MINNESOTA

LAKE WATER QUALITY

ASSESSMENT REPORT:

DEVELOPING NUTRIENT CRITERIA

Third Edition

Minnesota Pollution Control Agency

September 2005

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

520 Lafayette Rd. N.; Saint Paul, MN 55155-4194; (651) 296-6300 (Voice); (651) 282-5332 (TTY); www.pca.state.mn.us
St. Paul • Brainerd • Detroit Lakes • Duluth • Mankato • Marshall • Rochester • Willmar

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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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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:

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

Considerations for Lake Water Quality Criteria Development – 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.

Lake Pollution Control Programs in Minnesota – 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.

Deriving Eutrophication Criteria by Ecoregion – 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

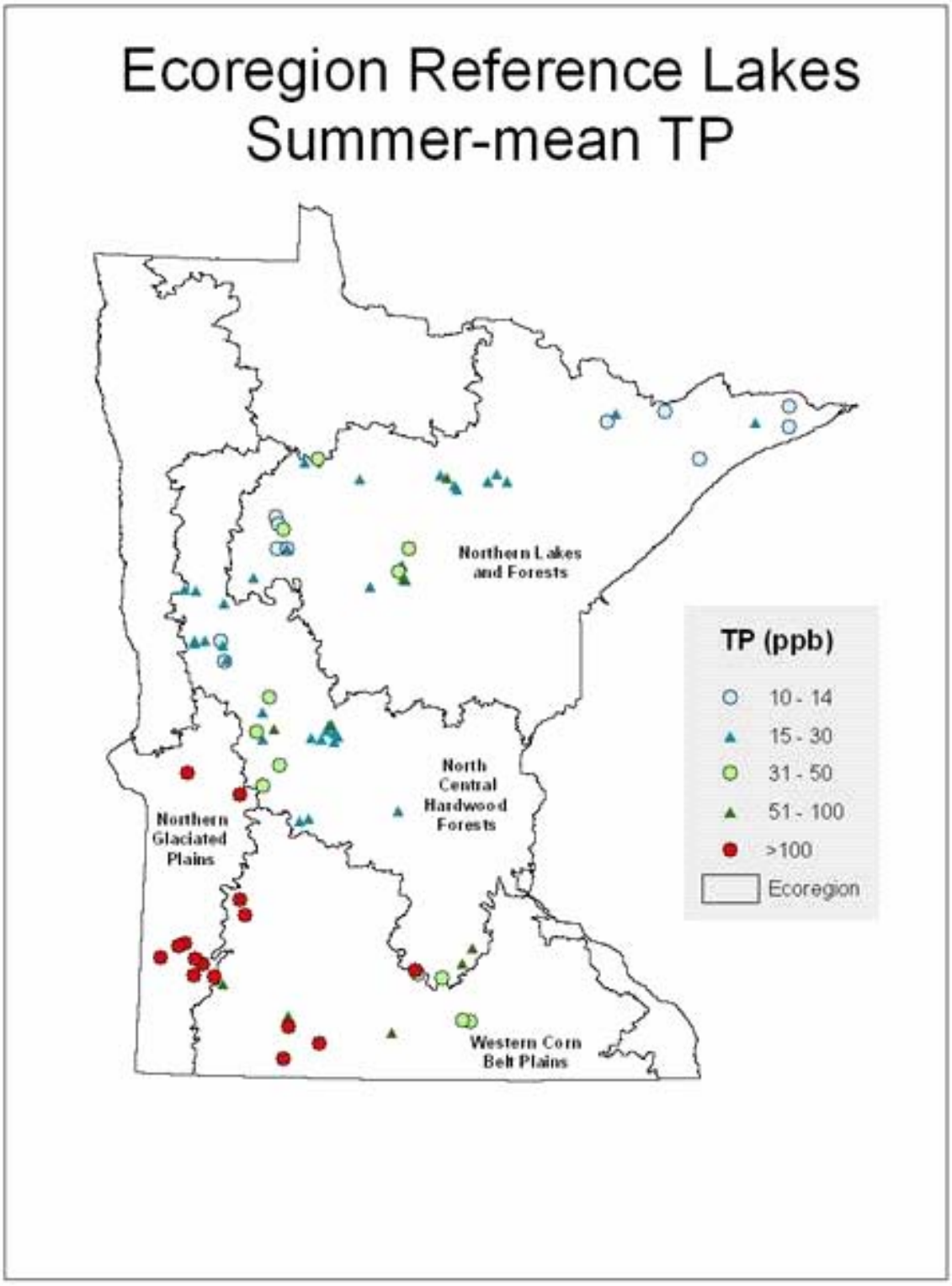
A second database is referred to as the reference lake database, which is a subset of the assessment database. The aquatic ecoregion framework (Omernik 1987) was used as a basis for selecting these lakes and analyzing the data. Though most of this work was conducted in the mid to late 1980's it is still 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

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 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

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.

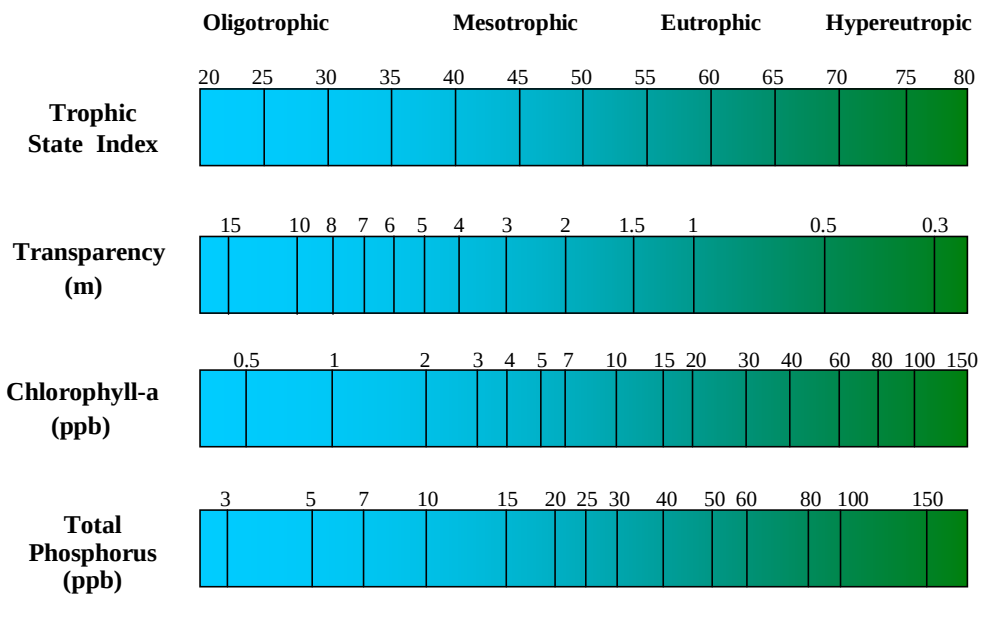
TSI 40 - 50 Water moderately clear, but increasing probability of anoxia in hypolimnion during summer.

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.

Trophic Status and Regional Water Quality Patterns for Minnesota Lakes

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).

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} Chl-a = 1.31 Log_{10} TP - 0.95 R^2 = 0.88; n = 108 (Eq. 1)
Log_{10} Secchi = -0.59 Log_{10} Chl-a + 0.89 R^2 = 0.85; n = 108 (Eq. 2)
Log_{10} Secchi = -0.81 Log_{10} TP + 1.51 R^2 = 0.81; n = 108 (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} Chl-a = 1.08 Log_{10} TP - 0.66 R^2 = 0.80; n = 31 (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).

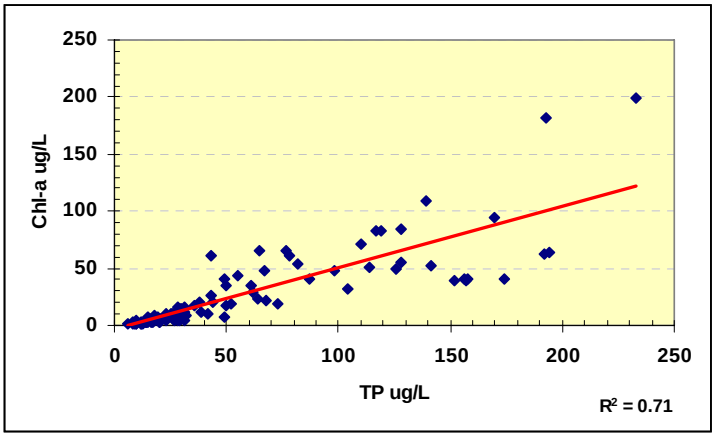
Smith (2003) notes that in general, comparative studies of freshwater eutrophication strongly suggest that efforts to control external nutrient loading to lakes will tend to achieve similar reductions in their average algal biomass. However, he notes that growing season average biomass (chlorophyll-a) 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).

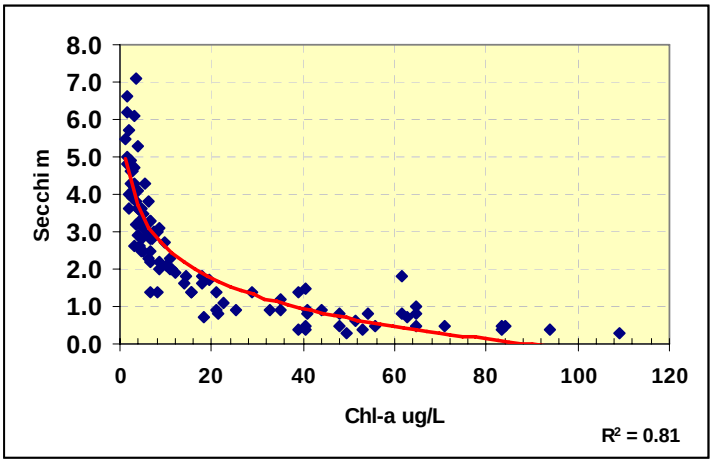
$$\text{Chl-a (max)} = 1.33\ \text{Chl-a (mean)} + 5.15 \qquad R^2 = 0.89;\ n = 108 \qquad \text{(Eq. 5)}$$

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.





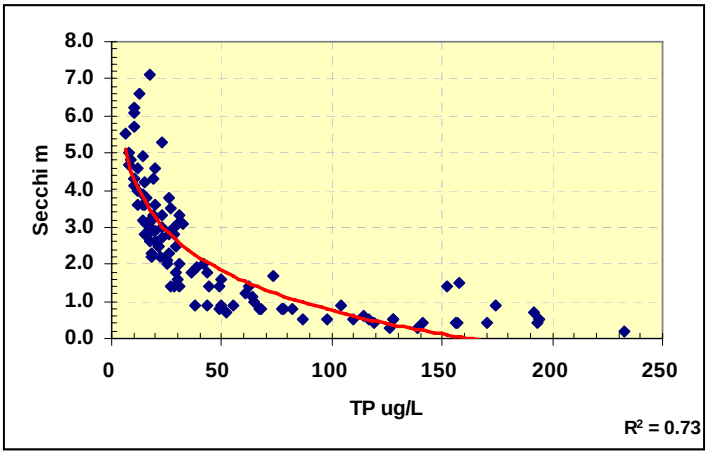
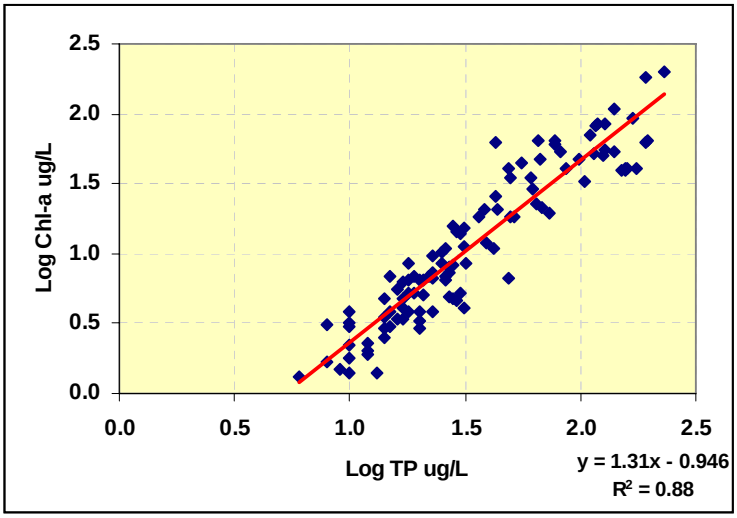
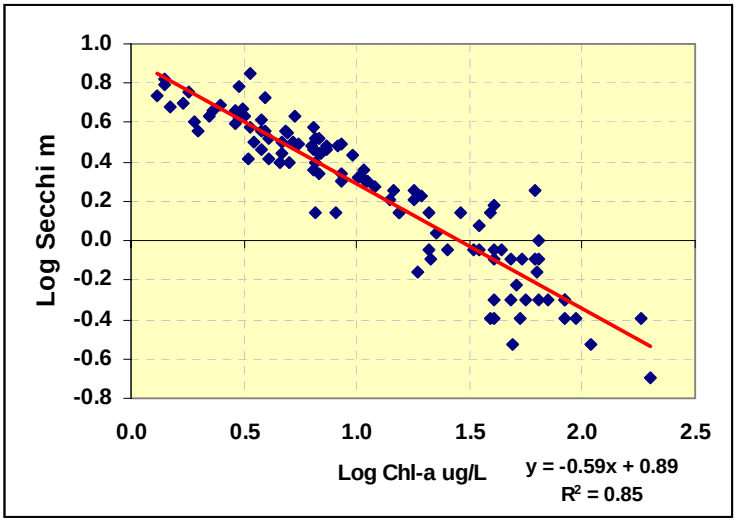
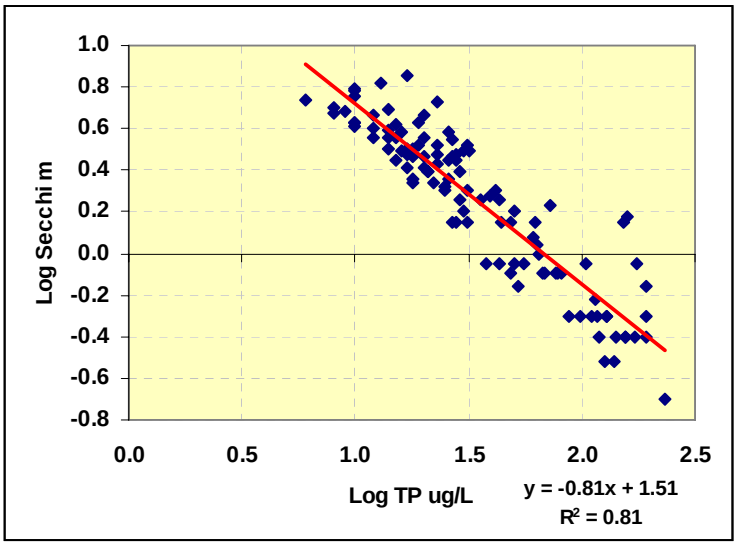


Figure 4b. Summer-mean TP, chlorophyll-a and Secchi log-log relationships. Based on reference lake data.







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.

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.

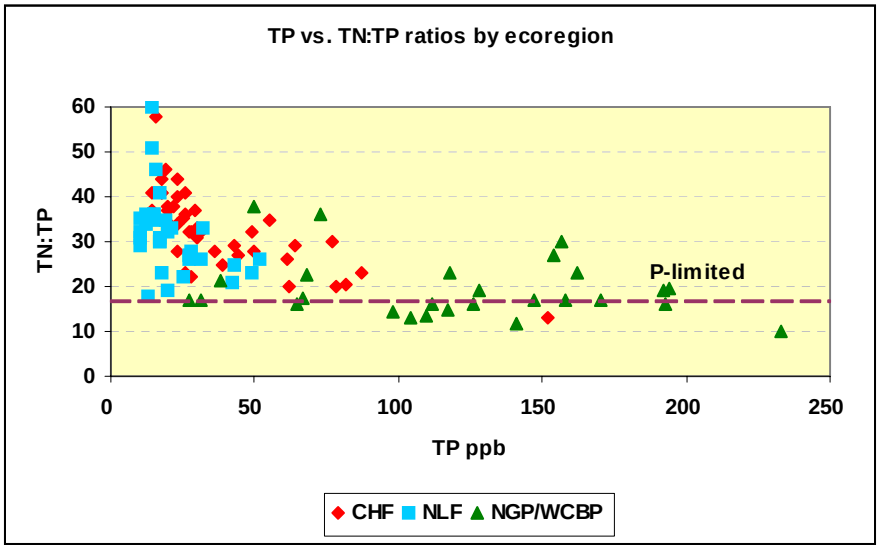
TKN is rather low and exhibits a small range in the NLF lakes (Table 2). The CHF, in contrast, is characterized by higher values and a larger range that can be attributed to the more nutrient-rich soils of this region and anthropogenic inputs from the increasingly agricultural or urban landuse that is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

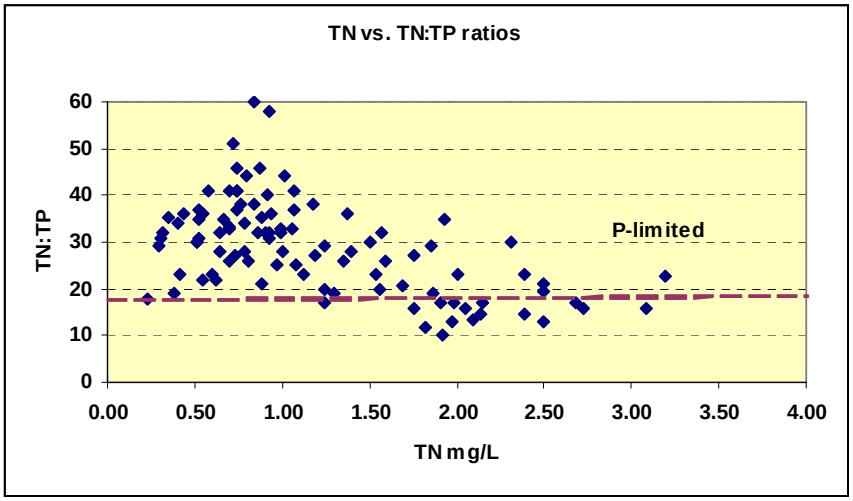
TN:TP ratios (TN calculated as sum of TKN + nitrate N) are often used as a basis for defining "P-limitation (when TN:TP >17:1)" vs. "N-limitation (when TN:TP<10:1)" with lakes in between being either P or N-limited (Smith, 1983). However, even when lakes are likely to be "N-limited" this generally means, 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.

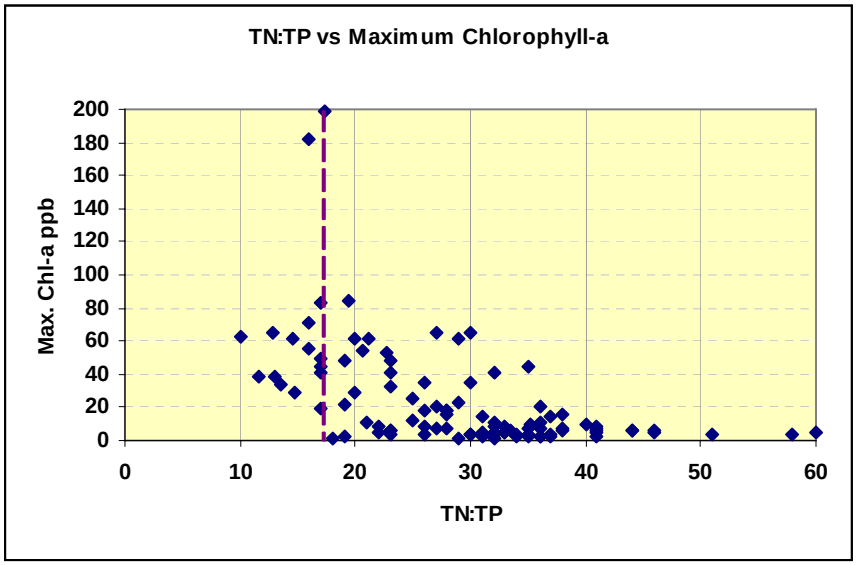
Based on the reference lake data NLF and CHF lakes will generally be considered P-limited, while WCP and NGP lakes may tend toward N-limitation (Table 2, Figure 7). TN:TP ratios (based on the reference lakes) are more highly correlated with TP ($R^2 = 0.63$) than to TN ($R^2 = 0.28$). Below TP concentrations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

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)

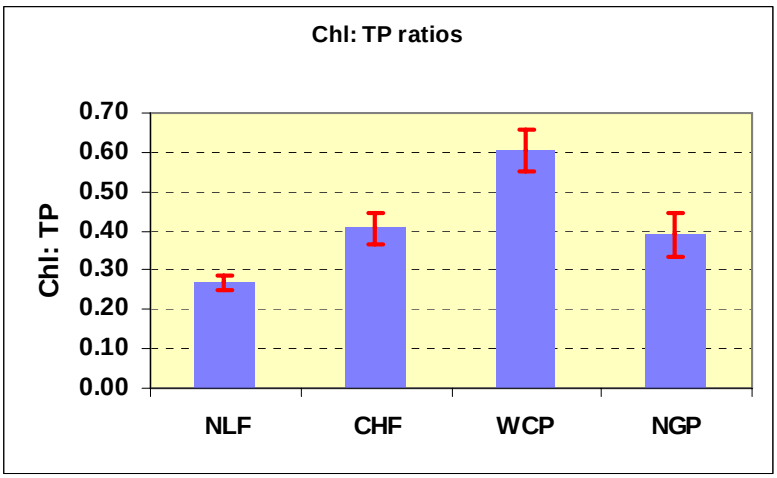
Some distinct regional patterns are evident in these data. In the NLF, differences in phosphorus concentration among the three mixing-status categories appears to be minimal up to the 75th percentile. In the CHF distinct differences in phosphorus concentrations exist among mixing-status categories over 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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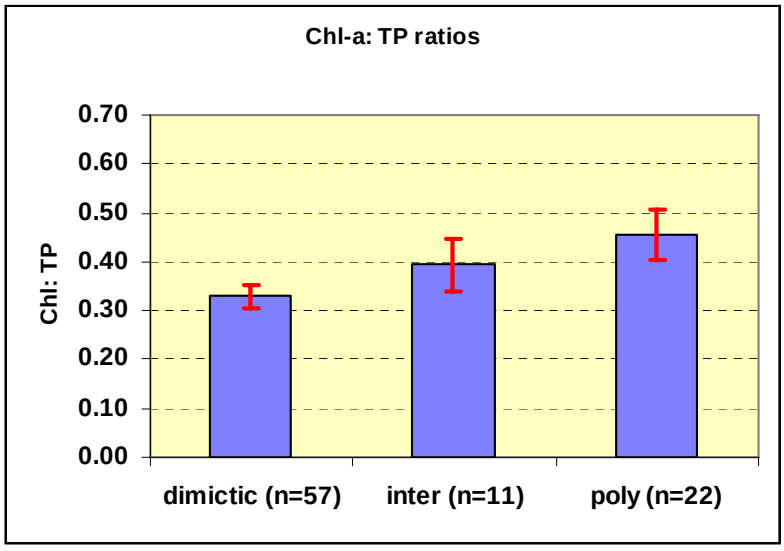

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Case Study 1.
Summer TP trends for a dimictic lake (Mary; 77-0019), intermittently mixing lake (Smith; 21-0016) and polymictic lake (Budd 46-0030)

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

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).

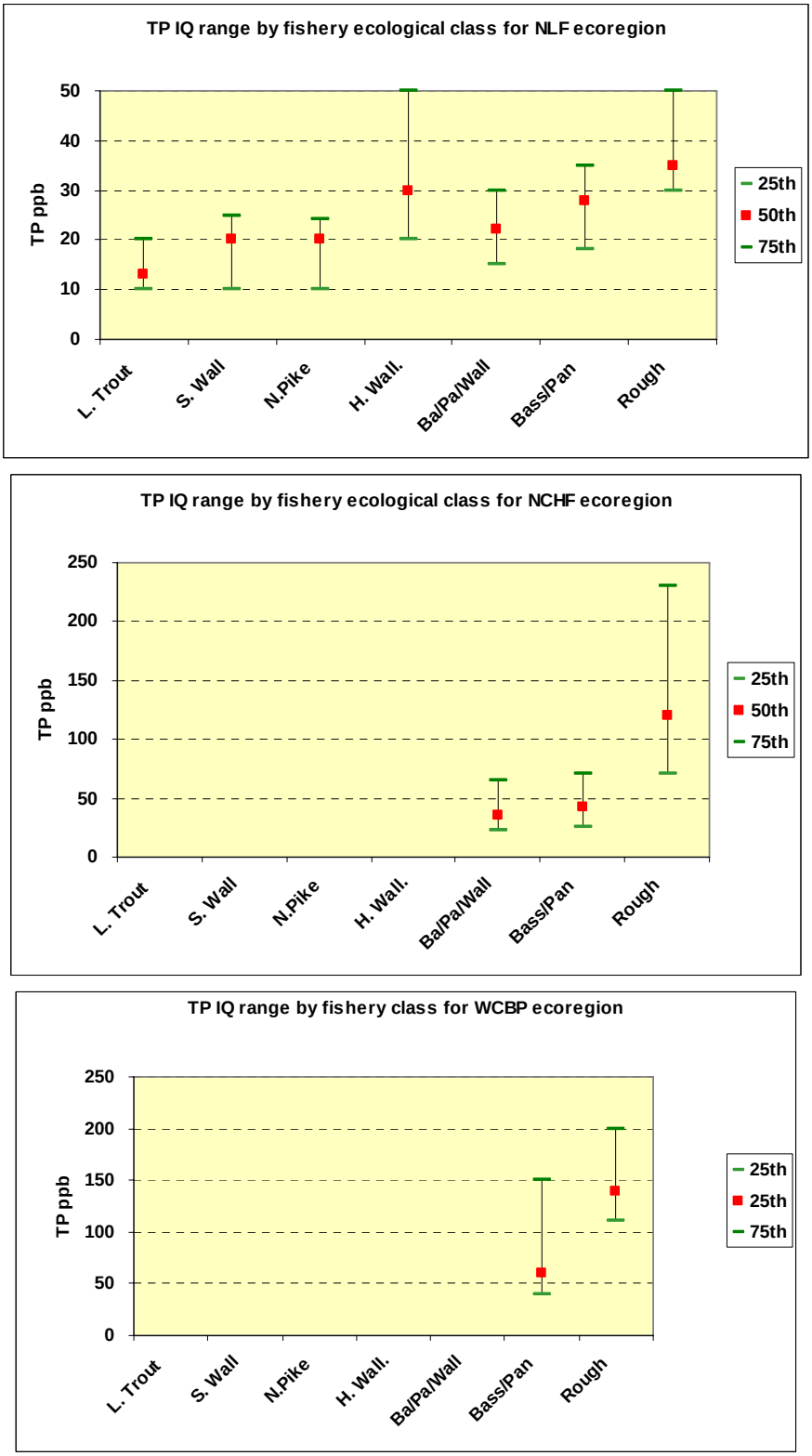
In an attempt to expand on Moyle's description of the geographic distribution of lake fisheries in Minnesota, 1,800 lakes were sorted by ecological and management types into the various ecoregions (Heiskary, et al., 1987). The lakes in the NLF are most typically classified as bass-panfish, bass-panfish-walleye, 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.

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).

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.

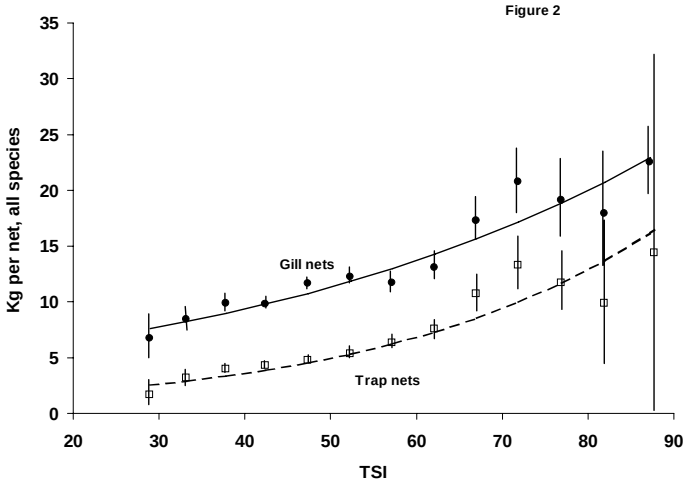
Perhaps the best fish indicators of water quality are two of the three bullhead species present in Minnesota (Schupp and Wilson, 1993). Yellow bullheads are found in the highest numbers in lakes with clear water. In contrast, black bullheads reach their highest abundance in very turbid, eutrophic waters (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.



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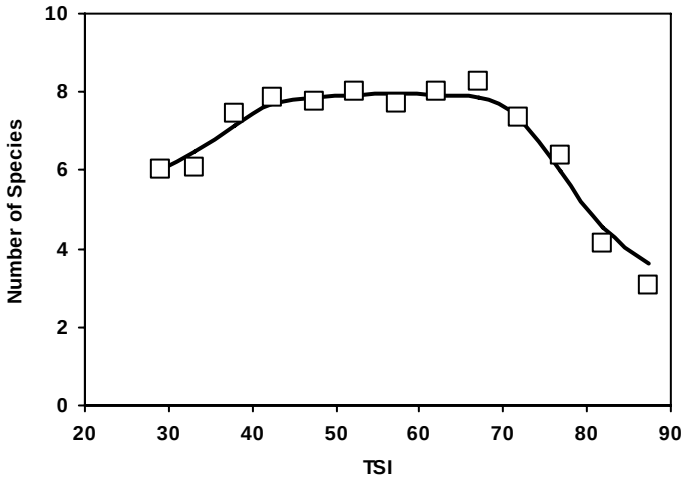

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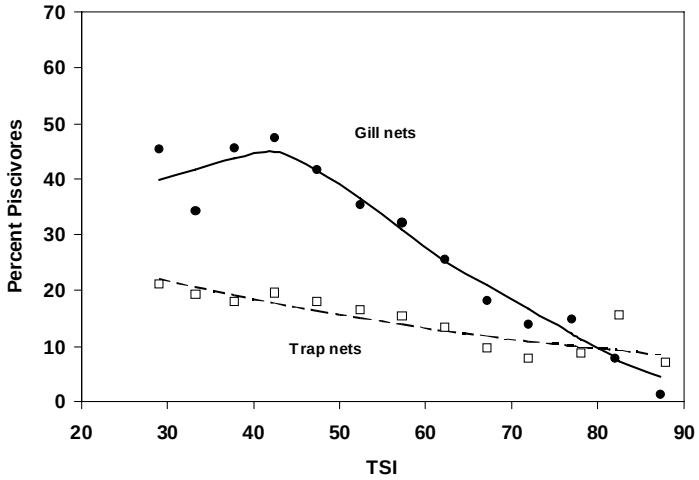
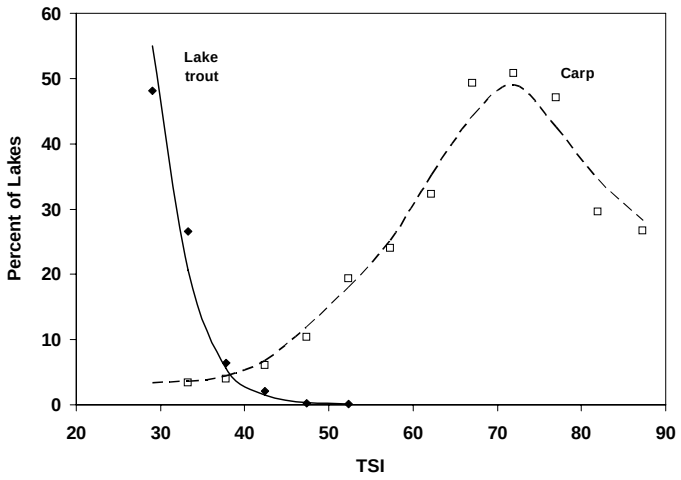
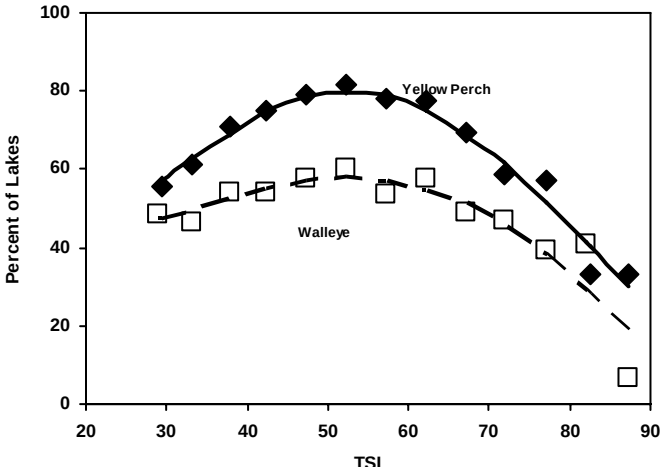
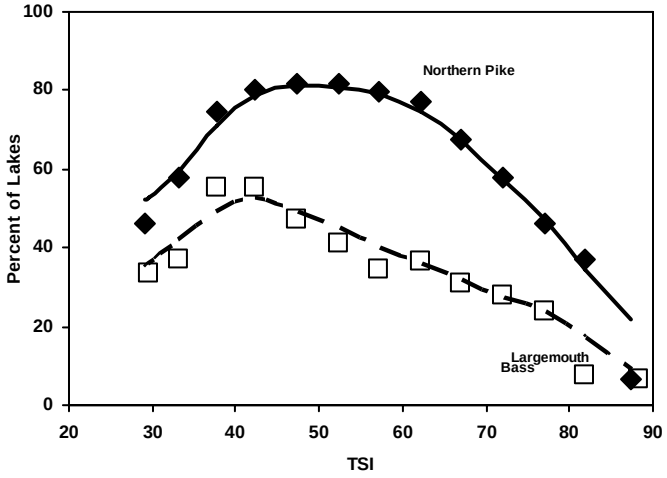
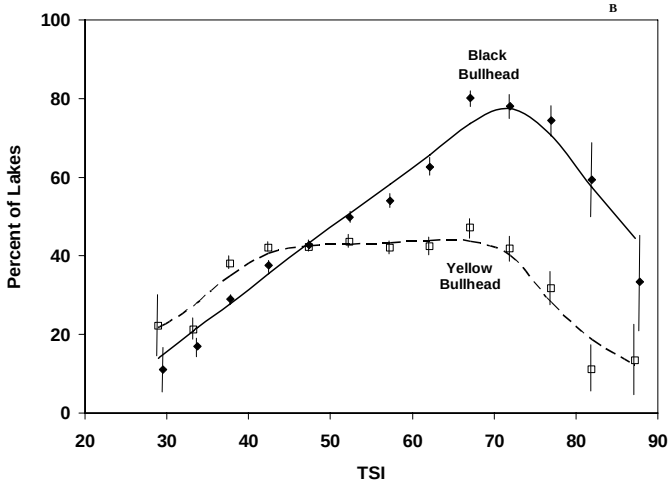


Figure 13 (cont.) d, e, f & g









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:

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

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.

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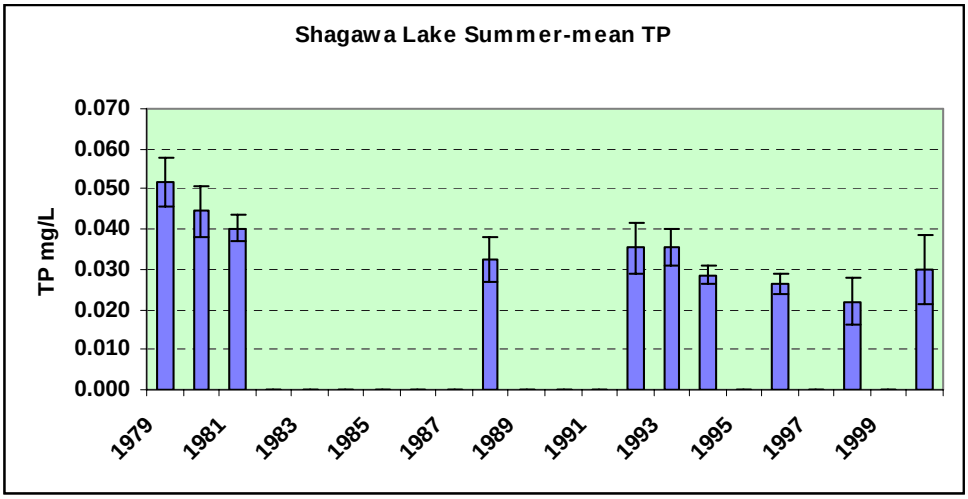
Case Study 4. Interrelatedness of lake morphometry, stratification stability and internal phosphorus loading: Shagawa Lake (69-0069).

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.



(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.

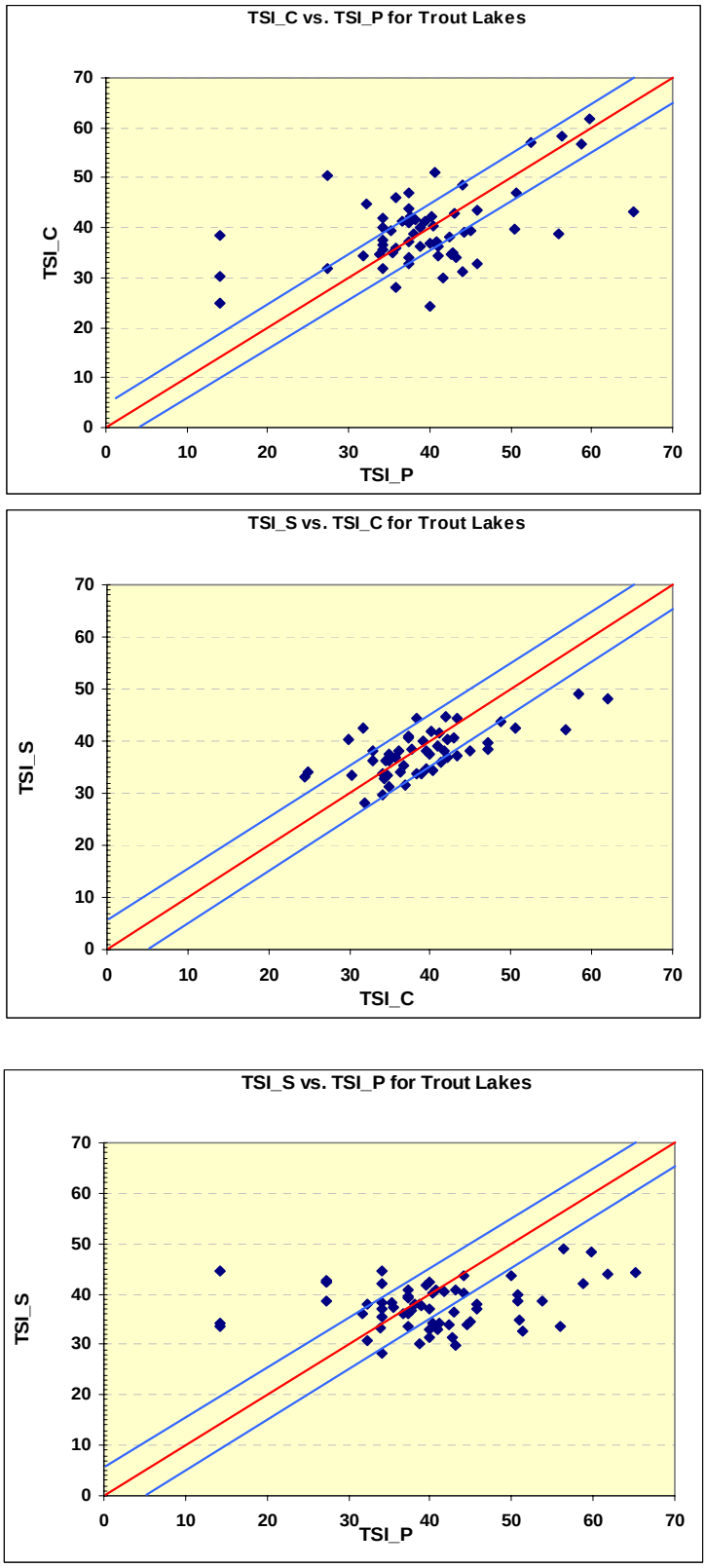
Maintaining adequate oxygen (~5-7 mg/L) in the hypolimnion is essential to maintaining a lake trout fishery as previously noted. Dillon et al. 2003 define the optimal habitat boundary for lake trout as the portion of the lake having more than 6 mg/L DO and temperature less than 10C – though they do acknowledge 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 20. Trophic state index comparison for trout lakes



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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 $\simeq$ 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."

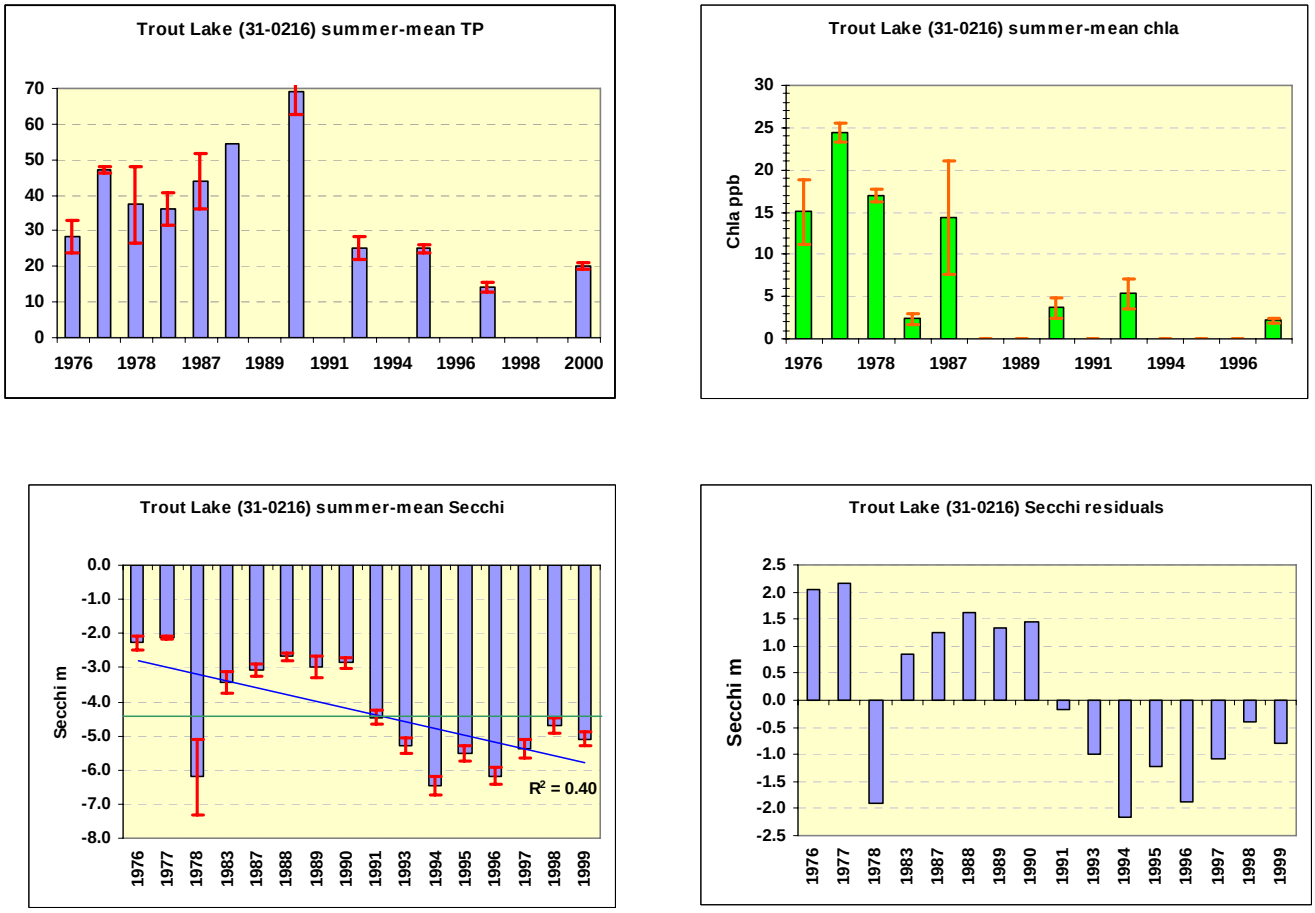
Significant declines in TP and chlorophyll-a are evident for Trout Lake based on MPCA and Itasca County data (Fig. 1 below). TP concentrations from 1993-2000 ranged from 15-25 µg/L, while summer-mean chlorophyll-a concentrations were in the 3-5 µg/L range. The decline in chlorophyll-a (algae) has resulted 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



"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).

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).

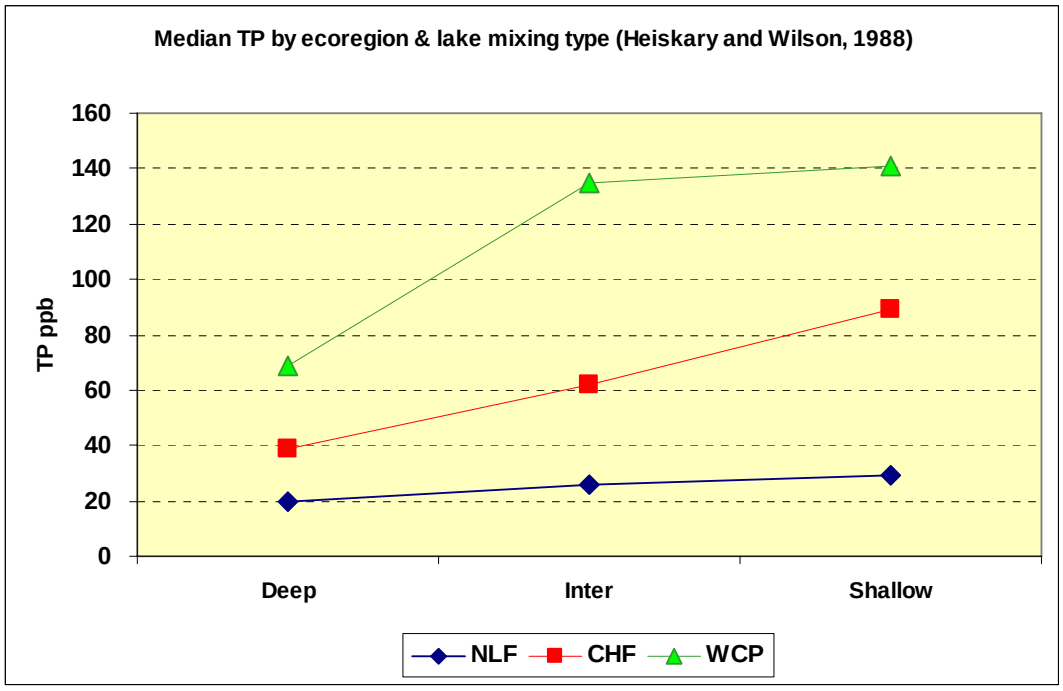
Work on shallow lakes was first initiated in 2002, with the sampling of several lakes in the WCP and NGP ecoregions. The purpose of that effort was to assess trends and condition in several of the reference lakes in these regions, supply baseline data for select lakes that lacked basic water quality 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.

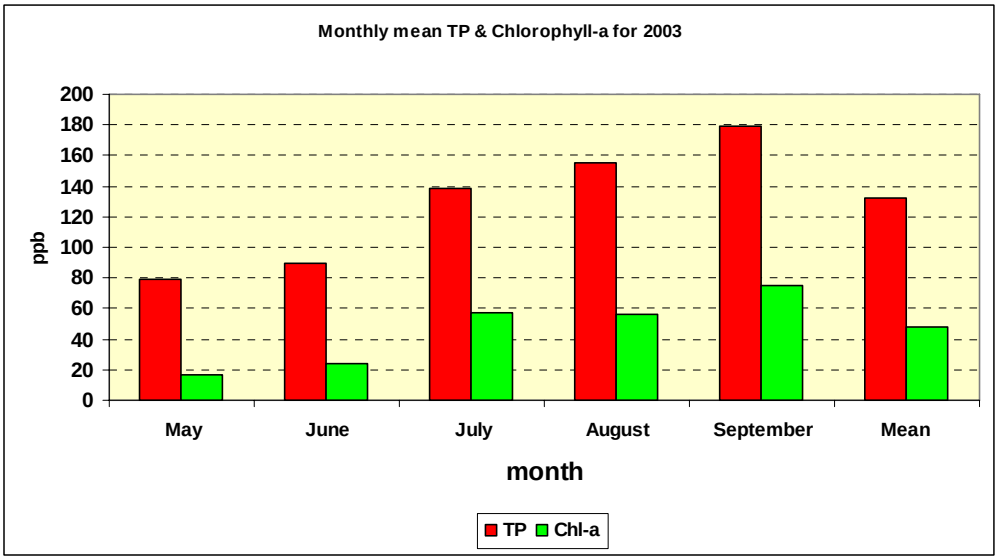
While excessive rooted macrophytes in a lake may impede recreation – macrophytes are essential to the overall ecology and stability of lakes – shallow lakes in particular. Madsen, (2001) notes that macrophytes help to slow water velocities increasing sedimentation, decreasing turbidity, thus increasing 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).