Lake eutrophication

July 2025

Development of eutrophication standards for northern lakes in Minnesota

MINNESOTA POLLUTION CONTROL AGENCY

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This report is available in alternative formats upon request, and online at www.pca.state.mn.us.

Document number: wq-bsm4-05

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Acronyms or abbreviations

a440	Absorptivity at 440 nm
CDF	Cumulative distribution function
CDOM	Colored dissolved organic matter
chl-*a*	Chlorophyll-*a*
CWA	Clean Water Act (33 U.S.C. § 1251 et seq.)
DLA	Driftless Area
DOC	Dissolved organic carbon
FQI	Floristic quality index
EPA	U.S. Environmental Protection Agency
EQuIS	Environmental Quality Information System
FIBI	Fish index of biological (biotic) integrity
HSPF	Hydrological Simulation Program - FORTRAN
HUC 8	8-digit hydrological unit code
IBI	Index of Biological (biotic) Integrity
mg/L	Milligrams per liter
Minn. R.	Minnesota Rules
Minn. Stat.	Minnesota Statutes
MN	Minnesota
MNDNR	Minnesota Department of Natural Resources
MPCA or Agency	Minnesota Pollution Control Agency
NCHF	North Central Hardwood Forest
NGP	Northern Glaciated Plains
NLA	National Lake Assessment
NLF	Northern lakes and forests ecoregion
NMW	Northern Minnesota Wetlands
PCU	Platinum-cobalt units
RRV	Red River Valley
TMDL	Total maximum daily load
TP	Total phosphorus
µg/L	Micrograms per liter
UMN	University of Minnesota
WCBP	Western Corn Belt Plains
WQS	Water Quality Standards
WRAPS	Watershed Restoration and Protection Strategy

Definitions

The following definitions of terms used in this document are based on standard use and are provided for the convenience of the reader. Unless otherwise specified, these definitions are specific to this document.

Aquatic biota: The aquatic community composed of game and nongame fish, minnows and other small fish, mollusks, insects, crustaceans, and other invertebrates, submerged or emergent rooted vegetation, suspended or floating algae, substrate-attached algae, microscopic organisms, and other aquatic-dependent organisms that require aquatic systems for food or to fulfill any part of their life cycle, such as amphibians and certain wildlife species. See Minn. R. 7050.0150, subp. 4.

Aquatic life use: A designated use that protects aquatic biota including fish, insects, mollusks, crustaceans, plants, microscopic organisms, and all other aquatic-dependent organisms. Attainment of aquatic life uses are measured directly in Minnesota using biological indices and biological criteria. Chemical and physical standards are also used to protect aquatic life uses.

Aquatic life use goals: A goal for the condition of aquatic biota; required by the Clean Water Act (CWA). Minimum aquatic life use goals are established using the CWA interim goal ("…water quality which provides for the protection and propagation of fish, shellfish, and wildlife…" CWA Section 101(a)(2) e). The objectives for these goals are established in Minnesota Rule using narrative standards, numeric standards, or both. Attainment of these goals is directly measured in Minnesota using biological indices and associated "Biological Criteria" or "Biocriteria."

Beneficial use: A designated use described under Minn. R. 7050.0140 and listed under Minn. R. 7050.0400 to Minn. R. 7050.0470 for each surface water or segment thereof, whether or not the use is being attained. The term "designated use" may be used interchangeably. See also "existing use."

Biological criteria or biocriteria: Specific quantitative measures of the structure and function of aquatic communities in a water body necessary to protect the designated aquatic life beneficial use. See Minn. R. 7050.0150, subp. 4.

Biological Integrity: The condition where "the biota is a balanced, integrated, adaptive system having a full range of ecosystem elements (genes, species, assemblages) and processes (mutation, demographics, biotic interactions, nutrient and energy dynamics, metapopulation dynamics) expected in areas with no or minimal human influence" (Karr 2000).

Criteria: Narrative descriptions or numerical values, which describe the chemical, physical, or biological conditions in a water body necessary to protect designated uses. See also the definitions for "biological criteria/biocriteria" and "standard".

Designated use: See "beneficial use."

Dimictic lake: Lakes which mix twice a year in the spring and fall and are stratified during the summer and winter. Compared to polymictic lakes, dimictic lakes tend to be deeper with the littoral zone comprising a lower proportion of the total area.

Existing use: Those uses actually attained in a surface water on or after November 28, 1975. See Minn. R. 7050.0255, subp. 4.

Index of biological integrity or index of biotic integrity (IBI): An index developed by measuring attributes of an aquatic community representing the health of that community and that change in quantifiable and predictable ways in response to human disturbance.

Polymictic lake: Lakes with frequent mixing of the water column during the ice-free period. In general, these are shallow and are largely consistent with Minnesota's shallow lake definition in Minn. R. 7050.0150, subp. 4.

Shallow lake: Shallow lakes are defined in Minn. R. 7050.0150, subp. 4, Item as "*an enclosed basin filled or partially filled with standing fresh water with a maximum depth of 15 feet or less or with 80 percent or more of the lake area shallow enough to support emergent and submerged rooted aquatic plants (the littoral zone). It is uncommon for shallow lakes to thermally stratify during the summer.*"

Standard: Regulatory limits on a particular pollutant, or a description of the condition of a water body, which supports or protects the beneficial use or uses. Standards may be narrative or numeric and are commonly expressed as a chemical concentration, a physical parameter, or a biological assemblage endpoint. See also the definitions for "biological criteria/biocriteria" and "criteria".

Stressors: Physical, chemical, and biological factors that can adversely affect aquatic organisms. The effect of stressors is apparent in biological responses because stressor conditions are outside the conditions for which an organism is adapted. This leads to changes in the fitness of organisms and changes in the composition of organisms found in aquatic communities. Under the effect of stressors, the normal functioning of organisms is interrupted (e.g., increased metabolism, interruption of behavior) which results in negative impacts such as decreased fitness, reduced growth, increased disease prevalence, interruption of reproductive behavior, increased emigration, and increased mortality. Examples of stressors in aquatic systems are low levels of dissolved oxygen, suspended sediments, toxic pollutants, habitat alteration, altered hydrology, and reduced connectivity.

Water Quality Standards (WQS): A law or regulation that consists of the beneficial use or uses of a water body, the narrative or numerical WQS that are necessary to protect the use or uses of that water body, and antidegradation.

Overview

A major threat to Minnesota's lakes is cultural eutrophication which degrades the beneficial uses provided by lakes including recreation (swimming, boating), aquatic life (fishing, wildlife), and aesthetics. To protect these important resources, the Minnesota Pollution Control Agency (MPCA) has been a national leader in lake nutrient criteria development. In 2008, eutrophication criteria for Minnesota lakes were promulgated into water quality standards (Minn. R. 7050.0222). Minnesota's abundant lake resources are naturally diverse due to differences in geography, geology, and land cover, which can affect the types of beneficial uses supported in lakes and the specific criteria needed to protect those uses. To ensure that appropriate and protective standards are applied to lakes, Minnesota's lake eutrophication standards are subdivided by region, lake stratification type, and the most sensitive designated beneficial use (e.g., aquatic recreation, coldwater fisheries, or aquatic life) supported by the lake.

The implementation of Minnesota's lake eutrophication standards has been effective for determining the status of lakes and driving restoration or protection strategies when needed. However, after over a decade of lake assessments, a subset of relatively un-impacted shallow lakes within the Northern Lakes and Forests (NLF) ecoregion have been identified that exceed Class 2B (cool and warm water) water quality standards (WQS). Unlike Minnesota's other lake nutrient regions (i.e., Central and South), there is currently no distinction in rule between WQS for dimictic and polymictic lakes[1] in the North region. A single standard was assigned to all northern, cool/warm water lakes because, during technical development of Minnesota's lake eutrophication standards, strong differences were not identified between dimictic and polymictic lakes in the NLF ecoregion. However, this was partly due to a small sample size of polymictic NLF lakes used for the development of the 2008 lake eutrophication standards. As a result, the adopted standards are largely based on a dataset of dimictic, NLF lakes. This fact and the determination that a subset of undisturbed, polymictic lakes do not meet the current standards indicate that separate standards may be needed for polymictic and dimictic lakes in the North region. If these two lake types have naturally different trophic states, but standards do not account for these differences, lakes with trophic states at or near natural-background conditions will be listed as not meeting beneficial use goals. As a result, the MPCA initiated a detailed analysis of lake eutrophication criteria for North region lakes to determine if different standards are needed for dimictic and polymictic lakes to ensure appropriate assessment and management outcomes.

Considering that the NLF ecoregion contains approximately 47% of all lakes in Minnesota, and within these lakes, 28% are shallow lakes (depths < 5m) (Olmanson et al. 2014), it is important that appropriate WQS are assigned to these waters. This study was focused on lakes in the NLF and Northern Minnesota Wetlands (NMW) ecoregions (herein referred to as the "North region" or "northern lakes") to determine if dimictic and polymictic lakes should be assigned different eutrophication standards to protect cool and warm water aquatic life and recreation (Class 2B). The questions this study addressed were:

- Do water quality and the relationships between eutrophication parameters differ between polymictic and dimictic lakes within the North region? If differences in trophic condition between dimictic and polymictic northern lakes are present, are these the result of natural characteristics or are there differences in cultural eutrophication impacts between these lake types?
- Due to the impact of colored dissolved organic matter (CDOM) on Secchi depth, is the use of Secchi depth a reasonable WQS parameter in northern lake eutrophication standards? If CDOM

[1] In Minnesota's current rules (Minn. R. 7050.0222), polymictic lakes are referred to as "shallow lakes" and dimictic lakes are referred to as "lakes."

introduces undesirable levels of error to Secchi depth assessments, can this effect be mitigated to allow for the use of Secchi depth data?

- What trophic conditions are protective of aquatic life and recreation beneficial uses in dimictic and polymictic northern lakes? If these differ between polymictic and dimictic lakes in the North region, which standards should be revised to protect beneficial uses in northern lakes?

An analysis of beneficial use indicators demonstrated that different eutrophication standards are needed for dimictic and polymictic lakes. Using thresholds for the most sensitive indicators, protective lake eutrophication standards were determined for each lake type (Table 1). The currently adopted lake eutrophication standard for chlorophyll-*a* (chl-*a*; 9 µg/L) was determined to be too stringent for polymictic lakes but was appropriate for dimictic lakes. However, updated total phosphorus (TP) and Secchi depth models indicated that changes to criteria for these parameters is appropriate for both dimictic and polymictic lakes (Table 1). These values are based on long-term summer means, per MPCA's assessment methodology (MPCA 2021). For designation of impairment, both TP and chl-*a* or Secchi depth must exceed the criterion. Secchi depth may be considered an inappropriate indicator of impairment due to naturally high CDOM. To address this, color and absorptivity at 440 nm (a_{440}) thresholds are provided to identify when Secchi depth is not a reliable indicator of chl-*a* in northern lakes.

Table 1. Current and draft lake eutrophication criteria for northern lakes.

Lake Type	Total phosphorus (µg/L)	Chlorophyll-*a* (µg/L)	Secchi depth (m)*
Current criteria			
Northern lakes	30	9	2.0
Draft criteria			
Northern polymictic lakes	32	18	0.9
Northern dimictic lakes	20	9	1.8

*lakes with color >73 platinum-cobalt units (PCU) or a_{440} >4 m^{-1} should not be assessed using Secchi depth and lakes with color >25 PCU or a_{440} >1.4 m^{-1} should be reviewed to determine the effect of CDOM on water transparency.

To determine these standards, the lines of evidence used in this study included:

- A compilation of northern lake data including water chemistry (TP, chl-*a*, Secchi depth, color, a_{440}, and dissolved organic carbon [DOC]), land use/land cover, paleolimnology, aquatic macrophyte, fish, and recreation use survey data.
- An analysis of the modern (1990-2020) status of water quality in dimictic and polymictic northern lakes. A focus of this assessment was to determine if water quality differed between dimictic and polymictic lakes in the northern region. This analysis demonstrated that there is a significant difference between dimictic and polymictic northern lakes. All eutrophication parameters (i.e., TP, chl-*a*, and Secchi depth) indicated that at a population level, polymictic lakes are more eutrophic compared to dimictic lakes. Measures of CDOM (i.e., color and a_{440}) and DOC were also higher in polymictic lakes. These results provide support for the need for different eutrophication standards between dimictic and polymictic northern lakes.
- An analysis of water quality relationships among TP, chl-*a*, and Secchi depth were compared to those developed as part of the original lake eutrophication standards (Heiskary and Wilson 2005) to determine if the new, larger dataset resulted in different relationships. The relationship between TP and chl-*a* was similar for dimictic and polymictic northern lakes and was also similar to the reference lakes used in the 2005 study. The relationship between chl-*a* and Secchi depth did indicate a difference between dimictic and polymictic lakes and the 2005 reference lakes.

Secchi depth was lower for polymictic northern lakes at similar chl-*a* concentrations indicating that there are other factors besides chl-*a* impacting clarity in these lakes.

- Colored dissolved organic matter (CDOM) is a largely natural component of freshwaters that can affect water clarity. The impact of CDOM on Secchi depth was analyzed because Secchi depth is affected by non-algal factors such as CDOM (Brezonik et al. 2019). As expected, CDOM reduced Secchi depth and negatively impacted a lake's ability attain Secchi depth standards, even in the absence of high levels of algae. Secchi depth standard of 2.0 m which demonstrated that a subset of lakes should not be assessed using Secchi depth.
- A reference condition analysis was performed to determine if water quality (i.e., TP, chl-*a*, Secchi depth, and color) for dimictic and polymictic northern lakes with minimal anthropogenic impact (i.e., watershed disturbance < 25%) differ and to identify natural water quality conditions for these lakes. Watershed disturbance for dimictic and polymictic lakes was similar, but the reference condition analysis indicated that dimictic and polymictic northern lakes were different with naturally higher eutrophication measures in polymictic lakes.
- Available paleolimnology data were reviewed and analyzed to determine if natural water quality differed between dimictic and polymictic lakes in the northern region and if water quality in these lakes is different from current conditions. This analysis demonstrated that under natural or background conditions, TP in dimictic and polymictic northern lakes are different with higher concentrations in polymictic lakes. In particular, TP and therefore trophic condition in many polymictic lakes is higher than current standards. Analysis of paleolimnological data also demonstrated that TP concentrations in most northern lakes are at or near natural or background levels.
- Protective chl-*a* thresholds were determined for three beneficial use endpoints including aquatic macrophytes, fish, and recreation uses.
    - **Macrophytes**: Analyses of macrophyte indices (taxa richness and floristic quality index [Radomski and Perleberg 2012]) demonstrated a strong response of macrophytes to increasing eutrophication. Based on a logistic regression, it was determined that chl-*a* thresholds of 13 and 18 µg/L are needed to protect macrophytes in dimictic and polymictic northern lakes, respectively. In addition, analyses indicated that the goals to protect aquatic macrophyte communities are also sufficient to protect wild rice from reduced transparency caused by suspended algae.
    - **Fish**: Analyses using fish index of biological integrity (FIBI) data demonstrated a strong relationship between chl-*a* and attainment of fish goals. Based on a logistic regression, it was determined that chl-*a* thresholds of 9 and 18 µg/L are needed to protect cool/warm water fish communities in dimictic and polymictic northern lakes, respectively.
    - **Recreational suitability:** Protective levels of chl-*a* for recreation uses were assessed using recreation survey endpoints. Recreational suitability use surveys were assessed against chl-*a* which indicated that in northern lakes, chl-*a* does impact recreation suitability scores and that these relationships differed between dimictic and polymictic lakes. To protect against high algal events that result in conditions that impair recreation in northern polymictic lakes, chl-*a* should be below 13 µg/L for dimictic lakes and below 39 µg/L for polymictic lakes. The recreation beneficial uses protected by these thresholds differ between the lake types with primary contact a driver of the standards for dimictic lakes and secondary contact in polymictic lakes.
- Based on the three beneficial use endpoints (2 aquatic life and 1 recreation), the most sensitive endpoint for polymictic lakes were macrophytes and fish (chl-*a* = 18 µg/L) and for dimictic lakes the most sensitive endpoint was cool/warm water fish (chl-*a* = 9 µg/L).

- Statewide quantile regression models were developed for TP – chl-*a* and chl-*a* – Secchi depth to model TP and Secchi depth criteria from the chl-*a* thresholds derived from analyses of beneficial use endpoints (Table 1).
- Additional details provided in this document include an overview of other relevant lake eutrophication criteria and details regarding the implementation of Minnesota's lake eutrophication standards.

The draft eutrophication standards for northern polymictic and dimictic lakes are based on a robust dataset of polymictic and dimictic lakes, which will protect applicable beneficial uses in these habitats. The refinement of the current eutrophication standards for northern polymictic lakes means that Minnesota's resources can be allocated appropriately to lakes in need of water quality restoration and protection. These proposed criteria can be refined pending additional data and research, including the potential to refine the natural background review process for eutrophication impairments in lakes within the northern ecoregion.

Introduction

To protect and manage Minnesota's important and diverse lake habitat resources, the MPCA is a national leader in lake nutrient criteria development. Existing MPCA research and analyses of lake water quality datasets have been previously summarized and reported elsewhere (e.g., Heiskary et. al. 1987, Heiskary and Walker 1988, Heiskary and Wilson 1989, Heiskary and Wilson 2005, and Heiskary and Wilson 2008), but will be briefly summarized here. In 2008 using a "weight of evidence" approach, eutrophication criteria for Minnesota lakes were promulgated into water quality standards based on region, lake type (lake[2] or shallow lake), and most sensitive designated use (e.g., aquatic recreation or coldwater fisheries; Table 2). The "weight of evidence" approach used to develop regional lake nutrient criteria was defined as the "collective summary of scientific information pertaining to identifiable lake response thresholds, linked with the most sensitive beneficial uses, attuned to regional and lake-type distinctions, and coupled with user perceptions of water quality" (Heiskary and Wilson 2008). Implementation of these standards for assessment include a review of a lake-specific data from the most recent 10 years. A minimum of 8 paired TP, chl-*a*, and Secchi depth measurements from two summers (June through September) are required for assessment. Lakes where TP and at least one of the response variables (chl-*a* or Secchi depth) exceed the standards are considered impaired. Lakes where all parameters are better than the standards are assessed as fully supporting lake eutrophication goals (MPCA 2021). Through this approach, Minnesota has successfully implemented lake eutrophication standards and demonstrated this framework to be a useful tool for the protection and restoration of lakes.

Although successful, more than a decade of experience implementing Minnesota's lake eutrophication standards has identified an aspect of these standards which needs revision. Specifically, numerous shallow, northern lakes have been identified which exceed lake eutrophication standards, but which have low levels of watershed disturbance or other indicators (e.g., paleolimnological data) demonstrating that these lakes are near a natural trophic state. For example, Bluebill Lake in Itasca County (Figure 1) is a polymictic lake with limited development around the lake (<3% developed land use). The lake drains a large wetland and forested watershed typical of the North region landscape. Recent assessment-level data in Bluebill Lake indicates that all three eutrophication parameters are exceeding Class 2B water quality standards (TP =34 µg/L, chl-*a* = 13 µg/L, Secchi depth = 1.1 m). Given the low potential of anthropogenic sources of TP to this water, it may not be reasonable to consider this lake as impaired as the current trophic status appears natural. Due to this example and other similar

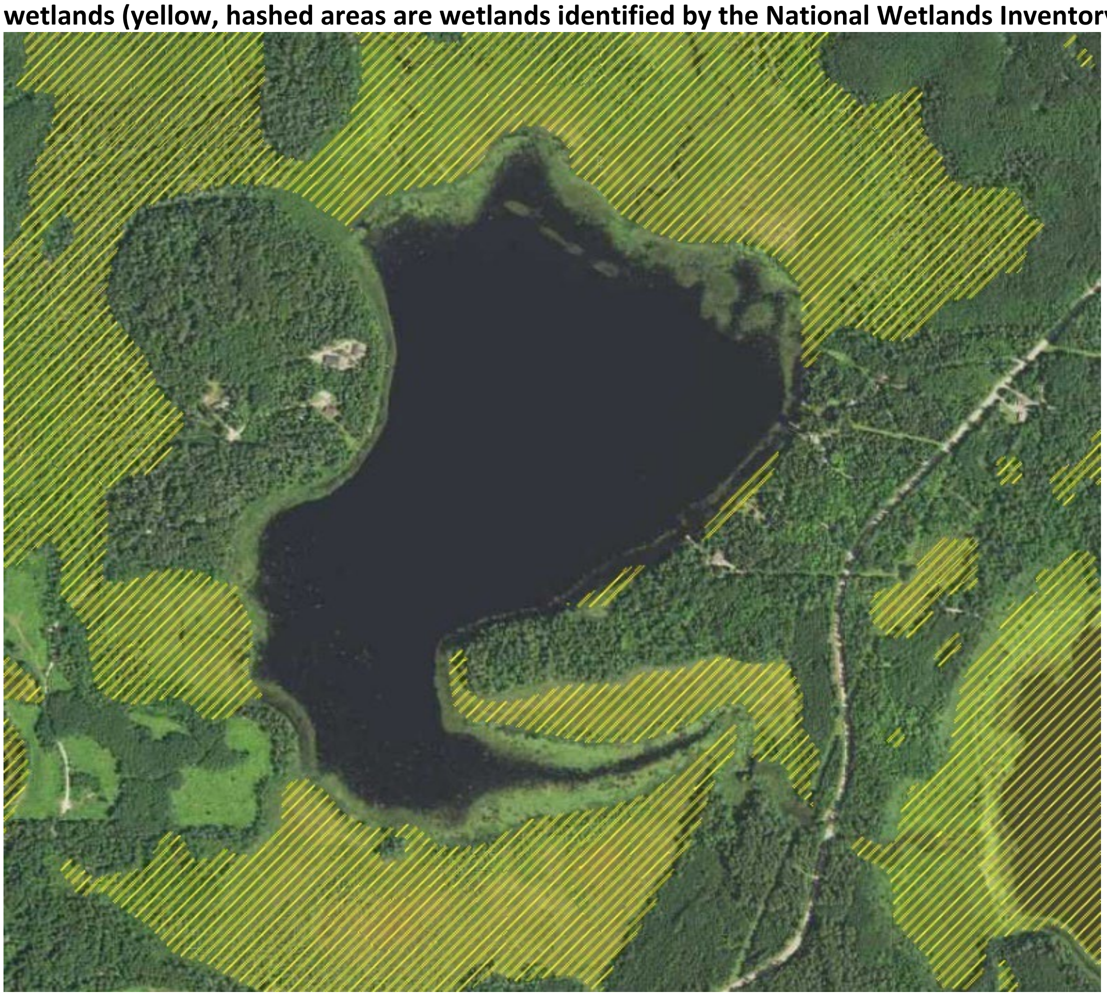
[2] In current rule, these waters are referred to as "lakes" and are defined as "an enclosed basin filled or partially filled with standing fresh water with a maximum depth greater than 15 feet" (Minn. R. 7050.0150, subp. 4, Item Q). These lakes tend to be dimictic and may also be referred to as deep lakes.

lakes, the MPCA initiated this study of lakes in the North region to determine whether revisions to the existing standards are needed and if so, which criteria are appropriate for the protection of northern lakes.

Table 2. Minnesota's current lake water quality standards by ecoregion and lake type (NLF = Northern Lakes and Forests, NCHF = North Central Hardwood Forests, WCBP = Western Corn Belt Plains, NGP = Northern Glaciated Plains, Class 2A = coldwater habitats, Class 2B/2Bd = cool and warm water habitats).

Ecoregion	Total phosphorus (µg/L)	chlorophyll-*a* (µg/L)	Secchi depth (m)
NLF – Class 2A: Lake trout	12	3	4.8
NLF – Class 2A: Stream trout	20	6	2.5
NLF – Class 2B/2Bd	30	9	2.0
NCHF – Class 2A: Stream trout	20	6	2.5
NCHF – Class 2B/2Bd	40	14	1.4
NCHF – Class 2B/2Bd: Shallow lakes	60	20	1.0
WCBP & NGP – Class 2B/2Bd	65	22	0.9
WCBP & NGP – Class 2B/2Bd: Shallow lakes	90	30	0.7

Figure 1. Bluebill Lake (31-0265-00), Itasca County showing high concentration of surrounding wetlands (yellow, hashed areas are wetlands identified by the National Wetlands Inventory).



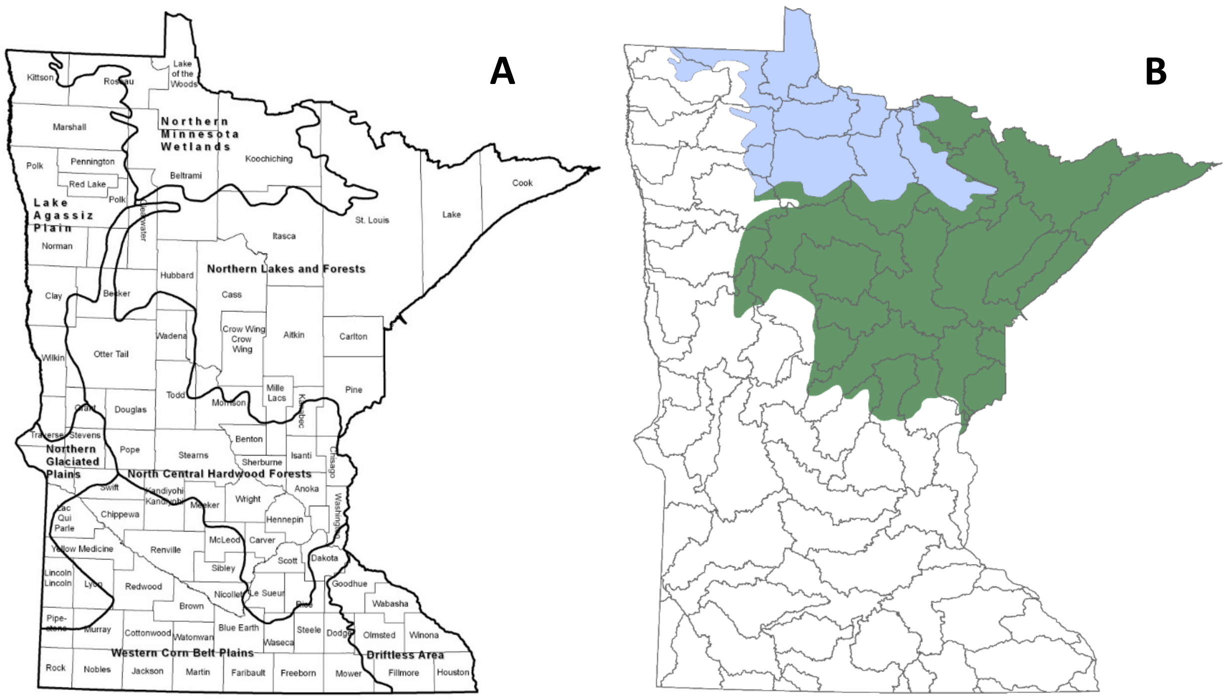
i. Lake typology

Lakes have naturally different characteristics that affect trophic status and thereby require classification for appropriate application of eutrophication standards. Lake typological frameworks can be complicated and some lakes challenge placement within such classifications, so it is necessary to describe in some detail the typology used in Minnesota and the nomenclature used for this framework. Minnesota has adopted a framework of eutrophication standards which accounts for regional differences (North, Central, and South nutrient regions), thermal conditions (coldwater [Class 2A] and warm/cool water lakes [Class 2B/2bd]), and stratification/depth characteristics (dimictic or polymictic/shallow; Table 2). When considering stratification/depth characteristics, it is useful to note that lakes often contain different habitat zones within their basins and each zone may support different biological communities which may overlap with other lake types. For example, lake trout lakes (Class 2A) are coldwater lakes which support cool, well-oxygenated water necessary for the survival of lake trout. However, the shallow portions of these lakes may also harbor cool/warm water species similar to those found in polymictic or shallow warm water lakes. Designation of lakes or placement within the lake typology is based on the most sensitive community within the lake and may also consider the overall characteristics of a lake. For example, some lakes coded as polymictic or shallow lakes may contain relatively small areas or basins which are deeper than 4.57 m (15 ft), but the overall character of these lakes is shallow, littoral habitat. There are several criteria or guidelines used for determining the placement of lakes within Minnesota's eutrophication lake typology (e.g., depth, percent littoral area, thermocline, and fish community; MPCA 2021, Appendix D). Although we can apply certain criteria thresholds to categorize most lakes, some lakes will not meet all criteria used to define a lake type and will require professional judgement to assign them to a lake category based on overall lake character (See Appendix A).

ii. Regionalization of lake eutrophication standards

Minnesota's abundant and diverse lake resources vary across the state naturally and are delineated in Minnesota's lake eutrophication standards using the United States Environmental Protection Agency's (EPA) aquatic ecoregion framework (Figure 2; Omernik 1987). Over 98% of Minnesota's lakes are within four of these ecoregions: NLF, North Central Hardwood Forests (NCHF), Northern Glaciated Plains (NGP), and Western Corn Belt Plains (WCBP). To account for different lake characteristic in these four ecoregions, Minnesota's lake eutrophication standards are currently divided into three regions: North (NLF), Central (NCHF), and South (NGP and WCBP; Minn. R. 7050.0222; Table 2). Different trophic criteria apply to lakes in each of these regions to account for natural differences in trophic state between regions. In the current rule framework, lakes in three lake-poor ecoregions, NMW, Lake Agassiz Plain, and Driftless Area, do not have eutrophication standards assigned to them. Instead, standards are assigned to lakes in these regions on a site-specific basis (Minn. R. 7050.0222).

Figure 2. Maps of (A) Minnesota's ecoregions (EPA Level III) and counties (Minn. R. 7050.0468) and the (B) Northern Lakes and Forests (green) and Northern Minnesota Wetlands (blue) Level I ecoregions with 8-digit hydrological unit code (HUC 8) watersheds.



In Minnesota, the Level I Northern Forests ecoregion is comprised of the NLF and NMW Level III ecoregions (Omernik 1987; Figure 2). Within the Midwestern United States, the Northern Forests ecoregion includes the land area in north central and northeast Minnesota, northern Wisconsin, and northern Michigan. The land cover in this ecoregion is dominated by forest, wetland, and open water which combined make up ~90% of the land area (White 2020, Wilson and Ryan 2015). Developed land (urban, mining, and agriculture) makes up the remaining land cover. In Minnesota, the NLF is particularly lake rich (Table 3) and contains over 5,600 lakes or approximately 47% of all lakes in Minnesota (Olmanson et al. 2014). Although not the dominant lake type, shallow lakes (depths <5m[3]) are abundant and number approximately 1,600 or 28% of NLF lakes in Minnesota (Olmanson et al. 2014). The true number of shallow lakes in the NLF is likely higher because in remote parts of the NLF, such as the Boundary Waters Canoe Area Wilderness, many lakes have never been surveyed and lack bathymetry data. The NMW ecoregion has many fewer lakes than the NLF with only 215 lakes. Although specific eutrophication criteria are not currently assigned to lakes in the NMW ecoregion, lakes in this ecoregion are most similar to lakes in the NLF ecoregion. As a result, the NLF criteria are typically applied to NMW lakes and in this study, NLF and NMW lakes are treated together with the intention that revised northern eutrophication standards should apply to lakes in the Northern Forests ecoregion (i.e., NLF and NMW ecoregion lakes).

[3] This is the threshold used by Olmanson et al. (2014) to define shallow lakes. This threshold does not correspond to the threshold used in Minnesota Rule or in this document.

Table 3. Descriptions of Minnesota's Northern Forest ecoregions (from White 2020).

Northern Minnesota Wetlands (49)
"Much of the Northern Minnesota Wetlands Level III ecoregion is a vast and nearly level wetland that is sparsely inhabited by humans and covered by conifer bog, mixed forest, and boreal forest vegetation. Formerly occupied by broad glacial lakes, much of the flat terrain in this ecoregion is still covered by standing water. Some low-gradient streams and eroded river channels occur in the east."
Northern Lakes and Forests (50)
"This Level III ecoregion has relatively nutrient-poor glacial soils, coniferous and northern hardwoods forests, undulating till plains, morainal hills, broad lacustrine basins, and areas of extensive sandy outwash plains. Soils are formed primarily from sandy and loamy glacial drift material and generally lack the arability of those in adjacent ecoregions to the south and west. Ecoregion 50, along with the [Northern Minnesota Wetlands] (49), have lower annual temperatures and a frost-free period that is considerably shorter than other ecoregions in Minnesota; this ecoregion also has the largest annual snowfall and the most days with snow cover. These conditions generally hinder agriculture; therefore, woodland and forest are the predominant land use/land cover. Numerous lakes dot the landscape."

Due to the low watershed disturbance and natural characteristics of the NLF and NMW regions, lakes in northern ecoregions on average have lower trophic state compared to other regions in Minnesota (MPCA 2020). National Lake Assessment (NLA) data indicated that in the northern region, TP and chl-*a* have been largely stable from 2007-2017 with a possible small increase in TP between surveys (MPCA 2020). However, other research has indicated that lakes in largely undisturbed areas, including northern Minnesota, have a relatively large increase in nutrients over recent decades (Stoddard et al. 2016). These changes are occurring at the continental scale and are likely attributable to a changing climate because anthropogenic activity in regions such as northern Minnesota remains low. Although there may be some increases in nutrient loading to lakes in Minnesota's northern region, most lakes are still mesotrophic or oligotrophic and conditions are near natural conditions (Ramstack et al. 2003). Many lakes are in largely undisturbed watersheds, which means that unlike other ecoregions, natural background conditions can be more easily determined. As a result, many lakes are near natural conditions, which allows for reexamination of the lake eutrophication standards when available water quality data indicate non-attainment of current standards. Indeed, the identification of a number of these northern lakes in undisturbed watersheds with relatively high nutrient levels were the trigger for this study.

iii. Lake beneficial uses

The Class 2 use designation protects two beneficial uses: aquatic life and recreation. Depending on the lake, these beneficial uses protect different types of aquatic life and recreation activities. The standards to protect these beneficial uses are often based on the most sensitive "sub-use." The most sensitive beneficial or designated use by lake type was defined by Heiskary and Wilson (2005; see Table 4). For example, lakes designated as trout lakes are specifically protected for trout and eutrophication standards are tailored to this sensitive portion of the aquatic life assemblage. Similarly, the distinction between dimictic and polymictic lakes in the Central and Southern nutrient regions recognizes differences in both the aquatic life and recreation activities that can be supported by these waters. Polymictic lakes may have a greater focus on secondary contact activities (e.g., boating, fishing, waterfowl production) whereas dimictic lakes may have a greater focus on the protection of primary contact activities such as swimming (Table 4; Heiskary and Wilson 2005). Overall, polymictic lakes standards were developed to minimize the risk of lakes shifting from a macrophyte dominated (i.e., clear water) state to an undesirable algal dominated (i.e., turbid) state (Heiskary and Wilson 2008). Specific sub-uses for shallow/polymictic lakes are codified in Minn. R. 7050.0150[4].

Table 4. Subcategories of aquatic life and recreation beneficial uses that eutrophication standards were designed to protect (from Heiskary and Wilson [2005]).

Waterbody type	Uses*
Lake trout Lakes	1. Protection of sensitive aquatic community. Specifically, maintenance of adequate dissolved oxygen in hypolimnion needed to support lake trout; 2. Water recreation of all types including swimming; and 3. Aesthetics.
Stream trout lakes	1. Protection of sensitive aquatic community. Specifically, maintenance of adequate dissolved oxygen in metalimnion needed to support stream trout; 2. Water recreation of all types including swimming; and 3. Aesthetics.
Lakes and reservoirs > 4.57 m (15 ft) deep	1. Water recreation of all types including swimming, at least part of the summer season; 2. Maintenance of the desired game fishery; and 3. Aesthetics.
Shallow lakes and reservoirs < 4.57 m (15 ft) deep	1. Protection of aquatic community. Specifically, the maintenance of a diverse community of emergent and submerged aquatic plants, and wildlife; 2. Water recreation of all types including primary body contact where usable; and 3. Aesthetics.

*The more "sensitive" use, which is the primary basis for the proposed standard, is listed as number 1. Other uses follow.

Attainment of aquatic life beneficial uses can be measured using water chemistry (e.g., lake eutrophication standards, toxics) or biological indices for assemblages such as macrophytes, zooplankton, macroinvertebrates, and fish. In Minnesota, the Minnesota Department of Natural Resources (MNDNR) has developed biological indices for macrophytes and fish in lakes (Radomski and Perleberg 2012, Bacigalupi et al. 2021). Historically, the primary tool for measuring lake recreation uses in Minnesota has been the use of lake observer surveys. Lake observer surveys of lake physical condition and recreation suitability have long been a component of MPCA and volunteer monitoring (Table 5). Recreation use and user expectations vary across Minnesota lakes and in general, transparencies for northern lakes are higher for a given response than other regions likely due to different expectations and sensitivities which are influenced by the region's predominance of high-quality lakes (Heiskary and Walker 1988). Regional differences are especially pronounced and significantly different at the lower end of the survey scale (Smeltzer and Heiskary 1990). Although existing analyses have not studied differences in user survey ratings between dimictic and polymictic lakes, the specific beneficial uses (e.g., swimming versus fishing) provided by these resources are different and could affect survey results. For example, user surveys may reflect the fact that swimming may not be a primary use in many of Minnesota's polymictic lakes due to lake depth, the presence of highly organic substances, and an abundance of aquatic plants. As a result, users may rate recreation suitability lower for shallow lakes based on factors other than suspended algae. These considerations have prompted the MPCA to further study beneficial uses in polymictic lakes to ensure that the most sensitive beneficial use is protected in polymictic and dimictic lakes.

[4] "The quality of shallow lakes will permit the propagation and maintenance of a healthy indigenous aquatic community and they will be suitable for boating and other forms of aquatic recreation for which they may be usable." Minn. R. 7050.0150, subp. 4, Item HH.

Table 5. Lake observer survey ratings and descriptions (MPCA 2016; modified from Garrison and Smeltzer 1987).

Rating	Description
Recreational suitability	
1	Beautiful, could not be any nicer.
2	Very minor aesthetic problems; excellent for swimming, boating, enjoyment.
3	Swimming and aesthetic enjoyment slightly impaired because of algae levels.
4	Desire to swim and level of enjoyment of the lake substantially reduced because of algae levels (would not swim, but boating is okay).
5	Swimming and aesthetic enjoyment of the lake nearly impossible because of algae levels.
Physical condition	
1	Crystal clear water
2	Not quite crystal clear—a little algae present/visible
3	Definite algae—green, yellow, or brown color apparent
4	High algal levels with limited clarity and/or mild odor apparent
5	Severely high algae levels with one or more of the following: massive floating scums on the lake or washed up on shore; strong, foul odor; or fish kill

iv. Lake stratification and depth

A review of possible revisions to northern lake eutrophication standards requires a description of the dichotomy of polymictic or shallow versus dimictic lakes, and the applicability of these factors in development of potential new standards in northern lakes. Lake stratification is a primary driver of lake productivity and is reflected in MPCA's shallow lakes definition (Minn. R. 7050.0150). Lake depth strongly influences if a lake stratifies, with shallow lakes much less likely to stratify. As a result, lake depth is a reasonable predictor of a lake's mixing status; although, other factors can be important such as lake area. Other measurable attributes, including lake geometry ratio[5] and lake temperature profiles, can be used to determine stratification status more accurately. Stratification, although not independent of depth, has been identified as a more important environmental factor than depth for developing nutrient-chl-*a* prediction models (Yuan and Pollard 2014) and is therefore used in this study to categorize lakes.

This document divides northern lakes into dimictic and polymictic lakes (see Appendix A). Lakes which are polymictic are generally consistent with Minnesota's current definition of "shallow" lakes: "*an enclosed basin filled or partially filled with standing fresh water with a maximum depth of 15 feet or less or with 80 percent or more of the lake area shallow enough to support emergent and submerged rooted aquatic plants (the littoral zone). It is uncommon for shallow lakes to thermally stratify during the summer*" (Minn. R. 7050.0150, subp. 4, Item HH). This definition largely focuses on the amount of the lake which is less than 4.57 m (15 ft) deep; although, the stratification status is also included. In this document, dimictic lakes largely correspond to waters defined as "lakes" in Minnesota rule: "an enclosed basin filled or partially filled with standing fresh water with a maximum depth greater than 15 feet" (Minn. R. 7050.0150, subp. 4, Item Q). Although this definition does not include stratification characteristics, many lakes in Minnesota meeting this definition are dimictic. As a result, the definitions of "shallow lake" and "lake" in rule are largely sufficient to appropriately categorize relevant lake type as part of a lake eutrophication typology. However, to clarify and emphasize the focus on lake stratification type we opt to use the more descriptive and accurate terms "dimictic" and "polymictic." The use of this

[5] Geometry ratio can be calculated as: $A_0^{0.25}/z_{max}$, where A_0 is lake surface area (m^2) and z_{max} is maximum depth (m) (Stefan et al. 1996).

terminology may also necessitate revisions to lake definitions in rule for consistency; although, the term "shallow lake" remains largely appropriate and may be used interchangeably with "polymictic lake." There are other lake types in Minnesota including meromictic[6] lakes and lakes which are intermediate between dimictic and polymictic lakes (e.g., discontinuous cold polymictic lakes[7]). Such lakes may require additional review to determine the applicable standard or if a site-specific standard is necessary.

Lake morphology affects nearly every lake attribute including productivity, internal loading, water movement, and biological communities. We can account for some of this variability by dividing lakes into polymictic and dimictic lakes. Polymictic lakes are by their nature inherently different from deeper, dimictic lakes because of their reduced depth and volume. As discussed, polymictic lakes continually turnover through the summer which affects temperature, oxygen, nutrient cycling, and biological assemblages. For example, shallower lake depths result in higher sediment temperatures in polymictic lakes which yield increased mineralization rates and thus sediment phosphorus release (Søndergaard et al. 2003). Attributes external to a lake's basin can also be important including water source (e.g., springs, streams), water outlet size, watershed slope, and watershed:lake ratios. In polymictic lakes, nutrient loading per unit volume can be higher, losses of nutrients to depositories such as sediments or lake outflow lower, and rates of nutrient recycling faster compared to dimictic lakes (Wetzel 2001). In some lakes, higher trophic states may be attributable to higher watershed:lake ratios (e.g., Webster et al. 2008). While dimictic lakes have greater volumetric buffering and higher net phosphorus sedimentation, polymictic lakes with large watershed areas have larger annual phosphorus loads relative to their volume, which may become legacy impacts via internal loading. Due to these differences in internal and external lake attributes, polymictic lakes on average have higher levels of TP, algal productivity (chl-*a*), DOC, and CDOM compared to dimictic lakes (Rasmussen et al. 1989, Nürnberg and Shaw 1998, Havens and Nürnberg 2004, Webster et al. 2008). Although increased CDOM would be expected to limit lake productivity through shading of algae and thereby offset the effect of increased TP, this has not been demonstrated in shallow lakes (e.g., Nürnberg 1996, Zwart et al. 2016).

In addition to lake and watershed morphology attributes, there are also important regional landscape drivers of water chemistry and lake productivity. Polymictic lakes within forest and wetland landscapes can naturally have higher productivity than deeper lakes (Havens and Nürnberg 2004). Gartner Lee Limited (2006), determined that TP was significantly higher in lakes located on organic terrain. Wetlands were also found to increase TP concentrations in northern Michigan headwater lakes (Zhang et. al. 2012). In 20 relatively undisturbed forested watersheds in Ontario, wetland extent was correlated with export of TP and DOC indicating that wetlands are TP sources to lakes (Dillon and Molot 1997). In fact, DOC concentrations in lakes can provide a useful index of watershed influence because it is primarily derived from surrounding wetlands (Gergel et al. 1999). In Minnesota, both northern dimictic and polymictic lakes have higher proportions of wetlands in their lakesheds compared to other regions; however, there is a significantly higher proportion of wetlands in polymictic lake watersheds (Figure 3; Mann-Whitney U Test: $W = 244229$, p-value < 0.0001). The influence of increased natural TP loading in northern polymictic lakes is also supported by Hydrological Simulation Program - FORTRAN (HSPF) modeling. Using HSPF modeling to determine TP loads to several impaired polymictic lakes in the St. Louis River watershed within the North region, it was estimated that the majority of the annual phosphorus load comes from watershed sources typical of the northern region (i.e., upstream forest, wetland, and water land cover) and anthropogenic sources were a smaller portion of annual loads (Figure 4). These results and the conclusions of other studies demonstrate that in some regions, like the Northern Lakes ecoregion, greater lake productivity in polymictic lakes compared to dimictic lakes can in part be attributed to greater sources of natural TP and DOC from wetlands. In northern Minnesota, natural watershed sources of nutrients and organic matter to lakes can be substantial, especially to

[6] Meromictic lakes do not completely mix.
[7] Discontinuous cold polymictic lakes stratify periodically during the summer (Wetzel 2001).

polymictic lakes, which challenges lake restoration options under the current lake eutrophication standards.

Figure 3. Violin plots comparing the percent of wetland landcover in northern dimictic and polymictic lake watersheds. Description of violin plots: grey circles = individual lakes; width of plot = kernel probability density; solid black lines = 10th, 25th, 50th, 75th, and 90th percentiles.

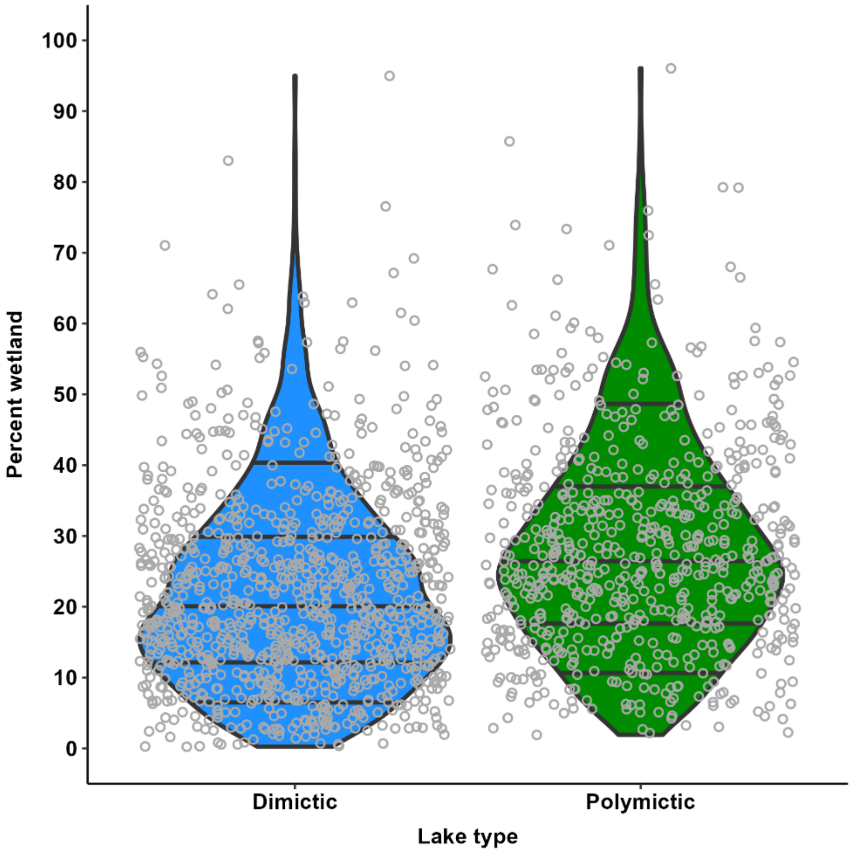
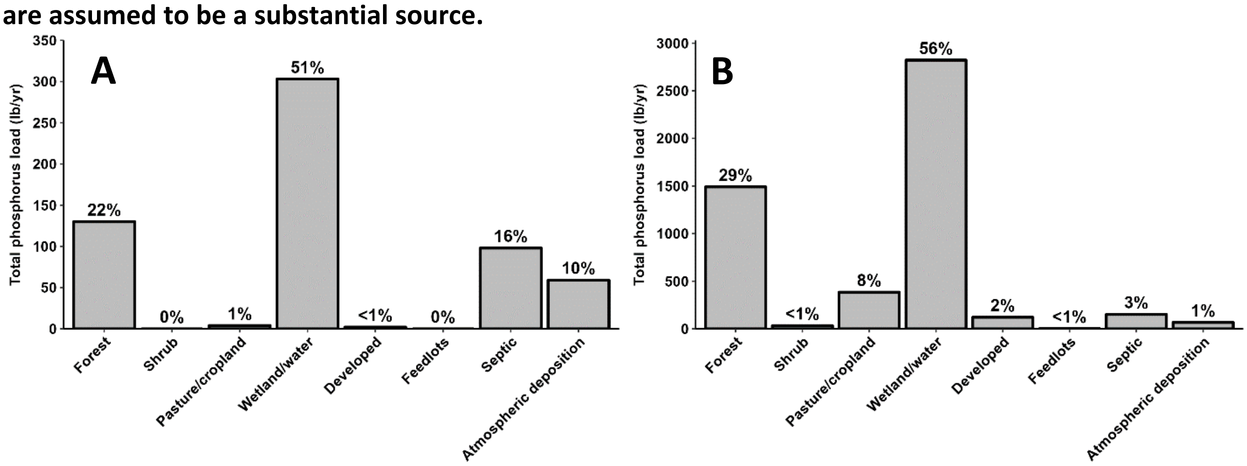
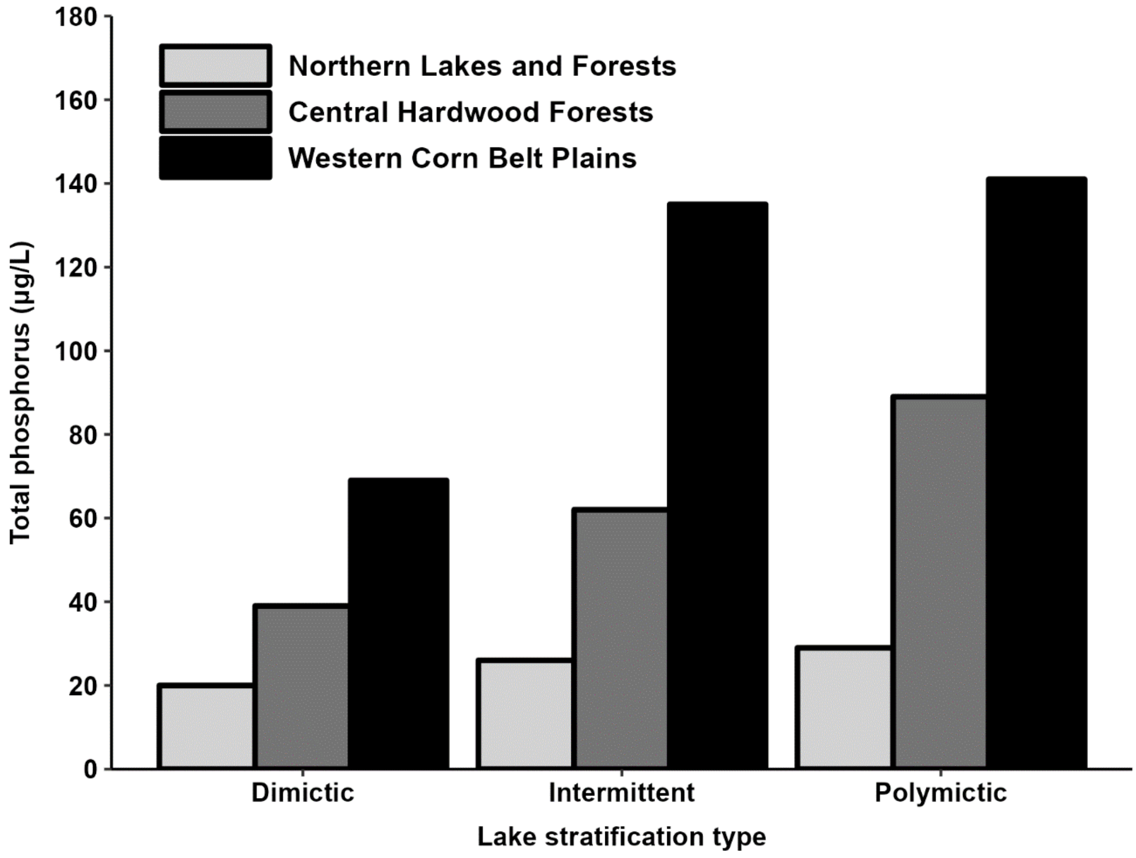


Figure 4. Estimated sources of total phosphorus to Long (A) and Strand (B) lakes (Plevan and Olson 2016). Phosphorus loading from internal loading and shoreland development were not quantified but are assumed to be a substantial source.



The effects of lake depth or stratification are currently recognized in Minnesota's lake eutrophication standards through different criteria (i.e., TP, chl-*a*, and Secchi depth) for "lakes" and "shallow lakes". However, this distinction is only made in the Central and South nutrient regions and the North nutrient region does not have separate standards for these lake types (Table 2). The lack of separate criteria for lakes with different stratification types in the North region was based on an analysis of the effect of mixing status on trophic state as part of the supporting technical documentation for the 2008 lake eutrophication rule (Heiskary and Wilson 2005). Mixing status was determined through a review of lake temperature and dissolved oxygen profiles for MPCA's Ecoregion Reference Lakes. These lakes were classified as either dimictic (stratified throughout the summer), polymictic (well-mixed throughout the summer), or intermittent (stratified during calm periods) (Heiskary and Wilson 2005). Based on these data, regional patterns were evident (Figure 5; Heiskary and Wilson 2005). In the NCHF (Central region) and WCBP (South region) ecoregions, differences in median TP between lake stratification types was used to support separate criteria for dimictic and polymictic and intermittent lakes (Heiskary and Wilson 2008; Figure 5). Although the difference in absolute TP concentrations between lakes with different stratification types was small for lakes in the NLF, the relative increase in median TP concentration was 45% between dimictic and polymictic lakes in the NLF ecoregion (Figure 5). This difference suggests a need to revisit the lake typology for northern lakes and the eutrophication standards used to protect these lakes. In addition, the identification of numerous northern lakes in largely undisturbed watersheds that do not meet the current standards, and the documented trophic differences observed between polymictic and dimictic lakes, demonstrate that it is reasonable to reexamine these standards to determine if it is appropriate to refine the lake eutrophication typology.

Figure 5. Median total phosphorus by ecoregion and lake-mixing type. Dataset comprised of the 1988 assessment lakes (see Heiskary and Wilson [2005]) .



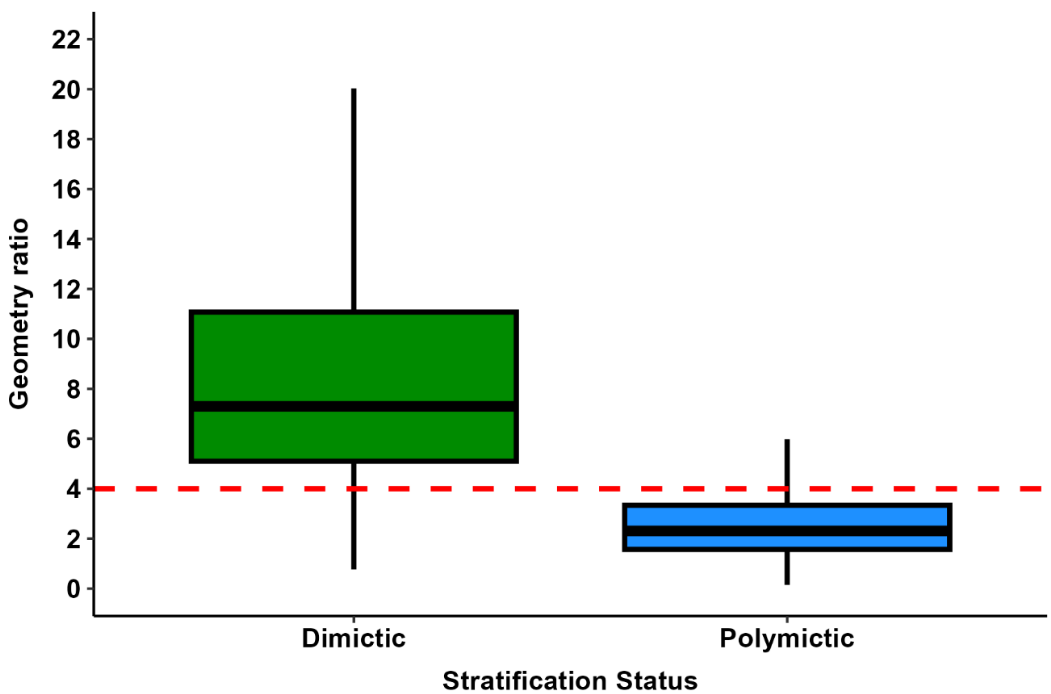
Data

The analyses in this study used a variety of datasets to characterize water quality in northern lakes and to identify protective eutrophication thresholds. Data were limited to water quality, biological, and recreation survey samples collected from 1990-2020. The analyses in this study include two main datasets: 1) a dataset of North region lakes only and 2) a combined North and Central region lake dataset. North region lake datasets included lakes from the NLF and NMW ecoregions. Lakes that are wholly or partially within one of these two ecoregions were included. Lakes in the NCHF ecoregion and Lake Agassiz Plain (LAP; or Red River Valley [RRV]) are part of the Central region and were included along with NLF and NMW lakes for the North/Central region dataset. Only lakes fully within the North or Central regions were included as part of these data. Lakes fully or partially in the Driftless Area (DLA), WCBP, and NGP ecoregions were not used in these analyses except for a single analysis comparing color between the three nutrient regions. The North-only datasets were largely used to characterize water quality in this population of lakes and to determine if there were differences between dimictic and polymictic lakes. The combined Central and North region datasets were used to assess the effects of disturbance on beneficial uses. The inclusion of the Central region lakes was necessary to create a more complete disturbance gradient due to the largely undisturbed condition of the North region.

i. Lake stratification

For most analyses, lakes were divided into dimictic and polymictic lakes using geometry ratio. Geometry ratio was calculated as: $A_0^{0.25}/z_{max}$, where A_0 is lake surface area (m^2) and z_{max} is maximum depth (m) (Stefan et al. 1996). A geometry ratio of 4 was used as a threshold to predict lake stratification where lakes with a geometry ratio of less than 4 were identified as dimictic. A geometry ratio of 4 was selected as a threshold because it reasonably distinguishes between dimictic and polymictic lakes (Jacobsen et al. 2010; Figure 6).

Figure 6. Comparison of geometry ratio for dimictic and polymictic lakes. Dimictic lakes were determined to be lakes with a temperature gradient of at least 1 °C per meter for more than 50% of lake oxythermal profiles. Red dashed line indicates the threshold used to predict lake stratification



type.

ii. Water quality data

Most water quality data were drawn from MPCA's data storage system: Environmental Quality Information System (EQuIS) database. The parameters queried from EQIS included TP, chl-*a*, Secchi depth, DOC, and color. Absorbance at 440 nm (a_{440}) data were compiled from MPCA and University of Minnesota (UMN) datasets (see Brezonik et al. 2019). All data queries were limited to 1990-2020.

Epilimnetic water samples were collected using either a 2-m long, 32-mm diameter integrated sampler or surface grab samples. Standard limnological methods were for used for TP, chl-*a*, DOC, and color (Table 6). Absorbance at 440 nm (a_{440}) data were collected by the MPCA and UMN following methods described in Brezonik et al. (2019). The reporting detection limit was variable, but the dataset was screened for samples with a reporting detection limit greater than the reported value and these measurements were censored. All negative values were also removed. To address non-detects, one half of the reporting detection limit was used for analyses.

Table 6. Summary of analytical methods used for water quality samples.

Water quality parameter	Analytical methods
Total phosphorus	365.1; 365.2; 365.3; 365.4; 4500-P (C, E, F, I)
Chlorophyll-*a*	10200-H; D3731-87; 445.0
Secchi depth	Field method
Dissolved organic carbon	5310-B; 5310-C; 9060A
Color	110.2; 110.3; 2120-B; 2120-C
Absorbance at 440 nm	see Brezonik et al. (2019)

The TP, chl-*a*, and Secchi depth datasets used in this study consisted only of lakes with at least two years of water quality data and at least four measurements collected during the summer index period (June-September) each year. Secchi depth measurements were eliminated from lakes that were too shallow to measure (i.e., Secchi depth > lake depth). Long-term summer averages for TP, chl-*a*, and Secchi depth were calculated as the average of individual summer averages[8]. No data minimum was required for DOC, color and a_{440} datasets. If multiple samples were available from a lake for these three parameters, the average of these values was calculated. High color lakes (color >25 or >73 PCU or a_{440} >1.4 or > 4 m^{-1}) were flagged and censored in some analyses. Additional information in these CDOM thresholds is provided in the ***Secchi depth and colored dissolved organic matter*** section (p. 24).

iii. 2005 reference lake data

As part of the development of the lake eutrophication standards adopted in 2008, a technical support document (Heiskary and Wilson 2005), was developed. Many of the analyses in that report were based on a set of “reference” lakes including models of the relationships between TP, chl-*a*, and Secchi depth. These lakes were originally selected as being representative of their ecoregion and had minimal disturbance from point and nonpoint sources of pollution. This dataset includes lakes from the four ecoregions in Minnesota which contain most of the lakes in the state: NLF, NCHF, WCBP, and NGP. These data are described in detail in Heiskary and Wilson (2005). In the current study, the reference lakes and the original models developed from these data are used for comparison with the updated datasets described above.

iv. Beneficial use indicator data

Fish and macrophyte biological monitoring data were obtained from the MNDNR as this agency largely performs biological monitoring for Minnesota lakes. Fish data were limited to survey years 2005 through 2019 since 2005 was the year when the sampling methods for the FIBI were initiated. See MNDNR’s “Manual instructions for lake survey” (2017) for a description of fish survey methods and Bacigalupi et

[8] EPA recommends the use of a geometric mean of summer samples because environmental data are often log-normally distributed (EPA 2021). However, Minnesota’s lake eutrophication standard is based on an arithmetic mean of summer samples. In the future, Minnesota could consider revising lake eutrophication standards to use a geometric mean, but to do so as part of this effort would require a revision to all lake eutrophication standards or the use of a mix of arithmetic and geometric means. As a result, Minnesota will retain the use of the arithmetic mean for northern lakes.

al. (2021) for a detailed description of the FIBI and associated biocriteria and their development. Aquatic macrophyte data were limited the survey years 1993 through 2019 as 1993 coincided with the start of survey data methods used in the original index development (Radomski and Perleberg 2012). See Radomski and Perleberg (2012) for a detailed description of the macrophyte index and associated biocriteria. Recreational user survey data from 1990-2020 collected by volunteer lake monitors and water quality survey staff were queried from EQuIS. See Smeltzer and Heiskary (1990) for additional information on Minnesota's recreation user survey methods. Data for the three beneficial use indicators were used to establish relationships between these indicators and trophic state and to identify chl-*a* thresholds which are needed to protect these beneficial uses (i.e., aquatic life and recreation).

Characterization of water quality

Before eutrophication thresholds for protecting beneficial uses could be determined, it was necessary to review available water quality data for northern lakes to characterize water quality and to compare these data to those used to develop the lake eutrophication standards adopted in 2008. Specifically, the following section analyzes if eutrophication parameters (i.e., TP, chl-*a*, Secchi depth, and CDOM) differ between dimictic and polymictic northern lakes and if the relationships among these parameters differ from those used to develop the current standard (Heiskary and Wilson 2005). If either indicates important and significant differences, then revisions to the standard would be warranted to ensure that protective and appropriate standards are assigned to northern lakes.

i. Water quality relationships

Water quality distributions and relationships between long-term summer average TP, chl-*a*, and Secchi depth for dimictic and polymictic lakes were assessed. The objective of this analysis was 1) to determine if water quality relationships differ between dimictic and polymictic northern lakes and 2) to determine if water quality relationships for northern lakes differ from statewide models used to develop the 2008 standards. The effect of light attenuating water quality parameters (i.e., CDOM) were also assessed in relation to their impact on Secchi depth.

1. Methods

Cumulative distribution functions (CDF) and violin plots were used to compare distributions of water quality data between dimictic and polymictic lake populations. The datasets used for these analyses did not censor lakes with high levels of CDOM. Cumulative distribution functions were plotted in R version 4.0.3 (R Core Team 2020) using the "stat_ecdf" function in the "ggplot2" package (Wickham 2016). Violin plots were created in R version 4.0.3 (R Core Team 2020) using the "geom_violin" function in the "ggplot2" package (Wickham 2016). The reference lakes from Heiskary and Wilson (2005) were also included in some violin plots for comparison with the original datasets used to develop the standards adopted in 2008. To compare water quality parameters between dimictic and polymictic lake populations, Mann-Whitney tests were used to test for significant differences using the "wilcox.test" function (R Core Team 2020). Environmental data is often non-normal, so the non-parametric Mann-Whitney test was used to compare groups of unpaired samples.

Least squares regressions were fit to log-transformed TP and chl-*a* data to analyze the relationships between these parameters. Linear regression was performed using the "lm" function in R version 4.0.3 (R Core Team 2020). The relationship between chl-*a* and Secchi depth was nonlinear so this relationship was modeled using a generalized additive model (GAM) in R version 4.0.3 (R Core Team 2020) with the "mgcv" package (Wood 2017). The datasets used for the TP – chl-*a* and chl-*a* - Secchi depth models included two datasets: 1) a dataset uncensored for CDOM and 2) a dataset censored for lakes with color >73 PCU or a_{440} >4 m^{-1}. Both the TP – chl-*a* and chl-*a* – Secchi depth models were compared to the original models developed by Heiskary and Wilson (2005).

We modeled the relationship between CDOM (measured as color and a_{440}) and Secchi depth using generalized additive models (GAMs) in R version 4.0.3 (R Core Team 2020) using the "mgcv" package (Wood 2017). The probability of attaining the current northern lake Secchi depth standard (2.0 m) as a function of color was assessed with generalized additive models (GAMs) using a logistic link function in R version 4.0.3 (R Core Team 2020) in the "mgcv" package (Wood 2017). To evaluate if censoring lakes with high color reduces false positive rates when using Secchi depth, receiver operating characteristic (ROC) curves were generated and false positive rates between the uncensored lake dataset and censored datasets (color >73 PCU or a_{440} >4 m^{-1} and color >25 PCU or a_{440} >1.4 m^{-1}) were compared. ROCs were modeled in R version 4.0.3 (R Core Team 2020) using the "pROC" package (Robin et al. 2011). Area under the curve (AUC) scores were used to evaluate each ROC model. For this analysis, an AUC value of 1 indicates that the model perfectly predicts an exceedance of the chl-*a* standard based on Secchi depth and a score of 0.5 indicates that the model has no predictive ability. Scores between 0.5 and 1 indicate different levels of predictive ability for the models, but there is no absolute threshold which indicates whether a model is good or not. Hosmer et al. (2013) assigned approximate discrimination guidelines for AUC values which we follow here to provide context: 0.5-0.7 = poor; 0.7-0.8 = acceptable; 0.8-0.9 = excellent; >0.9 = outstanding.

2. Comparison of water quality between dimictic and polymictic northern lakes

Plots of CDFs for TP, chl-*a*, and Secchi depth demonstrated a difference between populations of northern dimictic and polymictic lakes with higher TP, chl-*a*, and color and lower Secchi depth in polymictic lakes (Figure 7). Based on this dataset, 6% of dimictic lakes and 28% of polymictic lakes (Figure 7A) exceed the current northern lake TP standard (30 µg/L), 14% of dimictic lakes and 35% of polymictic lakes (Figure 7B) exceeded the current chl-*a* standard (9 µg/L), and 11% of dimictic lakes and 45% of polymictic lakes (Figure 7C) exceeded the current Secchi depth standard (2 m). Measures of CDOM were also different between dimictic and polymictic lakes with both color and a440 higher in polymictic lakes (Figure 8A,B). DOC was also higher in polymictic northern lakes compared to dimictic lakes (Figure 8C); however, the DOC dataset for northern lakes was relatively small (n=58; Table 7). There are no Class 2B/2Bd standards for color, a440, or DOC. Mann-Whitney tests between dimictic and polymictic lakes indicated a significant difference between lake stratification types for all six water quality parameters (Table 7). As with the CDF plots, violin plots for TP, chl-*a*, Secchi depth, and CDOM (Figure 9) demonstrated a difference in between populations of dimictic and polymictic lakes with higher TP, chl-*a*, and CDOM (color and a440) and lower Secchi depth in polymictic lakes. Despite the smaller sample size in the 2005 reference lakes dataset, violin plots indicated that these lake datasets had similar distributions to the 1990-2020 northern lake dataset.

Figure 7. Cumulative distribution plots of A) total phosphorus, B) chlorophyll-*a*, and C) Secchi depth for dimictic and polymictic northern lakes (1990-2020; red dotted line = current standard).

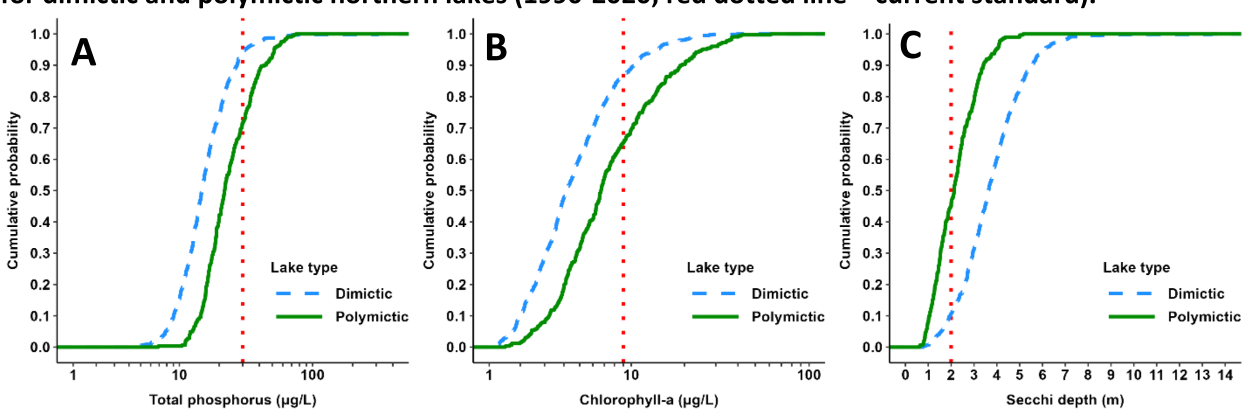


Figure 8. Cumulative distribution plots of A) color, B) a_{440} , and C) dissolved organic carbon for dimictic and polymictic northern lakes (1990-2020).

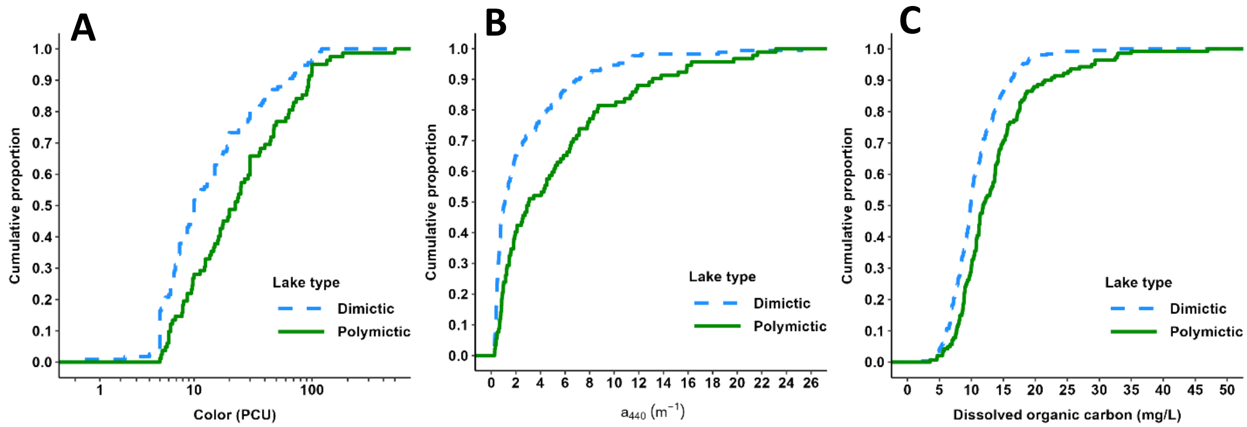
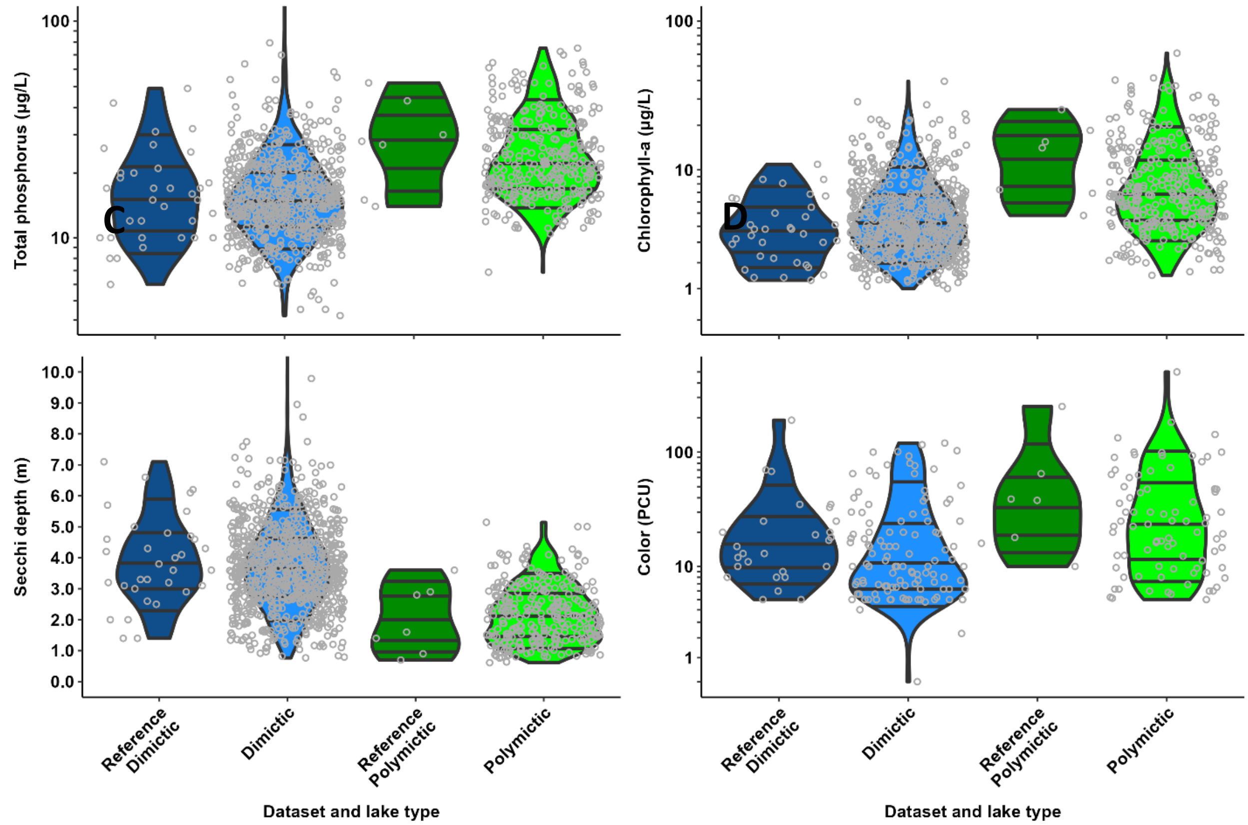


Table 7. Summary table for eutrophication parameters for dimictic and polymictic northern lakes(abbreviations: SD = standard deviation; MW = Mann-Whitney test).

					Percentile					
Parameter	**Lake type**	**N**	**Mean**	**SD**	**10**	**25**	**50**	**75**	**90**	**MW test**
Total phosphorus (µg/L)	Dimictic	633	17.3	14.4	9.1	11.5	14.8	20.2	27.2	W = 48800; p-value <0.0001
	Polymictic	324	26.0	12.5	14.6	16.9	22.0	31.8	43.0	
Chlorophyll-*a* (µg/L)	Dimictic	659	5.5	4.2	2.0	2.8	4.2	6.6	10.4	W = 64780; p-value <0.0001
	Polymictic	330	9.4	8.0	3.2	4.4	6.7	11.5	19.0	
Secchi depth (m)	Dimictic	822	3.7	1.4	2.0	2.8	3.6	4.6	5.6	W = 204240; p-value <0.0001
	Polymictic	302	2.2	0.9	1.1	1.5	2.1	2.8	3.4	
Color (PCU)	Dimictic	116	22.2	26.5	5.0	6.4	10.0	25.0	65.0	W = 3238; p-value = 0.0001
	Polymictic	82	41.5	62.8	6.2	9.8	22.5	47.7	93.3	
a_{440} (m^{-1})	Dimictic	169	2.7	3.8	0.4	0.5	1.1	3.7	6.9	W = 4809; p-value <0.0001
	Polymictic	92	5.4	5.5	0.7	1.2	3.1	7.8	13.1	
Dissolved organic carbon (mg/L)	Dimictic	33	3.0	3.8	0.5	0.6	1.1	4.6	6.6	W = 12856; p-value <0.0001
	Polymictic	38	4.7	4.5	0.8	1.5	3.0	6.8	10.5	

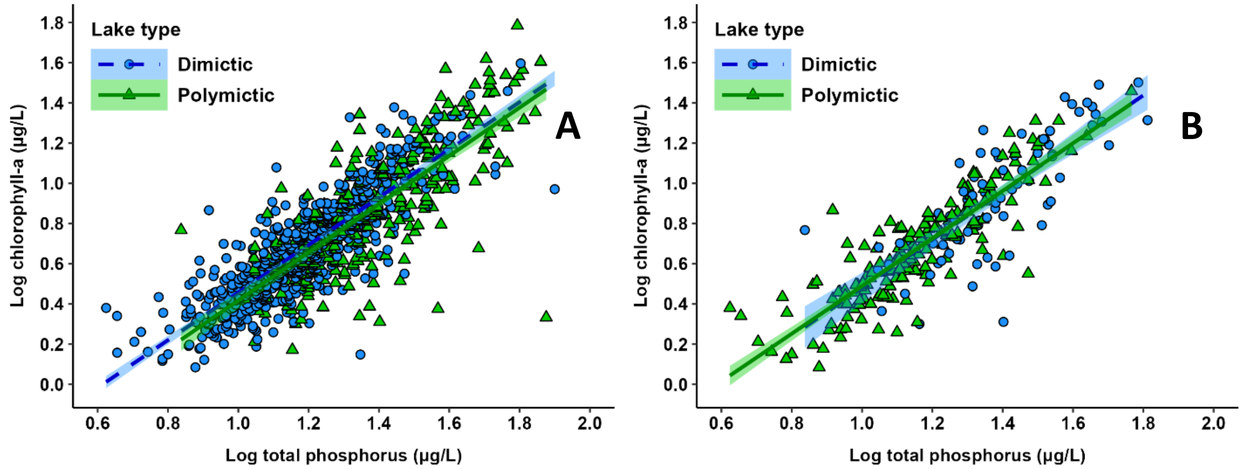
Figure 9. Violin plots of average A) total phosphorus, B) chlorophyll-*a*, C) Secchi depth, and D) color for northern reference lakes (Heiskary and Wilson 2005) and northern lakes (1990-2020). Description of violin plots: grey circles = individual lake measurements; width of plot = kernel probability density; solid black lines = 10th, 25th, 50th, 75th, and 90th percentiles. Determination of dimictic and polymictic lakes was based on geometry ratio.



3. Total phosphorus and chlorophyll-*a* relationships

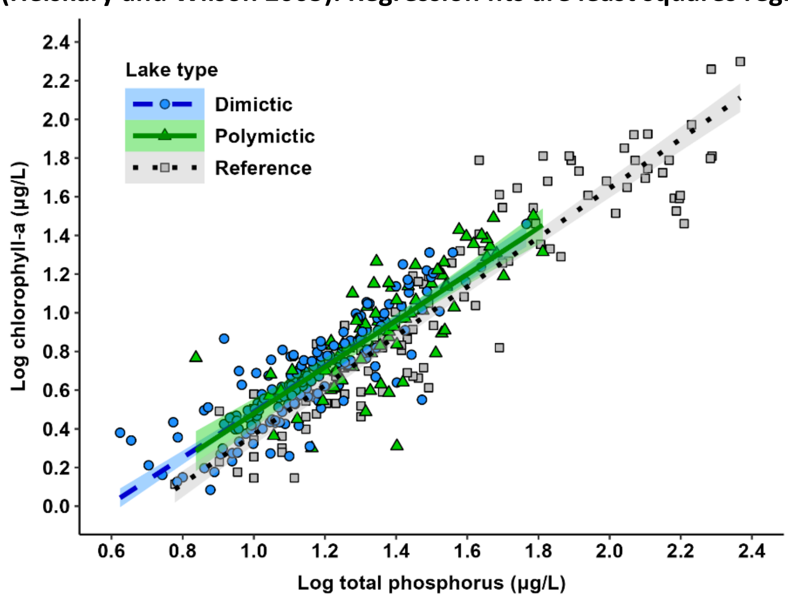
The development of Minnesota's 2008 lake eutrophication standards included regression models to predict the phosphorus concentrations necessary to meet chl-*a* targets (Heiskary and Wilson 2005). These analyses are repeated here to determine if these relationships differ between polymictic and dimictic northern lakes. Least squares regression using log-transformed data indicated that the relationships between these parameters were similar between dimictic and polymictic lakes (Figure 10). In general, these models predicted that polymictic lakes have slightly lower amounts of chl-*a* at the same TP concentration, but this difference is small especially at low concentrations of TP. In addition, the 90% confidence intervals for these models overlap along the entire TP gradient. This is observed for both the uncensored and censored datasets indicating that factors other than CDOM may be contributing to this small difference. All four models (dimictic and polymictic lakes using uncensored and censored data; Figure 10) estimate that a similar summer average TP concentration (24-28 µg/L) is needed to meet the current chl-*a* standard of 9 µg/L. This analysis indicates that it is not necessary to use different models for dimictic and polymictic lakes to estimate the TP concentrations needed to attain target chl-*a* concentrations.

Figure 10. Relationships between log-transformed, long-term summer average total phosphorus and chlorophyll-*a* for dimictic and polymictic northern lakes (1990-2020). Datasets include (A) all lakes and (B) lakes with color >73 PCU or a_{440} >4 m^{-1} censored. Regression fits are least squares regressions (all lakes dataset (A) - dimictic: $y = 1.1842x - 0.7275$, $F(1,588) = 1465$, p-value <0.0001, adjusted $R^2 = 0.71$; polymictic: $y = 1.1993x - 0.7821$, $F(1,300) = 415.7$, p-value <0.0001, adjusted $R^2 = 0.58$; censored dataset (B) - dimictic: $y = 1.1850x - 0.6962$, $F(1,174) = 523$, p-value <0.0001, adjusted $R^2 = 0.75$; polymictic: $y = 1.1978x - 0.7179$, $F(1,79) = 120$, p-value <0.0001, adjusted $R^2 = 0.60$).



The regression models for dimictic and polymictic northern lakes using the updated dataset were compared to the 2005 reference lake dataset model (Figure 11). Here the models using contemporary datasets with lakes censored for high CDOM are compared with the models used in Heiskary and Wilson (2005). As discussed above, the relationship between TP and chl-*a* is similar between dimictic and polymictic lakes for the contemporary dataset. These models were also similar to the reference lake dataset model (Figure 11). The 90% confidence intervals for these models overlap for part of the TP gradient, but in general, the reference lakes estimate lower amounts of chl-*a* at the same TP concentrations. This difference could be because the 2005 reference lake dataset included South and Central region lakes which are more likely to be impacted by other factors that could shade algae such as suspended sediment (EPA 2021a).

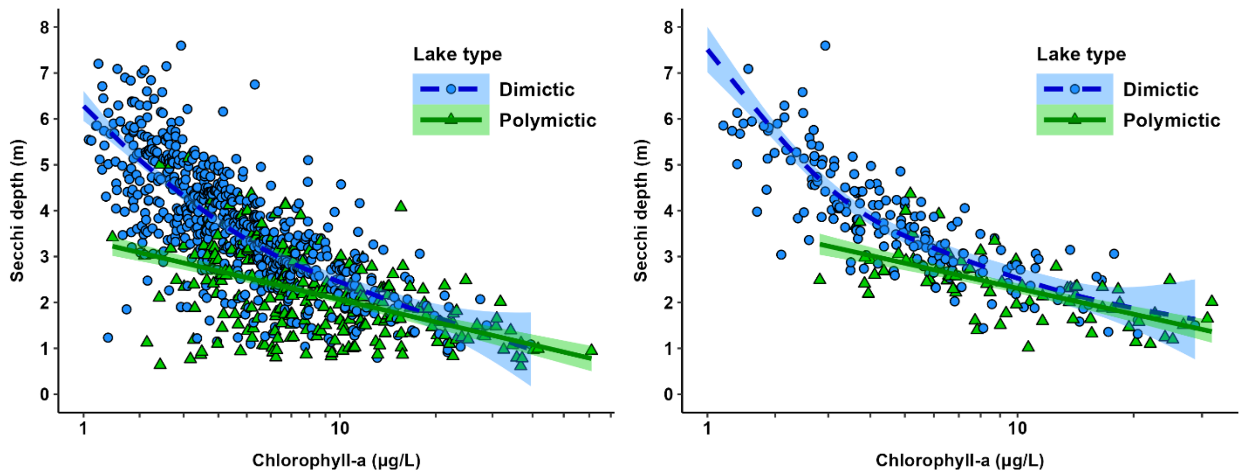
Figure 11. Log chlorophyll-*a* as a function of log total phosphorus for dimictic and polymictic northern lakes (1990-2020; lakes with color >73 PCU or a_{440} >4 m^{-1} censored) and for the reference lakes dataset (Heiskary and Wilson 2005). Regression fits are least squares regressions.



4. Chlorophyll-*a* and Secchi depth

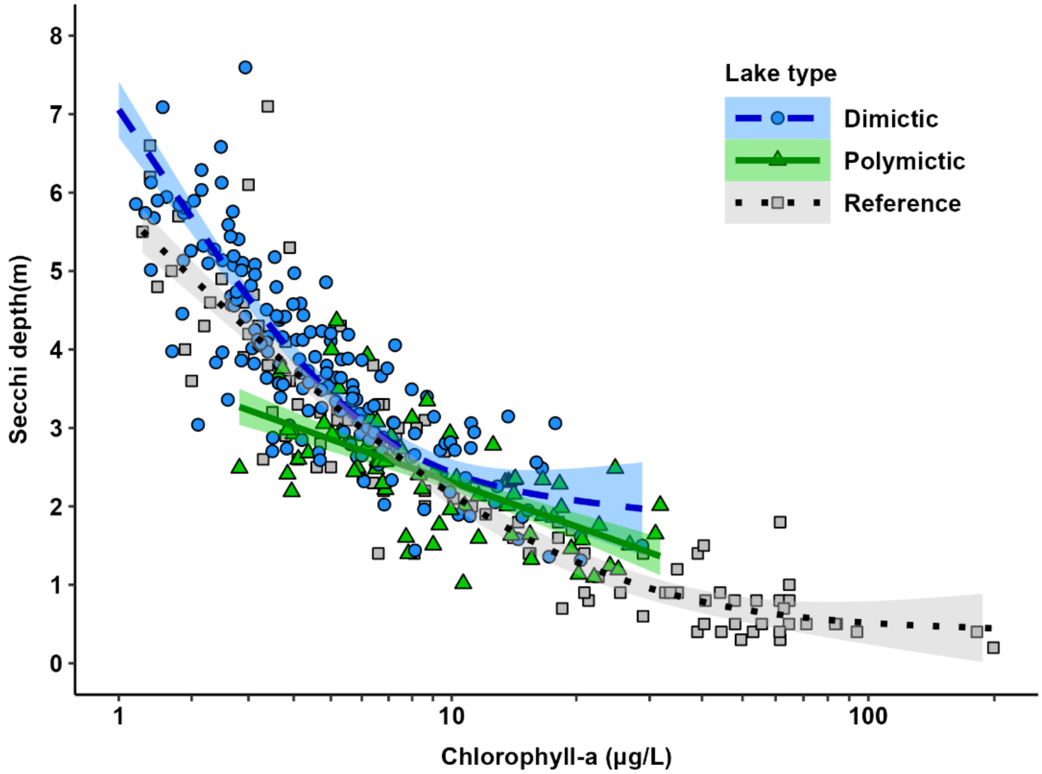
Unlike the relationship between TP and chl-*a* in northern lakes, an analysis of Secchi depth as a function of chl-*a* indicates this relationship is different between dimictic and polymictic lakes (Figures 12 and 13). Overall, Secchi depth is lower at equivalent chl-*a* concentrations in polymictic lakes compared to dimictic lakes especially at low chl-*a* concentrations (Figure 12). This pattern can in part be attributed to a limitation on Secchi depth by Z_{max} in polymictic lakes. However, as chl-*a* concentrations approach 10 µg/L, the relationship between chl-*a* and Secchi depth in dimictic and polymictic lakes is similar (Figure 12). The higher levels of CDOM in polymictic lakes could also affect this relationship; however, this general pattern also holds when lakes with high CDOM are censored (Figure 12B).

Figure 12. Relationship between log-transformed, long-term summer average chlorophyll-*a* and Secchi depth for dimictic and polymictic northern lakes (1990-2020). Datasets include (A) all lakes and (B) lakes with color >73 PCU or a_{440} >4 m^{-1} censored. Regression fits are generalized additive models (bs = "tp"; k = 5(A), 10(B); method = "REML").



A comparison of chl-*a* - Secchi depth models between the dimictic and polymictic northern lakes and the original (i.e., reference dataset) from Heiskary and Wilson (2005) indicated there are some differences. Overall, the chl-*a* - Secchi depth dataset model for the reference lakes falls between the dimictic and polymictic northern lakes (Figure 13). At low chl-*a* concentrations (<7 µg/L), the reference lake model is more similar to the dimictic northern lake model; although, the reference lake models estimate slightly lower Secchi depths likely due to the inclusion of both polymictic and dimictic lakes in the reference lake dataset. Above chl-*a* concentrations of ~10 µg/L, the reference lake model diverges from the northern lake models and has lower Secchi depth at similar chl-*a* levels. The phenomenon responsible for this pattern is not clear; although, factors other than algae (i.e., suspended sediment and CDOM) are likely important since the reference lake model also include lakes from the Central and Southern nutrient regions.

Figure 13. Long-term summer average Secchi depth as a function of chlorophyll-*a* for dimictic and polymictic northern lakes (1990-2020; lakes with color >73 PCU or a_{440} >4 m^{-1} censored) and reference lakes (Heiskary and Wilson 2005). Regression fits are generalized additive models (bs = "tp", k = 3, method = "REML").



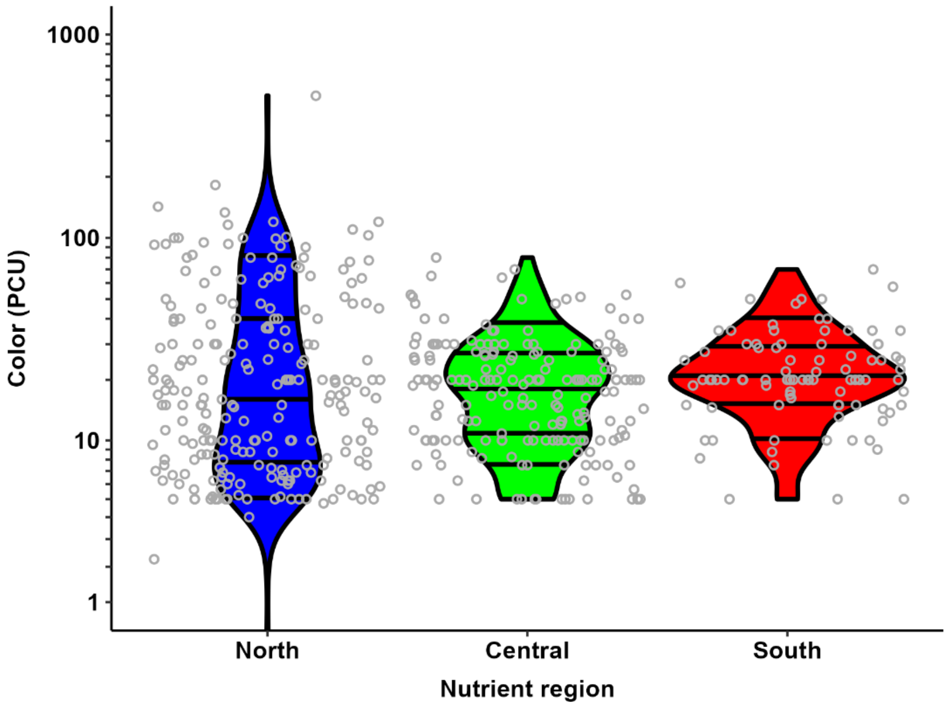
5. Secchi depth and colored dissolved organic matter

Light attenuation or scattering by any substance other than algal chlorophyll (i.e., non-algal turbidity) can cause errors in the estimates of chl-*a* from Secchi depth measurements (Carlson and Simpson 1996). As part of a detailed analysis of water quality in 299 reservoirs operated by the US Army Corp of Engineers, Walker (1982, 1985) determined that chl-*a* levels were more sensitive to nutrient concentrations in low turbidity reservoirs, and that light limitation effects were controlled primarily by non-algal turbidity in most impoundments. Non-algal turbidity often consists of CDOM from upstream watershed sources exported from wetland and forested landscapes. In highly colored lakes, transparency values are often lower than expected based on TP values and can result in false positive errors when assessing eutrophication with Secchi depth. The importance of considering CDOM when assessing Secchi depth may become more important due to increasing trends in CDOM (Roulet and Moore 2006).

The MPCA considers lakes with color values of >50 PCU as highly colored (MPCA 1999). Brezonik et al. (2019) determined that in lakes with high CDOM (>4 m^{-1} measured as absorptivity at 440 nm [a_{440}]), chl-*a* could not be accurately predicted from Secchi depth. Median values of color for lakes statewide are near 20 PCU for all three nutrient regions (Figure 14). However, North region lakes in Minnesota have more high color lakes compared to other regions of the state (Figure 14). This includes several, high-profile, high-quality lakes in the northern region, including Birch (69-0003-00) and White Iron (69-0004-00) lakes on the BWCA's Kawishiwi River system and Rainy (69-0694-00) and Vermilion (69-0378-00) lakes, which have levels of CDOM due to shallow bays that receive water from large, wetland dominated watersheds. The relatively large number of northern lakes with high CDOM and the effect of

CDOM on reducing Secchi depth, highlights the need to address CDOM as part of northern lake eutrophication standards if Secchi depth is part of the standard.

Figure 14. Violin plots of average color for lakes in Minnesota's nutrient regions. Description of violin plots: grey points = individual lake measurements; width of plot = kernel probability density; solid black lines = 50th percentile.



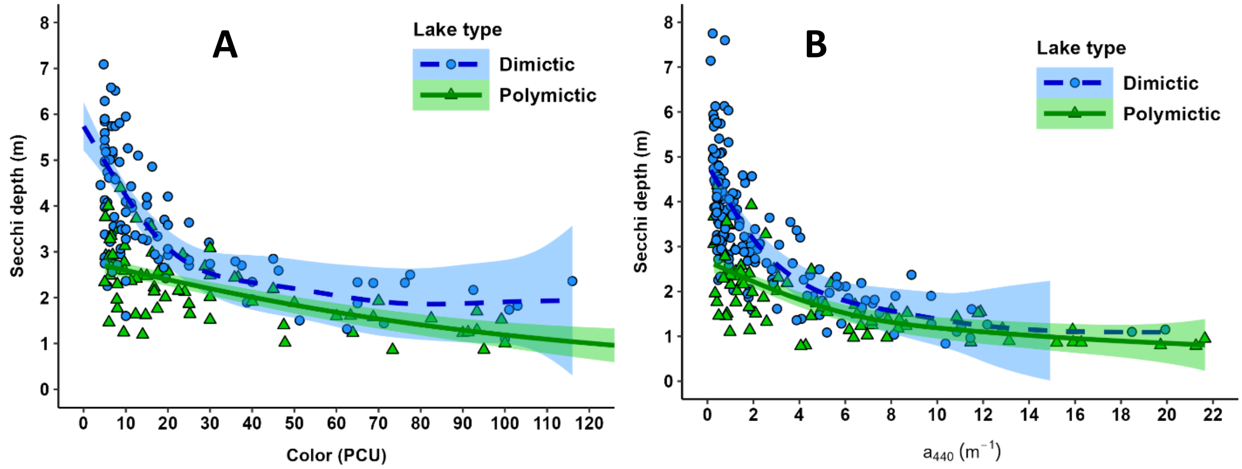
Light penetration is affected by CDOM which can influence the relationship between chl-*a* and Secchi depth (Garn and Parrott 1977, Brezonik et al. 2019). This effect is apparent in the Minnesota lakes dataset where Secchi depth declines and is apparently limited at high color levels (Figure 15). Heiskary and Wilson (2005) noted that in lakes with color below 50 PCU, there were no discernable effects of color on relationships between TP, chl-*a*, and Secchi depth. However, when the lake eutrophication standards were developed, the reference lake data included few lakes with color above 50 PCU and CDOM did not receive additional consideration in the standard. Brezonik et al. (2019) made use of more recent data from Minnesota, Wisconsin, and Michigan to estimate the effects of CDOM and other factors on Secchi depth. This research identified chl-*a*, suspended solids, and CDOM as the best predictors of Secchi depth although which of these factors were important differed among ecoregions. Relevant to eutrophication standards in northern Minnesota, Brezonik et al. (2019) determined that chl-*a* and CDOM were the most important factors for predicting Secchi depth in the NLF. This research also determined that in lakes with a_{440} values >~4 m^{-1}, CDOM has significant effects on Secchi depth, and that Secchi depth may not be a useful indicator of eutrophication in these lakes. When this a_{440} threshold was converted to color using the equation provided by Cuthbert and del Giorgio (1992), a value of 73 PCU was obtained. Northern lakes exceeding these colors or a_{440} thresholds have a low probability of attaining the northern lake Secchi depth standard. Logistic regression models for dimictic and polymictic northern lakes predicted that 100% of polymictic lakes and 66% of dimictic lakes will have a Secchi depth of less than 2.0 m when color is 73 PCU (Figure 16A). The Secchi depth exceedance rates were predicted to be lower when a_{440} was 4 m^{-1} with 65% of polymictic lakes and 30% of dimictic lakes not meeting the 2.0 m Secchi depth standard at this a_{440} value (Figure 16B). In many northern lakes, nonattainment of the Secchi depth standard is due to chl-*a* or a combination of CDOM and chl-*a*, but there are also many lakes where CDOM, largely independent of other factors, is an important limiting factor of Secchi depth.

Brezonik et al. (2019) included an equation for estimating Secchi depth based on chl-*a* and CDOM in NLF lakes. Using this we can estimate the a_{440} value at which the Secchi depth standard is not attained while the chl-*a* standard is met.

Eq. 1. log(Secchi depth) = 0.619 - 0.283 X log(chl-*a*) - 0.334 X log(a_{440}) (see Table 4 in Brezonik et al. [2019])

This model estimates that at Secchi depth of 2.0 m and chl-*a* of 9 µg/L, a_{440} will be 1.4 m^{-1}. Using the equation provided by Cuthbert and del Giorgio (1992), this can be converted to color giving 25 PCU. Using these CDOM thresholds, Minnesota lake datasets can be censored to determine if this reduces false positives with assessment. This was evaluated using receiver operating characteristic (ROC) curves by comparing false positive rates between the uncensored dataset and censored datasets (Figure 17). Secchi depth was an excellent or outstanding classifier[9] of for attainment of the chl-*a* standard for all datasets but censoring high CDOM lakes improved these determinations. Area under the curve (AUC) values for these datasets were as follows: all lakes = 0.8353; lakes with color < 73 PCU or a_{440} < 4 m^{-1} = 0.9454; lakes with color <25 PCU or a_{440} < 1.4 m^{-1} = 0.9600. This analysis demonstrates that Secchi depth is a good surrogate measure for chl-*a* for most lakes, but that caution is needed when assessing Secchi depth in lakes with high CDOM (>73 PCU or a_{440} values of >4.0 m^{-1}). The a_{440} value of >~4 m^{-1} provide by Brezonik et al. (2019) is likely to be the upper end where lakes are very unlikely to attain the Secchi depth standard of 2.0 m. Between a_{440} values of 1.4 and 4 m^{-1}, many lakes will attain the Secchi depth standard if chl-*a* is low, but there are some lakes where the combination of elevated chl-*a* and CDOM may result in an exceedance of the Secchi depth standard even when the chl-*a* standard is attained (Figure 16). The use of a_{440} values of 1.4-4.0 m^{-1} (or color values between 25 and 73 PCU) as a trigger to review data during the assessment process should reduce false positives and would make reliance on chl-*a* measurements more important to determine if the lake eutrophication standard is met.

Figure 15. Secchi depth as a function of (A) color and (B) absorbance at 440 nm for dimictic and polymictic northern lakes (1990-2020). Regression fits are generalized additive models (bs = "tp", k = 6(A), 10(B), method = "REML"). Shaded areas are 90% confidence intervals.



[9] AUC discrimination guidelines from Hosmer et al. (2013): 0.5-0.7 = poor; 0.7-0.8 = acceptable; 0.8-0.9 = excellent; >0.9 = outstanding.

Figure 16. Probability of Secchi depth not meeting the current lake eutrophication standard (2.0 m) as a function of (A) color and (B) absorbance at 440 nm for northern dimictic and polymictic lakes.Fits are generalized additive model (GAM) logistic regressions (bs = “tp”, k = 3, method = “REML”). Shaded areas are 90% confidence intervals and red dashed lines are possible color and a_{440} thresholds (color = 25 and 73 PCU; a_{440} = 1.4 and 4 m^{-1}) where color or a_{440} limits or affects attainment of the Secchi depth standard.

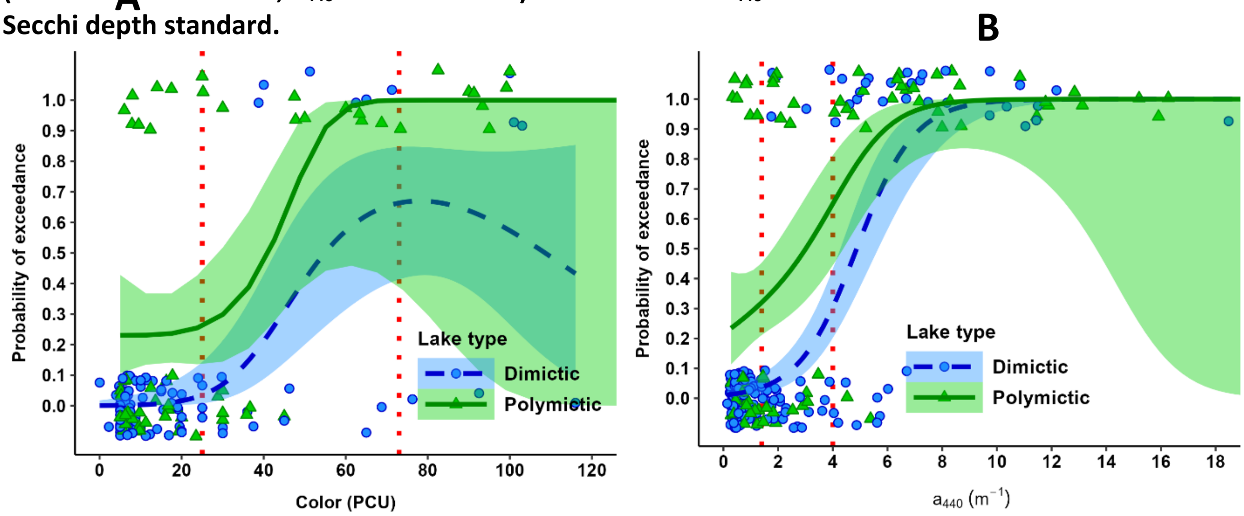
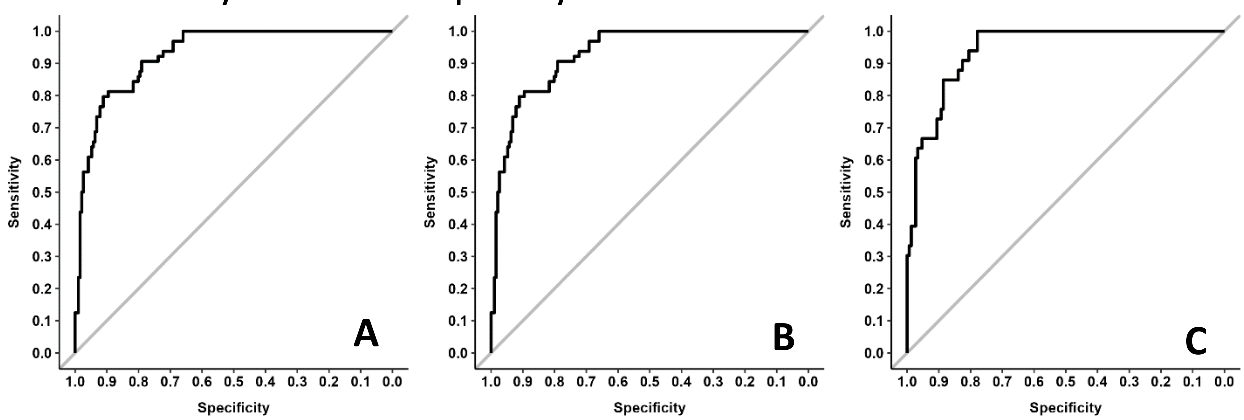


Figure 17. Receiver operating characteristic curves for predicting attainment of the current chlorophyll-*a* standard (9 µg/L) based on Secchi depth in northern lakes (1990-2020) for (A) all lakes (AUC = 0.8353), (B) lakes with color >73 PCU or a440 <4 m-1 censored (AUC = 0. 9454), and (C) lakes with color >25 PCU or a440 <1.4 m-1 censored (AUC = 0.9600). Specificity refers to the true negativity rate and sensitivity refers to the true positivity rate.



6. Nitrogen

The scope of this revision was narrowly focused on reviewing and revising the existing lake eutrophication standards for the North region in Minnesota. As a result, a nitrogen standard for lakes was not examined as part of this research. In addition, nitrogen water quality data are limited from Minnesota lakes which restricts analysis of the impact of nitrogen on eutrophication in these lakes. EPA’s recommended lake eutrophication standards (EPA 2021a) include models for determining candidate nitrogen criteria. However, these models also require DOC to model nitrogen criteria from chl-*a* targets and there is limited DOC data available from Minnesota lakes. Development of nitrogen standards for Minnesota lakes should be reviewed in the future to determine if it is appropriate and feasible to adopt nitrogen lake eutrophication standards statewide.

ii. Reference condition

1. Background

Reference condition analyses have been used as part of the development of water quality standards and were part of the analyses used to support Minnesota's current lake eutrophication standards (Heiskary and Wilson 2005). Two reference condition approaches for identifying candidate nutrient values in lakes were provided in EPA (2000): 1) calculation of the 75th percentile from reference lakes or 2) calculation of the 25th percentile of median values for eutrophication parameters from all lakes. The second approach is not appropriate for northern lakes due to the overall low disturbance in the region. The first method is more applicable but, it does not explicitly link criteria to attainment of beneficial use endpoints (EPA 2021). The EPA has now replaced the recommended lake nutrient criteria in EPA (2000) with EPA (2021). However, reference condition analyses can still provide insights into the overall condition of northern lakes in Minnesota and how this condition compares to candidate criteria.

2. Methods

Datasets of TP, chl-*a*, and Secchi depth (1990-2020) were compiled for northern dimictic and polymictic lakes with watershed disturbance information. Watershed disturbance was calculated as the percent of the watershed with pasture, cultivated crops, urban, and mining land uses based on the 2016 National Land Cover Database (NLCD; http://www.mrlc.gov/). Lakes with a watershed disturbance of more than 25% were eliminated as these were not considered to be reference lakes. Summer average values were calculated for TP, chl-*a*, and Secchi depth from lakes with at least two years of water quality data and at least four measurements collected each year during the summer index period (June-September). Distributions of eutrophication parameters between dimictic and polymictic lakes were compared for two different datasets: 1) all reference lakes and 2) reference lakes with color less than 25 PCU or a_{440} less than 1.4 m^{-1}. Comparisons were made using violin plots and differences between dimictic and polymictic lakes were tested using a Mann-Whitney test ("wilcox.test" function; R Core Team 2020).

Following methods described in EPA (2000) and used by Heiskary and Wilson (2005), the 75th percentile of TP and chl-*a* and the 25th percentile of Secchi depth for reference sites was calculated. The 75th and 25th percentiles of reference lakes were used as a safety factor because the reference site selection process is imperfect, and some reference lakes may not be truly reference or there may be other natural characteristics which make them unsuitable for the reference conditional analysis. Although this study does not rely on a reference condition analysis for criteria setting, it was used to compare eutrophication parameters between dimictic and polymictic reference lakes to assess if under minimally disturbed conditions, trophic status differed between these lake types. In addition, the 75th percentile of the reference dataset for chl-*a* was compared to thresholds determined from the beneficial use indicator analyses.

3. Results

Comparison of watershed disturbance between dimictic and polymictic northern lakes demonstrated that these two lake populations are similar in terms of anthropogenic disturbance (Figure 18). There was a significant, but probably not important, difference between these two populations (Mann-Whitney U Test: W = 353233, p-value = 0.0140) and polymictic lakes had on average lower disturbance levels in their watersheds (Table 8). In the North region, the percent of minimally disturbed lakes is high with 99.7% of dimictic lakes and 95.4% of polymictic lakes with watershed disturbance less than 25%. Since disturbance is low and similar between these lake types, differences in water quality characteristics can likely be attributed to natural differences between these lake types.

Figure 18. Comparison of percent watershed disturbance for dimictic and polymictic northern lakes using (A) violin plots and (B) cumulative distribution plots. Description of violin plots: grey circles = individual lakes; width of plot = kernel probability density; solid black lines = 10th, 25th, 50th, 75th, and 90th percentiles.

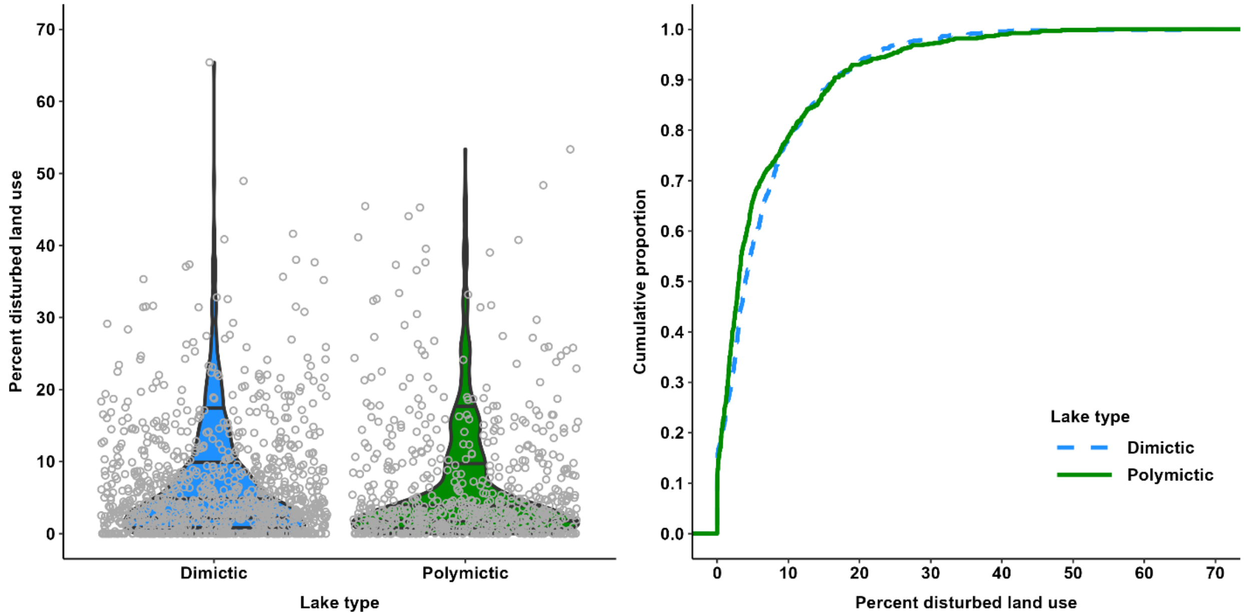


Table 8. Summary statistics for percent disturbed land use for northern region lakes (abbreviations: SD = standard deviation; MW = Mann-Whitney test).

Parameter	Lake type	N	Mean	SD	Percentile					MW test
					10	25	50	75	90	
% disturbed land use	Dimictic	983	6.6	7.6	0.0	1.4	4.0	8.8	16.4	W = 353233; p-value = 0.0140
	Polymictic	671	6.3	8.3	0.0	1.1	3.0	8.5	16.5	

There was a significant difference between reference dimictic and polymictic lakes for all three lake eutrophication parameters for both the uncensored lake dataset and the high CDOM censored lake dataset (Figure 19 and Table 9). However, the Secchi depth comparison is of limited use because Secchi depth is a function of lake depth, and this physical attribute creates an upper limit for polymictic lakes. For both reference lake datasets and all three eutrophication parameters, dimictic lakes had lower trophic condition measures compared to polymictic lakes (Table 9). Similarly, Havens and Nürnberg (2004) determined that TP and chl-*a* were higher in polymictic lakes in temperate North America despite greater CDOM levels in these lakes. The reference lake analysis also demonstrated that the current standards applied to northern polymictic lakes are likely too restrictive based on the relatively high non-attainment of eutrophication standards in lakes with low watershed disturbance. The reference lake distributions for eutrophication parameters can also be used to assess the suitability of thresholds developed from beneficial use indicator analyses.

Figure 19. Violin plots of average (A,D) total phosphorus, (B,E) chlorophyll-*a*, and (C,F) Secchi depth for reference (watershed disturbance < 25%) dimictic and polymictic northern lakes (1990-2020).Datasets include (A-C) lakes that were not censored high CDOM and (D-F) lakes that were censored for high CDOM (color < 73 PCU or a_{440} < 4 m^{-1}). Description of violin plots: grey circles = individual lake measurements; width of plot = kernel probability density; solid black lines = 10th, 25th, 50th, 75th, and 90th percentiles.

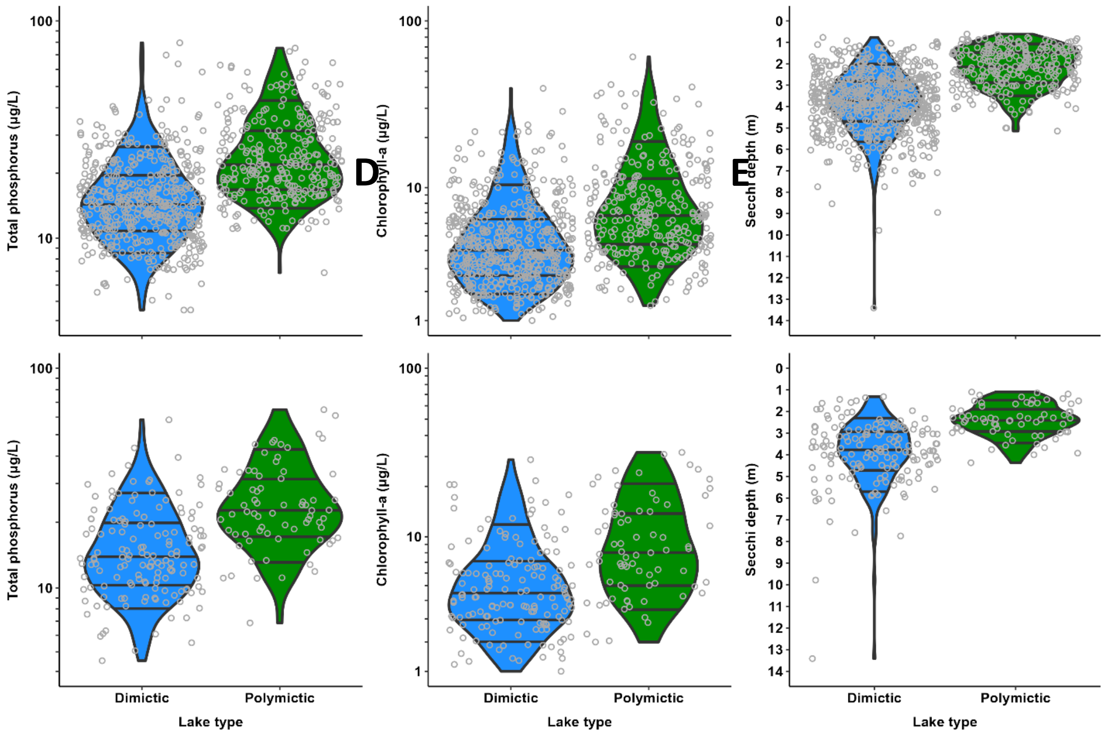


Table 9. Summary statistics for eutrophication parameters for northern references lakes(watershed disturbance < 25%; abbreviations: SD = standard deviation; MW = Mann-Whitney test; CDOM = colored dissolved organic matter).

Parameter	Lake type	N	Mean	SD	Percentile					MW test
					10	25	50	75	90	
Uncensored data										
Total phosphorus (µg/L)	Dimictic	472	16.3	8.5	8.6	11.0	14.4	19.5	26.5	W = 29932; p-value <0.0001
	Polymictic	283	25.7	12.3	14.6	16.7	21.9	31.7	41.4	
Chlorophyll-*a* (µg/L)	Dimictic	500	5.3	4.3	1.9	2.7	3.9	6.3	10.3	W = 41293; p-value <0.0001
	Polymictic	291	9.3	7.9	3.2	4.4	6.7	11.0	18.9	
Secchi depth (m)	Dimictic	471	3.7	1.4	1.9	2.7	3.6	4.6	5.5	W = 130421; p-value <0.0001
	Polymictic	255	2.2	1.0	1.0	1.5	2.1	2.9	3.5	
Lakes censored for high CDOM (color < 73 PCU or a_{440} < 4 m^{-1}).										
Total phosphorus (µg/L)	Dimictic	144	16.0	8.2	8.6	10.3	13.7	19.8	26.9	W = 2358; p-value = 0.0001
	Polymictic	74	25.8	11.9	13.3	17.7	22.5	32.2	44.7	
Chlorophyll-*a* (µg/L)	Dimictic	150	5.8	4.5	2.0	3.0	4.4	6.7	11.2	W = 2982; p-value <0.0001
	Polymictic	74	10.4	7.2	3.8	5.0	7.9	14.1	21.6	
Secchi depth (m)	Dimictic	144	3.8	1.2	2.3	2.8	3.7	4.5	5.4	W = 8690; p-value <0.0001
	Polymictic	69	2.4	0.8	1.5	1.8	2.3	2.9	3.5	

iii. Paleolimnology

Paleolimnological studies are useful for documenting background lake productivity or reviewing the applicability of eutrophication standards to specific lakes (e.g., VanderMeulen et al. 2016). Previous studies have demonstrated that on a population level, lakes in the North region are near historical conditions whereas lakes in the Central and South regions have trophic conditions elevated from background levels (Heiskary and Swain 2002, Ramstack et al. 2003). Such results can be used as part of site-specific analyses for individual lakes or used as part of water quality standards development. Here, the latter approach is used to determine if background trophic conditions differ between dimictic and polymictic lakes. The determination that current trophic conditions in northern lakes reflect background conditions is also valuable for putting eutrophication criteria in context.

Paleolimnological data were reanalyzed from three studies: Edlund (2005), Edlund et al. (2016), and Edlund et al. (2021). Edlund (2005) provided an analysis of differences between historical and modern TP concentrations for dimictic and polymictic northern lakes and concluded that modern TP concentrations were similar to historical conditions in both dimictic and polymictic lakes and that dimictic lakes had lower background levels of TP compared to polymictic lakes. The current study reanalyzed these data and included data from Edlund et al. (2016) and Edlund et al. (2021) (Table 10). This analysis was redone because the original dataset did not use the same polymictic (i.e., shallow) lake definition[10] used here to categorize lakes as dimictic or polymictic. Two lakes, Red Sand Lake (Edlund 2005) and Lac La Belle (Edlund et al. 2016), were removed from analyses due to uncertainties in the reconstructed TP concentrations caused by unusual diatom communities that did not have modern analogs. The resulting revised dataset included 8 polymictic and 16 dimictic northern lakes.

There was a significant difference between populations of polymictic and dimictic lakes (Mann-Whitney U Test: $W = 24.5$, $p\text{-value} = 0.0166$; Figure 20A). This analysis supports the determination of Edlund (2005) that TP concentrations are naturally higher in polymictic lakes compared to dimictic lakes in the North region. In addition to describing a difference between dimictic and polymictic northern lakes, Edlund (2005) also determined that for both populations of northern lakes, current nutrient conditions are similar to historical, pre-European conditions. The eight polymictic northern lakes used in the reanalysis indicated good correlation between modern and historical phosphorus concentrations (Figure 20B) which supports the conclusion of Edlund (2005). Therefore, we can infer that the current trophic status for most northern lakes is near background conditions and that modern data can be used to categorize these conditions.

Paleolimnological data were not used here as a line of evidence for setting criteria because they do not provide TP concentrations which are linked to the protection of beneficial use endpoints. However, the paleolimnological data were used to establish that under natural conditions, dimictic and polymictic lakes in the North region have different TP concentrations. This demonstrates that different standards may be appropriate because these lakes have naturally different trophic conditions which support different biological communities and recreation uses. It is also important to note that historical TP concentrations for four of the eight polymictic lakes in this dataset were above the current lake eutrophication standard (30 µg/L). This is further evidence that many polymictic lakes have nutrient levels that naturally do not meet the current lake eutrophication standard which supports the need for a revised standard.

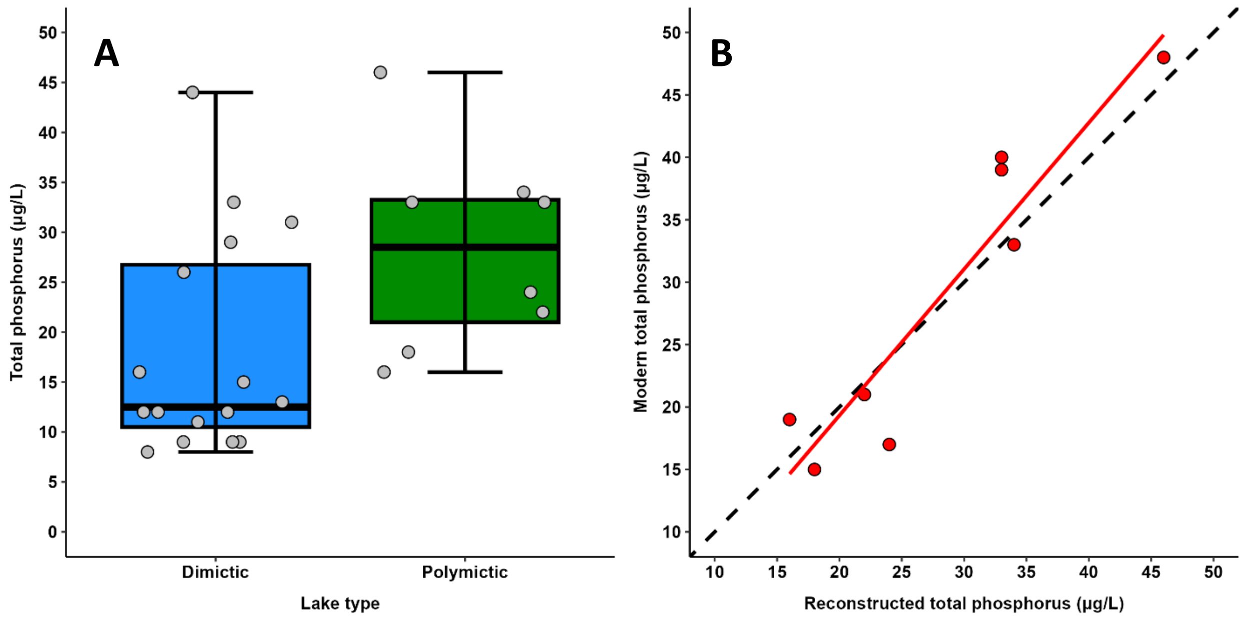
[10] Edlund (2005) defined shallow lakes as being <6 m deep or with a high littoral extent.

Table 10. Modern and diatom reconstructed total phosphorus concentrations for dimictic and polymictic northern lakes (Z_{max} = maximum depth; TP = total phosphorus; WID = Waterbody identification code).

Lake name	WID	Group	Z_{max} (m)	Pre-European settlement TP (µg/L)	Modern TP (µg/L)	Source
Snells	31-0569-00	Dimictic	15.2	12	24	Edlund (2005)
Long	31-0570-00	Dimictic	22.9	15	13	Edlund (2005)
Loon	31-0571-00	Dimictic	21.0	16	11	Edlund (2005)
Little Bass	31-0575-00	Dimictic	18.9	26	13	Edlund (2005)
Wilson	38-0047-00	Dimictic	14.9	44	13	Edlund (2005)
Windy	38-0068-00	Dimictic	11.9	9	12	Edlund (2005)
Nipisiquit	38-0232-00	Dimictic	5.5	33	16	Edlund (2005)
Wolf	38-0242-00	Dimictic	7.3	12	14	Edlund (2005)
Bear	38-0405-00	Dimictic	8.8	12	11	Edlund (2005)
Tooth	69-0756-00	Dimictic	13.1	9	12	Edlund (2005)
Loiten	69-0872-00	Dimictic	14.9	8	8	Edlund (2005)
Locator	69-0936-00	Dimictic	15.9	9	9	Edlund (2005)
Bean	38-0409-00	Dimictic	7.9	29	17	Edlund (2005)
Little Trout	69-0682-00	Dimictic	29	11	7	Edlund (2005)
Dyers	16-0634-00	Dimictic	6.1	31	27	Edlund (2005)
Ninemile	38-0033-00	Dimictic	9.1	13	17	Edlund (2005)
Lac La Belle	09-0011-00	Polymictic	unknown	*	40	Edlund et al. (2016)
Platte	18-0088-00	Polymictic	7.0	34	33	Edlund (2005)
Red Sand	18-0386-00	Polymictic	7.0	*	33	Edlund (2005)
Forsythe	31-0560-00	Polymictic	3.1	22	21	Edlund (2005)
Tetagouche	38-0231-00	Polymictic	4.6	24	17	Edlund (2005)
August	38-0691-00	Polymictic	5.8	18	15	Edlund (2005)
Net	58-0038-00	Polymictic	3.7	33	39	Edlund et al. (2016)
Shoepack	69-0870-00	Polymictic	7.3	16	19	Edlund (2005)
Long	69-0495-00	Polymictic	4.3	46	48	Edlund et al. (2021)
Strand	69-0529-00	Polymictic	4.9	33	40	Edlund et al. (2021)

***The diatom community from these lakes did not have a modern analog in the dataset and are excluded due to uncertainty in the reconstructed concentrations.**

Figure 20. Paleolimnological data from polymictic and dimictic lakes (data from Edlund (2005), Edlund et al. (2016), and Edlund et al. (2021)). (A) Box plots of diatom-inferred total phosphorus concentrations for pre-European settlement periods (1800s or earlier) from dimictic and polymictic northern lakes (plot description: grey circles = individual lake measurements; upper and lower hinges = 25th and 75th percentiles; whiskers extend from the hinge to the largest/smallest value no further than 1.5 * interquartile range from upper and lower hinges). (B) Comparison of total phosphorus concentrations from diatom-inferred reconstruction and modern water column summer averages for polymictic northern lakes (plot description: red solid line = least squares regression (R^2 = 0.88); black dashed line = 1:1 reference line).



Beneficial use thresholds

Three beneficial use endpoints for aquatic life and recreation were used to identify eutrophication thresholds for northern lakes: aquatic macrophytes (aquatic life), fish (aquatic life), and recreation suitability (recreation). These endpoints were selected because they encompass both beneficial use types in Class 2 (i.e., aquatic life and recreation) and because there are extensive datasets for these uses from both dimictic and polymictic lakes in Minnesota. The advantage of using multiple endpoints in standards development is that they can be used to identify the most sensitive endpoint to ensure the protection of beneficial uses applicable to northern lakes. Preliminary analyses indicated that the disturbance gradient was too truncated in the North region for threshold development, and it was necessary to include Central region lakes. South region lakes were excluded because of the high watershed disturbance compared to the North region. As a result, the dataset used for threshold analyses only included lakes from the North and Central regions. All three beneficial use endpoints were analyzed using similar methods to identify chl-*a* thresholds consistent with a low probability of non-attainment of each endpoint. Analysis of these three endpoints was also used to justify the need for separate standards for dimictic and polymictic northern lakes.

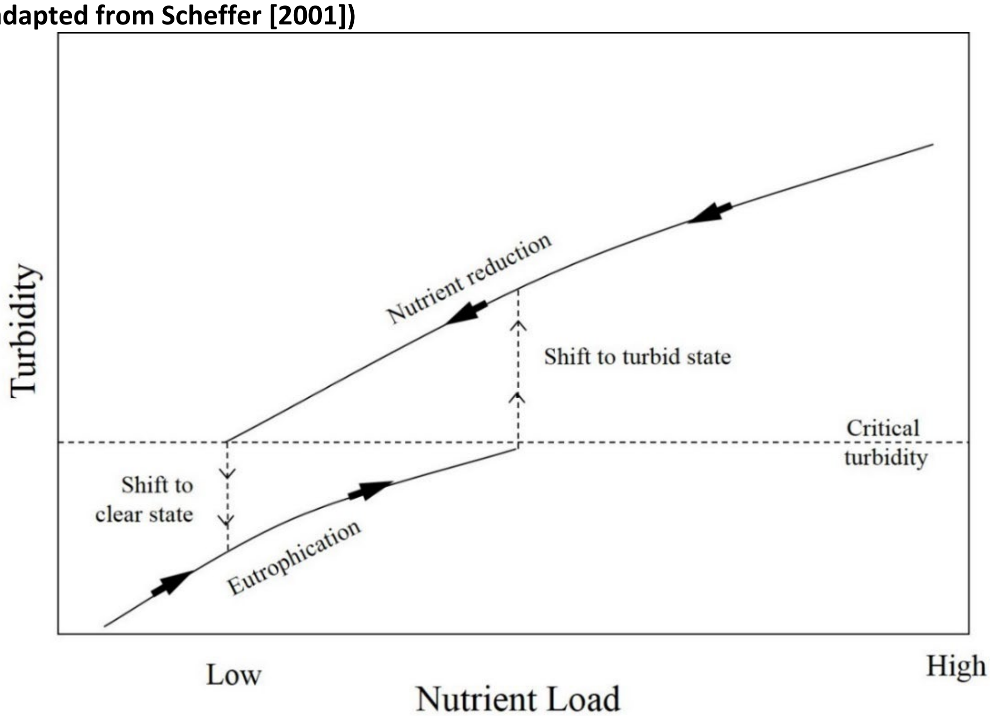
i. Aquatic Life Use

1. Aquatic macrophytes

Background

The maintenance of a healthy macrophyte community is important in most lakes including polymictic lakes and in the shallow areas of many dimictic lakes. However, macrophytes are often one of the defining attributes of polymictic or shallow lakes, and the alternative equilibria between macrophyte and algal dominated systems is often used as an indicator of lake condition in shallow lakes. Contrasting stable states in polymictic lakes alternate between clear water and abundant macrophytes to turbid water dominated by algal blooms and reduced aquatic habitat (Scheffer et al. 1993, Yuan 2021; Figure 21). The assumptions behind these stable states are that turbidity increases with nutrient level, submerged vegetation reduces turbidity, and vegetation disappears when a critical turbidity is exceeded (Scheffer et al. 2001, Vitense et al. 2021). In general, un-impacted polymictic lakes have clear water and a rich and diverse aquatic macrophyte community. Restoration of non-vegetated, turbid shallow lakes from the algal dominated state (principally cyanobacteria) is notoriously difficult. External loading reductions may have little effect on restoration since a large amount of phosphorus has been adsorbed by the sediments and internal loading often compensates for reductions in external sources (Scheffer 2004). The influence of turbulence and macrophytes in polymictic lakes has opposite effects (Scheffer 2004) where turbulence prevents excessive anaerobic phosphorus release by oxidizing the sediment surface, but it also promotes diffusion of phosphorus from the aerobic top sediment into the water. Macrophytes reduce turbulence, which enhances the probability that anaerobic conditions occur at the sediment surface, but they prohibit resuspension and limit diffusion of phosphorus out of the sediment. In Minnesota, this shift from a clear-water state to a turbid state has been observed in many lakes and rarely management actions have been successful in returning such lakes to the clear-water state (Perleberg et al. 2023). In general, a primary goal of lake eutrophication standards in polymictic lakes is to sustain trophic conditions that support the maintenance of a clear-water state, and in most Minnesota lakes, this includes the presence of a healthy macrophyte community.

Figure 21. Alternative stable states in polymictic lakes (from Emmons and Oliver Resources [2010] and adapted from Scheffer [2001])



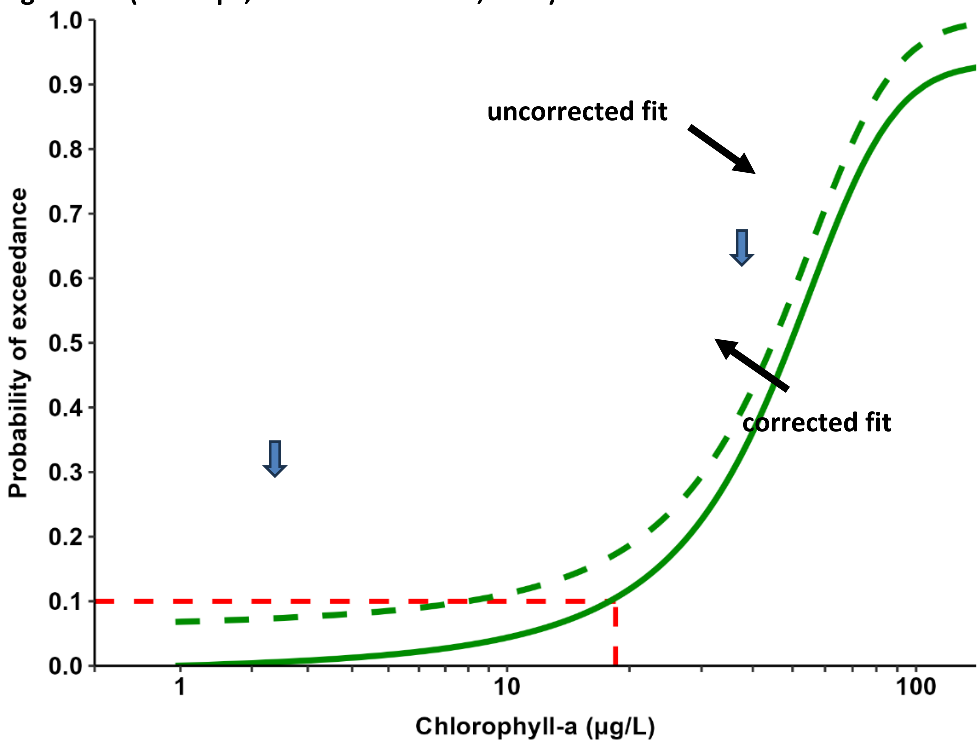
Aquatic macrophytes have been demonstrated to be a good indicator of trophic condition and their assessment provides a direct measurement of aquatic life uses (Radomski and Perleberg 2012). Macrophyte integrity, measured by the Floristic Quality Index (FQI), was shown to be negatively correlated with TP and chl-*a* in 27 west-central Minnesota Lakes (Heiskary and Lindon 2005). Similar findings have been reported in other studies of Minnesota Lakes (Beck et al. 2010, Hansel-Welsh et al. 2003, Radomski and Perleberg 2012). One of the most important factors influencing macrophyte community health is water transparency, which in northern Minnesota lakes is largely driven by lake productivity (i.e., increased algal production) and CDOM (Cheruvelil and Soranno 2008, Radomski and Perleberg 2012, Brezonik et al. 2019). As a result, aquatic macrophyte health in many lakes is associated with lake trophic condition which makes macrophytes an excellent indicator of the impacts of nutrient loading. Healthy aquatic macrophytes also support other beneficial uses such as waterfowl, fish, and invertebrates by providing habitat and food resources for these organisms (Radomski and Perleberg 2012). Another advantage of monitoring aquatic macrophytes is that they can be monitored in most shallow lakes where other indicators such as fish are not appropriate due to naturally limiting conditions (e.g., limited fish habitat, winterkill conditions).

An index for assessing the integrity of aquatic macrophytes was developed using Minnesota lake data by Radomski and Perleberg (2012). An advantage of this index is that it can make use of several aquatic plant survey types used in Minnesota. The result is a large database of Minnesota lakes with aquatic plant index scores which can be used to analyze attainment of aquatic life use goals. The Radomski and Perleberg (2012) index includes two metrics for determining the condition of the plant assemblage: (1) taxa richness and (2) FQI. Impairment thresholds for these two indices were also developed to specifically address nutrient impairments (Radomski and Perleberg 2012). The aquatic macrophyte threshold framework aligns with the MPCA's lakes standards in that it uses ecoregions and lake depth to set different metric expectations for lakes. The approach assigned different thresholds depending on the sampling methodology and created a matrix of thresholds to account for these differences (Radomski and Perleberg 2012). As a result, raw index scores between models are not necessarily equivalent in terms of attainment of aquatic life goals and were normalized in relation to thresholds. These normalized scores are called "relative macrophyte scores", and lakes with relative macrophyte scores of less than 0 do not meet the macrophyte goals.

Analysis

Average relative macrophyte index scores (1993-2018) were paired with long-term summer average chl-*a* data (i.e., ≥ 4 samples/year and ≥ 2 years; 1990-2020). For dimictic and polymictic lakes, the relationships between relative taxa richness or FQI scores and chl-*a* were modeled using GAMs. Logistic regression analysis was used to model the probability of attaining macrophyte thresholds at different concentrations of chl-*a*. To run the logistic regression analysis, an exceedance was assigned to lakes when either or both macrophyte indices (i.e., taxa richness or FQI) were below the thresholds in Radomski and Perleberg (2012). Logistic regression models used GAMs with a logistic link function. Generalized additive models were run in R version 4.0.3 (R Core Team 2020) using the "mgcv" package (Wood 2017) for both analyses. Even at low levels of nutrients (e.g., <5 µg/L), these models predicted some level of exceedance of macrophyte goals. Due to the low concentration of chl-*a* in these lakes, exceedances are likely to not be related enrichment, but rather due to other stressors (e.g., invasive species, lakeshore development, and suspended sediment) or chl-*a* or macrophyte sampling error and variability. To correct for these effects, the initial exceedance rate of the model (i.e., the modeled exceedance rate at the minimum chl-*a* concentration in the dataset) was subtracted from the model (Figure 22). In the case of the macrophyte logistic regression model for polymictic lakes, the lowest chl-*a* value in the dataset was 1 µg/L and at this value, the model estimated a 6.8% exceedance rate. This rate was subtracted from the model such that the initial exceedance of the model was 0% (Figure 22). From the logistic regression model, a 10% exceedance rate was used to interpolate a protective chl-*a* concentration. A 10% exceedance rate was selected to account for natural lake characteristics (e.g., naturally enriched lakes, lakes with atypical hydrology or morphology, lakes with high CDOM), lakes with more sensitive macrophyte communities compared to the population, and the combined effects of the lake productivity and other stressors. The 10% exceedance rate is a reasonable and practical threshold for northern lakes where the combined effect of other stressors is expected to be low.

Figure 22. Comparison of uncorrected and corrected logistic regression models for the exceedance of macrophyte index thresholds as a function of total chlorophyll-*a* for polymictic lakes (northern and central region lakes). Exceedance was assigned to lakes that do not meet either FQI or taxa richness thresholds (Radomski and Perleberg 2012) or both. Fit is a generalized additive model (GAM) logistic regression (bs = "tp", method = "REML", k = 3).



Results

Relative macrophyte index scores were significantly related to chl-*a* concentrations (Figure 23; FQI, polymictic lakes: adjusted R^2 = 0.48, p value <0.0001; FQI, dimictic lakes: adjusted R^2 = 0.31, p value <0.0001; taxa richness, polymictic lakes: R^2 = 0.42, p value <0.0001; taxa richness, dimictic lakes: R^2 = 0.24, p value <0.0001). There was a small difference between the polymictic and dimictic lake models with the 90% confidence intervals overlapping throughout most of the chl-*a* gradient. There was some separation of the different lake type models between chl-*a* concentration of 10 and 30 µg/L (Figure 23).

The rate of polymictic lakes exceeding the macrophyte index thresholds increased rapidly above a chl-*a* concentration of 20 µg/L (Figure 24). A 10% exceedance of macrophytes in polymictic and dimictic lakes was interpolated to correspond to chl-*a* concentrations of 18 and 13 µg/L, respectively (Figure 24). Based on this analysis; to protect aquatic life as measured by aquatic macrophytes in northern polymictic lakes, a chl-*a* threshold of 18 µg/L is recommended. The current chl-*a* standard for northern lakes is 9 µg/L which indicates this existing standard is sufficient to protect aquatic macrophyte uses in northern dimictic lakes.

Figure 23. Relative A) floristic quality index and B) taxa richness as a function of chlorophyll-*a* for dimictic and polymictic lakes. Macrophyte scores are scaled in relation to thresholds provided in Radomski and Perleberg (2012). Points represent average lake values for chlorophyll-*a* (1990-2020) and macrophyte index scores (1993-2018). Datasets are from northern and central region lakes. Fits are generalized additive models (bs = "tp", k = 10, method = "REML") and shaded areas are 90% confidence intervals.

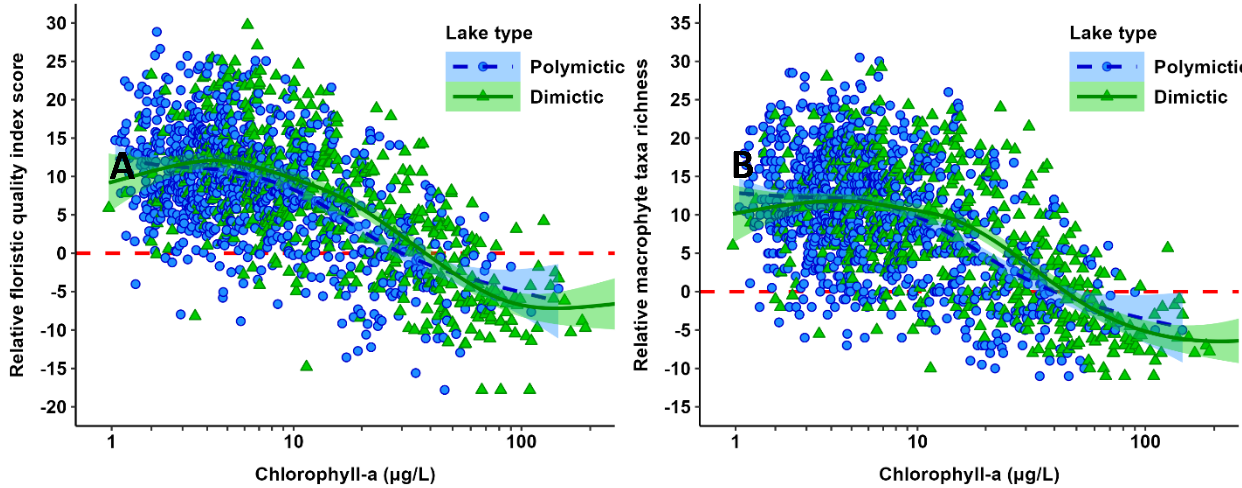
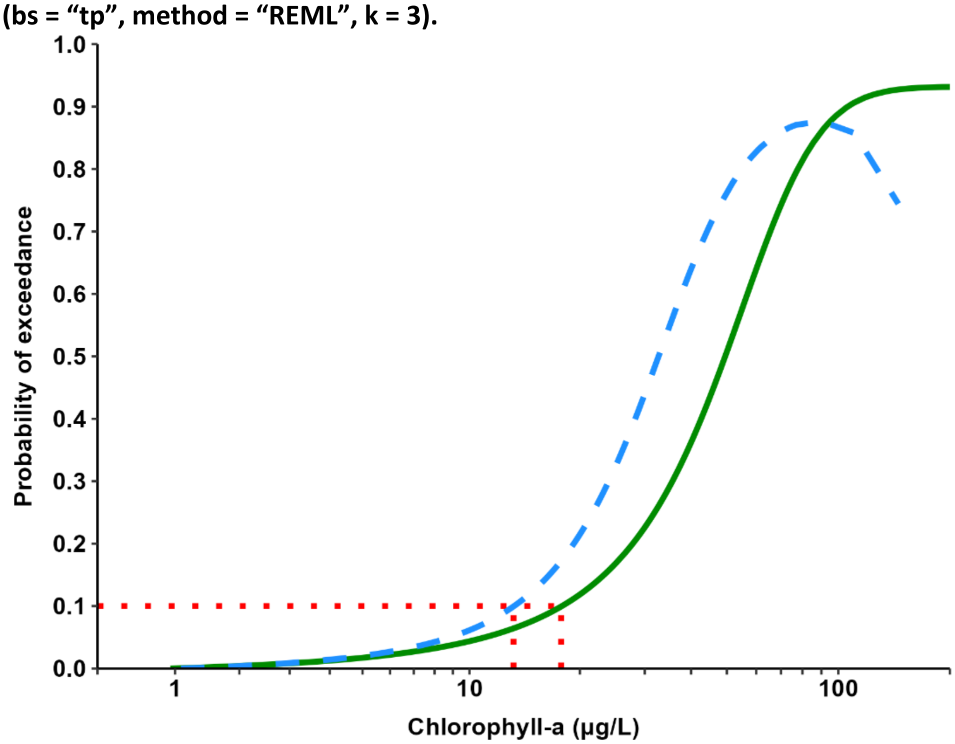


Figure 24. Exceedance of macrophyte index thresholds as a function of total chlorophyll-*a* for polymictic (green solid line) and dimictic (blue dashed line) lakes (northern and central region lakes). Exceedance was assigned to lakes that do not meet either FQI or taxa richness thresholds (Radomski and Perleberg 2012) or both. Red, dotted line shows interpolation of chlorophyll-*a* thresholds from a 10% probability of exceedance. Fits are generalized additive model (GAM) logistic regression (bs = "tp", method = "REML", k = 3).



Wild rice

Wild rice is biologically and culturally important and is a component of many macrophyte communities in northern and central Minnesota lakes (Hansen 2008). The distribution of wild rice in Minnesota and

Wisconsin has declined (Pillsbury and McGuire 2009), and this species faces threats from climate change and water quality degradation. Some specific factors that contribute to declines in wild rice stands include watershed hydrologic changes, pore-water sulfide, and declines in water transparency (Myrbo et al. 2017). Since increased productivity in lakes often results in greater levels of suspended algae and decreased water transparency, it is relevant to consider eutrophication thresholds for northern polymictic lakes and if they are sufficient to protect wild rice.

To assess the potential impact of revisions to northern lake standards, the probability of a lake supporting wild rice along a gradient of chl-*a* was modeled with GAMs using a logistic link function in R version 4.0.3 (R Core Team 2020) with the "mgcv" package (Wood 2017). The presence of wild rice was determined from the MNDNR's aquatic plant survey database where a lake with *Zizania* identified in any survey was coded as a lake supporting wild rice. Using the same logistic regression model methods, the probability of a lake supporting wild rice was also modeled as a function of macrophyte taxa richness and floristic quality index. To reduce the effect of different survey methodologies, relative macrophyte scores were used. The CDF of wild rice along the chl-*a* gradient was also plotted using the "stat_ecdf" function in the "ggplot2" package R version 4.0.3 (R Core Team 2020; Wickham 2016) and the 95% extirpation concentration of chl-*a* was calculated from this distribution.

Logistic regressions for polymictic and dimictic lakes indicated that the probability of wild rice occurrence as a function of chl-*a* and Secchi depth had a unimodal distribution (Figure 25). Overall, the probability of wild rice occurrence was slightly higher for polymictic lakes, and these lakes were less sensitive to chl-*a* and Secchi depth than dimictic lakes. The 95% extirpation values calculated were at chl-*a* concentrations of 21 and 37 µg/L for dimictic and polymictic lakes, respectively (Figure 26). It should be noted that this analysis relied on presence/absence data and did not account for the quality or size of wild rice populations in these lakes. However, the amount of buffer between the thresholds derived to protect macrophyte communities (i.e., chl-*a* = 9-18 µg/L) and the 95% extirpation concentration for wild rice indicated that the northern lake standards will be sufficient to protect wild rice from low transparency caused by elevated suspended algae.

Figure 25. Probability of occurrence of wild rice in a lake as function of A) chlorophyll-*a* and B) Secchi depth for polymictic and dimictic lakes. Datasets consist of northern and central region lakes. Fits are generalized additive model (GAM) logistic regression (bs = "tp", method = "REML", k = 10) and shaded areas are 90% confidence intervals.

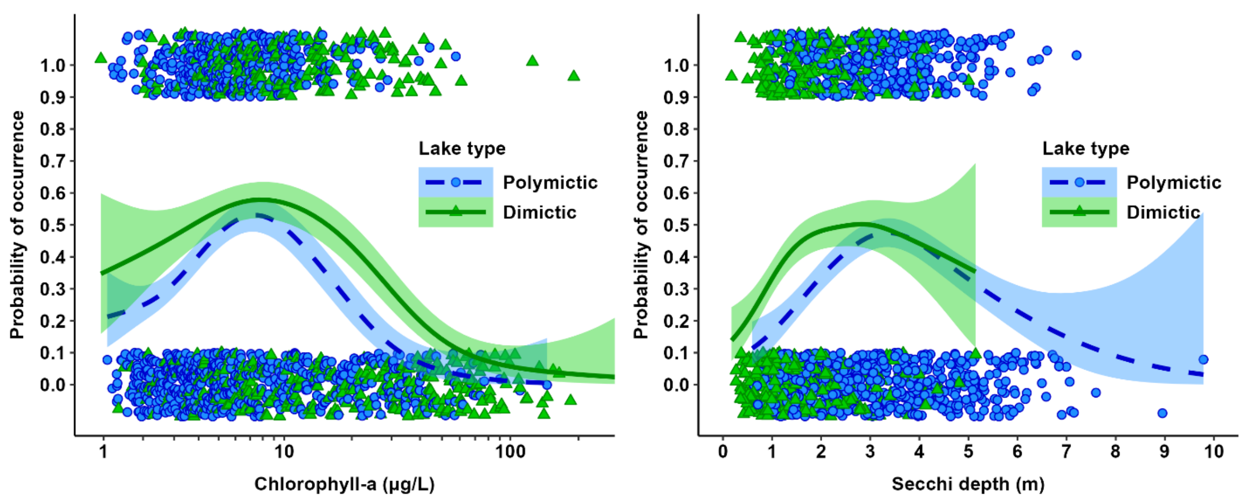
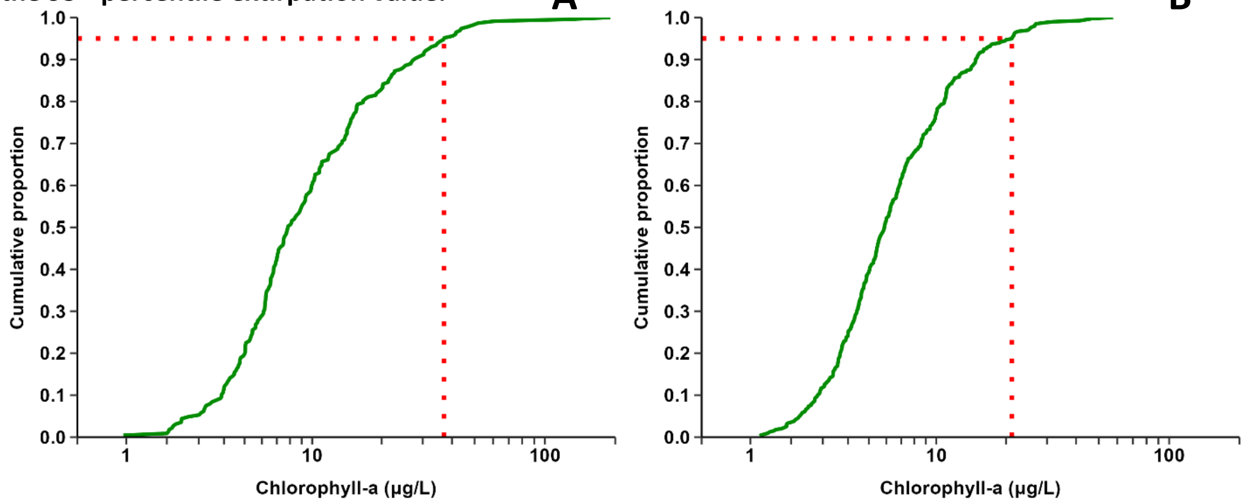
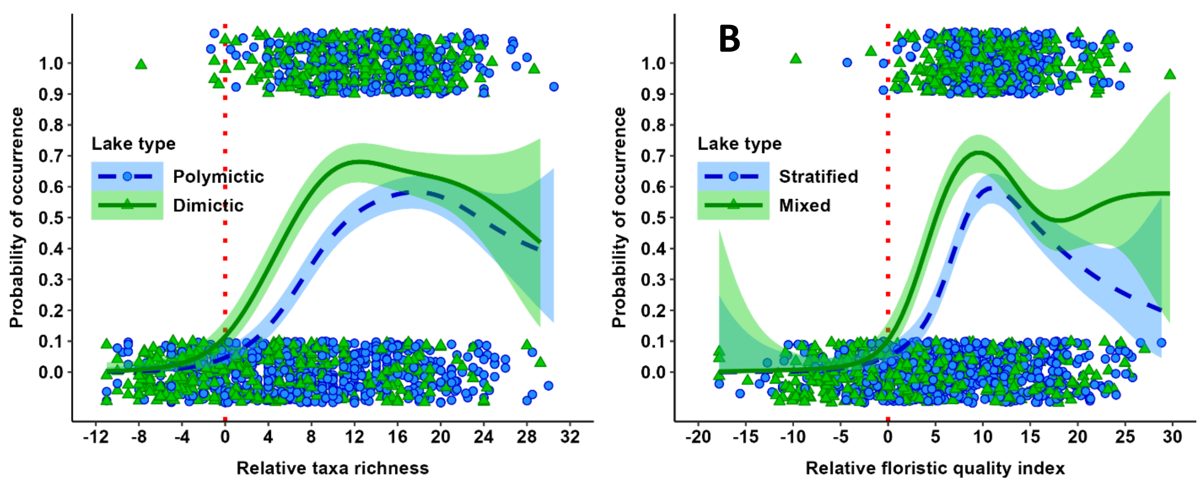


Figure 26. Cumulative distribution function of wild rice in A) dimictic and B) polymictic lakes along a gradient of chlorophyll-*a*. Datasets consist of northern and central region lakes. The red dashed line is the 95th percentile extirpation value.



In addition to an effect of transparency on wild rice, the presence of wild rice is often correlated with the presence of other macrophyte species (Myrbo et al. 2017). Therefore, a macrophyte index is not independent of wild rice presence and many lakes with good or healthy macrophyte communities would be expected to support wild rice assuming other ecological factors were suitable for this species. This is demonstrated by logistic regressions of wild rice presence as a function of relative macrophyte taxa richness and floristic quality index scores (Figure 27). Lakes with relative macrophyte scores of less than 0 do not meet the macrophyte thresholds provided in Radomski and Perleberg (2012), and there were very few lakes below this threshold which also supported wild rice. Importantly, as taxa richness or floristic quality index scores increased so did the probability of a lake supporting wild rice (Figure 27). Therefore, the use of eutrophication standards to broadly protect macrophytes will also collaterally protect wild rice. However, wild rice populations are also sensitive to other stressors such as pore-water sulfide and hydrological impacts which may need to be considered as part of protection goals for this sensitive and important species.

Figure 27. Probability of occurrence of wild rice in a lake as function of relative macrophyte A) taxa richness and B) floristic quality index for polymictic and dimictic lakes. Relative taxa richness and floristic quality index was calculated by subtracting index thresholds (Radomski and Perleberg 2012) from the raw values. Datasets consist of northern and central region lakes. Fits are generalized additive model logistic regressions (bs = “tp”, method = “REML”, k = 10) and shaded areas are 90% confidence intervals.



2. Fish

Background

Fish are important components of both polymictic and dimictic lakes and can be useful in determining attainment of aquatic life use goals. However, the aquatic life use goals, including the fish species supported or the benefits those fish communities provide, will differ between dimictic and polymictic lakes. For example, dimictic lakes thermally stratify in the summer which may create coldwater habitats for some fish species (e.g., Lake Trout, Cisco, and Lake Whitefish). Polymictic or shallow lakes generally support lower fish species diversity, and very shallow lakes that experience winterkill are less likely to support consistent populations of gamefish and generally have very low fish diversity. As a result, the standards and beneficial uses protected will be different between these lake types. For example, the application of EPA's recommended lake eutrophication standard for lake hypoxia is not appropriate to apply to polymictic lakes because hypoxia is less likely to occur and is unlikely to threaten fish species in polymictic lakes (Yuan and Jones 2020a, EPA 2021a). However, most polymictic lakes do support a fish community, including game fish in some lakes, which should be protected by water quality standards.

A fish index of biological integrity (FIBI) has been developed to measure the condition of fish communities in Minnesota lakes (Bacigalupi et al. 2021). Data used for calculating the lake FIBI consists of gill nets, trap nets, backpack electrofishers, and beach seines to comprehensively sample the cool/warm water fish community in a lake. The lake FIBI combines several functional group measures into a multimetric index to provide an overall measure of the condition of a lake's fish community. The FIBI tool also includes thresholds for non-attainment of aquatic life use goals (Bacigalupi et al. 2021). These goals were based on a biological condition gradient (BCG) model developed for Minnesota lakes and followed a similar process to that used for Minnesota streams (Bouchard et al. 2016). It should be noted that the FIBI is not applicable to all Minnesota lakes, including Canadian Shield lakes and lakes that experience winterkill, and the FIBI cannot be used with confidence for these lakes (Bacigalupi et al. 2021). As a result, there are some limitations in the applicability of the FIBI in the northern region since some lakes could not be included.

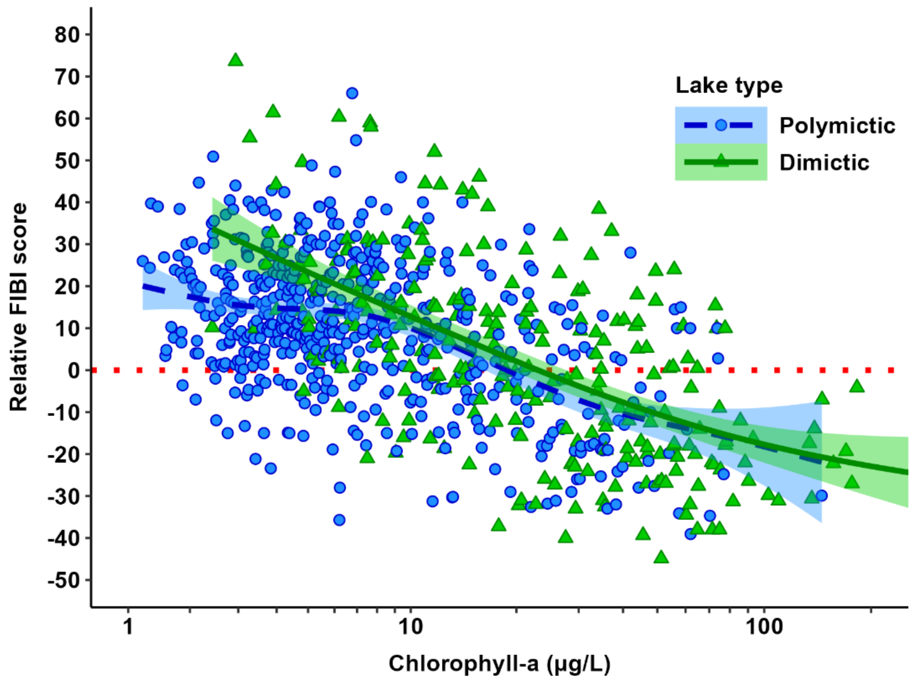
Analysis

FIBI scores were normalized in relation to the applicable thresholds (Bacigalupi et al. 2021) for each lake type because northern lakes include different FIBI lake types and raw FIBI scores are not necessarily equivalent. Average relative FIBI scores (2005-2019) were paired with average chl-*a* (1990-2020). The relationship between relative FIBI scores and chl-*a* was modeled using GAMs. The probability of attaining FIBI thresholds at different concentrations of chl-*a* were modeled using logistic regressions developed from GAMs with a logistic link function. Generalized additive models (GAM) were run in R version 4.0.3 (R Core Team 2020) using the "mgcv" package (Wood 2017). Even at low levels of nutrients (e.g., <5 µg/L), these models predicted some level of exceedance of fish goals. Due to the low concentration of chl-*a* in these lakes, exceedances are not likely related to enrichment, but rather due to other stressors (e.g., invasive species, lakeshore development, and suspended sediment) or chl-*a* or fish sampling error and variability. To correct for these effects, the initial exceedance rate of the model (i.e., the modeled exceedance rate at the minimum chl-*a* concentration in the dataset) was subtracted from the model. In the case of the fish logistic regression model for polymictic lakes, the lowest chl-*a* value in the dataset was 2 µg/L and at this value, the model estimated a 15.6% exceedance rate. This rate was subtracted from the model such that the initial exceedance of the model was 0%. From this model, a 10% exceedance rate was used to interpolate chl-*a* concentration. A 10% exceedance rate was selected to account for natural lake characteristics (e.g., naturally enriched lakes, lakes with atypical hydrology or morphology, lakes with high CDOM), lakes with more sensitive fish communities compared to the population, and the combined effects of the lake productivity and other stressors. The 10% exceedance rate was selected as a reasonable threshold for northern lakes where the effect of other stressors is expected to be low.

Results

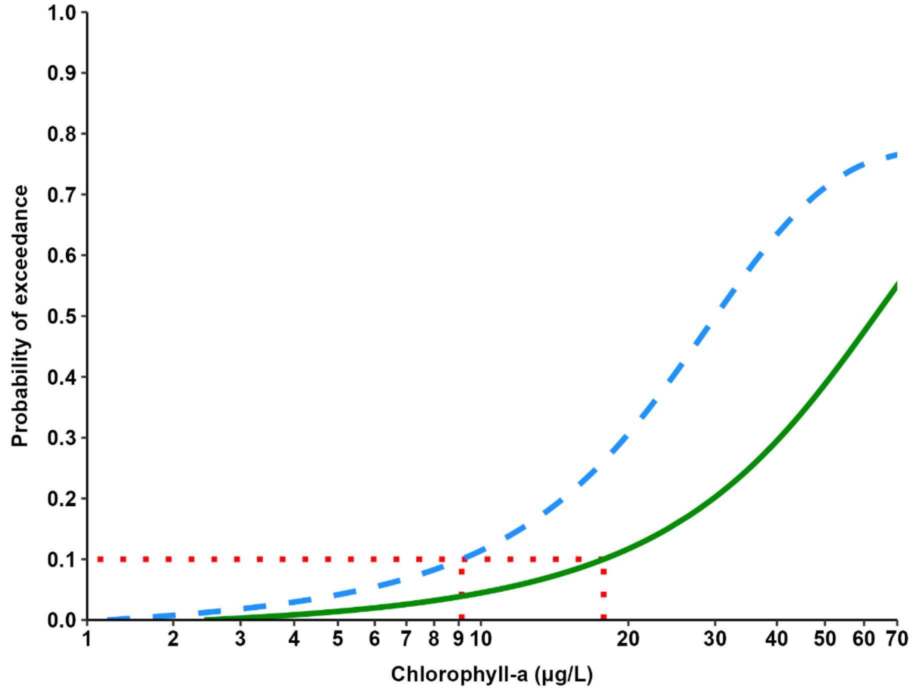
There was a negative relationship between chl-*a* and FIBI for both polymictic and dimictic lakes (Figure 28; polymictic lakes: adjusted R^2 = 0.35, p value <0.0001; dimictic lakes: adjusted R^2 = 0.24, p value <0.0001). The relationships between these two lake types differed with polymictic lakes indicating a lower sensitivity to increasing chl-*a*; although, there was overlap between the confidence intervals of these models (Figure 28). A 10% exceedance of the FIBI in polymictic and dimictic lakes was interpolated to correspond to chl-*a* concentrations of 18 and 9 µg/L, respectively (Figure 29). Based on this analysis; to protect aquatic life as measured by fish in northern polymictic lakes, a goal of 18 µg/L is recommended. The current chl-*a* standard for northern lakes is 9 µg/L which indicates this threshold is appropriate to protect cool/warm-water fish communities in northern dimictic lakes.

Figure 28. Relative fish index of biological integrity score as a function of average chlorophyll-*a* for Minnesota lakes. Points represent average lake values for chlorophyll-*a* (1990-2020) and fish index of biological integrity scores (2005-2020). Dataset consists of northern and central region lakes. Red, dashed line indicates the threshold for non-attainment of biological criteria. Fits are generalized additive models (GAM; bs = "tp", method = "REML", k = 10) and shaded areas are 90% confidence



intervals.

Figure 29. Probability of fish index of biological integrity scores exceeding applicable biocriteria as a function of chlorophyll-*a* in polymictic (green solid line) and dimictic (blue dotted line) lakes (northern and central region lakes). Red, dotted line shows interpolation of chlorophyll-*a* thresholds from a 10% probability of exceedance. Fits are generalized additive model (GAM) logistic regressions (bs = "tp",



method = "REML", k = 3).

ii. Recreational Use

Background

Protecting lake beneficial uses includes maintaining conditions that support the recreation beneficial uses for which a lake is useable. Depending on the lake, these specific recreation beneficial uses can consist of primary and secondary contact activities including swimming, boating, and fishing. Minnesota has had a lake survey program in place for over 30 years and extensive user survey data are available. This information is collected by volunteer lake monitors and water quality survey staff. Linking recreation use condition to eutrophication measures can be difficult because human interpretations of recreation suitability can be subjective (Smeltzer and Heiskary 1990), and data collected by volunteers can be problematic due to issues with coverage (EPA 2021b). However, lake volunteer monitors are a segment of the population who are active lakes users and most have a good familiarity with these resources. In addition, Minnesota's user survey database consists of over 360,000 individual surveys which makes it a useful tool for determining relationships between lake trophic status and recreation uses at the population level.

Minnesota's lake user survey asks respondents to score recreation suitability based on their opinion of current water conditions (Table 11). Depending on the habitat, ratings of 1 through 3 are generally considered to indicate attainment of primary contact recreation beneficial uses. A score of 4 indicates that primary contact recreation uses are likely not attained, but other secondary contact uses are still protected. Therefore, in lakes where swimming is not a use, a recreation survey scores of 4 may be acceptable because other recreation uses such as boating, wading, and fishing would be protected. In addition, other non-Class 2 uses may also be protected including waterfowl (Class 4) and aesthetics (Class 5). In nearly all cases, a rating of 5 would be considered to not be in attainment of recreation goals. However, it is also important to consider the duration and frequency of undesirable lake conditions. For example, ratings of 4 or 5 may be acceptable in a lake provided that condition is not frequent, nor does it persist for a long period of time.

Table 11. Lake user survey ratings for recreation suitability.

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

Analysis

Recreation suitability scores for North and Central region lakes were paired with chl-*a* data (1990-2020). Habitat type (i.e., polymictic versus dimictic lakes) could also impact user rating, so in the following analyses, these lake types were treated separately. The relationship between recreation suitability ratings and chl-*a* and depth category was analyzed with an ordinal regression using the polr function ("MASS" package; Venables and Ripley 2002) in R version 4.0.3 (R Core Team 2020). The interactions of effects from recreation suitability and lake depth category were analyzed using the "effects" package (Fox and Weisberg 2019). The relationship between average recreation suitability scores and chl-*a* was modeled using GAMs (R version 4.0.3 [R Core Team 2020]; "mgcv" package [Wood 2017]).

Recreation suitability scores were assessed as a function of long-term, summer average chl-*a*. For dimictic lakes, the probability of a recreation suitability score of 4 was used as an endpoint whereas polymictic lakes used the probability of a recreation suitability score of 5 (Table 11). The use of a higher recreation suitability score for polymictic lakes was because swimming is often not a beneficial use for these lakes (Heiskary and Wilson 2005). Algal blooms in Minnesota lakes are seasonally dynamic and as a result recreation suitability changes through the summer season. Typically, there is an increase in suspended algae and a decline in recreation suitability through the summer that often peaks in August; although, these patterns vary from summer to summer. Maximum concentrations or the probability of exceeding an undesirable level of chl-*a* are useful for determining recreation use thresholds (e.g., Bachmann et al. 2003). However, assessments of lake eutrophication standards are based on long-term summer averages, so recreation thresholds need to be expressed as summer averages. Therefore, recreation suitability score endpoints for each lake were expressed as the probability of a lake receiving a recreation suitability score of 4 or 5 (i.e., not meeting recreation goals) in 50% or more of the years it was surveyed. The probability of a lake exceeding this recreation goal was modeled as a function of chl-*a* with GAMs using a logistic link function in R version 4.0.3 (R Core Team 2020) with the "mgcv" package (Wood 2017). To correct for recreation suitability scores which exceeded thresholds due to other stressors (e.g., invasive species, lakeshore development, and suspended sediment) or chl-*a* or recreation survey sampling error and variability, the initial exceedance rate was subtracted from the model. From this model, a 10% exceedance rate was used to interpolate chl-*a* concentration. A 10% exceedance rate was selected to account for natural lake characteristics (e.g., naturally enriched lakes, lakes with atypical hydrology or morphology, lakes with high CDOM), and the combined effects of the lake productivity and other stressors. The 10% exceedance rate was selected as a reasonable threshold for northern lakes where the effect of other stressors is expected to be low.

Results

Ordinal regression analysis indicated that there were significant effects of lake depth category and chl-*a* on recreation suitability ratings (Figure 30). The probability of the five recreation suitability ratings was more strongly related to chl-*a* concentration (Figure 30). As expected, recreation suitability was poorer with increasing chl-*a*. The probably of recreation suitability scores of 4 and 5 (i.e., unsuitable for swimming) had a higher probability of occurring in lakes with higher chl-*a* concentrations (Figure 30). The opposite was true for recreation scores of 1 and 2 (i.e., excellent to good swimming conditions). The effect of lake type was smaller; although, there was some difference between lake types for recreation suitability scores of 1 (i.e., excellent swimming conditions) with these more likely in dimictic lakes (Figure 30). There was a negative relationship between chl-*a* and recreation suitability for both polymictic and dimictic lakes (Figure 31; polymictic lakes: adjusted $R^2 = 0.53$, p value <0.0001; dimictic lakes: adjusted $R^2 = 0.53$, p value <0.0001). There was only a small difference in the response between dimictic and polymictic lakes with confidence intervals overlapping between the two models (Figure 31). A 10% exceedance of recreation suitability goals for polymictic and dimictic lakes was interpolated to correspond to chl-*a* concentrations of 39 and 13 µg/L, respectively (Figure 32). Based on this analysis, to protect recreation beneficial uses (e.g., boating, waterfowl, and fishing) in northern polymictic lakes, a goal of 39 µg/L is recommended. This analysis indicates that the current chl-*a* standard of 9 µg/L for northern lakes is sufficient to protect recreation uses such as swimming in northern dimictic lakes.

Figure 30. Effects plots for ordinal logistic regression of recreation suitability (RS) as a function of chlorophyll-*a* and lake stratification category.

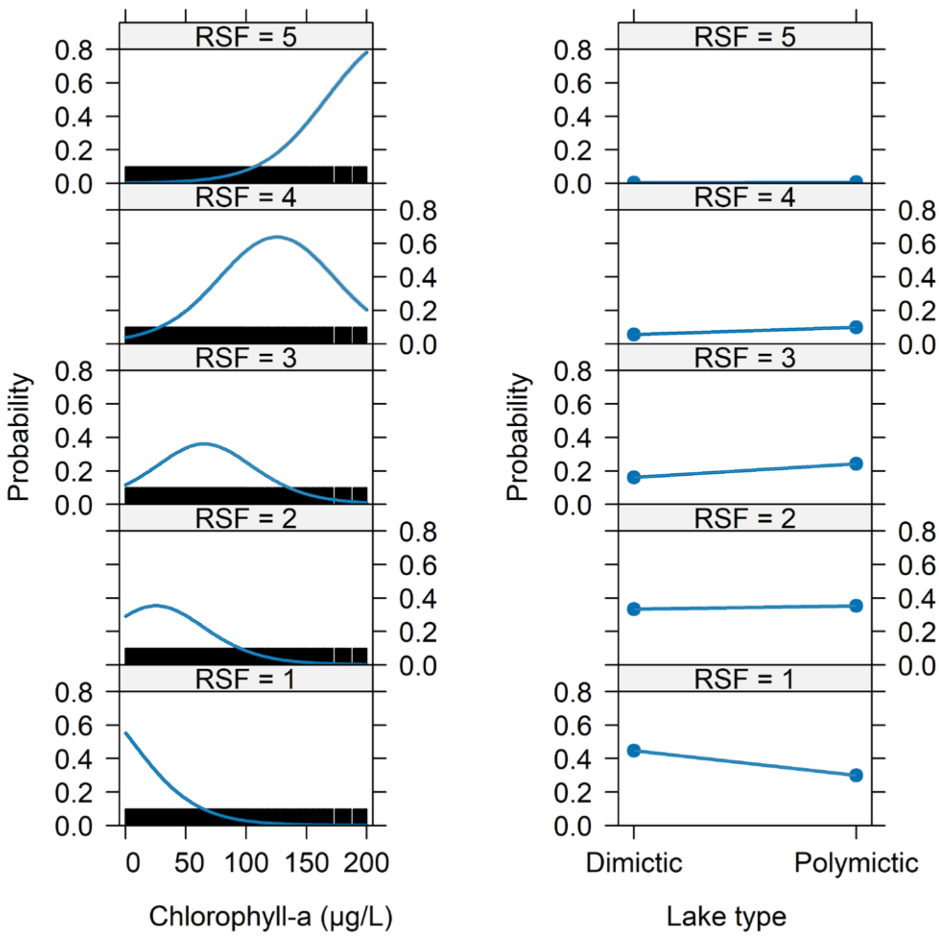


Figure 31. Recreational suitability scores as a function of chlorophyll-*a* for Minnesota lakes. Points represent average lake values for chlorophyll-*a* and recreation suitability (1990-2020). Dataset consists of North and Central region lakes. Fits are generalized additive models (GAM; bs = "tp", method = "REML", k = 10) and shaded areas are 90% confidence intervals. The red dashed line indicates the threshold between recreation suitable scores of 3 and 4 which approximates the protection levels of the current lake eutrophication standards.

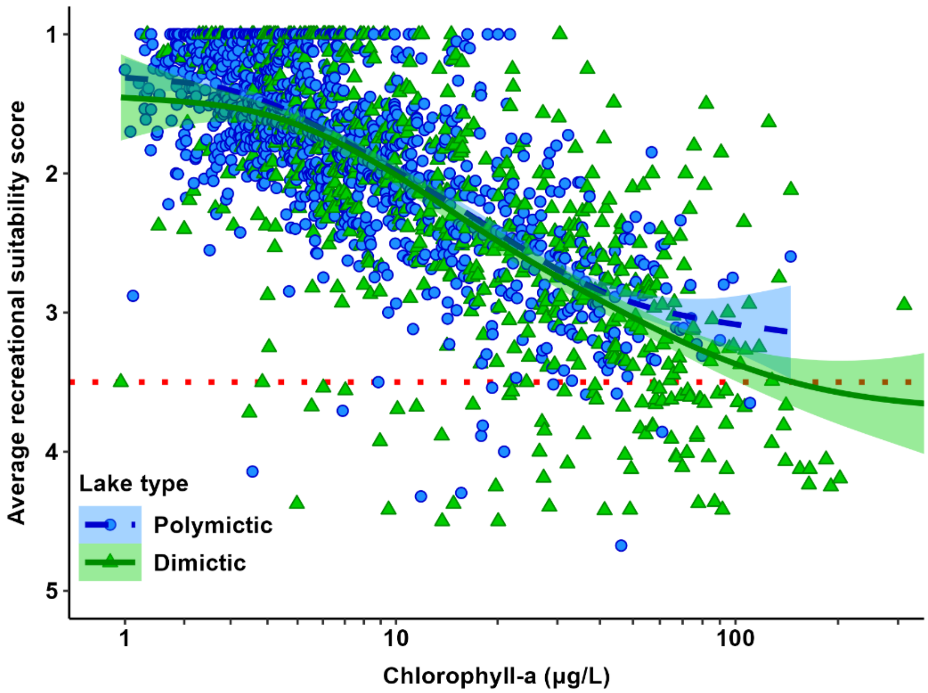
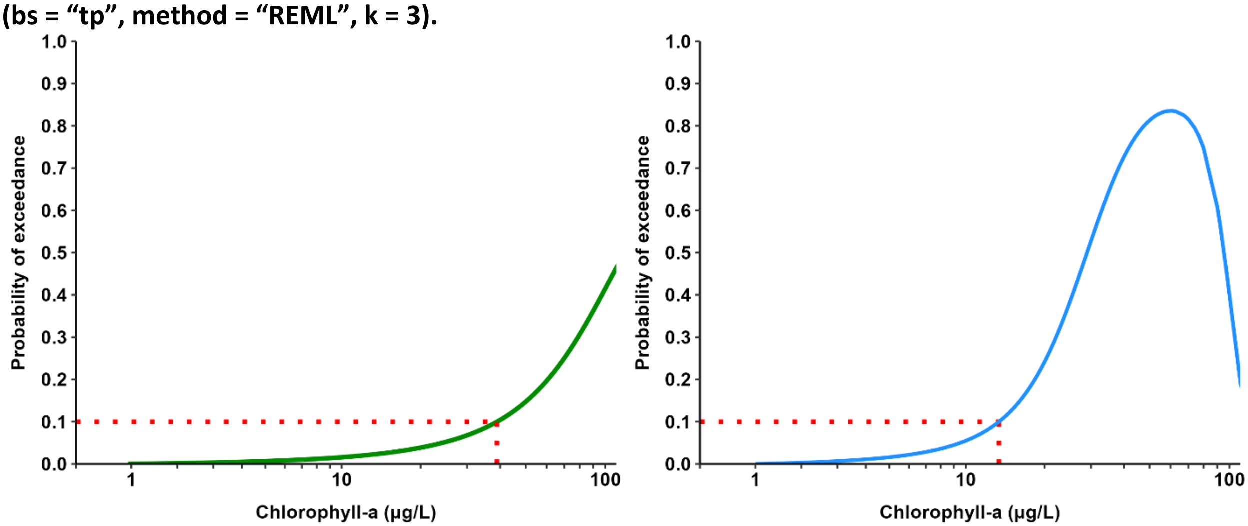


Figure 32. Probability of recreation suitability scores of A) 5 (polymictic lakes) or B) 4 (dimictic lakes) or as a function chlorophyll-*a* for polymictic (green solid line) and dimictic (dashed blue line) lakes (North and Central region lakes). Red, dotted line shows interpolation of chlorophyll-*a* thresholds from a 10% probability of exceedance. Fit are generalized additive model (GAM) logistic regressions (bs = "tp", method = "REML", k = 3).

A

B



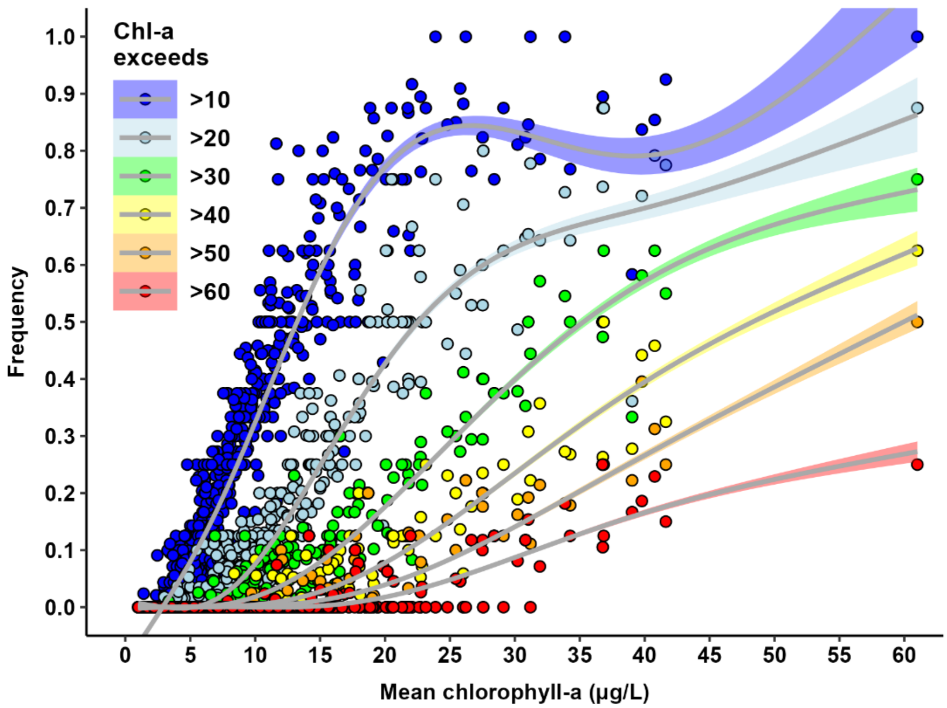
iii. Nuisance algal blooms

1. Frequency of nuisance algal blooms

A range of chl-*a* concentrations are used to identify severe or nuisance conditions in lakes. In South African impoundments, Walmsley (1984) used a chl-*a* threshold of 20-30 µg/L for nuisance and >30 µg/L for severe nuisance. The State of Florida uses a chl-*a* threshold of >40 µg/L to indicate an algal bloom (Havens 2003). Heiskary and Walker (1988) also reviewed these conditions in several studies and determined that undesirable algal conditions generally occur in the range of >30-40 µg/L of chl-*a*. Since recreation suitability is most impacted by extreme algal events in a lake, it is useful to model the percent of the summer season which will exceed undesirable chl-*a* concentrations for target summer-average chl-*a* concentrations. The percent of reference lakes (i.e., lakes with watershed disturbance <25%) with nuisance algal conditions (chl-*a* >30-40 µg/L) was determined for polymictic and dimictic lakes. This analysis was limited to lakes with watershed disturbance determinations and at least 2 years of chl-*a* data with four or more samples per year. The frequency of exceeding different chl-*a* concentrations during the summer season was modeled with GAMs in R version 4.0.3 (R Core Team 2020) using the "mgcv" package (Wood 2017).

There were 304 references northern polymictic lakes with at least 2 years and 4 samples per year of chl-*a* data. Of these lakes, 23% had at least one measurement of chl-*a* above 30 µg/L and 16% had at least one measurement of chl-*a* above 40 µg/L. The number of samples categorized as indicating nuisance conditions for these lakes was 2-75% and 1-63% of samples for 30 µg/L and 40 µg/L, respectively. There were 546 reference northern dimictic lakes with sufficient data. Of these lakes, 10% had at least one measurement of chl-*a* above 30 µg/L and 4% had at least one measurement of chl-*a* above 40 µg/L. The number of samples exceeding nuisance conditions for these lakes was 1-58% and 1-44% of samples for 30 µg/L and 40 µg/L, respectively. Dimictic lakes had lower rates of nuisance algae compared to polymictic lakes; however, in both lake types there are some lakes with minimal disturbance which have occurrences of nuisance algae for part of the summer. The chl-*a* thresholds derived from the most sensitive indicators for polymictic, and dimictic northern lakes were used to determine the percent of the summer that is predicted to have undesirable levels of suspended algae. Based on the most sensitive indicator for polymictic lakes (i.e., macrophytes), a mean chl-*a* of 18 µg/L was estimated to result in chl-*a* concentrations exceeding 30 or 40 µg/L for 13% and 5% of the summer, respectively (Figure 33). For dimictic lakes, the chl-*a* threshold necessary to protect the most sensitive endpoint was 9 µg/L for fish. Based on the chl-*a* frequency models, the percent of the summer where chl-*a* was predicted to be greater than 30 and 40 µg/L was 1-0%, respectively (Figure 33). This indicates that for polymictic and dimictic lakes, which meet chl-*a* thresholds protective of the most sensitive indicator, the frequency of nuisance algal blooms will be low and as a result recreation uses should also be protected.

Figure 33. Frequency of chlorophyll-*a* concentrations exceeding 10, 20, 30, 40, 50, and 60 µg/L as a function of mean summer chlorophyll-*a* (1990-2020) for northern lakes. Fits are generalized additive

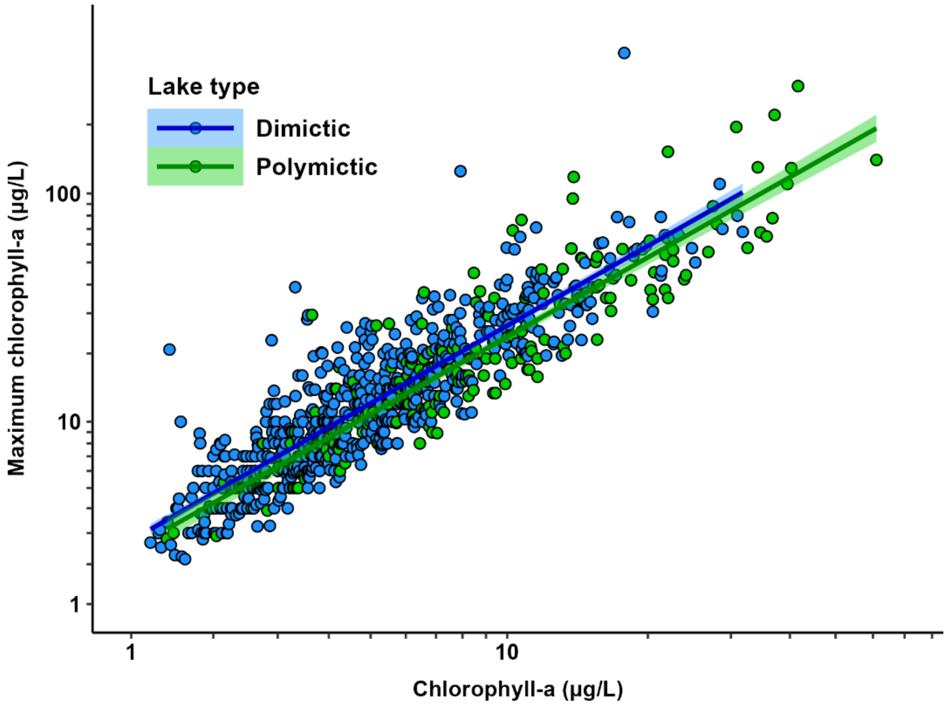


models (GAM; bs = "tp", k = 5) and shaded areas are 90% confidence intervals.

Maximum chlorophyll-*a*

The original work supporting the 2008 lake eutrophication rule (Heiskary and Wilson 2005) analyzed the relationship between summer average and maximum chl-*a*. Here we also modeled the relationship between average and maximum chl-*a* for dimictic and polymictic lakes using GAMs (R version 4.0.3 [R Core Team 2020]; "mgcv" package [Wood 2017]). As in the original work, there is a significant relationship between average and maximum chl-*a* in northern lakes (polymictic lakes: adjusted R^2 = 0.72, p-value >0.0001; dimictic lakes: adjusted R^2 = 0.44, p-value >0.0001; Figure 34). This relationship is similar between polymictic and dimictic lakes indicating that separate models are not necessarily needed for each lake type. These results can be used to determine how a chl-*a* standard, measured as a summer average, corresponds to extreme or nuisance algal bloom events to avoid such events. Based on the model of average and maximum chl-*a* concentrations (polymictic lake-only model; Figure 34), a summer average chl-*a* concentration of 18 µg/L is estimated to result in a maximum chl-*a* concentration of 51 µg/L. Although 51 µg/L is likely indicative of nuisance levels of algae, the analysis of algal bloom frequency demonstrated that such conditions will to be infrequent. The dimictic lake-only model (Figure 34) estimated that a summer average chl-*a* concentration of 9 µg/L will result in a maximum chl-*a* concentration of 30 µg/L. This indicates that most dimictic lakes meeting the draft chl-*a* threshold are not likely to have nuisance algal blooms.

Figure 34. Relationship between long-term summer average and maximum chlorophyll-*a* for dimictic and polymictic northern lakes. Fits are generalized additive model (GAM) logistic regressions (bs = "tp", k = 10) and shaded areas are 90% confidence intervals.



Development of eutrophication criteria

The chl-*a* thresholds determined from the analysis of individual beneficial use endpoints for polymictic, and dimictic lakes are compiled here, and the most sensitive endpoint is used for setting draft criteria. These chl-*a* endpoints were also used to interpolate protective levels of TP and Secchi depth from quantile regression models. The draft eutrophication criteria developed from this process provide protective goals for northern lakes which are consistent with the existing standards framework, but also provide more refined tools to manage diverse lake habitats in northern Minnesota.

i. Determination of total phosphorus and Secchi depth thresholds

Background

The current lake eutrophication standards framework includes nutrient (TP) and response (chl-*a*, and Secchi depth) criteria. In this framework, TP and at least one response parameter are required for assessments. An exceedance of TP and one or both response parameters results in an impairment determination. The TP criteria are important for the linkage of response criteria exceedance (i.e., chl-*a* and Secchi depth) to elevated nutrients and because much of lake eutrophication management (e.g., WQBELs, total maximum daily load (TMDL)s, protection plans) is based on TP targets. Secchi depth thresholds are developed for a different purpose than TP thresholds. Secchi depth is useful as a surrogate for chl-*a* when sufficient chl-*a* data are not available and CDOM levels are not too high. The relationship between chl-*a* and Secchi depth is affected by factors such as CDOM and suspended sediment. However, in northern lakes, suspended sediment is not an important factor; although, CDOM can be important in many of these lakes. (Brezonik et al. 2019, EPA 2021). If the effect of CDOM is not addressed, the use of Secchi depth alone can cause assessment error but, when accounted for, Secchi depth is a reasonable predictor of chl-*a* concentration (see Figure 37). As such, Secchi depth can be effective for making decisions regarding the attainment status of lake eutrophication standards. For these reasons and to be consistent with the existing framework, TP concentrations and Secchi depths were modeled using the chl-*a* thresholds established for the protection of beneficial use endpoints in northern lakes.

Analysis

To determine protective TP and Secchi depth values, 90th and 10th percentile quantile regressions were fit to statewide TP - chl-*a* and chl-*a* - Secchi depth relationships using the "rq" function in the "quantreg" package (Koenker 2019) and "bs" function in the in "splines" package in the program R version 4.0.3 (R Core Team 2020). The effect of CDOM on relationships between TP, chl-*a*, and Secchi depth was assessed by modelling these relationships using datasets with different levels of CDOM censored. These models were run using three datasets: 1) not censored for high CDOM, 2) lakes with color >73 PCU or a_{440} >4 m^{-1} censored, and 3) lakes with color >25 PCU or a_{440} >1.4 m^{-1} censored. The CDOM thresholds used for censoring lakes were based on CDOM models in Brezonik et al. (2019) and were discussed in the section ***Secchi depth and colored dissolved organic matter*** (p. 24). If these models indicated an important effect of CDOM, then censored datasets could be used to model these relationships. The 90th and 10th percentiles were selected to address the effects of other sources of TP (e.g., dissolved phosphorus or sediment-associated phosphorus) or factors that could limit algal growth or lake transparency (e.g., shading, other nutrients). This approach was conceptually similar to EPA's recommended lake eutrophication criteria (see Yuan and Jones 2019, 2020b, EPA 2021a); although, EPA (2021a) used a different statistical approach to model chl-*a* based on an estimate of the phosphorus bound to phytoplankton. Criteria for TP and Secchi depth were interpolated from the quantile regression models using chl-*a* thresholds identified from the beneficial use endpoints.

Modeled TP and Secchi depth thresholds for the most sensitive endpoints (Table 12) were assessed in terms of their ability to protect or predict chl-*a* levels in lakes. ROCs and error rate plots were used to determine if TP and Secchi depth are good predictors of chl-*a* and if selected thresholds will minimize errors. This type of analysis has also been used to develop nutrient criteria using an approach to select thresholds based on error rate minimization (Smeltzer et al. 2016). ROCs were modeled in R version 4.0.3 (R Core Team 2020) using the "pROC" package (Robin et al. 2011). False positive error rates (FPR) were calculated as:

(2) FRP = NFP/(NTN + NFP)

False negative error rates (FNR) were calculated as:

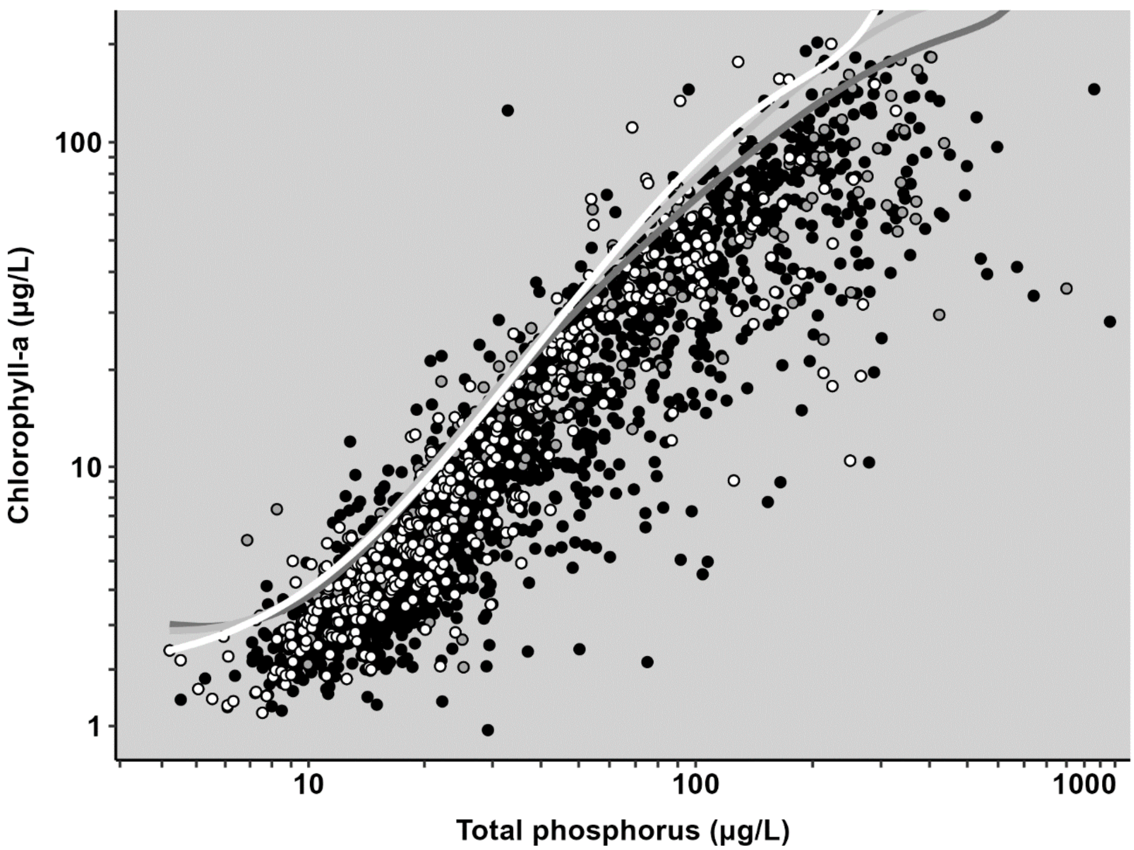
(3) FNR = NFN/(NFN + NTP)

where NFP = number of false positives, NTN = number of true negatives, NFN=number of false negatives, and NTP = number of true positives. Receiver operating characteristic curves were plotted, and AUC values were used to evaluate how well chl-*a* predicts TP and Secchi depth in lakes. False positive rates and false negative rates were plotted to determine error rates associated with modeled TP and Secchi depth criteria. For this analysis, false positives occurred when TP or Secchi depth for a lake exceeded thresholds, but the applicable chl-*a* threshold was below the threshold. False negatives were coded when TP or Secchi depth met thresholds and chl-*a* did not. For this analysis, the TP dataset was not censored, but the Secchi depth dataset censored lakes with color >73 PCU or a_{440} >4 m^{-1}.

Results

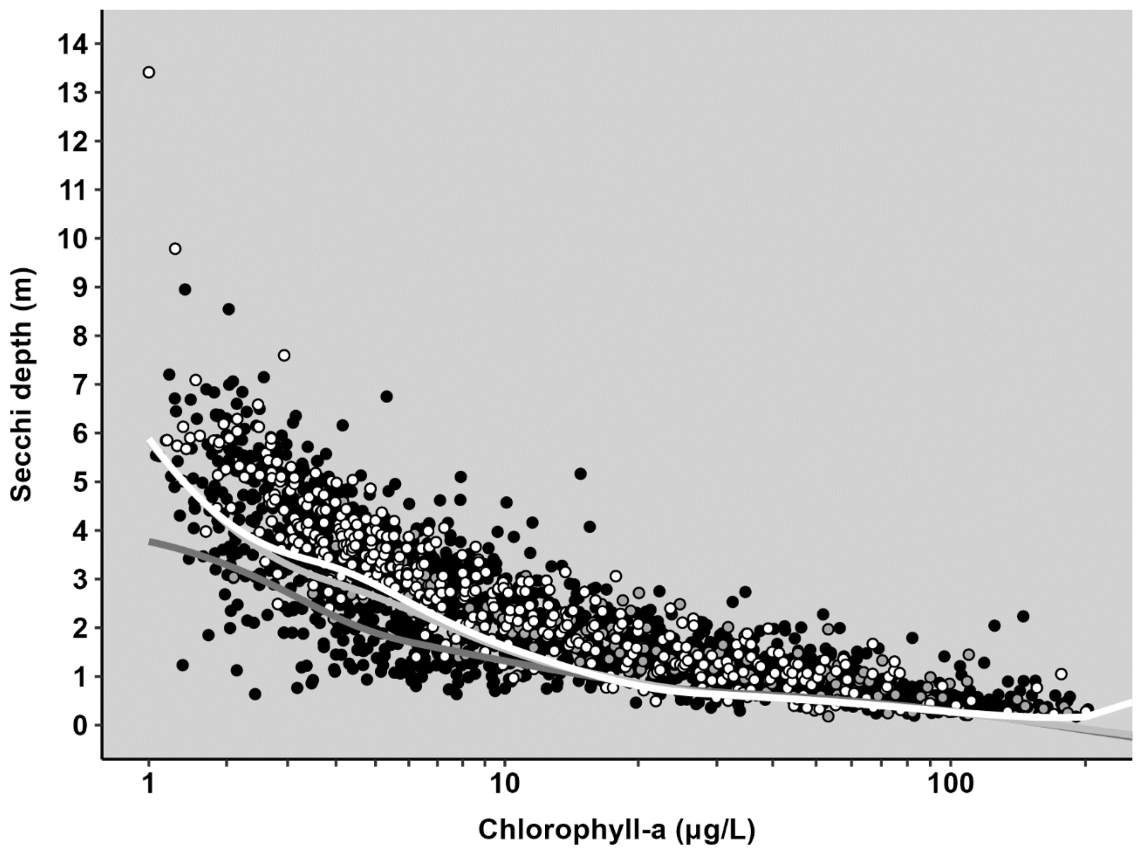
Regardless of dataset (i.e., lakes censored for high CDOM or not), 90th percentile quantile models for estimating chl-*a* from TP concentrations were similar (Figure 35). There were some differences including a gap between the uncensored model and both of the censored models at chl-*a* concentrations above approximately 40 µg/L. In general, this supports previous analyses (e.g., Figure 10) which demonstrated that CDOM does not have an important effect on the TP – chl-*a* relationship. This has also been observed in other studies where populations of shallow lakes were determined to have higher chl-*a* concentrations than deep lakes despite higher CDOM levels (e.g., Nürnberg and Shaw, 1998, Webster et al. 2008). This appears to be the case at chl-*a* concentrations below 40 µg/L, but at higher levels there may be an effect of CDOM on chl-*a*. These analyses indicated that for northern lakes, the highly censored dataset (i.e., color <73 PCU or a_{440} <4 m^{-1}) is not needed to model chl-*a* concentrations from TP. However, due to the possible effect of CDOM at higher chl-*a* concentrations, the lakes with moderate censoring (i.e., color <25 PCU or a_{440} <1.4 m^{-1}) are selected to model TP and Secchi depth. Using this dataset compared to the more highly censored dataset results in a larger dataset and provides more certainty for the model.

Figure 35. Quantile regression models of chlorophyll-*a* as a function of total phosphorus using uncensored and censored datasets. Datasets include statewide data and points are summer average values for lakes (dark grey line and black circles = uncensored data; grey line and grey circles = color <73 PCU or a_{440} <4 m^{-1}; white line and white circles = color <25 PCU or a_{440} < 1.4 m^{-1}).



There was a difference in the 90th percentile quantile models for estimating Secchi depth from chl-*a* concentrations between the datasets (i.e., lakes censored for high CDOM or not; Figure 36). There was very little difference between the models using censored data, but the model using uncensored data estimated lower Secchi depth at comparable chl-*a* concentrations. This is expected since lakes with higher CDOM will have lower Secchi depths. However, above chl-*a* concentrations of approximately 10 µg/L, all three quantile regression models estimated similar Secchi depths (Figure 36). In general, these analyses indicated that for northern lakes, censored data are needed to model Secchi depth from chl-*a*. Either dataset of CDOM censored lakes resulted in similar estimates, but using the larger dataset provides more certainty. Therefore, we used the dataset with color >73 PCU or a_{440} >4 m^{-1} censored to model Secchi depth criteria.

Figure 36. Quantile regression models of Secchi depth as a function of chlorophyll-*a* using uncensored and censored datasets. Datasets include statewide data and points are summer average values for lakes (dark grey line and black circles = uncensored data; grey line and grey circles = color <73 PCU or a_{440} <4 m^{-1}; white line and white circles = color <25 PCU or a_{440} < 1.4 m^{-1}).



Quantile regression models were used to determine TP concentrations and Secchi depths consistent with attainment of chl-*a* targets based on the northern lake endpoint thresholds. The most sensitive indicator for dimictic lakes was the fish IBI which resulted in a protective chl-*a* threshold of 9 µg/L. Fitting quantile regressions to TP – chl-*a* data predicted that 90% of lakes with a TP concentration of 20 µg/L will meet the 9 µg/L chl-*a* threshold (Figure 37A, Table 12). Using the chl-*a* – Secchi depth model (Figure 37B, Table 12), 90% of lakes at the 9 µg/L chl-*a* threshold will have a Secchi depth of at least 1.8 m. The most sensitive indicators for polymictic lakes were macrophytes and fish which had a protective chl-*a* threshold of 18 µg/L. For TP, a concentrations of 32 µg/L was estimated to result in attainment of the chl-*a* 18 µg/L threshold (Figure 37A, Table 12). A Secchi depth of less than 0.9 m was predicted to indicate a high likelihood that the chl-*a* target for macrophytes and fish will be exceeded (Figure 37B, Table 12). These TP and Secchi depth criteria combined with chl-*a* thresholds are used to establish protective standards for northern lakes which are consistent with the existing lake eutrophication standards framework.

Figure 37. Statewide quantile regression models for determining (A) total phosphorus levels needed to meet chlorophyll-*a* thresholds and (B) Secchi depths associated with an exceedance of chlorophyll-*a* criteria. Points are summer average values for lakes (1990-2020). Both datasets were censored for lakes with color >73 PCU or a_{440} >4 m^{-1}. Fits are quantile regression smoothing splines for 90^{th} and 10^{th} quantiles (total phosphorus model: degree = 4, df = 6; Secchi depth model: degree = 3, df = 7).

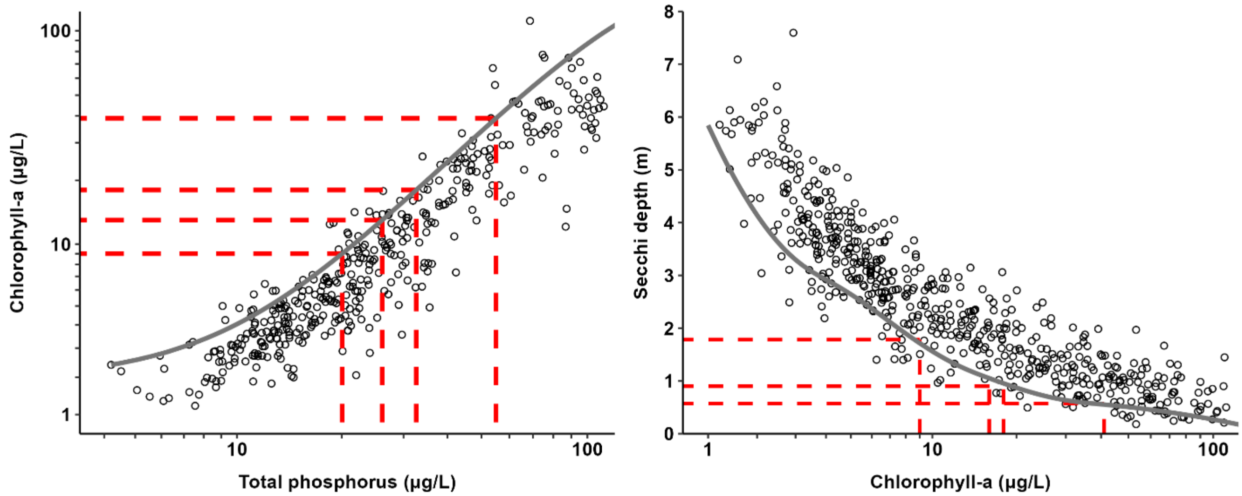


Table 12. Chlorophyll-*a* targets for the protection of aquatic life and recreation beneficial uses in northern polymictic and dimictic lakes with associated total phosphorus concentrations and Secchi depths. Total phosphorus and Secchi depth values were determined from chlorophyll-*a* targets using quantile regression models (see Figure 37). The blue highlight indicates the most sensitive endpoints for each lake type.

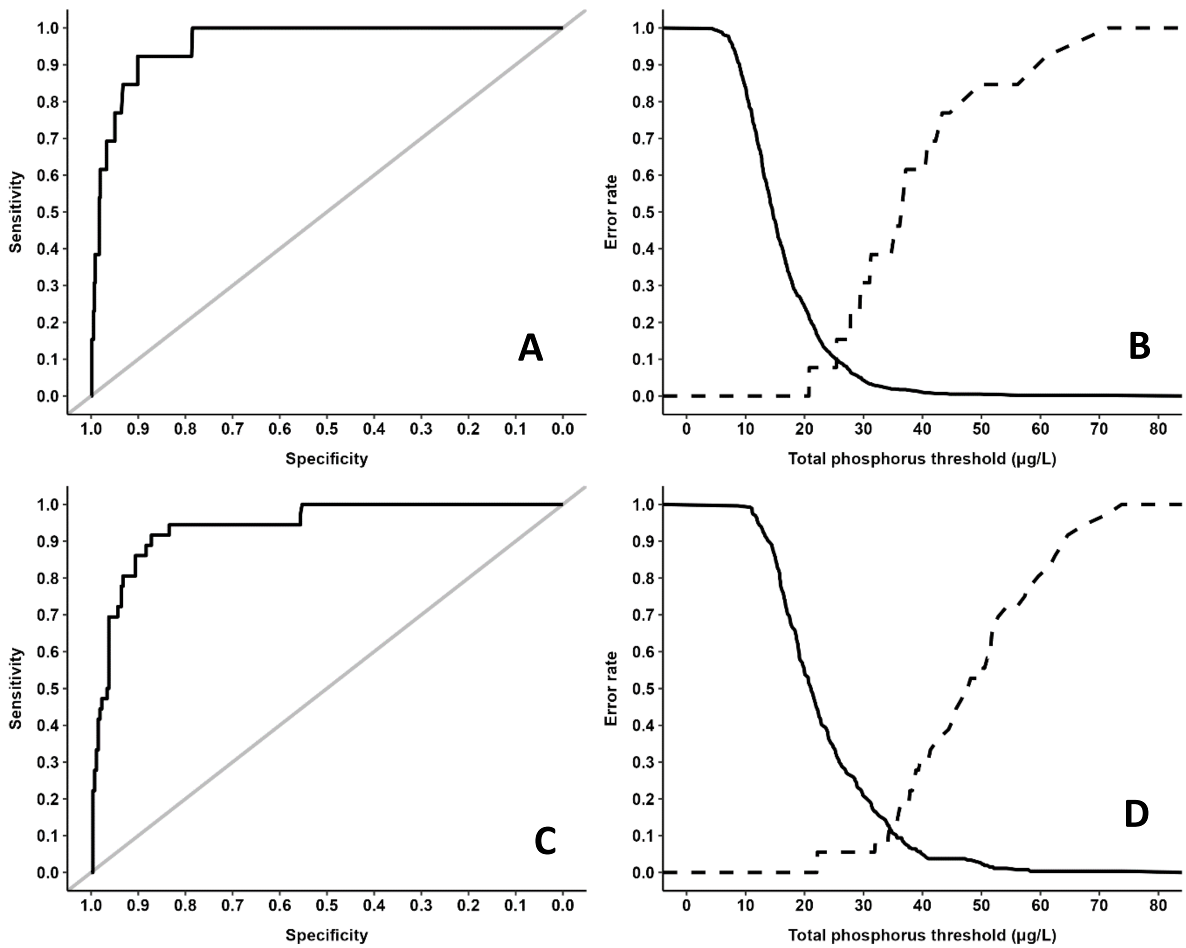
Beneficial use	Beneficial use endpoint	Total phosphorus (µg/L)	Chlorophyll-*a* target (µg/L)	Secchi depth (m)
		Northern polymictic (shallow) lakes		
Aquatic life	Aquatic macrophytes	32	18	0.9
Aquatic life	Fish: Index of biological integrity	32	18	0.9
Recreation	Recreational suitability	55	39	0.6
		Northern dimictic lakes		
Aquatic life	Aquatic macrophytes	26	13	1.2
Aquatic life	Fish: Index of biological integrity	20	9	1.8
Recreation	Recreational suitability	26	13	1.2

The draft TP criteria for dimictic northern lakes are more stringent than current values because the current standards were based on a least squares regression model (Heiskary and Wilson 2005). The existing least-squares model reasonably predicts chl-*a* based on TP for Minnesota lakes; however, there is a higher likelihood of false negatives compared to the updated model (Figure 37A), particularly for lakes near thresholds. Such lakes often fall into an “inconclusive” assessment category because TP is not exceeded, but chl-*a* is high. Since chl-*a* is a more direct measure of productivity than TP and some lakes are more productive at lower nutrient levels, it is reasonable to establish criteria that will acknowledge these lake attributes. The updated models result in more stringent criteria, but this will reduce false negative errors. These more protective standards will also be more likely to maintain the clear-water state in northern lakes (see Figure 21) and avoid lakes tipping into a stable turbid-water state. However, there are lakes where chl-*a* is lower than predicted by this model which can be attributed to non-algal bound phosphorus or other limiting factors. For these lakes, it may be appropriate to develop site-specific TP criteria, to determine the amount of bioavailable phosphorus or to identify other factors limiting algal growth (e.g., shading, TN:TP ratio).

The proposed Secchi depth criteria for northern lakes are less stringent compared to current values. As with TP this is due to differences between the current least squares chl-*a*-Secchi depth model (Heiskary and Wilson 2005) and the quantile regression model used in this study (Figure 37B). Assessments should ideally be based on chl-*a* and TP when these data are available because chl-*a* provides a direct measure of lake productivity. Secchi depth is also a good predictor of lake productivity, but it may be affected by other factors that can introduce error into assessments. As a result, the 10^{th} percentile was used to minimize these errors while still retaining the information Secchi depth can provide to an assessment, even when chl-*a* data are not available. In addition, assessments relying on Secchi depth will need to account for CDOM and suspended sediment as part of assessments to reduce false positive errors (see *Appendix A*).

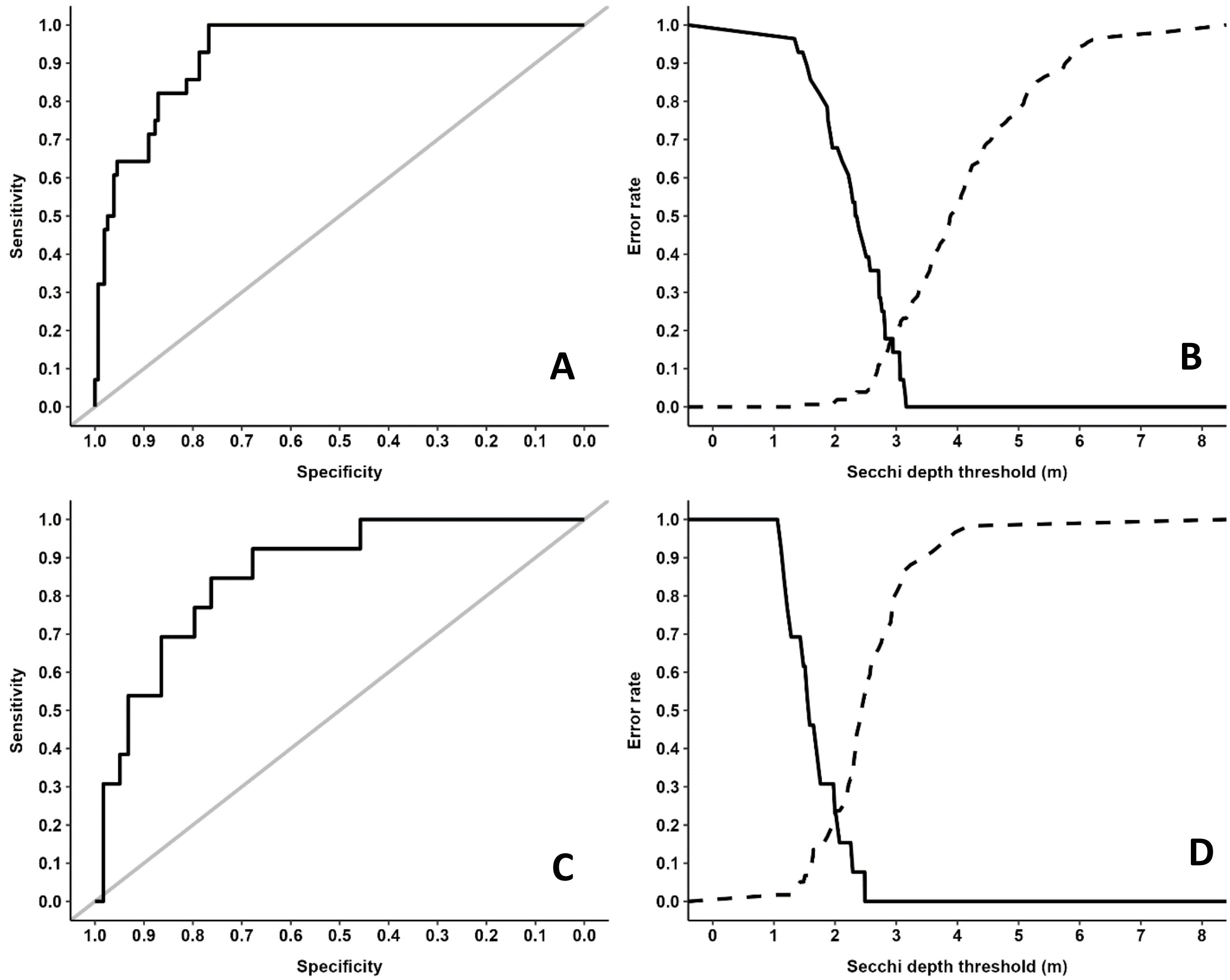
Characterizing decision errors when assessments are based on chl-*a* and TP is useful to understand and minimize potential assessment errors. The AUC value for dimictic lakes was 0.9585 indicating that TP was an outstanding predictor of chl-*a* assessment outcomes (Figure 38A). The lowest combined error rate for dimictic lakes occurred at a TP concentration of approximately 25 µg/L (FNR = 8%; FPR = 10%; Figure 38B). This is higher than the draft TP criterion (20 µg/L) for dimictic lakes which has higher error rates (FNR = 0%; FPR = 24%). For polymictic northern lakes, TP was also an outstanding predictor of chl-*a* attainment with a high AUC value (AUC = 0.9392; Figure 38C). The lowest combined error rate occurred at a TP concentration of approximately 34 µg/L (FNR = 8%; FPR = 13%; Figure 38D). The modeled TP concentration needed to meet a chl-*a* concentration of 18 µg/L was 32 µg/L and is near the threshold concentration with the lowest error rates (see Table 12, Figure 37); although, error rates are higher (FNR = 8%; FPR = 17%). The use of TP criteria of 20 or 32 µg/L for dimictic and polymictic lakes decreases false negative rates at a cost to false positive error rates which is reasonable to provide protections for these lakes. In other words, setting more stringent TP criteria will increase the likelihood that these nutrient goals will achieve desired chl-*a* targets and therefore also protect beneficial uses.

Figure 38. Analysis of error rates for predicting chlorophyll-*a* based on total phosphorusfor (A,B) dimictic (9 µg/L) and (C,D) polymictic (18 µg/L) northern lakes: (A,C) receiver operating characteristic (ROC) curves (AUC: dimictic lakes = 0.9585; polymictic lakes = 0.9392) and (B,D) error rate plots. Data not censored for lakes with high CDOM. For ROC curves (A,C), specificity refers to the true negativity rate and sensitivity refers to the true positivity rate. For error rate plots (B,D), solid lines are false positive rates, and dashed lines are false negative rates.



As with TP, it is useful to characterize decision errors when assessments are based on only Secchi depth and TP to understand and minimize potential errors. Secchi depth was an outstanding predictor of chl-*a* assessments for northern, dimictic lakes and had an AUC value of 0.9290 (Figure 39A). The lowest combined error was at a Secchi depth threshold of 3.2 m (FNR = 0%; FPR = 23%; Figure 39B). For polymictic, northern lakes, Secchi depth was an excellent predictor of chl-*a* attainment (AUC = 0.8592; Figure 39C) with the lowest combined error rates at a Secchi depth of approximately 2.1 m (FNR = 15%; FPR = 24%; Figure 39D). The interpolated Secchi depth thresholds (see Table 12, Figure 37) resulted in a low FPR (FPR <2%) for both lake types. Since Secchi depth is a surrogate for chl-*a*, it is reasonable to use thresholds which minimize false positives. The disadvantage of this approach is a relatively high FNR; however, it may be appropriate to use Secchi depth data to identify lakes where follow up monitoring is needed. For example, lakes which exceed the TP standard and have a Secchi depth between 0.9 and 2.1 m for polymictic lakes or 1.8 and 3.2 m for dimictic lakes may be candidates for additional monitoring to determine if the chl-*a* standard is exceeded. However, as discussed, a review of CDOM conditions within lakes should be part of lake eutrophication assessments utilizing Secchi depth when it is the sole response parameter.

Figure 39. Analysis of error rates for predicting chlorophyll-*a* based on Secchi depthfor (A,B) dimictic (9 µg/L) and (C,D) polymictic (18 µg/L) northern lakes: (A,C) receiver operating characteristic (ROC) curves (AUC: dimictic lakes = 0.9290; polymictic lakes = 0.8592) and (B,D) error rate plots. Data are censored for lakes with high CDOM (color > 73 PCU and a_{440} < 4 m^{-1}). For ROC curves (A,C), specificity refers to the true negativity rate and sensitivity refers to the true positivity rate. For error rate plots (B,D), solid lines are false positive rates, and dashed lines are false negative rates.



ii. Development of protective lake eutrophication standards for Northern Forest ecoregion lakes

1. Review of lake eutrophication standards in other CWA programs

As part of this study, we provide a short review of lake eutrophication standards from northern states and tribes. The jurisdiction of these states and tribes include part of the Northern Forests Level I ecoregion for comparison with the draft standards for northern lakes in Minnesota. Several of the tribes and states included in this ecoregion have not adopted numeric lake eutrophication criteria. However, most states and tribes without numeric criteria do implement protections through narrative WQS which may be used to limit phosphorus discharges from point sources or describe conditions under which it may be necessary to limit nutrient loading. In some cases, narrative criteria are implemented as a numeric translator (e.g., Grand Portage Band of the Minnesota Chippewa Tribe). In addition, site-specific standards have been adopted for some lakes even if statewide standards have not been implemented.

States and tribes with adopted numeric lake eutrophication criteria include the Fond du Lac Band of the Minnesota Chippewa Tribe and the states Minnesota, Vermont, and Wisconsin. These states and tribes have adopted different numeric lake eutrophication criteria which consist of different eutrophication parameters for assessment, but all have adopted TP criteria. Vermont and Minnesota have developed and adopted criteria for TP, chl-*a*, and Secchi depth. Vermont has also adopted nitrogen criteria which apply to lakes. The Fond du Lac Band of the Minnesota Chippewa Tribe has adopted numeric lake eutrophication criteria for TP, TN, chl-*a*, and Secchi depth, and Wisconsin has adopted only TP criteria. The Fond du Lac Band of the Minnesota Chippewa Tribe and Vermont are the only Northern Forest ecoregion programs to have adopted nitrogen criteria as part of lake eutrophication standards. Lake classifications for implementation of lake eutrophication standards also differ between these jurisdictions. Most lake classifications in this region are based on lake type and region, but some classifications are refined enough for the application of lake-specific criteria (e.g., Fond du Lac). An overview of individual state and tribal lake eutrophication standards are provided below.

Fond du Lac Band of the Minnesota Chippewa Tribe

The Fond du Lac Band has adopted TP, TN, chl-*a*, and Secchi depth criteria for 9 primary fisheries lakes with 3 of these lakes divided into separate basins. Eutrophication criteria were determined by calculating the 90th percentile of samples for each lake. Phytoplankton community data were used to confirm that these numeric criteria were protective of aquatic life uses. As developed, these standards are site-specific standards which was feasible due the relatively small number of lakes to which these standards apply and the large dataset available to develop these standards. In addition, these lakes have the advantage of occurring in a relatively undisturbed landscape which provides a baseline for trophic conditions. Total phosphorus criteria for these lakes range from 15-47 µg/L and chl-*a* criteria range from 3-44 µg/L. The range of eutrophication criteria for these lakes reflects natural differences among Fond du Lac's lakes and indicate that some lakes in the northern region have naturally higher levels of nutrients. This supports these efforts to refine Minnesota's northern lake eutrophication standards to address the natural diversity of lake trophic state in these systems. Secchi depth standards adopted by the Fond du Lac Band are 0.3-2.5 m with most lakes with criteria at or below 1 m. Many of these lakes have relatively low Secchi depth criteria due to CDOM affecting lake transparency demonstrating the need to consider CDOM as part of a Secchi depth standard.

Grand Portage Band of the Minnesota Chippewa Tribe

Although the Grand Portage Band of the Minnesota Chippewa Tribe has not yet adopted numeric lake eutrophication criteria, it is useful to review their standards because the Grand Portage Band's lakes are in the NLF ecoregion and are comparable to lakes analyzed as part of this study. The tools used by the Fond du Lac Band for implementing lake eutrophication standards are also used by the Grand Portage Band. Although the Grand Portage Band of the Minnesota Chippewa Tribe has not formally adopted numeric lake eutrophication standards, narrative criteria are implemented as a numeric translator for 15 lakes. Total phosphorus criteria for these lakes range from 29-97 µg/L and chl-*a* criteria range from 4-67 µg/L. Twelve of these lakes have a $Z_{max} < 4.57$ m (15 ft) indicating that most of these lakes would be considered shallow based on Minnesota's shallow lake definition. The ranges of lake-specific eutrophication criteria demonstrate the diversity of protective conditions for lakes in this region. The Grand Portage Band's lake eutrophication criteria also do not include Secchi depth due to high CDOM in many of these lakes. As with the Fond du Lac Band's standards, the tools used by the Grand Portage Band demonstrate a need for refined lake eutrophication criteria and a need to consider CDOM as part of Secchi depth criteria in northern Minnesota.

Wisconsin

The state of Wisconsin has adopted lake eutrophication standards for lakes greater than 4.05 ha (10 acres) which includes TP standards for five lake types (Wisconsin Department of Natural Resources 2019; Table 13). These categories are based on thermal regime, stratification, and lake hydrology. The thermal regime is addressed by assigning a separate TP criterion to two-story lakes (i.e., lakes supporting coldwater and warm water fish communities). Different criteria are applied to stratified and non-stratified lakes with more stringent standards applied to stratified lakes. Wisconsin uses an equation similar to the lake geometry ratio to determine if a lake is stratified. Lakes without surface water inflow or outflow are considered seepage lakes and those with surface water inflow or outflow are categorized as drainage lakes. It is difficult to make comparisons between Minnesota and Wisconsin lake eutrophication standards because the lake typologies differ. The largest differences being the inclusion of regionalization in Minnesota and a hydrological component in Wisconsin's typology. The polymictic northern lakes are most comparable to Wisconsin's unstratified lake types which are assigned a TP standard of 40 µg/L. The dimictic northern lakes are most comparable to Wisconsin's stratified lake types which have TP standards ranging from 20-30 µg/L. It is also important to note that Wisconsin's TP standards are higher for polymictic lakes compared to stratified lakes which is consistent with the results in this study.

Table 13. Wisconsin's lake eutrophication standards.

Lake type	Total phosphorus standard (µg/L)
Two-story fishery lakes	15
Lakes that are both drainage and stratified lakes	30
Lakes that are drainage lakes, but are not stratified lakes	40
Lakes that are both seepage and stratified lakes	20
Lakes that are seepage lakes, but are not stratified lakes	40

Although not codified in rule and not used for impairment decisions, Wisconsin may consider trophic status index (TSI) for the integrated report under CWA Section 305(b) (Wisconsin Department of Natural Resources 2019). Lakes meeting the TSI threshold may be placed in Category 2 in the integrated report. The Wisconsin DNR uses the transition between a fair and poor condition as the threshold for lakes meeting aquatic life goals. For shallow lakes, this threshold is a TSI of 71 which corresponds to a TP concentration of 100 µg/L. This threshold was selected because in shallow lakes this TP concentration is associated with a switch from a clear water state (i.e., aquatic plant dominated) to a turbid state (i.e., algal dominated) (Jeppesen et al. 1990).

Vermont

Vermont has adopted lake nutrient criteria for TP, chl-*a*, and Secchi depth (Table 14). Vermont's lake classification divides lakes into 3 groups[11] which include excellent (Class A(1)), very good (Classes B(1) and A(2)), and good (Class B(2)) aesthetic conditions. The good classification (Class B(2)) likely best aligns with Minnesota's minimum eutrophication goals for lakes. For good (Class B(2)) lakes, Vermont assigns the following criteria for TP, chl-*a*, and Secchi depth: 18 µg/L, 7 µg/L, and 2.6 m. This is similar to the draft northern dimictic lake standard but is more stringent than the draft thresholds for northern polymictic lakes in Minnesota. As in Minnesota, Vermont uses a nutrient coupled with response parameters for assessment where nonattainment of the standard requires both exceedance of the nutrient and a response parameter. Vermont has also adopted nitrogen standards for lakes which applies to all lakes, ponds, and reservoirs, regardless of classification. The nitrogen standard requires that levels of nitrate do not to exceed 5.0 mg/L.

[11] These groups are similar to Minnesota's tiered aquatic life use (TALU) framework which establishes goals for some waters which are more protective than the CWA minimum goal.

Table 14. Summary of Vermont's criteria for selected lake eutrophication parameters.

Parameter	Class A(1)	Classes A(2) and B(1)	Class B(2)
Total phosphorus (µg/L)	12	17	18
Chlorophyll-*a* (µg/L)	2.6	3.8	7.0
Secchi depth (m)	5	3.2	2.6

2. Comparison to EPA's lake eutrophication criteria recommendations

The EPA has developed numeric lake eutrophication standards which states and tribes may use to develop and adopt their own standards (EPA 2021a). In some regards, the structure and framework for EPA's recommended lake standards (EPA 2021a) differs from Minnesota's lake eutrophication standards and the draft revisions to the northern lakes' standards. For example, some of EPA's lake eutrophication tools result in lake-specific targets or candidate criteria; whereas Minnesota's framework is based on a lake typology which applies standards based on thermal regime, region, and lake stratification. However, EPA's lake standards framework is flexible enough to allow states and tribes to modify EPA's recommended standards such that they may more closely match existing frameworks and typologies. Even without these modifications, some comparisons can still be made between Minnesota's draft northern lake standards and EPA's recommended lake eutrophication standards. Broadly, EPA's recommended lake eutrophication standards consist of several beneficial use endpoints including zooplankton (aquatic life), deepwater hypoxia (aquatic life), and microcystin (recreation and drinking water). In general, this approach is similar to Minnesota's which also considers multiple endpoints (macrophytes, fish, and recreation suitability) and beneficial use types (aquatic life and recreation). In both frameworks, thresholds or criteria are determined for chl-*a* based on empirical responses of beneficial use indicators to increasing eutrophication. Nutrient levels (TP and total nitrogen: EPA (2021a); TP: Minnesota) are then estimated to determine ambient nutrient conditions necessary to achieve chl-*a* thresholds. Overall, Minnesota is following the general approach described by EPA (2021) to revise its existing lake eutrophication standards, and therefore Minnesota's draft standards for northern lakes are broadly consistent with EPA's recommendations.

The zooplankton tool developed by the EPA divides lakes into three depth categories: <3.8 m, 3.8-8 m, and >8 m (EPA 2021a). Minnesota's definition of shallow lake includes a depth threshold of 15 ft or 4.57 m which falls within EPA's middle zooplankton depth category. There is also a distinct separation of stratification types based on depth between dimictic and polymictic lakes in Minnesota with the 10^{th} percentile of Z_{max} for dimictic lakes and the 90^{th} percentile of polymictic lakes both at 6.1 m (Figure 40). Again, this falls within the middle zooplankton depth category which indicates that there is not an exact correspondence between the EPA zooplankton lake categories and Minnesota's lake typology. To determine chl-*a* targets for zooplankton, EPA's model requires the selection of a slope threshold and certainty limit. If we select a 90% certainty limit, different lake depth categories and slope thresholds produce different chl-*a* thresholds with lower targets for shallower lake groups (Table 15). The draft chl-*a* thresholds for northern lakes are comparable to the results from EPA's zooplankton models where, depending on the slope threshold selected, the polymictic lake threshold (18 µg/L) is most similar to the 3.8-8 m lake category although the draft Minnesota criteria are consistently higher. The dimictic lake threshold (9 µg/L) is most similar to the 3.8-8 m and >8 m lake categories.

Figure 40. Box plots of maximum depth (Z_{max}) for polymictic and dimictic lakes (box plot description: open circles = outliers; upper and lower hinges = 25th and 75th percentiles; whiskers extend from the hinge to the largest/smallest value no further than 1.5 * interquartile range from upper and lower hinges).

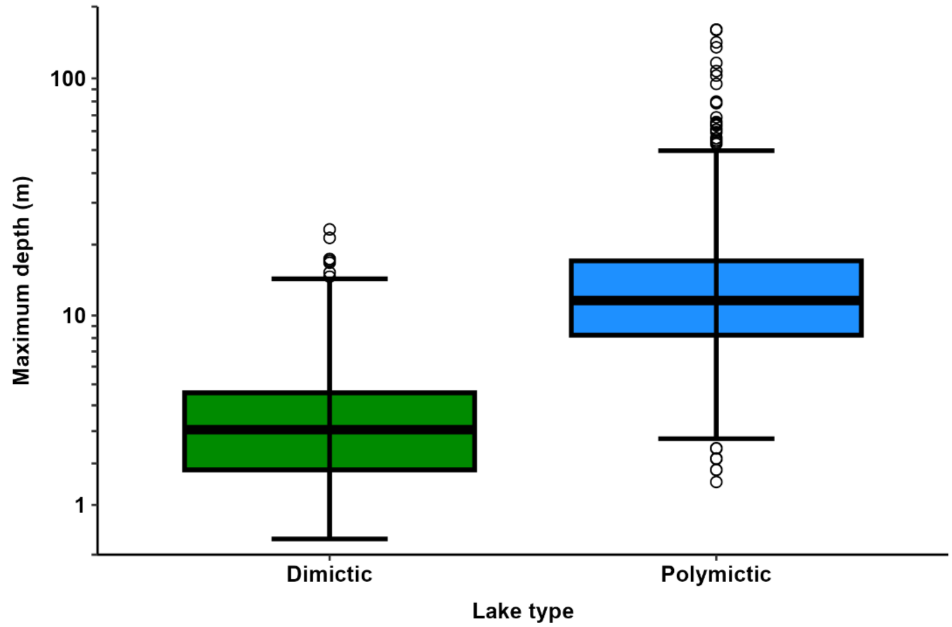


Table 15. Chlorophyll-*a* targets based on different lake depth categories and slope thresholds using EPA's draft lake nutrient criteria model (90% certainty level; https://chl-zooplankton-prod.app.cloud.gov/; accessed January 27, 2022).

Slope threshold	<3.8 m	3.8-8 m	>8 m
0	72	13	11
0.05	56	9	9
0.1	43	6	7

The EPA's deepwater hypoxia criteria are difficult to compare to Minnesota's draft northern lake eutrophication criteria because EPA's models require longitude/latitude, elevation, DOC, and the depth below the thermocline to determine lake-specific chl-*a* targets (Yuan and Jones 2020a, EPA 2021a). While all of these parameters can be determined for some Minnesota lakes, the resulting criteria are lake specific. In general, the deepwater hypoxia criteria would also not be applicable to polymictic lakes so it is not appropriate to use this tool to develop criteria for these lakes. Minnesota has developed draft coldwater lake standards (MPCA 2024) which include an oxythermal habitat measure (i.e., T_{DO3}) and is similar in some regards to EPA's deepwater hypoxia criteria. The adoption of an oxythermal habitat criteria in Minnesota will protect the most sensitive lakes and their aquatic life in a manner comparable to EPA's recommended standards. Although oxythermal habitat criteria would not apply to dimictic lakes which support only cool/warm water fish communities in Minnesota, the draft lake eutrophication standards for northern lakes were derived directly from fish and macrophyte community goals and will be sufficient to protect these assemblages.

The EPA's recommended lake eutrophication criteria also include a tool for determining chl-*a* concentrations based on microcystin targets (EPA 2021a). The MPCA has not determined if the EPA approach or a state-specific approach is needed to protect beneficial uses (domestic consumption and recreation) from harmful algal blooms. However, adoption of standards for cyanotoxins with sufficient toxicity data is under consideration in other rulemaking efforts focused on human health.

In the EPA's approach, chl- *a* targets determined from the zooplankton, deepwater hypoxia, and microcystin tools are used to determine TP and total nitrogen criteria. The EPA's recommended nitrogen criteria were not reviewed here in detail because development of nitrogen criteria was not considered for Minnesota lakes due to data limitations and the narrow scope of this project. The models for determining candidate TP criteria include targeted chl-*a* concentration, ecoregion, and lake maximum depth as parameters (EPA 2021a). As depth increases, candidate TP criteria decline until maximum depth reaches 4-5 m (Table 16). This depth approximates Minnesota's threshold between dimictic and polymictic lakes (Figure 40). The draft chl-*a* criteria for dimictic and polymictic lakes were used to calculate TP criteria from the draft EPA lake eutrophication models for lakes in the NLF using a certainty level of 0.9[12] (Table 16). Although not completely comparable due to differences in lake classifications, the results of the quantile regressions using Minnesota data (dimictic lakes 20 µg/L, polymictic lakes 32 µg/L) are similar to those determined by EPA's draft lake eutrophication models (Table 16). The draft TP criteria for northern Minnesota lakes fall within the outputs EPA models. This demonstrates that the draft Minnesota criteria are generally comparable to the EPA model criteria. In addition, the criteria developed for Minnesota's North region lakes is based on Minnesota-only data and these resulting criteria should be applicable to these lakes.

Table 16. Total phosphorus candidate criteria based on different chlorophyll-*a* targets using EPA's draft lake nutrient criteria models (Northern Lakes and Forest ecoregion, 90% certainty level; https://tp-tn-chl-prod.app.cloud.gov/; accessed November 8, 2024).

Chl-*a* target (µg/L)	**9**														
Depth (m)	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
TP criterion (µg/L)	28	24	21	21	20	20	19	19	19	19	19	18	18	18	18
Chl-*a* target (µg/L)	**18**														
Depth (m)	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
TP criterion (µg/L)	37	34	31	30	29	29	29	28	28	28	28	27	27	27	27

3. Summary of thresholds development of protective standards

The development of lake eutrophication standards for northern polymictic (shallow) and dimictic lakes was based on several analyses including a review of background conditions and a determination of thresholds to protect aquatic life (macrophytes and fish) and recreation (recreation suitability). As demonstrated by reference condition and paleolimnology analyses, background conditions are different between dimictic and polymictic northern lakes. This difference can be attributed to natural lake characteristics including lake morphology, watershed ratio, and watershed land cover types. Differences in chl-*a* thresholds for aquatic life and recreation between dimictic and polymictic northern lakes demonstrate that different lake eutrophication standards should be adopted for these lake types as they currently are for the Central and South regions in Minnesota.

Eutrophication thresholds for three aquatic life and recreation endpoints were determined for polymictic and dimictic northern lakes (Table 17). Aquatic life thresholds were developed for aquatic macrophytes and fish, and aquatic recreation thresholds were developed for recreation suitability. The recreation suitability goals differ between polymictic and dimictic lakes with a focus on primary contract (e.g., swimming) in dimictic lakes and secondary contact (e.g., boating, fishing) in polymictic lakes. The focus of these analyses was to identify protective chl-*a* concentrations for each of these endpoints. The thresholds selected were based on logistic regression models to determine the chl-*a* concentration at which a 10% non-attainment rate is estimated to occur. From the chl-*a* thresholds, protective TP criteria were determined using a 90th percentile quantile regression model (Figure 37A) to ensure a high

[12] The credible interval is analogous to a confidence interval in frequentist statistics and describes model uncertainty. A credible interval of 0.9 predict that the target chl-*a* will be attained 90% of the time.

likelihood that TP targets will achieve beneficial use goals. Secchi depth criteria were determined using a 10^{th} percentile quantile regression model (Figure 37B) to allow the use of this measure as a surrogate for chl-*a* while minimizing false positive errors.

Background conditions for northern lakes were characterized using reference condition and paleolimnology datasets (Table 17). Analyses using these datasets identified the trophic condition for these lake populations under minimally disturbed conditions (i.e., reference condition) and background/natural conditions (i.e., paleolimnology). This evidence does not necessarily establish thresholds for the protection of beneficial uses, but rather provides a background for thresholds determined from stressor-response models. For example, if the draft criteria were more protective than background conditions, this would potentially raise questions regarding the accuracy of the background conditions or determined thresholds. As a result, the endpoints developed to protect specific beneficial uses can be put in the context of natural background and modified if necessary. In general, the most-sensitive endpoint values for eutrophication parameters were comparable to reference or background conditions (Table 17). Reference or background conditions for chl-*a* and Secchi depth had eutrophication parameters that were indicative of lower trophic state compared to the most-sensitive endpoint values (Table 17). In contrast, the TP from the paleolimnological analyses were higher than the TP concentrations modeled from the most sensitive chl-*a* endpoint. The TP values from reference sites matched those modeled from the chl-*a* endpoints.

The selection of draft lake eutrophication thresholds for northern lakes was based on the most sensitive beneficial use endpoint. This approach is consistent with EPA guidelines (EPA 2021a). For northern polymictic lakes, the most sensitive endpoints were macrophytes and fish and a protective chl-*a* threshold of 18 µg/L was identified (Table 17). The modeled response parameter thresholds associated with this chl-*a* concentration were a TP of 32 µg/L and a Secchi depth of 0.9 m. The chl-*a* endpoint was higher (39 µg/L) for recreation in polymictic lakes due in part to the use of a maximum recreation survey score of 4 (i.e., protective of boating, fishing, waterfowl production) to determine protective chl-*a* concentrations. This threshold was selected because many polymictic lakes are not used for swimming. In addition, user perception surveys of recreational suitability are negatively impacted by conditions other than suspended algae. For example, a user may not want to swim in a lake due to extensive macrophyte beds or fine, mucky sediments. These conditions are typical and natural for many polymictic or shallow lakes which can create false positive errors for a standard based on suspended algae (i.e., chl-*a*). In cases where swimming is a use for a polymictic lake, a decision can be made to change the classification of the lake to a dimictic lake to ensure that the protective standards are applied (see Appendix A: Determination of nutrient region and lake type). A site-specific standard may also be appropriate for such lakes. In addition, the draft chl-*a* standard for polymictic lakes is near the recreation threshold that is protective of swimming in dimictic lakes which indicates that even without a change to the lake type, 18 µg/L will protect most polymictic lakes used for swimming. The most sensitive endpoint for dimictic lakes was the warm/cool water fish community which resulted in a protective chl-*a* concentration of 9 µg/L (Table 17). The response parameter thresholds modeled from this chl-*a* concentration were a TP of 20 µg/L and a Secchi depth of 1.8 m. Implementation of these standards will be consistent with existing guidelines (MPCA 2021). This chl-*a* threshold indicates that the current standard is appropriate to protect dimictic lakes. However, the modeled TP and Secchi Depth values differ from the current standard due to the updated models used to determine these values. The new TP criterion will assign protective nutrient goals to lakes to ensure that protective chl-*a* targets are attained. The updated Secchi depth criterion will reduce false positive errors. In addition, consideration of CDOM levels should be given to both polymictic and dimictic lakes to avoid false positive errors when assessing Secchi depth. Due to the prevalence of high CDOM limiting Secchi depth in many northern lakes, Secchi depth should not be used as a primary assessment parameter in lakes where there is high CDOM or the level of CDOM is unknown. Implementation of the draft standards are described in Appendix A. The draft standards for northern lakes are consistent with Minnesota's existing lake eutrophication framework and represent a refinement to these standards. The draft revisions to lake eutrophication standards will require amendments to [Minn. R. 7050.0222](), subparts 3 and 4.

Table 17. Summary water quality condition and beneficial use endpoints for northern lakes. Light grey highlighted fields provide background or minimally disturbed conditions and do not necessarily provide protective goals. Orange highlighted rows indicate the most sensitive beneficial use indicators assessed for each lake type and blue highlighted rows indicate the draft lake eutrophication standards.

Beneficial use	Analysis/beneficial use endpoint	Total phosphorus (µg/L)	Chlorophyll-*a* (µg/L)	Secchi depth (m)
Aquatic life and **Recreation**	Current northern lakes standard **(dimictic and polymictic lakes)**	**30**	**9**	**2.0**
	Polymictic (shallow) northern lakes			
	Reference lakes*	32	11	1.5
	Paleolimnology‡	33	-	-
Aquatic life	Aquatic macrophytes	32	18	0.9
Aquatic life	Fish: Index of biological integrity	32	18	0.9
Recreation	Recreational suitability	55	39	0.6
Aquatic life and recreation	**Draft northern polymictic lake criteria**	**32**	**18**	**0.9**
	Dimictic northern lakes			
	Reference lakes*	20	6	2.7
	Paleolimnology‡	27	-	-
Aquatic life	Aquatic macrophytes	26	13	1.3
Aquatic life	Fish: Index of biological integrity	20	9	1.8
Recreation	Recreational suitability	26	13	1.3
Aquatic life and recreation	**Draft northern dimictic lake criteria**	**20**	**9**	**1.8**

*** The reference analysis is based on datasets of lakes with <25% watershed disturbance. Total phosphorus and chl-*a* statistics were determined from uncensored datasets and Secchi depth statistics were determined from datasets censored for lakes with high CDOM (measured as color >73 PCU or a_{440} >4 m^{-1}). Total phosphorus and chl-*a* values are based on the 75th percentile for these lakes and Secchi depth is based on the 25th percentile.**

‡ Based on the 75th percentile of lakes.

Conclusions

The draft eutrophication standards for polymictic (shallow) and dimictic northern lakes are based on protections for the most sensitive beneficial use endpoint and protect both aquatic life and recreation beneficial uses. These standards and the supporting analyses demonstrate that most lakes in the NLF and NMW ecoregions have good water quality compared to other Minnesota ecoregions. As in other regions, the draft standards for polymictic lakes are less stringent compared to dimictic lakes, but both fall within the ranges of mesotrophic and eutrophic lakes (Carlson and Simpson 1996, EPA 2017). The existing lake eutrophication standard for chl-*a* applied to all northern lakes (9 µg/L) is sufficient to protect dimictic lakes, but new analyses and revised models indicate TP and Secchi depth criteria should be adjusted for dimictic lakes and that all three eutrophication parameters should be revised for polymictic lakes. Overall, polymictic lakes have naturally higher trophic condition and support different beneficial uses which should be reflected in applicable standards. This research also demonstrated that assessment of Secchi depth criteria requires additional scrutiny due to the prevalence of lakes with high CDOM in the northern ecoregion. In lakes with high CDOM or where CDOM levels are unknown, Secchi depth should not be used as a primary endpoint for assessment and assessments should rely on TP and chl-*a*. Specific CDOM thresholds are provided to determine when Secchi depth should not be used in lake eutrophication assessments.

The methods used to determine northern lake eutrophication criteria were specifically selected to identify chl-*a* thresholds which, if attained, will result in a high probability of beneficial use protection. In addition, TP and Secchi depth criteria were selected to reduce assessment errors and to ensure a high probability that lakes will attain chl-*a* targets. This approach to setting protective standards is important because establishing baseline standards that are under protective can result in the loss of beneficial uses in lakes. Managing aquatic resources is more efficient and cost effective when protection strategies are implemented as opposed to reactive measures to restore degraded waterbodies (Radomski and Carlson 2018).

The methods for assigning standards and the long-tailed distributions of TP and chl-*a* in reference lake datasets (Figure 19) indicate that a portion of northern lakes have naturally high TP and chl-*a*. There are several reasons why a subset of these dimictic lakes have a higher tropic state. The current lake type determination may be incorrect, and some dimictic lakes should be classified as polymictic following a more detailed review. In addition, some lakes have unique characteristics that may require development of a site-specific standard. For example, lakes may have flowage characteristics (e.g., a large watershed lake ratio) or may be heavily influenced by adjacent wetlands. Other standards development approaches have also recognized a need for lake-specific eutrophication criteria (e.g., Soranno et al. 2008, EPA 2021a). The implementation of site-specific standards in many cases will be contingent on a determination that a lake's beneficial uses are currently supported and given the overall low disturbance in this region, this is likely to be the case for many lakes. Analyses provided in this document and in EPA's recommended eutrophication criteria (EPA 2021a) provide tools for setting site-specific standards for atypical lakes. In general, the draft standards for polymictic and dimictic northern lakes provide a protective, regional baseline which may be modified for specific lakes as needed.

Minnesota's diverse aquatic resources require refined standards to ensure the application of appropriate and protective goals for the maintenance of beneficial uses. Natural differences in water quality and the beneficial uses supported in northern dimictic and polymictic lakes demonstrate that different standards are appropriate to manage these habitats for aquatic life and recreation. It is important that standards account for natural differences between aquatic resources such that they accurately reflect goals for the protection of beneficial uses. The revised lake eutrophication standards along with modifications to assessment procedures will improve assessment outcomes and reduce assessment errors. This is critical because much of the management of these aquatic resources stems from these standards including permitting, TMDLs, and Watershed Restoration and Protection Strategies (WRAPS). Refining lake eutrophication standards for northern lakes will improve Minnesota's ability to protect and restore these important resources through application of more appropriate goals and by improving management outcomes.

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Appendix A: Implementation of lake eutrophication standards

The development of water quality standards also requires a description of how these standards are intended to be implemented. A description of these methods, including determination of lake type and assessment of eutrophication parameters, is provided below. These protocols largely follow the existing methods described in Minnesota's assessment guidance manual (MPCA 2021). As such, the protocols outlined below are intended to reiterate some important elements of the existing methods and to augment methods where necessary.

i. Determination of nutrient region and lake type

The draft lake eutrophication standards were developed for application to dimictic and polymictic lakes in the northern region including the NLF and NMW ecoregions. Lakes on or near the ecoregion borders may need to be reviewed to determine a lake's placement into the three lake eutrophication regions. For example, watershed land use may be considered to determine affiliation with the lake nutrient regions (MPCA 2024).

Although the definition for shallow lakes is codified in rule (Minn. R. 7050.0150), there are lakes where this definition does not accurately reflect the conditions which define the appropriate lake stratification type. The distinction between dimictic and polymictic lakes in Minnesota's eutrophication standards is largely based on differences in mixing status between these waters. Dimictic lakes tend to thermally stratify during the summer while polymictic lakes do not stratify during the summer; although, they may periodically stratify between periods of mixing. The definition for shallow lakes in Minnesota Rule is a good predictor of lake mixing status. However, some lakes that meet the definition of a shallow lake do stratify and some lakes that do not meet this definition are polymictic lakes. As a result, it may be necessary to review some lakes in more detail using several lines of evidence to determine lake type. The objective is to characterize the overall condition of the lake, and no single attribute may be sufficient (Table 18). When water profile temperature data are available, these data may be used to ascertain the mixing status of these waters especially when measurements are available from multiple summers. Geometry ratio is also a good predictor of lake stratification status (Table 18; see Figure 6). However, there are many other lake attributes which may also be relevant for determining lake stratification including dominant substrate types, fishery, and beneficial uses (Table 18). For example, lakes that meet the definitions or criteria for shallow lakes (i.e., >80% of lake are littoral) or polymictic lakes (i.e., maximum temperature gradient <1 °C per meter or geometry ratio >4) with swimming beaches or other evidence of primary contact uses, may be appropriately assigned the dimictic lake eutrophication standard to protect those uses. Although many lakes will clearly fall into either of these categories, consideration of these factors is important to assign the appropriate eutrophication standards to lakes. In addition, the lake classification process may indicate the need to assign a site-specific standard to a lake.

Table 18. Attributes that may be used to determine lake stratification type (modified from MPCA [2024], Appendix D).

Attribute	Dimictic lakes	Polymictic lakes
Temperature gradient	At least >1 °C per meter	Maximum <1 °C per meter
Geometry ratio	<4 $m^{-0.5}$	>4 $m^{-0.5}$
Maximum lake depth (Z_{max})	Typically >15 feet (4.57 m)	Typically <15 feet (4.57 m)
Littoral habitat	Typically <80% of lake area	Typically >80% of lake area
Fetch	Significant fetch depending on size and shape	Fetch is variable depending on size and shape
Substrate	Consolidated sand/silt/gravel	Consolidated to mucky
Emergent vegetation and relative amount of open water	Shoreline may have ring of emergents; vast majority of basin open water	Emergents common, may cover much of fringe of lake; basin often has high percentage of open water
Submergent vegetation	Common in littoral fringe, extent dependent on transparency	Abundant in clear lakes; however, may be lacking in algal-dominated turbid lakes
Dissolved Oxygen	Aerobic epilimnion; hypolimnion often anoxic by midsummer	Aerobic epilimnion but wide diurnal flux possible
Fishery	Typically managed for a sport/game fishery. May be stocked. MNDNR fishery assessments typically available.	May or may not be managed for a sport fishery. If so, fishery assessment should be available. Winter aeration often used to minimize winterkill potential.
Uses	Wide range of uses including boating, swimming, skiing, fishing; boat ramps and beaches common	Boating, fishing, waterfowl production, hunting, aesthetics; limited swimming; may have boat ramp, beaches uncommon
Protected Waters Inventory (PWI) Code	Typically coded as “L or LP” in PWI	May be coded as either “L, LP or LW” in PWI

ii. Assessment of lake eutrophication standards in northern lakes

Implementing assessments for the draft northern lake eutrophication standards is not substantially different from the existing standards (MPCA 2024). This protocol is slightly modified for the northern lakes and this decision process is outlined in Figure 41. Assessments are based on the exceedance of the nutrient (i.e., TP) and one or both of the response parameters (i.e., chl-*a* or Secchi depth). To perform assessments, a minimum of two years of data within the last 10 years is needed. Each year requires a minimum of 4 sampling events, and these samples should be reasonably spaced through the summer index period (June through September). For monitoring and assessment purposes, chl-*a* should be considered preferable to Secchi depth because it provides a more proximate measure of beneficial use attainment. However, Secchi depth is useful in lakes where chl-*a* data are lacking and CDOM is low (see below) because when Secchi depth standard is exceeded, there is a high likelihood that the chl-*a* standard is also exceeded. In addition, if chl-*a* data are lacking and Secchi depth is between 0.9 and 2.1 m for polymictic lakes and 1.8 and 3.2 m for dimictic lakes (see Figure 39), this may indicate possible nonattainment of beneficial uses and follow up monitoring, especially for chl-*a*, may be appropriate.

The most substantial modification to lake eutrophication assessment protocols is the addition of explicit CDOM considerations when using Secchi depth in assessments. As part of the development of northern lake eutrophication standards, the effect of CDOM on Secchi depth was assessed to determine the level of CDOM that has important impacts on Secchi depth measurements used for assessment. This effect can result in inappropriate assessments of the lake eutrophication standards because high CDOM can negatively impact Secchi depth in the absence of high productivity (Brezonik et al. 2019). Previous and current research identified potential thresholds for CDOM (color or a_{440}) where Secchi depth is not appropriate for assessment or where Secchi depth should be scrutinized. Brezonik et al. (2019) determined that when CDOM, measured as a_{440}, exceeded 4 m^{-1}, Secchi depth did not provide a good determination of trophic status. We determined an additional a_{440} threshold of 1.4 m^{-1} as the level where CDOM begins to negatively affect Secchi depth. Equivalents for these a_{440} thresholds using color are 73 and 25 PCU. As a result, lakes with color >73 PCU or a_{440} >4 m^{-1}, should not be assessed using Secchi depth. Lakes falling between a color of 25-73 PCU or an a_{440} of 1.4-4 m^{-1} should be reviewed to determine if CDOM is affecting Secchi depth to the point where it does not provide an accurate measure of trophic status. For example, if Secchi depth is well below the threshold and CDOM is relatively close to the lower CDOM threshold, it may be appropriate to proceed with a recommendation for an impairment. As a result, lakes should not be assessed using Secchi depth if there are high or unknown levels of CDOM. Such lakes should only be assessed using TP and chl-*a*. Lake CDOM levels may be measured or estimated using several approaches including direct measurement of color or a_{440} within the lake. There is no minimum number of samples required to estimate lake CDOM. Other methods for estimating CDOM can include use of remote sensing (e.g., Olmanson et al. 2020), user surveys (e.g., EPA 2021b), or other relevant information such as field notes or photos.

Figure 41. Northern lake eutrophication assessment decision chart.