving, water skiing and other forms of recreation where immersion and the possibility of inadvertently ingesting water is likely) may not be the principal or primary use of some lakes. Very shallow and/or macrophyte-dominated lakes are good examples where the shallowness of the lake, highly organic substrates, and/or dominance of macrophytes may either prohibit or discourage primary contact recreation. While these lakes are used for "aquatic recreation" it is more often in the form of secondary body contact, e.g. boating, fishing or wading. In many of these types of lakes ecological endpoints may be a more important focus for criteria development and emphasis may be best placed on maintenance of adequate transparency or minimizing the magnitude and frequency of nuisance algal blooms to allow for the propagation of submergent and emergent macrophytes that in turn support fish and waterfowl production.

In summary, we propose nutrient criteria for the aforementioned "most sensitive sub-uses" for several lake types, including lake trout (coldwater fishery) and stream trout lakes (coolwater fishery), primary body contact recreation (deeper lakes), and a healthy aquatic plant and wildlife community (shallow lakes). These uses are summarized in the following table.

Subcategories of beneficial uses within the aquatic life and recreation use that the nutrient criteria are designed to protect.

Waterbody type	**Uses. The more "sensitive" use, which is the primary basis for the proposed standard, is listed as number 1. Other uses follow.**
Lake trout Lakes	1. Protection of sensitive aquatic community. Specifically, maintenance of adequate dissolved oxygen in hypolimnion needed to support lake trout 2. Water recreation of all types including swimming 3. Aesthetics
Stream trout lakes	1. Protection of sensitive aquatic community. Specifically, maintenance of adequate dissolved oxygen in metalimnion needed to support stream trout 2. Water recreation of all types including swimming 3. Aesthetics
Lakes and reservoirs > 15 feet deep	1. Water recreation of all types including swimming, at least part of the summer season 2. Maintenance of the desired game fishery 3. Aesthetics
Shallow lakes and reservoirs < 15 feet deep	1. Protection of aquatic community. Specifically the maintenance of a diverse community of emergent and submerged aquatic plants, and wildlife 2. Water recreation of all types including primary body contact where usable 3. Aesthetics

While our original effort (Heiskary and Wilson, 1988) focused solely on the development of phosphorus criteria, this current effort will consider the development of chlorophyll-a and Secchi criteria as well. This is a result of EPA's recommendations for nutrient criteria development (USEPA, 2000) and MPCA rulemaking for the "determination of impairment" for 305(b) reports and the 303(d) list (MPCA, 2003). A primary concept in that rulemaking was the identification of ecoregion-based thresholds for a causal factor (i.e., phosphorus) and response factors (chlorophyll-a and Secchi) (Table 19). These considerations will be employed in this effort as well. Our approach will involve the use of the previously identified data sets, consider interrelationships of the "causal" variables (TP) and the "response" variables (chlorophyll-a and Secchi), user perception responses, literature review and related information.

The following discussion seeks to compare various data sets, allows us to consider EPA's percentile approach for criteria setting (USEPA, 2000), make some linkages among the variables and allow for comparisons to our previously defined phosphorus criteria and current thresholds for listing of impaired lakes (Table 19). Datasets used in this analysis have been previously defined and include: Minnesota's ecoregion reference lake data set (originally assembled in 1985 – 1987); our most recent 305(b) assessment of trophic status data for Minnesota lakes (MPCA assessed lakes, 2004); and EPA's ambient water quality criteria recommendations for Aggregated Nutrient Ecoregions VI, VII, and VIII (EPA assessed lakes; USEPA, 2000). These aggregated ecoregions correspond as follows to Minnesota's seven ecoregions for criteria development purposes: a) Ecoregion VI includes the WCP, NGP, and Red River Valley (RRV); b) Ecoregion VII includes the CHF and Paleozoic Plateau (PP, formerly referred to as Driftless Area); and c) Ecoregion VIII includes the NLF and Northern Minnesota Wetlands (NMW). The RRV, NMW and PP ecoregions contain very few lakes as a result of the geomorphology of these regions and they were not included in MPCA's reference lake data set. Likewise, they are poorly represented in MPCA's and EPA's assessed data sets. Given the general lack of data, for lakes in these regions, criteria might best be set based on an adjacent region and/or established on an individual basis as needed. We have previously summarized assessed data for the NMW, PP, and RRV, in Table 1 and paired the ecoregions as noted above with the exception that we have included the few RRV lakes with those of the CHF. The following discussion assumes that criteria developed for the NLF would be applied to the NMW lakes and that CHF criteria would be applied to the RRV and PP lakes as well.

A comparison of the number of lakes in each database was made to provide some perspective for each (Table 18**).** The number of observations in MPCA's assessed database (MPCA, 2004) may be larger than EPA's because it considers data from 1970 through 2002, while EPA's data was compiled based on STORET and other data for the time period from 1990 through 1998. Also, EPA's database considers data from adjacent states that share the same ecoregion and as a result the EPA database may be larger than that for Minnesota alone. Values are expressed as summer-mean concentrations unless otherwise noted. EPA percentiles are based on all summer observations available for that ecoregion (Table 18).

Table 18. Comparison of number of lakes in MPCA reference, MPCA 305(b) assessed (2004), and EPA assessed (EPA, 2000) databases: by ecoregion and parameter.

Region (#)	Ref. All	Assess TP		Assess Chl-a		Assess Secchi	
	MPCA	MPCA	EPA	MPCA	EPA	MPCA	EPA
NLF (50)	30	863	406	521	128	1,394	581
CHF (51)	38	691	469	622	273	968	559
WCP (46)	12	89	92	79	57	109	108
NGP (47)	10	30	69	27	24	37	55

The MPCA assessment database (MPCA, 2004) for the NLF, CHF, and WCP ecoregions is generally larger than the EPA assessment database and as such, likely provides a better description of the population and the 25th percentile. The MPCA has TP data for 863 NLF lakes, which represents about 16 percent of the approximately 5,558 lakes in this ecoregion. This is higher than the CHF where we have TP data for 691 lakes, which represents about 15 percent of the approximately 4,765 lakes in the region. The percentages for the WCP are similar at 15 percent, however the NGP is poorly represented with about four percent of the lakes with P data. In the WCP and NGP reference lakes are an appreciable portion of the assessed lakes for each region. These two regions share many similarities in terms of lake morphometry and watershed landuse and were previously addressed together for purposes of developing ecoregion-based P criteria.

The EPA dataset also provides some data for the lake-poor ecoregions: RRV, NMW and DA ecoregions. For example, TP data are available for 11, 22 and 32 lakes respectively for the three ecoregions. For our assessment purposes (Table 1) we have included the NMW lakes with the NLF lakes given similarity in land use and lake trophic status (TP IQ range for NMW is 10-23 µg/L based on EPA data set). The RRV lakes have been included with the CHF lakes for our assessments, since many of the lakes we have assessed are very near the transition between the CHF and RRV ecoregions and some of the watersheds may actually be located in the CHF ecoregion. However, EPA has included this region in Aggregated Region VI with the NGP and WCP lakes. Based on the EPA data the TP IQ range is 85-295 µg/L, which is more consistent with the NGP ecoregion (Table 1); however chlorophyll-a at 4-23 µg/L and Secchi at 1.4 – 2.0 m are more consistent with CHF lakes. For now we will continue to use the CHF distributions as a basis for assessing the RRV lakes. EPA included the DA ecoregion with the CHF in the Aggregated ecoregion VII. There are very few lakes in this portion of Minnesota, absent Mississippi pools or backwaters or other pooled stream reaches and we have used the CHF as a basis for assessing lakes that are located in the DA ecoregion. However, as is the case with Lake Pepin, a run-of the-river reservoir on the Mississippi, we are inclined to establish site-specific water quality goals (e.g., Heiskary and Walker, 1988).

The following data summaries and discussions will focus on the interquartile (IQ) range -- values between the 25th to 75th percentiles (inclusive). This range is a good indicator of the central tendency of a population and we have previously used it as an indicator of the "typical range" of values for reference lakes (Table 2) and EPA focuses on the 25th and 75th percentiles for one criteria setting approach they promoted (USEPA, 2000). In addition we will draw upon the preceding analysis of data and the various interrelationships that were established. As a part of discussions for each ecoregion we will consider the most sensitive sub-uses of lakes in that region as a basis for developing ecoregion-specific criteria for TP, chlorophyll-a and Secchi.

The previous sections of this report, and studies that preceded it, provide the basis for establishing eutrophication criteria for Minnesota lakes. In the following section we summarize much of this information and data by ecoregion and use these multiple sources of information to propose likely ranges for the criteria values. While there is no exact formula to this approach the following considerations and relevant data were considered prominently in proposing criteria ranges and ultimately in developing the criteria:

- The typical (IQ range) trophic status range of <u>reference</u> lakes represents a starting point and we tend to focus first on TP. Emphasis is placed on the 75th percentile of the reference population; however this is not a sole consideration and the focus may vary among regions.

- Next we consider the IQ range for the MPCA and EPA assessed populations and how they relate to the reference population. With these populations we may focus more on the 25^{th} and 50^{th} percentiles.
- Pre-European (diatom-inferred) TP concentrations are brought into this assessment to provide perspective on background condition for each ecoregion. Here emphasis is placed on the 75^{th} percentile and acknowledging that, in some instances, these data sets are rather small.
- The analysis is extended to chlorophyll-a and Secchi and we pursue similar comparisons among the reference and assessed data sets. In addition we make linkages among TP, chlorophyll-a, and Secchi based on regressions developed for Carlson's TSI (Carlson, 1977) and/or those developed from the reference lakes (e.g. Eq. 1-3). In general, there is not a great deal of difference among the Carlson and Minnesota-based regression equations for oligotrophic and mesotrophic lakes; however we do note that the Carlson TSI equations tend to overestimate chlorophyll-a for hypereutrophic lakes (assumes a more linear relationship). In addition we have made further linkages to nuisance bloom frequency as it relates to summer-mean chlorophyll-a (Fig. 5b) and TP (Fig. 6a &b) and this figures prominently as well.
- User perceptions relative to Secchi transparency and chlorophyll-a help provide meaning to Secchi and chlorophyll-a data. Our focus here is on minimizing the occurrence of conditions characterized as "swimming impaired" or "high algae" in the NLF and CHF ecoregions; whereas in the WCP and NGP we seek to minimize the occurrence of conditions characterized as "no swimming" or "severe algal blooms." Again as these thresholds are identified we can use the interrelationships of TP, chlorophyll-a and Secchi to further refine proposed criteria ranges.
- In the case of coldwater fisheries, where protection of lake trout is of primary concern, we emphasize interrelationships among TP, chlorophyll-a, Secchi and hypolimnetic oxygen depletion based on the literature and data from assessed lake trout lakes. For coolwater fisheries, where maintenance of stream trout fisheries is an emphasis, HOD is not as significant of a concern; however there is a need for well-oxygenated cooler water in the metalimnion to ensure trout are not stressed and forced into the warmer epilimnetic waters. As for other fish populations we use established relationships among lake trophic status (TSI) and where we see species shifts as a further basis for establishing criteria ranges (Fig. 13).
- Lake morphometry and mixing can influence lake trophic status. For Minnesota the difference between shallow and deep lakes is most pronounced in the CHF and WCP ecoregions. This influence, combined with differing uses of shallow lakes (e.g. swimming is often not a primary use), prompted us to consider linkages among the trophic status variables and rooted macrophytes as a basis for recommending criteria ranges (Fig. 26).

a. Northern Lakes and Forests

In the NLF ecoregion most sensitive sub-uses that will be addressed in our development of nutrient criteria include: coldwater fisheries (lake trout), coolwater fisheries (stream trout), and primary contact. Shallow lakes were not considered as a separate category for this ecoregion as there was minimal difference among the trophic status (based on TP) of deep versus shallow lakes in this ecoregion (Table 4). Also, Secchi transparency for NLF lakes is typically 1.8 m or greater (75th percentile) and chlorophyll-a is typically less than 8 µg/L based on the assessed lakes (Table 1), which should allow adequate light for rooted plant growth over much of the littoral zone. Our background information for lake trout and stream trout fisheries was previously addressed. Data summaries for lake and stream trout (Tables 6 & 7), combined with established interrelationships with nutrients, DO, HOD (Fig. 21), and overall environmental requirements (Table 5) of these fisheries served to guide selection of criteria values.

Summer-mean TP values are quite similar among the NLF reference, MPCA assessed and EPA assessed databases (Fig. 29). Further, these ranges compare favorably with diatom-inferred pre-European TP concentrations. Based on a comparison of the reference and EPA or MPCA assessment data it is evident that the 25th percentile would not be a reasonable concentration for establishing criteria. At this level a vast majority of the reference lakes would be in excess of the criterion level, as would 75 percent of the lakes in this predominantly forested ecoregion – for which there would be no reasonable management alternative for achieving a criteria value this low (~10 – 13 µg/L) in a majority of the lakes. Further, based on diatom reconstructions of 20 NLF lakes many lakes in this ecoregion may have been above this concentration range in pre-European times (Table 17 and Fig. 29). A TP concentration of 30 µg/L corresponds to the 75th percentile for the assessed databases (Fig. 29). This concentration is only slightly above the pre-European and reference lake 75th percentiles and would seem to be a reasonable level for criteria establishment given the predominantly forest and wetland land uses that are characteristic of most of the watersheds in this region. From a fisheries standpoint TP concentrations of 30 µg/L or less would seem to be favorable for piscivore dominance and tend to favor walleye, northern pike and perch; in contrast to common carp and bullhead species (Fig. 13). Further, a concentration of 30 g/L is often used as a boundary between mesotrophic and eutrophic conditions (Nurnberg, 1996).

The IQ range for chlorophyll-a in the NLF is relatively small and similar among the reference and assessed lakes (Fig. 29). Based on previous work chlorophyll-a greater than ~10 µg/L is perceived as a bloom and greater than ~20 µg/L nuisance to severe nuisance bloom in this region. At a summer-mean of 10 µg/L maximum chlorophyll-a concentrations should remain below 20 µg/L (Figure 5a, Eq. 4) with about 40 % of summer exhibiting chlorophyll-a of 10 µg/L or greater (Figure 5b). A reasonable goal for the NLF would be to minimize events of 10 µg/L or greater, which corresponds to a P concentration of 30 µg/L or less (Figure 6a). This translates to a summer-mean chlorophyll-a in the 8 – 10 µg/L range, which would be near the 75th percentile for the reference database and just above the 75th percentile for MPCA and EPA assessment databases. Also, a concentration of 9 µg/L is often used as a boundary between mesotrophic and eutrophic conditions (Nurnberg, 1996).

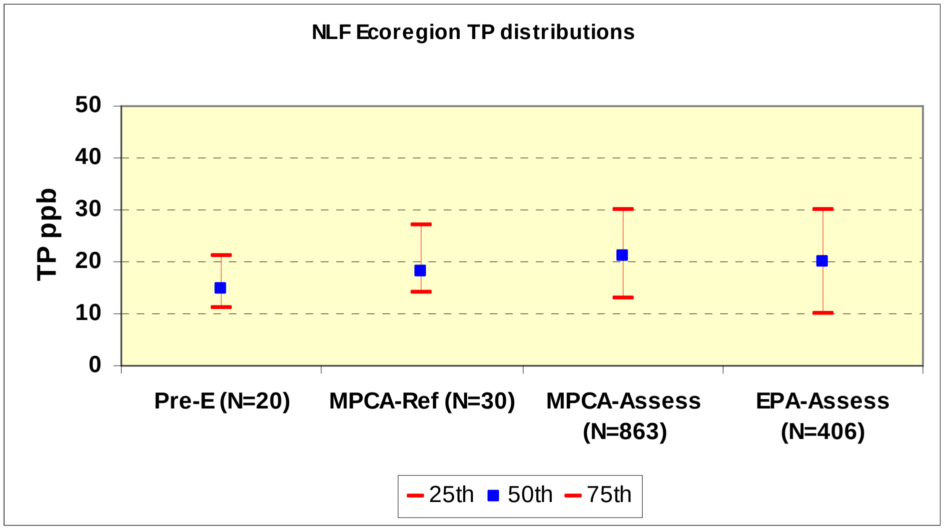
Extensive work has been done examining user perception relative to Secchi transparency for Minnesota (Heiskary and Wilson, 1989; Heiskary and Walker, 1988; and Smeltzer and Heiskary, 1990). This work demonstrates distinct linkages and regional patterns in perception. In the NLF user perception responses of “definite algal green “ or “use slightly impaired” are associated with Secchi readings of about 2.0 - 2.2 m on average for assessed observers (Table 10). A response of “no swimming or high/severe algae” corresponds to Secchi of about 1.5 m or less (Fig. 27). Given user perception, a reasonable goal would be to keep Secchi readings above 2.0 m for a majority of the summer and avoid occurrence of Secchi < 1.5 m. Achieving the latter would require a summer-mean Secchi on the order of 2.0 m or greater which is near the 75th percentile for all assessed and reference lakes (Fig. 29). A mean Secchi of 2.0 m would correspond to a P concentration on the order of 25 µg/L and chlorophyll-a of 7 µg/L (Figure 4b and Eq. 2 & 3). As noted previously these concentrations are near the 75th percentile for the reference lakes. A summer-mean P concentration of 30 µg/L would result in about 25 percent of the summer with Secchi readings of 2.0 m or lower (slight impairment). Based on the IQ ranges of the databases and user perception information a Secchi criteria value in the 1.8 – 2.2 m range may be appropriate.

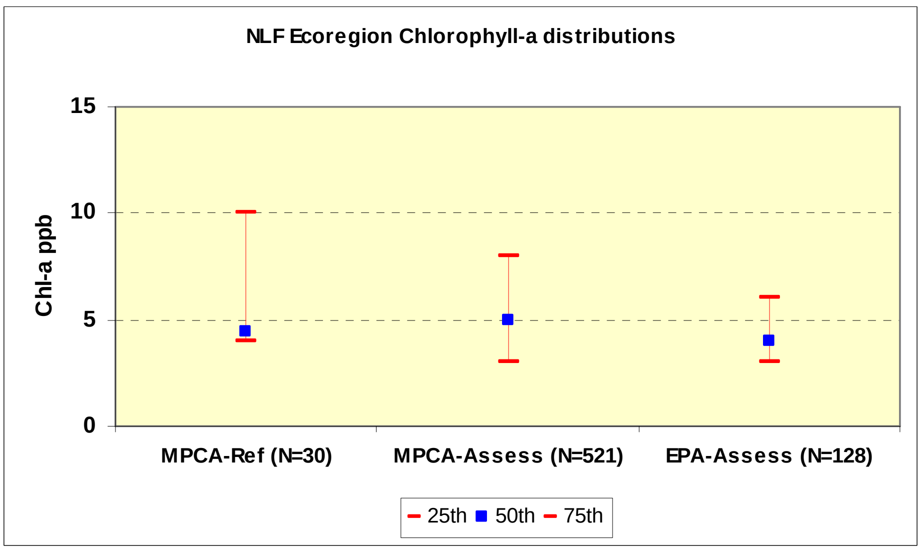
In summary we would propose the following ranges for criteria development:

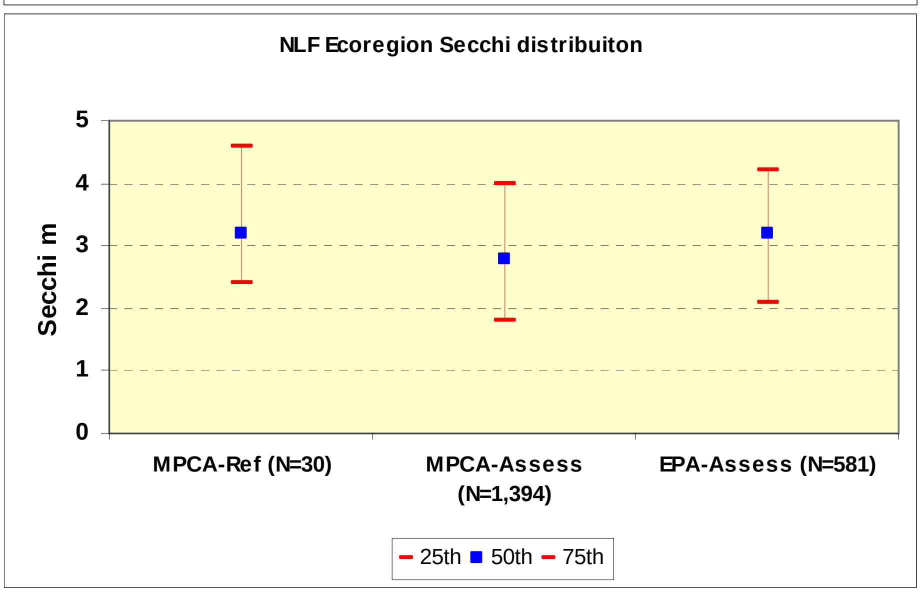
Sub-use	TP µg/L	Chl-a µg/L	Secchi meters
Lake trout[1]	10 - 15	2 - 4	4.0 – 6.0
Stream trout[2]	10 - 25	2 - 6	3.0 – 5.0
Aquatic recreation	20 - 30	5 - 10	1.8 – 2.2

1. see pages 56-60, Table 6, Figures 17-23 and Case Study 5.
2. see pages 69-70 and Table 7

Figure 29. NLF Ecoregion Summer-mean IQ ranges for TP, chlorophyll-a and Secchi







b. North Central Hardwood Forests

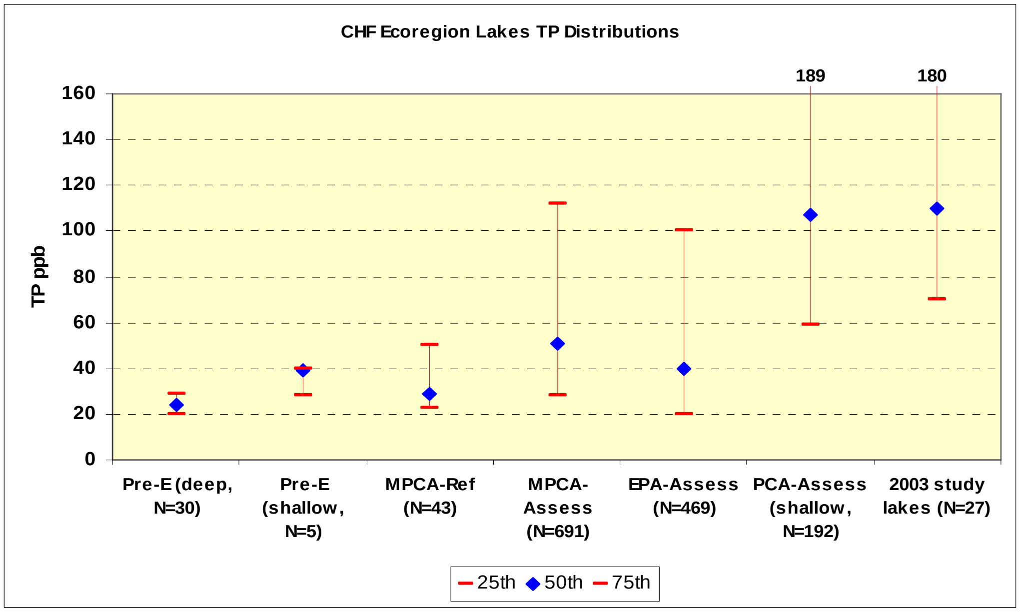
In the CHF ecoregion most sensitive sub-uses addressed in our development of nutrient criteria include: stream trout lakes, aquatic recreational use, and shallow lakes. This ecoregion contains no natural lake trout lakes and very few stream trout lakes. The previous analysis and criteria recommendations for NLF stream trout lakes applies to the CHF ecoregion as well and no further discussion is offered on this topic. This ecoregion does, however, have a large number of shallow lakes that may represent on the order of 25% to 40% of the lakes in this region based on estimates offered in Tables 1 and 5. As previously discussed, shallow lakes have their own specific management challenges. These challenges, on one hand, stem from the multiple and possibly unattainable uses the lakes are protected for. However, the challenges also stem from the limited capacity of the lake to assimilate excess nutrients because of shallowness, potential for internal recycling from several sources, and their tendency for "alternate states" – potentially from macrophyte-dominated to algal dominated and the inherent problems with either extreme.

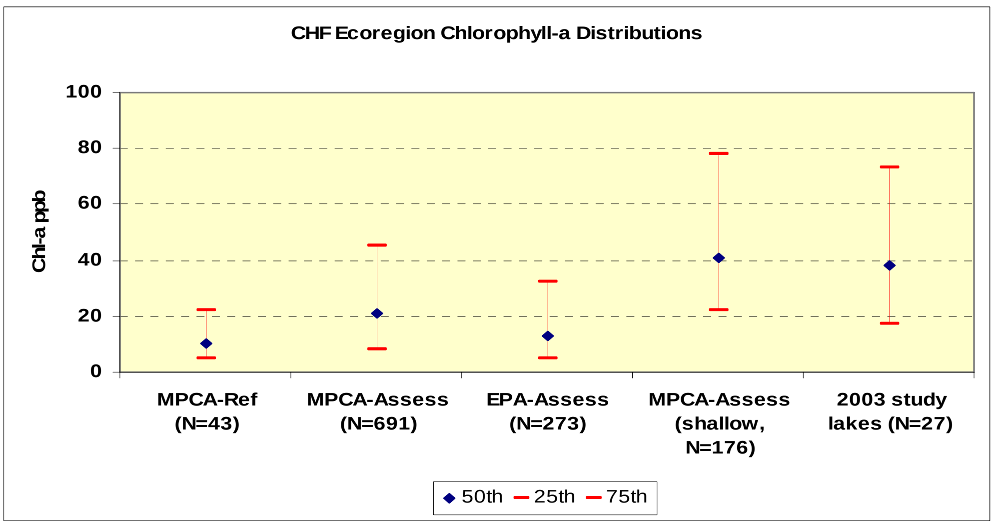
CHF – Aquatic recreational use

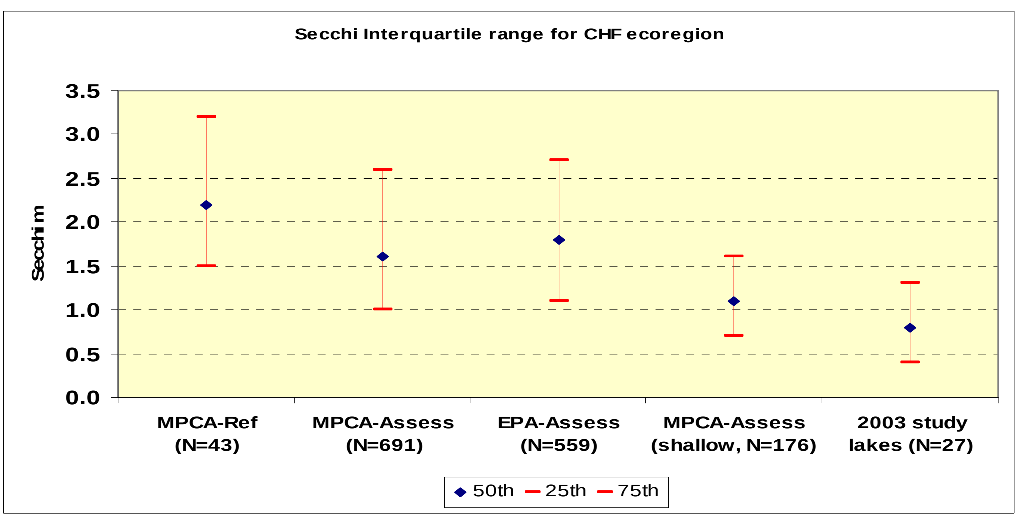
To aid the derivation of nutrient criteria for aquatic recreational use support various distributions (Figure 30) were reviewed – with a primary focus on the "deep" lakes. For TP the MPCA and EPA assessment databases exhibit somewhat similar IQ values (Fig. 30) with an overall range (75^{th}- 25^{th} percentiles) of 84 and 80 µg/L, respectively. This large range is a function of the heterogeneity of the lakes in this region (e.g., morphometry, watershed characteristics and anthropogenic impacts). The reference database exhibits a much smaller IQ range (27 µg/L) than the assessed database. Comparison of the 25^{th} percentile for the MPCA and EPA assessed and the 75^{th} percentile for the reference lakes suggests that an appropriate P criteria level may lie between 20 - 28 µg/L (25^{th} percentile assessed lakes) and 50 µg/L (75^{th} percentile reference lakes; Fig. 30). Also the 25^{th} percentiles for the reference and assessed populations correspond to about the 75^{th} percentile for the "deep" pre-European diatom-inferred P concentrations. Also as TP increases above about 40 – 50 µg/L (TSI's of about 55-60) the number of piscivores start to decline and relative abundance of carp and black bullhead tend to increase (Fig. 13).

For chlorophyll-a the IQ and overall range of the EPA assessment database is more similar to MPCA's reference lakes than MPCA's assessment database (Fig. 30). Again, as with TP, the overall range of values in the CHF is larger than the NLF. The EPA database exhibits a range of 27 µg/L in the CHF as compared to < 4 µg/L in the NLF and in the MPCA assessed lakes the range is 37 µg/L in the CHF and 5 µg/L in the NLF. If we consider values from the 75^{th} percentile (reference) and 25^{th} percentile (assessed) we have a potential range for criteria setting from 5 – 22 µg/L. Based on CHF user perceptions chlorophyll-a greater than ~20 µg/L is typically perceived as a nuisance bloom and greater than ~30 µg/L as a severe nuisance bloom. Here, a reasonable goal would be to minimize events of 20 µg/L or greater. At a summer-mean of 15 µg/L 70 % of summer chlorophyll-a is greater than 10 µg/L, 20 % greater than 20 µg/L (Figure 5b) and maximum chlorophyll-a should remain below 25 µg/L (Figure 5a). While summer-mean chlorophyll-a near the MPCA and EPA assessed 25^{th} percentile values (7 and 5 µg/L, respectively) would keep these events at five percent or less of the summer (Figure 5b) this concentration range is quite low and corresponds to the 25^{th} percentile for the reference database.

Figure 30. CHF summer-mean IQ ranges for TP, chlorophyll-a and Secchi







A more reasonable concentration range, considering both user perception information and distribution of chlorophyll-a concentration for this region, would be in the 12 - 15 µg/L range. This concentration range falls near the median of the three databases and corresponds to a summer-mean P of 35 to 40 µg/L (Figure 4b).

For Secchi, the EPA and MPCA assessment databases for the CHF are quite similar (Fig. 30). All three databases exhibit a relatively small IQ range of 1.7 m. 75th percentile values range from 1.0 – 1.5 m in the three databases. Fifty percent of volunteer observers in the CHF associate responses of "definite algal green" or "use slightly impaired" with Secchi readings of 1.3 m or less. Responses of "no swimming" correspond to Secchi readings of about 1.0 m or less (Table 10). A reasonable goal in this region would be to minimize the frequency of Secchi < 1.5 m and avoid occurrence of Secchi < 1.3. A summer-mean P concentration of 40 µg/L would yield Secchi readings less than 1.0 m about five percent of the summer and less than 2.0 m about 50 percent of the summer (Heiskary and Walker, 1988). Considering user perception information and 75th percentile values for the CHF ecoregion, a summer-mean Secchi of 1.2 - 1.5 m may be appropriate, which would correspond to a summer-mean P of about 35 - 40 µg/L (Figure 4a & b). This range of Secchi values would fall between the median and 75th percentiles for the assessment databases.

CHF - Shallow lakes

Our primary focus in setting eutrophication criteria for shallow lakes is to allow for a healthy and diverse population of macrophytes and to minimize the chance for a shift to algal-dominated conditions. As such, maintaining adequate transparency to allow plants to establish themselves over much of the basin, minimizing the occurrence of nuisance algal blooms, and keeping TP concentrations below a range that promotes excessive algal growth are all important considerations upon which to base eutrophication criteria. And of these three variables (parameters), transparency may be the most important. In turn transparency can be directly related to TP and chlorophyll-a, though several biotic factors, such as dominance of benthivorous (e.g., carp and bullhead) or planktivorous fish, and abiotic factors such as suspended sediments, lake depth, wind erosion and resuspension may also influence transparency and the ability of the lake to support macrophytes.

Distributions of TP, chlorophyll-a and Secchi for shallow lakes based on a subset of the assessed population and the west central study lakes help offer some perspective (Fig. 30). For TP we have comparisons for pre-European (diatom-inferred) TP and modern-day distributions for all assessed CHF lakes and a subset of that "shallow lakes" as bases for comparison (Fig. 30). For this exercise the shallow lakes distribution represents all assessed CHF lakes with a maximum depth of less than 20 feet, which should include most of the lakes that meet our definition of shallow lakes – "maximum depth of 15 feet or less or 80% or more of the lake is littoral." As noted previously the shallow, polymictic lakes of the CHF ecoregion tend to be more nutrient-rich as compared to the overall population (Fig. 30) and in particular when compared to deeper, dimictic lakes of the same ecoregion (Fig. 24). For example the 25th percentile for the shallow lakes is at about 60 ppb, which is slightly above the median for the MPCA assessed lakes (Fig. 30). The IQ range for the 2003 shallow west-central lakes (Heiskary and Lindon, 2005) is fairly similar to the TP range for shallow lakes and would appear to be a reasonable representation of the range in TP found in shallow CHF lakes.

As with TP the shallow lakes exhibit higher chlorophyll-a as compared to the overall MPCA assessed lakes. In this case the 25th percentile for the shallow lakes – 22 µg/L is near the median for the MPCA assessed lakes. Again the IQ range for the 2003 study lakes is quite similar to the assessed shallow lakes (Fig. 30). As anticipated, Secchi transparency for the assessed shallow lakes is less than the overall assessed lakes with a median value of 1.1 m corresponding to the 75th percentile for the assessed population (Fig. 30), i.e., 75 % of the assessed lakes have transparencies greater than 1.1 m. The 2003 study lakes exhibited a similar but slightly lower range of Secchi as compared to the assessed shallow lakes.

Based on Table 20 and Figure 26, transparency should remain above about 0.7 m and ideally 1.0 m or more to minimize the likelihood of low FQI and a reduced number of rooted plant species. Relative to the assessed shallow CHF lakes 0.7 m and 1.0 m correspond to about the 25th and 50th percentiles respectively (Fig. 30). A summer average transparency of 0.7 – 1.0 m should allow for SAV colonization to a depth of about 1.5 – 2.0 m (~5 - 6 ft.) based on equations developed by Canfield et al. (1985) and Chambers and Kalff (1985) and data from the west-central lakes (Heiskary and Lindon, 2005). This would represent an appreciable portion of the lake-basins such as those included in the west-central shallow lakes study where mean depths were typically 1 – 2 m (3- 7 ft.) (Heiskary and Lindon, 2005).

Lake user perception can provide some perspective as well in determining an appropriate level of transparency – even though swimming may not be the primary use of shallow lakes. Based on user perception for CHF lakes a transparency of 0.7 m represents the average transparency associated with "severe nuisance blooms" and/or "swimming and aesthetic enjoyment nearly impossible." Whereas 1.0 m was the average transparency associated with "high algal levels" and/or "desire to swim reduced because of algae levels (Smeltzer and Heiskary, 1990). This would suggest a transparency closer to 1.0 m may be more desirable based on potential lake users.

Chlorophyll-a is the next consideration and based on Carlson's TSI (Fig. 3) and interrelationships developed in this study corresponding chlorophyll-a concentrations would be on the order of 20 (Secchi = 1 m) to 30 µg/L (Secchi = 0.7 m). However based on a desire to minimize nuisance blooms a concentration closer to 20 µg/L would be more desirable since the frequency of nuisance blooms (chlorophyll-a > 30 µg/L) increases from about 15% at 20 µg/L up to about 45% at a chlorophyll-a of 30 µg/L (Fig. 5b) and would likely lead to an algal-dominated system. Also the average chlorophyll-a associated with high FQI was 19 µg/L (Table 19). As a frame of reference, a chlorophyll-a concentration of about 20 µg/L ranks near the 25th percentile for shallow CHF lakes (Fig. 30).

A corresponding range of TP concentrations to yield a transparency of 0.7 - 1.0 m would be on the order of 48-68 µg/L based on Carlson's TSI (Fig. 3) and about 60-80 µg/L based on Fig. 4b. TP concentrations greater than about 60-80 µg/L would be undesirable since the frequency of nuisance blooms increases substantially (Fig. 6b) and the number of rooted plants declines (Fig. 26) and with perhaps a few notable exceptions signals a shift to algal-dominated systems. And as noted earlier the average TP associated with high FQI or lakes supporting 15 or more species was 47 - 49 µg/L (Table 19). Lake response to increased TP over the range from 60-90 µg/L is rather variable for these shallow lakes in terms of Secchi and chlorophyll-a and the number of rooted plants (Figs. 6b & 26); however the general pattern is toward increased chlorophyll-a, more frequent nuisance blooms, and declining numbers of rooted plants. As a frame of reference a TP of 60 µg/L ranks near the 25th percentile based on assessed shallow lakes (Fig. 30).

Table 19. Shallow lakes summary of morphometric, water quality and plant metrics (Floristic Quality Index and number of plant species). Based on 27 west-central Minnesota lakes (Heiskary and Lindon, 2005)

FQI / # plants	Depth →	feet	Area acres	TP ppb	Chl-a ppb	Secchi m	TSS ppm
FQI	**Mean**	**Max.**	**Mean**	**mean / median**	**mean / median**	**mean / median**	mean / median
High	11	23	792	49 / 34	19 / 10	2.0 / 1.9	8 / 3
Medium	5	9	555	142 / 130	54 / 48	0.7 / 0.7	33 / 15
Low	5	8	502	194 / 210	74 / 45	0.5 / 0.4	43 / 38
# of plant species							
> 15 sp.	10	20	708	47 / 38	17 / 10	1.9 / 1.7	8 / 3
< 15 sp.	5	8	548	171 / 155	65 / 48	0.6 / 0.5	39 / 32

In summary, based on the reference and assessed lakes, various interrelationships among trophic status variables, user perceptions, fishery considerations, rooted plant metrics and other considerations it appears that appropriate ranges for selecting eutrophication criteria values for protection of aquatic recreation uses in deeper CHF lakes and emphasizing ecological considerations in shallow lakes in the CHF ecoregion are as follows:

Sub-use	TP µg/L	Chl-a µg/L	Secchi meters
Aquatic recreation Deep lakes	30 - 50	12 - 15	1.2 – 1.5
Shallow lakes	60 – 80	20 - 30	0.7 – 1.0

Given this range of values for shallow lakes, and acknowledging that other biotic and abiotic factors can be very significant in determining whether a lake can support a healthy and diverse population of rooted macrophytes, we are inclined to recommend criteria be set at the lower end of each range of the aforementioned values, i.e., maintain summer average Secchi of 1.0 m or greater, summer average chlorophyll-a of 20 µg/L or lower, and summer average total phosphorus of 60 µg/L or lower. Maintaining values is this range will not absolutely ensure that a shallow lake will remain in a plant-dominated state but should reduce the likelihood that the lake will switch to an algal-dominated state, which as repeatedly noted in the literature can be rather hard to reverse once the change has occurred.

c. Western Corn Belt Plains and Northern Glaciated Plains

Because of similarities in land use (Table 11), lake morphometry (Table 1), water quality (Table 2), and user perceptions (Figure 27) we have tended to lump these two regions together for the purpose of nutrient criteria development. USEPA (2000) shares a similar view since the WCP and NGP are characterized by relatively small databases and are in the same EPA aggregated ecoregion (USEPA, 2000). For these two regions we will consider criteria for "aquatic recreation use support" (intended for the few "deep" lakes in the WCP) and "shallow lakes" criteria for the majority of the lakes in the two regions.

The IQ ranges for the EPA and MPCA assessment databases are somewhat similar and exhibit an overall range on the order of 95 – 135 µg/L, respectively for the WCP (Fig. 31) and 210 - 90 µg/L, respectively for the NGP (Fig. 32). The "reference" lakes exhibit slightly lower IQ ranges but the overall range remains high -- on the order of 85 (WCP) and 44 (NGP) µg/L. Use of the 75^{th} (reference) and 25^{th} percentiles (assessed) would place potential criteria in a range from 60-150 µg/L in the WCP and 80 - 160 µg/L in the NGP. Based on previous assessments we estimate that perhaps 10% or less of the lakes in the WCP might be considered dimictic (deep) and these lakes will tend to have lower TP concentrations (Table 5) as compared to the typical polymictic (shallow) lakes that are characteristic of the region (Table 1). All lakes in the reference and assessment databases for these two regions would be considered eutrophic to hypereutrophic. We previously described "partial support of aquatic recreational use" as a reasonable goal for many of these shallow lakes, with a focus on reducing the severity and frequency of nuisance algal blooms (Heiskary and Wilson, 1989). Mean transparency associated with "impaired" and "no swimming" is 0.6 m in these two regions (Table 10). A concentration range of 70 – 90 µg/L P should yield transparencies greater than 0.5 m over 90 % of summer." As P increases above 80 – 90 µg/L TN:TP declines below 29:1 and blue-green algae may dominate (Smith 1983) and above 100 µg/L N limitation is increasingly likely (Fig. 7). These observations, combined with the 25^{th} percentile for the MPCA assessed data for the WCP (99 µg/L) and NGP (104 µg/L) (Table 18), suggest that a P concentration in the 70 – 90 µg/L range may be an appropriate criteria level for most lakes in these two regions. Also this would be just below the ~ 100 µg/L boundary between eutrophic to hypereutrophic lakes (Nurnberg, 1996). Further, as TSI (based on TP) increases above 70 (~100 µg/L) a decline in the number of fish species, percent piscivores (declines in walleye, northern pike and largemouth bass) and increases in the abundance of carp and black bullhead are evident (Figure 13).

Diatom-inferred pre-European P concentrations for five "deep" WCP and six "shallow" NGP/WCP lakes provide further perspective (Fig. 28). Based on the five deep WCP lakes median pre-European P is near 50 µg/L and the IQ range is from 38-56 µg/L. In comparison, median pre-European P for shallow lakes is 69 µg/L with a IQ range from 50 - 89 µg/L. This suggests that along with the distribution of modern-day P concentrations (Tables 1 and 5) it may be appropriate to set concentrations for deep lakes at a slightly lower level.

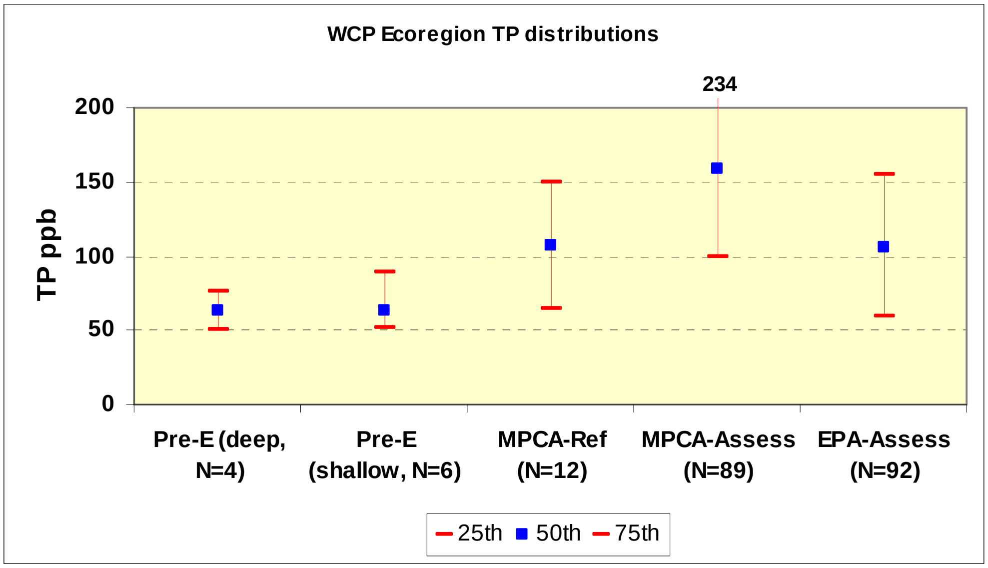
Chlorophyll-a is high in the WCP and NGP based on the MPCA reference and assessed databases (Figs. 31 & 32). The EPA database exhibits slightly lower values but a similar range (~30 µg/L). In the NGP the MPCA reference and assessed ranges are similar but the EPA values are slightly lower and exhibit a smaller range (Fig. 32). Based on user perceptions chlorophyll-a greater than 30 µg/L perceived as a nuisance bloom and greater than 60 µg/L a very severe nuisance bloom. At a summer-mean of 25 µg/L (near MPCA assessed 25th percentile) chlorophyll-a is greater than 20 µg/L about 60 percent, greater than 30 µg/L about 25 percent, and greater than 60 µg/L about five percent of the summer. However, a concentration of 25 µg/L is very low for the WCP or NGP ecoregion (Table 1) and would correspond to TP concentrations in the 60 – 70 µg/L range, near the 25th percentile for WCP reference lakes (Fig. 31). Given the distribution of TP and chlorophyll-a values for the WCP and NGP ecoregions a slightly higher chlorophyll-a concentration (on order of 30 – 35 µg/L) could be considered. However, as the mean chlorophyll-a concentrations rises the frequency and intensity of algal blooms and maximum chlorophyll-a increases as well (Fig. 5a & b), which should be taken into account as well. Chlorophyll-a concentrations of 30 – 35 µg/L correspond to TP concentrations on the order of 90 - 100 µg/L, which for lakes in the NGP and WCP is still rather low.

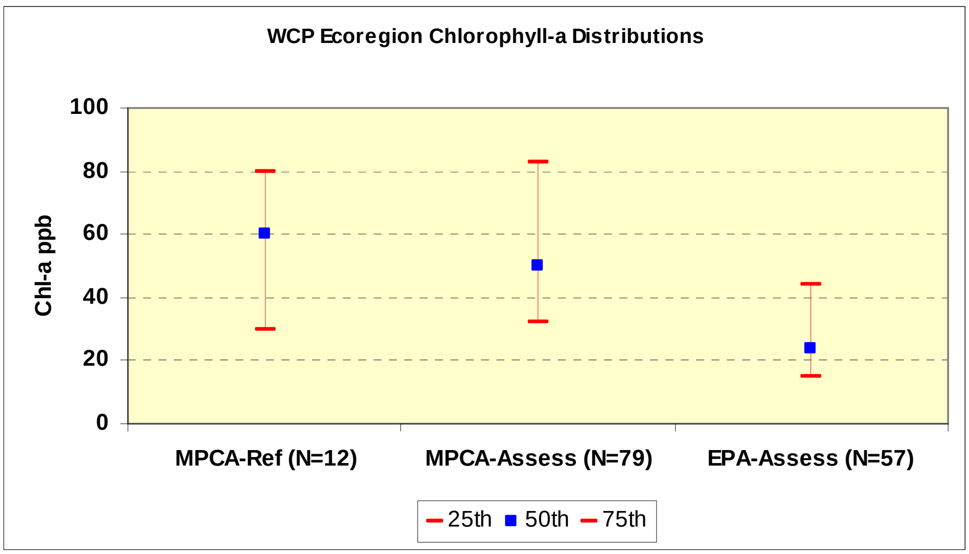
Secchi readings are quite low in most WCP and NGP lakes, typically ranging between about 0.5 – 1.2 m (Fig. 31 & 32). The reference lakes exhibit a lower 25th percentile value than the assessment databases, however the reference population is rather small. In these regions user responses of "definite algal green " or "use slightly impaired" correspond to Secchi readings of 0.8 m or less for 50 percent of observers (Table 10). Responses of "no swimming" correspond to Secchi readings of about 0.7 m or lower (Fig. 21). A reasonable goal in these regions may be to minimize frequency of Secchi below about 0.7 - 0.8 m. Maintaining a summer-mean transparency of 0.7 – 0.8 m requires a P concentration on the order of 65 – 70 µg/L, which may be too low for many lakes in these two regions, but may be a reasonable range for deeper lakes (Table 4). In contrast, summer-means in the 0.5 – 0.6 m range would correspond to summer-mean P concentrations of 70 – 90 µg/L which may be more reasonable considering modern-day and pre-European P distributions for these two regions (Fig. 31 & 32). A summer-mean P concentration of 90 µg/L should keep Secchi above 0.5 m over 90 percent of the summer and over 1.0 m about 60 percent of the summer (Heiskary and Walker, 1988).

In summary, based on the reference and assessed lakes, various interrelationships among trophic status variables, user perceptions, fishery considerations, rooted plant metrics (as demonstrated in the shallow west-central CHF lakes) and other considerations it appears that appropriate ranges for selecting eutrophication criteria values for protection of aquatic recreation uses in deeper WCP lakes and emphasizing ecological considerations in shallow lakes that predominate in the WCP and NGP ecoregions are as follows:

Sub-use	TP µg/L	Chl-a µg/L	Secchi meters
Aquatic recreation Deep lakes	50 - 70	20 - 25	0.7 – 1.2
Shallow lakes	70 - 90	25 - 30	0.7 – 1.0

Figure 31. WCP Ecoregion Summer-mean IQ ranges for TP, chlorophyll-a and Secchi





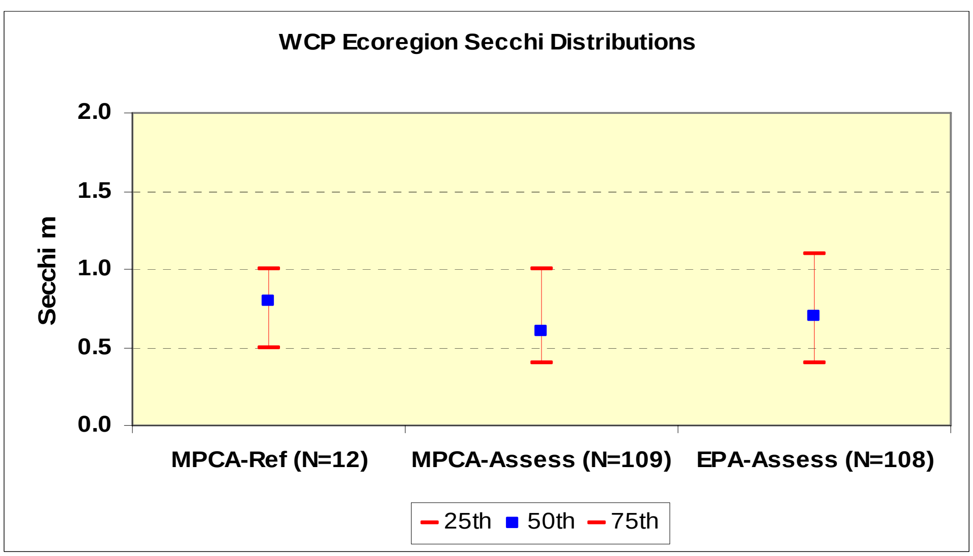
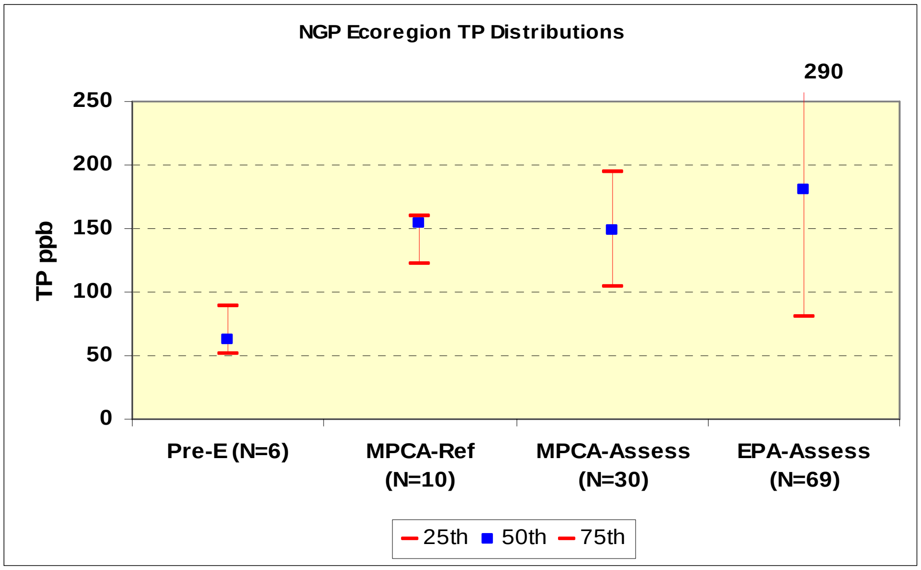
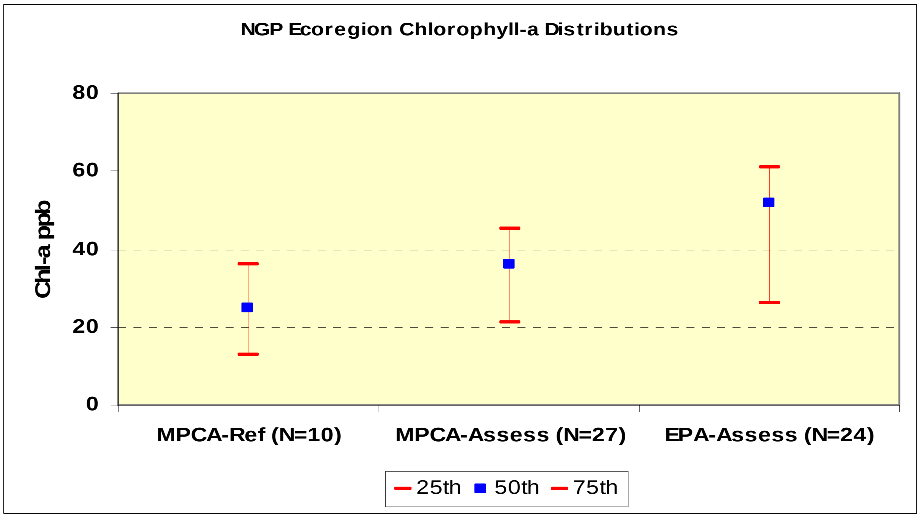
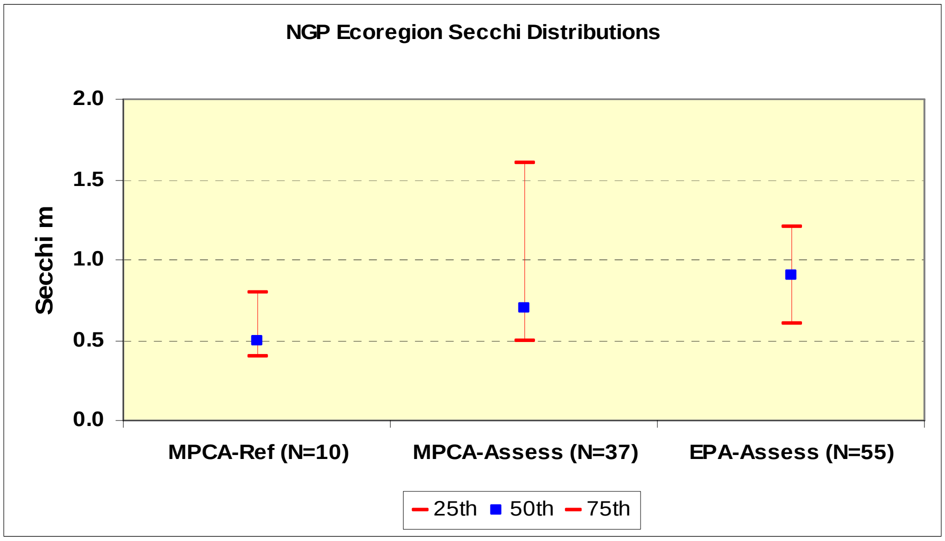


Figure 32. NGP Ecoregion Summer-mean IQ ranges for TP, chlorophyll-a and Secchi







Eutrophication Criteria: Summary by Ecoregion

The previous sections of this report, and studies that preceded it, provide the basis for establishing eutrophication criteria for Minnesota lakes. The following considerations and relevant data were considered prominently in developing the criteria (as noted in the preceding section):

- The typical trophic status of reference lakes and IQ range for the MPCA and EPA assessed lakes;
- Pre-European (i.e. background) TP concentrations;
- Interrelationships among TP, chlorophyll-a, Secchi, and nuisance bloom frequency (with an emphasis on MPCA-derived regressions, e.g., Eq. 1 & 2);
- The influence of lake mixing status (morphometry) on trophic status and differences among deep and shallow lakes;
- User perceptions relative to trophic status indicators;
- Most sensitive sub-uses of lakes including fishery concerns, aquatic recreation use support, secondary contact, and support for wildlife uses were considered;
- Review of the literature relative to these topics and nutrient criteria development in general.

Based on these considerations, previous data analysis and discussions we propose the following set of ecoregion-specific criteria for Minnesota's lakes (Table 20).

Table 20. Proposed eutrophication criteria by ecoregion and lake type.

Ecoregion	TP	Chl-a	Secchi
	ppb	ppb	meters
NLF – Lake trout (Class 2A)	< 12	< 3	> 4.8
NLF – Stream trout (Class 2A)	< 20	< 6	> 2.5
NLF – Aquatic Rec. Use (Class 2B)	< 30	< 9	> 2.0
CHF – Stream trout (Class 2a)	< 20	< 6	> 2.5
CHF – Aquatic Rec. Use (Class 2b)	< 40	< 14	> 1.4
CHF – Aquatic Rec. Use (Class 2b) Shallow lakes	< 60	< 20	> 1.0
WCP & NGP – Aquatic Rec. Use (Class 2B)	< 65	< 22	> 0.9
WCP & NGP – Aquatic Rec. Use (Class 2b) Shallow lakes	< 90	< 30	> 0.7

Criteria Considerations and Applications

MPCA developed guidance for assessing nutrient-impairment for lakes, for purposes of 303(d) listing and 305(b) reporting (MPCA 2003). This guidance document was in support of formal rulemaking (Minn. R. Ch.7050). The assessment factors and approach used relied heavily on the data bases and ecoregion-based approach previously described in this report. As such it is worthwhile to review some of the considerations and information acquired (e.g., public input) during that process in our current effort to develop eutrophication criteria. Among the questions considered in the TMDL assessment methodology rulemaking were: a) what thresholds to assign for the eutrophication variables used in the assessment; b) need for both causal and response thresholds; c) amount of data required to do an assessment; d) index period for the assessments and d) identifying any special cases where thresholds may not apply or when site-specific thresholds may be necessary.

The TP thresholds, developed for the "assessment factor rule," drew heavily from the previously developed TP criteria (i.e., Heiskary and Wilson, 1988). Corresponding chlorophyll-a and Secchi thresholds were then developed based on a combination of user perception information and the relationship among TP, chlorophyll-a, and Secchi based on Carlson's TSI (Fig. 3) and MPCA regression equations (e.g., Equation 1-3). In the guidance MPCA elected to assign a range of values to be used in the assessments (Table 21). In this fashion it was acknowledged that there may be some close calls when assessing a lake near the threshold values and this allowed for increased scrutiny of the data for those lakes.

Based on public feedback during that rulemaking process and in response to an early draft 303(d) list it was evident that we needed to establish thresholds for both the causal (TP) and response (chlorophyll-a and Secchi) variables. And that a combination of the two would be needed to determine whether a lake is meeting the criteria. Table 21 provides the threshold values that were used to assess lakes in the 2002 and 2004 assessments and place lakes on the 303(d) list. The nutrient criteria (Table 20), as described herein, would replace the threshold values currently used for 303(d) assessment and listing (Table 21).

As we look ahead to application of the criteria some basic definitions and our overall approach needs to be described so that the criteria are applied appropriately. A typical approach for evaluating the impact of nutrients on lakes requires a defined index period, sampling regimen, and a standard suite of parameters to measure. In Minnesota a typical approach for assessing lake condition (or compliance with nutrient criteria) is as follows:

- Index period - we focus on the summer period (roughly June through September);
- Sample the lake at one or more mid-lake or mid-bay sites about four - eight times with sampling distributed over the summer index period. These data can be augmented with Secchi measures and user perception data collected by volunteers in the Citizen Lake Monitoring Program to provide a more complete picture of inter- and intra-summer variability in Secchi and user perception.
- Water chemistry samples (standard trophic status variables and supporting physical/chemical data) are collected with a two meter integrated sampler. Oxygen and temperature profiles are taken to document stratification and provide information on hypolimnetic oxygen concentrations.

- Trophic status and related water quality data are averaged over the summer. These averages would provide a basis for evaluating whether criteria are being met and can provide a basis for estimating the frequency of occurrence of extreme events (e.g., Figure 6a). Standard error of the mean should be calculated as well to provide a basis for describing the variability of the data (e.g., CV or coefficient of variation of the mean) and comparing means between years and perhaps with an ambient criteria value.
- A minimum of two years of data may be desirable as well to ensure accurate characterization of trophic status, establish linkages between causative and response variables, and provide some sense of inter-year variability. It will be important to understand the anticipated inter-year variability in the trophic status parameters that will occur naturally. For 303(d) listing purposes 12 paired (TP, chlorophyll-a and Secchi) observations were established as the minimum amount of data required to conduct an assessment for 303(d) purposes.

Table 21. Trophic Status Thresholds for Determination of Use Support for Lakes. (Carlson's TSI Noted for Each Threshold.)

Ecoregion (TSI)	TP ppb	Chl ppb	Secchi m	TP Range ppb	TP ppb	Chl ppb	Secchi m
305(b):	**Full Support**			**Partial Support to Potential Non-Support**			
303(d):	**Not Listed**			**Review**	**Listed**		
NLF	< 30	<10	≥ 1.6	30 – 35	> 35	> 12	< 1.4
(TSI)	(< 53)	(< 53)	(< 53)	(53-56)	(> 56)	(> 55)	(> 55)
CHF	< 40	< 15	≥ 1.2	40 - 45	> 45	> 18	< 1.1
(TSI)	(< 57)	(< 57)	(< 57)	(57 – 59)	(> 59)	(> 59)	(> 59)
WCP & NGP	< 70	< 24	> 1.0	70 - 90	> 90	> 32	< 0.7
(TSI)	(< 66)	(< 61)	(< 61)	(66 – 69)	(> 69)	(> 65)	(> 65)

TSI = Carlson trophic state index; Chl = Chlorophyll-a; ppb = parts per billion or µg/L; m = meters

Several other considerations rose in the course of developing the assessment factors rule that merit inclusion here as it may affect how criteria are developed and/or applied. A summary of these considerations as follows:

- <u>Defining lakes</u> – MPCA has routinely relied on Bulletin 25 (MDNR, 1968) as the primary basis for identifying lakes and reservoirs. However some lakes listed in Bulletin 25 are actually wetlands. If a lake basin listed in Bulletin 25 is listed as a wetland on the MDNR Public Waters Inventory (that has essentially replaced Bulletin 25) it is considered a Class 2D wetland and the applicable wetland water quality standards apply. The eutrophication criteria described in this document are not intended to be applied to wetlands for purposes of 303(d) listing or 305(b) assessment, however the general principles included herein may be of value as nutrient or biological criteria are developed for wetlands.

- Shallow lakes – As defined in this document we are referring in general to lakes with a maximum depth of 15 feet or less or a littoral (percent of lake 15 feet deep or less) area of 80 percent or more. This definition was derived based on observations by Schupp (1992) and personal conversation with MDNR staff. This definition assumes that the lake is listed in Bulletin 25 and is not classified as a wetland in the Public Waters Inventory.
- Lake size – Bulletin 25 is limited to lakes of ten acres or greater. When we talk about the total number of lakes in Minnesota that is generally the smallest size that is considered a "lake." While 303(d) assessment is limited to lakes of ten acres or greater, based on the guidance manual (MPCA 2003), the eutrophication criteria included herein apply to water bodies smaller that ten acres that are shown to be lakes. However, in general, when we refer to lakes for purposes of applying the criteria we are referring to water bodies 10 acres or greater that are listed in Bulletin 25 and/or the Public Waters Inventory.
- Reservoirs - Sampling regimen and assessments for aquatic recreational use for reservoirs may be different from those used for lakes. Since reservoirs typically exhibit distinct zones, often referred to as inflow segment, transitional segment, and near-dam segment, calculation of "whole reservoir" mean TP may not be an appropriate basis for assessing support of aquatic recreational use. Rather, the MPCA may want to evaluate the status of the reservoir based on a specific segment – most likely the near-dam segment. Also, water residence time may vary substantially as a function of river flow (e.g., Lake Pepin, Heiskary and Walker 1995) and may influence algal response to available nutrients. In addition, reservoirs often have very large watersheds that may drain portions of one or more ecoregion. Hence the ecoregion criteria, based on where the reservoir is located, may not be an appropriate basis for evaluating use support. Thus it is our expectation that site-specific eutrophication criteria may need to be developed for many reservoirs. This will be especially true for reservoirs that drain very large land areas that span more than one ecoregion (e.g. Lake Pepin).
- Bayed lakes – Lakes that comprised of numerous individual bays, such as Lake Minnetonka, may present a similar situation. The bays may need to be assessed on an individual basis (our current method for storing data for "bayed lakes" readily allows for this). In some instances a single bay may exceed the listing thresholds (eutrophication criteria) while other bays in the lake do not. In this case it should be determined whether the entire lake should be listed (e.g., there is distinct interaction between the bays and the entire lake must be addressed) or simply the individual bay. This will likely require knowledge of flow-through patterns in the lake and assistance from local cooperators to make an appropriate determination.
- Residence time – Residence time is the primary basis for separating reservoirs from rivers for purposes of applying lake nutrient thresholds for 303(d) assessment. Based on the guidance manual reservoirs with a residence time of less than 14 days will not be assessed as lakes (as assessed based on summer-mean low flow for the four-month summer season, June – September, with a once-in-ten year recurrence interval). This is consistent with EPA's current guidance (USEPA 2000a & Kennedy 2001). As a practical manner in rivers one would use the interrelationships among TP, chlorophyll-a and other

factors (derived from rivers) as a basis for assessing nutrient impacts (e.g. Heiskary and Markus, 2002).

As we look ahead to application of the nutrient criteria it might be worthwhile to provide an estimate of the number or percentage of lakes in each ecoregion that may be above criteria levels (and perhaps in need of TMDL studies), those near criteria (which may be a priority for more detailed sampling to determine use-support), and those below criteria (which may require various levels of "protection" to ensure that they remain in good condition and below nutrient criteria thresholds). There are various ways to derive these estimates – for example we can compare the distribution of assessed (sampled) lakes (Table 1) to the criteria values as is the case in Table 22. We can also derive estimates based on remote sensing data. Figure 33 provides a distribution of late summer Secchi transparency by aggregated EPA ecoregion based on a recent remote sensing project conducted by the University of Minnesota, Water Resources Laboratory (2000). These estimates of peak growing season (e.g., late July through August) transparency, for over 10,000 lakes, when compared to the proposed Secchi criteria yield the estimates of the number of lakes above and below the ecoregion-based Secchi criteria values (Table 23). Based on these very general comparisons we see that a majority of the lakes in the NLF ecoregion are likely in compliance with the TP and Secchi criteria values proposed for support of aquatic recreational use. In contrast, in the WCP ecoregion a majority of the lakes have TP concentrations above the 70 - 90 µg/L range and Secchi readings above (shallower than) the 0.7 - 0.9 m range. In the CHF ecoregion 50 percent or more of the lakes are likely above the proposed TP criteria. Based on TP distributions for shallow lakes (Fig. 30) it seems likely that 75 percent or more may be above the proposed TP criteria for shallow lakes. These estimates can be refined in future work but for now they provide some perspective on how these criteria relate to the overall populations of lakes in each ecoregion.

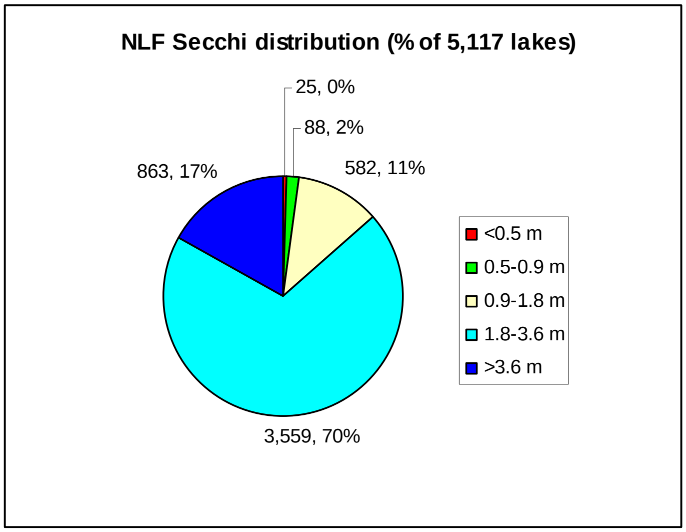
**Table 22. Percent of lakes above or below proposed TP criteria.
Based on comparison with Table 1.**

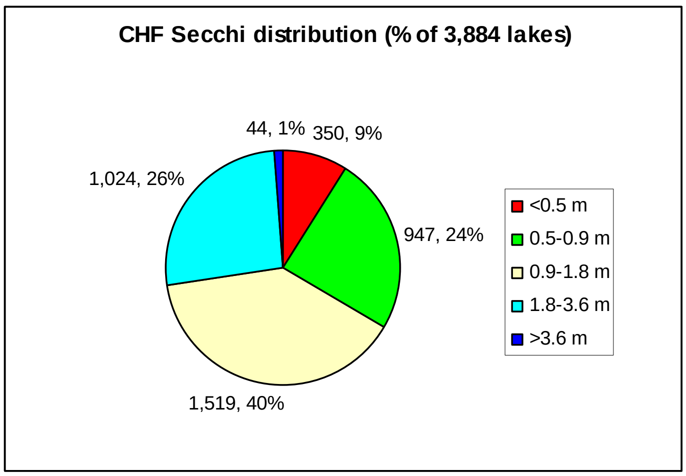
Ecoregion	Criteria TP ppb	% of lakes above criteria (higher TP)	% of lakes below criteria (lower TP)
NLF	< 30	~ 25%	~ 75%
CHF: deep - shallow	< 40 - 60	~ 50%	~ 50%
WCP: deep - shallow	< 65 - 90	~ 80 %	~ 20%

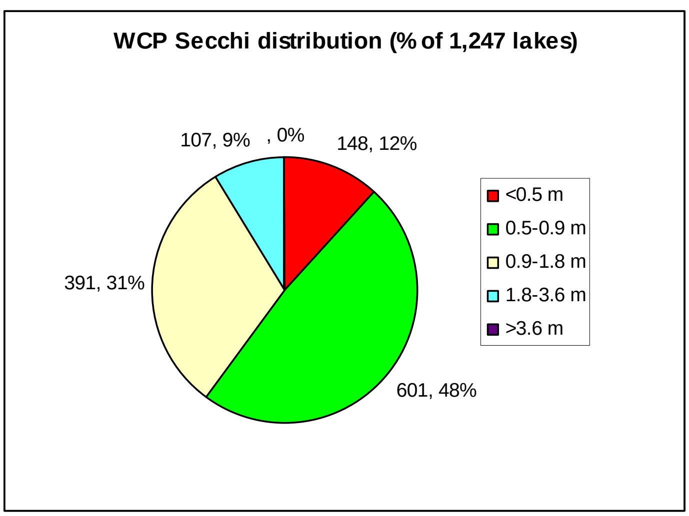
**Table 23. Percent of lakes above or below proposed Secchi criteria range.
Based on Figure 33.**

Ecoregion	Criteria (m)	% of lakes above (shallower Secchi)	% of lakes below (deeper Secchi)
NLF	> 2.0	~ 13%	~ 87%
CHF: deep - shallow	> 1.4 – 1.0	~33%	~ 67%
WCP: deep - shallow	> 0.9 – 0.7	~ 60%	~ 40%

Figure 33. Late-summer Secchi distribution based on remote sensing (UM-WRL, 2000). Presented by EPA aggregated ecoregion.







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Appendix

I. Ecoregion reference lakes
II. Diatom reconstruction lakes
III. Designated lake trout lakes
IV. Case Studies

I. Ecoregion Reference Lakes

		Lake		Watershed		Land use						
LakeID	**Lake**	**Depth**	**Area**	**Wshed**	**Wrat**	**For**	**Wat**	**Res**	**Pas**	**Cul**	**County**	**Eco.**
		m	**ha**	**mi2**		**%**	**%**	**%**	**%**	**%**		
03-0107	**Toad**	5.6	729.8	12.6	4.5	48.2	20.5	4.0	25.1	1.7	Becker	CHF
03-0381-01	**Detroit (Main Bay)**	4.8	1261.0	69.1	14.1	30.0	10.0	9.0	26.0	20.0	Becker	CHF
03-0576	**Big Cormorant**	7.7	1368.0	26.8	5.1	11.0	39.0	11.0	16.0	20.0	Becker	CHF
03-0657	**Turtle**	8.4	74.2	1.7	5.9	.	.	.	.	.	Becker	CHF
21-0016	**Smith**	4.6	235.1	14.9	16.4	3.0	6.4	2.0	24.4	59.4	Douglas	CHF
21-0057	**Carlos**	13.3	1020.0	222.9	56.4	13.0	20.0	7.0	22.0	35.0	Douglas	CHF
21-0076	**Irene**	6.0	257.0	9.3	9.3	24.0	11.0	4.0	18.0	40.0	Douglas	CHF
21-0079	**Maple**	5.1	336.7	5.4	4.1	5.7	20.6	4.0	18.8	48.8	Douglas	CHF
21-0085	**Andrews**	8.9	402.0	5.7	3.7	5.0	22.0	6.0	17.7	49.8	Douglas	CHF
21-0145	**Chippewa**	6.3	710.0	3.7	1.3	10.5	50.0	7.0	11.3	21.5	Douglas	CHF
34-0079	**Green**	6.4	2188.0	115.9	13.7	15.0	14.0	6.0	22.0	34.0	Kandiyohi	CHF
34-0142	**George**	4.9	100.0	0.7	1.8	9.0	27.0	45.0	18.0	0.0	Kandiyohi	CHF
40-0033	**Volney**	7.0	114.1	3.2	7.2	0.0	11.9	1.0	8.9	73.9	Le Sueur	CHF
40-0117	**Washington**	3.6	609.0	12.3	5.2	1.0	14.0	13.0	12.0	57.0	Le Sueur	CHF
49-0140	**Cedar**	11.7	102.0	2.0	5.1	29.8	19.3	0.0	28.0	22.9	Morrison	CHF
56-0302-01	**Silver**	4.8	221.0	2.1	2.5	0.0	30.0	0.0	3.0	52.0	Otter Tail	CHF
56-0306	**Elbow**	6.3	76.0	3.8	12.9	10.0	20.0	0.0	21.0	49.0	Otter Tail	CHF
56-0358	**Scalp**	9.8	98.0	1.4	3.7	36.4	30.2	15.0	4.7	14.0	Otter Tail	CHF
56-0382	**East Twin**	4.5	136.7	1.7	3.2	12.7	38.8	0.0	0.0	41.1	Otter Tail	CHF
56-0475	**Pickerel**	8.5	335.0	7.2	5.5	15.0	19.0	8.0	2.0	57.0	Otter Tail	CHF
56-0695	**Heilberger**	6.2	90.0	0.9	2.6	25.7	36.5	3.0	15.1	20.3	Otter Tail	CHF
56-0784	**Long**	7.8	304.8	8.6	7.3	41.9	18.6	4.0	7.3	28.2	Otter Tail	CHF
56-0877	**Jewett**	7.0	297.2	2.4	2.1	12.2	42.1	14.0	5.8	26.2	Otter Tail	CHF
61-0023	**Grove**	2.8	151.6	12.7	21.6	4.3	5.9	3.0	30.5	50.1	Pope	CHF
61-0037	**Linka**	6.7	73.4	1.6	5.6	0.1	13.2	2.0	16.4	68.7	Pope	CHF
61-0041	**Scandinavian**	4.5	170.6	2.6	3.9	8.9	21.8	16.0	19.5	33.5	Pope	CHF
66-0029	**Fox**	6.1	125.0	12.9	26.6	6.0	10.0	6.0	13.0	61.0	Rice	CHF
66-0052	**Cedar**	2.7	363.3	6.3	4.5	7.6	20.9	4.0	12.9	42.8	Rice	CHF
77-0007	**Mound**	8.1	106.0	1.3	3.2	53.1	14.6	9.0	23.8	0.0	Todd	CHF
77-0019	**Mary**	7.5	41.9	1.4	8.6	19.9	13.2	0.0	35.4	31.6	Todd	CHF
77-0026	**Moose**	4.0	50.0	1.5	7.7	12.3	5.0	0.0	38.1	40.5	Todd	CHF
77-0027	**Long**	9.9	150.0	7.1	12.2	19.9	12.7	7.0	24.6	36.3	Todd	CHF
77-0032	**Lady**	6.5	83.5	2.1	6.5	3.0	28.0	13.0	18.3	35.4	Todd	CHF
77-0089	**Little Birch**	10.6	321.0	56.4	45.3	25.0	9.0	3.0	26.0	37.0	Todd	CHF
77-0149	**Long**	5.1	57.7	1.9	8.5	46.9	18.4	0.0	26.0	8.7	Todd	CHF
81-0055	**Reeds**	4.4	73.8	0.9	3.1	0.0	35.2	12.0	17.6	34.8	Waseca	CHF
86-0223	**Indian**	3.3	52.8	0.9	4.4	10.0	.	.	.	.	Wright	CHF
86-0233	**Sugar**	7.3	372.0	9.3	6.5	0.4	13.0	18.0	11.0	42.0	Wright	CHF
06-0002	**Artichoke**	1.8	811.7	10.4	3.3	0.0	33.4	0.0	15.2	50.9	Big Stone	NGP
41-0021	**Dead Coon**	1.5	232.3	19.0	21.1	1.3	3.6	0.0	14.2	81.3	Lincoln	NGP
41-0043	**Benton**	2.0	1152.0	38.7	8.7	0.0	9.4	2.0	14.5	72.1	Lincoln	NGP
42-0047	**Yankton**	7.9	154.0	1.5	2.5	0.0	29.7	18.0	8.7	43.7	Lyon	NGP
42-0052	**Rock**	1.6	146.8	4.3	7.6	0.0	17.0	0.0	5.3	77.7	Lyon	NGP

42-0093	**East Goose**	2.1	54.0	1.5	7.2	0.0	12.4	0.0	4.2	83.4	Lyon	NGP
51-0063	**Sarah**	1.1	483.0	16.9	9.0	0.0	12.8	0.0	7.1	79.5	Murray	NGP
51-0082	**Currant**	1.6	215.7	2.5	3.0	0.0	25.4	3.0	6.8	63.1	Murray	NGP
75-0200	**Hattie**	1.7	192.3	7.7	10.3	0.0	10.3	0.0	3.9	85.8	Stevens	NGP
76-0086	**Hassel**	1.1	266.5	22.7	22.0	0.0	4.9	0.0	23.8	69.9	Swift	NGP
03-0029	**Hungry Men**	1.2	37.0	0.7	4.9	29.9	34.1	0.0	0.0	32.4	Becker	NLF
03-0030	**Boot**	13.10	140.4	1.8	3.3	45.4	24.8	4.0	9.5	14.0	Becker	NLF
04-0011	**Moose**	5.6	231.9	6.2	6.9	68.9	21.5	.	5.8	0.0	Beltrami	NLF
04-0166	**Julia**	4.80	177.8	5.3	7.7	63.2	15.6	0.0	17.0	3.8	Beltrami	NLF
04-0230	**Deer**	4.70	108.1	4.4	10.5	80.3	11.1	1.0	6.2	0.3	Beltrami	NLF
11-0059	**Washburn**	6.00	614.1	30.7	12.9	69.0	22.3	7.0	0.7	0.0	Cass	NLF
11-0092	**Little Sand**	0.80	172.6	3.1	4.6	58.0	21.7	3.0	13.0	0.0	Cass	NLF
11-0101	**George**	1.50	339.5	20.7	15.7	90.0	4.5	0.0	3.1	0.0	Cass	NLF
11-0102	**Island**	6.70	111.7	4.5	10.4	72.8	17.7	3.0	5.6	1.1	Cass	NLF
11-0105	**Upper Trelipe**	7.20	164.9	1.5	2.3	47.9	39.1	9.0	4.5	0.0	Cass	NLF
11-0116	**Stevens**	4.30	35.5	0.5	3.6	56.5	33.4	7.0	1.8	0.0	Cass	NLF
11-0250	**Ada**	6.60	398.0	12.8	8.3	58.3	22.3	0.0	3.4	1.5	Cass	NLF
15-0005	**Squaw**	7.40	60.9	7.7	32.6	91.9	8.1	0.0	0.0	0.0	Clearwater	NLF
15-0010	**Elk**	10.50	109.3	3.1	7.3	85.8	14.2	0.0	0.0	0.0	Clearwater	NLF
15-0057	**Long**	13.50	58.5	0.9	4.0	84.5	7.1	8.0	0.0	0.0	Clearwater	NLF
16-0049	**Trout**	10.20	104.7	1.8	4.4	80.0	20.0	0.0	0.0	0.0	Cook	NLF
16-0077	**Greenwood**	10.62	828.6	10.9	3.4	69.0	30.0	1.0	0.0	0.0	Cook	NLF
16-0182	**Ball Club**	2.6	82.4	1.3	4.1	81.0	19.0	0.0	0.0	0.0	Cook	NLF
29-0312	**Cedar**	4.10	39.3	0.5	3.3	37.5	27.8	0.0	22.6	0.0	Hubbard	NLF
31-0051	**Stingy**	4.60	142.1	2.3	4.2	82.7	16.5	0.0	0.8	0.0	Itasca	NLF
31-0069	**Buck**	4.90	183.5	1.6	2.3	38.5	28.6	33.0	0.4	0.0	Itasca	NLF
31-0424	**Burnt Shanty**	2.60	73.8	0.8	2.8	66.8	8.0	25.0	0.0	0.0	Itasca	NLF
31-0438	**Sand**	5.90	60.5	0.7	3.0	36.0	32.5	17.0	5.3	0.0	Itasca	NLF
31-0671	**Big Island**	2.60	96.0	1.3	3.5	62.6	27.7	10.0	0.0	0.0	Itasca	NLF
38-0047	**Wilson**	5.9	260.3	4.9	4.9	74.0	26.0	0.0	0.0	0.0	Lake	NLF
38-0529	**Snowbank**	15.1	1860.4	57.6	8.0	68.0	32.0	0.0	0.0	0.0	Lake	NLF
38-0664	**Dunnigan**	2.4	32.6	0.4	2.9	46.0	34.0	0.0	0.0	0.0	Lake	NLF
69-0085	**Fenske**	4.40	42.3	1.4	8.5	90.4	4.3	5.0	0.0	0.0	St. Louis	NLF
69-0118	**Burntside**	15.0	4144	58.0	5.1	72.0	26.0	2.0	0.0	0.0	St. Louis	NLF
69-0923	**Hobson**	4.70	26.6	0.5	0.5	81.7	0.0	0.0	17.7	0.5	St. Louis	NLF
07-0047	**George**	2.9	57.0	1.3	5.9	5.0	15.0	0.0	5.0	75.0	Blue Earth	WCP
07-0054	**Ballantyne**	2.4	143.0	5.8	10.5	0.0	17.0	11.0	8.0	59.0	Blue Earth	WCP
17-0007	**Bingham**	2.0	106.0	1.7	4.1	0.0	21.1	16.0	10.1	52.7	Cottonwood	WCP
22-0074	**Bass**	3.1	74.6	1.0	3.5	12.2	28.2	14.0	1.4	38.3	Faribault	WCP
32-0018	**Fish**	4.5	115.7	2.1	4.7	0.0	9.0	15.0	5.9	70.4	Jackson	WCP
32-0022	**Clear**	1.9	182.7	1.8	2.5	0.0	26.3	0.0	0.0	73.7	Jackson	WCP
46-0133	**Big Twin**	2.3	180.2	1.8	2.6	0.3	40.0	0.0	3.3	56.3	Martin	WCP
51-0040	**Bloody**	1.8	107.7	1.0	2.4	20.8	23.3	19.0	0.0	36.5	Murray	WCP
74-0023	**Beaver**	4.4	38.7	0.4	2.7	16.2	0.0	36.0	28.1	20.0	Steele	WCP
81-0003	**St. Olaf**	4.4	35.9	0.3	2.2	21.8	1.0	16.0	0.0	61.8	Waseca	WCP
87-0016	**Curtis**	1.8	130.2	7.7	15.3	0.4	9.0	0.0	5.2	82.3	Yellow Medicine	WCP
87-0030	**Wood**	1.6	185.9	9.0	12.5	1.3	0.0	0.0	1.5	97.2	Yellow Medicine	WCP

II. Diatom reconstruction lakes

Diatom-inferred TP for 69 Minnesota lake cores (Edlund, 2005). Two samples (1800s, 1750s) represent sediments deposited before European settlement, and one sample (surface, 0-2 cm) represent modern lake conditions (c. early 1990s). Diatom-inferred based on 79 lake model (west-central shallow lakes not used in this version of the model). Lakes sorted by project and ecoregion.

Project	Group	Eco	Name	ID #	Z max	Z mean	obs TP	1990s DI-TP	1800 DI-TP	1750 DI-TP
					m	m	ppb	ppb	ppb	
55MNLakes	Metro	NCHF	Marcott	19-0042	10.1	2.0	19	20	19	25
55MNLakes	Metro	NCHF	Dickman	19-0046	2.4	2.4	105	—	26	18
55MNLakes	Metro	NCHF	Fish_D	19-0057	10.1	1.8	79	40	43	29
55MNLakes	Metro	NCHF	Schultz	19-0075	4.9	2.4	26	22	19	21
55MNLakes	Metro	NCHF	Harriet	27-0016	25.0	8.8	27	38	17	20
55MNLakes	Metro	NCHF	Calhoun	27-0031	27.4	10.7	28	41	18	23
55MNLakes	Metro	NCHF	Sweeney	27-0035-01	7.6	3.7	46	33	34	27
55MNLakes	Metro	NCHF	Twin	27-0035-02	17.1	9.4	22	19	21	23
55MNLakes	Metro	NCHF	Christmas	27-0137	26.5	11.0	15	37	27	36
55MNLakes	Metro	NCHF	Little Long	27-0179-02	23.2	18.7	10	14	29	22
55MNLakes	Metro	NCHF	Gervais	62-0007	12.5	5.8	33	35	55	57
55MNLakes	Metro	NCHF	McCarrns	62-0054	17.4	7.6	48	46	24	29
55MNLakes	Metro	NCHF	Owasso	62-0056	12.2	3.4	40	27	25	25
55MNLakes	Metro	NCHF	Turtle	62-0061	8.8	3.7	27	21	21	18
55MNLakes	Metro	NCHF	Johanna	62-0078	12.5	5.2	36	42	39	33
55MNLakes	Metro	NCHF	L_Carnelian	82-0014	19.2	3.9	9	17	20	20
55MNLakes	Metro	NCHF	Square	82-0046	20.7	9.1	12	13	24	12
55MNLakes	Metro	NCHF	Elmo	82-0106	42.7	13.4	10	17	20	20
55MNLakes	Metro	NCHF	Tanners	82-0115	13.7	7.0	56	20	33	31
55MNLakes	Metro	NCHF	Carver	82-0166	11.0	4.6	36	35	21	23
55MNLakes	NCHF	NCHF	Diamond	34-0044	8.2	4.8	80	87	36	35
55MNLakes	NCHF	NCHF	Long_K	34-0066	13.4	5.4	18	24	20	23
55MNLakes	NCHF	NCHF	Hendersn	34-0116	12.2	6.6	22	29	38	24
55MNLakes	NCHF	NCHF	George_K	34-0142	9.1	4.9	15	23	36	49
55MNLakes	NCHF	NCHF	Hook	43-0073	5.5	2.1	68	64	58	51
55MNLakes	NCHF	NCHF	Stahl	43-0104	10.7	4.8	46	46	23	21
55MNLakes	NCHF	NCHF	Dunns	47-0082	6.1	3.6	139	56	22	19
55MNLakes	NCHF	NCHF	Richardson	47-0088	14.3	5.9	98	48	34	28
55MNLakes	NCHF	NCHF	Sagatagan	73-0092	14.3	3.0	27	15	33	20
55MNLakes	NCHF	NCHF	Kreighle	73-0097	17.1	5.2	11	14	19	17
55MNLakes	NLF	NLF	Dyers	16-0634	6.1	2.8	27	22	27	24
55MNLakes	NLF	NLF	Forsythe	31-0560	3.1	1.8	20	15	15	16
55MNLakes	NLF	NLF	Snells	31-0569	15.2	6.0	24	17	15	10
55MNLakes	NLF	NLF	Long	31-0570	22.9	5.4	12	16	15	11
55MNLakes	NLF	NLF	Loon	31-0571	21.0	6.3	11	13	22	15
55MNLakes	NLF	NLF	Little Bass	31-0575	18.9	7.6	13	12	29	29
55MNLakes	NLF	NLF	Ninemile	38-0033	9.1	2.4	17	14	12	10
55MNLakes	NLF	NLF	Wilson	38-0047	14.9	5.9	13	25	40	36

55MNLakes	NLF	NLF	Windy	38-0068	11.9	7.0	12	9	11	10
55MNLakes	NLF	NLF	Tettegouche	38-0231	4.6	1.3	17	17	14	15
55MNLakes	NLF	NLF	Nipisiqt	38-0232	5.5	4.3	16	18	24	25
55MNLakes	NLF	NLF	Wolf	38-0242	7.3	3.7	14	17	8	12
55MNLakes	NLF	NLF	Bear	38-0405	8.8	4.5	11	20	15	11
55MNLakes	NLF	NLF	Bean	38-0409	7.9	4.6	17	20	21	22
55MNLakes	NLF	NLF	August	38-0691	5.8	2.5	15	12	20	14
55MNLakes	NLF	NLF	Little Trout	69-0682	29.0	13.1	7	8	11	10
55MNLakes	NLF	NLF	Tooth	69-0756	13.1	6.0	12	10	11	12
55MNLakes	NLF	NLF	Shoepack	69-0870	7.3	4.3	18	20	21	14
55MNLakes	NLF	NLF	Loiten	69-0872	14.9	9.1	7	11	10	10
55MNLakes	NLF	NLF	Locator	69-0936	15.9	8.2	9	8	10	10
55MNLakes	WCP	WCP	George	07-0047	8.5	2.9	130		76	41
55MNLakes	WCP	WCP	Duck	07-0053	7.6	3.0	65	41	29	53
55MNLakes	WCP	WCP	Bass	22-0074	6.1	3.1	81	52	55	90
55MNLakes	WCP	WCP	Fish	32-0018	8.2	4.5	38	48	87	83
55MNLakes	WCP	WCP	Beaver	74-0023	8.2	4.4	30	35	49	58
CentMN	NCHF	NCHF	Quamba	33-0015	3.4	1.8	90	60	49	52
CentMN	NCHF	NCHF	Johanna	61-0006	3.7	2.1	80	47	28	50
CentMN	NCHF	NCHF	Fremont	71-0016	3.0	2.1	110	36	27	26
CentMN	NCHF	NCHF	Silver	72-0013	2.7	1.4	160	81	27	29
CentMN	NCHF	NCHF	McCormic	73-0273	3.7	2.1	60	55	36	45
CentMN	NCHF	NCHF	Monson	76-0033	6.4	3.7	90	29	76	71
CentMN	NLF	NLF	Platt	18-0088	7.0	3.0	30	38	69	39
CentMN	NLF	NLF	Red Sand	18-0386	7.0	2.1	30	22	21	18
SWMN	NGP	NGP	Shaokatan	41-0089	3.7	2.4	124	105	48	52
SWMN	NGP	NGP	East Twin	42-0070	6.7	3.0	104	186	110	86
SWMN	WCP	WCP	Cottonwood	17-0022	2.7	1.5	84	122	94	98
SWMN	WCP	WCP	Clear	32-0022	3.0	1.8	110	67	34	27
SWMN	WCP	WCP	Big Twin	46-0133	3.0	2.4	141	114	42	72
SWMN	WCP	WCP	Bloody	51-0040	3.4	1.5	89	181	60	77

Appendix III. MDNR Designated Lake Trout Lakes

Minnesota Rules Part 7050.0470 Excerpts

Lake Trout Lakes Adopted as Outstanding Resource Value Waters, Restricted Dischargers Category, March 7, 1988.

Subpart 1. **Lake Superior Basin.**

(8) *Bearskin Lake, West, [3/7/88R] (T.64, 65, R.1): 1B, 2A, 3B; **16-228**

(11) *Birch Lake, [3/7/88R] (T.65, R.1, 2): 1B, 2A, 3B; **16-247**

(42) *Echo Lake, [3/7/88R] (T.59, R.6): 1B, 2A, 3B; **38-28**

(59) *Greenwood Lake, [3/7/88R] (T.64, R.2E): 1B, 2A, 3B; **16-77**

(65) *Kemo Lake, [3/7/88R] (T.63, R.1): 1B, 2A, 3B; **16-188**

(86) *Moss Lake, [3/7/88R] (T.65, R.1): 1B, 2A, 3B; **16-234**

(138) *Trout Lake, [3/7/88R] (T.62, R.2E): 1B, 2A, 3B; **16-49**

(142) *Twin Lake, Upper (Bear Lake), [3/7/88R] (T.56, R.8): 1B, 2A, 3B; **38-408**

Subp. 2. **Lake of the Woods Basin.**

(19) *Burntside Lake, [3/7/88R] (T.63, 64, R.12, 13, 14): 1B, 2A, 3B; **69-118**

(22) *Caribou Lake, [3/7/88R] (T.58, R.26): 1B, 2A, 3B; **31-620**

(68) *Gunflint Lake, [3/7/88R] (T.65, R.2, 3, 4): 1B, 2A, 3B; **16-356**

(85) *Johnson Lake, [3/7/88R] (T.67, 68, R.17, 18): 1B, 2A, 3B; **69-691**

(96) *Larson Lake, [3/7/88R] (T.61, R.24W, S.16, 21): 1B, 2A, 3B; **31-317**

(99) *Loon Lake, [3/7/88R] (T.65, R.3): 1B, 2A,

3B; **16-448**

(103) *Magnetic Lake, [3/7/88R] (T.65, R.3, 4): 1B, 2A, 3B; **16-463**

(107) *Mayhew Lake, [3/7/88R] (T.65, R.2): 1B, 2A, 3B; **16-337**

(120) *North Lake, [3/7/88R] (T.65, R.2): 1B, 2A, 3B; **16-331**

(124) *Ojibway Lake (Upper Twin), [3/7/88R] (T.63, R.9, 10): 1B, 2A, 3B; **38-640**

(152) *Spring Lake, [3/7/88R] (T.68, R.18): 1B, 2A, 3B; **69-761**

Subp. 4. **Upper Mississippi River Basin.**

(7) *Blue Lake, [3/7/88R] (T.46, 47, R.27): 1B, 2A, 3B; **1-181**

(8) *Blue Lake, [3/7/88R] (T.141, R.34): 1B, 2A, 3B; **29-184**

(9) *Bluewater Lake, [3/7/88R] (T.57, R.25): 1B, 2A, 3B; **31-395**

(18) *Kabekona Lake, [3/7/88R] (T.142, 143, R.32, 33): 1B, 2A, 3B; **29-75**

(36) *Pokegama Lake, [3/7/88R] (T.54, 55, R.25, 26): 1B, 2A, 3B; **31-532**

(38) *Roosevelt Lake, [3/7/88R] (T.138, 139, R.26): 1B, 2A, 3B; **11-43**

(49) *Trout Lake, Big, [3/7/88R] (T.57, 58, R.25): 1B, 2A, 3B; **31-410**

(50) *Trout Lake, Big, [3/7/88R] (T.137, 138, R.27, 28): 1B, 2A, 3B; **18-315**

(51) *Trout Lake, Little, [3/7/88R] (T.57, R.25): 1B, 2A, 3B; **31-394**

Subp. 6. **Saint Croix River Basin.**

(1) *Grindstone Lake, [3/7/88R] (T.42, R.21): 1B, 2A, 3B; **58-123**

Appendix IV. Lake Case Studies

Case Studies

The following case studies are included to provide examples of how eutrophication criteria may be applied for goal setting, listing of impaired waters, and protection of lakes that are currently below criteria levels. The examples are organized by ecoregion and draw on actual data and studies that have been conducted on the lakes. Several case studies from Heiskary and Wilson (1988) and have been included and updated where data permit. Additional studies from more recent work have been added as well. Many of the case studies refer to LAP studies and/or previously developed case studies from Heiskary and Swain (2002). The complete reports may be found on the MPCA web site at: http://www.pca.state.mn.us/water/lakequality.html#reports .

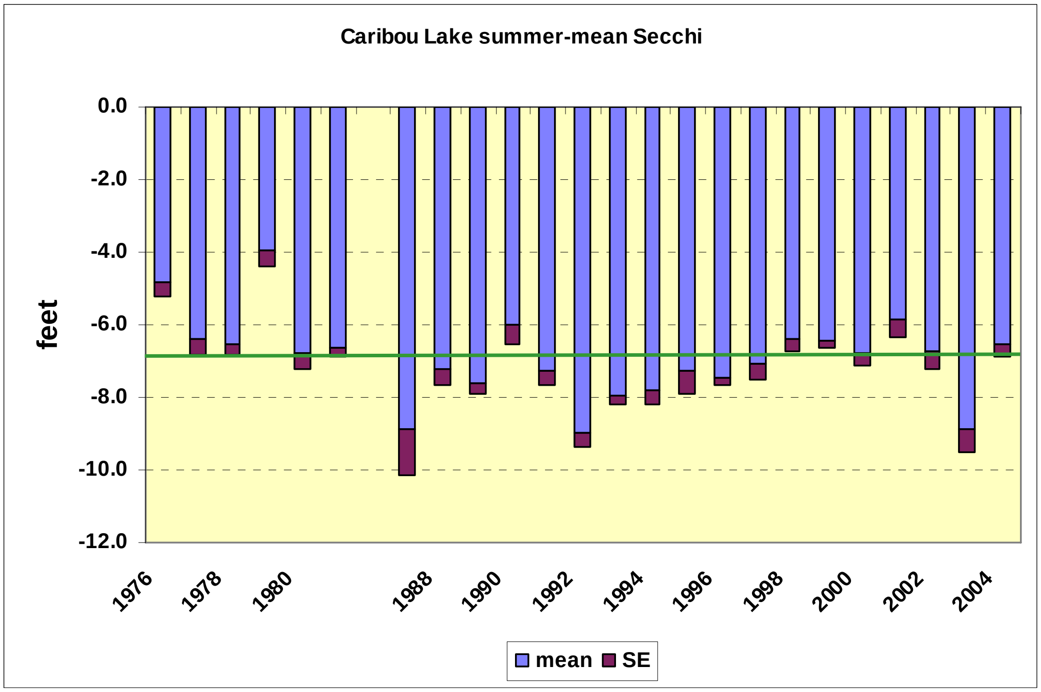
Northern Lakes and Forests

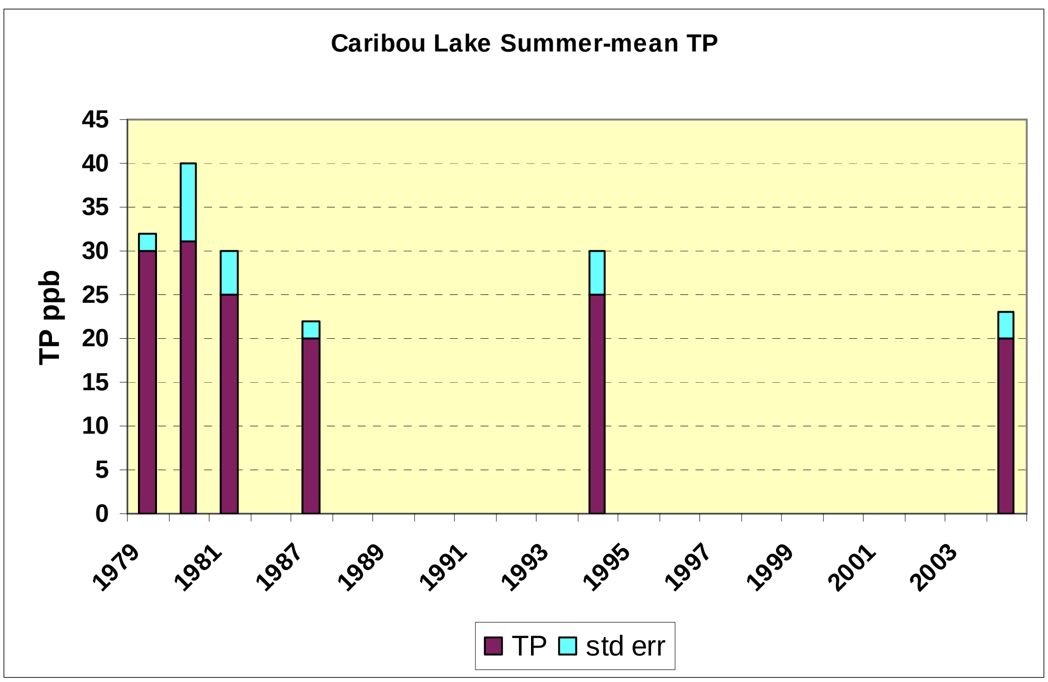
6. Caribou Lake (16- 0360) is located near Lutsen on the North Shore of Lake Superior. At 728 acres it ranks near the 90th percentile in terms of size for lakes in the NLF but with a maximum depth of 32 feet it is rather shallow. Caribou Lake was the subject of a 1987 LAP study. That study found an average TP of 20 µg/L, chlorophyll-a 7.5 µg/L, and Secchi of 2.7 m. These values are near the median for assessed lakes in the NLF (Table 1). The study also acknowledged that the lake may be subject to intermittent mixing as a result of its shallowness. An improving trend in Secchi was noted as compared to values in the 1970s (Fig. 1, below). It was surmised that this may be attributed to reductions in logging in the upper watershed since that time. However Secchi data collected since 1987 suggests a decline in transparency (Fig. 1).

The lake was again assessed in 1994. That study revealed increased TP and chlorophyll-a (25 and 14.5 µg/L respectively) as compared to 1987. Secchi transparency has declined since 1993 when a transparency of 2.7 m was measured. Development in the shoreland area and in the watershed has become an increasing concern with respect to the water quality of the lake. In 2003 a development proposal prompted Cook County to require an alternative area-wide review (AUAR) to assess the impact of this development on the water quality of the lake. Storm water from the development was of particular interest. That study examined various scenarios and their potential impact on the lake. MPCA monitored the lake in 2004 to support this effort and provide updated baseline data for this effort.

Based on data from 1994 and 2004 Caribou Lake is below the 30 µg P/L criteria value (Fig. 1). This suggests the lake should be protected from further increases in phosphorus loading as increased phosphorus will lead to increased chlorophyll-a and decreased Secchi and an increased frequency of nuisance blooms. Given the shallowness of the lake the responses may be more marked than that of a deeper lake and there is a potential for internal recycling to become a problem in the lake should TP concentrations increase further. This internal loading would contribute to the further decline in the water quality of the lake. In this case a no net increase in P loading might be an appropriate approach for protection of this lake.

Figure 1. Caribou Lake summer-mean Secchi transparency and TP.





7. Trout Lake (31-0216) near Coleraine is rather large (1,890 acres) and very deep (maximum depth 135 feet). This lake received wastewater discharges from the cities of Coleraine and Bovey from 1910 through early 1987. In-lake conditions from a 1987 LAP study were as follows: mean TP – 44 µg/l, mean chlorophyll-a – 14 µg/l, maximum chlorophyll-a – 34 µg/l and mean Secchi 3.6 meters. The TP and chlorophyll-a values would rank near the 90th percentile for lakes in the NLF ecoregion (Table 1). Blue-green algal blooms were common and algae frequently form thick scums on the downwind shore of the lake. By early August the hypolimnion (top $\cong$ 45 feet) was anoxic. At that time the lake was not extensively used by the public, due in part to its poor water quality relative to other lakes in this region. Plans to develop the near shore area were just underway at that time and it was anticipated this would increase usage and lake users will likely expect lake conditions comparable to similar lakes in this region. At that time a reasonable phosphorus goal for Trout Lake was thought to be in the 15 – 20 µg/l range, considering its morphometry and composition of land use in its watershed.

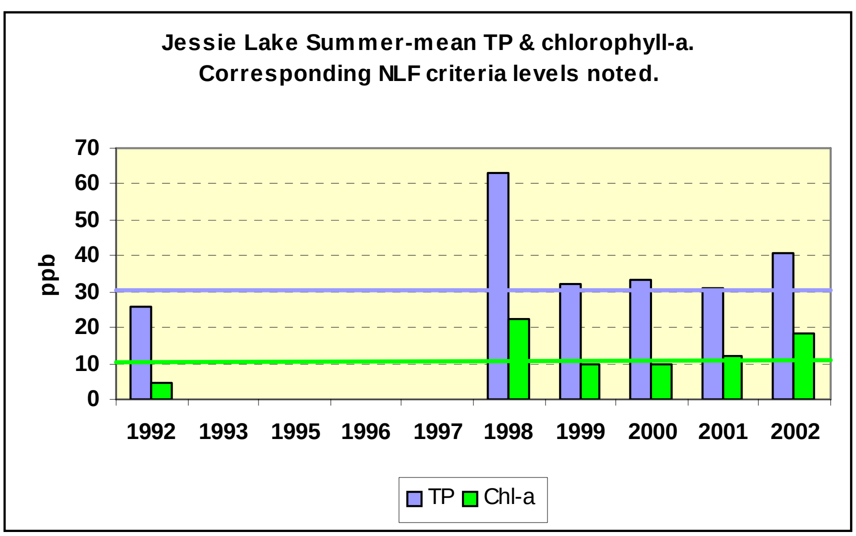
With the cessation of wastewater inputs in-lake TP and chlorophyll-a have declined dramatically (Case Study 5). TP and chlorophyll-a concentrations are now below ecoregion-based thresholds and there are indications of improving hypolimnetic oxygen (Case Study 5). However, while this has resulted in a reduction in the intensity and frequency of nuisance algal blooms current TP and chlorophyll-a concentrations may not be low enough to allow for a well-oxygenated hypolimnion, which would be required before lake trout could be re-introduced to the lake. Discussions of potential projects that might allow for dilution-flushing with water from nearby mine pits as been discussed as one means to hasten the recovery of the lake. Efforts are also underway to address storm water inputs that can potentially increase nutrient loading to the lake. Given its current water quality and the improvements that have been realized it is important to protect and if possible further improve the quality of Trout Lake.

8. Jessie Lake (31-786), one of the larger lakes of Itasca County of northern Minnesota, has experienced severe episodes of poor water quality over the past 10 years exceeding levels that should be reasonably expected, given its size, depth, watershed size, and ecoregion (CWP study summary). The watershed of Jessie Lake is 24.7 mi^2 and lies within the larger Rainy River watershed. Water from Jessie Lake flows into the Bowstring River, north through the Big Fork River to the Rainy River, and ultimately empties into Hudson Bay. Forest and wetlands constitute the largest percentage of landuse types within the watershed. The watershed consists mostly of forest (60%) and wetland (24%), with some agricultural land (9%). Over 100 private residences and four resorts are located within the Jessie Lake watershed. Most of the homes are along the Jessie Lake shoreline, or within a short vicinity of the lake (Reed and Watkins 1999).

Interest in the health of Jessie Lake and the surrounding watershed increased in the 1990's when lakeshore residents expressed concerns about algal blooms possibly leading to decreased water quality. To address residents' concerns, The Jessie Lake Watershed Association partnered with the Itasca County Soil and Water Conservation District (SWCD), the Chippewa National Forest, and the Minnesota Department of Natural Resources (DNR) to conduct water quality testing during 1998 and 1999. DNR grants were obtained to fund the lab analysis. This study revealed an in-lake total phosphorus concentration of 57 ug/l, an increase of 135% with respect to a similar study completed in 1992 (24 ug/l). This increase placed Jessie Lake in the 90th percentile with respect to total phosphorus concentrations in the NLF ecoregion. Severe algae blooms were frequent throughout the growing season, and consequently the lake suffered a decline in aesthetic and recreational value. It is evident that the water quality of Jessie Lake is unstable at best. These swings in phosphorus loading not only affect the lake seasonally (recreation and fishery potential), but are detrimental to the long-term "health" of the water body as well, and may be an indicator that Jessie Lake is approaching the point at which accelerated eutrophication could result from relatively minor increases in nutrient loading. Jessie Lake has been considered a premiere walleye fishery and destination for tourists and area residents, prior to these recent episodes. Changes in weather patterns, including increases in the amount of the ice free time as well as the length of growing season, have also been noted. Citizen and governmental agencies concerns over these highly valued waters resulted in obtaining a Clean Water Partnership diagnostic grant to study Jessie Lake in detail.

This Clean Water Partnership diagnostic effort relied upon state-of-the-art stream, lake water, and sediment investigations, working with the best diagnostic resources of the University of Minnesota including the Natural Resources Research Institute and the St. Anthony Falls Laboratory. The well defined results show: (1) that this is a extremely sensitive lake from a nutrient-eutrophication perspective; (2) the lake has received a succession of cumulative impacts from logging, agricultural and development stages; (3) that the lake is polymictic from a temperature perspective but not always for oxygen; (4) that the lake has significant and worsening internal phosphorus loading from it's sediments; and (5) that the lake can be improved significantly over time, by incremental reductions in external and internal P sources.

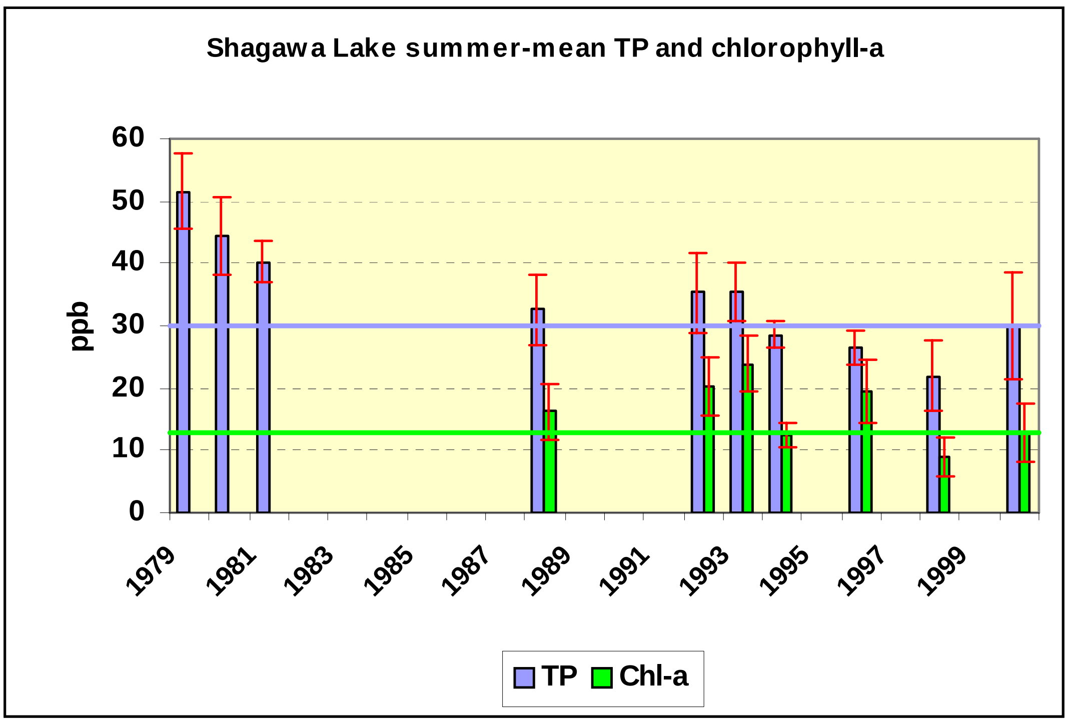
Lake management goals have been developed that will minimize watershed nutrient and sediment income to the lake. Present nutrient levels are relatively low, but improvable. Long-term lake restoration may require reduction of the primary P loading source – the lake's sediments. Whole lake sediment treatments are the typical approach, however this approach would be cost-prohibitive. And, internal loading appears to have been detected at a sufficiently early stage where incremental reductions will achieve longer-term stability (Hondzo et al, 2002). Research efforts continue to focus on cost-effective approaches for addressing internal loading in the lake. For Jessie Lake reducing in-lake P to 30 µg/L or less (NLF criteria) is a reasonable goal and should provide chlorophyll-a and transparency values more consistent with lakes in the NLF ecoregion.



9. Shagawa Lake (69-0069) is among the more studied lakes in the NLF ecoregion. In the 1970s it was the subject of EPA studies to determine the affect of tertiary wastewater treatment at Ely on the lakes' water quality. It has been studied by several other investigators as well and has been routinely monitored by MPCA since the mid 1980s.

Distinct reductions in TP and chlorophyll-a are evident based on available data in STORET. However, as stated in Case Study 4 improvement in lake condition has slowed somewhat in recent years as a result of internal P loading and intermittent (wind-driven) mixing of meta- and hypolimnetic waters with the epilimnion. As of 2004, Shagawa Lake was just below the NLF ecoregion 303(d) listing thresholds for TP and chlorophyll-a (Figure 1). Its TP and chlorophyll-a values rank near the 75^{th} and 90^{th} percentiles for the NLF ecoregion, respectively. It is important to minimize any further increases in P loading to Shagawa Lake and look for opportunities to reduce loading wherever possible (e.g., urban storm water) so the lake does not exceed these thresholds. Likewise it is important to continue to monitor this lake in order to assess trends over time and to determine whether the lake should be included on a future 303(d) list.

Figure 1. Shagawa Lake summer-mean TP and chlorophyll-a. NLF trophic status thresholds noted.



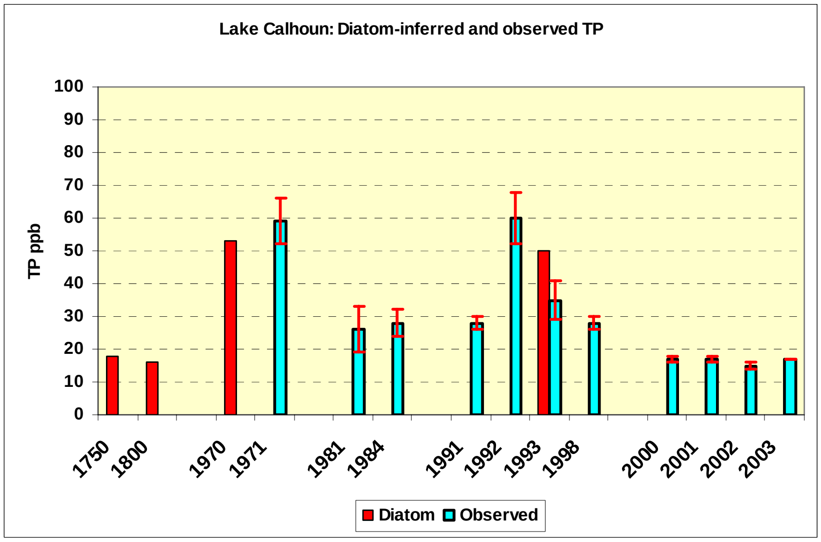
North Central Hardwood Forests

10. Lake Calhoun (27-0031) is among the largest lakes in the Metro area at 169 ha and with Lake Harriet, is the centerpiece for the Minneapolis park system. Calhoun has a very large drainage area (primary = 1,236 ha and total = 2,515 ha), while Harriet, which is immediately adjacent to Calhoun has a rather small watershed (387 ha). Extensive storm sewering of Calhoun's watershed (i.e., City of Minneapolis) led to increased nutrient loading and associated increases in in-lake TP by the 1960s and 1970s. The majority of the storm sewering activity was completed by the late 1980s.

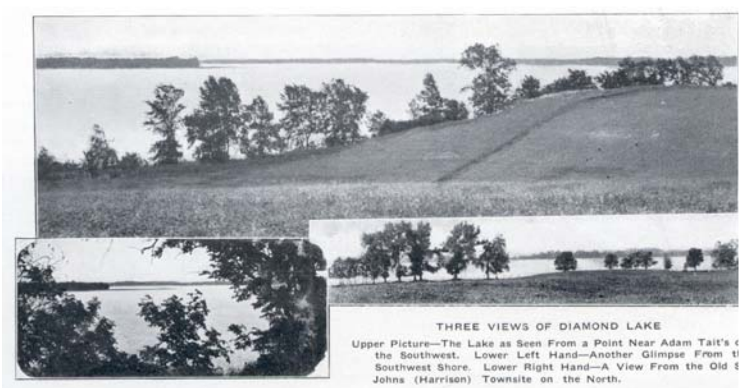
The diatom reconstructions for Calhoun suggest pre-European P concentrations that were on the order of 16 – 19 µg/L. However, by the 1970s, TP concentrations were on the order of 50 - 60 µg/L based on the diatom reconstructions. This peak in TP followed the extensive storm sewering that had occurred during the 1920s-1960s. Water quality monitoring data from the early to mid-1970s would appear to corroborate the diatom reconstruction (Figure 1). The reconstruction for c. 1990 indicated a small non-significant decrease in TP in Calhoun. Observed data from 1993 through 2003 indicate a decline in TP.

Recent declines in TP could be attributed to aggressive implementation of best management practices including street sweeping, improvements in storm water treatment, and related activities. Recent activities, conducted as a part of the Clean Water Partnership Project on the Chain of Lakes contribute to the recent improvements and should ensure protection of water quality into the future. Based on the most recent assessment (MPCA, 2004) TP and chlorophyll-a concentrations (31 and 8 µg/L respectively) below CHF threshold values and protection of current water quality should be a priority.

Figure 1. Lake Calhoun diatom-inferred and observed TP (Heiskary and Swain, 2000)



11. <u>Diamond Lake (34-0044)</u> has a large watershed (3,590 ha) that is characterized primarily (74%) by agricultural uses including row crop, pasture, and feedlots. Wetlands comprise about nine percent of the watershed, however many basins have been drained or ditched. Pre-settlement vegetation tended toward prairie to the south and oak-aspen to the north of the lake (Marschner, 1930) and this land now is largely in cultivated or other agricultural uses. Photographs from the late 1800's show a tree-lined shore and prairie/pasture-like upland. This shoreline is now heavily developed with over 350 cottages and year-round residences.



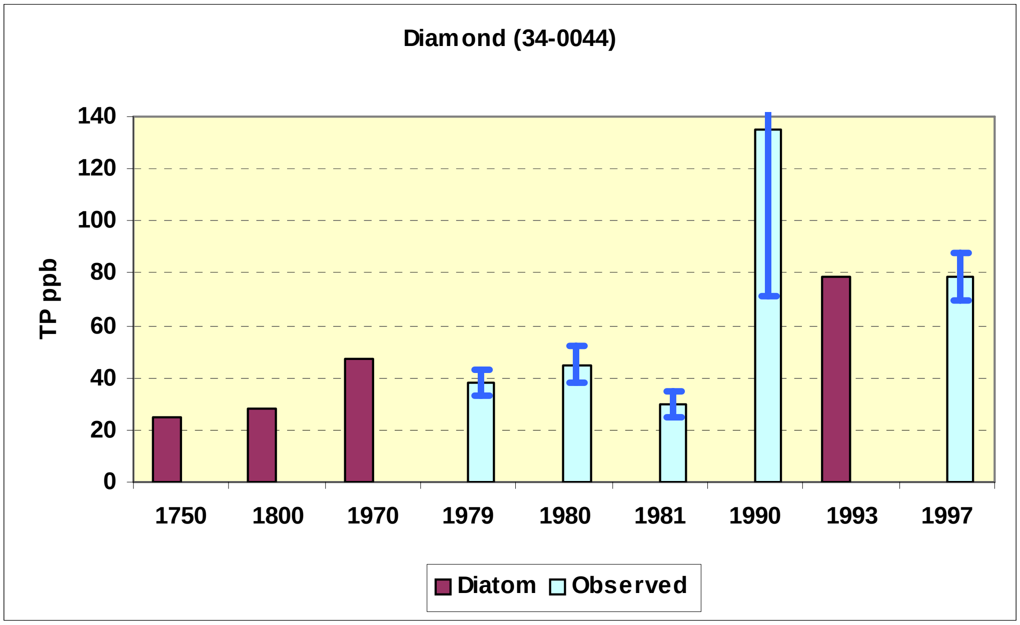
Diamond Lake, late 1800's (Lawson and Nelson, 1905)

There is extensive evidence of concern regarding the water quality of Diamond Lake based on various reports and memoranda. In 1972, for example, the MPCA, in a report to Kandiyohi County, noted in-lake TP concentrations of 40-50 µg/L (essentially equivalent to the 1970 diatom-inferred P value, Figure 1, below) and very high TP concentrations in tributaries to the lake.

- In 1986 a private consultant documented high nutrient concentrations.
- In 1988 AGNPS modeling, conducted by the SWCD and MPCA, attempted to estimate relative contributions of TP to the lake from different land uses. Row crop cultivation was thought to be an important source. Documentation for the modeling effort identified 23 feedlots in the watershed.
- In 1990 a public meeting was held to address water quality concerns. Shortly thereafter, Diamond Lake entered the CWP in response to declining water quality.

TP concentrations were already above pre-settlement by the 1970s, however these levels increased further by the 1990s based on diatom-inferred and observed data (Figure 1). TP has increased further by c. 1993 based on DI-P and monitored data. While a large increase in TP was noted between 1800 and 1970 an almost equal increase was noted over the 1970 to 1993 timeframe. Assessed data for the most recent ten-year timeframe note a mean TP of 79 µg/L and chlorophyll-a of 38 µg/L (MPCA, 2004), both of which are well above the CHF eutrophication criteria levels and the lake will be included on the 2006 draft 303(d) list. The modern-day increases in P and overall eutrophication of the lake can be attributed to the extensive agricultural activities in the watershed, combined with runoff from its urbanized shoreline. These data, diatom-inferred and observed, suggest that the CHF P criteria value of 40 µg/L is a reasonable goal for the lake and that achievement of this goal would lead to perceptibly improved water quality in the lake.

Figure 1. Diamond Lake diatom-inferred and observed TP (Heiskary and Swain, 2002)



12. Lake Volney (40-0033) is a dimictic lake in the southeastern portion of the North Central Hardwood Forests. It has a relatively small (watershed: lake surface ratio ~7:1) but highly agricultural watershed (83 percent). MPCA received numerous complaints from lake shore residents in 1985 and 1986 regarding degraded water quality. A LAP study in 1986, in response to these complaints, revealed mean chlorophyll-a concentrations in the 35 – 40 µg/l range and maxima in the 90 to 100 µg/l range (Wilson, 1987). These very high levels of algae kept residents from swimming most of the summer. Near-shore scums were evident during much of the summer (as evidenced in the inset picture). TP was very high and averaged 160 µg/l during the summer of 1986. Relative to lakes in the North Central Hardwood Forest, this would be between the 75th and 90th percentile (Table 1). A minimally impacted lake in this region would typically have a concentration less than 50 µg/l.

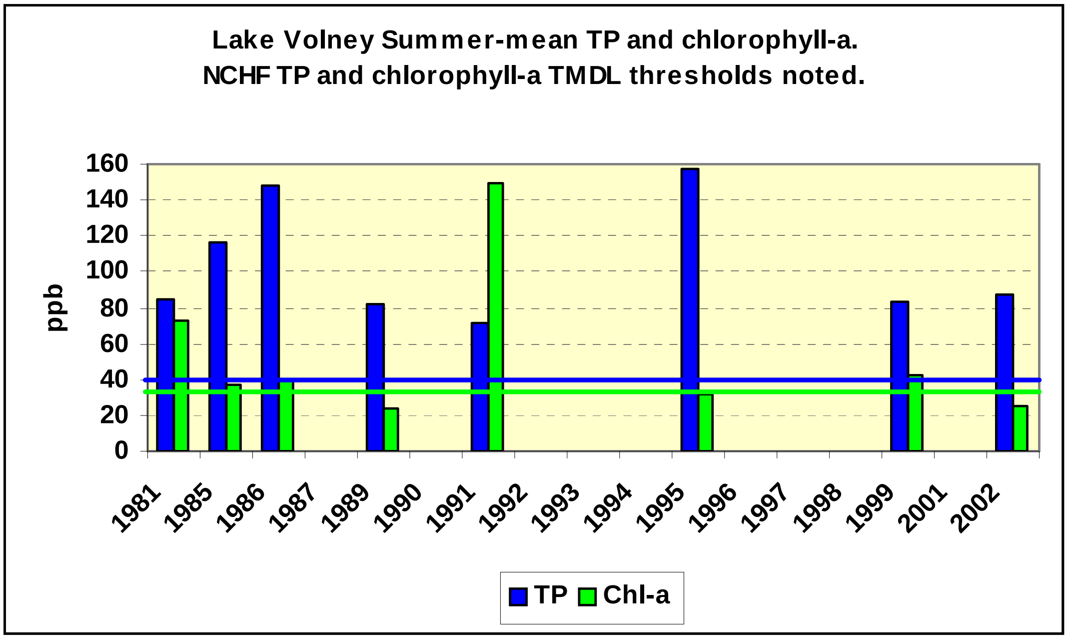
During the summer of 1987, Secchi transparency averaged 1.4 m and ranged between 0.8 to 2.1 m. The CLMP participant characterized recreational suitability as either “swimming impaired” or “no swimming” throughout the summer. No swimming was generally associated with transparencies less than 1.2 m. Based on the above information, Lake Volney was considered non-supportive of aquatic recreational use.

Modeling of watershed and in-lake sources of nutrients provided an estimate of what TP concentrations may be attainable for Lake Volney. Models by Chapra and Reckhow (1986) and Walker (1986) were used to estimate the amounts of water and phosphorus that reach Lake Volney (Wilson, 1987). Each model did a reasonable job of predicting current in-lake phosphorus with concentrations of 145 and 178 µg/l, respectively. These models suggest that Lake Volney receives nutrients at rates much greater than expected, based upon land use in the watershed. Watersheds with comparable land uses may be expected to generally contribute 0.1 to 1.3 pounds P/acre/year (Omernik, 1977). Whereas, Lake Volney appears to be receiving phosphorus at a rate of 3.8 pounds/acre/year or 3 to 10 times what may reasonable be expected, based upon land use (Wilson, 1987). A reduction in phosphorus export by a factor of four, i.e., a more typical export based on land use, should result in an in-lake phosphorus concentration in the 70-90 µg/l range. This analysis suggests that Lake Volney should respond favorably to a reduction in phosphorus loading from its watershed and should be able to attain a phosphorus concentration more typical of a dimictic lake in this region.

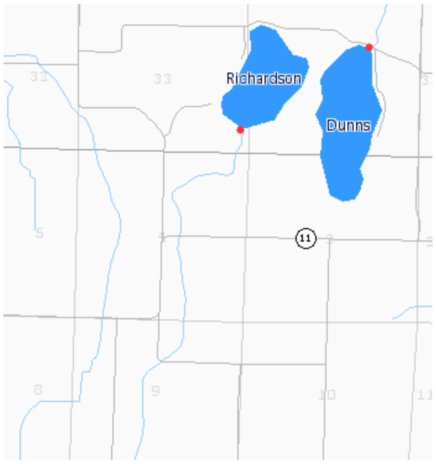
Further studies were conducted as a part of the CWP and local monitoring efforts (MPCA 1996). A sediment core study, based on cores collected in 1995 provides further insights on trends in P and sediment accumulation in the lake and landuse changes that have occurred since European settlement of the watershed (Umbanhower et al. 2003). Pre-European watershed land use was characterized as a mixture of forest, big woods and wetlands. With European settlement (c. 1880) the watershed was rapidly converted to cropland. Wetland drainage became prominent in the early 1900s with the tiling of a large wetland complex adjacent to the lake. Animal agriculture in the watershed increased steadily from the late 1800s through the 1970s as well (Umbanhower et al. 2003). Estimates of pre-European P accumulation suggest a P loading rate on the order of 226-277 kg P / yr. Back-calculation of TP in the water column using BATHTUB suggests an in-lake P of 33 µg/L, which is well within the range of pre-European P based on diatom reconstructions from 15 west-central and southern Minnesota lakes (Heiskary and Swain, 2002). The reconstruction of P loading to the lake (Umbanhower et al. 2003) and diatom reconstructions from other lakes in this region clearly indicate that Volney (and other lakes studied) were not naturally eutrophic. In the case of Lake Volney, Umbanhower et al. (2003) state that the CWP project goal to reduce TP to 60-80 µg/L may not be aggressive enough.

TP and chlorophyll-a remain high (Figure 1), well above ecoregion-based TMDL assessment thresholds, and severe nuisance blooms are common (inset photo above). These water quality characteristics caused Lake Volney to be listed on the 2002 303(d) list. A TMDL study of the lake will hopefully lead to solutions for reducing the excessive external and internal P loading that contributes to the extremely poor water quality in the lake. The study will also determine the extent of reductions needed to achieve the ecoregion-based criteria or other appropriate goals for the lake.

Figure 1. Lake Volney summer-mean TP and chlorophyll-a.



13. Dunns (47-0082) and Richardson (47-0088) are two adjacent lakes in the same watershed.

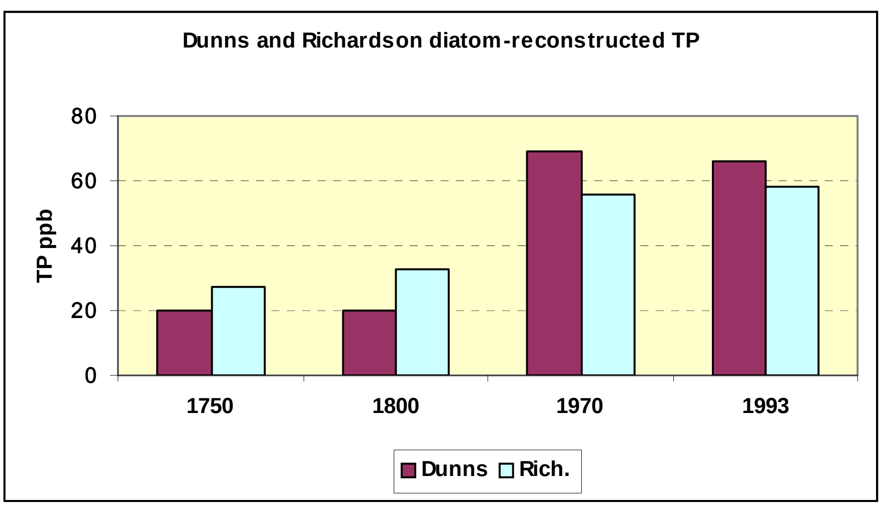


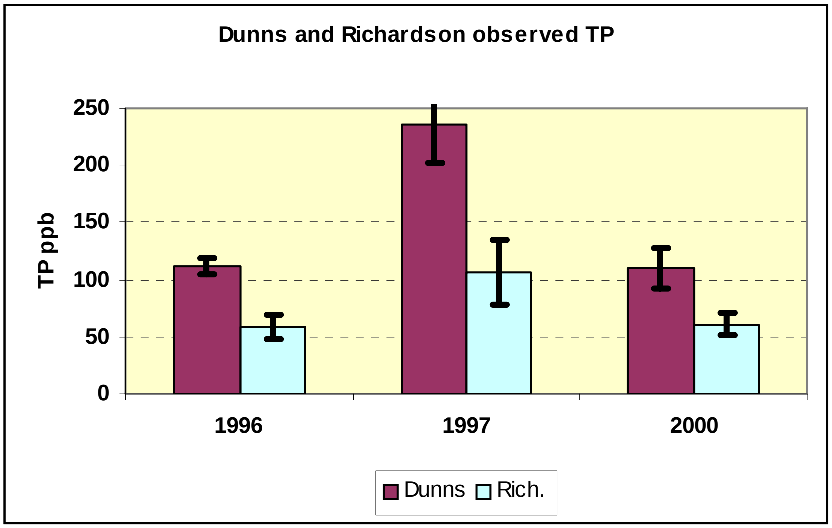
Richardson is the smaller but deeper of the two and receives the majority of the runoff from its 1,168 ha watershed. It, in turn, drains to Dunns Lake, which is somewhat larger (63 ha) but is much shallower with a maximum depth of 6.1 m. It has a small direct watershed of 165 ha in addition to Richardson's watershed.

The modern-day watershed is characterized by extensive agricultural landuse and is dotted by numerous wetlands. A 1996 LAP study (Heiskary et al. 1997) noted numerous animal feeding operations in the watershed with the following estimated numbers of animals: 220 dairy, 80 beef, 1.6 million chickens, 80,000 turkeys, 850 hogs, and 6,000 mink, as estimated by the Meeker SWCD. Since that time there has been a reduction in hog and mink farming, however there has been an increase in poultry farming in the watershed. The high number of animals in the watershed suggests that large amounts of manure are present in the watershed, and unless properly land-applied or otherwise processed, it could represent a significant source of pollution to the lakes. There were about 69 homes around the two lakes with the majority on Dunns. Following the LAP study a CWP project was undertaken on these lakes.

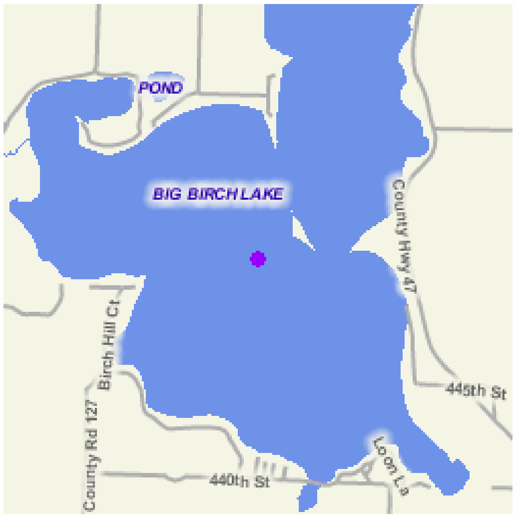
Pre-European TP levels for the lakes suggest fairly stable conditions between 1750 and 1800 (Figure 1, below) and that TP was higher in Richardson as compared to Dunns. This makes intuitive sense as Richardson would serve as a sedimentation basin for the upper watershed of the two lakes, hence reducing the load to Dunns. Modern-day TP show a dramatic increase in both lakes, with TP increasing three-fold in Dunns and two-fold in Richardson; however we note that TP concentrations in Dunns Lake are now greater than those in Richardson Lake. This was observed in the LAP study and was attributed in part to internal recycling of TP within the lake as a result of periodic wind-mixing of the lake during the summer. For Richardson the 1970 and c. 1993 diatom values compare quite well with observed values from 1996 and 2000 (Figure 1) and no significant trend is noted. The 1996 observed TP for Richardson was consistent with the TP predicted by the MINLEAP model, which implies that Richardson exhibits in-lake TP values consistent with that expected for a lake of its volume and watershed area in the CHF ecoregion. Based on these data it appears that major increases in TP loading to these lakes occurred prior to 1970 and have likely been sustained since that time. These data also suggest that as certain trophic “thresholds” are passed in shallow lakes that internal recycling can become a significant source of P to the lake. These data (diatom-reconstructed and observed) suggest that the CHF eutrophication criteria would be a reasonable goal for these lakes, however ultimately there may be need to address internal P loading as well as the external load.

Figure 1. Dunns and Richardson Lakes’ diatom-reconstructed and observed TP (Heiskary and Swain, 2002)





14. Big Birch Lake (77-0084) is a large lake (2,108 acres) with two distinct basins. Each bay is considered dimictic although there is evidence that mixing may occur during the summer after extended windy periods. No single land use dominates in the watershed typical for lakes in the CHF.



The upper (northeast) basin receives the drainage from most of the watershed and exhibits poorer water quality as a result. Based on a 1987 LAP study the two basins trophic conditions can be characterized as follows: upper basin – mean TP 53 µg/l, mean chlorophyll-a 13 µg/l, maximum chlorophyll-a 30 µg/l and mean Secchi 1.7 m; lower basin – mean TP 24 µg/l, mean chlorophyll-a 8 µg/l, maximum chlorophyll-a – 14 µg/l and Secchi 2.0 meters.

The difference in condition between the two basins is perceptible to residents on the lake and they also noted a decline in water quality over time. A trend assessment of CLMP data dating back to 1971 revealed a declining trend in transparency (MPCA, 1993). Average summer transparencies ranged from 4.6 m to 2.6 m in the 1970's compared to 3.0 m to 1.8 m in the 1980's (Figure 1). This implies a shift from mesotrophy, typical of minimally-impacted lakes in this region, to eutrophy.

This trend and differences in water quality among the two basins were recognized by the lake association, Todd County SWCD and the Sauk River Watershed District. A 100 percent locally-funded diagnostic study was conducted in the mid 1990s, leading to a state-funded CWP implementation grant. The Big Birch Lake Association also provided innovative funding to aid the implementation. These groups focused on the tributaries that drained to the upper basin. Using flow-weighted mean TP data from these tributaries in comparison to CHF minimally-impacted stream water quality data: Bass Creek – 140 µg/L, Calahan Creek 100 µg/L, Hoffman Creek – 240 µg/L and Fish Creek – 380 µg/L they determined that Fish and Hoffman Creeks were well in excess of the 60 – 150 µg/L range that is typical for minimally impacted CHF streams. Further for Fish Creek they noted that, although its subwatershed accounted for 44 percent of the drainage area to the upper lake it contributed 66 percent of the P loading. From there they set a goal of 150 µg/L for Fish Creek and concentrated their attention on that subwatershed. The group subsequently collaborated with local land owners to find solutions to reduce P loading to the stream.

Subsequent monitoring following successful implementation of the projects revealed reductions in in-stream TP and TP concentrations for Fish Creek are now in the 150 µg/L range based on data supplied by Sauk River Watershed District (personal communication). Reductions in in-lake TP is evident and the once distinct differences in TP among the upper and lower basin were no longer evident (Figure 2, below). Summer-mean transparency has increased lake-wide and as of about 1999 there was no longer a distinct difference in transparency between the two basins (Figure 1). The lake is currently below CHF eutrophication threshold values.

Figure 1. Big Birch Lake summer-mean Secchi: basin-specific means.

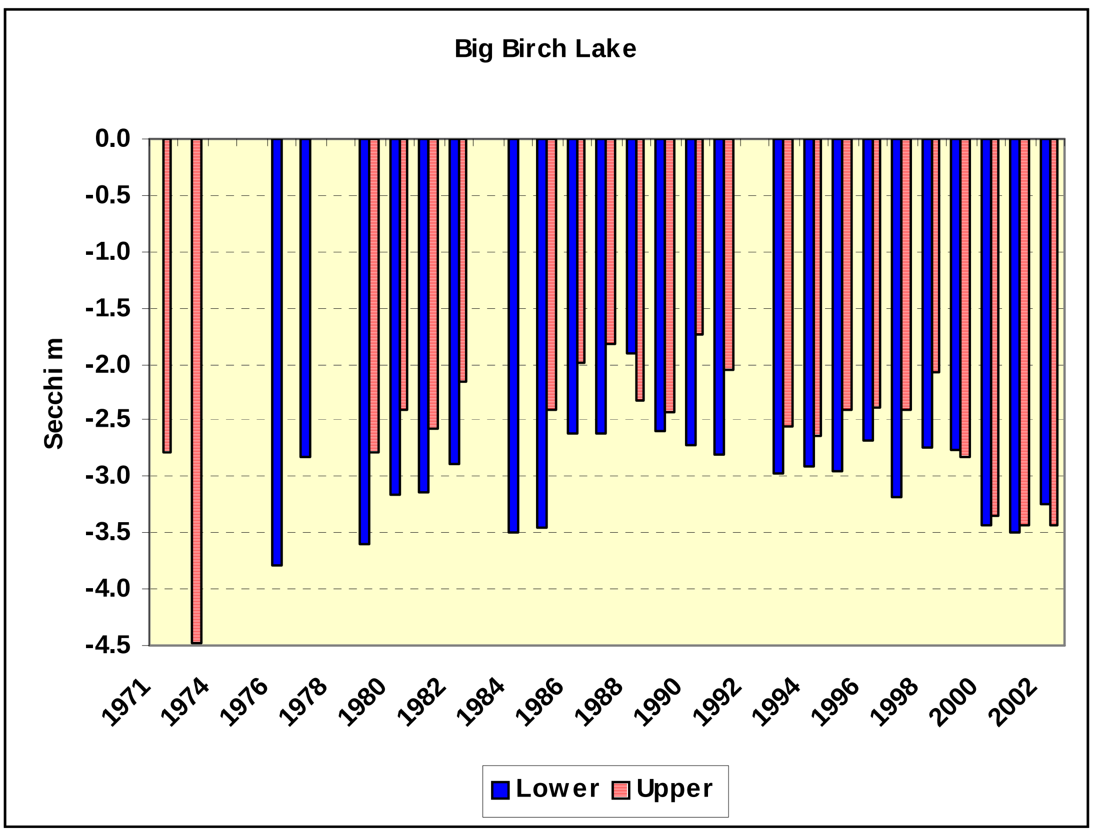
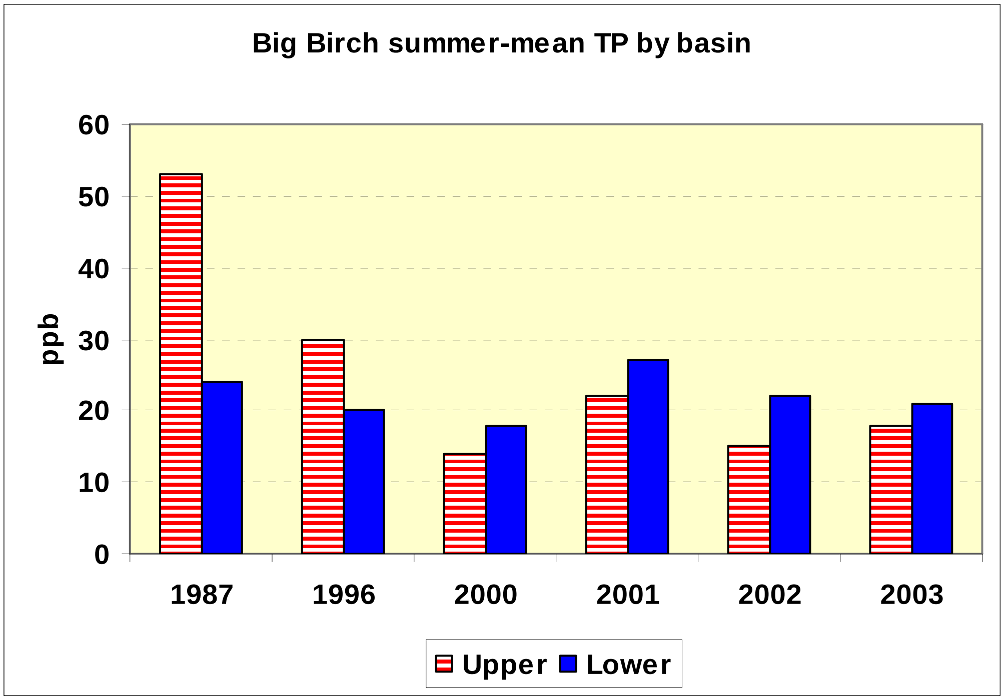
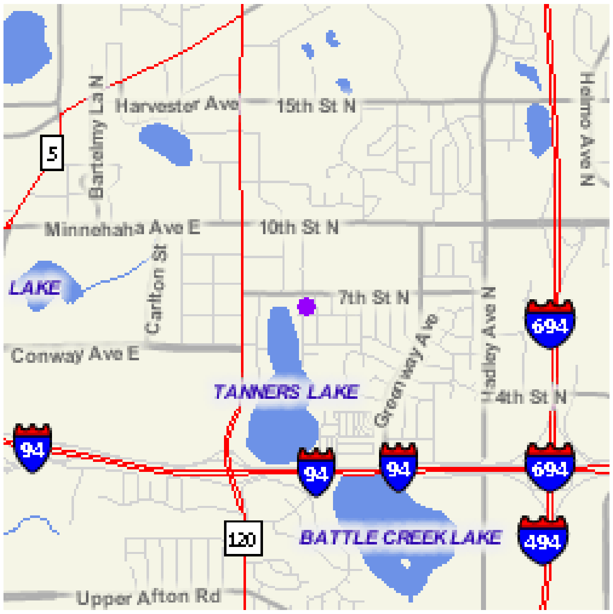


Figure 2. Big Birch Lake summer-mean TP by basin



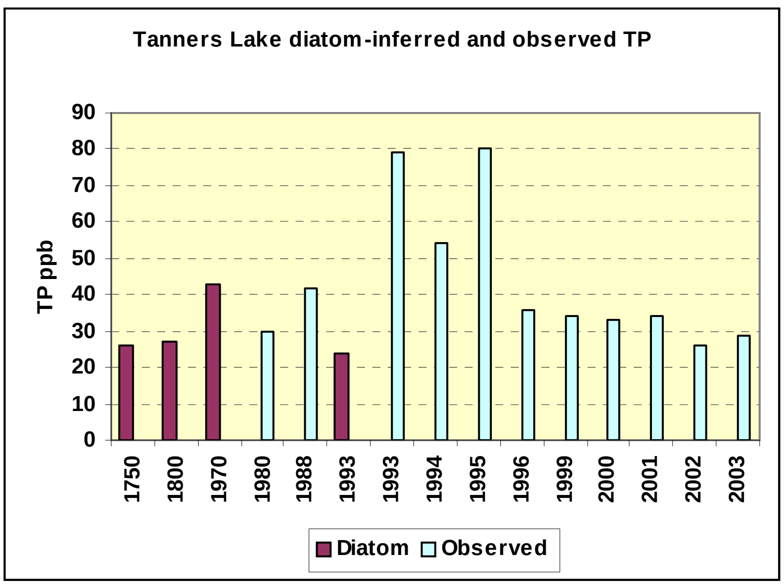
15. Tanners Lake (82-0115) is a small (30 ha) lake with a moderate sized watershed (22:1 ratio). Its modern-day watershed is characterized by built up (80%) land use and has a significant portion of the watershed in roads (8%).



Pertinent history of the lake and its watershed were summarized by Aichinger (2002, personal communication). Tanners Lake was a resort lake for St. Paul residents until about 1950. The first development dates to the 1930's and 40's with several cabins and homes built on the lake. Additional homes, businesses and condominium projects were built around the lake in the late 1950's and 1960's. A major freeway, I-94, was completed just south of the lake in the early 1960's. Storm sewers and sanitary sewers were constructed concurrent with development. Most storm sewers drained directly to the lake. Much of the remaining watershed was developed in commercial and high density residential in the 1960's and storm water from this area was directed to the lake as well. Watershed projects aimed at improving the quality of storm water that entered the lake date to the late 1980s when the Ramsey-Washington-Metro Watershed (RWMW) District completed a project to modify a wetland on the north end of the lake. In 1996 the Tanners Lake Water Quality Improvement project included the construction of a multi-cell storm water treatment pond, extended detention pond and alum treatment of storm water from the northern sub watershed.

Pre-European TP levels were somewhat higher than the other Ramsey County lakes and suggest meso-to mildly eutrophic conditions for the lake. TP increased almost two-fold by c. 1970 based on diatom-inferred and observed data (Figure 1). Development that occurred in the two to three decades prior to that time, combined with I-94 to the south of the lake would readily explain the increase in TP. This trend was reversed by the mid 1990s based on diatom-inferred and observed TP (1995-96), which corresponds to the timeframe of the water quality improvement projects. TP and chlorophyll-a available for the 2002 assessment period were just above the TMDL threshold values and the lake was included on the 2002 TMDL list. However completion of watershed BMPs, which included alum treatment of a primary inflow has led to reduced TP and chlorophyll-a concentrations in the lake (Figure 1). As these values are below CHF listing thresholds (Table 22) the lake will be delisted. Again the diatom-inferred and observed data suggest that a P criteria of 40 µg/L or less is a reasonable goal. And, in the case of Tanners Lake the interim P goal, developed by the watershed district, is 30 ± 5 µg/L (BARR, 2003).

Figure 1. Tanners Lake diatom-inferred and observed TP.

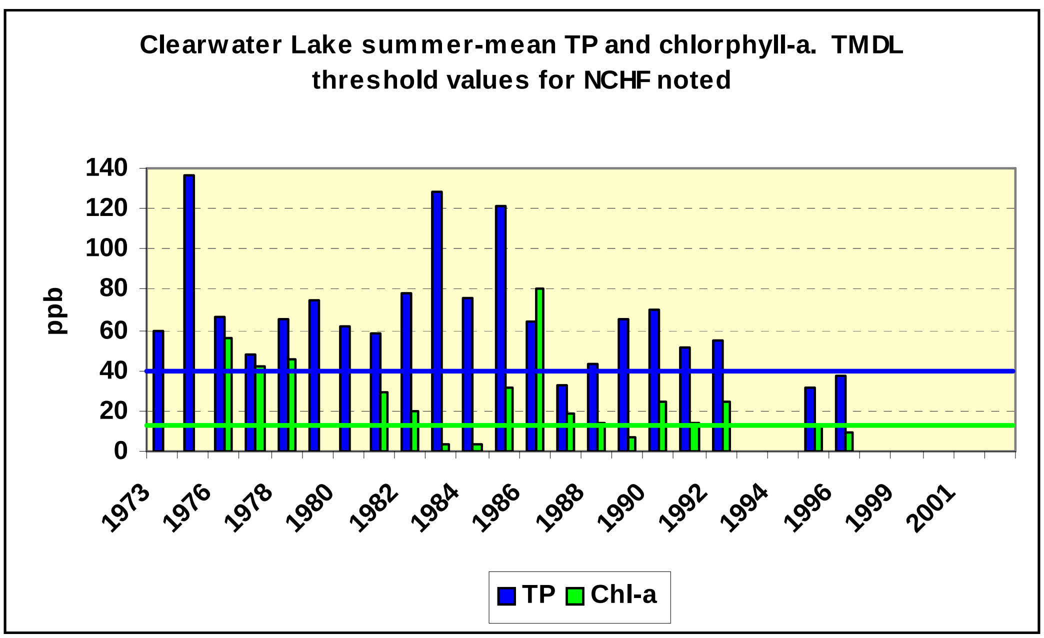


16. Clearwater Lake (86-0252) in Wright County is a very large (3,182 acres), dimictic lake, with two distinct basins. Its watershed is highly agricultural, but with increasingly dense urban development -- fairly typical of CHF lakes near the Metro area. The west basin receives the drainage from the Clearwater River Watershed (100 mi^2) and also has the outlet of the lake. In contrast, the east basin has a much smaller watershed and generally tends to flow towards the west basin.

Clearwater Lake was the focus of a major 314(a) Clean Lakes project. Phase II of this project was initiated in July of 1980. Efforts in the watershed to reduce phosphorus loading included wetland modifications and removal of point source discharges. In-lake measures have included hypolimnetic aeration and rough fish removal. Implementation of best management practices in the watershed is a continuing focus of the watershed district.

Complaints about the water quality of Clearwater Lake date back to the 1960's. In 1970, MPCA studied the lake and its watershed in response to complaints from the property owners (MPCA, 1971). TP was very high ranging from 60 to over 120 µg/L (Figure 1). Agricultural runoff and a seasonal discharge from the City of Annandale's wastewater treatment facility were the primary sources of excess TP to the lake. Wastewater was diverted out of the lake by 1982; however TP and chlorophyll-a remained high throughout the early to mid 1980s. The lake was considered not supportive of aquatic recreational use at that time. For example, in 1987 CLMP participants on Clearwater Lake noted transparencies in the one - two meter range and perceptions of "swimming impaired" or "no swimming" were common. By the mid 1990s TP declined below 40 µg/L and chlorophyll-a concentrations were on the order of 12 µg/L. From 1995 to 2002 summer-mean Secchi ranged from 2.2 to 2.8 m. These water quality measurements are below the ecoregion-based eutrophication criteria for the CHF and user perceptions generally indicate only "minor impairment" as compared to the swimming impaired or no swimming responses that were common in the prior years (http://www.pca.state.mn.us/water/clmp/lkwqReadFull.cfm?lakeID=86-0252).

Figure 1. Clearwater Lake summer-mean TP and chlorophyll-a.



17. Duck Lake (07-0053) is a moderate-sized lake north of Madison Lake in Blue Earth County. It has a small watershed relative to its size (3.3:1) that is dominated by agricultural land use, which is typical for the region. This lake was the focus of a LAP study and a CWP project.

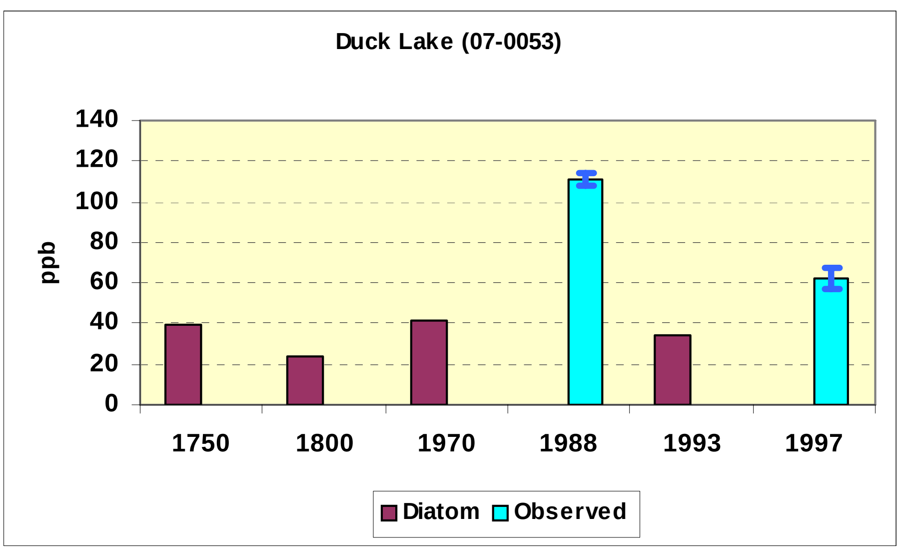
MDNR fishery files indicate lakeshore homes numbered 52 in 1950, 76 in 1956 and 121 by 1974. A 1988 LAP study indicated high development with about 100 permanent homes (Dindorf, 1989). Although there are no major tributaries to the lake, about 15 culverts and tiles drain the surrounding watershed.

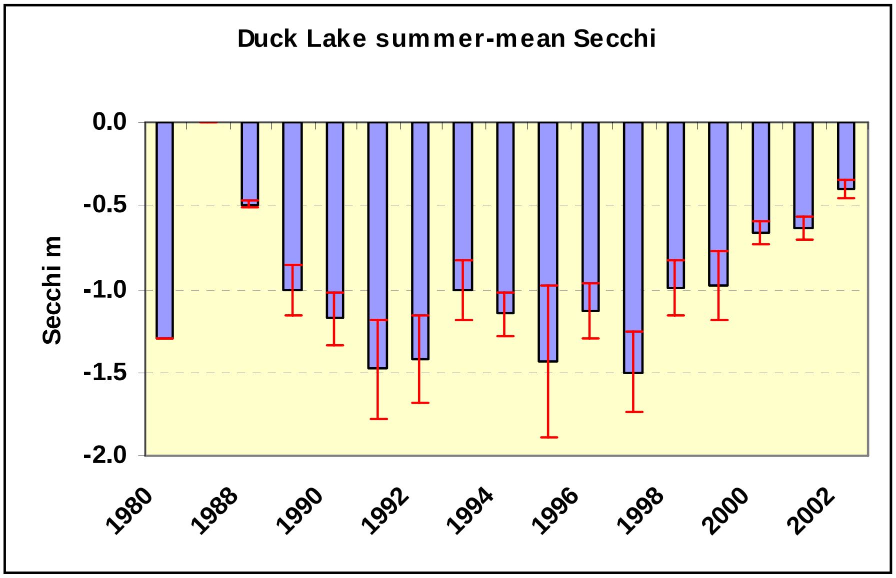
Historical information assembled as a part of the descriptive history of the lake (Dindorf, 1989) revealed a healthy northern pike fishery and abundant lily pads in the shallow bays in the 1940's to 1960's. One observer noted that commercial fisherman fished carp in the lake in the 1950's. A file review by Hugh Valiant (MDNR area fishery manager, personal communication) revealed fish surveys were conducted in 1939, 1950, 1956, 1970, 1974, 1985, 1990, 1995 and 2000. Winterkill of fish was documented once, in 1955-56. Curly-leaf pondweed was first documented in 1970. Submerged macrophytes seemed to be more diverse in 1950 and grew to depths of 9 or 10 feet (consistent with anecdotes provided by citizens as a part of the LAP study). Fishery surveys also noted that livestock watered along the shoreline during the 1950 and 1956 surveys.

The pre-European DI-P values suggest the lake was mildly eutrophic but was quite variable (Figure 1). No distinct trend is evidenced in the comparison of these values with the modern-day values; however variability is evident in the observed TP and Secchi data (Figure 2). Monitoring data from the 1988 LAP study suggest hypereutrophic conditions for the lake during the drought that was characteristic of the late 1980s. Slight improvement was noted by 1997 (Figure 1); however recent Secchi data suggest a return to hypereutrophic conditions (Figure 2).

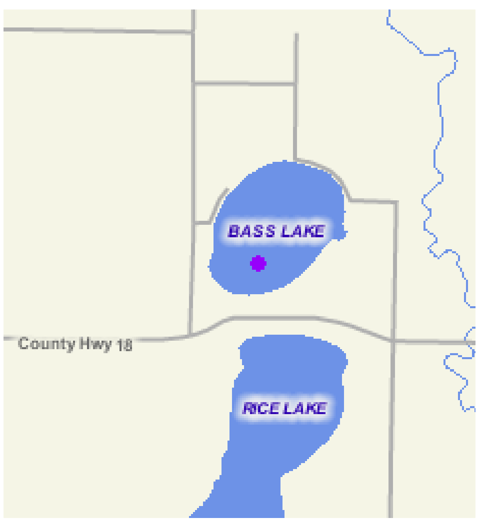
These data suggest that achieving a TP concentration of 70 µg/L or less is quite reasonable for Duck Lake. Given its relatively small watershed and that it is among the deeper lakes in the region (25 feet) a goal of 40 µg/L, consistent with full support of primary aquatic recreational use, may be achievable for the lake. However, based on recent Secchi data (Figure 2) it appears the trophic status of the lake may be declining rather than improving. Based on CWP studies to date, there appears to be a need to address both internal and external sources of TP to allow for improvements in in-lake water quality.

Figure 1. Duck Lake diatom-inferred and observed TP





18. Bass Lake (22-0074), a small lake near Winnebago in Faribault County, was one of the "reference" lakes in the 1980s. It has a small watershed relative to its surface area (1.8:1) that is characterized by a mix of agricultural and forested land. Its shoreline is fairly well developed with about 50 homes based on a 1993 LAP study (Weir, 1993). This represented an increase over the eight homes noted in a 1947 MDNR fishery survey. In a review of these surveys Valiant (DNR area fishery manager, personal communication) noted near-shore land use changed from 45% deciduous, 30% cultivated, 10% pasture and 15% pastured savannah in 1947 to a much higher percentage of cultivated land in recent surveys. There are no major tributaries in the watershed, however the lake receives tile drainage from about 5-10 field tiles (Weir, 1993). Water level has been a long-term concern and a tile was constructed in 1932 to allow flow from Rice Lake to augment Bass Lake. This tile was blocked in 1988 as a part of a fish reclamation project.



Fish surveys from 1947 – 1954 suggest abundant bluegill and crappie populations, however winterkills in 1977, 1978 and 1979 allowed for dominance by black bullhead. In early fish surveys the lake was considerably influenced from time to time by the riverine fish community (Valiant, personal communication). Clarity was relatively low at times; there was evidence of significant carp and bigmouth buffalo populations and even a 1956 letter on file complaining about loss of submergent macrophyte. Fish surveys also revealed that curly-leaf pondweed was the dominant macrophyte by 1986 and it was likely present in the lake since 1978. Senescence of curly-leaf was implicated in the mid-summer pulse in in-lake TP (Weir, 1993).

Pre-European TP data suggest variability between the 1750 to 1800 timeframe (Fig.1). Modern-day (observed and diatom-inferred) TP values suggest a marked increase from 1970 to mid

1990s. However, TP measurements were quite stable from 1986 to 1988 with values that correspond quite well with the c. 1993 sediment core TP. Secchi peaked at about 1.2 m in 1994-1997 with some decline since that time. Achieving a TP concentration of 65µg/L (WCP criteria) or less is a very reasonable goal for Bass Lake and in fact a 2004 study of the lake revealed summer-mean TP of 57µg/L, which is similar to the TP concentration in the 1980s. Recent reductions in TP were attributed to implementation of watershed BMPs and use of CRP on some of the previously cultivated land (Thompson, 2005 personal communication). However some concern was expressed on recent ditching activities and loss of CRP that may increase TP loading to the lake. Based on the LAP study attaining and maintaining lower concentrations will not only require reductions in watershed loading but would likely involve addressing internal sources such as curly-leaf pond weed and the periodic dominance of black bullhead, which continue to remain dominant based on a 2002 MDNR survey (http://www.dnr.state.mn.us/lakefind/showreport.html?downum=22007400).

Figure 1. Bass Lake diatom-inferred and observed total phosphorus.

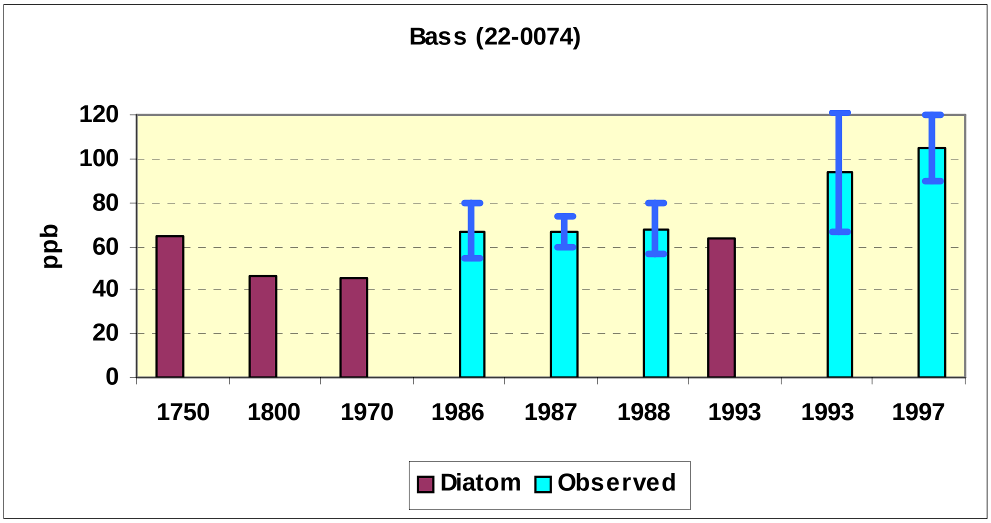
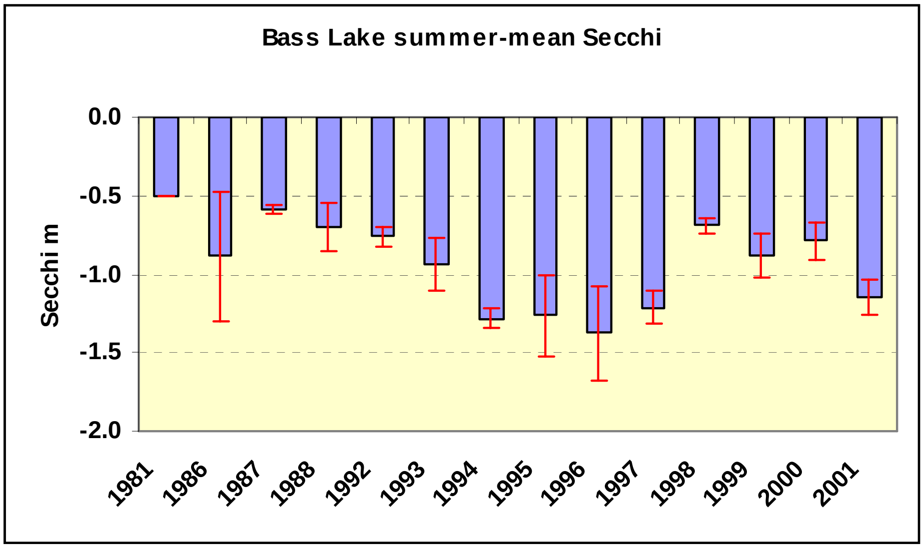
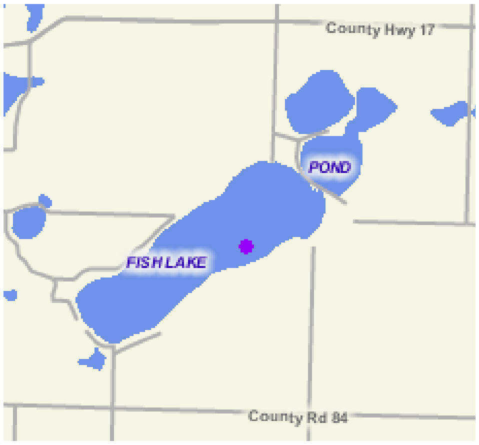


Figure 2. Bass Lake summer-mean Secchi.



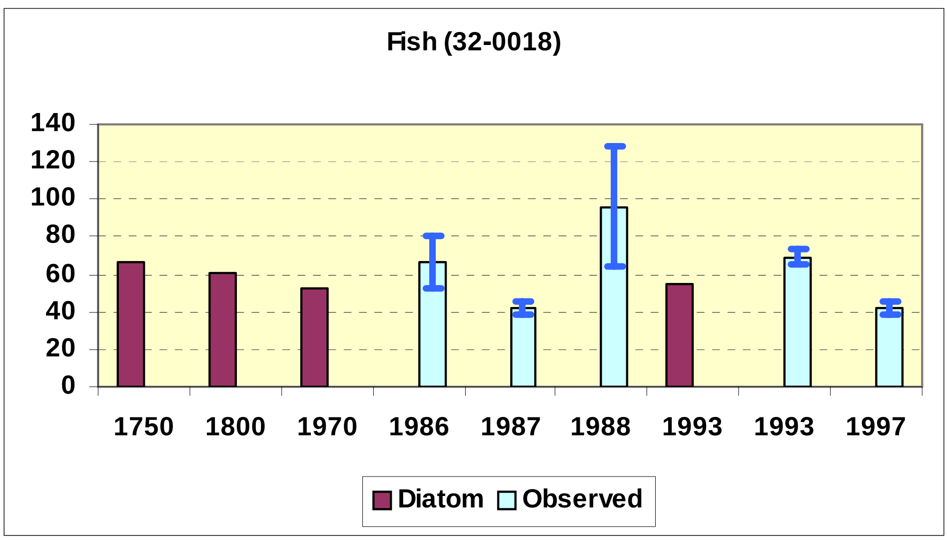
19. Fish Lake (32-0018) is located 2.5 miles southeast of Windom. Fish Lake is among the deepest lakes in the WCP ecoregion with a maximum depth of 26 feet. It has a moderate sized watershed relative to its size (5.4:1) that is dominated by agricultural uses. Its shoreline is fairly well developed, with about 88 homes based on a 1993 LAP study (Runholt, 1994). The remainder of the shoreline is forested. Fish Lake was one of the WCP reference lakes.



Pre-European TP values were quite stable and suggest eutrophic conditions for the lake. Modern day TP values suggest a decline in TP as compared to pre-European values. Modern day diatom-reconstructed TP values correspond quite well to monitored data from the 1980s to the 1990s and suggest no recent trends in trophic status, though year-to-year variability is evidenced.

Owing to its greater depth and small watershed TP concentrations in Fish Lake are among the lowest in the ecoregion (upper 5% based on Table 1) and hence it is reasonable to expect that the lake can attain (and maintain) better water quality than the majority of the shallow lakes with large highly agricultural watersheds that are typical for the ecoregion. Since Fish Lake's TP and chlorophyll-a concentrations are below WCP TMDL thresholds (for secondary aquatic recreational use) the lake should be a high priority for protection.

Figure 1. Fish Lake observed and diatom-reconstructed TP



20. Clear Lake (32-0022) is located in Jackson County, within the Missouri River Basin and the Western Corn Belt Plains (WCP) ecoregion. It was one of the reference lakes for this region. It has a maximum depth of 10 feet and a total surface area of 451 acres. It has a small watershed relative to its size (2.6:1 ratio). The majority of the land is cultivated based on 1968-69 data; however there may be some acreage enrolled in Conservation Reserve (CRP) in recent years.

Clear Lake is one of the more data-rich lakes in these two ecoregions. Distinct increases in transparency were evident for the period from 1975 to the mid 1980s (Figure 1). A decline in Secchi was noted in 1987 that would have coincided with the onset of the 1987-1989 drought period. Lake levels in Clear Lake were about four feet below the OHW in 1988-1990 and thereafter increased, peaking in 1993 (Heiskary et al. 2003). Transparency in 2002 was quite similar to measures from the mid 1970s. TP and chlorophyll-a are quite variable (Figure 2). Both 1988 (drought) and 1993 (high precipitation) were characterized by low TP and chlorophyll-a as compared to the other years (Figure 2). The lake was above the trophic status thresholds in four of six years. Based on the review of all data no long-term trend is evident rather there has been some cycling of trophic status that is driven in part by climatic cycles. An assessment of watershed land use changes and fishery management over time (both of which could influence P loading and trophic status) would provide some useful information for understanding the principal causes of variation in trophic status for this lake.

Figure 1. Clear Lake summer-mean Secchi.

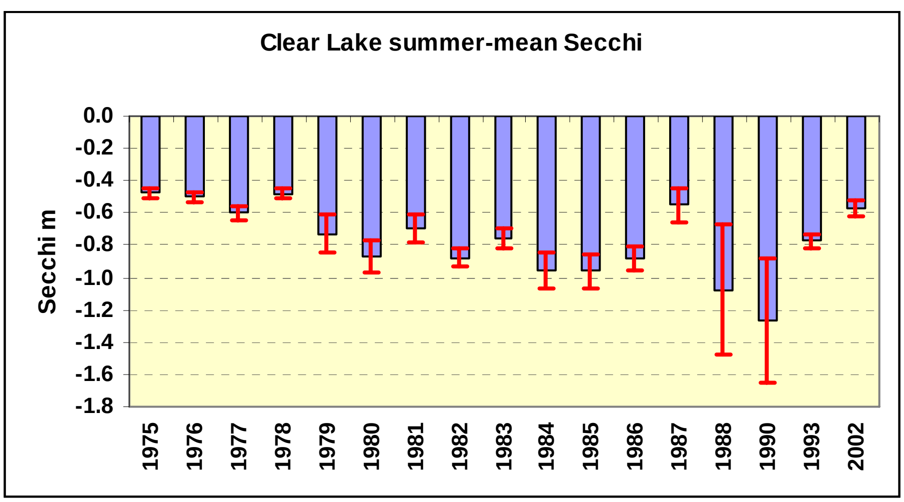
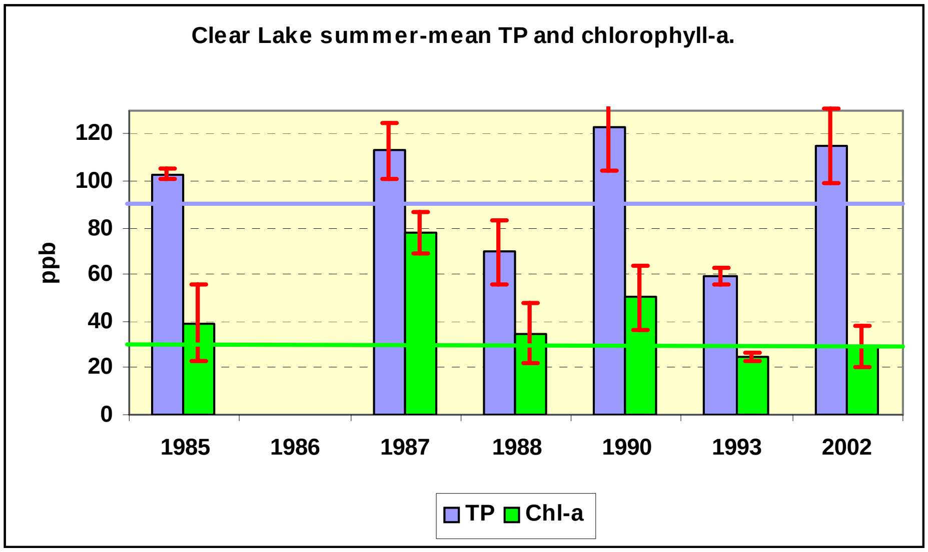
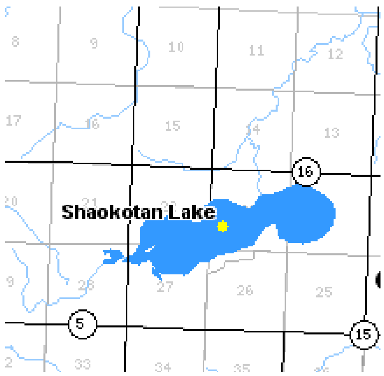


Figure 2. Clear Lake summer-mean TP. WCP thresholds noted.

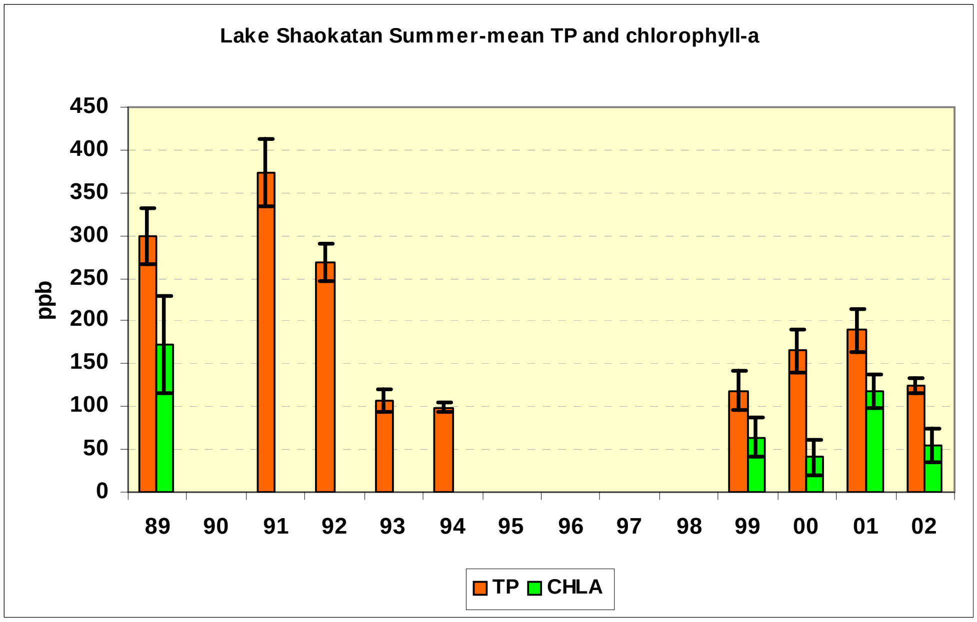


21. Lake Shaokotan (49-0089) is among the most studied lakes in southwest Minnesota. With a maximum depth of about 12 feet and a predominately agricultural watershed it is fairly typical of lakes in this ecoregion. The lake has a history of water quality problems including severe nuisance blue-green blooms, summer and winter anoxia, and periodic fish kills. These problems were the result of excessive nutrient loading to the lake. A detailed CWP Phase I diagnostic study was initiated in 1989 and restoration efforts were underway by 1991. This detailed monitoring allowed for the characterization of phosphorus exports for several subwatersheds. Subwatershed land uses ranged from relatively low intensity land uses, such as Conservation Reserve Program (CRP) acres to high intensity uses such as row crop cultivation and feedlots. Phase II implementation included rehabilitation of three animal feedlots, four wetland areas, and shoreline septic systems.



By 1994 significant reductions in in-lake P were realized with concentrations approaching the ecoregion-based P goal of 90 µg/L, in contrast to the 200 to 350 µg/L noted in previous summers (Fig. 1). This resulted in reductions in the frequency and severity of nuisance algal blooms. Transparency increased and anecdotal evidence in 1999 suggested macrophyte populations were increasing. However subsequent plant surveys in 2000 and 2002 found essentially no rooted plants. Water chemistry data indicated an increase in TP and chlorophyll-a from 1999 – 2001. This increase was largely attributed to an abandoned feedlot operation in the near shore area of the lake. Subsequent efforts by the Yellow Medicine Watershed District, Lincoln County and the local sportsman's group sought to address the problem. TP and chlorophyll-a remained above the trophic status thresholds for the NGP ecoregion and the lake was included on the 2002 303(d) list. Planning is now underway to initiate the TMDL study and this will hopefully complete the work that was initiated in the CWP and result in reduced (and more stable) TP and chlorophyll-a, which should lead to a reduction in the frequency of severe nuisance blue-green blooms that have characterized recent summers. This, combined with some improvement in transparency, may allow the return of macrophytes to the lake. At this point the eutrophication criteria (for secondary uses) appear to be very reasonable and achievable goals for the lake.

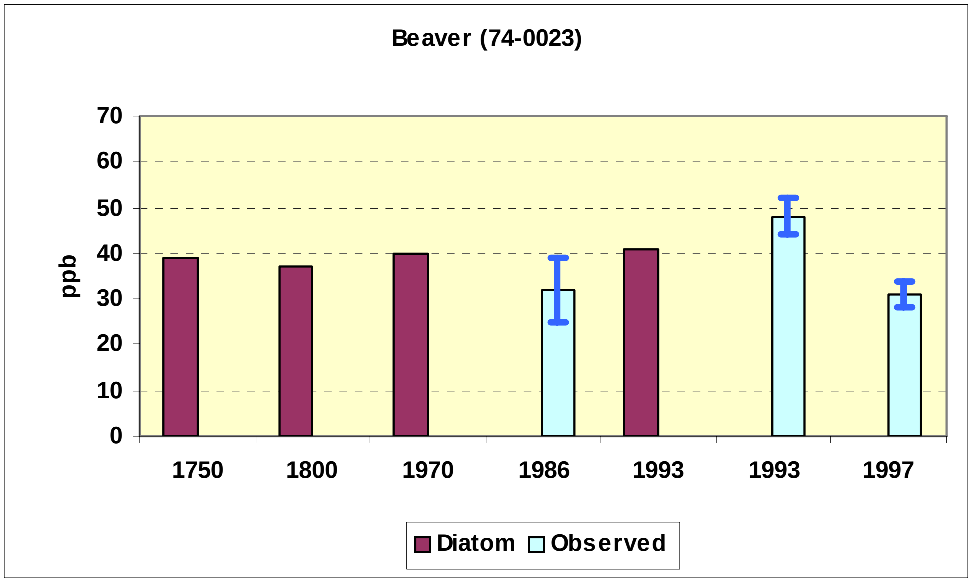
Figure 1. Lake Shaokatan summer-mean TP and chlorophyll-a



22. Beaver Lake (74-0023) is one of the few lakes in Steele County and with a maximum depth of 30 feet it is among the deepest lakes in the WCP (Table 1). It has a small watershed relative to its surface area (1.9:1) that is characterized by a mix of agricultural and forested land use. It has a highly developed shoreline. Beaver Lake was used as a reference lake and was the subject of a 1992 LAP study (Ganske, 1993). Water level fluctuations were common here as well as in other lakes with small watersheds and a system was developed to allow diversion of water from an adjacent watershed, though it was seldom used. There are no significant tributaries to the lake but there is a ditched inlet on the north side and about 11 drain tiles inlets enter the lake.

Pre-European TP values were quite stable and suggest mildly eutrophic conditions for the lake. Modern-day TP values did not exhibit any significant change from the pre-European values (Figure 1). There is good correspondence between observed TP values from 1986, 1993, 1997 and the diatom-inferred value for c. 1993. Curly-leaf pondweed is present in Beaver Lake and may contribute to variability in TP concentrations. The lake is well below TMDL TSI thresholds for the WCP and as such, protection of its currently good water quality should be a high priority.

Figure 1. Beaver Lake diatom-inferred and observed TP.



24. <u>Clear Lake (81-0014)</u> at Waseca is a moderate sized (652 acres) and relatively deep (maximum depth ~30 feet) lake in the WCP. It intermittently stratifies during the summer (Barten, 1983). This lake represents one of the first Clean Lakes projects in Minnesota and a valuable resource for this part of the state. Prior to restoration, summer total phosphorus averaged about 130 µg/l, chlorophyll-a values ranged between 20 to 60 µg/l throughout the summer, and Secchi transparency was commonly less than 1 m. Frequent and intense blue-green algal blooms were also common (Barten, 1983). These conditions tended to limit the recreational use of the lake.

After restoration, summer total phosphorus ranged from 85 to 100 µg/l and chlorophyll-a values were variable ranging from 5 to 35 µg/l in 1981 and 25 to 65 µg/l in 1982. Secchi transparency was generally unchanged over that period. Transparency measures in 1987, were on the 1.0 to 1.5 meter range and increased usage of the lake was noted and attributed to the “decrease in the intensity duration and frequency of algae blooms” after restoration (Barten, personal comm.),. At that time it was felt that, Clear Lake had probably achieved a reasonable TP concentration for a lake in the Western Corn Belt Plains (Figure 1).

In our most recent assessment (MPCA, 2004) it was evident that TP and chlorophyll-a had increased and were above the trophic status thresholds for lakes in the WCP ecoregion (Figure 1). This resulted in the lake being included on the 2004 303(d) list. Hopefully the TMDL study will identify sources of excess nutrient loading and projects to reduce the loading and return the lake to conditions that were achieved during the CLP implementation of the 1980s.

Figure 1. Clear Lake summer-mean TP and chlorophyll-a. WCP trophic status thresholds noted.