Como Lake: Water Quality Drivers Analysis Study

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DRAFT
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ACRONYMS AND ABBREVIATIONS

μg	Micrograms
BMP	Best Management Practices
Chl-a	Chlorophyll
CLP	Curlyleaf Pondweed
CRWD	Capitol Region Watershed District
DO	Dissolved Oxygen
EPA	Environmental Protection Agency
ha	Hectare
IQR	Interquartile Range
lbs	Pounds
m²	Square Meter
mg	Milligram
MNDNR	Minnesota Department of Natural Resources
MPCA	Minnesota Pollution Control Agency
N	Nitrogen
NH_3	Ammonia
NH_4	Ammonium
NO_2	Nitrite
NO_3	Nitrate
NOAA	National Oceanic and Atmospheric Administration
P	Phosphorus
RCPW	Ramsey County Public Works
RWMWD	Ramsey Washing Metro Watershed District
SRP	Soluble Reactive Phosphorus
TDS	Total Dissolved Solids
TKN	Total Kjeldahl Nitrogen
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids

Executive Summary

Como Lake is a hyper-eutrophic lake in St. Paul, MN that has experienced water quality problems associated with frequent algal blooms and winter fish kills for decades. Como Lake has annually exceeded shallow lake eutrophication standards for the North Central Hardwood Forest Ecoregion for total phosphorus (TP; < 60 μg/L or 0.06 mg/l) and Chlorophyll-a (Chl-a; <20 μg/l or 0.02 mg/l)), but has met the standards for Secchi depth (> 1 m) (CRWD 2015; MN Statute 7050.0222). Excessive nutrient loading, particularly phosphorus (external and internal), is the primary cause of water quality issues in Como Lake. Capitol Region Watershed District (CRWD) has installed numerous stormwater best management practices (BMPs) throughout the watershed, which have reduced external nutrient loading to Como Lake. Continued efforts to further reduce external loading in addition to controlling internal loading of phosphorus will be required to improve water quality conditions in Como Lake.

The overall objective for this project was to determine the primary water quality drivers in Como Lake, which is an important step prior to the development of a lake management plan. Therefore, CRWD contracted LimnoTech to conduct a water quality drivers analysis study. We analyzed long-term records in chemical, biological, and physical data collected by Ramsey County Public Works in order to assess the primary drivers of water quality. The results of this study will serve as a foundation for development of a lake management strategy, which will be conducted in a later phase. This report presents the results of the diagnostic study of water quality drivers in Como Lake. Key findings are highlighted below.

High phosphorus concentrations in Como Lake have resulted in excessive algal blooms during the summer growing season (May – September). Decomposition of algae results in depleted, or anoxic, dissolved oxygen concentrations in the lake. Depleted oxygen in this shallow lake has serious implications for the success of biological communities and will favor species that are tolerant to suboptimal habitat conditions. Furthermore, dissolved phosphorus is released from the sediments under anoxic conditions which further contributes to the water quality issues.

The phytoplankton community analysis revealed a dominance of blue-green algae (*Cyanobacteria)* with peak densities in mid-late summer each year along with the presence of toxin producing species (i.e. *Microcystis* and *Anabaena*). The presence of toxins from blue-green algae in Como Lake has not been measured. An increase in blue-green algal density often coincided with total nitrogen to total phosphorus mass ratios less than 10, which is indicative of nitrogen limiting conditions. Blue-green algae are capable of fixing atmospheric nitrogen giving them a competitive advantage over other algal taxa under these conditions. Nitrogen limiting conditions in a eutrophic lake indicates that phosphorus concentrations are too high and further reductions are needed.

Long-term changes in the zooplankton community structure included increased density of cladocerans over time. Higher densities of small-bodied cladocerans relative to large-bodied zooplankton taxa was likely the result of predation pressure by planktivorous fish (i.e. fish that primarily feed on zooplankton), which dominates the fish community in Como Lake. Size selective feeding on zooplankton (i.e. consumption of large zooplankton species) from planktivorous fish have likely contributed to the changes in zooplankton structure. The consequence of these changes to the zooplankton community is reduced grazing pressure on algal species as small-bodied zooplankton do not graze on phytoplankton as efficiently as large-bodied zooplankton species. The low density of piscivorous fish (i.e. fish that primarily feed on other fish)

in conjunction with continued fish stocking of planktivorous fish is likely contributing to the imbalanced food web in Como Lake. As a result of these top-down (i.e. regulated by fish) and bottom-up controls (i.e. regulated by nutrient inputs), the water quality issues in Como Lake are complex and will require a combination of additional external and internal load reductions and biological community management for water quality improvements.

We conducted a screening-level assessment to evaluate the relative magnitude of internal P loading through various mechanisms, which included:

- Watershed loads (external loads)
- Release of dissolved phosphorus from lake bottom sediments (internal load)
- Wind-driven resuspension of sediment phosphorus (internal load)
- Bioturbation and macrophyte community influence on sediment P flux (internal loads)

External loading and internal diffusive flux from sediments were found to be the primary drivers of phosphorus loading. Watershed loads based on modeling from a previous study and diffusive flux estimates based on monitoring data provided the greatest amount of certainty with respect to loading rates. However, the estimated range of diffusive sediment flux calculated in this study is large, and depends on the temporal and spatial extent of anoxia at the sediment-water interface. Consequently, CRWD is continuously monitoring dissolved oxygen in Summer 2017 to determine the spatial and temporal extent of anoxia which will better constrain flux estimates.

Considerable uncertainty exists with quantification of phosphorus loading rates due to bioturbation and macrophyte senescence. Curlyleaf pondweed, an invasive species, is of particular concern because it dies off (senesces) earlier in the year than native macrophytes. As a result, it releases nutrients into the water column when conditions are optimal for algal growth. These source loads are inherently challenging to quantify and literature estimates are limited. Additional monitoring and/or site-specific studies would be required to better constrain estimates of phosphorus loading rates from these sources. Consequently, we were unable to estimate a load from bioturbation with the available information. We estimated a potential phosphorus loading rate from CLP senescence, but this likely represents an upper bound estimate as it assumes that all phosphorus within the CLP present in the lake is released to the water column through senescence. Mechanical harvesting reduces CLP occurrence prior to senescence, but the data show that partial communities of CLP remain after harvesting which contribute P upon senescence.

In summary, the water quality issues in Como Lake are complex with many different contributing factors. Despite considerable reductions in external loading, nutrients loads from the watershed continue to be significant. After decades of watershed loading to Como Lake, internal nutrient loads are also significant and will be difficult and costly to control. Given the extent of eutrophication and the imbalanced ecological functioning in Como Lake, there is no one-size-fits-all approach or technique. A holistic approach that incorporates multiple strategies to managing all aspects of the lake will be required to improve water quality conditions in Como Lake. Therefore, we recommend that the following actions be investigated for future management:

- Continue ongoing efforts to reduce external loads

- Manage anoxia (e.g. aeration)

- Alum treatment to inactivate sediment derived dissolved phosphorus

- Macrophyte management, which should include long-term strategies to control CLP and establish healthy native aquatic vegetation

- Biomanipulation of the fish community to establish a balanced foodweb

Due to the long history of nutrient loading to Como Lake, and the complexity of causal factors, improvements in water quality will be long-term and costly. There is not a one-size-fits-all approach to remedy Como Lake. Rather, a holistic approach that integrates multiple strategies along with adaptive management will be required to improve water quality conditions in Como Lake.

A "healthy" Como Lake will have reduced nutrient loading (external and internal) to the extent that the lake meets water quality standards; it will be free of dense floating algae; possess water clear enough to support native macrophyte vegetation that can take up nutrients, stabilize sediments and provide refugia for zooplankton and fish; and it will support a balanced aquatic food web.

1
Introduction

LimnoTech was contracted by Capitol Region Watershed District (CRWD) to conduct a water quality drivers analysis study of Como Lake, which is a hyper-eutrophic shallow lake in St. Paul, MN.

1.1 Background on Como Lake

Como Lake has experienced water quality problems associated with frequent algal blooms and winter fish kills since at least 1945 (Noonan 1998). Management efforts to improve water quality in Como Lake have been ongoing for decades with intermittent improvement through fisheries biomanipulation and application of rotenone (piscicide), application of copper sulfate treatment (algal fungicide), aeration and mechanical harvesting of macrophytes (see Noonan 1998 for details). Excessive nutrient loading, particularly phosphorus (external and internal), is the primary cause of water quality issues in Como Lake (Noonan 1998; CRWD 2015). CRWD has installed numerous stormwater BMPs throughout the watershed, which have reduced some external nutrient loading to Como Lake. Further improving water quality conditions in Como Lake will also require addressing internal loading of phosphorus.

Como Lake has annually exceeded shallow lake eutrophication standards for the North Central Hardwood Forest Ecoregion for total phosphorus (TP; < 60 µg/L or 0.06 mg/l) and Chlorophyll-a (Chl-a; <20 µg/l or 0.02 mg/l), but has met the standards for Secchi depth (> 1 m) (CRWD 2015; MN Statute 7050.0222). For impairment determination, Minnesota assesses TP and Chl-a data collected during the summer season with summer-average calculations limited to the upper 3 meters of the water column. Minnesota defines the summer season from June 1 – Sept 30 (MN Statute 7050.0150). Como Lake has been listed on the MPCA's 303(d) list of impaired waterbodies since 2002 (CRWD 2015).

1.2 Study Purpose

The purpose of this study was to evaluate long-term physical, biological and chemical data to better understand drivers of water quality in Como Lake. This report presents the results of the study, which included the following objectives:

(1) Analysis of in-lake biological, chemical and physical data to determine long-term and seasonal data trends, assess within-lake spatial variability, determine potential periods of nitrogen (N) limitation, and determine typical mixing and stratification patterns in Como Lake.

(2) Develop a screening-level phosphorus (P) mass balance to evaluate the relative contribution of various sources and mechanisms of internal P loading, assess the uncertainty in source contributions, and determine the potential primary versus secondary drivers of P loading and in-lake dynamics.

(3) Develop a planning-level conceptual aquatic food web model to assess current biological community structure and functional feeding groups to describe food web interactions and highlight key drivers of community structure.

The results of these analyses, and data collection information, are presented in the following sections. An overview of the available monitoring data, sampling frequency and station information is discussed in Chapter 2.

2

Lake Characteristics and Monitoring Data

The watershed area of Como Lake is 1,856 acres while the lake surface area is 70.5 acres (CRWD 2015). Como Lake has a large watershed to lake area ratio of 26.3, which indicates that the watershed strongly influences the lake through runoff from the drainage basin. Watershed to lake area ratios greater than 10 are considered to have a large watershed area relative to the lake surface area (Holdren et al. 2001). Additionally, the drainage area of Como Lake is dominated by urban land use with artificial drainage networks and direct storm sewer inputs to the lake.

Ramsey County Public Works (RCPW) conducts monitoring in Como Lake through funding provided by CRWD. CRWD provided this data to LimnoTech, which includes chemical, biological and physical data generally collected biweekly during May through October from three stations over the period of 1984-2016 (Table 1; Figure 1).

Table 1. Depth and period of record at the three sampling stations in Como Lake. Historical station name shown in parentheses.

Station	Water Depth (m)	Period of Record
102 (5404)	2.3	1984-1993
201 (5401)	4.5	1984-2016
202 (5402)	2.7	1985, 1987-1988
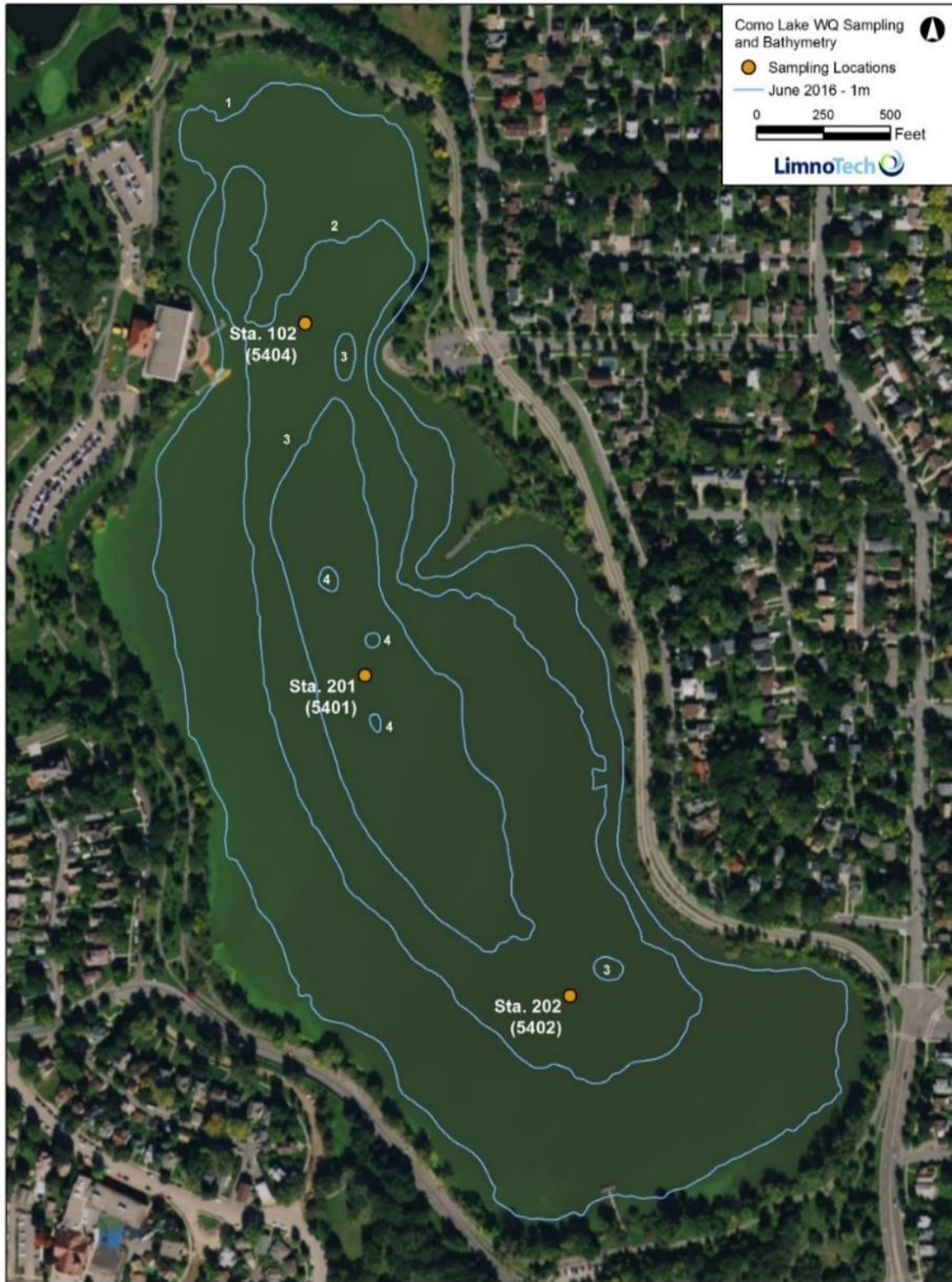
Figure 1. Bathymetry of Como Lake with sampling stations. Historical station names are shown in parentheses.

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3
Primer on Shallow Lake Ecology

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"A healthy shallow lake is characterized by water clear enough to see to the bottom, with fairly thick weed beds with fish and other aquatic organisms using the plants for both food and protection. Everything is connected as the aquatic plants use nutrients to grow, hold sediment on the bottom, and provide a place for zooplankton to hide from predators." (MSU 2010).

3.1 Shallow Lake Ecology

Shallow lakes are characterized in Minnesota as having aquatic plants and water depths less than 15 feet (MDNR 2017a). In many ways, shallow lakes have unique characteristics as compared to deeper lakes. The following sections describe some characteristics of shallow lakes to provide an ecological foundation for understanding water quality drivers in Como Lake.

3.1.1 Clear Versus Turbid Lake States

Clear, shallow lakes with low nutrient concentrations are typically dominated by submersed macrophytes. Shallow lakes with continued nutrient loading often transition to a turbid state dominated by algae that no longer supports a healthy macrophyte community (Wetzel 2001; Scheffer 2004; MSU 2010). The clear or turbid state of a shallow lake is illustrated in Figure 2. The shift to turbid, phytoplankton-dominated shallow lake can be the caused by several factors (Wetzel 2001; Scheffer 2004):

1) High external and/or internal nutrient loading;
2) Proliferation of algae that can outcompete submersed plants for nutrients and light;
3) Establishment of invasive species;
4) Absence of submersed plants that provide refuge for large zooplankton that would graze on algae. Lack of zooplankton habitat leaves zooplankton vulnerable to predation by planktivorous fish, which has cascading effects on the success of the algal community; and
5) Establishment of cyanobacterial community (i.e. blue-green algae), which contain many species that are inedible by zooplankton.

Reversal of the turbid, algal-dominated state to a healthy, clear, macrophyte-dominated shallow lake is challenging and costly. The challenge is even greater in urban areas with high watershed nutrient loading and human utilization. Typically, when a lake suffers from multiple factors causing degradation, one management technique will not address all of the issues causing impairment. In these cases, an integrated strategy that is tailored for the lake with multiple management actions will be the most successful plan for water quality improvement.
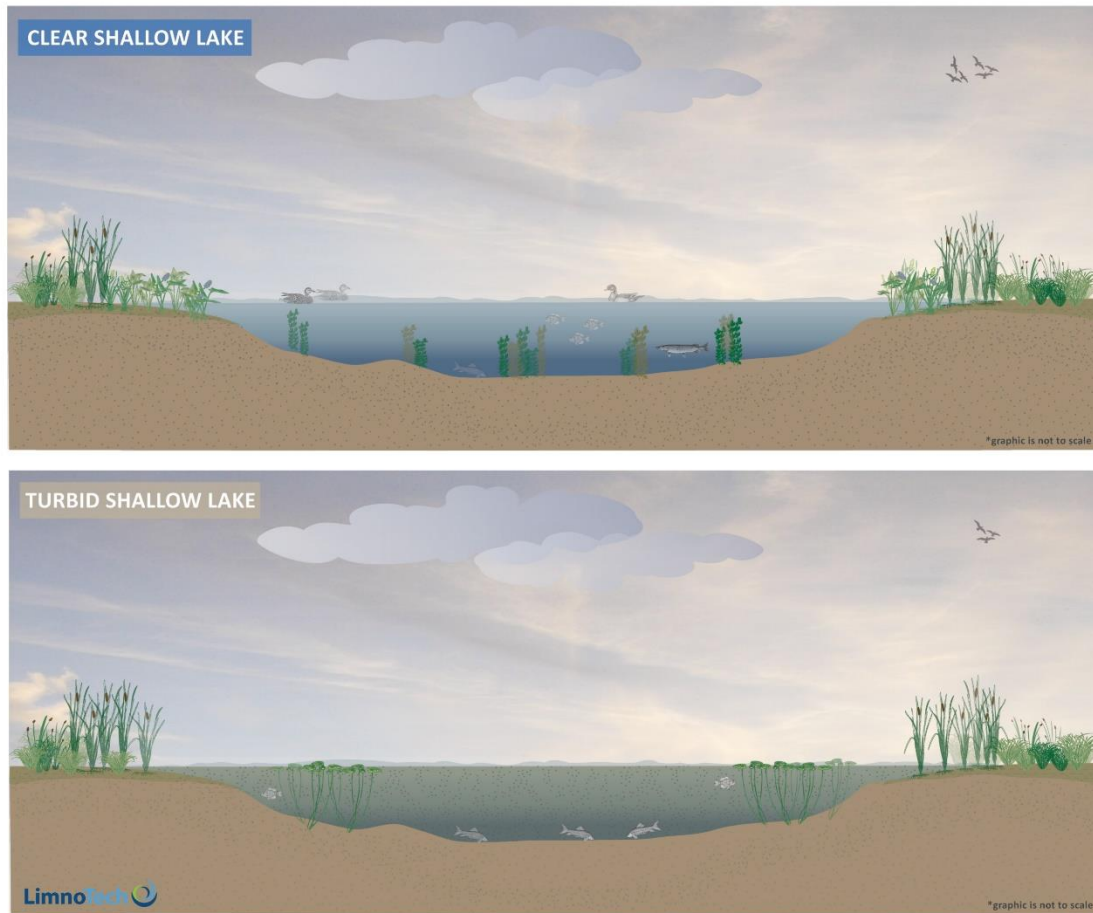
Figure 2. Schematic representation of a clear shallow lake with submerged aquatic vegetation (upper panel) and a shallow lake in a turbid state with dense algae and few aquatic plants (lower panel). Graphic adapted from Scheffer (2004).

3.1.2 Thermal stratification and mixing regimes

Deep lakes develop a vertical temperature gradient where well-mixed surface waters (epiliminion) are warmer than the colder, denser bottom waters (hypolimnion). The zone of transition is referred to as the thermocline. This thermal stratification is usually fairly stable until forces such as wind cause the stratified layers to mix completely. The mixing event is referred to as "turnover" and typically occurs in spring and fall. Shallow lakes, in contrast, may mix several times a year (i.e. polymictic) and tend to exhibit weak thermoclines that can set up fairly quickly on hot summer days. The frequent mixing in shallow lakes implications for chemical and biological processes and related impacts on water quality. Furthermore, shallow lakes have high areal sediment-water contact, which means sediment processes can have a disproportionate impact on the chemistry of overlying water column. For example, organic matter deposited on the sediments (e.g. algal cells) consumes oxygen as it decomposes. This can lead to depletion of oxygen in the surficial sediments and hypolimnion leaving insufficient oxygen for aquatic life in the bottom waters of the shallow lake. Additionally, phosphorus can be released from the sediments when

oxygen is depleted (discussed further in the next section) which then provides a source of phosphorus for algal growth when the lake mixes. Depending on the magnitude and duration of external nutrient loading, shallow lakes may still exhibit high internal phosphorus loading even after external loads are reduced or removed. The primary mechanisms of internal P loading are briefly discussed in the following section.

3.2 Internal P loading

There are four primary pathways of internal P loading in shallow lakes (Figure 3). Each of these processes are briefly discussed below.
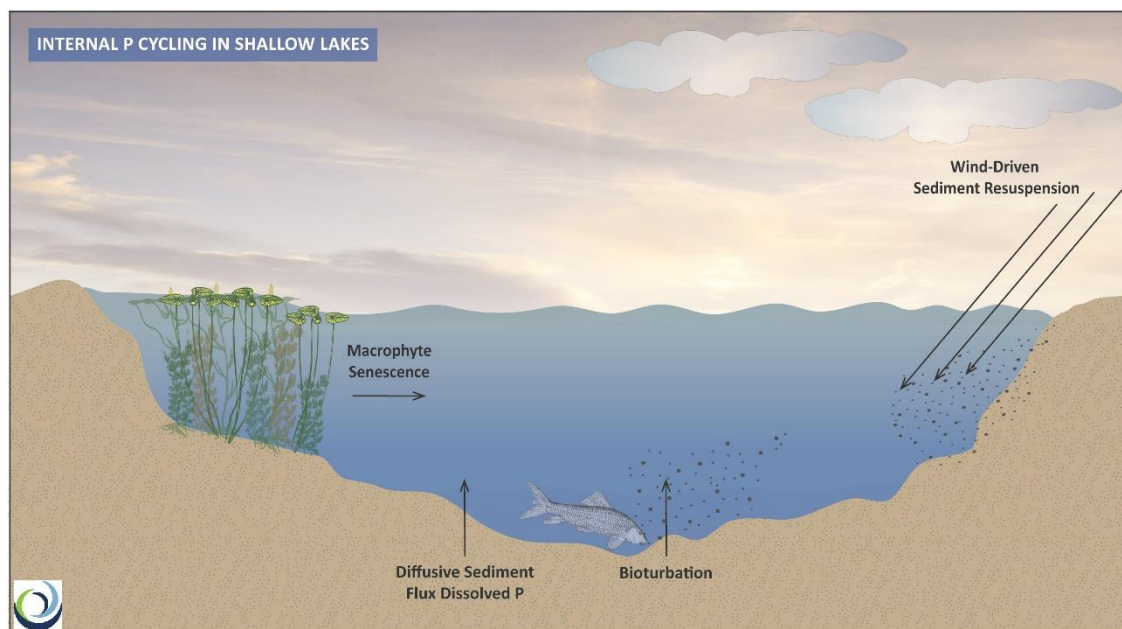
Figure 3. Major pathways of internal phosphorus loading in shallow lakes.

1) Diffusive sediment flux of dissolved P
The release of iron-bound dissolved P in the sediments under anoxic conditions is a classically described mechanism for P transport across the sediment-water interface. Hypolimnetic accumulation of P during stratification becomes available to upper water layers when the lake mixes. Unassimilated P will then return to the sediments during natural sedimentation processes continuing the internal loading cycle (Wetzel 2001).

2) Wind-driven sediment resuspension
Sediment derived phosphorus can be released following wind events capable of inducing sediment resuspension. Fetch length, which is the uninterrupted distance over the surface of the lake which winds acts, can influence wind-driven sediment resuspension. Lakes with long fetch distances have more expanse of water for wind energy to cause wave action (Wetzel 2001). These physical conditions of the lake allow wind-driven surface turbulence to reach the lake bottom and

promote P release through sediment resuspension. It is challenging to quantify, but has been shown to increase several orders of magnitude within a few days of wind events in some lakes (Sondergaard et al. 2003).

3) Bioturbation

Bioturbation involves disturbance of the sediments via bottom-dwelling fish and/or benthic-dwelling invertebrates (Wetzel 2001). Fish can disturb sediments through predation and reproductive behavior (e.g., nest building).

4) Macrophyte senescence

Macrophytes can have a positive and negative impact on nutrient dynamics in lakes. While macrophytes take up nutrients during the growing season and may stabilize sediments (i.e. guard against resuspension), they also release P at senescence (i.e. die-off) (Sondergaard et al. 2003). Curlyleaf pondweed, an invasive macrophyte, can be a significant source of P to lakes due to the timing of its decay. Curlyleaf pondweed (CLP) tends to die-off in late June in the Upper Midwest, which results in release of mobile P as temperature and light conditions are ideal for algal growth and development. This early season pulse of P occurs when light and temperature conditions are optimal for phytoplankton growth, which can lead to algal blooms.

4
Water Quality Trends

LimnoTech evaluated the temporal and spatial trends in the available biological, chemical and physical data to establish a baseline understanding of historical and current conditions in Como Lake. These analyses served as a foundation for deeper assessmnet into sources and mechanisms of P dynamics in Como Lake (see Chapter 6), which will be used to inform management strategies for improving water quality.

4.1 Typical Growing Season Conditions

LimnoTech evaluated bi-monthly dissolved oxygen (DO) and temperature profiles collected during the growing season (May – September) in Como Lake during the period of record (1984-2016). Spatial and temporal patterns in DO conditions throughout the lake can provide important information on lake mixing regimes, habitat quality for aquatic life, and conditions for dissolved P flux from the sediments.

4.1.1 Dissolved Oxygen & Temperature Patterns

LimnoTech assessed dissolved oxygen (DO) and temperature profiles from Station 201 to determine typical mixing and stratification patterns in Como Lake. Figure 4 shows the measured DO and temperature profiles for 2016. Similar plots for 2010-2015 are shown in Appendix A. These plots show that Como Lake typically mixes in early spring following ice-off, thermally stratifies in June and then mixes again in August or September.   The profiles further show that the epilimnetic and hypolimnetic waters in Como Lake may partially or fully mix in some years, which indicates that the lake is polymictic, a mixing pattern typical of shallow lakes (Scheffer 2004). The profiles show that the thermocline is often fairly weak, but tends to set up fairly quickly following a water column mixing event. The more noticeable and interesting pattern in these figures is the presence of strong, persistent oxycline (i.e. oxygen concentration gradient) that is typically present even with a weak thermocline. This condition points to a high demand for oxygen in the hypolimnion and/or surficial sediments. High hypolimnetic or sediment oxygen demand is typical for eutrophic lakes with lots of organic matter (i.e. high algal productivity) that consumes oxygen during decomposition (Horne and Goldman 1994; Wetzel 2001). The duration of summer temperature and oxygen conditions at Station 201 are illustrated in Figure 5 and Figure 6. These values were interpolated between the bi-weekly sampling events to create the temperature and DO isopleths. These isopleths further show that even with no, or a very weak thermocline, Como Lake develops very strong oxycline with hypoxic or anoxic bottom waters.

These patterns have implications for field sampling practices as well. Upon review of the data, and according to Noonan (1998), the decision to sample the hypolimnion during a given sampling event appears to be based on the presence or absence of a thermocline, which is determined in the field. However, the persistence of a strong oxycline in the absence of a thermocline emphasizes the need to base hypolimnetic sampling on the oxycline as well as the thermocline. The presence or absence of oxygen regulates key biogeochemical processes as well as diffusive flux of soluble P from sediments. Thus, monitoring beneath the oxycline in the absence of a thermocline can provide important information for in-lake management in the ongoing effort to improve water quality conditions in Como Lake.
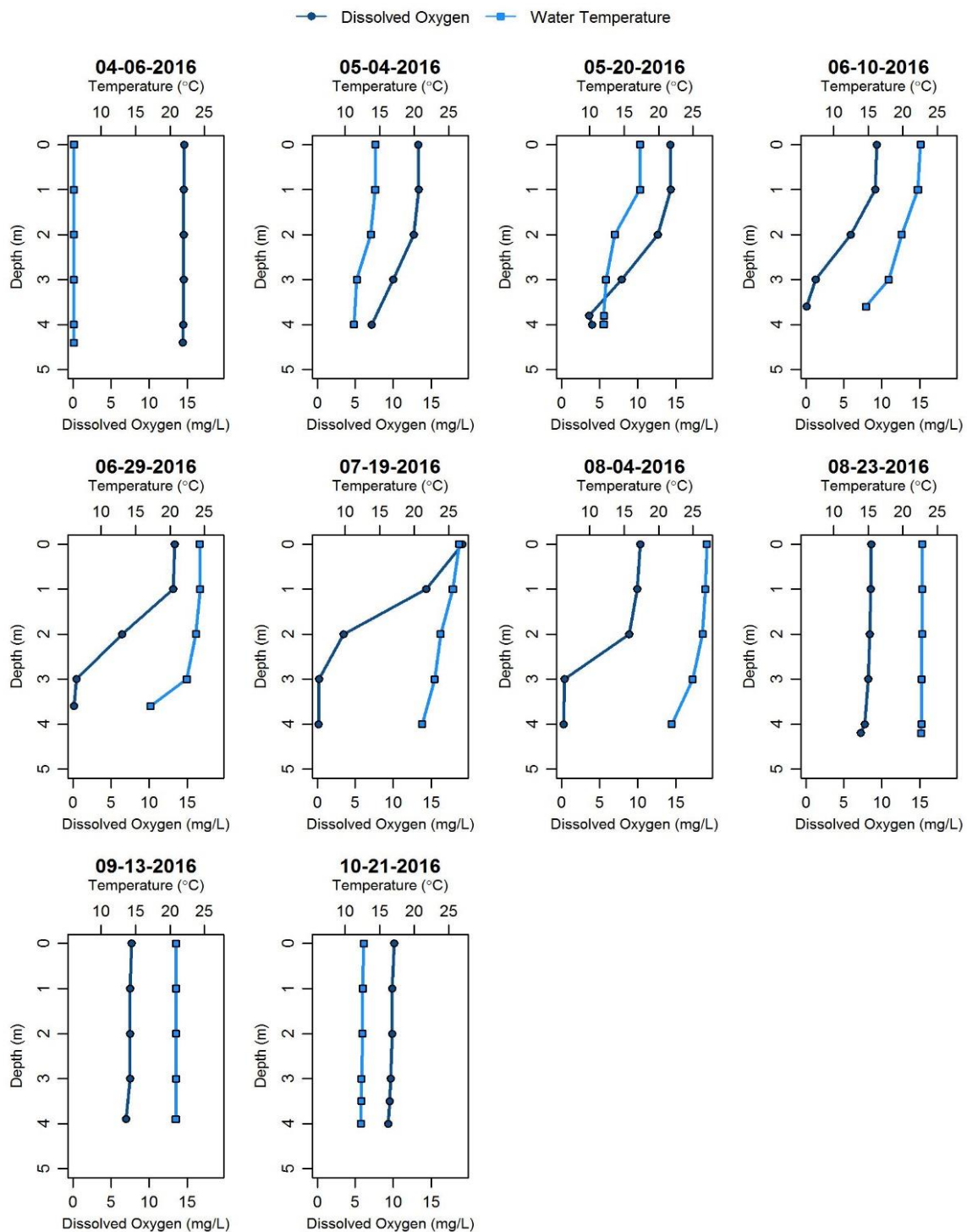
Figure 4. Dissolved oxygen (DO) and temperature profiles measured at Station 201 in 2016.
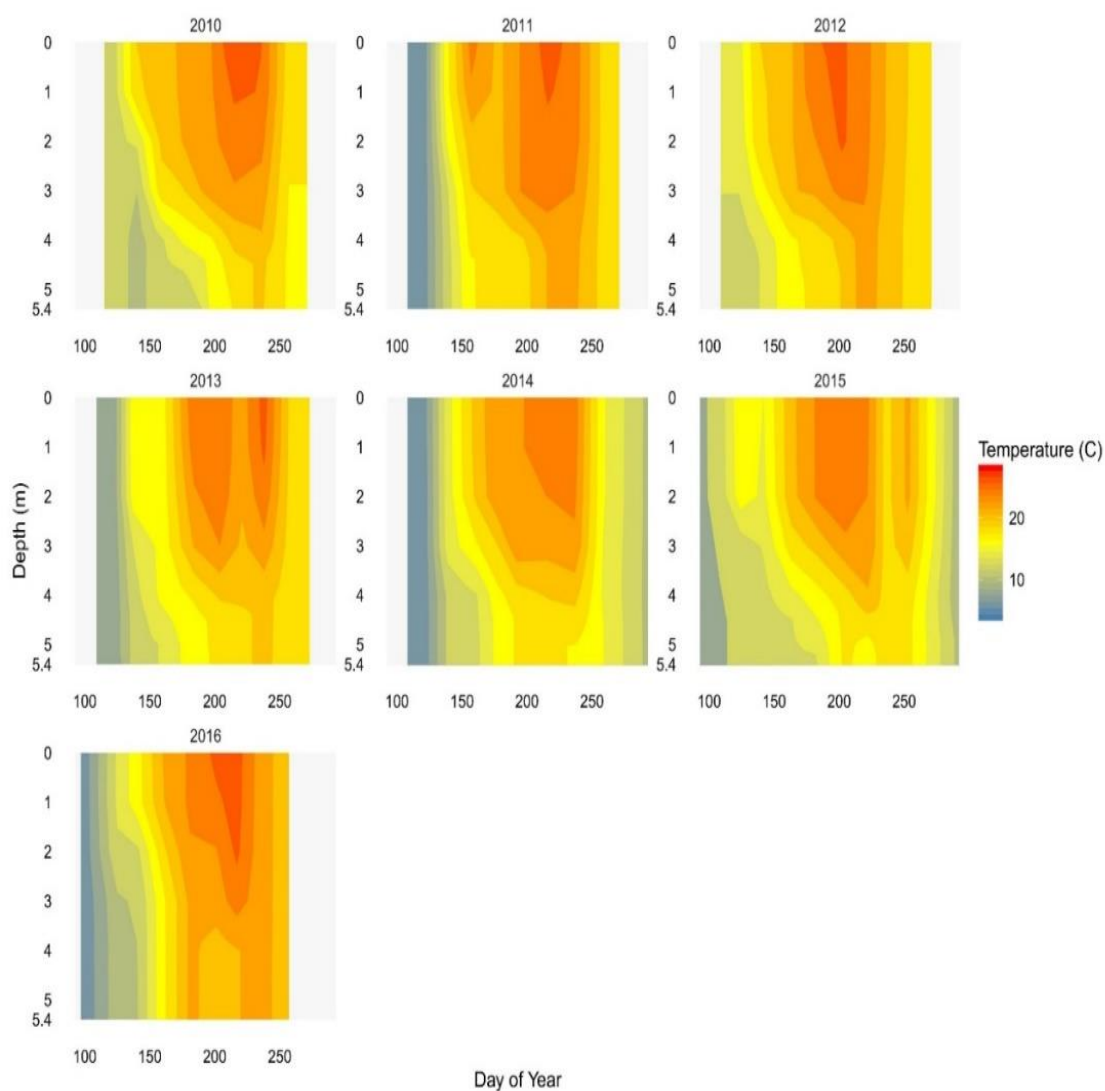
Figure 5. Temperature isopleths created from approximate bi-weekly measurements at Station 201 (2010-2016). For reference, July 19 corresponds to Julian Day 200 non-leap years, and Julian Day 201 in leap years. Note that 2016 was a leap year.
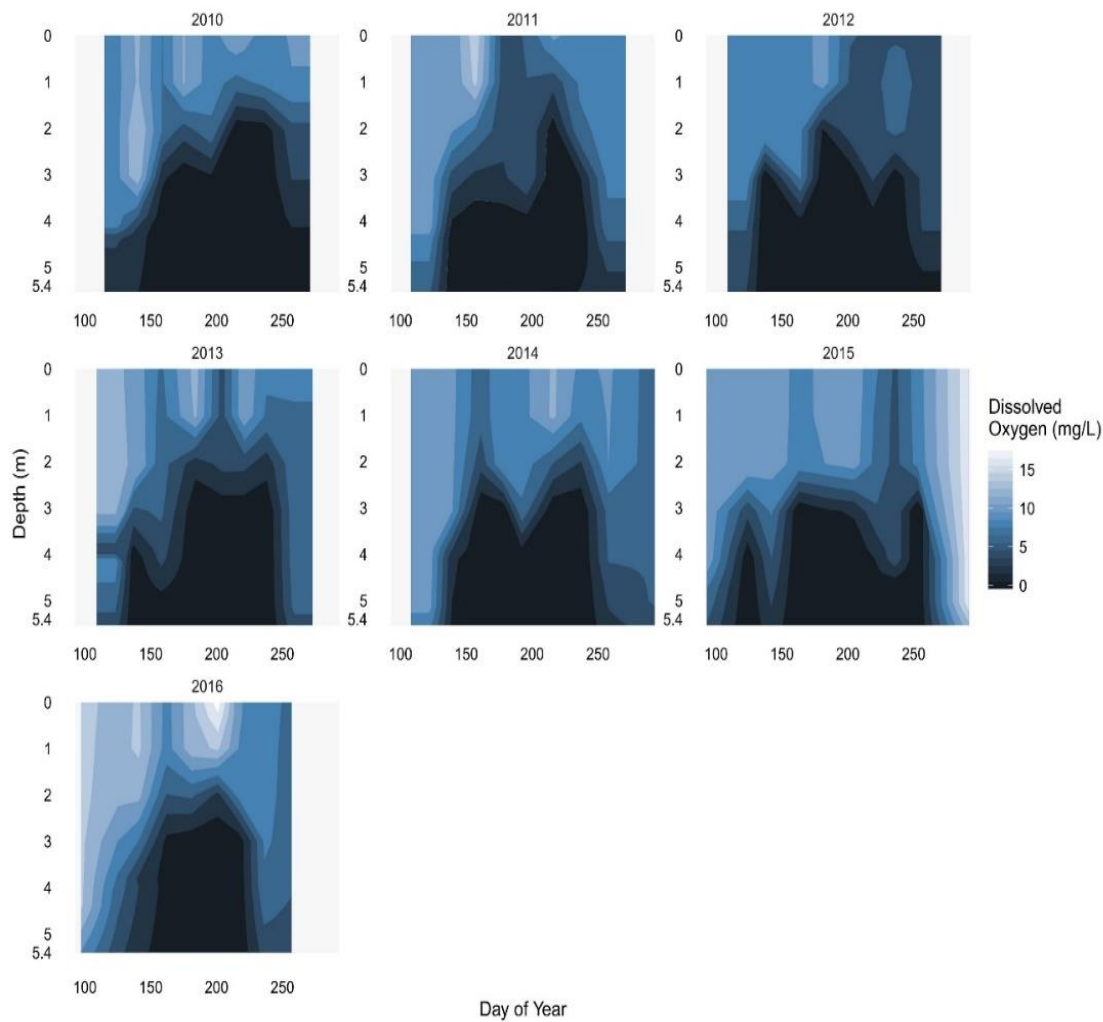
Figure 6. Dissolved oxygen isopleths created from approximate bi-weekly measurements as measured at Station 201 (2010-2016). For reference, July 19 corresponds to Julian Day 200 non-leap years, and Julian Day 201 in leap years. Note that 2016 was a leap year.

4.1.2 **Spatial and Temporal Anoxia**

The longest period of record for data in Como Lake exists at Station 201 (5401), which spans the years 1984-2016 (See Chapter 2 for details). Data also exist for Station 102 (5404) and Station 202(5402) for the years 1984-1993, and 1985, 1987-1988 respectively.

For the period of comparison (1984-1993), we found considerable differences in DO dynamics among stations. These differences indicate that spatial uniqueness exists between shallow and deep stations suggesting that the shallow station should be reestablished as a routine monitoring station to better understand P dynamics and related biological responses.

Far fewer hypolimnetic samples have been collected over time compared to epilimnetic samples (e.g.,Table 2 – discussed in more detail below). As mentioned above, Como Lake is polymictic, which means that the lake may mix multiple times throughout the year and typically does not

exhibit *sustained* thermal stratification during the growing season when the lake is sampled most frequently. It is our understanding that the decision to collect hypolimnetic samples is based on the presence of thermal stratification (RCPW 2009). Therefore, a hypolimnetic sample may not always be collected, which may explain the difference in sample size between the epilimnion and hypolimnion data. Figure 7 and Figure 8 illustrate DO and temperature patterns throughout the water column at the deep station (201/5401) and shallow station (5404/102) for the years 1991-1993. At Station 201 (5401), anoxic conditions are apparent throughout most of the summer growing season, which would be expected given that depth and the lake's trophic status. However, these figures show that anoxic conditions also developed at the shallow station 5404/102 for some or all of the summer growing season in the years 1991-1993, which is less commonly reported in shallow lakes. Figure 8 further shows that an oxycline develops at both stations even when a strong thermocline does not, which points to high oxygen demand near and/or in the surficial sediments. The data suggest that the areal extent of hypoxic or anoxic conditions is potentially larger than what is represented by deep station profiles only. This points to the need to collect DO and temperature profiles throughout the water column at shallow and deep stations even when a thermocline is not present. To determine if current conditions reflect historical patterns, CRWD has installed continuous DO data loggers in Como Lake at all three sampling stations to monitor the period of May - October 2017. These data will be used to quantify the spatial and temporal extent of anoxic conditions in Como Lake which will be used to refine the estimate of dissolved sediment P flux (discussed further in Chapter 6).
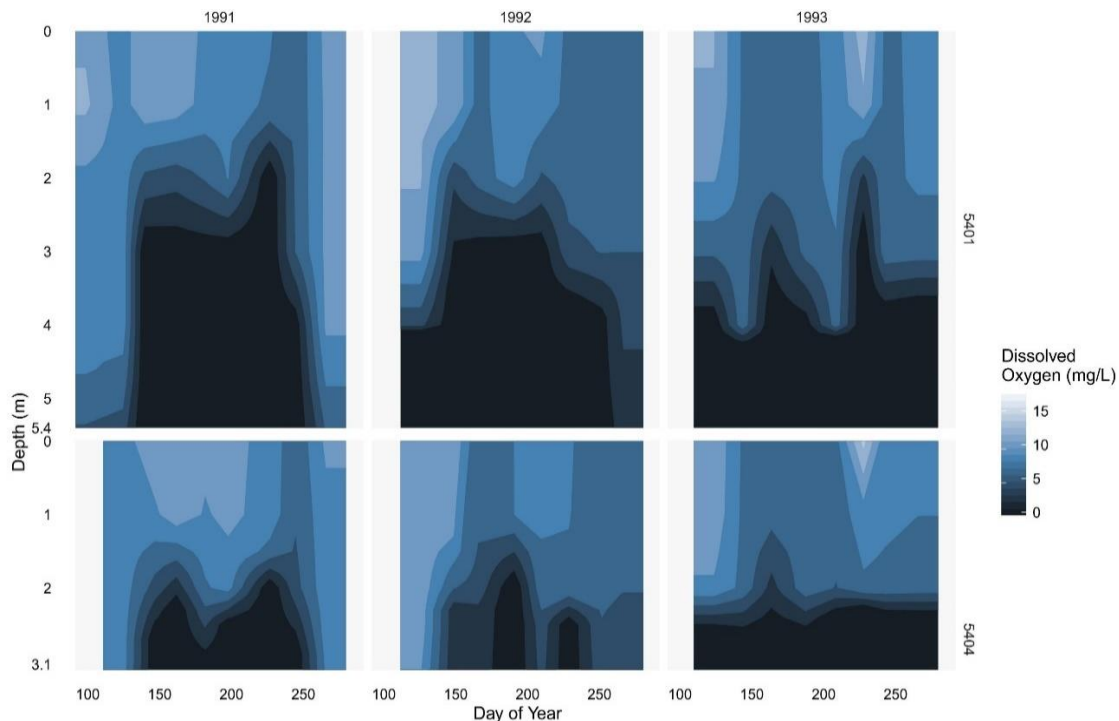
Figure 7. Temperature isopleths showing low dissolved oxygen concentration in the deep station (201/5401) and shallow station (5404/102) for the years 1991-1993.
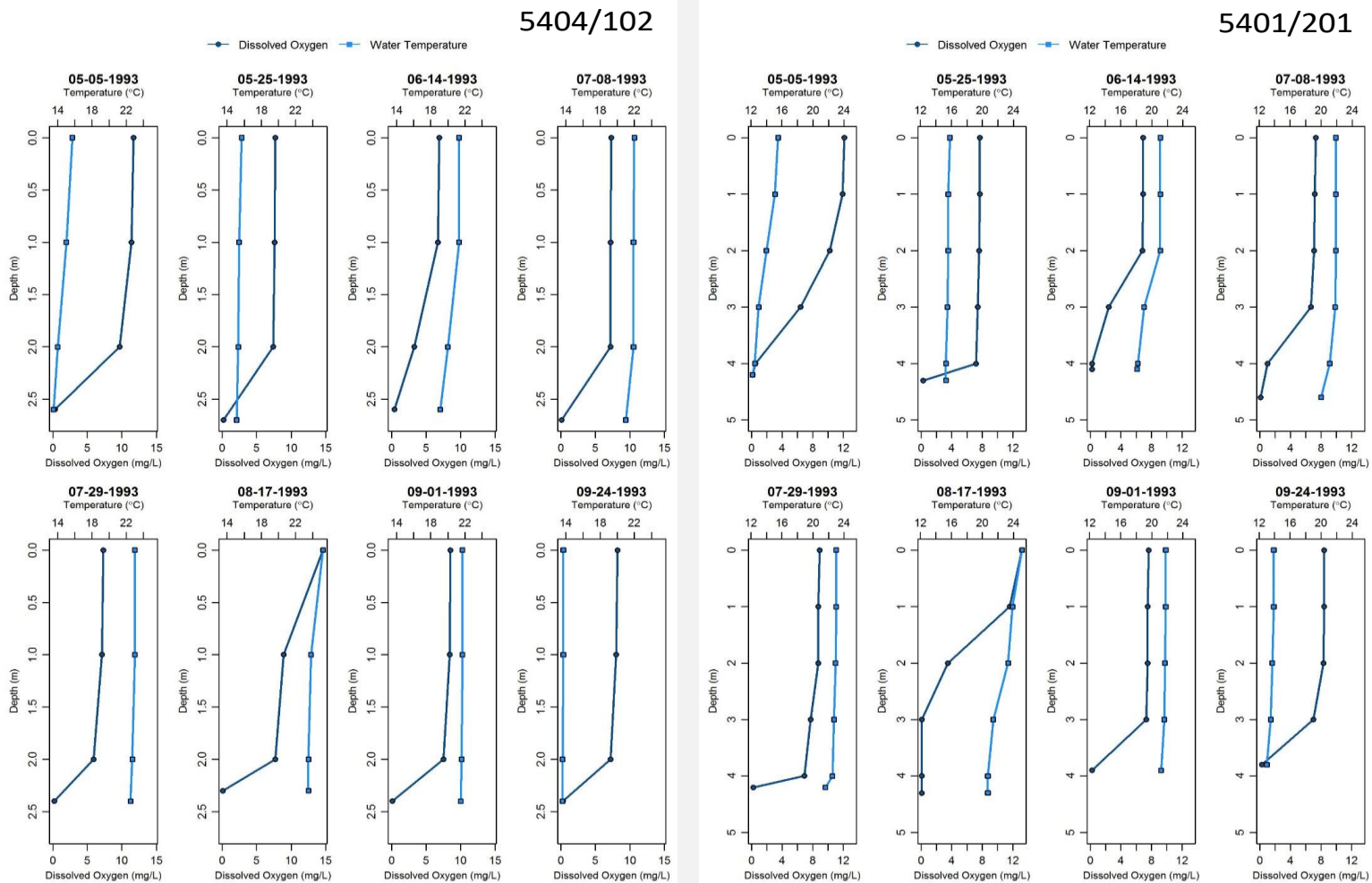
Figure 8. Comparison of dissolved oxygen and temperature profiles at the deep (5401/201) and shallow (5404/102) stations in 1993.

4.2 Trends in Eutrophication Related Parameters

Inter-annual and intra-annual trends in eutrophication related parameters are discussed this section followed by an analysis of the spatial variability in data trends. Trends in the non-eutrophication related parameters (i.e. conductivity, chloride, water temperature, water surface elevation, and precipitation) can be found in Appendix B. While these are important parameters that inform water quality conditions, they do not directly drive eutrophic conditions. The primary goal of this project is to diagnose drivers of water quality as it relates to internal nutrient loading and eutrophication. Therefore, only the parameters directly related to eutrophication are included in the main body of the report.

4.2.1 Summary of Statistics and Analysis Methods

LimnoTech used R Statistical Software (version 3.3.1) in all data analyses. Basic summary statistics were calculated for data from all stations during the growing season (May – Sept) across the entire data record. The long-term mean, standard deviation, range of historical values, and number of samples for each strata are shown in Table 2. The long-term mean epilimnetic TP concentration (0.2 mg/l; Table 2) exceeds the TP water quality criteria for shallow lakes in this region. Long-term mean epilimnetic Chl-a was 39.8 µg/l, which exceeds water quality criteria for shallow lakes in this region area. The long-term mean Secchi depth (Table 2) does meet water quality criteria. Applicable water quality standards are discussed in Chapter 1. Note that these comparisons are used to assess current water quality conditions, and not impairment determination which is to be conducted by the MPCA.

All references to ammonia (NH_3) or ammonium (NH_4) throughout this report refer to the equilibrated measurement of un-ionized ammonia plus ionized ammonium (i.e. $NH_3 + NH_4^+$) and are simply noted as NH_4. All reported Chl-a measurements have been corrected for pheophytin (RCPW 2012). Pheophytin is a degradation product of Chl-a, and represents detrital or decaying algal cells. The Chl-a measurements can be overestimated without this correction (Lorenzen 1965). Details on analysis methods and corrections can be found in Eaton et al. (2012). Total nitrogen (TN) was not measured directly, but was calculated by summing nitrate (NO_3) and total Kjeldahl nitrogen (TKN), which is a measure of NH_4 and organic nitrogen. Note that NO_3 reported here represents nitrate+ nitrite (NO3+NO2). In many cases, NH_4 and NO_3 values were reported at the limit of detection (0.02 mg/l for each) indicating that the true values are equal to, or below this limit and could not be accurately quantified by the reporting laboratory. In these instances, LimnoTech maintained the reported value of 0.02 mg/l for NH_4 and NO_3 in the data analyses. Consequently, estimates of NH_4, NO_3 and TN represent upper bounds, and the true values could be slightly lower than what is reported here.

Table 2. Summary statistics for historical data in Como Lake. Data represent the growing season (May – Sept) across the entire period of record (1984-2016) from all stations.

Parameter	Strata	Mean	SD	Min	Max	N	Date Range	
Ammonium (mg/l)*	Epilimnetic	0.2	0.45	0.01	6.4	755	5/15/1984	9/13/2016
Ammonium (mg/l)*	Hypolimnetic	0.5	0.68	0.01	4.6	125	5/15/1984	7/19/2016
Chl-a (µg/l)	Epilimnetic	39.8	40.99	0.2	238.3	751	5/15/1984	9/13/2016
Chl-a (µg/l)	Hypolimnetic	33.6	26.58	0.2	131.0	40	6/29/1984	7/19/2016
Chloride (mg/)	Epilimnetic	103.9	52.21	28.0	290.0	654	5/15/1984	9/13/2016
Chloride (mg/)	Hypolimnetic	127.3	49.51	29.0	275.0	136	6/21/1985	8/4/2016
Conductivity (µS/cm)	Epilimnetic	436.5	187.47	128.0	1028.0	1058	5/15/1984	9/13/2016
Conductivity (µS/cm)	Hypolimnetic	585.1	193.23	123.0	1103.0	342	5/15/1984	8/4/2016
Dissolved Oxygen (mg/l)	Epilimnetic	7.1	3.59	0.0	19.2	1486	5/15/1984	9/13/2016
Dissolved Oxygen (mg/l)	Hypolimnetic	2.0	3.17	0.0	15.4	479	5/15/1984	8/4/2016
Ferrous iron (µg/l)	Epilimnetic	144.7	99.28	19.0	421.0	40	5/3/1989	9/28/1994
Ferrous iron (µg/l)	Hypolimnetic	1074.2	1340.32	19.0	5670.0	31	5/17/1989	9/8/1994
Hardness (mg/l)	Epilimnetic	55.3	17.81	20.0	140.0	605	5/15/1984	9/30/2005
Hardness (mg/l)	Hypolimnetic	58.7	17.84	24.0	134.0	104	5/15/1984	8/1/2005
Nitrate (mg/l)*	Epilimnetic	0.03	0.05	0.01	0.5	760	5/15/1984	9/13/2016
Nitrate (mg/l)*	Hypolimnetic	0.02	0.03	0.01	0.2	125	5/15/1984	7/19/2016
Organic Particulate Matter (mg/l)	Epilimnetic	8.7	6.99	0.0	35.0	632	5/15/1984	9/30/2005
Organic Particulate Matter (mg/l)	Hypolimnetic	9.4	6.85	1.0	26.0	32	5/15/1984	5/5/1993
pH	Epilimnetic	8.2	0.94	6.1	10.4	1062	5/15/1984	9/13/2016
pH	Hypolimnetic	7.4	0.69	6.3	9.7	339	5/15/1984	8/4/2016
Secchi depth (m)		1.4	0.93	0.2	4.2	370	5/15/1984	9/13/2016
Soluble reactive phosphorus (mg/l)	Epilimnetic	0.1	0.07	0.0	0.7	855	5/15/1984	9/13/2016
Soluble reactive phosphorus (mg/l)	Hypolimnetic	0.1	0.11	0.0	0.6	197	5/15/1984	8/4/2016
Total alkalinity (mg/l)	Epilimnetic	48.5	13.60	21.0	91.0	622	5/15/1984	9/30/2005
Total alkalinity (mg/l)	Hypolimnetic	52.6	14.46	21.0	119.0	105	5/15/1984	8/1/2005
Total iron (mg/l)	Epilimnetic	0.6	0.34	0.1	1.3	40	5/3/1989	9/28/1994
Total iron (mg/l)	Hypolimnetic	1.6	1.28	0.2	6.3	32	5/17/1989	9/8/1994
Total Kjeldahl nitrogen (mg/l)	Epilimnetic	1.5	0.67	0.3	5.2	750	5/15/1984	9/13/2016
Total Kjeldahl nitrogen (mg/l)	Hypolimnetic	1.6	0.96	0.4	8.1	123	5/15/1984	7/19/2016
Total nitrogen (mg/l)*	Epilimnetic	1.5	0.67	0.3	5.2	733	5/15/1984	9/13/2016
Total nitrogen (mg/l)*	Hypolimnetic	1.7	0.94	0.4	8.1	120	5/15/1984	7/19/2016
Total nitrogen: Total phosphorus*	Epilimnetic	10.1	5.30	0.7	68.3	716	5/15/1984	9/13/2016
Total nitrogen: Total phosphorus*	Hypolimnetic	7.2	4.21	0.9	22.0	118	5/15/1984	7/19/2016
Total particulate matter (mg/l)	Epilimnetic	11.0	8.37	0.0	60.0	645	5/15/1984	9/30/2005
Total particulate matter (mg/l)	Hypolimnetic	12.5	8.27	2.0	31.0	34	5/15/1984	5/5/1993
Total phosphorus (mg/l)	Epilimnetic	0.2	0.19	0.0	2.7	858	5/15/1984	9/13/2016
Total phosphorus (mg/l)	Hypolimnetic	0.4	0.42	0.1	2.4	198	5/15/1984	8/4/2016
Turbidity (NTU)	Epilimnetic	9.1	8.52	0.3	62.5	817	5/15/1984	9/13/2016

Parameter	Strata	Mean	SD	Min	Max	N	Date Range	
Turbidity (NTU)	Hypolimnetic	11.7	12.45	1.5	110.0	195	5/15/1984	8/4/2016
Water temperature (°C)	Epilimnetic	20.3	4.18	7.3	28.0	1480	5/15/1984	9/13/2016
Water temperature (°C)	Hypolimnetic	18.5	3.69	10.0	28.9	480	5/15/1984	8/4/2016

* denotes upper bound estimates due to the presence of values reported at the limit of detection (i.e. 0.02 mg/l for ammonia/ammonium and nitrate)

LimnoTech employed non-parametric techniques to assess statistical relationships over time, between select parameters and among stations following methods in Helsel and Hirsch (2002). The Mann-Whitney-Wilcoxon test was used to compare epilimnetic and hypolimnetic TP and SRP at Station 201 (Section 4.2.2). The significance of temporal trends was assessed by applying the Mann-Kendall Trend test. Mann-Kendall Trends test were computed with values collected during the growing season only (May – Sept) to remove seasonal variability from the analyses, and thus represent temporal trends in the summer growing season for the period of record 1984-2016 (Section 4.2.2). The Kruskal-Wallis test was used to compare data among stations (Section 4.2.4). Finally, statistical correlations among key environmental parameters were assessed using Spearman's rank-order correlation tests based on whole-lake data (i.e. all depths, all stations) for the growing season only (Section 4.2.5).

4.2.2 Inter-annual Trends

Boxplots showing inter-annual trends in the chemical and physical parameters for the growing season (May – Sept) are shown in Figure 9 through Figure 16. Data shown in these figures represent values from station 201 only. The center line in all boxplots represent the median whereas the box represents the interquartile range (IQR; 25th-75th percentile), and the upper and lower whiskers represent 1.5* IQR.

Many of the parameters shown in Figure 9 through Figure 16 exhibit high inter-annual variability. Inter-annual variations in climate conditions (i.e. precipitation, cloud cover, air temperature) can have significant effects on external loading, and chemical and biological responses of the lake (Scheffer 2004). Noonan (1998) observed distinct cyclical patterns in water quality between the pre-treatment (rotenone application) and the post-treatment period (1982-1985, and 1986-1997, respectively) as well as variability in the biological responses to manipulations. Some of the patterns were attributed to the observed inter-annual variability in precipitation, but significant ecosystem changes were also detected from fisheries manipulation (i.e. rotenone and fish-stocking) and in-lake management during the study period, which included copper sulfate application (algaecide), and mechanical harvesting of macrophytes.

Over the period of record in this study (1984-2016), mean hypolimnetic TP and SRP exceeded mean epilimnetic concentrations (Table 2), which may point to internal loading of P. On average, epilimnetic and hypolimnetic TP differed by 71%, and epilimnetic and hypolimnetic SRP differed by 110%.The Mann-Whitney-Wilcoxon tests indicated that the median hypolimnetic TP and SRP concentrations were significantly greater than median epilimnetic TP and SRP (Table 3). Median annual TP and SRP concentrations showed high inter-annual variability (Figure 9 and Figure 10) similar to that discussed in Noonan (1998) which described conditions through 1997. Over the period of record described in this current study (1984-2016), median epilimnetic TP decreased slightly while hypolimnetic TP, and epilimnetic and hypolimnetic SRP increased (Table 4). These trends may indicate an increase in internal loading of P over time. However, these trends should

be loosely interpreted as there are clearly strong inter-annual forces influencing nutrient dynamics in Como Lake that also likely vary from year to year and must be kept in mind when evaluating long-term changes in the lake.

The period of 2010-2016 was unique for hypolimnetic TP and SRP compared to previous years. Median annual hypolimnetic TP and SRP concentration was highest in 2014 and 2012, respectively. In addition, the range of TP and SRP in each of those years is much greater than preceding years (exception being 1995 and 2000-2001), pointing to a high degree of intra-annual variability. Median epilimnetic TP and SRP began to increase in 2010-2011, which could be influencing patterns observed in hypolimnetic SRP in 2012-2014. However, the elevated hypolimnetic SRP that sustained the growing season 2012 and 2013 suggest a higher degree of internal P loading compared to other years. Median hypolimnetic NH_4 was high in 2012 and 2013 (Figure 12) as well, which can only persist under anoxic conditions further supporting the likelihood of P loading from diffusive sediment flux.

While some epilimnetic and hypolimnetic TP or SRP peaks occur correspond in some years, the lack of consistent temporal and spatial (i.e. between epilimnetic and hypolimnetic concentrations) coherence in TP and SRP among lake strata points to a complex interaction of external and internal loading of P with differential controls among years (see Figures 9 and 10). It's important to note that disentanglement of sources and sinks of P in eutrophic urban lakes is challenging due to the temporal and spatial complexity of interacting mechanisms. There are many different ways to approach data analysis in this type of study. In this study, we focused on identification of the trends in chemical, biological and physical data which provided the foundation for deeper investigation into potential mechanisms of internal P loading that can be addressed through lake management (see Chapter 6).

Table 3. Mann-Whitney-Wilcoxon tests to compare epilimnetic and hypolimnetic TP or SRP.

Parameter	Estimate	Statistic	p-value (α = 0.05)
Total phosphorus (mg/l)	-0.087	40,167	1.52E-15
Soluble reactive phosphorus (mg/l)	-0.025	42,135	2.30E-10

Table 4. Mann-Kendall Trend tests for in-lake water quality data by strata at Station 201 only. Data represent the growing season only (May – Sept) for the 1984-2016 period of record. Bold italic number indicate significant trends (α = 0.05).

Parameter	Strata	Kendall's tau	p-value (α = 0.05)
Secchi depth (m)		-0.013	0.749
Turbidity (NTU)	epilimnion	*0.065*	*0.006*
Chl-a (µg/l)	epilimnion	*-0.072*	*0.012*
Soluble reactive phosphorus (mg/l)	epilimnion	*0.079*	*0.003*
Soluble reactive phosphorus (mg/l)	hypolimnion	*0.192*	*<0.0001*
Total phosphorus (mg/l)	epilimnion	*-0.081*	*0.002*
Total phosphorus (mg/l)	hypolimnion	*0.075*	0.120
Ammonia/ammonium (mg/l)	epilimnion	*-0.300*	*<0.0001*
Ammonia/ammonium (mg/l)	hypolimnion	-0.113	0.065
Total Kjeldahl nitrogen (mg/l)	epilimnion	0.008	0.768
Total Kjeldahl nitrogen (mg/l)	hypolimnion	0.015	0.807
Nitrate (mg/l)	epilimnion	0.019	0.478
Nitrate (mg/l)	hypolimnion	-0.031	0.589
Chloride (mg/l)	epilimnion	*0.193*	*<0.0001*
Chloride (mg/l)	hypolimnion	*0.202*	*0.001*
Conductivity (µS/cm)	epilimnion	*0.252*	*<0.0001*
Conductivity (µS/cm)	hypolimnion	*0.144*	*<0.0001*
Water temperature (°C)	epilimnion	0.026	0.530
Water temperature (°C)	hypolimnion	-0.006	0.854
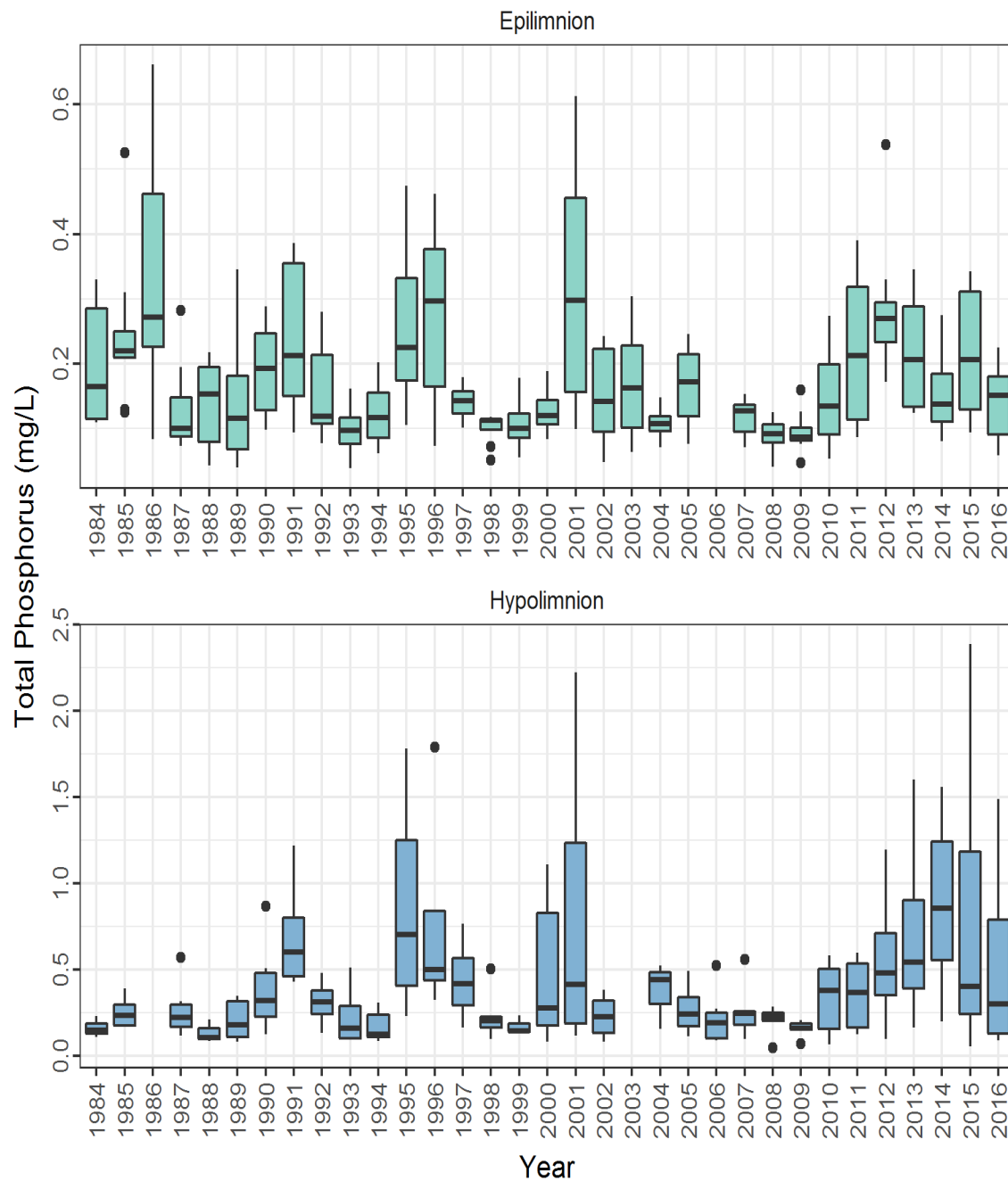
Figure 9. Inter-annual variability in epilimnetic and hypolimnetic total phosphorus during the growing season (May – Sept) at Station 201.
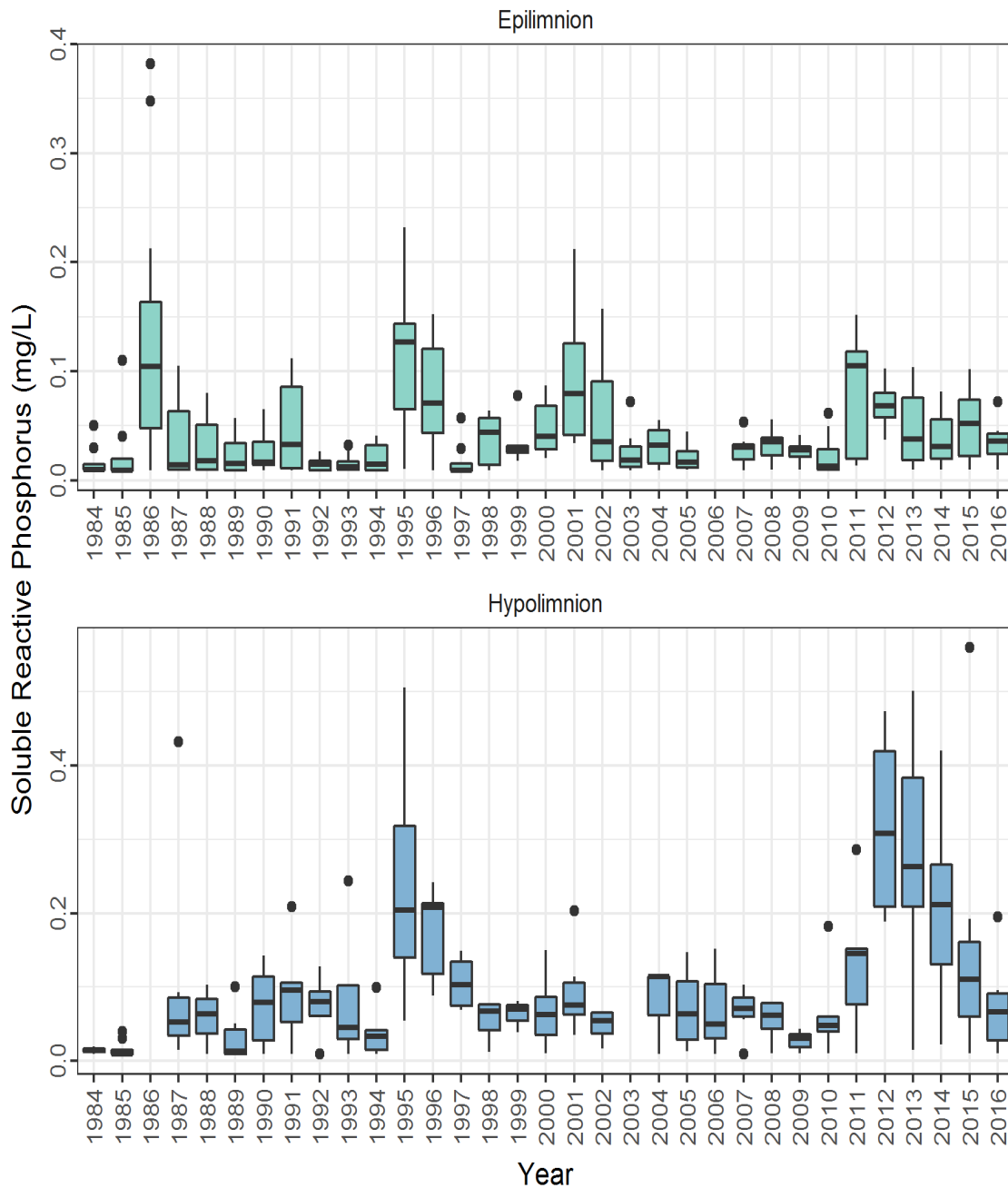
Figure 10. Inter-annual variability in epilimnetic and hypolimnetic soluble reactive phosphorus during the growing season (May – Sept) at Station 201.

Epilimnetic total nitrogen (TN) trends (Figure 11) in Como Lake exhibit a cyclical pattern similar to TP and SRP but do not show an anomalous pattern in recent years as described above with TP and SRP. Epilimnetic and hypolimnetic NH_4 and NO_3 (Figure 12 and Figure 13) were often near the limits of detection throughout the period of record with sparse hypolimnetic measurements which is reflected in hypolimnetic TN measurements. It's important to note that NH_4 rapidly

oxidizes to NO_3 in the presence of oxygen, which is a biogeochemical process termed nitrification (Wetzel 2001). Thus, NH_4 will persist in that chemical form in anoxic hypolimnetic waters. Additionally, NH_4 is a byproduct of organic matter degradation (i.e. decaying algae or fish) so accumulation in hypolimnetic waters usually points to decomposition under anoxic conditions. As noted above, hypolimnetic N measurements are somewhat sparse especially in the last ten years. The presence of NH_4 in the hypolimnion in 2012 could point to organic matter degradation under anoxic conditions, which would also lead to increased diffusive flux of soluble P from the sediments, a pattern discussed above (Figure 10). Coupled with data reported at the limit of detection, the sparsity of hypolimnetic N data prevent additional investigation into this relationship in other years. This situation further supports the practice of sampling beneath the oxycline in the absence of a distinct thermocline.

As noted above, TKN is a measure of NH_4 and organic N. When calculated for the entire period of record, TKN comprised 99% of TN concentrations in Como Lake. The low concentrations of NH_4 indicate that the majority of TKN is comprised of organic N which is cellular-bound in proteins, amino acids and other organic compounds in algae and bacteria (Horne and Goldman 1994; Scheffer 2004). Therefore, these N compounds are not readily available for phytoplankton uptake and indicate potential intermittent N limitation (see Section 4.3). Only epilimnetic NH_4 showed a significant trend during the growing season (May – Sept) over the period of record (Table 4). As noted above, however, trends should be interpreted loosely as 1) hypolimnetic data are sparse and 2) strong inter-annual forces are driving observed patterns.
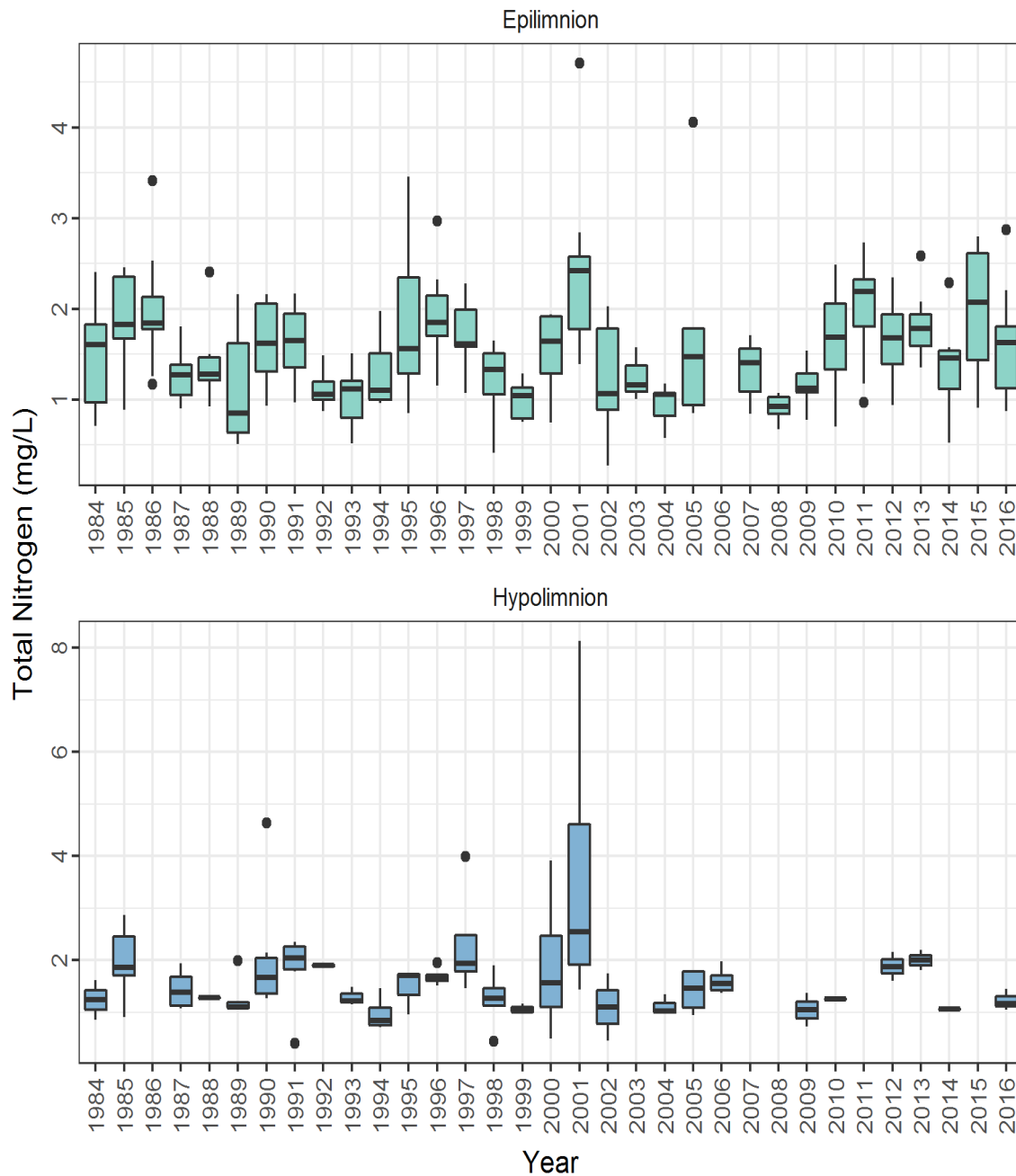
Figure 11. Inter-annual variability in epilimnetic and hypolimnetic total nitrogen during the growing season (May – Sept) at Station 201.
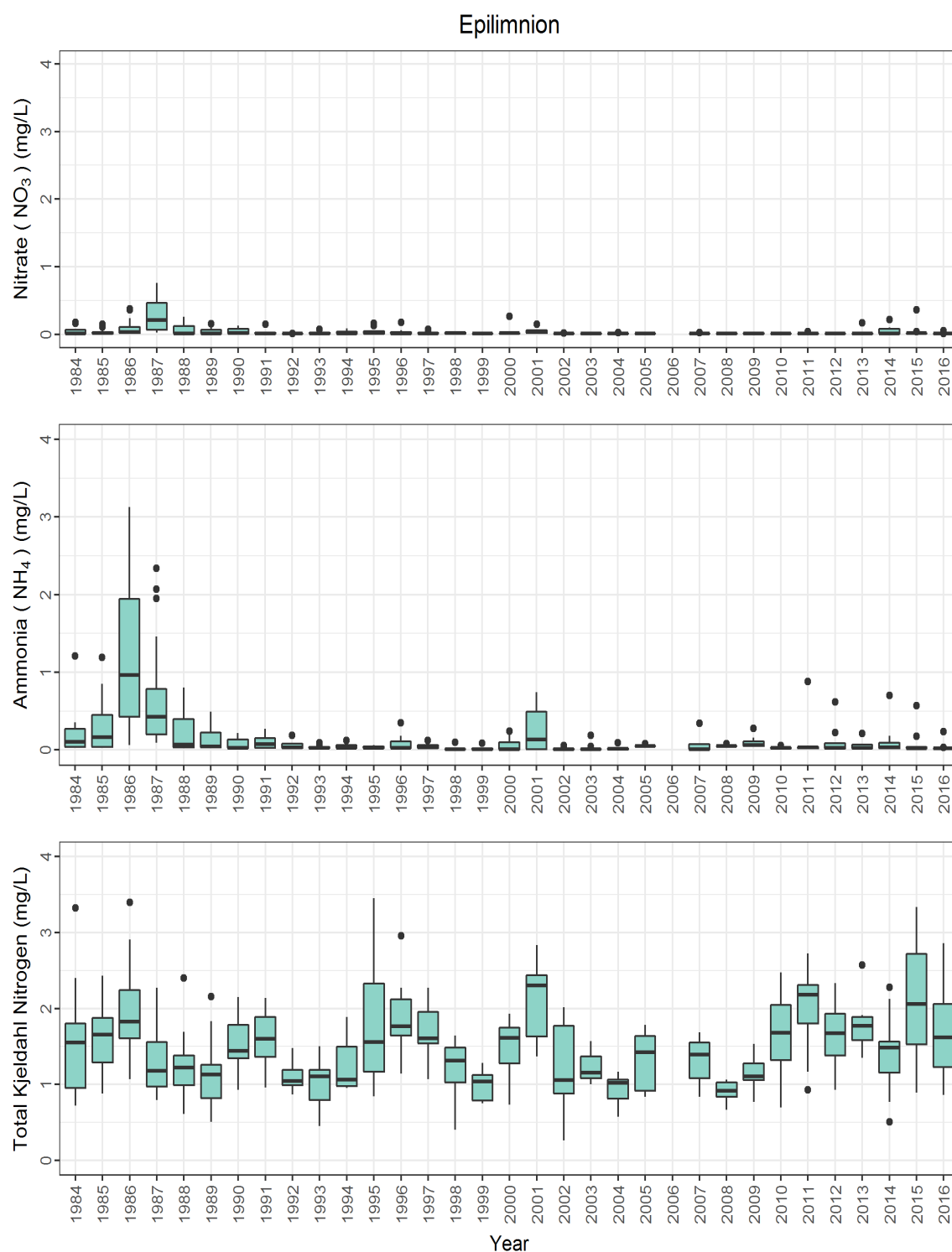
Figure 12. Inter-annual variability in epilimnetic nitrate, ammonia/ammonium, and total Kjeldahl nitrogen during the growing season (May – Sept) at Station 201. Note that the majority of reported values for ammonia/ammonium and nitrate were at or below the laboratory limit of detection.
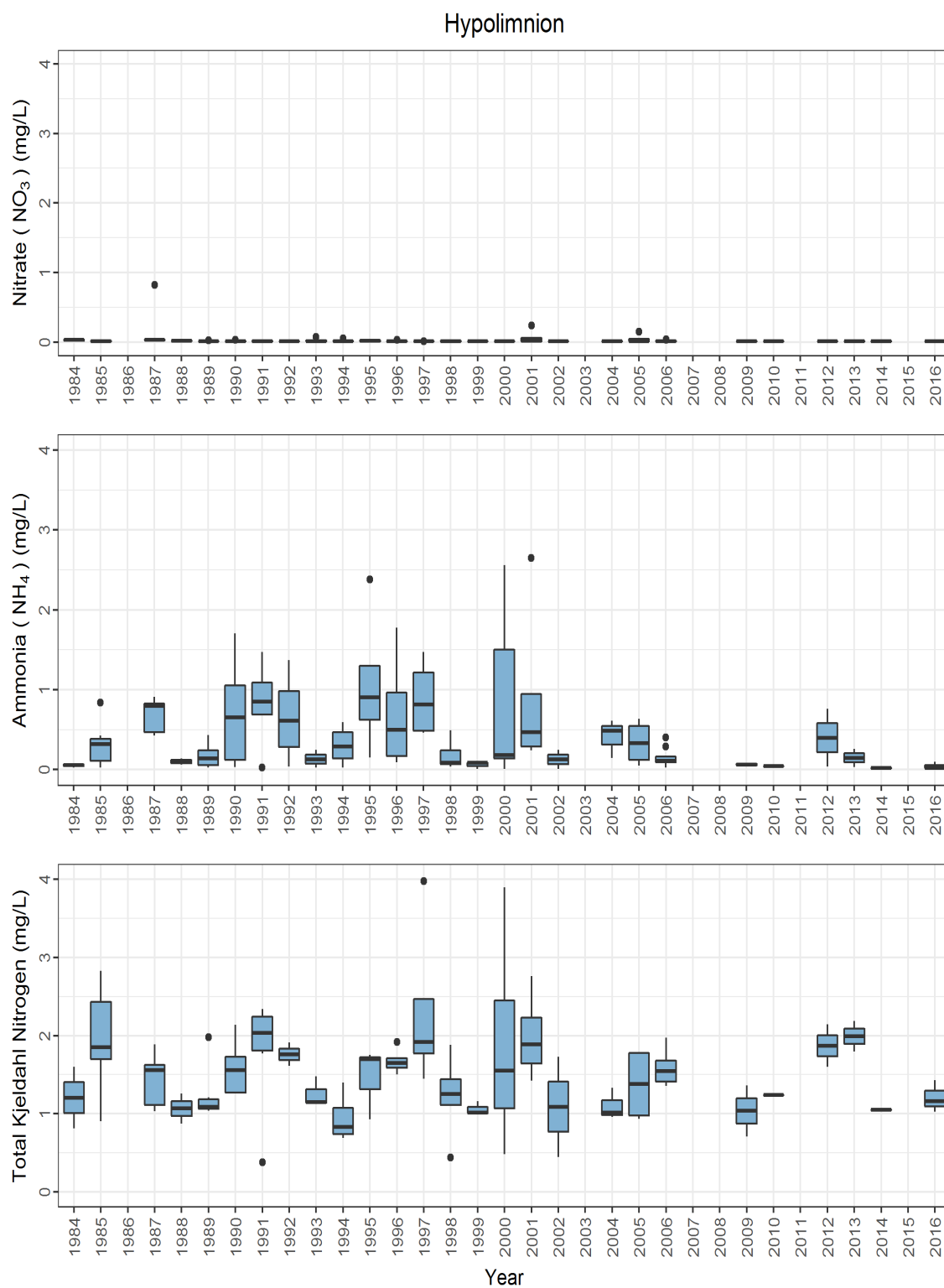
Figure 13. Inter-annual variability in hypolimnetic nitrate, ammonia/ammonium, and total Kjeldahl nitrogen during the growing season (May – Sept) at Station 201.

Chl-a provides an estimate of algal biomass. Algal growth is limited by N and P availability, so the Chl-a patterns should correspond with the most limiting nutrient during the growing season, which appears to be the case for the epilimnetic values (Figure 14). Epilimnetic Chl-a concentrations were highest in the mid-1980's with inter-annual patterns similar to P and N in subsequent years. Epilimnetic Chl-a during the growing season decreased slightly with time (Table 4), which is likely a function of the higher Chl-a values in the 1980s relative to later years.
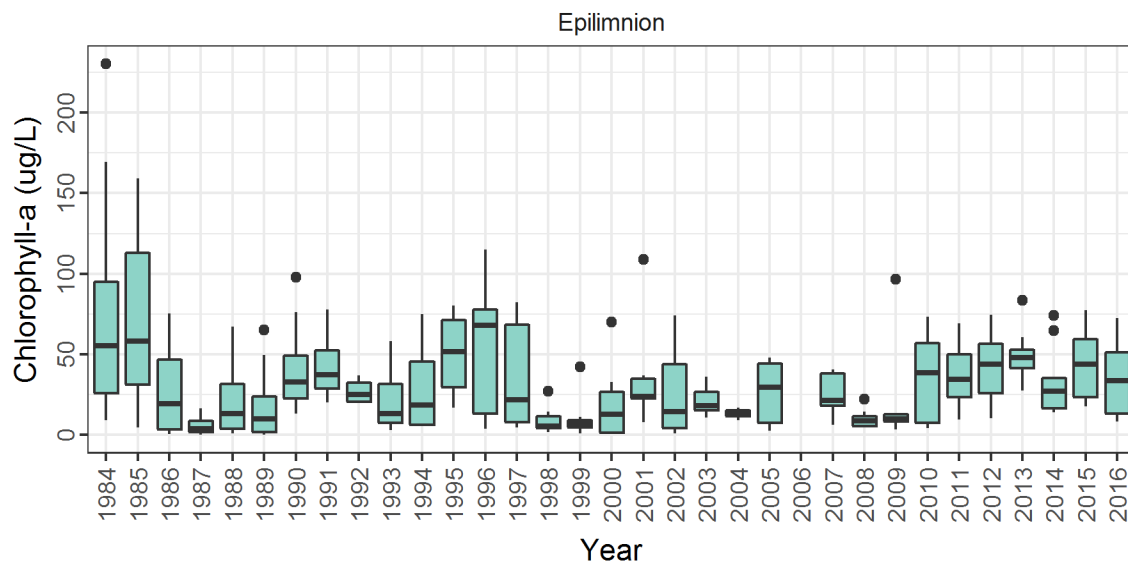
Figure 14. Inter-annual variability in epilimnetic Chl-a during the growing season (May – Sept) at Station 201.

Secchi disk depth is a measure of water transparency and thus provides an indicator of the turbidity of a lake, which can be caused by organic and inorganic particulate matter, as well as dissolved substances (e.g. dissolved organic carbon) in the water column. In Como Lake, Secchi depth and Chl-a were significantly correlated (see Section 4.2.5). Indeed, the deepest median Secchi depth measurements correspond to the years with the lowest and least variable Chl-a concentrations (Figure 14 and Figure 15). Over the past six years, the range of Secchi depth measurements during the growing season were the lowest in the period of record. The greatest median Secchi depth occurred in 1998 and 1999. No significant temporal trend was found with Secchi depth over the period of record (Table 4).
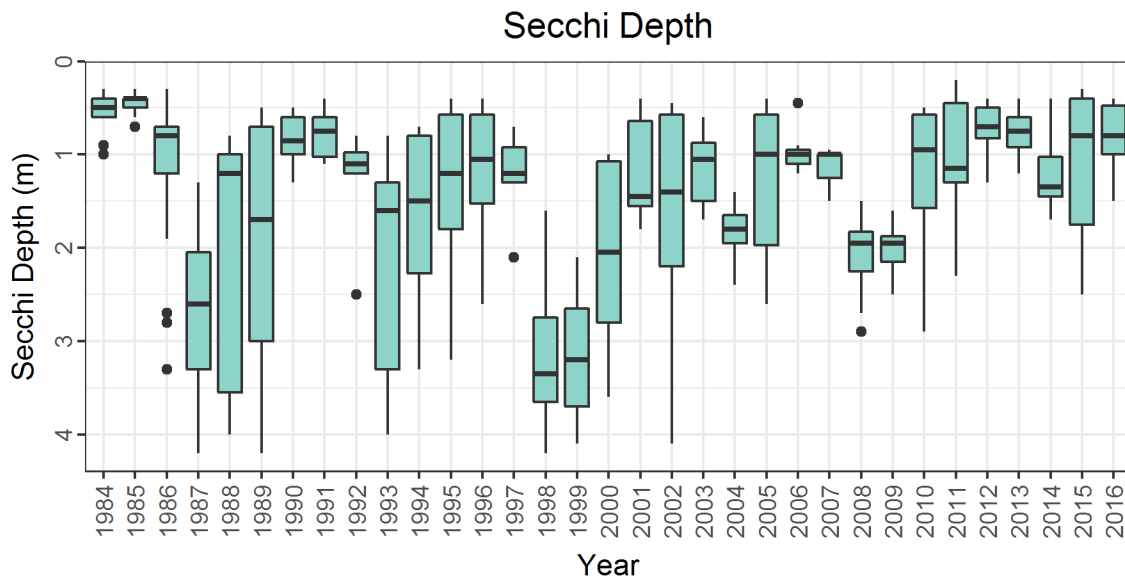
Figure 15. Inter-annual variability in Secchi depth during the growing season (May – Sept) at Station 201.

Epilimnetic turbidity also showed high inter-annual variability with a marked increase from 2010-2016 (Figure 16). For the growing season only (May – Sept), epilimnetic turbidity increased significantly through time (Table 4). As mentioned above, turbidity is a measure of water clarity and can be influenced by organic (e.g., algae) and inorganic materials (e.g., suspended solids). Comparison of the Chl-a and turbidity boxplots show somewhat similar patterns for the data record prior to 2010. Turbidity and Chl-a were significantly correlated, which simply means that turbidity is higher when Chl-a is higher and vice versa (see Section 4.2.5). However, Chl-a does not show the same sharp increase since 2010 as turbidity, which suggests an increase in non-algal turbidity (e.g. external loading of particulates) especially in 2015 and 2016. Total particulate matter and organic particulate matter have not been collected in Como Lake since 2005, which prevents further determination of the inorganic or organic nature of the turbidity values.
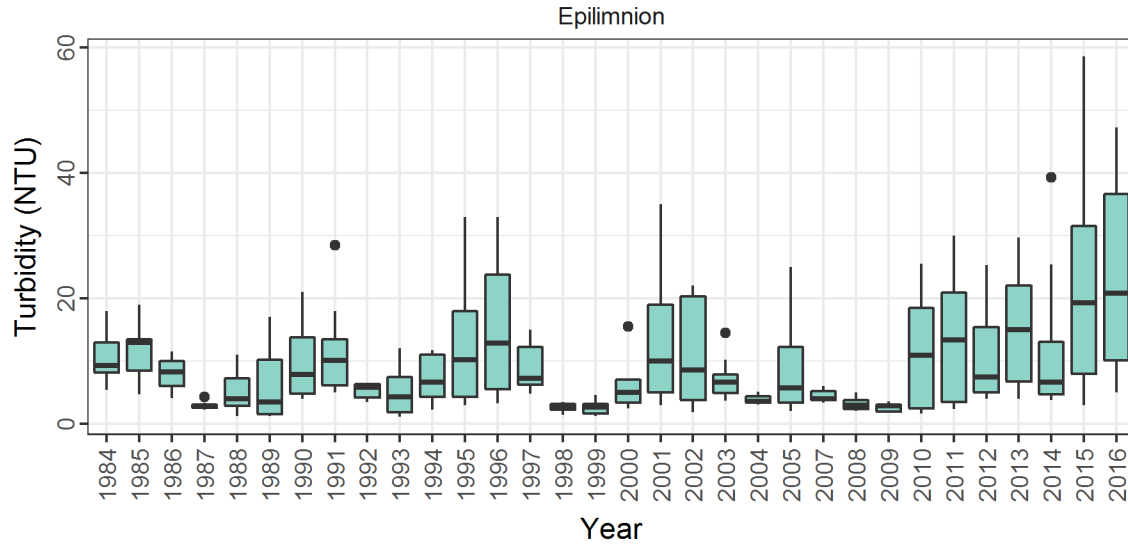
Figure 16. Inter-annual variability in turbidity during the growing season (May – Sept) at Station 201.

4.2.3 Intra-annual Trends

The majority of the data for Como Lake has been collected during the summer growing season with a few samples collected during April or October. The lack of data for non-growing season months prevent robust comparisons among seasons. Therefore, we focused our inter-annual data assessment on observations collected during the growing season (May – Sept) to determine within season variability in water quality variables (Figures 17 through 22).

Nutrients (TP, SRP, TN), Chl-a, and Secchi depth were aggregated by month and strata using the entire period of record (1984-2016) from Station 201, to evaluate trends during the growing season (May through Sept).

In both strata, median TP is highest in August, but the greatest variability and highest observed concentrations have occurred in July (Figure 17). Epilimnetic SRP exhibits a similar pattern to TP in that the highest median concentration occurs in July and August with the highest observed concentrations in July. Median and peak hypolimnetic SRP is typically greatest in August (Figure 18), which also corresponds to the lowest observed hypolimnetic DO concentrations (Figure 19). Along with the significant correlation between DO and SRP, this provides additional support of redox driven sediment P flux (see Chapter 6).
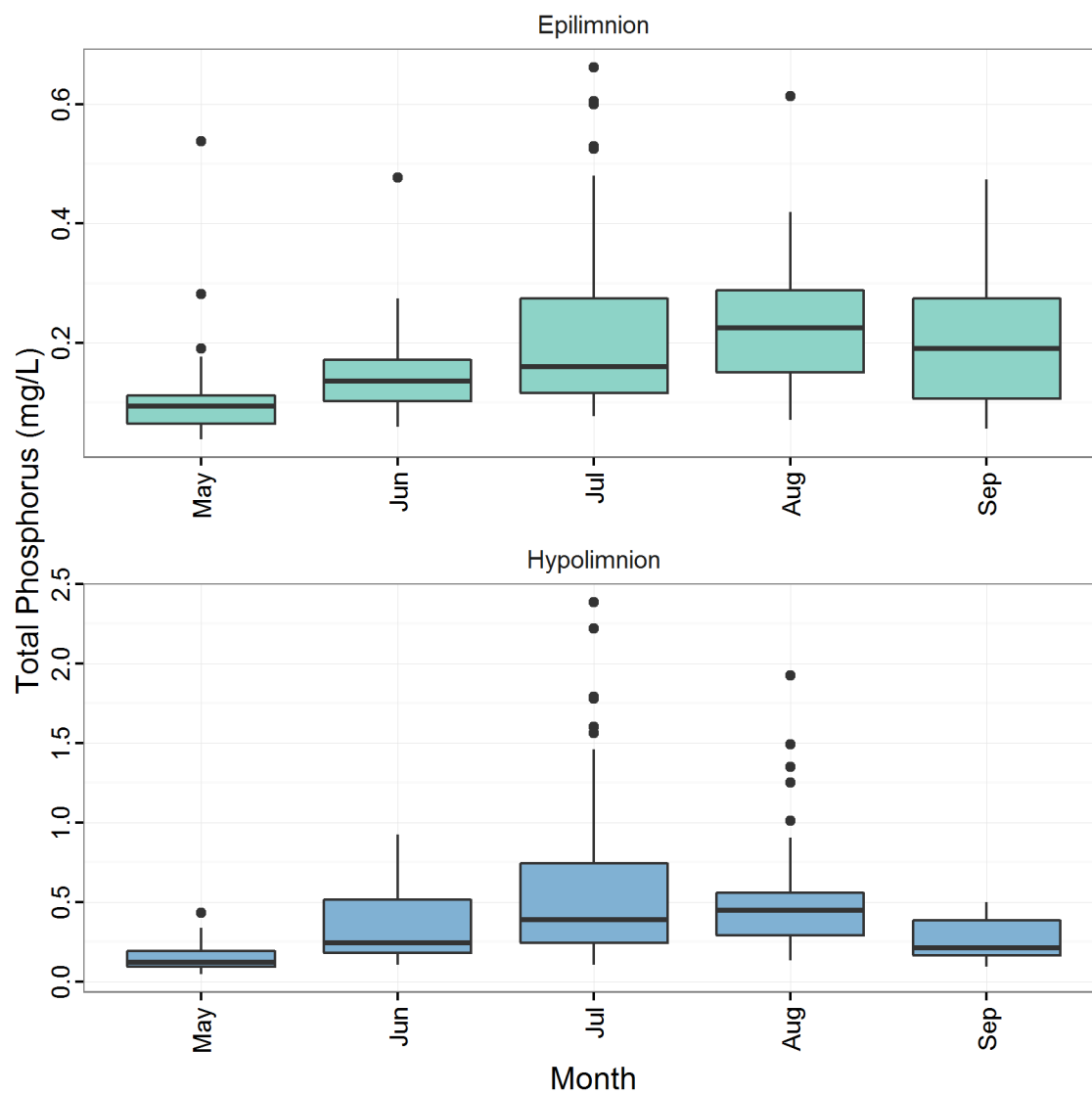
Figure 17. Range of epilimnetic and hypolimnetic total phosphorus concentration throughout the summer growing at Station 201 (1984-2016).
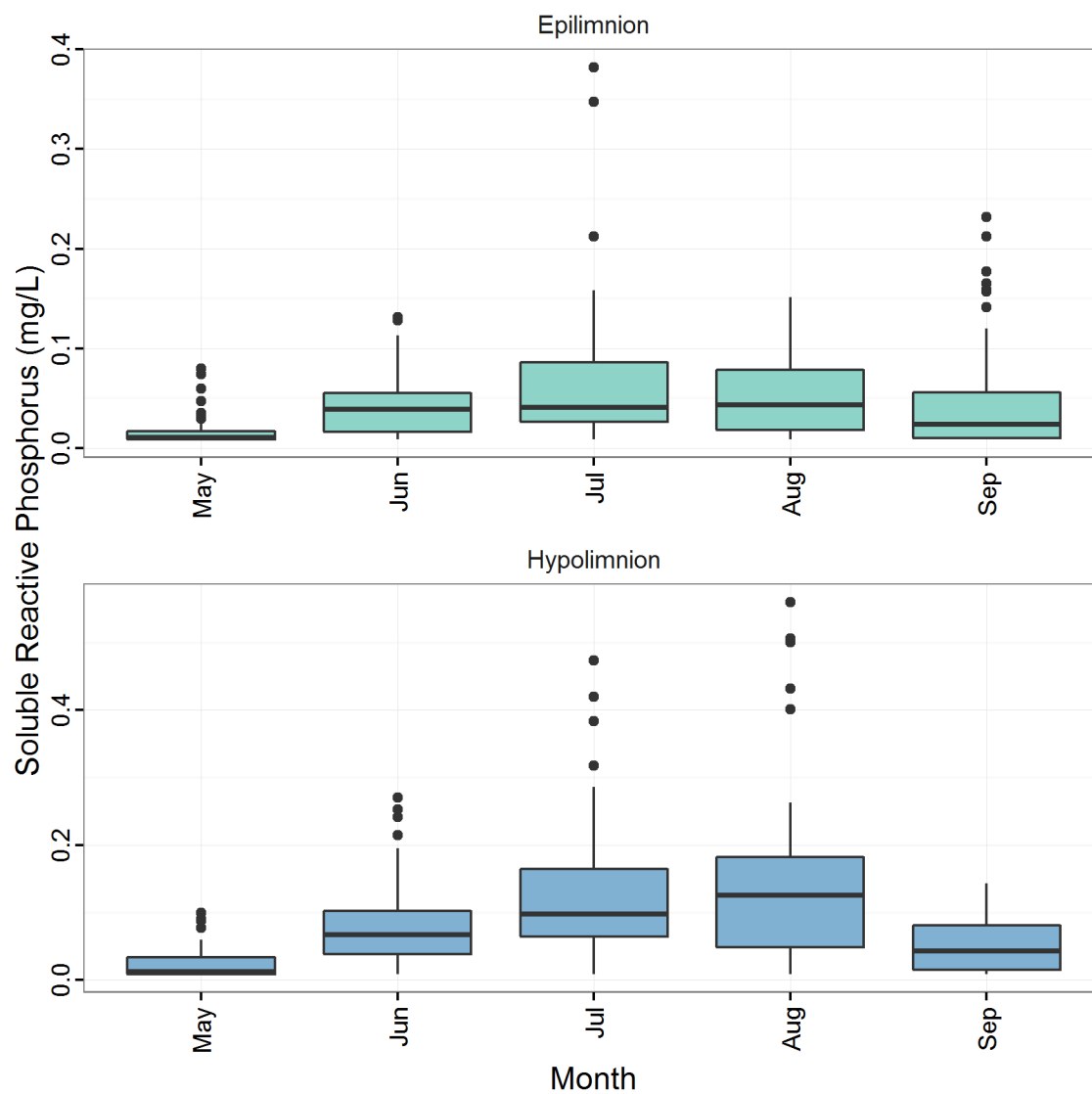
Figure 18. Range of epilimnetic and hypolimnetic soluble reactive phosphorus concentration throughout the summer growing at Station 201 (1984-2016).
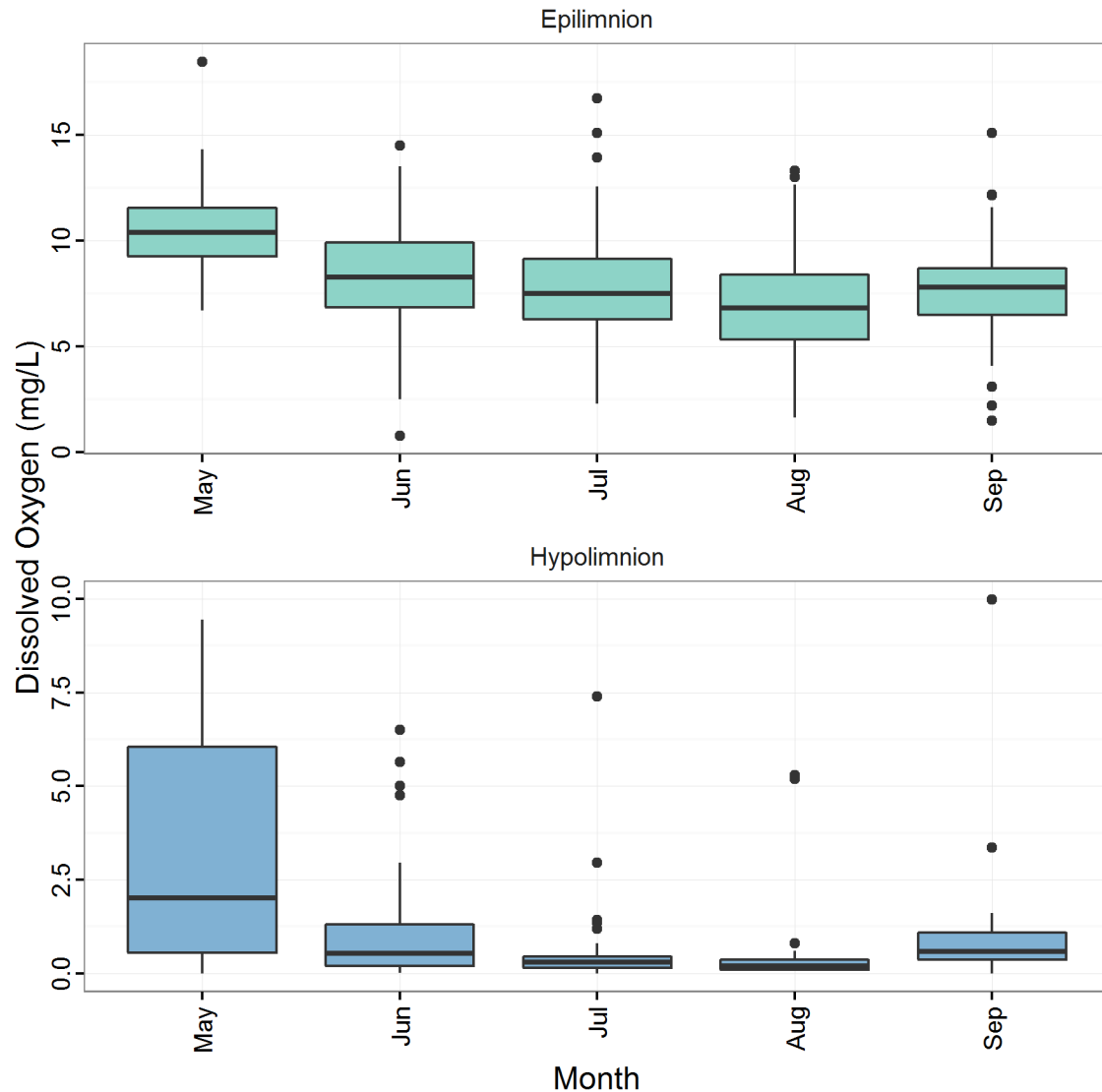
Figure 19. Range of epilimnetic and hypolimnetic dissolved oxygen throughout the summer growing at Station 201 (1984-2016).

TN tends to increase throughout the summer growing season in both strata (Figure 20). As discussed above, TN in Como Lake is comprised of 99% TKN, which is a measure of organic N and NH_4. Given the bulk of NH_4 measurements are at the limit of detection, we can conclude that a large portion of the TKN, and thus TN, is comprised of organic N, which is tied up in cellular material, which is tied up in algal or bacterial cells and is not always readily available in a form that can be taken up by phytoplankton. Thus, the increase in TN across the summer is indicative of the organic N in algal community biomass.
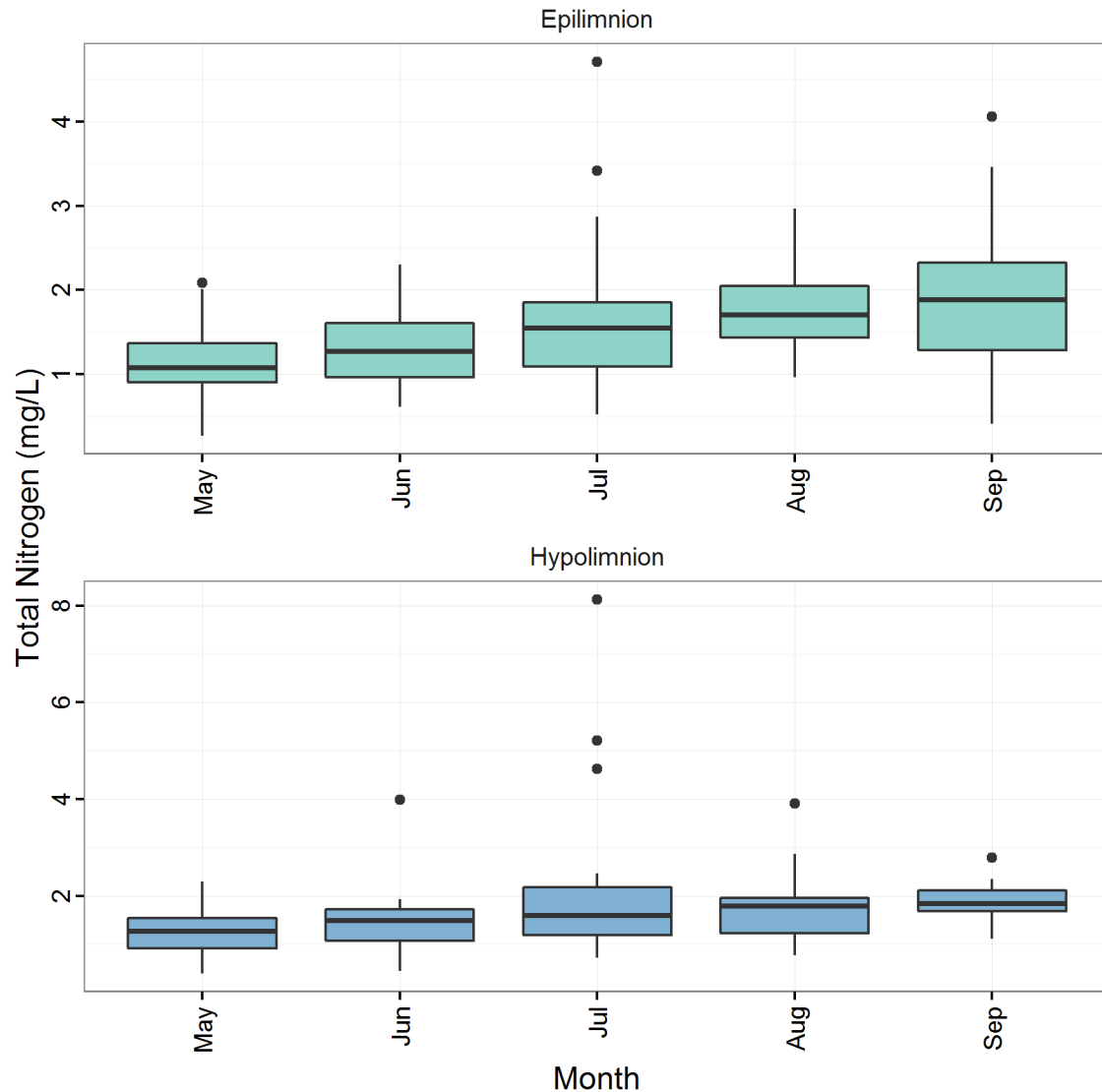
Figure 20. Range of epilimnetic and hypolimnetic total nitrogen concentration throughout the summer growing at Station 201 (1984-2016). Note that calculated TN values are conservatively estimated due to the presence of values reported at the limit of detection for nitrogen constituents (i.e. ammonia and nitrate).

Chl-a tends to increase throughout the summer months (Figure 21), with the highest median Chl-a concentration typically occurring in August. Chl-a begins to decrease in September which is a function of reduced algal growth due to decreasing temperature and light availability.
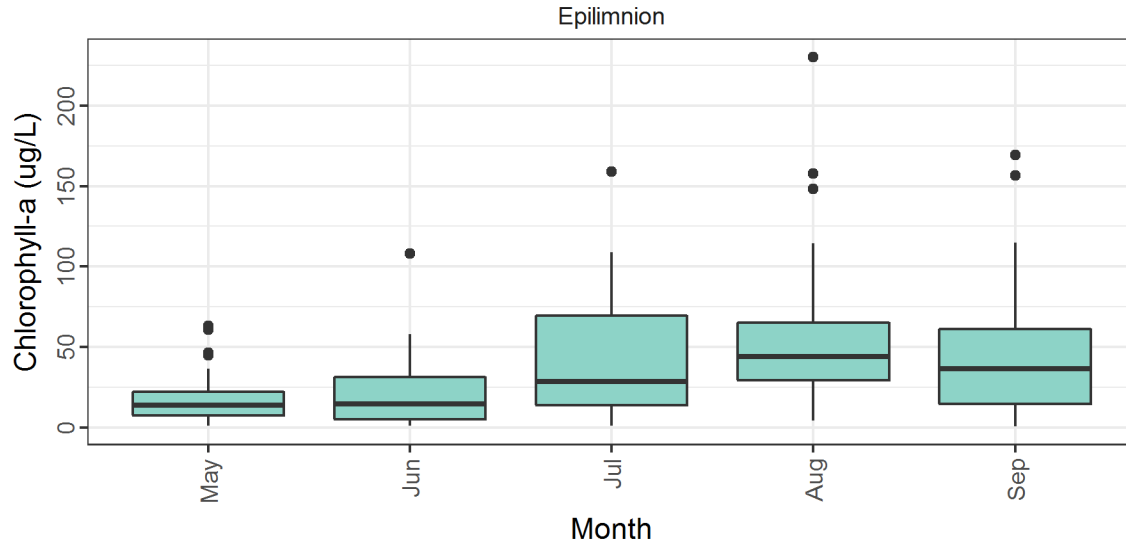
Figure 21. Range of epilimnetic Chl-a concentration throughout the summer growing at Station 201 (1984-2016).

Expectedly, median Secchi depth tends to be lowest in the late summer when median Chl-a is typically higher.
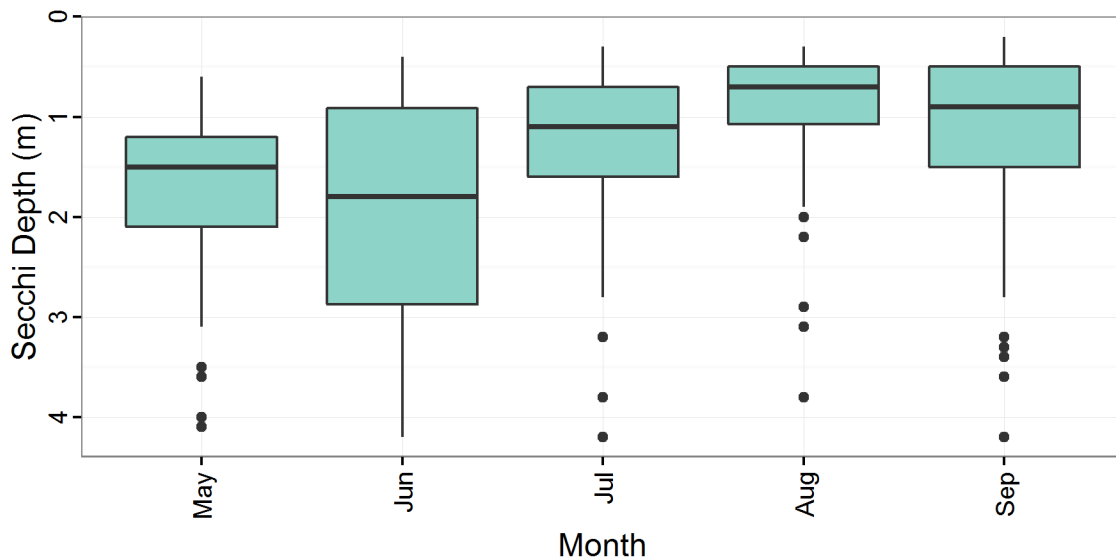
Figure 22. Range of Secchi depth measurements throughout the summer growing at Station 201 (1984-2016).

4.2.4 **Within-lake Spatial Variability**

The longest period of record for data in Como Lake exists at Station 201 (5401), which spans the years 1984-2016. Data also exist for Station 102 (5404) and Station 202 (5402) for the years 1984-1993, and 1985, 1987-1988 respectively. Acknowledging that the older data may not be representative of current conditions, we compared the spatial variability in epilimnetic TP, SRP, Chl-a, and Secchi depth for the years where data overlap among stations (Table 5) to evaluate spatial variability within Como Lake. Note that none of the differences discussed below were statistically significant when the Kruskal-Wallis test was applied (Table 5), but that may be due in part to low sample size at Station 202 in particular.

Table 5. Results of the Kruskal-Wallis rank sum test to determine spatial variability in TP, SRP, Chl-a or Secchi depth among stations during the growing season (May – Sept).

Parameter	Kruskal-Wallis test statistic (K)	p-value (α = 0.05)
TP (mg/l)	0.39	0.82
SRP (mg/l)	0.45	0.80
Chl-a (ug/l)	0.11	0.95
Secchi depth (m)	1.18	0.55

4.2.5 **Correlations among Environmental Parameters**

Relationships among environmental variables were evaluated using Spearman's rank correlation tests (Table 6). Correlation tests were based on whole-lake values (i.e. all depths, all stations) for the growing season only (May – Sept). As mentioned above, the primary objective of the current study centers on understanding drivers of water quality and potential mechanisms of internal loading of P in Como Lake. Therefore, in this discussion, we focus on parameters that are significantly correlated to eutrophication related parameters. Trends in other non-eutrophication related variables are shown in Appendix B.

TP was strongly positively correlated with Chl-a, TKN, NH_4 SRP, turbidity, and weakly correlated with water temperature. TP was negatively correlated with DO, conductivity, chloride and Secchi depth. Given that Chl-a provides an estimate of the biological response to nutrient availability, a significant correlation among those variables is expected. The strong positive correlation between TP and turbidity (Table 6) suggests that a significant proportion of turbidity in Como Lake is caused by algal biomass, which is supported by the fact that Chl-a and turbidity, and Chl-a and Secchi depth, were also significantly correlated. A potential deviation to that relationship includes the period of 2010-2016 where something else appears to be driving increased turbidity (see Section 5.2.2). The negative relationships between DO and nutrients point to complex redox conditions that could include sediment P release under anoxic periods, and/or decomposition of organic matter in the water column and sediments.

Table 6. Statistical correlations (Spearman's rank correlation) between environmental variables representing all stations, all depths during the growing season (May – Sept) for the 1984-2016 period of record. Bold italic numbers indicate significant correlations (α = 0.05).

Parameter	TP (mg/l)	Chl-a (mg/l)	TKN (mg/l)	NH$_4$-N (mg/l)	SRP (mg/l)	DO (mg/l)	Water Temp(°C)	Turbidity (NTU)	Conductivity (µS/cm)	Chloride (mg/l)	Secchi depth (m)	Precip (in)	Water level (ft)
TP (mg/l)	1.00	0.61	0.68	0.44	0.66	-0.59	0.20	0.70	-0.21	-0.33	-0.48	-0.10	0.31
Chl-a (mg/l)	0.61	1.00	0.65	0.02	0.09	-0.13	0.15	0.81	-0.30	-0.28	-0.86	0.03	0.44
TKN (mg/l)	0.68	0.65	1.00	0.29	0.36	-0.28	0.12	0.72	-0.25	-0.30	-0.60	-0.03	0.36
NH$_4$-N (mg/l)	0.44	0.02	0.29	1.00	0.35	-0.61	-0.10	0.12	-0.20	-0.18	-0.01	-0.07	0.24
SRP (mg/l)	0.66	0.09	0.36	0.35	1.00	-0.43	0.33	0.32	-0.11	-0.24	-0.07	-0.06	0.18
DO (mg/l)	-0.59	-0.13	-0.28	-0.61	-0.43	1.00	0.04	-0.22	0.06	0.17	0.07	0.03	-0.09
Water Temp (°C)	0.20	0.15	0.12	-0.10	0.33	0.04	1.00	0.19	-0.28	-0.33	-0.13	-0.10	0.00
Turbidity (NTU)	0.70	0.81	0.72	0.12	0.32	-0.22	0.19	1.00	-0.24	-0.32	-0.78	-0.12	0.40
Conductivity (µS/cm)	-0.21	-0.30	-0.25	-0.20	-0.11	0.06	-0.28	-0.24	1.00	0.94	0.28	0.13	-0.38
Chloride (mg/l)	-0.33	-0.28	-0.30	-0.18	-0.24	0.17	-0.33	-0.32	0.94	1.00	0.29	0.14	-0.33
Secchi depth (m)	-0.48	-0.86	-0.60	-0.01	-0.07	0.07	-0.13	-0.78	0.28	0.29	1.00	0.14	-0.55
Cumulative daily precip (in)	-0.10	0.03	-0.03	-0.07	-0.06	0.03	-0.10	-0.12	0.13	0.14	0.14	1.00	-0.44
Mean daily water level (ft)	0.31	0.44	0.36	0.24	0.18	-0.09	0.00	0.40	-0.38	-0.33	-0.55	-0.44	1.00

4.3 Periods of Nitrogen Limitation

LimnoTech evaluated the data for potential conditions where Como Lake might exhibit N limitation. Typically, a spring pulse of nutrients (N and P) from the watershed promotes early algal blooms which can also deplete the supply of N for continued phytoplankton growth. On a mass basis, ratios of TN:TP < 10:1 typically indicate N limitation to phytoplankton (Downing and McCauley 1992; Horne and Goldman 1994; Scheffer 2004; Lehman 2011). Figure 23 shows the range of TN:TP ratios for period of record at Station 201 for the growing season only (May-Sept). The median TN:TP ratio was less than 10 in June through Sept with fairly low variability in each month suggesting that N limitation may occur rather frequently during the growing season.

Under N limiting conditions, blue-green algal abundance can increase because some species are capable of fixing atmospheric N (Lehman 2011). Algal abundance is controlled by nutrient supply, grazing pressure, and temperature (Johnson and Luecke 2012). As discussed above, true TN values in the Como Lake dataset may be slightly lower due to the inclusion of NH_4 and NO_3 data that were reported at the limit of detection. Therefore, the TN:TP ratios represent upper bound estimates and could be slightly lower than what's presented here. Combined with high summer temperatures in Como Lake, it is likely that capable species of blue-green algae are fixing atmospheric N to sustain growth and competitively outcompete other algal groups.
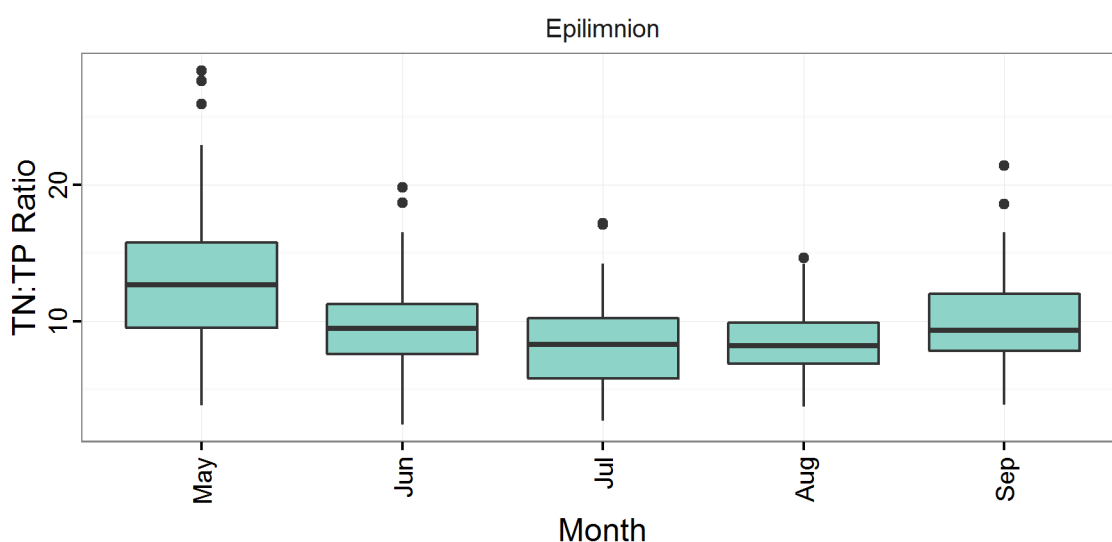
Figure 23. Range of epilimnetic monthly TN:TP ratios at Station 201 (1984-2016).

LimnoTech compared phytoplankton abundance to TN:TP ratios for 2014-2016 (Figure 23-Figure 26). Note that trends in phytoplankton abundance are discussed in more detail in Chapter 6. On 8/23/2016, the TN:TP mass ratio was less than 10 which was followed by an increase in the abundance of blue-green algae. The relationship between phytoplankton abundance and TN:TP ratios is more pronounced in 2014 where TN:TP was less than 10 for much of the summer growing season. As discussed above, TN:TP mass ratios less than 10 typically indicate N limitation. The increase in blue-green algae following a period of where TN:TP mass ratios were less than 10 is likely a function of their ability to fix atmospheric N under N limiting conditions.
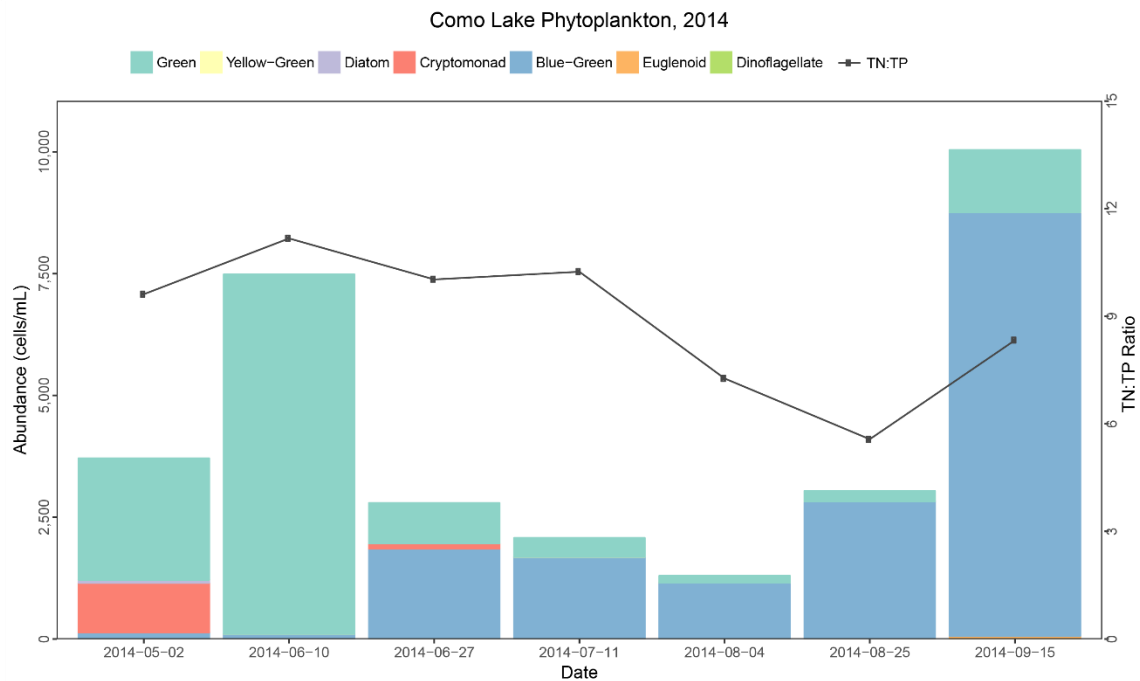
Figure 24. Phytoplankton abundance and epilimnetic total nitrogen to total phosphorus (TN:TP) ratios during the growing season (May – Sept) in 2014.
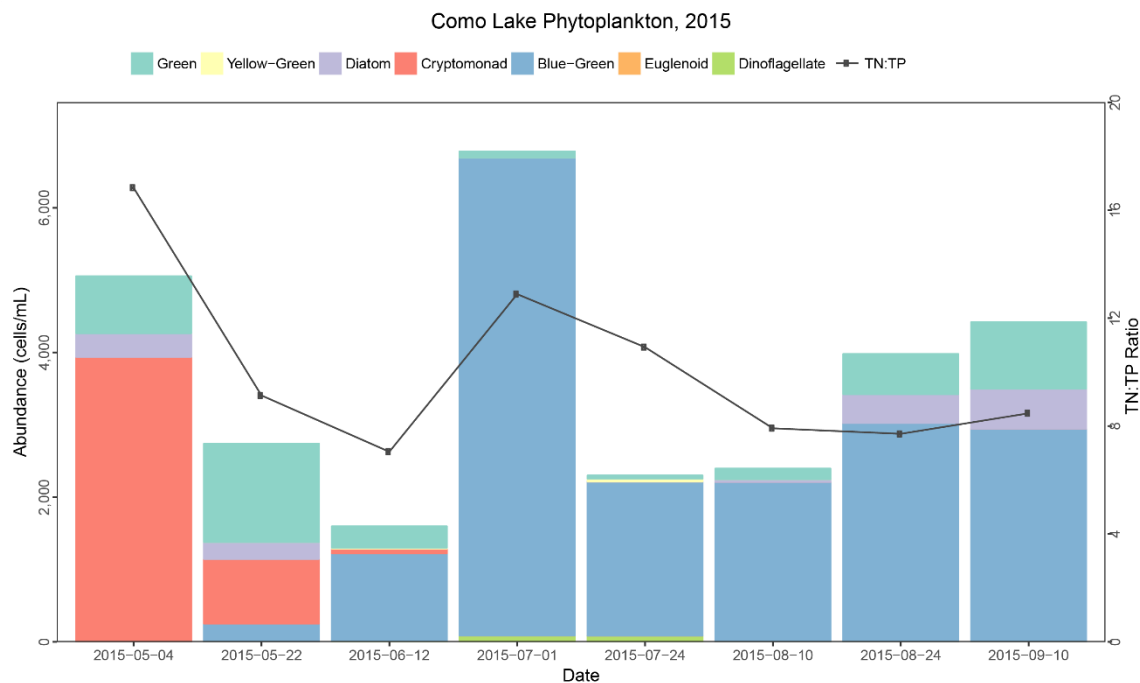
Figure 25. Phytoplankton abundance and epilimnetic total nitrogen to total phosphorus (TN:TP) ratios during the growing season (May – Sept) in 2015.
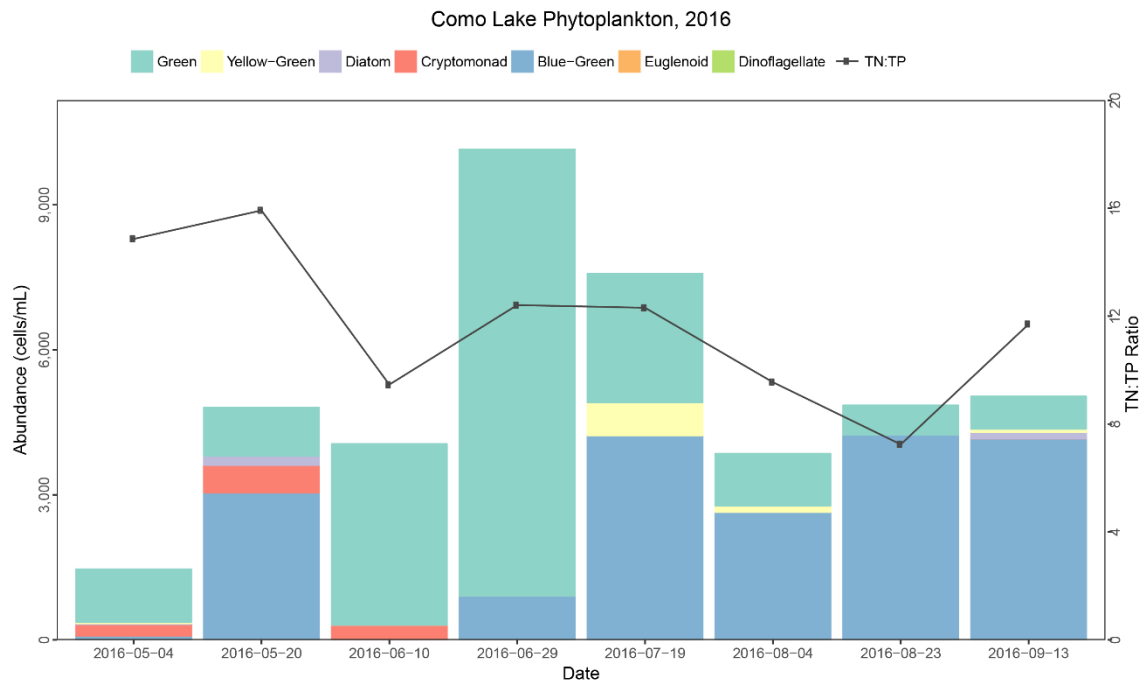
Figure 26. Phytoplankton abundance and epilimnetic total nitrogen to total phosphorus (TN:TP) ratios during the growing season (May – Sept) in 2016.

4.4 Conclusion

In summary, there is considerable inter-annual variability in the chemical data, which was also reported by Noonan (1998). Year to year variability is expected in a shallow lake as external drivers such as precipitation and air temperature also vary among years. The results of the data analysis were used to inform additional analyses including food web analysis (Chapter 5) and estimation of P loading from source categories (Chapter 6). Some key findings of the baseline data evaluation include the following observations:

- A strong oxycline often persists in the absence of a thermocline indicating high hypolimnetic and/or sediment oxygen demand, which emphasizes the need to base hypolimnetic sampling on the oxycline as well as the thermocline.

- The data suggest that the areal extent of hypoxic or anoxic conditions is potentially larger than what is represented by deep station profiles only.

- Nitrogen limiting conditions appear to occur in mid-late summer as evidenced by the TN:TP mass ratios < 10, which also tends to coincide with an increase in blue-green algae density. These algae are capable of capturing atmospheric nitrogen, which gives them a competitive edge over other algal groups when nitrogen is limiting.

- Despite the intermittent periods of apparent nitrogen limitation in summer, as a whole, nitrogen and phosphorus are in excess which is driving algal growth in Como Lake.

5
Imbalanced Aquatic Food Web Structure

As part of the overall goal to understand drivers of water quality in Como Lake, LimnoTech evaluated long-term trends in phytoplankton, zooplankton, fisheries and macrophyte data. The results of these analyses were used to develop a planning-level conceptual trophic food web model, which was designed to:

- Assess current aquatic community structure and functional feeding groups through available phytoplankton, zooplankton and fisheries data;

- Qualitatively describe food web interactions to highlight key drivers of community structure; and

- Develop a planning-level conceptual model to describe how the existing trophic cascade effects the lake to better understand operating food web dynamics.

The results of these analyses are discussed in the following sections.

5.1 Long-term Trends in Biological Data

Phytoplankton and zooplankton cell counts were grouped by major taxa and evaluated on an inter-annual and intra-annual basis for the summer growing season only (May – Sept). Total abundance and relative percent abundance of the major phytoplankton and zooplankton taxa during the growing season are discussed in the following sections. Total growing season abundance of each phytoplankton and zooplankton major taxonomic group for each year are listed in Appendix C. It is our understanding that CRWD has increased phytoplankton and zooplankton monitoring in the last 3-4 years, so LimnoTech focused on the biological data for the years 2014-2016 to evaluate intra-annual patterns.

Available fish abundance data was generally grouped by species and evaluated on annual basis. Surveys were conducted by MNDNR every five years, and by Wenck Associates annually from 2014-2016. The total abundance and relative percent abundance was calculated for each year of available data.

Macrophyte surveys have been conducted in Como Lake since 2005 and are reported as a percent occurrence basis (i.e. number of times a plant species was observed divided by the total number of sampling sites with observed vegetation). Since 2013, multiple surveys per summer were conducted where in years past, only a single survey was conducted. Therefore, we focused our assessment on years with multiple surveys thereby allowing for temporal comparison.

5.1.1 Phytoplankton

The major taxonomic groups (and common names for each) found in Como Lake are listed below:

- Diatoms (*Bacillariophyceae*)
- Green-algae (*Chlorophyta*)
- Blue-green algae (*Cyanophyta*)

- Yellow-green algae (*Chrysophyta*)
- Dinoflagellates (*Pyrrhophyta*)
- Cryptomonads (*Cryptophyta*)
- Euglenoids (*Euglenophyta*)

LimnoTech summed phytoplankton abundance data that was collected approximately bi-weekly during the summer growing season (May – Sept). Therefore, total abundance represents growing season conditions. We evaluated the total abundance of available phytoplankton taxa (Figure 27) and the percent relative abundance of each group (Figure 28) for the growing season over the period record (1984-2016). Total abundance of phytoplankton per growing season was higher in the mid-1980s with a fair amount of variability among years later in the data record. High phytoplankton density in the early 1980s was also noted by Noonan (1998) prior to fisheries manipulation and copper sulfate treatment to reduce algal biomass.

In all years, blue-green algal cells appear to be most dominant taxa present with a few exceptions where green algae and the cryptomonads were more abundant than blue-green algae (Figure 28). Dominance by blue-green algae is typical of eutrophic lakes due in part to many competitive advantages over other algal species. These advantages include the ability to fix atmospheric N when dissolved inorganic N is low (see Section 4.3), shade tolerance enabling them to proliferate at high turbidity (i.e. self-shading by blue-green algae density), and the ability to form dense colonies or produce toxins making them more resistant to predation pressure (Wetzel 2001; Scheffer 2004).
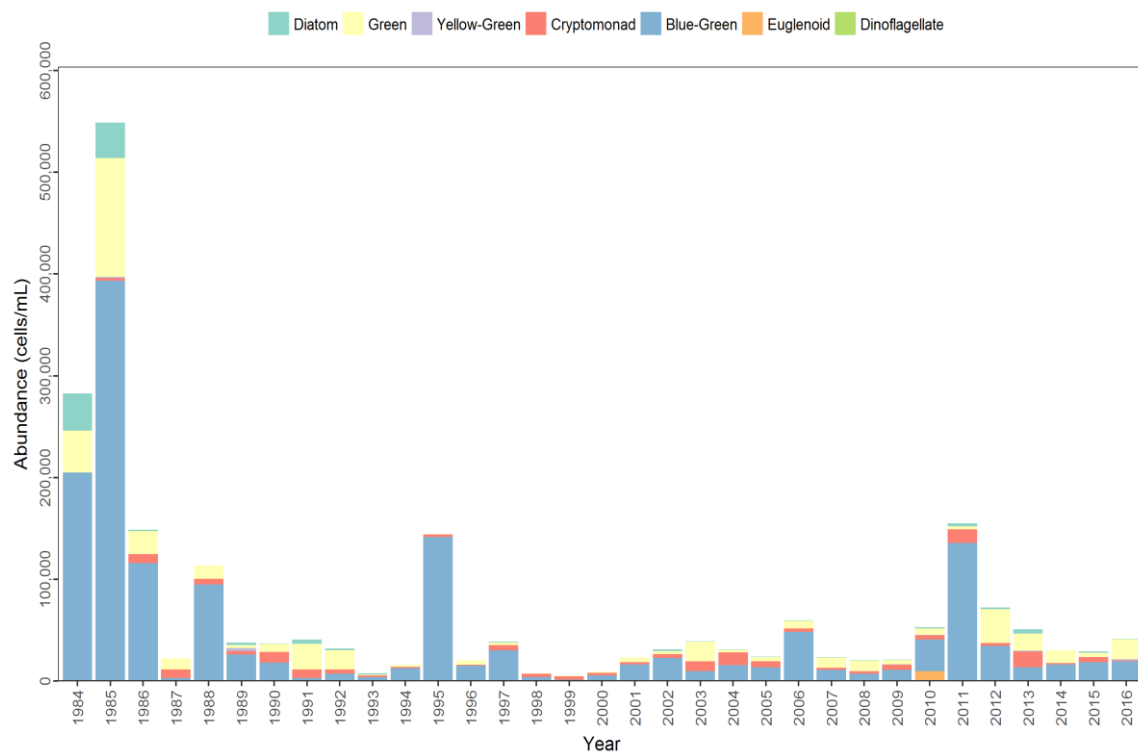
Figure 27. Total growing season (May – Sept) abundance of phytoplankton groups (1984-2016).

Figure 28. Percent relative abundance of phytoplankton groups during the growing season (May – Sept; 1984-2016).

Phytoplankton groups typically show predictable seasonal successional patterns which is driven by nutrient availability, light availability, temperature, degree of thermal stratification, species competition, and predation by zooplankton (Wetzel 2001; Arhonditsis et al. 2004; Scheffer 2004). In temperate regions, often a spring bloom of dinoflagellates and diatoms occurs before or during ice off, followed by increasing density of green algae and yellow-green algae. In mid to late-summer, blue-green algae tend to peak along with or followed by another bloom of diatoms and dinoflagellates (Horne and Goldman 1994; Wetzel 2001). These patterns are often more evident when assessing phytoplankton biomass compared to abundance. In Como Lake, early diatom blooms are evident, the most abundant early season taxa for 2014-2016 include green algae, crytomonads and euglenoids (Figure 29-Figure 31).

Phytoplankton taxa have different nutrient requirements and respond differentially to available limiting nutrients (Lee et al. 2015). Consequently, fluctuating nutrient concentrations are often coincident with bloom cycles of various taxa (Scheffer 2004). Figure 29 through Figure 31 show phytoplankton abundance with corresponding epilimnetic TP concentration for each sampling date in 2014-2016, respectively. In all three summers, TP tends to increase throughout the growing season, which doesn't necessarily show increased total abundance of phytoplankton. However, these figures do show that blue-green algal abundance is associated with increasing TP. Other notable patterns include a "clear-water phase" in late spring and late summer/early fall.

The "clear-water phase" is due to increased growth and predation by zooplankton in response to increased phytoplankton density (Horne and Goldman 1994; Scheffer 2004). During the clear-water phase, zooplankton grazing rates exceeds phytoplankton growth rates resulting in a brief phytoplankton community crash, which releases nutrients during decay of phytoplankton and zooplankton cells. The release of nutrients can foster growth by different phytoplankton taxa (Scheffer 2004). In all three years, blue-green algae are the most abundant taxa present by mid-late summer. Blue-green algae can quickly reproduce at higher temperature and some species are capable of fixing atmospheric N, which can be a competitive advantage over other algal species if N becomes limiting (Horne and Goldman 1994; Scheffer 2004). Conditions under which N may be a limiting factor in Como Lake is discussed in more detail below.
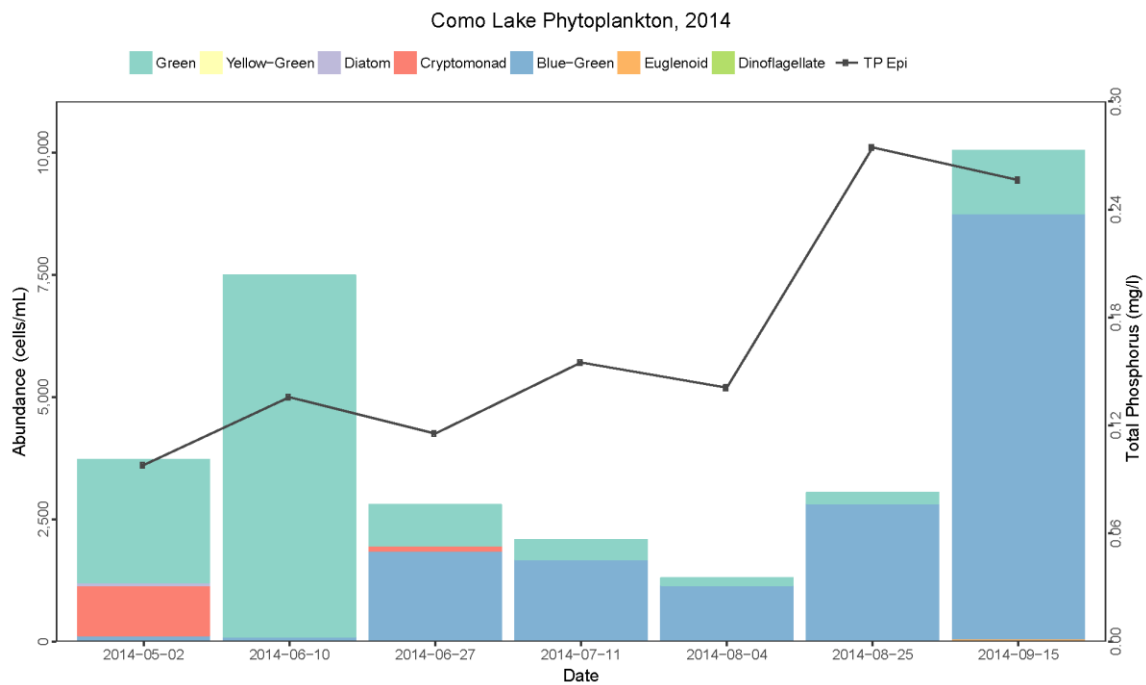
Figure 29. Phytoplankton abundance and epilimnetic total phosphorus in 2014.
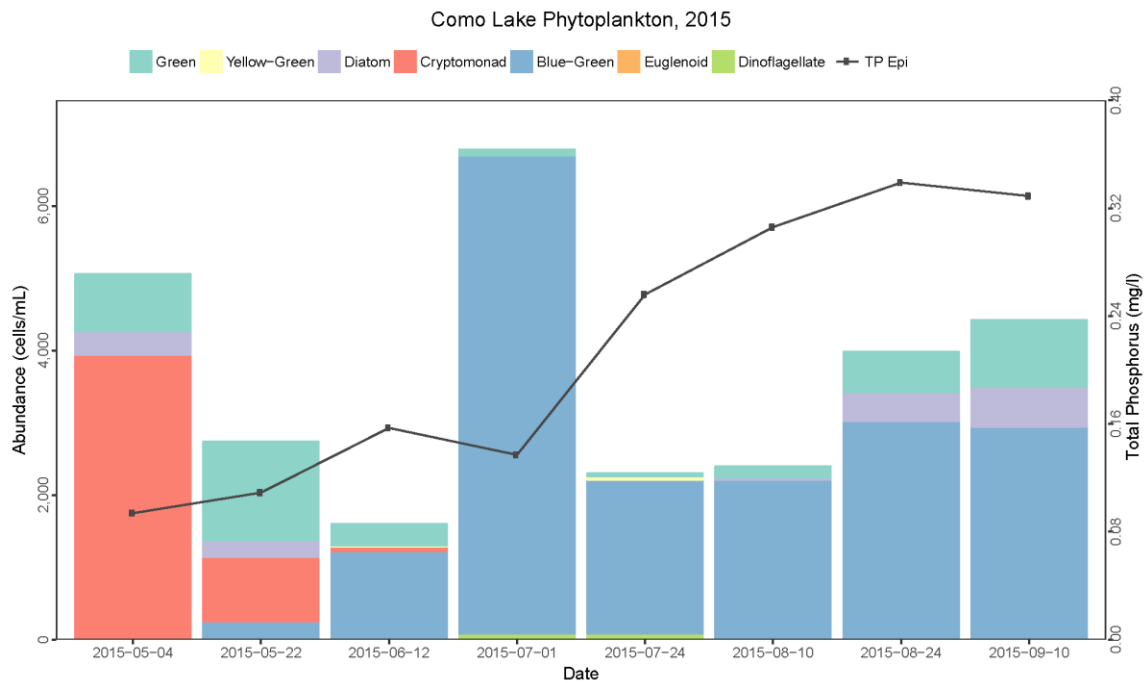
Figure 30.Phytoplankton abundance and epilimnetic total phosphorus in 2015.
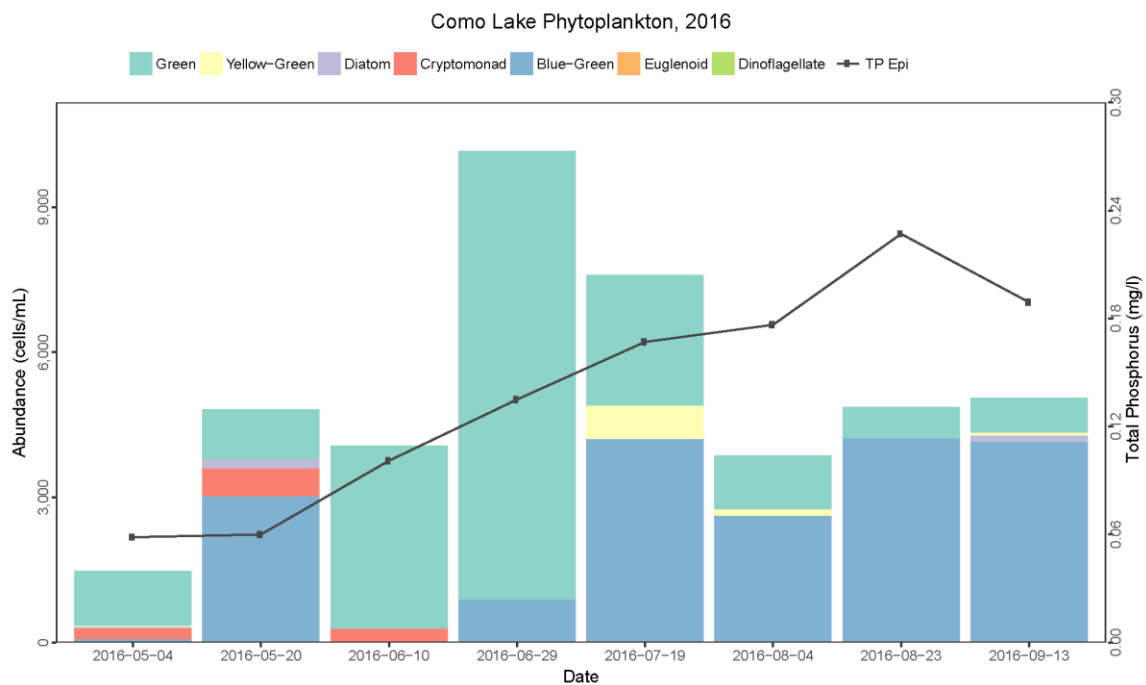
Figure 31. Phytoplankton abundance and epilimnetic total phosphorus in 2016 (May – Sept).

5.1.2 **Zooplankton**

The major zooplankton groups found in Como Lake are listed below:

- Calanoid copepods

- Cyclopoid copepods

- Nauplii

- Cladocerans

- Rotifers

LimnoTech evaluated the total growing season (May-Sept) abundance of major taxa (Figure 32) and the percent relative abundance of each group (Figure 33). Similar to phytoplankton abundance, the total density of zooplankton taxa was highest in the mid-1980s. Total annual zooplankton abundance shows a fair amount of inter-annual variability. Zooplankton are in the center of aquatic food chain and as such their density is regulated by the interactions of available food resources (i.e. phytoplankton or bacteria) and predation pressure (i.e. fish).
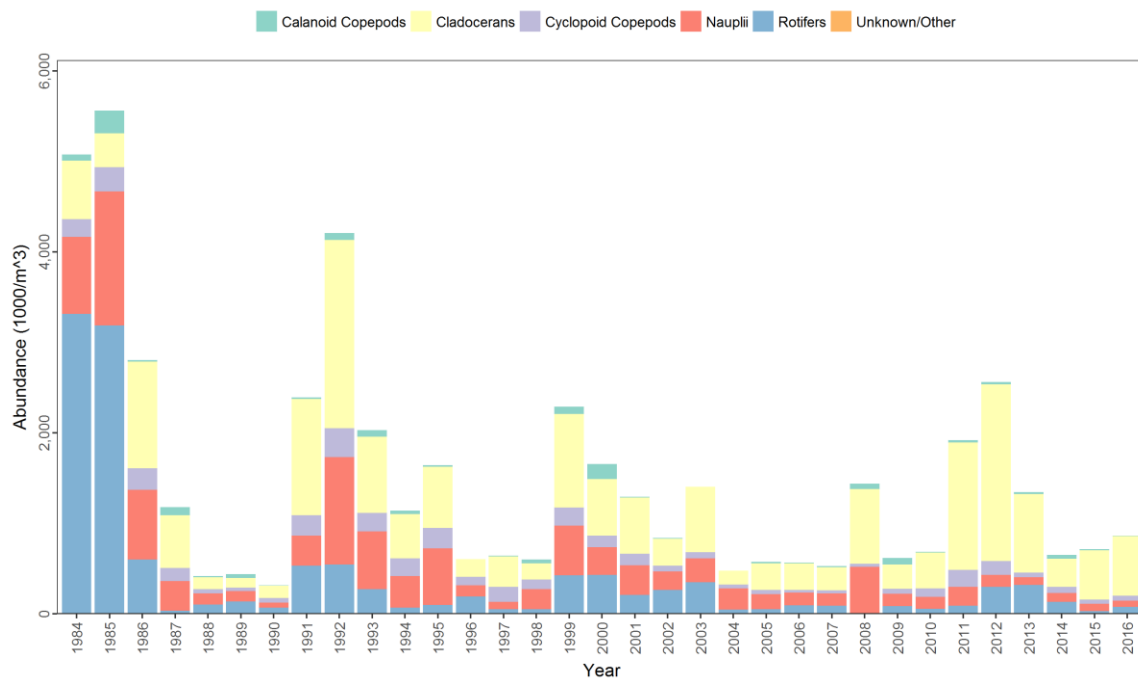
Figure 32. Total growing season (May – Sept) abundance of zooplankton groups (1984-2016).

Perhaps the most notable pattern in zooplankton data is the increasing density of cladocerans (Figure 33). In many lakes, the crustacean communities are typically dominated by copepod and cladoceran taxa. In more productive lakes, cyclopoid copepods tend to be more abundant than calanoid copepods, and cladoceran density is typically higher than calanoids (Hsieh et al. 2011). Cladocerans can reproduce quickly and have high demand for P relative to N (Johnson and Leucke 2012). Copepods contain more N in their cellular makeup compared to cladocerans. Conversely, cladocerans are richer in P compared to copepods (Sommer and Sommer 2006).

Zooplankton taxa recycle nutrients in lakes through consumption of bacteria and algae, excretion and decomposition. Consequently, zooplankton can be major sources of N and P (Horne and Goldman 1994). Given the high density of cladocerans in Como Lake, it is likely that zooplankton play an important role in the internal cycling of P.
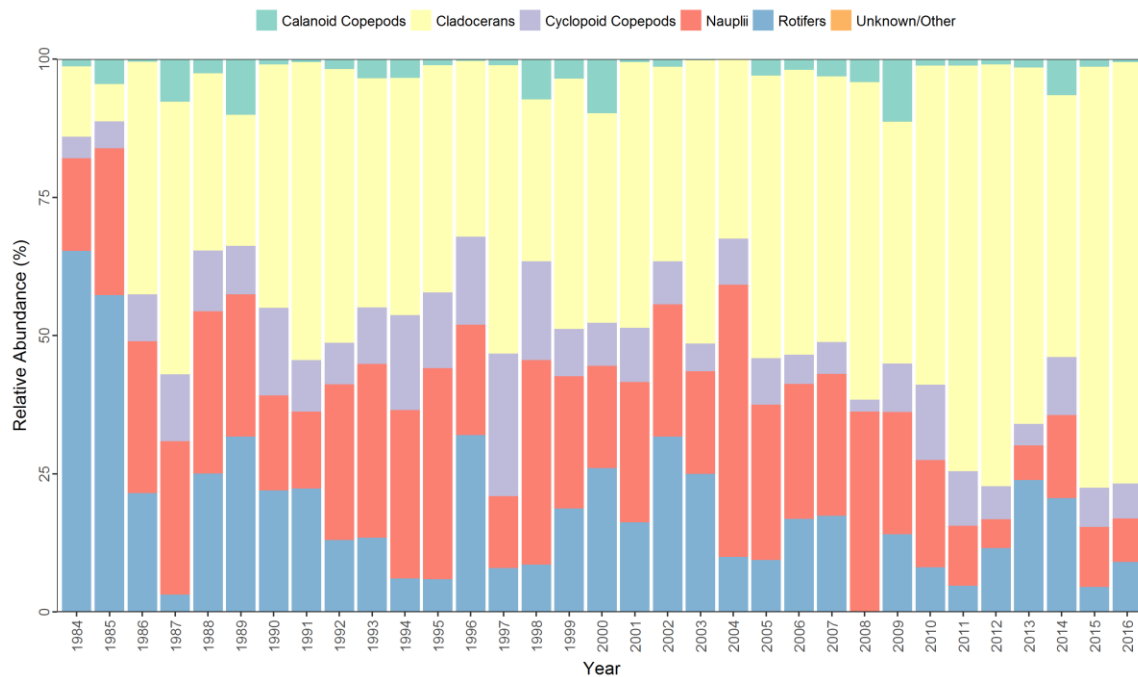
Figure 33. Percent relative abundance of zooplankton groups during the growing season (May – Sept; 1984-2016).

Figure 34 through Figure 36 show phytoplankton abundance with corresponding epilimnetic Chl-a concentration for each sampling date in 2016. As would be expected from ecological theory on predator-prey interactions, a periodic lag phase can be observed where zooplankton abundance increases following an increased in Chl-a (a measure of phytoplankton biomass) and vice versa. Consumption rates on phytoplankton would differ among zooplankton size classes and would therefore differentially influence Chl-a dynamics.
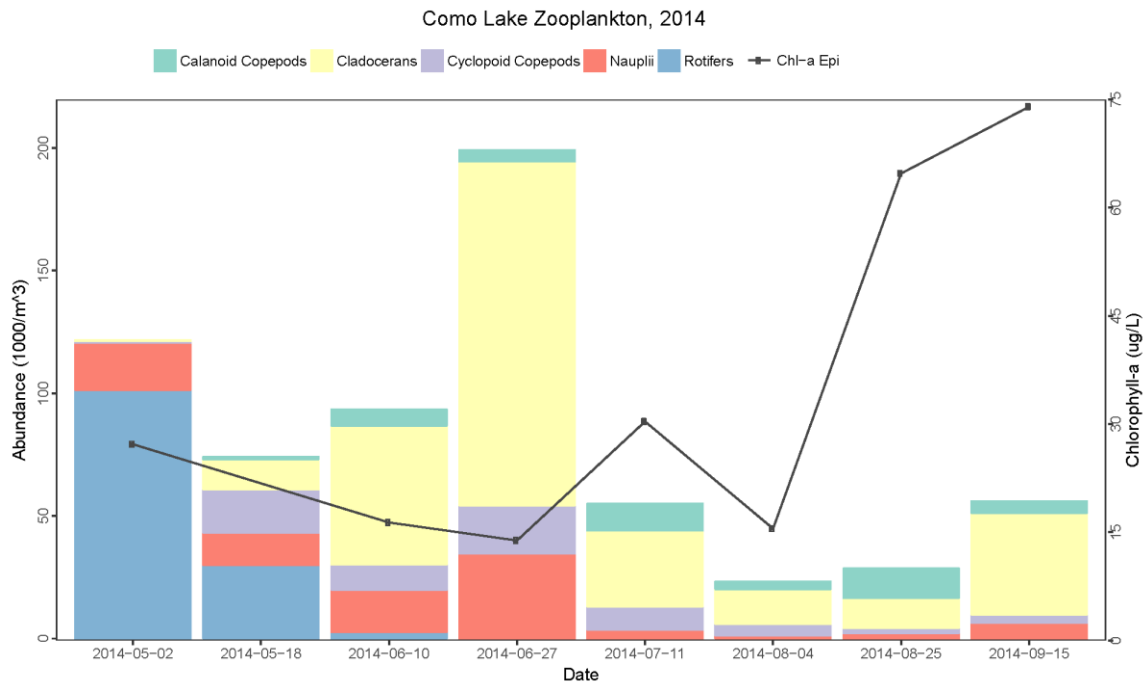
Figure 34. Zooplankton abundance and epilimnetic Chl-a in 2014.
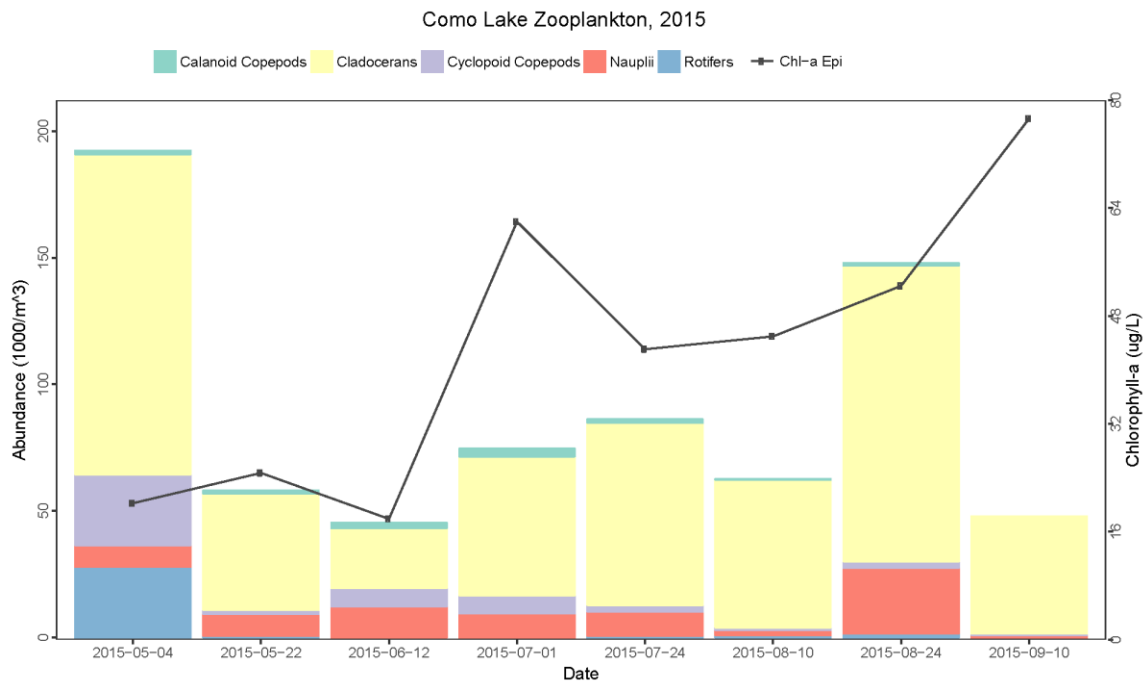
Figure 35. Zooplankton abundance and epilimnetic Chl-a in 2015.
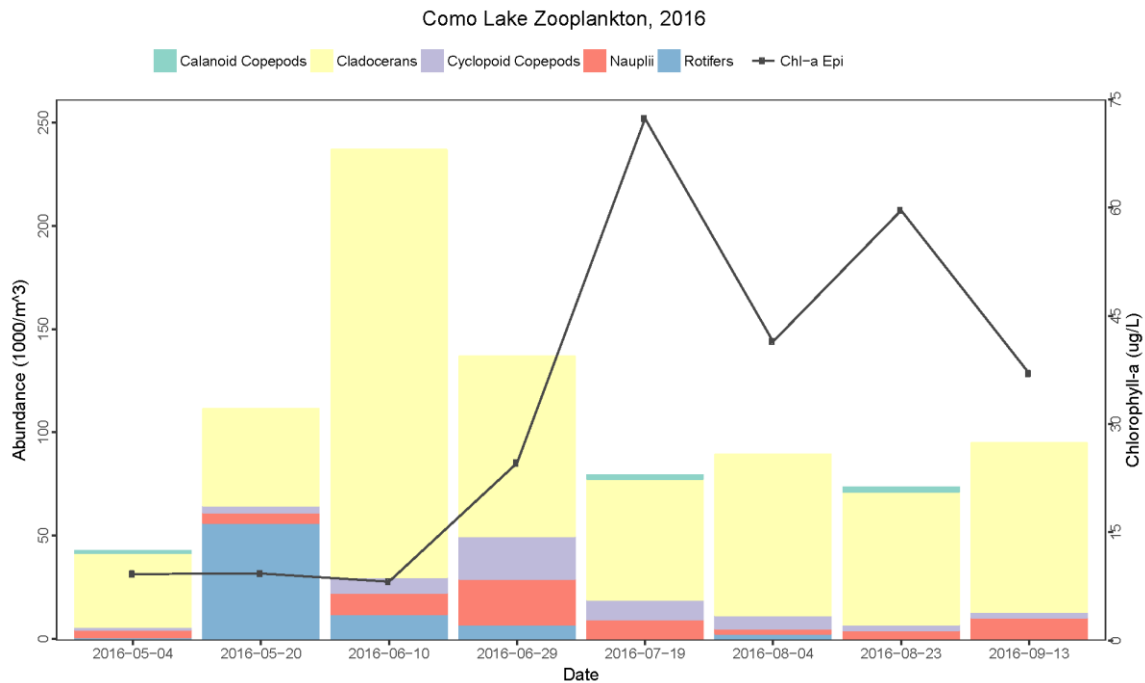
Figure 36. Zooplankton abundance and epilimnetic Chl-a in 2016.

5.1.3 Fisheries Data Trends

LimnoTech summarized available fish survey data for the years 2001-2016. Fish surveys were generally conducted in July or August. The number of individuals caught in the surveys, and relative proportion to total fish surveyed are shown in Table 7. The most abundant fish observed in recent surveys include black crappie, black bullhead and bluegills. Size class and fish stocking data were also provided with the fish survey data.

Table 7. Summary of annual fish community surveys in Como Lake. Most abundant taxa per survey shown in bold.

Species	2001	% catch	2006	% catch	2011	% catch	2014	% catch	2015	% catch	2016	% catch
Black Bullhead	71	6.3%	603	**52.9%**	71	9.6%	14	7.0%	190	38.0%	155	28.1%
Black Crappie	162	14.3%	47	4.1%	271	**36.6%**	121	**60.5%**	233	**46.6%**	258	**46.7%**
Bluegill Sunfish	4	0.4%	329	28.9%	236	31.8%	6	3.0%	2	0.4%	106	19.2%
Brown Bullhead	1	0.1%	0	0.0%	0	0.0%	0	0.0%	1	0.2%	0	0.0%
Channel Catfish	1	0.1%	0	0.0%	18	2.4%	3	1.5%	10	2.0%	3	0.5%
Common Carp	0	0.0%	0	0.0%	0	0.0%	0	0.0%	1	0.2%	0	0.0%
Golden Shiner	690	**60.7%**	12	1.1%	2	0.3%	18	9.0%	6	1.2%	4	0.7%
Green Sunfish	0	0.0%	6	0.5%	0	0.0%	0	0.0%	0	0.0%	0	0.0%
Hybrid Sunfish	1	0.1%	4	0.4%	9	1.2%	0	0.0%	2	0.4%	11	2.0%
Largemouth bass	0	0.0%	0	0.0%	1	0.1%	0	0.0%	0	0.0%	0	0.0%
Northern Pike	104	9.2%	65	5.7%	49	6.6%	15	7.5%	20	4.0%	7	1.3%
Pumpkinseed	0	0.0%	17	1.5%	29	3.9%	4	2.0%	0	0.0%	3	0.5%
Walleye	0	0.0%	20	1.8%	5	0.7%	6	3.0%	19	3.8%	2	0.4%
White Sucker	13	1.1%	1	0.1%	3	0.4%	0	0.0%	3	0.6%	0	0.0%
Yellow Bullhead	81	7.1%	31	2.7%	33	4.5%	0	0.0%	12	2.4%	0	0.0%
Yellow Perch	8	0.7%	5	0.4%	14	1.9%	13	6.5%	1	0.2%	3	0.5%
Total	**1136**	**100.0%**	**1140**	**100.0%**	**741**	**100.0%**	**200**	**100.0%**	**500**	**100.0%**	**552**	**100.0%**

The total annual abundance and relative percent abundance of fish groups are shown in Figure 37 and Figure 38, respectively. Note that fish groupings are more broadly categorized in Figure 37 and Figure 38 than what is reflected in Table 7. Total fish abundance was highest in 2001 and 2006 and lowest in 2014. Since 2006, the fish community was dominated by black crappie followed by black bullheads or bluegill sunfish.
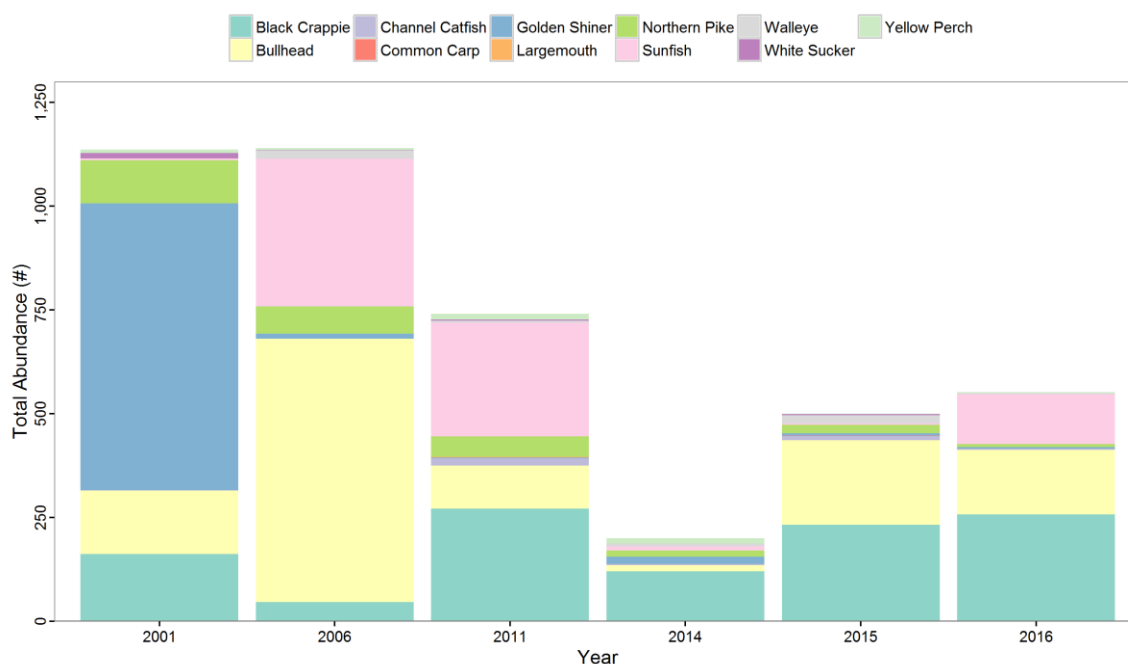
Figure 37. Total fish abundance among sampling years. Fish surveys were generally conducted in July or August.
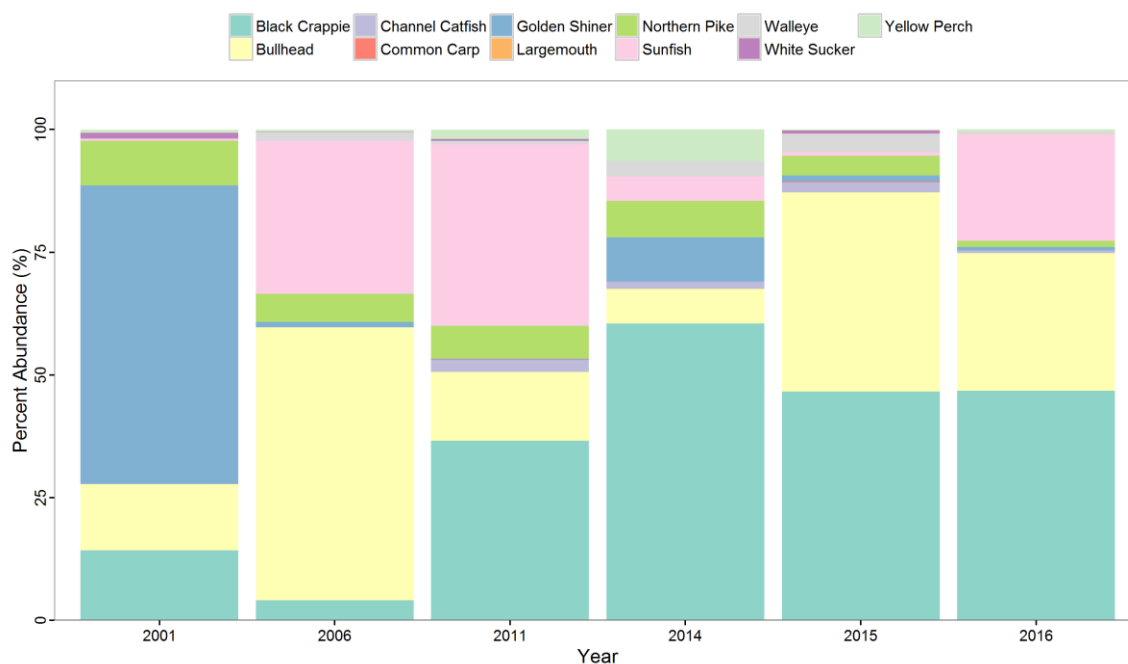
Figure 38. Relative percent abundance for fish among sampling years. Fish surveys were generally conducted in July or August

Como Lake is stocked by the MNDNR through their Fishing in the Neighborhood Program (MNDNR 2017b), which aims to increase angling opportunities in urban lakes. The species stocked by the MNDNR include bluegill, channel catfish, largemouth bass, walleye and yellow perch. The historical record for the number of each fish species stocked in Como Lake is shown in Table 8. There is a lack of survey information for the years 2006-2011, which also appears to be years for heavy fish stocking. The density of fish taxa in Como Lake is controlled by the interactive effects of fish stocking, fish predation, fishing pressure and water quality conditions in the lake. Given the data gaps, it is difficult to assess the impact of fish stocking on the fisheries in Como Lake and is beyond the scope of this project. One goal of this project is to evaluate the fish community for potential influences on P cycling through bioturbation and foodweb interactions (see Chapters 5 and 6, respectively).

Table 8. Historical record of the number of fish stocked in Como Lake.

Year	Bluegill	Channel catfish		Largemouth bass		Walleye			Yellow perch	
	Adult	Adult	Yearling	Adult	Yearling	Fry	Yearling	Fingerling	Adult	Yearling
2004	959	-	-	-	-	-	-	-	-	-
2005	-	-	-	-	-	-	-	-	-	-
2006	1,384	-	-	-	-	-	-	-	-	-
2007	284	150	3,864	179	414	-	-	-	-	-
2008	-	155	4,502	-	-	-	-	-	-	-
2009	-	-	3,457	-	-	-	-	-	816	1,200
2010	24	91	3,900	-	-	-	4	-	-	-
2011	-	124	3,900	-	-	-	-	3,593	-	-
2013	-	-	-	-	-	-	-	486	-	-
2014	-	-	130	8	-	71,000	-	4,108	1,211	-
2015	-	181	-	-	-	-	2,840	-	-	-
2016	-	-	-	-	-	-	903	-	-	-
Total	2,651	701	19,753	187	414	71,000	3,747	8,187	2,027	1,200

5.1.4 Macrophytes

Macrophyte surveys have been conducted in Como Lake since 2005. Since 2013, multiple surveys per summer were conducted where in years past, only a single survey was conducted. Therefore, we focused our assessment on years with multiple surveys thereby allowing for temporal comparison. Curlyleaf pondweed (CLP) has been present in all surveys. CLP is an aggressive growing invasive species, and has been documented to contribute to internal P loading (James et al. 2002; Heiskary and Valley 2012). Macrophytes, including CLP, are mechanically harvested in Como Lake for recreational purposes in targeted areas of the lake. Table 9 shows the percent of occurrence for macrophytes on survey dates from 2013-2016 along with dates of mechanical harvesting. The macrophyte community in Como Lake appears to be dominated by CLP, Canada waterweed, and in some years, coontail. Table 9 shows a clear reduction in the proportion of CLP present in Como Lake following mechanical harvesting, but CLP reoccurs each year due in part to rapid overwinter growth (Heiskary and Valley 2012). As mentioned above, CLP is known to release P upon senescence which tends to occur in mid-late June (James et al. 2002; Heiskary and Valley 2012). The data suggest that the decrease in CLP abundance may be due to combined effects of senescence and mechanical harvesting. More specific evaluation of the potential effects of CLP on internal P loading are discussed in the Chapter 7.

Table 9. Percent Occurrence of Macrophytes.

Common Name	Scientific Name	% Occurrence of Macrophytes										
CLP Mechanical Harvest Dates →		*6/27/13 - 6/29/13*		*07/03/14 - 7/9/14*			*07/14/15*			*6/21/2016 - 6/22/2016*		
	Survey Dates →	06/06/13	07/24/13	06/13/14	07/23/14	08/28/14	05/28/15	07/17/15	08/28/15	06/02/16	07/11/16	08/23/16
Canada waterweed	*Elodea canadensis*	65	100	67	74	92	50	92	93	0	81	88
Flatstem pondweed	*Potamogeton zosteriformis*	0	52	0	24	0						
Coontail	*Ceratophyllum demersum*	21	30	24	31	19	0	13	0			
Sago pondweed	*Potamogeton pectinatus*	0	4	67	0	0				2	0	
Lesser duckweed	*Lemna minor*	0	4	0	0	19						
Curly-leaf pondweed	*Potomogeton crispus*	85	0	88	17	0	94	21	14	95	63	23
Slenderleaf naiad	*Najas flexilis*	3	0							19	11	4
Muskgrass	*Chara sp.*			2	0	0						
Leafy pondweed	*Potomogeton foliosus*			0	0	8	6	13	0	2	7	
Greater Duckweed	*Spirodela polyriza*						0	4	0	0	4	
Wild Celery	*Valisneria americana*						0	4	0			

5.2 Characterization of Trophic Levels in Shallow Lakes

Food webs are used to illustrate the transfer of energy among trophic levels through the consumption of food resources. In aquatic ecosystems, algae (i.e., phytoplankton) and macrophytes obtain solar energy through photosynthesis (i.e., primary production). That energy is transferred from lower trophic levels (i.e., phytoplankton) to higher trophic levels (i.e., zooplankton and fish) through consumption. An example of a simplified food web diagram in shown in Figure 39 to illustrate the transfer of energy among trophic levels.
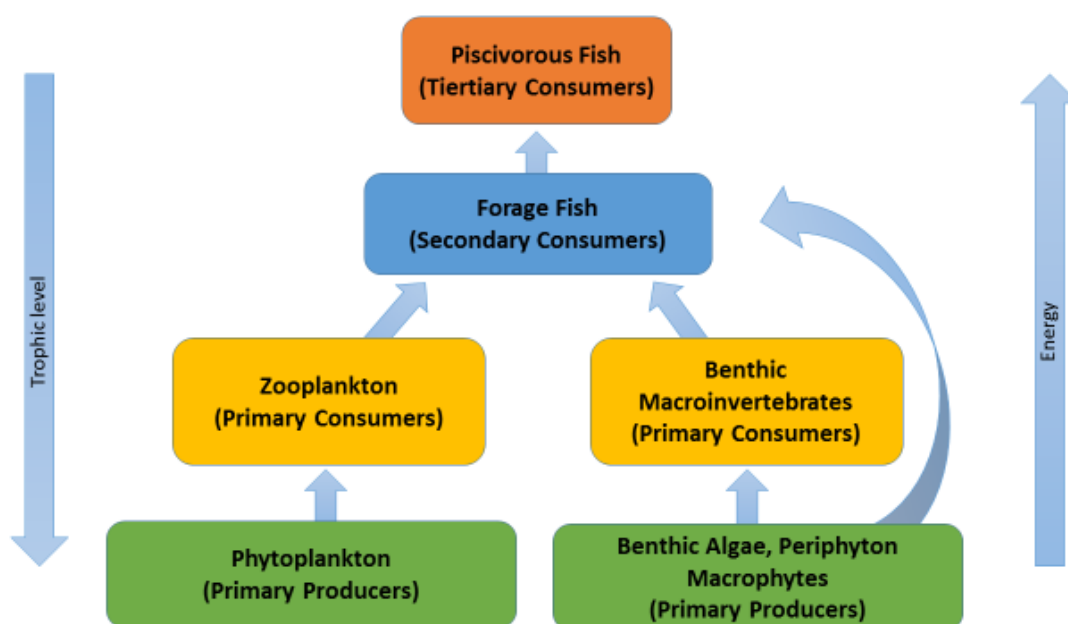
Figure 39. Simplified aquatic food web diagram. Light blue arrows indicate the direction of energy flow among trophic levels. This model forms the foundation for the simplified conceptual model for Como Lake. The model includes two primary producer and consumer pathways that generally separate water column and benthic pathways.

For background purposes, generic characteristics of each trophic level are briefly discussed below, followed by a discussion of food web characteristics specific to Como Lake.

5.2.1 Phytoplankton

The lowest level of the food web is comprised of two pathways of primary production. The first pathway is through phytoplankton, which are primary producers (i.e., autotrophs) that dwell in the water column and on the sediment surface. These organisms capture energy from the sun and produce biomass through photosynthesis. Thus, their abundance and community composition is driven by nutrient availability, light availability, temperature, thermal stratification, species composition, and predation (Wetzel 2001; Arhonditis et al. 2004; Scheffer 2004; MNDNR 2016). They are a vital part of the community in that they transfer energy to higher trophic levels, via consumption. While nutrients in the water are required to sustain primary producers, excessive

nutrients, like in Como Lake, have the potential to cause excessive growth of phytoplankton and can lead to changes in the higher level community (i.e., trophic cascade) and water quality.

5.2.2 Benthic Algae, Periphyton and Macrophytes

The second primary production pathway is via benthic algae, periphyton and macrophytes, which are important primary producers in shallow lakes. Shallow lakes exhibit seasonal transitions of primary producer types and biomass, which are then followed by consumer transitions (Brönmark and Hansson 1998). Shallow lakes have a high sediment area to water contact, which in clear lakes, allows for ample light penetration that promotes benthic algal growth and healthy macrophyte community development. In turbid lakes, like Como Lake, the biomass of benthic algae is dampened because of limited light penetration. The lack of nutrient uptake by the benthic algae may then be available for other producer types. Periphyton refers to attached algae on objects such as rocks or macrophtye leaves, and can be an important food resources for grazing zooplankton and herbivorous fish. When abundant vegetation is present, periphyton can be a dominant food resource for zooplankton grazers. Macrophytes play multiple roles in the ecological function of shallow lakes (Hanson et al. 2012). They offer attachment surfaces for periphyton as well as critical habitat and refuge for a wide range of organisms, which can be the key to a successful and diverse community structure among trophic levels (Persson and Crowder 1998). Further, macrophytes provide a critical nutrient link between sediment and water quality, and offer a key mechanism for decreasing sediment-bound nutrients and function to provide a balance in nutrient dynamics caused by resuspension and erosion (Scheffer 2004; Barko and James 1998).

5.2.3 Zooplankton

The next level of the food web in shallow lakes is comprises two pathways of primary consumption by invertebrates. The first pathway is via zooplankton, which are small heterotrophic (i.e., depend on external sources of energy) animals living in the water column. They feed mainly on phytoplankton, but some carnivorous species also consume smaller zooplankton (NOAA 2009). They are preyed upon primarily by fish, thereby providing an energy source to higher trophic levels through consumption. Zooplankton provide a crucial linkage between phytoplankton and fish in the lake, so they are influenced by bottom-up (i.e., nutrient availability) and top-down (i.e., fish predation) controls. When abundant, they can depress phytoplankton populations and minimize nuisance algal blooms, while also providing a food source for many species of fish across the year (Figure 40).
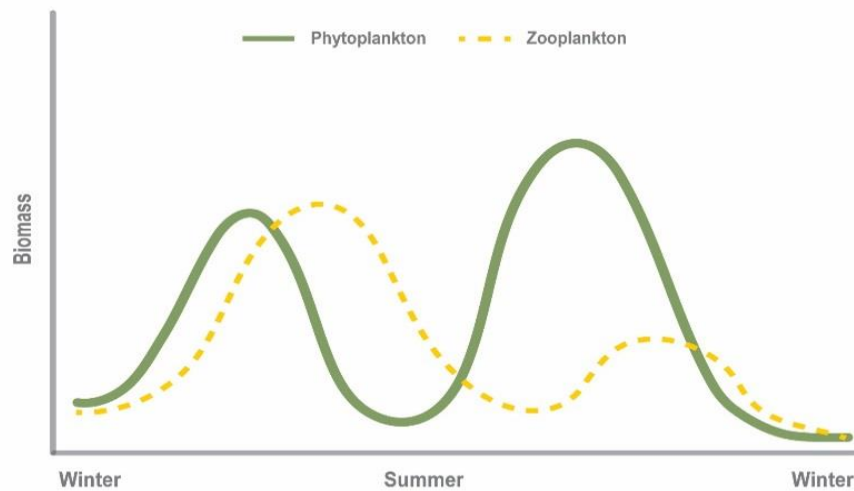
Figure 40. The diagram depicts the general patterns of algal and zooplankton biomass patterns across the year, in eutrophic systems. Zooplankton follow a typical response pattern where late spring-summer zooplankton biomass is greater than the fall-winter biomass because of the type of algae available for zooplankton consumption. Adopted from Brönmark and Hansson (1998).

5.2.4 Benthic macroinvertebrates

The second primary consumption pathway is via benthic macroinvertebrates, which are small invertebrate animals that live in the sediment of the lake and play a key role in the aquatic food web. Their role in food web dynamics is substantially more important in shallow lakes when compared to deep lakes in terms of biomass production and energy transfer, because of the strong associations between fish predation and higher proportions of macrophyte abundance typically found within these systems (Scheffer 2004; Hanson et al. 2012). Benthic macroinvertebrates tend to consume aquatic plants, phytoplankton, detritus (i.e. dead particulate organic matter), and sometimes other benthic macroinvertebrates. They are also consumed, like zooplankton, by fish and thus form an important link between producers (aquatic plants and phytoplankton) and fish (Scheffer 2004; Hanson et al. 2012).

5.2.5 Forage Fish

Forage fish are secondary consumers that mainly feed on phytoplankton, zooplankton and/or benthic macroinvertebrates. Forage fish that feed on primarily on phytoplankton or zooplankton are referred to as planktivorous. These fish typically feed primarily on zooplankton and, when overly abundant, can lower the density of zooplankton which graze on phytoplankton, thus leading to increased algal densities (Scheffer 2004; Brönmark and Hansson 1998; Moss et al. 2013). Forage fish that tend to feed on benthic macroinvertebrates are referred to as benthivorous, and can disturb the sediments through their foraging behavior, increasing turbidity and affecting other components of the food web (Bernes et al. 2013). Some forage fish are also omnivores that may feed in the benthos as well as the water column, and may selectively feed on young fish of all levels when found in large numbers (Brönmark and Hansson 1998). The U.S.

EPA (EPA 1993) suggests that intermediate densities of forage species are best for maintaining balance in both phytoplankton and zooplankton levels.

5.2.6 **Pisciviorous Fish**

Piscivores are carnivorous fish that primarily consume other fish and are generally the top predators in an aquatic food web. Therefore, they are regarded as tertiary (third level) consumers that do not normally eat zooplankton or phytoplankton, although it is important to note that some piscivorous fish may be planktivorous or omnivorous in their juvenile life stages. Overall, piscivores are important for controlling both forage and piscivorous fish populations through predation (Figure 41). In productive, densely vegetated systems, piscivores should compose >30% of the fish biomass, although the stability of the food web seems to function better when the piscivore age-class and community are diverse (EPA 1993). Piscivores also tend to be target fish species for recreation fishing, and can destabilize food webs in systems not managed to account for fishing pressures (EPA 1993). If the piscivore population becomes dominant and out of balance, the forage fish base can become suppressed, creating a trophic cascade (Hobbs and Suding, 2009), which is further discussed below.
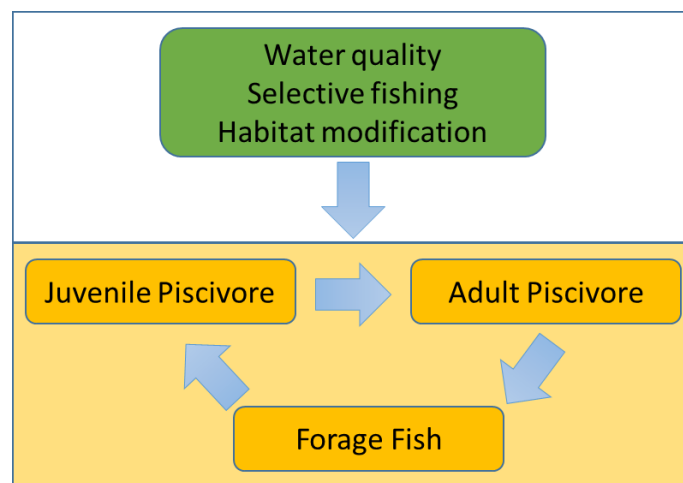
Figure 41. Generalized trophic triangle of fish with primary inputs affecting fish abundance of a managed system (Adopted from Hobbs and Suding 2009). The fish pathways primarily depict how unbalanced piscivores and forage fish affect one another to create situations of trophic cascades (described below). For example, an excessive abundance of adult piscivores can diminish the forage fish base, allowing more recruitment of juvenile piscivores, and a lack of controls on the primary consumers, leading to increased abundance of a few zooplankton and macroinvertebrates, which then allow for increased nuisance algae presence – a scenario of a trophic cascade.

5.3 Trophic Cascade

Extensive research has addressed aquatic food webs in lakes and impoundments as resource managers try to untangle the complexities of the interactions and responses to the range of management objectives for a water body (see EPA 1993). Nutrients and the food web composition are important for developing the foundation of a lakes ecosystem (Brönmark and Hansson 1998). In stocked lake systems, it is important to consider how all trophic levels respond to changes in the upper trophic levels—piscivore and planktivore fishes. The term trophic cascade refers to how

changes in the upper trophic levels cascade down towards lower trophic levels, ultimately affecting the primary producers (Brönmark and Hansson 1998). For fishes, this cascade (Figure 41) can result from an overabundance of either the top (piscivores) or second (planktivores) tiers, which can be observed from data among the trophic levels. A site-specific food web model can help identify trophic levels and their components that may be out of balance, and support a management plan that helps meet the objectives of the system.

5.4 Conceptual Trophic Food Web Model in Como Lake

LimnoTech built upon the biological data trend analyses and conducted additional research to inform categorization of functional feeding groups of species present in the biological survey data. For this assessment, LimnoTech focused on the most recent data, which were collected during the growing seasons (May-Sept) of 2014 through 2016. Species were segregated into the trophic levels as described above (Figure 39; i.e., primary producers, primary consumers, secondary consumers, and tertiary consumers). The relative abundance of each species was calculated, which is simply the percent composition of that species within a trophic level for that sampling year.

Planning-level food webs based on Como Lake data are illustrated in Figure 42 through Figure 44. For simplicity, the adult feeding behavior of fishes is primarily considered in assigning to a trophic level. Species were given their own box when they composed greater than 5% relative abundance. Species with less than 5% relative abundance were grouped according to their general trophic position and categorized as "Rare." At less than 5% relative abundance, these organisms are not expected to significantly impact community dynamics in the lake. No abundance data are available for benthic macroinvertebrates, benthic algae or periphyton, so no percentages were included in the boxes. Macrophytes' presence was reported as % occurrence by area, whereas other biological data are calculated as relative percent abundance, so numbers were not included in the food web models for consistency among units and representation.

A light blue arrow connecting trophic levels indicates the direction of energy transfer among groups. It is likely that some fish species selectively feed on certain types of zooplankton, but too much uncertainty exists with most consumers and predators to specify that level of detail. Accordingly, the primary objective of this task is to understand the greater influences that higher trophic levels may be exerting on lower trophic levels to understand the relative condition of the lake. As can be observed in Figure 42 through Figure 44, the fish community in Como Lake has become dominated by forage fish that typically feed on plankton and/or benthic-dwelling macroinvertebrates. As discussed above, this distribution can result in a situation where a disproportionately large forage fish community suppresses zooplankton abundance to such a degree that fewer controls are placed on phytoplankton density - a model example of a trophic cascade.

It should be noted that not all possible linkages between species are included in the model, and thus represent a planning-level food web model of dominant relationships. That is, the relationships represent those that are identified in the literature as being common and substantial. For example, northern pike and walleye, which are piscivores, feed on zooplankton and benthic macroinvertebrates when they are young, but quickly transition to a diet of mainly fish as they mature. Similarly, although the preference for black bullhead and yellow perch are planktonic invertebrates, both are known to feed on newly recruited fish (e.g., piscivores) in situations where imbalance fish communities exist. However, for the purposes of this conceptual food web evaluation, only the adult dominant feeding behavior was considered for trophic level positioning.
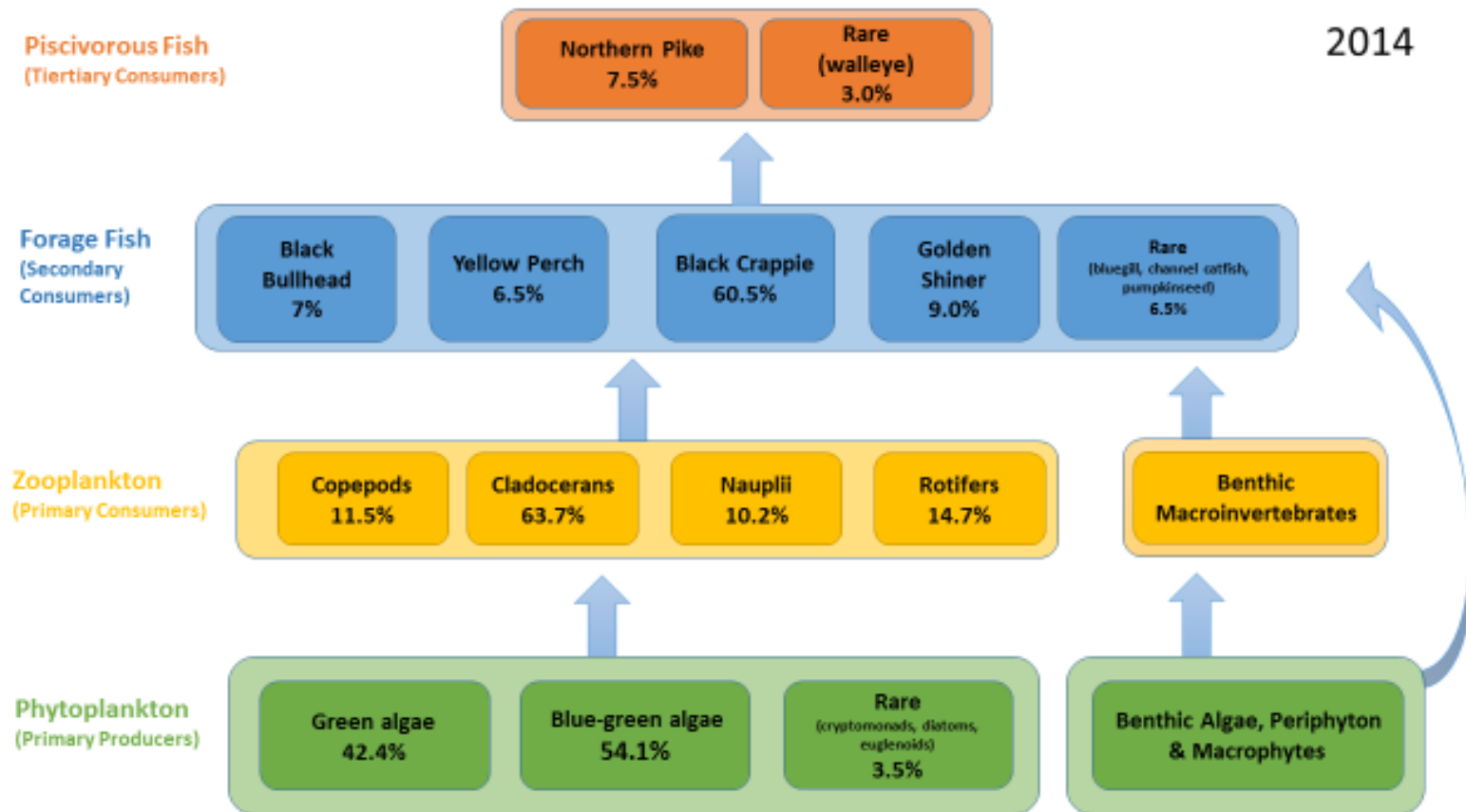
Figure 42. Planning-level food web model for the 2014 growing season (May – Sept). Proportional estimates are not available for the benthic algae-macroinvertebrate pathway, while estimates are made for the water column pathway for phytoplankton and zooplankton. The upper two trophic levels depict a dominance of forage fish, especially black crappie, and relatively few piscivores. It is likely that the dominance of the forage fish is suppressing piscivore recruitment as the forage fish are likely feeding, at some level, on the recruited piscivore.
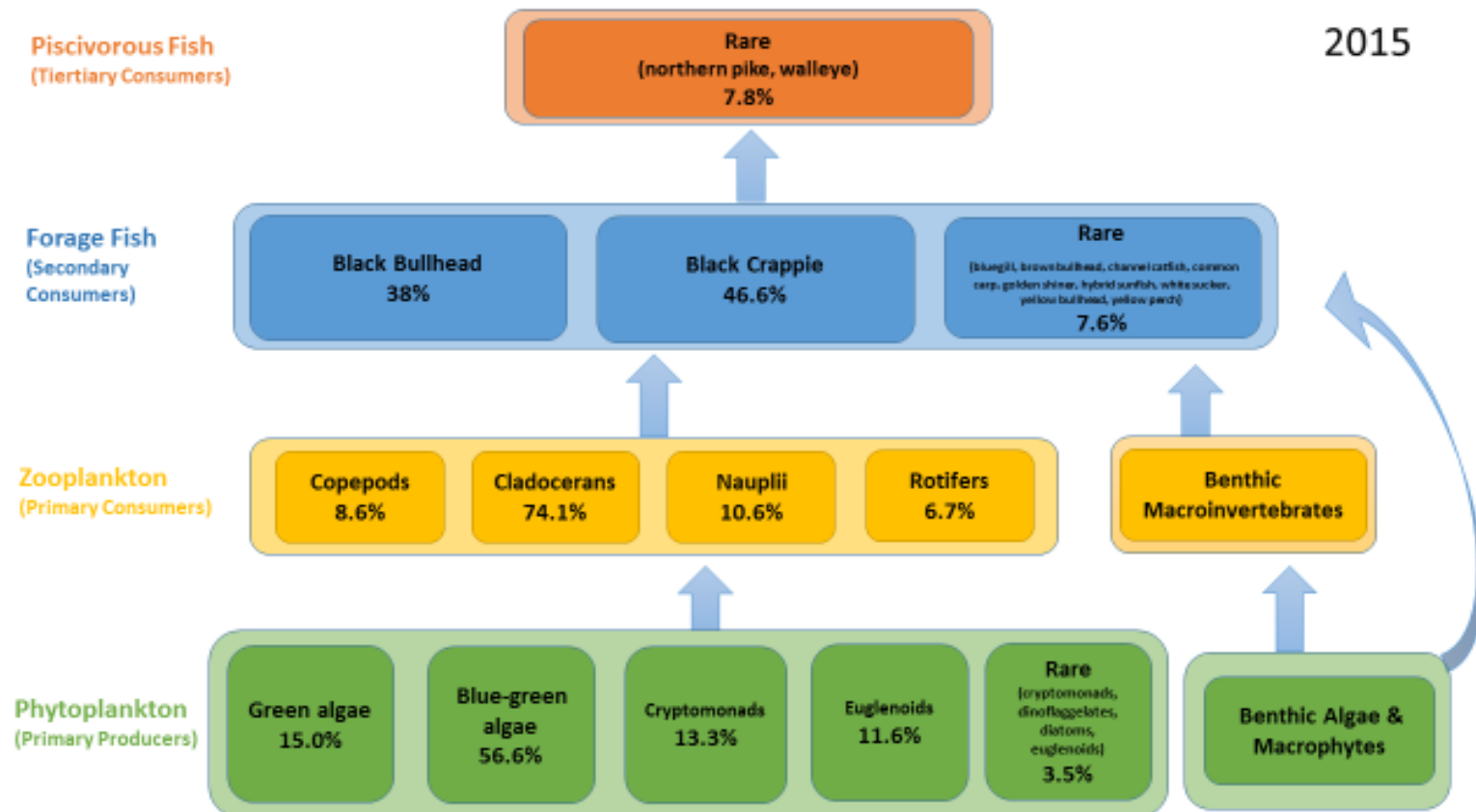
Figure 43. Planning-level food web model for the 2015 growing season (May – Sept). The upper two trophic levels depict a greater dominance of forage fish (now black crappie and bullhead are co-dominant) and even fewer piscivores than 2014. It is likely that the increasing dominance of the forage fish, and particularly black bullhead, are increasingly suppressing piscivore recruitment. Black bullhead, in particular, are known as omnivores that will feed on recruited piscivores (i.e. juvenile forms).
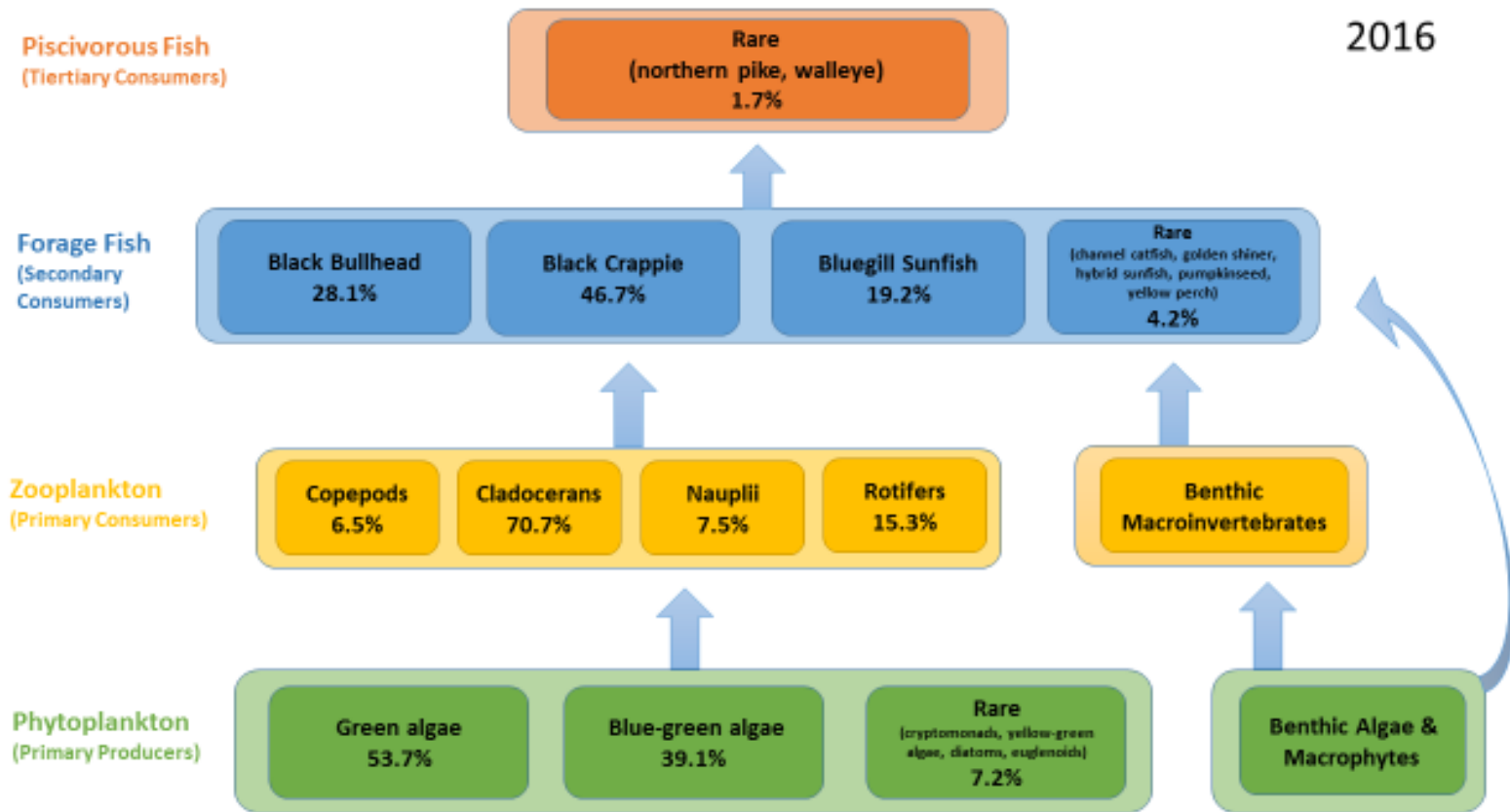
Figure 44. Planning-level food web model for the 2016 growing season (May – Sept). The upper two trophic levels depict a greater dominance of forage fish (now black crappie and bullhead are co-dominant, with a high proportion of bluegill) and even fewer piscivores than 2014 and 2015. It is unlikely that the piscivores can recover from this imbalanced trophic system without some level of active management support.

Characteristics of the dominant groups represented in each trophic level, and their significance in Como Lake, are briefly described in the following sections.

5.4.1 Phytoplankton

In recent years, blue-green algae (Cyanophyta) and green algae (Chlorophyta) have been the most dominant phytoplankton groups. Blue-green algae, which typically dominate in eutrophic lakes, can form blooms in the late summer due to excessive growth. Many species are not desirable food resources by zooplankton due to size or toxin production (Persson and Crowder 1998; Scheffer 2004). Consequently, blue-green algae abundance can increase significantly in the absence of grazing pressure by zooplankton. Green algae are single-celled organisms that most likely form the main nutritional basis of the food web, but can also be composed of nuisance species that may be undesirable food resources (Scheffer 2004; NOAA 2009), which could be the case for Como Lake.

Other species present include the cyrptomonads (Cryptophyta), euglenoids (Euglenophyta), diatoms (Bacillariophyceae), yellow-green algae (Chrysophyta), and dinoflagellates (Pyrrhophyta), but these are generally less abundant than the blue-green algae or green algae. It is difficult to differentiate which species specifically are consumed by larger organisms, but in general, phytoplankton are preyed upon by zooplankton, benthic macroinvertebrates, and herbivorous/omnivorous fishes.

5.4.2 Benthic Algae, Periphyton and Macrophytes

No benthic algae or periphyton data are available for Como Lake. Given the abundance of benthic-dwelling forage fish in Como Lake and the areal extent of the shallow area (~96% littoral zone), it is possible that benthic algae are a significant food resource. In shallow lakes, benthic algae and periphyton greatly affect whole-like primary production, although a direct trophic response of primary consumer densities is less well understood (Vadeboncoeur et al. 2008). It would be assumed that the density of the primary consumer community would increase in response to a high primary production condition, but that is speculative. There is substantial macrophyte data that have been collected in Como Lake. Macrophytes play more of an indirect role in food webs, as they provide critical habitat for zooplankton seeking refuge from fish predation. Macrophtytes are also consumed by herbivorous fish (Scheffer 2004).

5.4.3 Zooplankton

The zooplankton community in Como Lake has been dominated by cladocerans in recent years, with rotifers, nauplii, and copepods (cyclopoid and calanoid) also being present. Cladocerans have generally increased in abundance from 1984- 2016. Cladoceran density is often high in eutrophic lakes (Hsieh et al. 2001), so this condition is not surprising for Como Lake. Of particular interest, however, is that the cladoceran community appears to be dominated by small-bodied zooplankton. In most years, total cladoceran abundance is dominated by *Bosminae* or *Chydorus*, which are known to be small-bodied zooplankton (NOAA 2006; Jiang et al. 2016). In contrast, *Daphnia,* a genus of large-bodied cladocerans, are common in some years, but never appear in high densities. *Daphnia* are an important component of forage fish diets in Minnesota lakes (Hanson et al. 2012). Selective predation on large-bodied zooplankton by planktivorous fish can drastically shift zooplankton dominance to small-bodied zooplankton (Vanni and Findlay 1990; Jiang et al. 2016).

Zooplankton filter-feed large quantities of phytoplankton, which can help control excessive algal growth (NOAA 2009). In particular, large-bodied zooplankton are more efficient grazers than small-bodied zooplankton (Vanni and Findlay 1990). Furthermore, abundance of small-bodied zooplankton have been positively correlated with blue-green algal blooms because they tend to avoid feeding on toxic blue-green algae (Hansson et al. 1998; Jiang et al. 2016). Consequently, zooplankton size can differentially impact top-down and bottom-up feeding strategies and preferences by consumers (Scheffer 2004). Zooplankton

size data in Como Lake is limited to *Daphnia* and was not explored in more detail for this planning-level assessment. However, the high density of small-bodied zooplankton relative to large-bodied zooplankton suggest that planktivorous fish may be selectively feeding on large-bodied zooplankton leading to reduced zooplankton grazing pressure on phytoplankton.
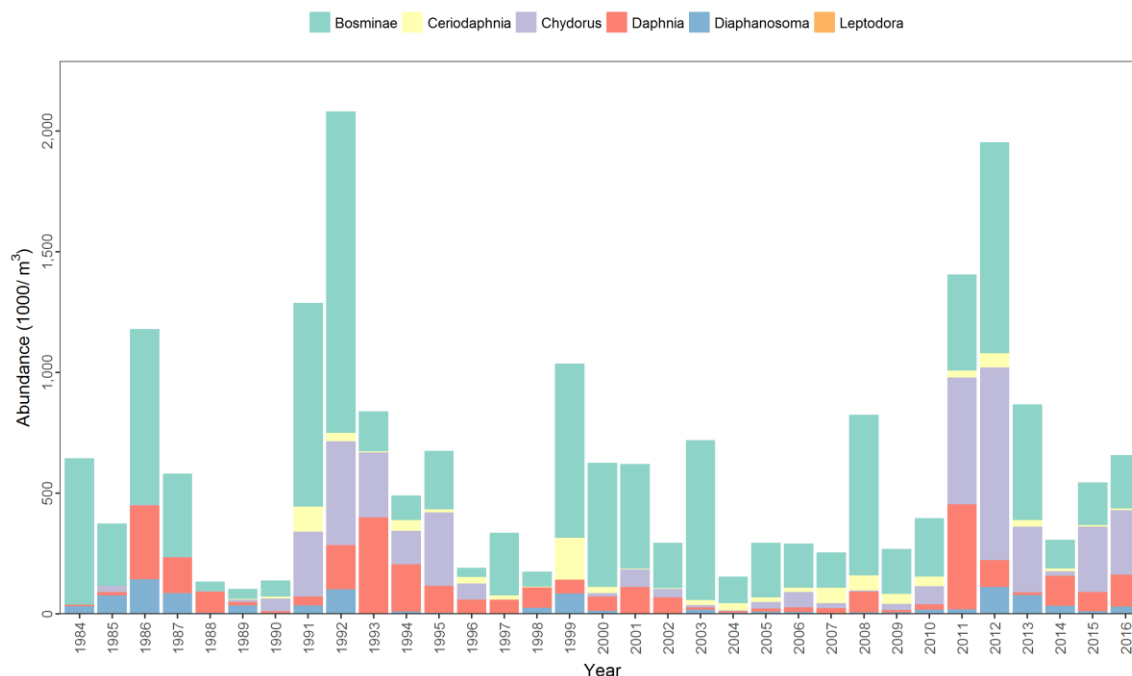
Figure 45. Relative abundance zooplankton groups in the Order Cladocera. Total cladoceran abundance shows a cyclical inter-annual pattern dominated by the genus *Bosmina* in most years.

5.4.4 Benthic Macroinvertebrates

No benthic macroinvertebrate data are available for Como Lake.

5.4.5 Forage Fish

In recent years, Como Lake has been dominated by omnivorous forage fish, specifically black crappie and black bullhead. Black crappie feed on zooplankton, benthic macroinvertebrates, and small fish (Holtan 1997). Similarly, black bullheads tend to be planktivores as juveniles (zooplankton and phytoplankton), but also feed on benthic macroinvertebrates and small fish as adults (Stuber 1982; Dehring & Krueger 2008). Other less common forage fish in Como Lake have included bluegill sunfish, golden shiner, and yellow perch. Rarer species in recent years include brown bullhead, channel catfish, common carp, hybrid sunfish, pumpkinseed, white sucker, and yellow bullhead. The dominance of forage fish that could be suppressing the zooplankton population is a more important observation than what food resources are utilized by specific species. Age of the forage fish also play a role in suppressing zooplankton taxa. Hannson et al. (1998) reported that lakes with high abundance of young-of-the-year fish (i.e. less than one-year-old) had lower densities of large-bodied zooplankton. Fish stocking practices often involve young-of-year or yearling fishes and has been the case for Como Lake in some years (Table 14). Consequently, age as well as species of forage fish may be contributing to the trophic cascade in Como Lake. The results of this planning-level evaluation suggest that secondary consumers are controlling the abundance and distribution of zooplankton taxa, which may be impacting phytoplankton density in Como Lake.

5.4.6 **Pisciviorous Fish**

The only piscivorous fish present in Como Lake in recent years (2014-2016) were northern pike and walleye. According to the fish surveys, these species have been relatively uncommon, with the highest abundance (10.5%) in 2014. A higher abundance of piscivorous fish that prey on forage fish would lower the density of fish exerting predation pressure on zooplankton.

5.5 Conclusion

The results of this Como Lake planning-level food web analysis finds an imbalanced food web likely resulting from a trophic cascade, which seems to be driven by an overabundance of planktivorous fishes. The fish community in Como Lake has few top predators, and is dominated by planktivorous forage fish due to low predation pressure. As a result, cascading effects on the zooplankton community can be observed. The zooplankton data shows an increasing abundance of cladoceran zooplankton that is dominated by small-bodied taxa. The low density of large-bodied zooplankton is likely the result of intense size-selective feeding pressure by planktivorous fish leading to increased abundance of small-bodied zooplankton. Small-bodied zooplankton are less efficient grazers on phytoplankton than large-bodied zooplankton which allows for increased phytoplankton growth.

Water quality conditions remain poor, primarily due to high phosphorus loading and excessive algal growth. This condition combined with a forage fish dominated fishery, suggests a need for in-lake management strategies that aim to reduce phosphorus loading, but also focus on rebuilding the piscivore fish community in Como Lake to establish a more stable food web. An approach for rebuilding the trophic structure and diversity in Como Lake could take advantage of the long term trophic-level trend data as reported by Noonan (1998), along with the recent data provided in this report. It is likely that the ecosystem within this popular lake will not rebound without some active management, and the lake managers are in a unique position to have an abundant dataset from which to build a well-designed plan to initiate specific actions.

<div align="right">6</div>

Sources of Internal and External Phosphorus Loading

CRWD has installed numerous stormwater BMPs throughout the watershed, which have reduced some external nutrient loading to Como Lake from the watershed. Improving water quality conditions in Como Lake will also require in-lake management strategies to reduce internal loading of phosphorus. LimnoTech evaluated available data and information to characterize and estimate the magnitude of P loading from external and internal sources. The results of that assessment are presented in the following sections.

6.1 Sources and Magnitude of P Loading

We evaluated the relative magnitude of internal P loading through various source mechanisms through screening-level mass flux calculations, which were designed to:

- Estimate the mass flux of phosphorus to Como Lake from various source categories,
- Assess the uncertainty in those source contributions, and
- Determine potential primary versus secondary drivers of phosphorus loading and in-lake dynamics.

The source mechanisms considered consisted of:
- Watershed loads,
- Release of dissolved phosphorus from lake bottom sediments,
- Wind-driven resuspension of sediment phosphorus, and
- Bioturbation and macrophyte community influence on sediment P flux.

6.1.1 External Watershed Loading

Capitol Region Watershed District (CRWD) has installed numerous permanent stormwater treatment BMPs to control external watershed loading of total phosphorus (TP) to Como Lake (Houston Engineering 2016). LimnoTech summarized available watershed loading estimates developed in a recent study. Through P8 modeling, Houston Engineering recently estimated a total watershed load of 1,198 lbs. TP/yr but also found that approximately 37% of that total annual TP load from the watershed is removed through existing stormwater treatment practices. The estimated watershed load actually entering Como Lake is approximately 759 lbs. TP/yr. Details on the modeling, calculations and assessment can be found in Houston Engineering (2016).

6.1.2 Internal Loading

Phosphorus loads to Como Lake were estimated for the following internal sources: Sediment Diffusive Flux, Wind-driven Resuspension of Sediment Phosphorus, and Bioturbation and Macrophyte Community Influence. The calculations and resulting loads from each category are discussed below.

6.1.2.a Sediment Diffusive Flux

The loading from sediment diffusive flux is calculated as the product of sediment flux rates (i.e. mass of P released per surface area per time) and bottom surface area. Flux rates vary over time and space, because the flux is strongly dependent upon the dissolved oxygen concentration at the sediment-water interface.

This section first describes how flux rates were calculated for oxic (oxygen present) and anoxic (oxygen absent) conditions, then merges that information with data on the times and locations where oxygen is present in order to estimate an annual phosphorus load from diffusive flux.

Flux Rates

Flux rates of the diffusion of soluble phosphorus from Como Lake sediments was estimated in two separate manners: 1) Explicitly, via application of sediment flux measurements reported in "Sediment Characterization for Lake McCarrons and Como Lake" (Wenck 2016), and 2) Implicitly, via examination of the increase in hypolimnetic phosphorus concentrations during periods of stratification. The explicit method provides results for locations where sediment flux measurements were conducted, and consist of both oxic and anoxic conditions at a shallow lake station, as well anoxic fluxes at a deep lake station. The implicit method provides an estimate only for anoxic conditions in the deeper portion of the lake, and is conducted solely to provide an independent verification of the explicit rate measured for anoxic conditions at the deep station.

Sediment phosphorus diffusive flux rates were directly measured from sediment cores taken from two locations in Como Lake. As described in Wenck (2016), Station 1 represented a shallow lake station while Station 2 was located near the deepest portion of the lake. Laboratory flux measurements were conducted to calculate sediment flux at Station 1 for both oxic and anoxic conditions, while flux measurements were conducted only for anoxic conditions for Station 2. The results, shown in Table 10, demonstrate the large difference in release rates between oxic and anoxic conditions.

Table 10. Como Lake anoxic and oxic release rates for intact sediment cores (from Wenck, 2016).

Station	Oxic Release Rate (mg/m^2/d)	Anoxic Release Rate (mg/m^2/d)
1	0.2	5.8
2	--	6.5

Anoxic sediment flux rates were implicitly estimated based upon observed increases in hypolimnetic phosphorus concentrations during periods of anoxia. The specific steps taken in this analysis consisted of:

1. Examining the entire time series of data at station 5401-201
2. Finding the first date that has stratification and bottom DO <= 1, that persists for more than one date
3. Calculating the average hypolimnetic TP for that date
4. Find the last date of stratification corresponding to the event identified in Step 1
5. Calculating the average hypolimnetic TP for that date
6. Calculating the rate of rate of change in TP concentration over time (e.g. [TP2 -TP1]/ΔT)
7. Calculating the volume and bottom surface area of hypolimnion, based upon average reported thermocline depth during stratification period
8. Calculating flux rate as rate of concentration change x volume ÷ surface area

A total of 19 anoxic stratification events were identified and assessed, with these events documented in Appendix D. The average phosphorus flux rate calculated using total phosphorus was 5.6 mg/m²/day. This value is very close to the range of explicitly measured anoxic release rates reported in Wenck (2016) of 5.8 to 6.5 mg/m²/day, and serves as an independent verification of that measurement.

Annual Loading Calculation

Given the large difference in release rates between oxic and anoxic conditions, calculation of the overall mass loading from sediment diffusive phosphorus flux should consider the amount of time and spatial extent that each of the different release rates occurs in Como Lake as depicted in Equation 1.

Equation 1:

Total diffusive flux =
   No. of days that oxic conditions are present x Area of oxic sediment x Oxic Release Rate
 + No. of days that anoxia is present x Area of anoxic sediment x Anoxic Release Rate

Application of Equation 1 becomes slightly more complicated for Como Lake due to the nature of its bathymetry and the effect of this bathymetry on spatial variability in anoxia. Como Lake consists of: 1) a deeper mid-lake section that undergoes distinct thermal stratification each summer, and 2) a shallower littoral zone that does not exhibit distinct thermal stratification but still demonstrates oxygen stratification. Because thermal stratification serves to prevent oxygen-rich surface waters from mixing with water near the bottom sediments, areas of the lake that exhibit thermal stratification are expected to have a higher frequency of anoxia. Equation 1 can be expanded to account for difference in the number of days where anoxia is present between the mid-lake and littoral areas, as shown in Equation 2:

Equation 2:

Total diffusive flux =
   No. of days that oxic conditions are present mid-lake x Area of mid-lake sediment x Oxic Release Rate
 + No. of days that anoxia is present mid-lake x Area of mid-lake sediment x Oxic Release Rate
 + No. of days that oxic conditions are present littoral x Area of littoral sediment x Oxic Release Rate
       + No. of days that anoxia is present in littoral zone x Area of littoral sediment x Oxic Release Rate

Recent routine dissolved oxygen monitoring provides an estimate of the number of days that oxic vs. anoxic conditions exist for the mid-lake area, but no such recent data are available for the littoral zone (i.e. 1993 is most recent year). Wenck (2016) used a shallow lake Anoxic Factor developed by Nürnberg (2005) to estimate the number of days that anoxia is present. This Anoxic Factor was based upon an empirical regression from other shallow lakes as a function of lake morphometry and TP concentrations. Nürnberg's regression, when applied by Wenck to Como Lake, resulted in an estimated 67.4 days of anoxia per year that applies to the entire area of the lake.

While the calculations in Wenck (2016) are certainly reasonable, it is noted that the calculation of the number of days that anoxia is present is based upon an empirical regression that: 1) was based on data from lakes other than Como Lake, and 2) contains a significant degree of scatter. LimnoTech examined historical Como Lake dissolved oxygen data to estimate the number of days per year that anoxia is present at both the deep and shallow stations, using two different criteria for defining whether anoxic conditions are present. Results are summarized in Table 11.

Table 11. Estimated number of anoxic days per year under different low dissolved oxygen (DO) conditions.

Station	Number of Days per Year of Anoxia (Anoxic Factor)	
	DO < 1 at deepest measurement	DO < 2 at next-to-deepest measurement
Site 5401-201 (Deep)	221	138
Site 5404-102 (Shallow)	195	42

This information can be input into Equation 2, using the measured oxic release rate of 0.2 mg/m²/day and anoxic release rate of 6.5 mg/m²/day to generate estimates of total diffusive flux using each of the anoxic condition criteria. For the criterion "DO < 1 at deepest measurement", the resulting flux is 819 lbs/yr, while the resulting flux for the criterion "DO < 2 at next-to-deepest measurement" is 293 lbs/yr (Table 12). The specific calculations behind these results are provided in Appendix E.

Table 12. Estimated diffusive phosphorus flux under different low dissolved oxygen (DO) conditions.

Station	Phosphorus Flux (lbs/yr)	
	DO < 1 at deepest measurement	DO < 2 at next-to-deepest measurement
Site 5401-201 Oxic	4	6
Site 5401-201 Anoxic	205	128
Site 5404-102 Oxic	16	30
Site 5404-102 Anoxic	594	128
Total	**819**	**293**

These results highlight three important facts:

1. Assumptions made regarding the percent of time that anoxic conditions are extremely important. The difference in flux amounts between the two assumptions in the shallow areas of the lake (594 − 128 = 466 lbs) is much larger than the total flux rate for the entire lake calculated using the second assumption (293 lbs).
2. The dissolved oxygen profile data used in the shallow station calculations are from 1984-1993, so may no longer represent current conditions.
3. The surface area assumed to be represented by shallow versus deep stations is a significant factor. Surface areas for this report were calculated from the digital elevation map of lake bathymetry for those portions of the lake shallower and deeper than 2.6 m (the average observed depth of the thermocline).

The combination of the above facts highlights the need to collect dissolved oxygen profile data at shallow lake stations in the future if a more accurate estimate of overall annual flux from the sediments is desired. The uncertainty in average annual internal load can currently be estimated as ranging from the 283 pounds estimated by Wenck (2016) to the upper bound load estimated in this report of 819 pounds.

6.1.2.b *Wind-driven Resuspension of Sediment Phosphorus*
This sub-task was defined to determine the relative influence of wind-driven sediment resuspension on overall phosphorus loading to the lake. It was conducted in the following steps:

1. Verify, using observed data, the relationship between water column total particulate matter and total phosphorus concentrations.
2. Define those sampling events which contained much larger than normal levels of particulate material.
3. Define the climatic conditions that occurred prior to each of the "high solids" sampling events.
4. Review the extent to which the presence of high wind and/or significant rainfall occurred in the days immediately prior to the "high solids" sampling events.
5. Make a qualitative assessment of the extent to which high wind events could be deemed responsible for elevated solids concentrations, and thereby serve as a source of phosphorus to the lake's water column.

Relationship between Water Column Solids and Total Phosphorus

The first step in assessing the relative influence of wind-driven sediment resuspension on phosphorus loading to the lake is to verify that increases in particulate matter correspond to increases in observed TP. All available May through October water quality data for sample depths less than one meter were compiled for the Stations 201 and 102, and regressions were developed between observed TP and total particulate matter. The results are shown in Figure 46 and Figure 47. The relatively low R^2 values (~0.2) indicate that solids are not a perfect predictor of phosphorus, but the strong statistical significance (p<0.0001) confirms that with higher solids concentrations correspond to higher TP values.

Como Lake: Station 201

$$y = 0.104 + 0.00774\,x$$
$$R^2 = 0.21$$
$$p\text{-value} = 9.06\text{e-}12$$
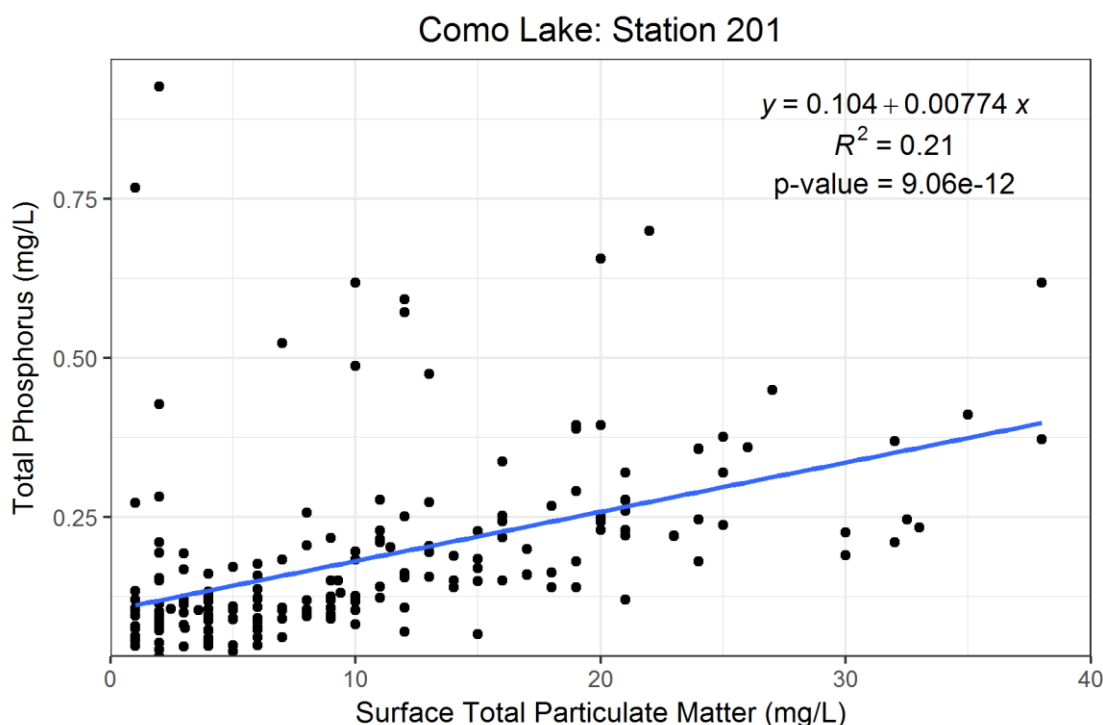
Figure 46. Relationship between Total Phosphorus and Surface Total Particulate Matter for May-October Surface Samples at Station 201.
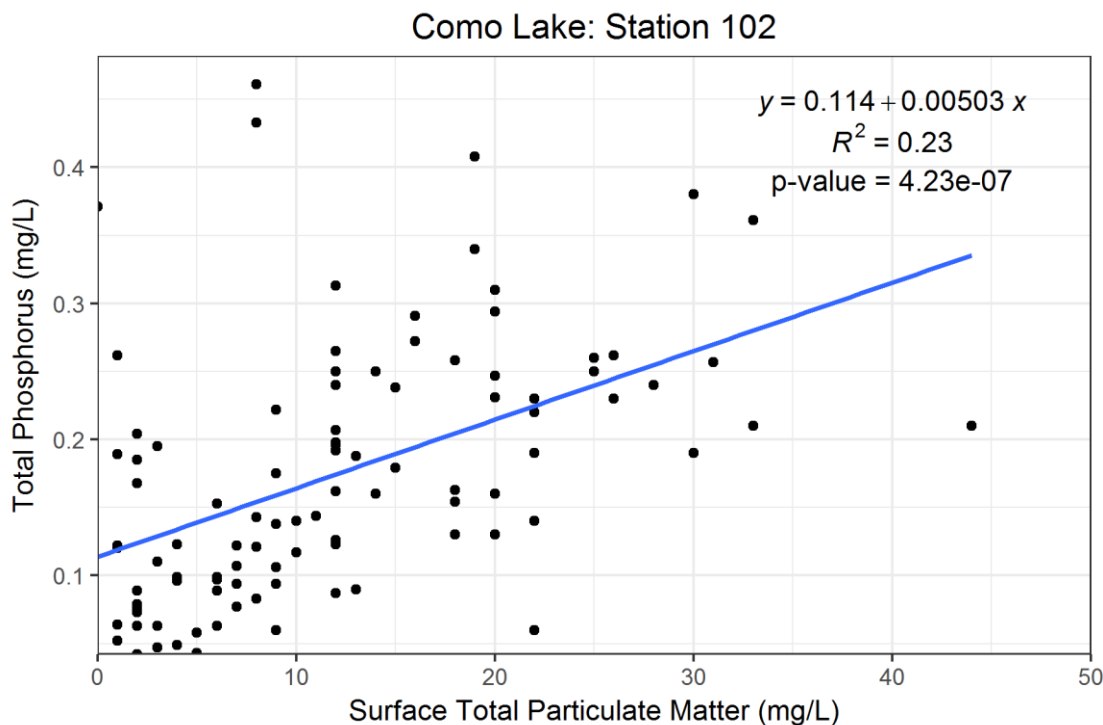
Figure 47. Relationship between Total Phosphorus and Surface Total Particulate matter for May-October Surface Samples at Station 102.

Define "High Solids" Sampling Events

The second step in assessing the relative influence of wind-driven sediment resuspension on phosphorus loading to the lake was to identify those sampling events that contained much higher than average levels of total particulate matter, defined using best professional judgement for this assessment as greater than 25 mg/l (roughly the 90th percentile) at either lake monitoring station. A total of 25 sampling events were identified that met this criterion.

Define the Climatic Conditions Prior to "High Solids" Events

The third step in assessing the relative influence of wind-driven sediment resuspension on phosphorus loading to the lake was to define the climatic conditions that occurred in the four days prior to each observed "high solids" event. The four-day period was selected to account for the time necessary for the added particulate matter to settle out of the lake. Historical wind data were retrieved from Minneapolis St. Paul International Airport, while precipitation data were obtained from the University of Minnesota Climate Data Center. The climatic condition associated with each high solids event are plotted below (Figure 48).
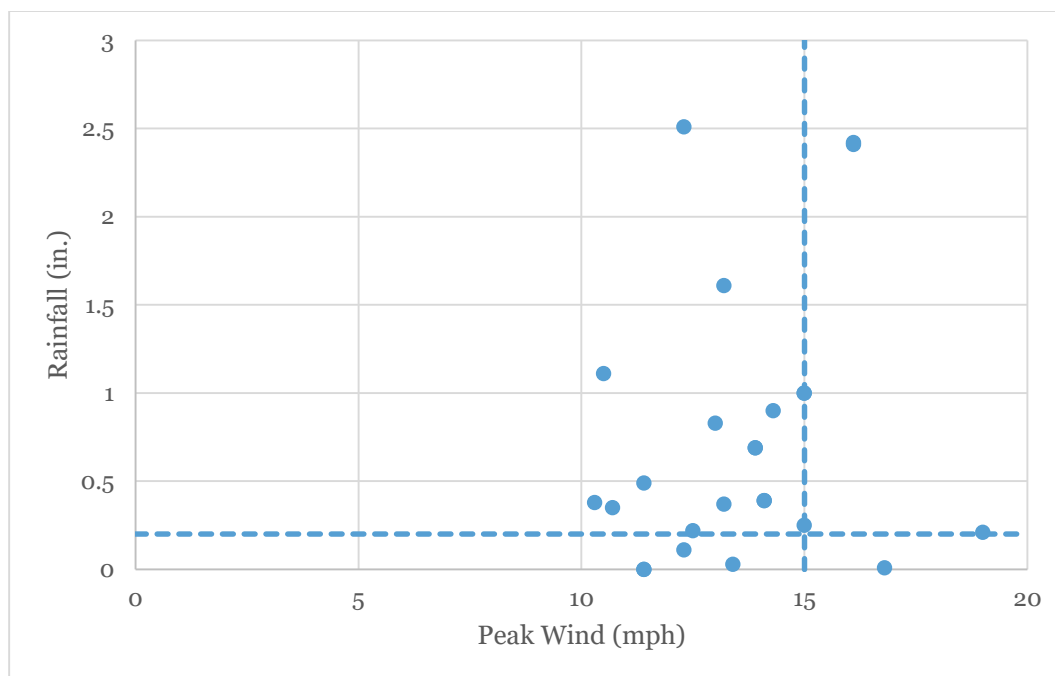
Figure 48. Antecedent Rainfall and Peak Wind Speed Associated with High Solids Events.

A simple wind-wave model was developed for Como Lake, based on work published by Christoffersen, and Jonsson (1985). The model calculates bottom shear stress as a function of wind speed, water depth and fetch (i.e. wind can travel over water before intersecting a land mass). Application of this model with Como Lake bathymetry, along with a common assumption that 1 dyne/cm is an approximate threshold above which resuspension occurs (e.g. James et al. 2004), identified 15 mph as a threshold wind speed necessary to trigger significant wind-induced resuspension. A threshold value of 0.2 inch of rainfall was selected as representing "wet weather" conditions. The following conclusions can be drawn from the data in Figure 48:

- The large majority (21 out of 25, 84%) of high solids events were associated with antecedent rainfall >0.2"
- The median rainfall antecedent to a high solids event was 0.39"
- Of the five events that had sufficient wind to cause resuspension, four of them also had antecedent rainfall >0.2"

Given that all but one of high wind events also had rainfall associated with them, it is not possible to fully distinguish potential wind-induced resuspension events from events with wet-weather runoff as a cause of increased solids and phosphorus concentrations. The fact that high solids events were four times more likely to be associated with rain as opposed to high wind (21 occurrences vs. five) strongly suggests that rainfall-driven watershed loading is a more important source of phosphorus to the lake than wind-induced resuspension. The conclusion that high solids events in the lake were more likely to occur following rainfall still apply, even for threshold values different 15 mph for wind and 0.2" for rainfall.

6.1.2.c Bioturbation Influence on Sediment Phosphorus

The contribution of phosphorus to the lake via fish bioturbation and macrophyte senescence was assessed by evaluating the composition and relative abundance of the fish and macrophyte communities, and

comparing those to literature-based estimates of species-specific bioturbation and senescence-induced sediment phosphorus release.

Bioturbation (i.e., fish behavior)

The behavior of benthic-dwelling fish can contribute phosphorus to the lake by stirring up sediment via two main mechanisms: 1) benthic feeding, and 2) spawning activities. Based on the list of species sampled in the lake from 2001-2016 (Table 7), the following benthic-dwelling species are present and have the potential to cause bioturbation: black bullhead, black crappie, bluegill sunfish, brown bullhead, channel catfish, common carp, green sunfish, hybrid sunfish, largemouth bass, pumpkinseed, white sucker, and yellow bullhead. Detailed information on species-specific behavior causing bioturbation of sediments can be found in Table 13 below.

Quantitative estimates of the P load via fish bioturbation are limited and are generally restricted to carp biomass. Carp are not found in large numbers in Como Lake, but have appeared sporadically in historical fish surveys (Table 14). However, many of the fish species in Como Lake exhibit benthic-dwelling behavior through feeding or spawning (Table 13), so some general qualitative assumptions on their contribution to internal P loading through bioturbation can be made. The fish abundance data provide qualitative information about the likelihood of bioturbation as a source and can also be used to ensure that fish stocking and management practices are in alignment with water quality improvement goals for Como Lake.

Table 13. List of species sampled from 2001-2016 and their potential to increase phosphorus concentrations via bioturbation, based on the literature.

Species	Bioturbation via Feeding?	Bioturbation via Spawning?	References
Black Bullhead	Yes	Yes, April-July	Dehring & Krueger 2008
Black Crappie	No	Yes, May-July	Holtan 1997
Bluegill Sunfish	No	Yes, May-August	Thorp 1988; Paukert et al. 2004
Brown Bullhead	Yes	Yes, April-July	Dehring & Krueger 2008
Channel Catfish	Yes	Yes, summer	McKaye et al. 1994; Zimmerman 2011
Common Carp	Yes	Yes, spring & summer	Weber & Brown 2009; Huser et al. 2015
Golden Shiner	No	No	Hall et al. 1979; ODNR 2016
Green Sunfish	No	Yes, May-August	Thorp 1988; Zimmerman 2011
Hybrid Sunfish	No	Yes	Thorp 1988; Zimmerman 2011
Largemouth Bass	No	Yes, April-June	Thorp 1988; ODNR 2016
Northern Pike	No	No	Becker 1983; ODNR 2016
Pumpkinseed	No	Yes, April-August	Becker 1983; Thorp 1988
Walleye	No	No	Mathias & Li 1982; Becker 1983
White Sucker	Yes	No	Zimmerman 2011
Yellow Bullhead	Yes	Yes, April-July	Dehring & Krueger 2008
Yellow Perch	No	No	Herman et al. 1959

In 2016, three fish species were most abundant in the fish survey (Figure 37 and Figure 38): black crappie (47%), black bullheads (28%) and bluegill sunfish (19%). All three of these species/families have the potential to increase phosphorus levels in the lake through bioturbation.

- Black crappie, which spawn May-July, construct disc-shaped nests by using their caudal fins to fan out small depressions in the benthos near emergent vegetation (Holtan 1997). This fanning, which sweeps away silt and debris, resuspends phosphorus into the water column.
- Bullheads (black, brown, and yellow) are known to cause bioturbation by their feeding and spawning activities. During spawning, which occurs April-July, they use their lower fins to move sediment and debris around to excavate saucer-shaped nests. The parents also continually fan the nest after spawning to protect the eggs. Bullheads are known to destroy aquatic vegetation and stir up sediment on the bottom in their search for food, increasing turbidity and resuspending phosphorus (Dehring & Krueger 2008). It should be noted that there are questions in the literature as to whether bullheads' impact on phosphorus concentrations is caused by physical bioturbation or excretion (Keen & Gagliardi 1981; Fischer et al. 2013).
- Sunfish (bluegill, green, hybrid, largemouth bass, and pumpkinseed) contribute to bioturbation during spawning, which occurs April-August. They use their tails to generate turbulence and sweep out vegetation, silt, and debris in the creation of saucer-shaped nests in the benthos (Thorp 1988).

Additionally, channel catfish and common carp, which have been found in the lake but not in large quantities, are notorious for increasing P levels through bioturbation. Catfish feed by burrowing into the mud and moving side to side to stir up food particles out of the benthos, thereby resuspending sediment and phosphorus in the process (McKaye et al. 1994). While only one common carp has been collected in the lake since 2001, carp are most often cited regarding bioturbation impacts and thus require mentioning here. Carp are a generalist species that dig into the sediment while searching for food, uprooting aquatic plants, resuspending sediment, and increasing turbidity in the process (Huser et al. 2015). Due to their size, large carp can penetrate up to 12 cm into the substrate, allowing them to stir up significant quantities of nutrients (Weber & Brown 2009). A study by Huser et al. (2015), which studied carp bioturbation in a shallow, Midwestern lake that was undergoing alum treatment, found that carp increased the amount of mobile phosphorus available for release by 55-92%, depending on the location in the lake. Additionally, LaMarra (1975) used enclosures to determine that phosphorus release rates are linearly correlated to the density of the carp, and that a carp population of 200 kg/ha can internally load a lake with 1.07-2.18 mg/m^2/day of TP. The Ramsey-Washington Metro Watershed District (RWMWD 2006) scaled this loading rated based on an estimated density of 40 kg/ha of carp in Phalen Lake, which corresponded to a range of release rates of 0.44 - 0.87 mg TP/m^2/day via bioturbation. Estimates of P loading due to bioturbation from other benthic-dwelling species are lacking. While carp have been observed in Como Lake, their numbers appear to be very low according to the fish surveys and thus, are not likely significantly contributing to bioturbation. While other benthic-dwelling fish are present, and stocked somewhat regularly in Como Lake, we cannot confidently quantify an estimate of P loading from this source category with the available information.

Given the presence of benthic-dwelling fish in Como Lake, bioturbation is likely a contributing factor to increased internal phosphorus loading in the lake, but the magnitude is uncertain. Additional investigation would be needed to specifically quantify rates of P loading due to fish bioturbation in Como Lake. However, these estimates, coupled with fish abundance and stocking information, have important implications for improving water quality in Como Lake. Lake management and fish stocking practices should consider species that do not increase phosphorus concentrations in the lake via bioturbation. Since 2004, species that have been stocked in the lake include bluegill, channel catfish, largemouth bass, walleye, and yellow perch (Table 14). Of these, bluegill, channel catfish, and largemouth bass are expected to impact internal TP loading through bioturbation despite the range of uncertainty. The potential to influence sediment phosphorus release should be compared against the goals of the stocking when adding these species to Como Lake in the future.

Table 14. Historical record of the number of fish stocked in Como Lake.

Year	Bluegill	Channel catfish		Largemouth bass		Walleye			Yellow perch	
	Adult	Adult	Yearling	Adult	Yearling	Fry	Yearling	Fingerling	Adult	Yearling
2004	959	-	-	-	-	-	-	-	-	-
2005	-	-	-	-	-	-	-	-	-	-
2006	1,384	-	-	-	-	-	-	-	-	-
2007	284	150	3,864	179	414	-	-	-	-	-
2008	-	155	4,502	-	-	-	-	-	-	-
2009	-	-	3,457	-	-	-	-	-	816	1,200
2010	24	91	3,900	-	-	-	4	-	-	-
2011	-	124	3,900	-	-	-	-	3,593	-	-
2013	-	-	-	-	-	-	-	486	-	-
2014	-	-	130	8	-	71,000	-	4,108	1,211	-
2015	-	181	-	-	-	-	2,840	-	-	-
2016	-	-	-	-	-	-	903	-	-	-
Total	2,651	701	19,753	187	414	71,000	3,747	8,187	2,027	1,200

6.1.2.d Curly-leaf Pondweed Senescence and Harvesting

Macrophytes in the lake possess the potential to contribute large quantities of sediment-derived phosphorus into the water column during senescence, which is the decomposition and leaching of sediment-derived phosphorus (James et al. 2002). Curlyleaf pondweed (CLP; *Potamogeton crispus*) is an invasive macrophyte that can outcompete native vegetation and also decomposes earlier in the summer growing season (mid-late June) than most native vegetation, which typically decompose later in the summer (Cooke et al. 2005; James et al. 2002; Heiskary & Valley 2012). The significance of the timing of this P source in contrast to decay from native macrophytes is that P becomes available for biological uptake much earlier in the season when other conditions are optimal (e.g. temperature and sunlight) for algal growth as opposed to later in the summer growing season when conditions are less ideal for excessive algal growth. CLP is one of the most abundant macrophytes in Como Lake, which has been surveyed to occur in up to 95% of the areas of the lake with vegetation (Table 9). CLP is common in Minnesota, where it grows profusely in the spring and then senesces in mid-late June, thereby contributing to algal blooms in many lakes (James et al. 2002; Heiskary & Valley 2012).

The impact of CLP on water quality varies between lakes, specifically based on the water depth, biomass of CLP, and presence of native vegetation (Heiskary & Valley 2012). It has been estimated that senescing CLP in Kohlman and Keller Lakes (MN), which have abundant macrophyte communities, contributes 2.2 mg TP/m^2/day (RWMWD 2006). Applied to the entire surface area of Como Lake, this translates into an annual loading rate of 505 lbs TP/yr. The littoral zone in Como Lake comprises approximately 94% of the total surface area so this estimate is plausible. An evaluation of the contribution of CLP to a shallow, eutrophic, urban lake in Western Australia determined that the 220,000 kg of CLP, which had an average phosphorus concentration of 6.3 mg P/g of CLP, would release 1.56 mg P/L to the 888,000 m^3 lake (Leoni et al. 2016).

James et al. (2002) directly measured the rates of P leaching from CLP in a laboratory incubation study. They found that rates of P loss from CLP was highest in the first week with 40% of the total loss in the first two days of incubation. They experimentally determined that decomposing CLP leaches phosphorus at a rate of $18.732e^{-0.3935x}$ (mg/m^2/day), where x represents the number of days of decomposition. They further determined that CLP decomposition accounted for 20% of the internal phosphorus loading during the summer in Half Moon Lake, an urban lake in Wisconsin.

Macrophytes, including CLP, are mechanically harvested in Como Lake for recreational purposes in targeted areas of the lake. If CLP begins to senescence in Como Lake prior to mechanical harvesting of macrophyte biomass, then timing of harvest is an important factor in regulating the mass load of P from CLP senescence. The typical date of CLP senescence in Como Lake is unknown, but if that process occurs in mid-late June as it does in other lakes in the region, then CLP is at least partially senescing prior to mechanical harvesting.

Figure 49 through Figure 52 show TP, SRP, % CLP occurrence and CLP harvest dates through the growing season (May – Sept) of 2013-2016. In all four years, % CLP occurrence was measured on three dates throughout the summer. Additionally, CLP was mechanically harvested between the first and second macrophyte sampling event for each of those years as noted by the vertical grey bars in each figure. In each year, a distinct decrease in % CLP occurrence was observed between the first and second sampling events. Based on the available information, it is uncertain if CLP had begun to senesce prior to mechanical harvesting. It appears as though the decline in % CLP occurrence between the first and second sampling events is a combined function of senescence and mechanical harvesting, which complicates assessment of the potential P load from CLP senescence in Como Lake.
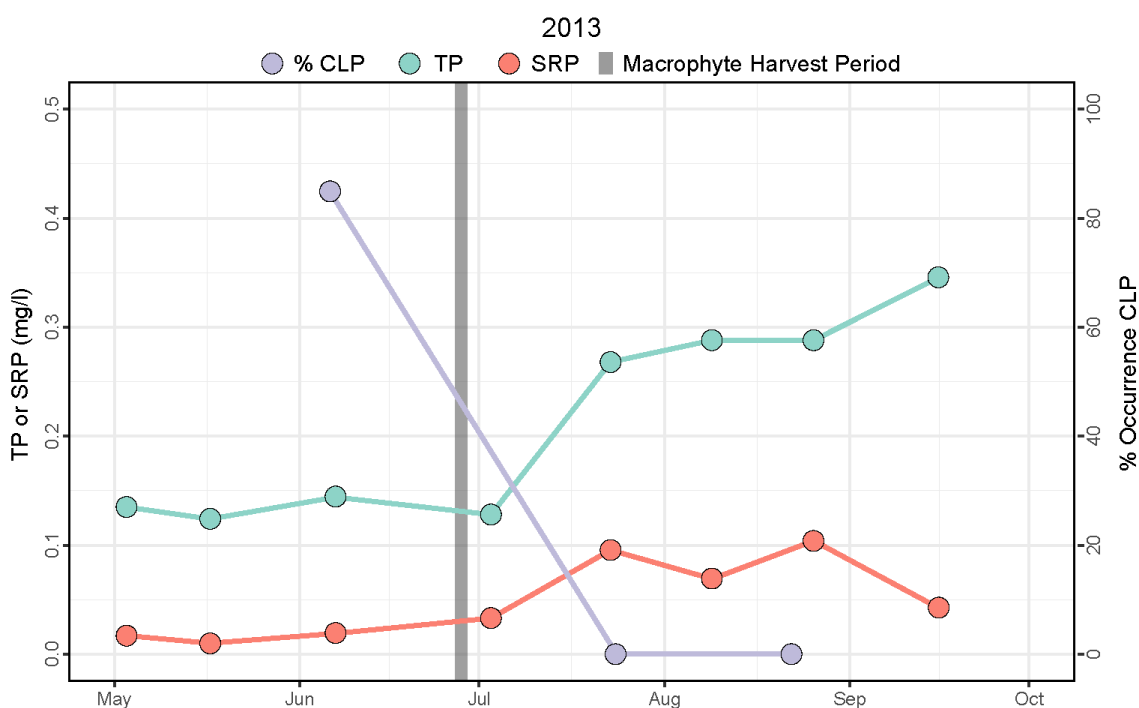
Figure 49. Mean epilimnetic total phosphorus (TP) and soluble reactive phosphorus (SRP), % lake-wide curlyleaf pondweed (CLP) occurrence and macrophyte harvesting period in Como Lake in the summer of 2013.
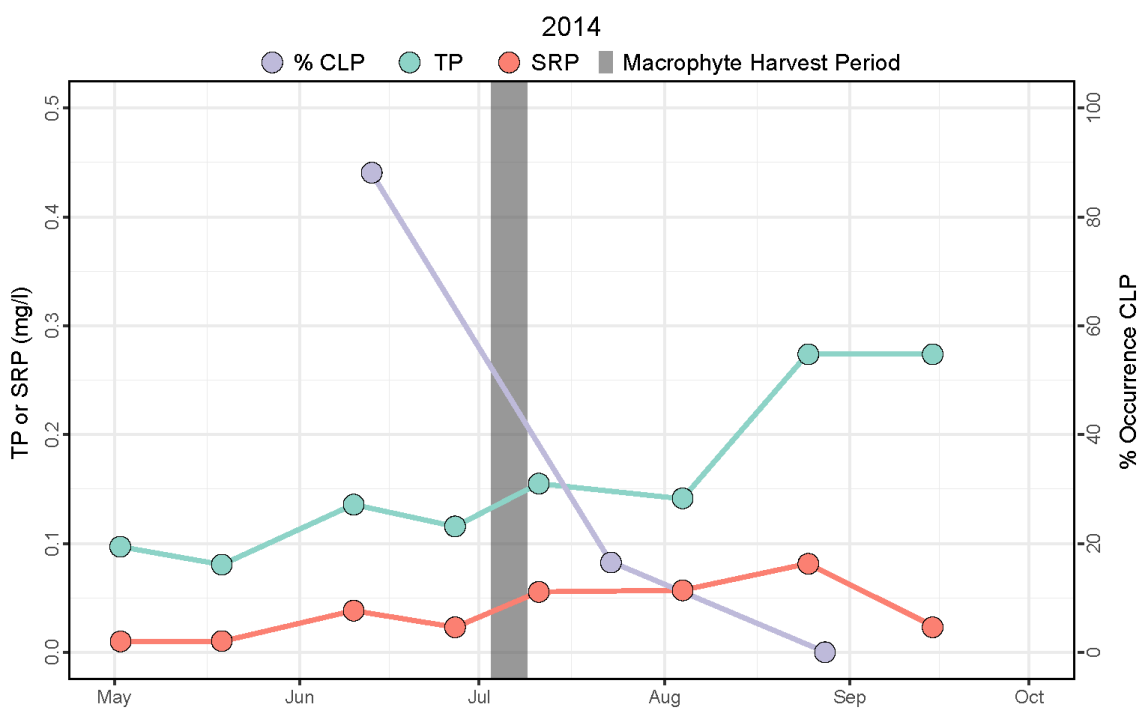
Figure 50. Mean epilimnetic total phosphorus (TP) and soluble reactive phosphorus (SRP), % lake-wide curlyleaf pondweed (CLP) occurrence and macrophyte harvesting period in Como Lake in the summer of 2014.
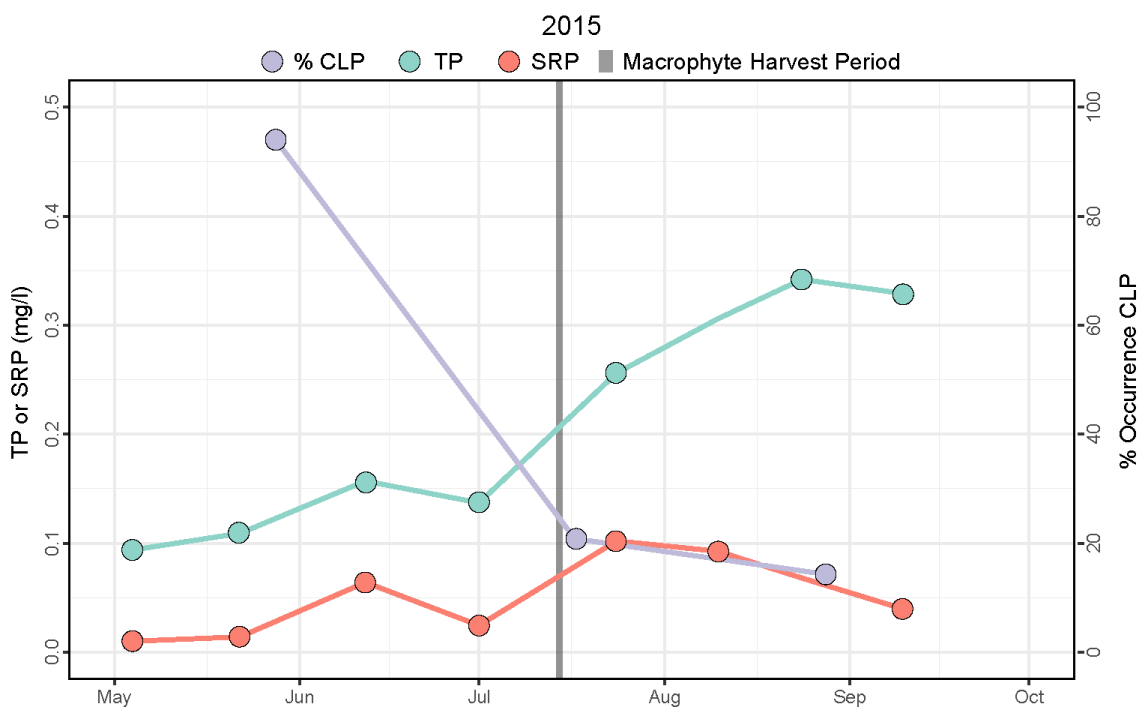
Figure 51. Mean epilimnetic total phosphorus (TP) and soluble reactive phosphorus (SRP), % lake-wide curlyleaf pondweed (CLP) occurrence and macrophyte harvesting period in Como Lake in the summer of 2015.
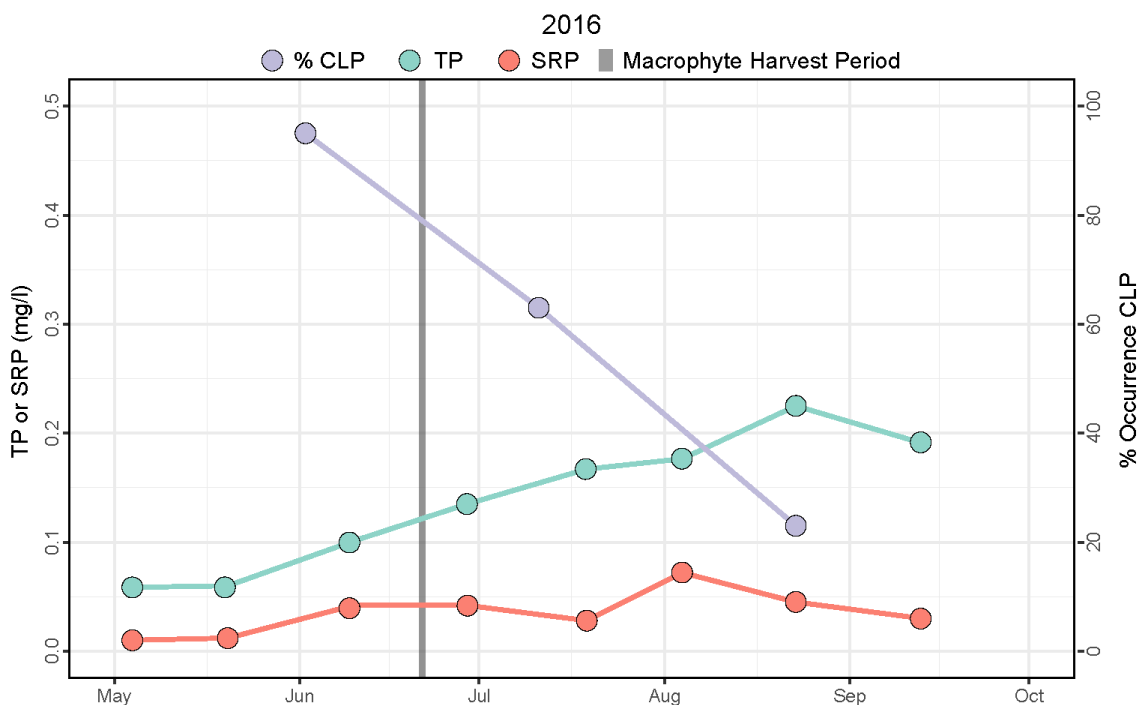
Figure 52. Mean epilimnetic total phosphorus (TP) and soluble reactive phosphorus (SRP), % lake-wide curlyleaf pondweed (CLP) occurrence and macrophyte harvesting period in Como Lake in the summer of 2016.

In some years, there appears to be an increase in TP or SRP coupled with a decrease in % CLP occurrence. For example in 2013, there was a distinct increase in both TP and SRP between the first and second macrophyte/CLP sampling events, suggesting that CLP senescence was a source of P. However, as mentioned above, mechanical harvesting also occurred between those two dates so it is uncertain if the change in CLP occurrence was due primarily to senescence or harvesting. In 2014, there was a sharp increase in TP between the second and third CLP sampling events, which occurred on 7/23/2014 and 8/24/2017, respectively. Both of these dates are beyond the typical period of CLP senescence in this region suggesting that the increase in TP is due to some other mechanism. In 2015, mechanical harvesting occurred just before second sampling event. Consequently, the large decrease in % CLP occurrence in that year appears to be due to senescence which is supported by an associated increase in TP and SRP. In 2016, TP increased while the % occurrence of CLP decreased, but there was not a marked increase in SRP. SRP would be expected to increase with CLP decomposition as this is a primary leachate of the decomposition process suggesting that the TP increase may be due to other mechanisms. Macrophyte harvesting occurred shortly after the first sampling event in 2016 suggesting that the decrease in % CLP was due to mechanical harvest, not senescence. The relationships between TP, SRP and % CLP suggest that CLP is likely contributing P loads to Como Lake, and the data indicate that this load is occurring prior to mechanical harvesting in some years. Macrophyte management strategies in Como Lake should consider the typical CLP senescence date, and manage CLP accordingly to in order to reduce P loading from this source. Additionally, management strategies should target invasive species control, but should include ways to enhance or restore healthy communities of native aquatic vegetation.

6.2 Conclusion

Where possible with available data, annual mass loading estimates were developed for the key source categories contributing phosphorus to Como Lake, with results summarized in Table 15. Note that these estimates do not consider nutrient recycling processes of available phosphorus (e.g. plant uptake or fish excretion), and therefore, represent gross flux estimates from each source mechanism.

Watershed loads based on modeling from a previous study (Houston Engineering 2016) and diffusive flux estimates based on monitoring data calculated in this study provided the greatest amount of certainty with respect to loading rates. However, diffusive flux estimates represent P flux rates that would occur under specific oxygen conditions. The actual rate depends on the temporal and spatial extent of anoxia at the sediment-water interface, which is unknown (see Section 6.1.1). CRWD has installed continuous DO data loggers and will be monitoring DO in the bottoms waters in Summer 2017 to better constrain the extent of anoxia which will be used to refine the diffusive flux estimates in Como Lake.

Considerable uncertainty exists with quantification of P loading rates attributed to bioturbation and macrophyte senescence (i.e. CLP in particular). These source loads are inherently challenging to quantify and literature estimates are significantly limited. Site-specific studies would be required to better constrain P loading rates from these sources. For example, these site-specific studies could include fish enclosure studies coupled with frequent P measurements (e.g. LaMarra 1975 and Huser et al. 2015) or experiments that monitor P accumulation with CLP decomposition (e.g. James et al. 2002). Consequently, we were unable to determine the estimated load from bioturbation with the available information.   We estimated a potential phosphorus loading rate from CLP senescence, but this likely represents an upper bound estimate as it assumes that all phosphorus within the CLP present in the lake is released to the water column through senescence. Mechanical harvesting aims to reduce CLP occurrence prior to senescence although the data show that partial communities of CLP remain after harvesting which is likely a source P upon senescence.

Table 15. Summary of Estimated Phosphorus Loading from Various Source Categories.

Source Category	Estimated Load (lbs. TP/yr)	Basis of Estimate, and Key Assumptions
Watershed (external)	759	Houston Engineering (2016), with loads from Como 3 and Como 7 subwatershed verified in this study
Diffusive Flux (internal)	293-819	This report, which assumes that historical dissolved oxygen profiles from shallow Como Lake stations are representative of current conditions. CRWD is currently measuring continuous DO at multiple stations to better constrain this estimate.
Wind-induced resuspension (internal)	Indeterminate, but likely much less than watershed loading	This report.
Bioturbation (internal)	Indeterminate	This report. Literature estimates on P load from fish are lacking except for carp, which haven't been found in Como Lake in appreciable densities.
Macrophytes (internal)	505	Scaling of RWMWD (2006) rates for Kohlman and Keller Lakes (which are reported to have "abundant" macrophytes) to Como Lake surface area

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7

Summary

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This report presents the results of LimnoTech's analysis of chemical, biological and physical data collected in Como Lake for the period of 1984-2016. The results of this diagnostic study will be used to inform future lake management strategies to improve water quality in Como Lake. A summary of the key findings in our diagnostic study are highlighted below.

Long-term Chemical, Biological and Physical Data Analysis

- A strong oxycline often persists in the absence of a thermocline indicating high hypolimnetic and/or sediment oxygen demand, which emphasizes the need to base hypolimnetic sampling on the oxycline as well as the thermocline.

- The areal extent of hypoxic or anoxic conditions is potentially larger than what is represented by deep station profiles only.

- A significant amount of nitrogen is organically bound in Como Lake as evidenced by the TKN and NH_4 relationships. Consequently, inorganic nitrogen is likely consumed early on in the growing season leading to intermittent conditions where nitrogen is limited. Intermittent nitrogen limitation was further evidenced by periodic TN:TP mass ratios below 10.

- Cladoceran zooplankton density increased substantially over the period of record during the growing season (May – Sept). Cladocerans contain more intracellular phosphorus than other dominant zooplankton taxa such as copepods and are likely playing a major role in internal P cycling in Como Lake.

Planning-level Food Web Analysis

- The aquatic food web in Como Lake is imbalanced likely resulting from a trophic cascade, which seems to be driven by an overabundance of planktivorous fishes.

- The fish community in Como Lake has few top predators, and is dominated by planktivorous forage fish due to low predation pressure. As a result, cascading effects on the zooplankton community can be observed.

- The zooplankton data shows an increasing abundance of cladoceran zooplankton that is dominated by small-bodied taxa. The low density of large-bodied zooplankton is likely the result of intense size-selective feeding pressure by planktivorous fish leading to increased abundance of small-bodied zooplankton. Small-bodied zooplankton are less efficient grazers on phytoplankton than large-bodied zooplankton which allows for increased phytoplankton growth.

Relative Magnitude of Phosphorus Loading from Various Source Mechanisms

External Loads:

- Watershed loads were found to be the dominant source of phosphorus to Como Lake.

Internal Load Mechanisms:

- Release of dissolved phosphorus from lake bottom sediments was also a primary driver of phosphorus loading.

- Bioturbation of bottom sediments due to spawning and bottom-feeding activity of fish contribute phosphorus loading to the lake. However, a large amount of uncertainty exists due to lack of literature estimates for abundant benthivorous fish species (e.g. bullheads). These fish are known to disturb sediments, and while some studies suggest magnitude of disturbance similar to carp, we were unable to better constrain phosphorus load estimates with available information.

- Senescence of macrophytes, mainly CLP, contribute phosphorus loading to the lake, potentially at levels similar to sediment dissolved phosphorus flux. However, a large amount of uncertainty exists in the loading estimate due to unknown volume of CLP that may senescence prior to mechanical harvesting.

In summary, water quality conditions in Como Lake remain poor, primarily due to high phosphorus loading (external and internal) and excessive algal growth. This condition combined with a forage fish/planktivorous dominated fishery, suggests a need for in-lake management strategies that aim to reduce phosphorus loading, but also focus on rebuilding the piscivorous fish community in Como Lake to establish a more stable food web. It is likely that the ecosystem within this popular lake will not rebound without a holistic, active, and adaptive management strategy. CRWD is in a unique position to have an abundant dataset from which to build a well-designed plan to initiate specific actions. The results of this study have provided a thorough diagnosis of major drivers of water quality problems in Como Lake, which lays the foundation for a successful in-lake management strategy. A few recommendations for consideration in future lake management planning is presented in the following section.

8

Recommendations

From the data analysis, we revealed some opportunities to improve understanding of the nutrient dynamics and uncertainty with loads from various sources. To better constrain some of the uncertainty, we have highlighted some recommendations below for data collection and management.

Recommendations for Additional Data Collection and Monitoring

- Reinstate sampling from shallow stations.

  - The historical data indicates some unique properties of Station 202 including higher SRP in some years (despite lack of statistical power and significance).

  - DO should be collected at shallow stations to better understand the spatial and temporal extent of anoxia under current conditions (*Note that continuous DO is being measured at all three stations in Summer 2017*)

- Hypolimnetic sample collection should be based on the oxycline as well as the thermocline. As shown above, anoxic conditions can persist in the absence of a thermocline indicating that chemical reactions are occurring in the bottom waters regardless of thermal stratification.

- Collection of particulate matter and organic particulate matter to constrain the proportion of organic to inorganic particulate matter, which can inform whether the increased turbidity in recent years is derived from organic or inorganic sources(Section 4.2.2) .

- Collection of hypolimnetic chloride samples to better understand current conditions with chloride accumulation in the hypolimnion.

- Watershed nitrogen loads should be calculated using existing stormwater monitoring data to better understand mass loading of nitrogen to Como Lake.

- Site-specific determination of CLP senescence dates.

- Laboratory incubations of CLP decomposition to quantify the P release rate.

- Fish enclosure experiments to quantify P loading from benthic-dwelling fish.

- Fish sampling techniques that permit more reliable sampling for carp (i.e. electrofishing).

Recommendations for Management

Due to the long history of nutrient loading to Como Lake, and the complexity of causal factors, improvements in water quality will be long-term and costly. There is not a one-size-fits-all approach to remedy Como Lake. Rather, a holistic approach that integrates multiple strategies along with adaptive management will be required to improve water quality conditions in Como Lake. A "healthy" Como Lake will have reduced nutrient loading (external and internal) to the extent that the lake meets water quality standards; it will be free of dense floating algae; possess water clear enough to support native macrophyte vegetation that can take up nutrients, stabilize sediments and provide refugia for zooplankton and fish; and it will support a balanced aquatic food web.

Moving forward, we recommend that the following actions be investigated for future management.

o **_Holistic management:_** Como Lake needs to be managed holistically for nutrient reductions and ecological health, and thus, water quality improvements will be realized. Continued management of major aspects (e.g. macrophytes, fisheries, stormwater, etc) independent of each other rather than holistically, the lake will remain ecologically imbalanced. A stable shallow lake cannot thrive in the absence of macrophytes or algae. Lakes require a source of energy through photosynthesis. In the clear state, a healthy shallow lake has abundant macrophytes that provide that energy, stabilize sediments, and provide habitat for zooplankton and fish. Without macrophytes, a lake will turn towards the turbid state characterized by dense algal communities with little vegetative cover. Although difficult to control, invasive aquatic vegetation also provides this ecological function. Invasive plant management also needs to include strategies that enhance development of native macrophyte communities. Similarly, management of the fisheries in Como Lake needs to consider overall ecological health of the lake and impacts to the zooplankton community (i.e. trophic cascade). The aquatic food web in Como Lake is imbalanced and should be restored as part of a holistic lake management strategy, which could include the following approaches:

o **_Manage anoxia:_** Future management strategies should consider actions to reduce anoxic conditions (temporally and spatially) in the lake. Phosphorus is released from the sediments under anoxic conditions, so reducing anoxia will also minimize this source of internal loading.

o **_Alum treatment:_** Alum could be beneficial if used in conjunction with ongoing efforts to reduce external loading. An alum treatment could provide a jump start to improvement in water quality conditions by significantly reducing internal phosphorus, however this should not be considered as a single remedy.

o **_Macrophyte management:_** Evaluate strategies to effectuate long-term control of curlyleaf pondweed, which could include herbicidal treatment and/or adjustment to timing of mechanical harvesting to occur prior to senescence. Establish healthy native macrophyte community to provide refugia for fish and zooplankton, stabilize sediments, and nutrient uptake.

o **_Biomanipulation:_** Biomanipulation has potential to improve water quality conditions in Como Lake if used in conjunction with ongoing efforts to reduce external loading and other appropriate strategies. Recommended approaches for biomanipulation include the following:

  ▪ Remove bullheads, and reduce populations of benthivorous fish and planktivorous fish.

  ▪ Avoid stocking young-of-the-year planktivorous fish until Como Lake is restored, then re-evaluate fisheries management based on ecological health and long-term goals.

  ▪ Introduce piscivorous fish species that will control the planktivorous fish that are consuming large-bodied zooplankton species. Restoring balance to the zooplankton community will increase grazing pressure on the algae population.

  ▪ Stock with piscivorous fish tolerant of current conditions (i.e. low dissolved oxygen) in Como Lake unless other improvements are made.

  ▪ Re-establish native macrophyte with removal of curlyleaf pondweed. Cease removal of native macrophytes for recreational purposes and target invasive species control. The data suggest that current macrophyte harvesting strategies are providing short-term control, so we recommend re-evaluation of management strategies for long-term success.

o **_External Watershed Controls:_** Continue ongoing efforts to reduce external nutrient loading through watershed controls.

9
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Appendix A: Dissolved oxygen and temperature profiles (2010-2015)
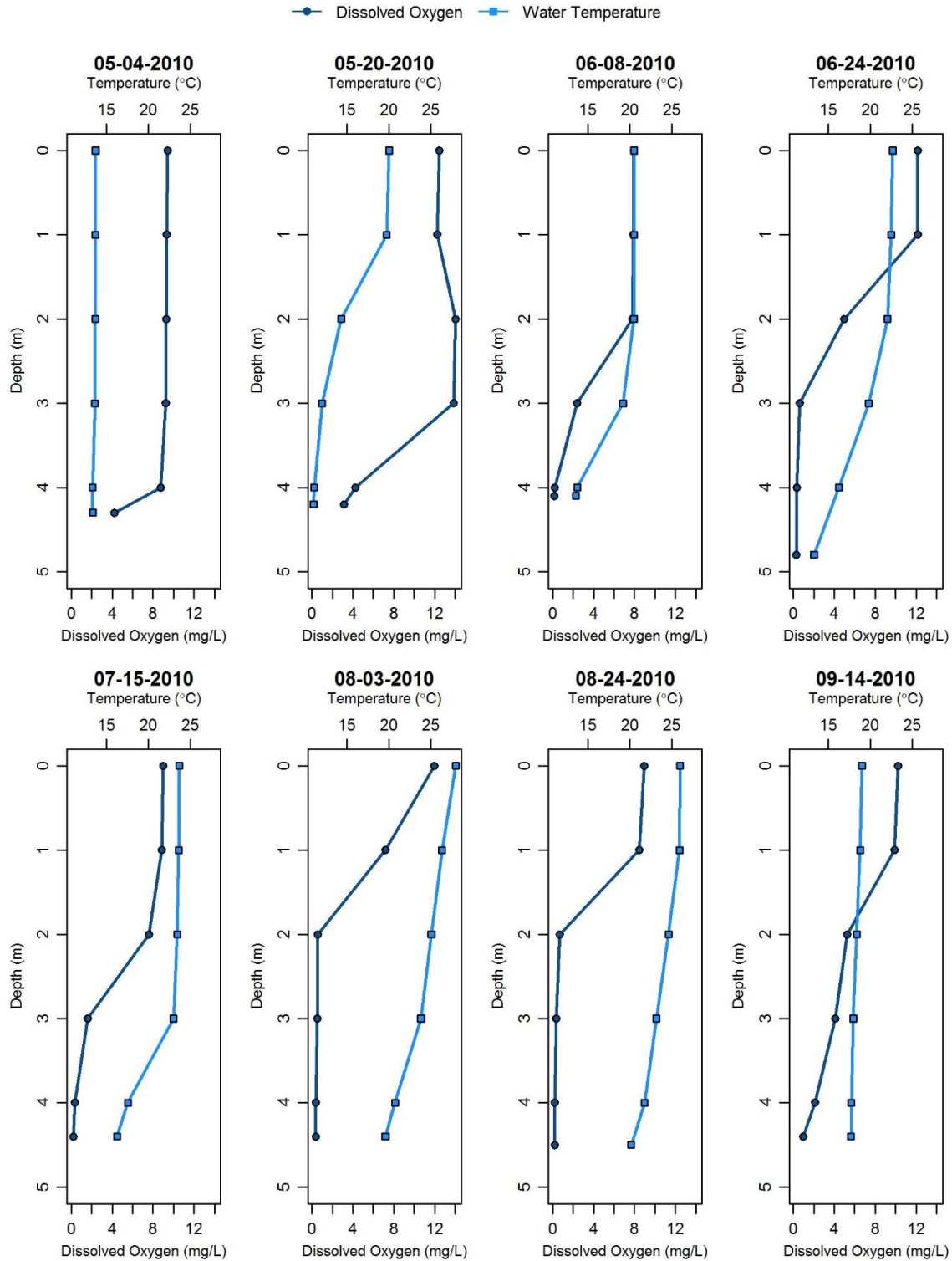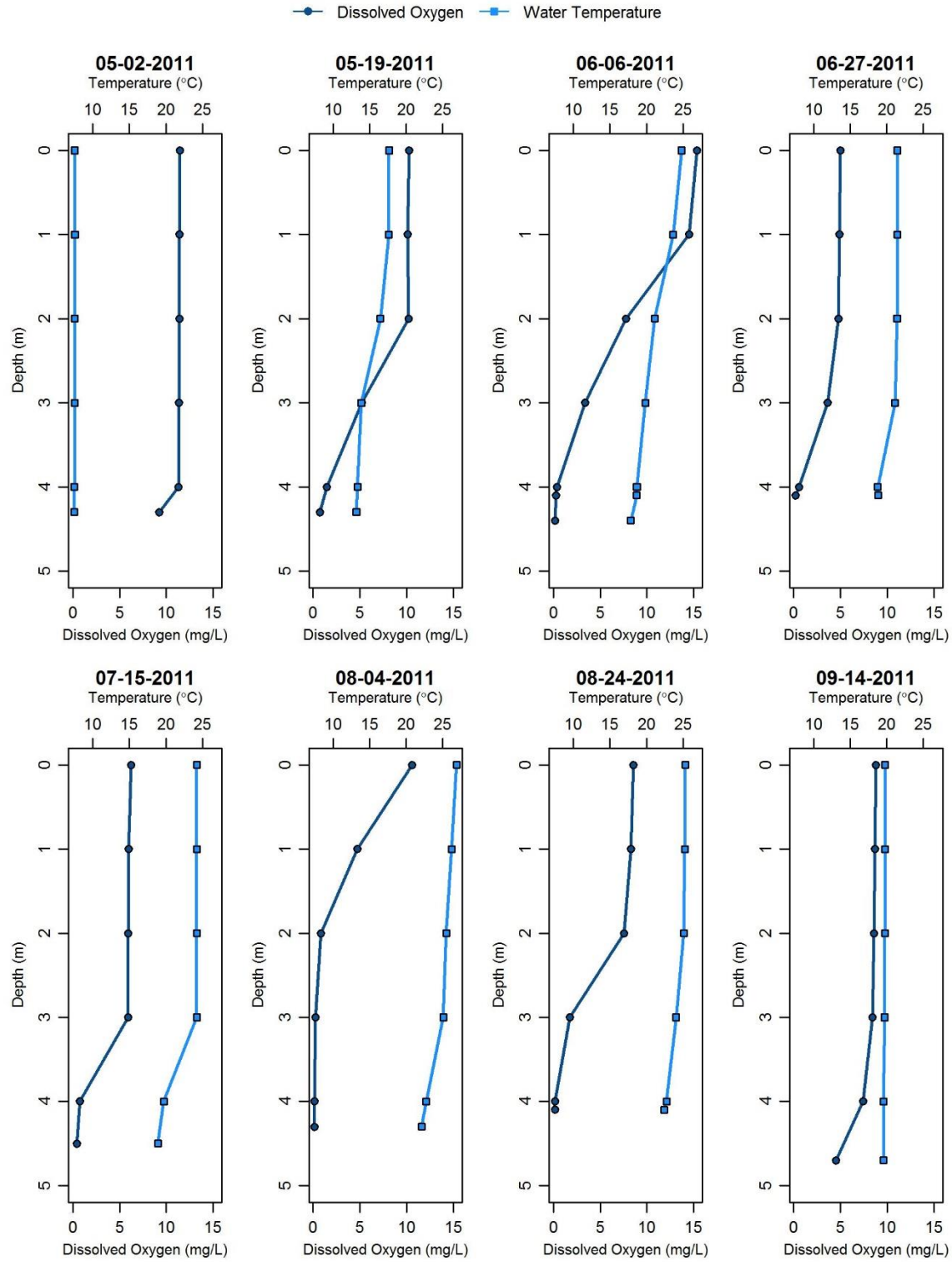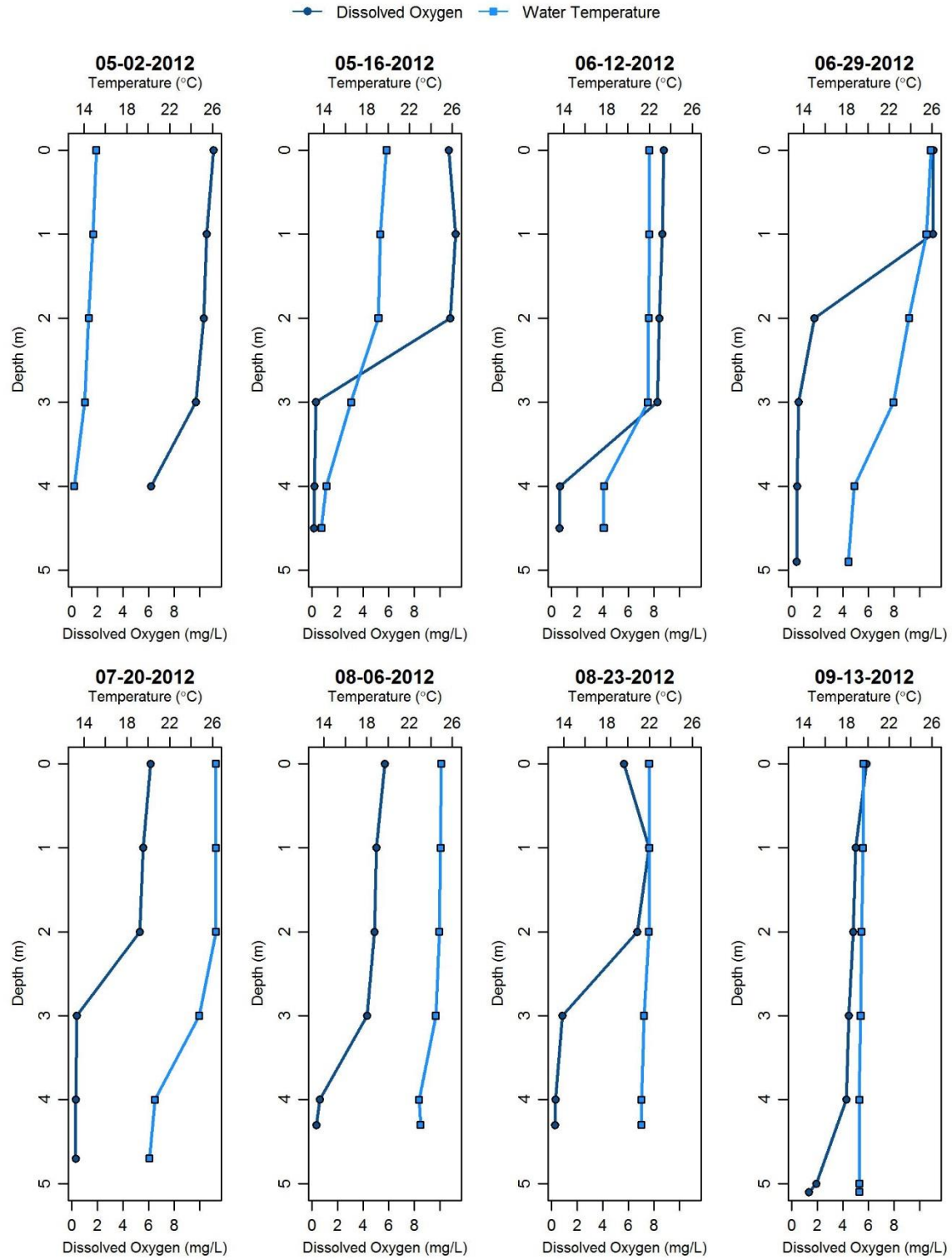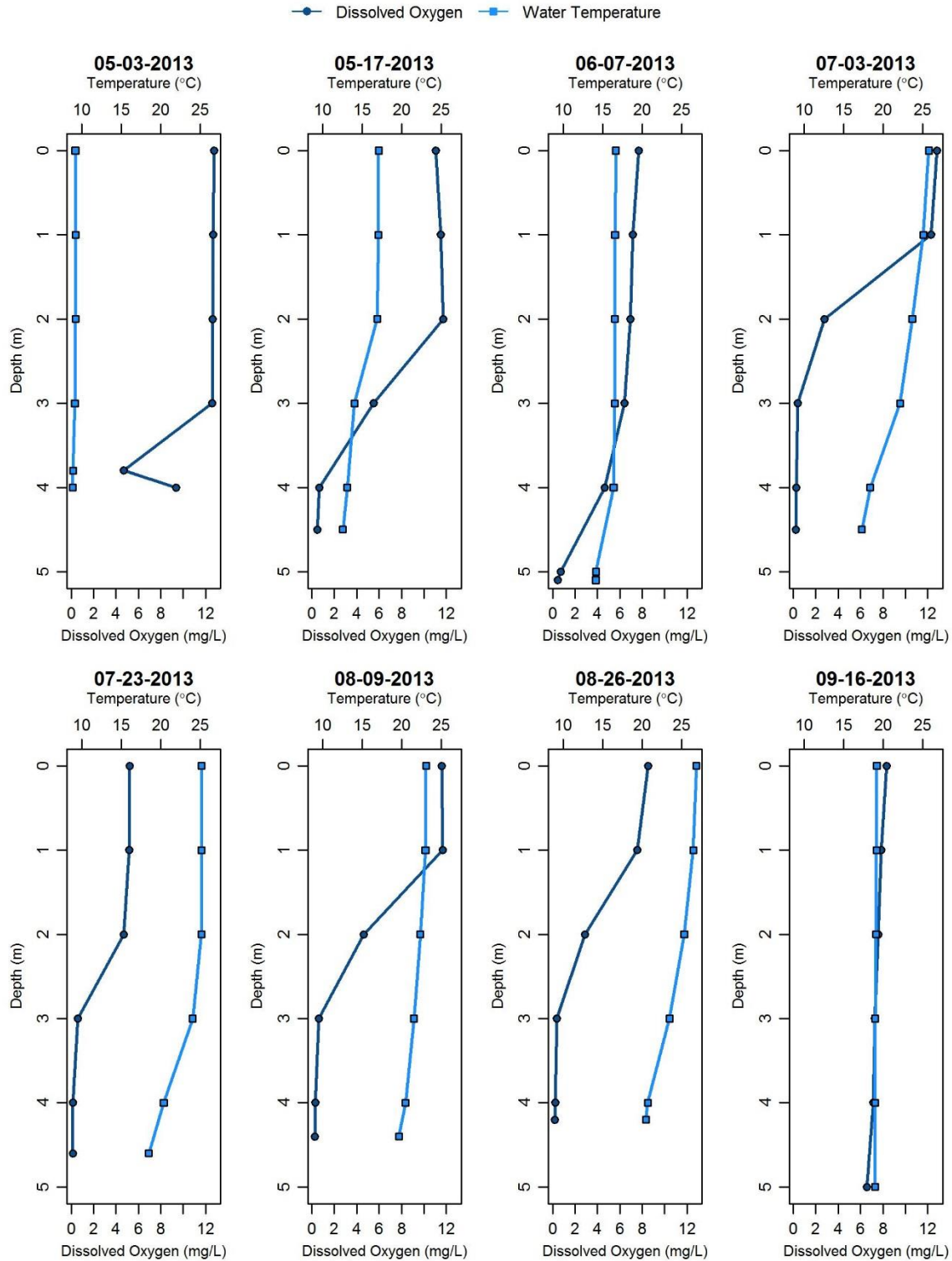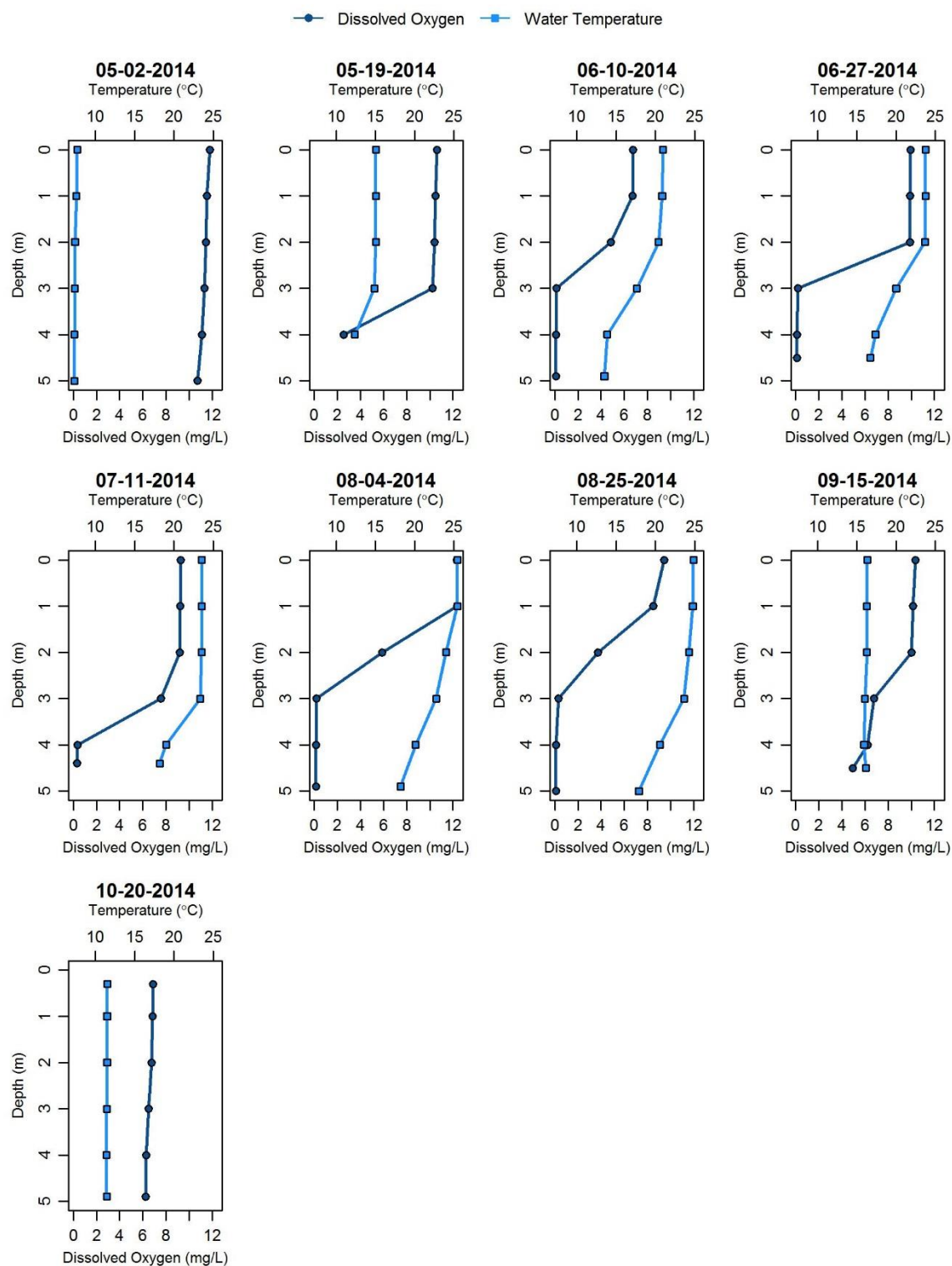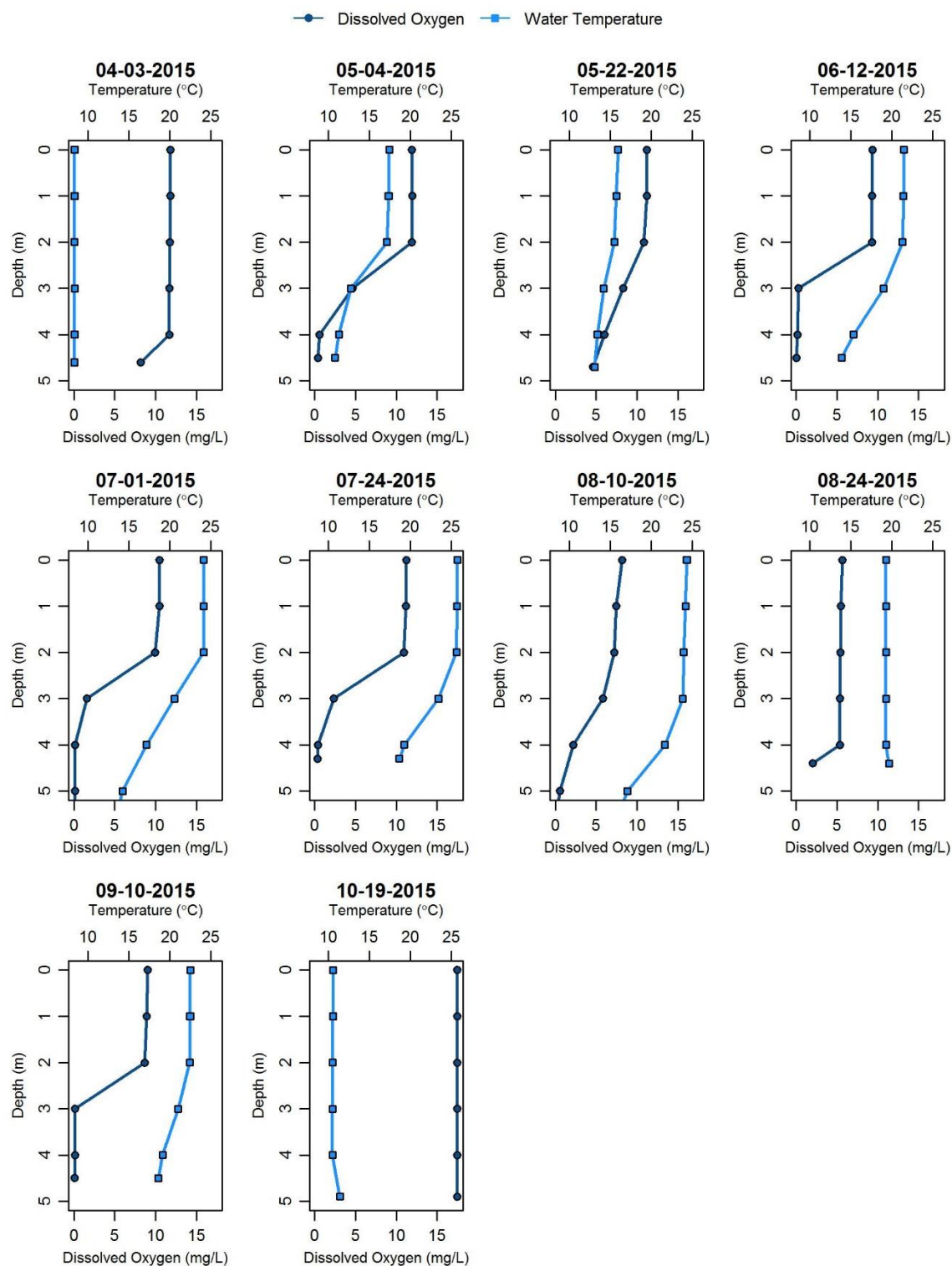
Appendix B: Trends in Non-Eutrophication Parameters

Conductivity is a measure of dissolved inorganic ions in water, which primarily include calcium, magnesium, sodium, potassium, bicarbonate, sulfate and chloride (Wetzel 2001; MPCA 2010). Ions present in watershed runoff significantly influence conductivity in lakes, and the range of values are indicative of water quality (Horne and Goldman 1994). In Minnesota, conductivity greater than 400 µS/cm is considered high (MPCA 2010), and conductivity measurements greater than 1000 µS/cm approaches saline conditions. These high values can have significant implications for less tolerant aquatic organisms thereby influencing biological community structure and diversity. Mean epilimnetic and hypolimnetic conductivity in Como Lake was 436.5 µS/cm ± 187.47 SD, and 585.1 µS/cm ±193.23 SD, respectively (Table 2). These values are in the high range for Minnesota lakes and should be taken into consideration for biological community management.

Conductivity and chloride were highly variable over the period of record and appear to show correspondent patterns in magnitude and range (Figure B1 and Figure B2, respectively) and were highly correlated (Table 2). This relationship would be expected in regions where winter road salt applications are common. Over the period of record, epilimnetic and hypolimnetic conductivity and chloride concentrations during the growing season have increased significantly (Table 4). However, few hypolimnetic chloride samples were reported in the last decade (Table 2 and Figure B2). Chloride is a conservative constituent and is not processed through biological activity. Consequently, chloride tends to accumulate in the hypolimnion of lakes creating bottom waters with density gradients that over time can prevent whole-lake mixing (Novotny et al. 2008), which is referred to as meromictic (Wetzel 2001). Lack of whole-lake mixing would further exacerbate oxygen-depleted conditions in the hypolimnion. The water quality standard for chloride in Minnesota lakes is 230 mg/L for chronic exposure and 860 mg/L for acute exposure (MN Statute 7050.0222). Mean epilimnetic and hypolimnetic chloride in Como Lake was 103.9 ± 52.21 SD and 127.3 ± 49.51 SD, respectively, which does not exceed state water quality standards. These values are on the high end and should be considered in lake management strategies. However, recent hypolimnetic chloride data is lacking (Figure), and would be needed to evaluate current conditions in Como Lake.
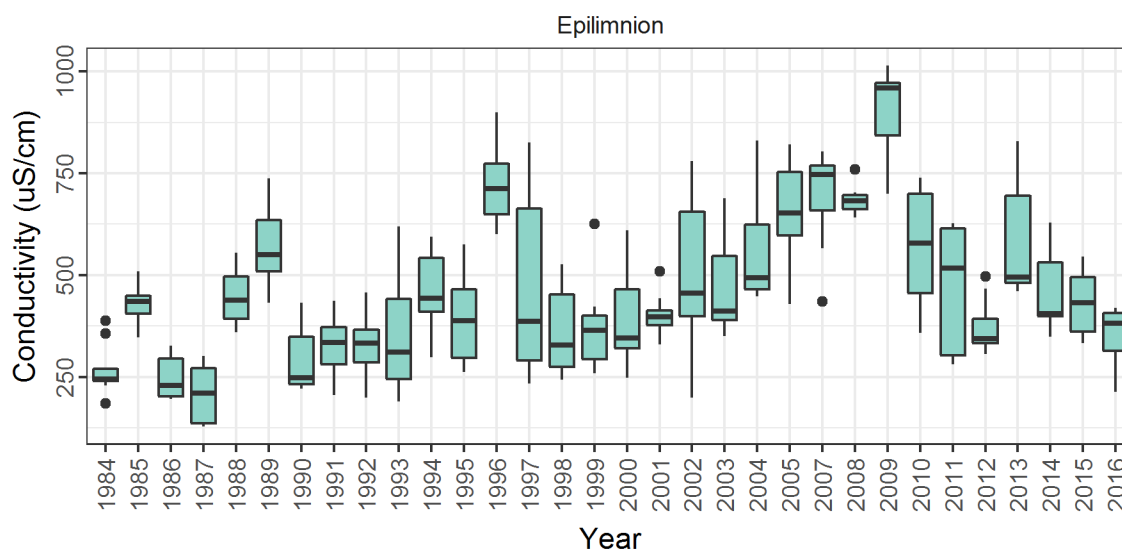
Figure B1. Inter-annual variability in conductivity during the growing season (May – Sept) at Station 201.

Figure B2. Inter-annual variability in epilimnetic and hypolimnetic chloride during the growing season (May – Sept) at Station 201.

No significant trend in epilimnetic or hypolimnetic median temperature during the growing season was found (Table 4; Figure B3). On a global basis, however, surface water temperatures have increased significantly for lakes in the mid to high-latitudes of the northern hemisphere since 1985 (Schneider and Hook 2010). While median temperatures have not changed significantly in Como Lake, other climate induced impacts are expected for the Upper Midwest, which have major implications for lake productivity and water quality. These implications include longer ice-free periods with extended thermal stratification and summer growing seasons (i.e. strength of the thermocline), and changes to the mixing regime (Adrian et al. 2009; Jacobson et al. 2010; Jiang et al. 2012; Butcher et al. 2015). These factors can have

implications for DO, biological activity and community structure in lakes and thus should be considered for in-lake management of Como Lake.
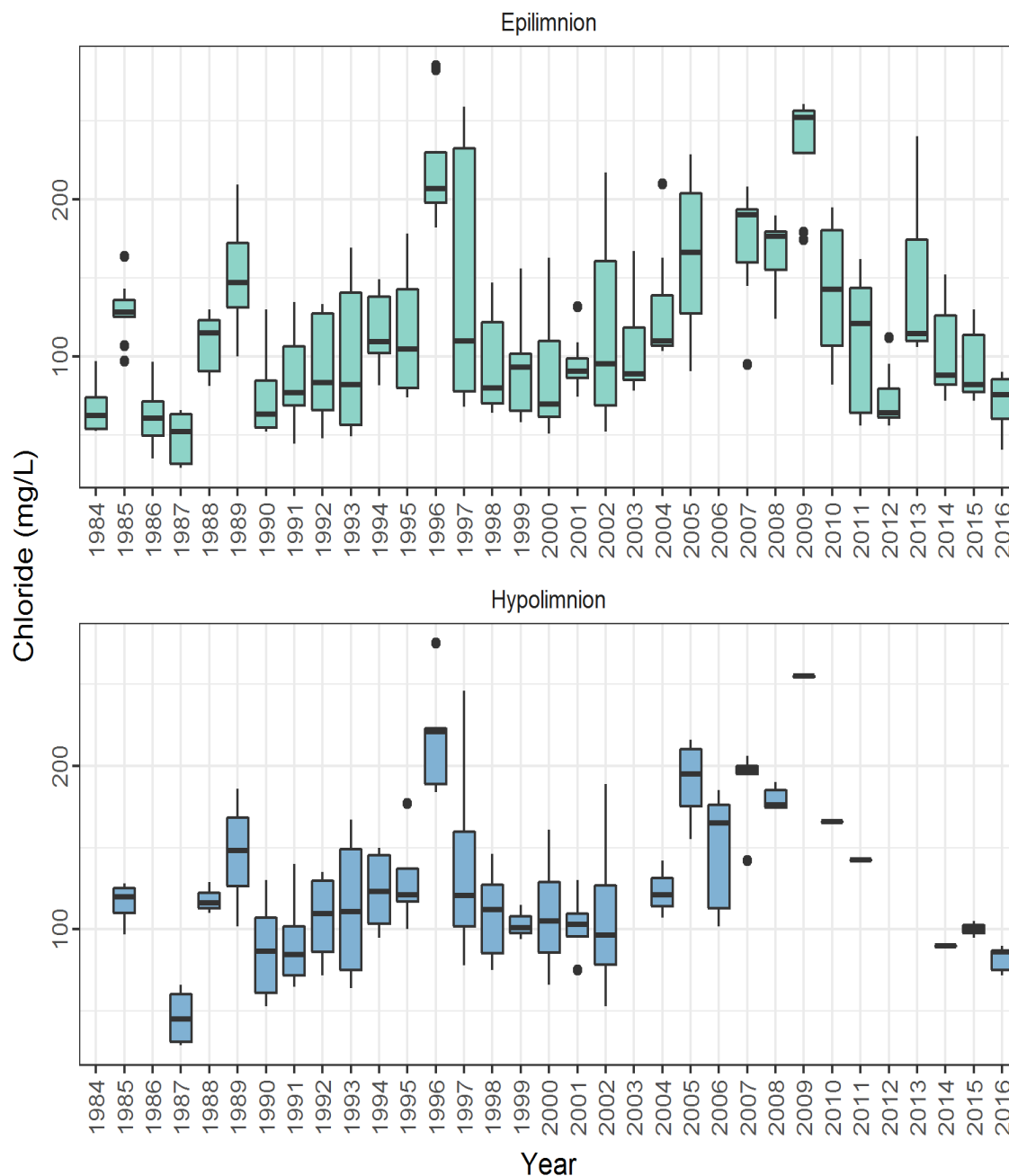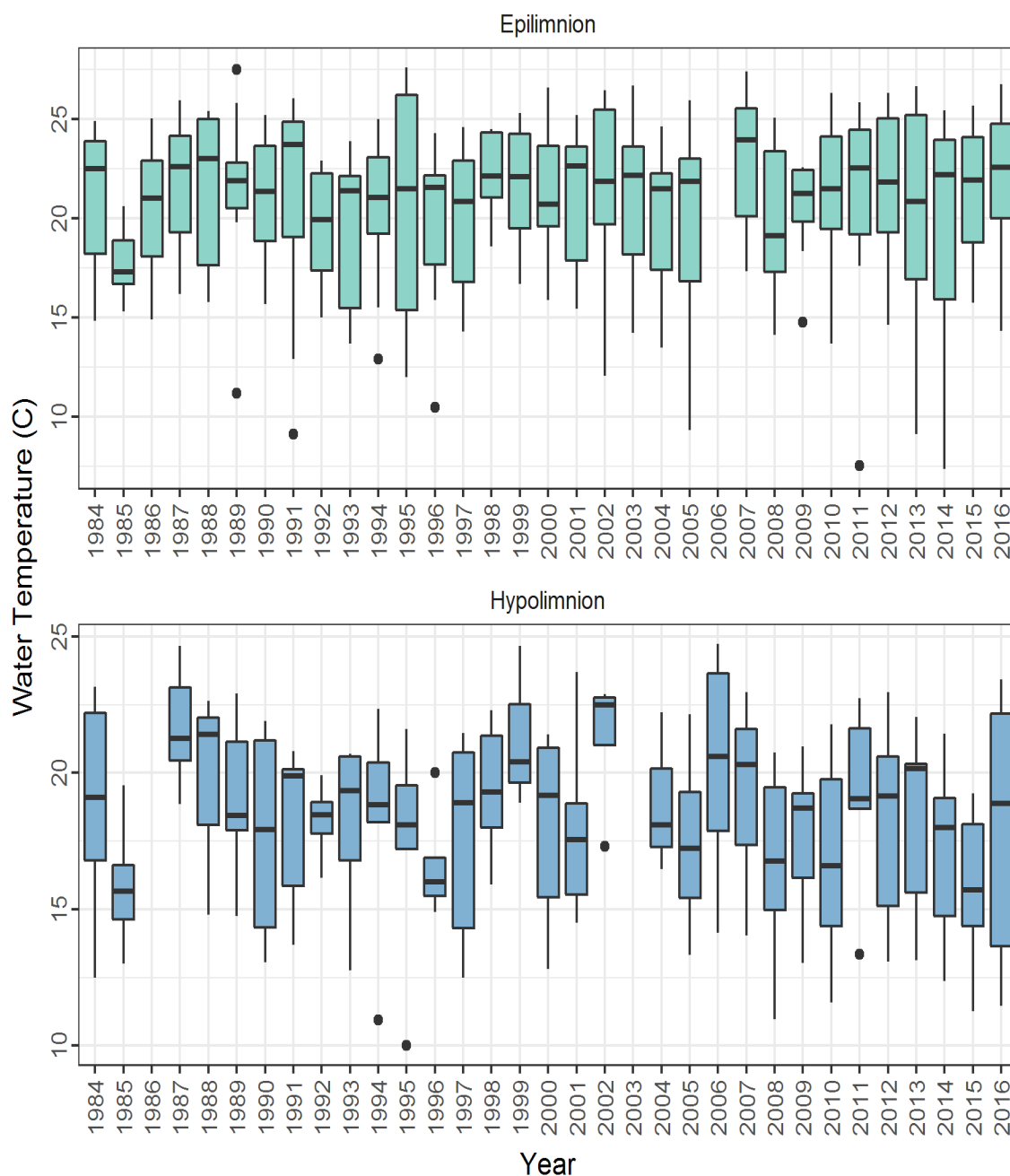

Figure B3. Inter-annual variability in epilimnetic and hypolimnetic water temperature during the growing season (May – Sept) at Station 201. Epilimnetic temperature represents surface measurements only (0 – 1 m).

Median water surface elevations in Como Lake have oscillated around the Minnesota DNR-defined ordinary high water level elevation of 881.24 ft (Figure B4), which is a function of the control weir at the outlet of Como Lake. TP, Chl-a, TKN, and turbidity, were positively correlated to mean daily water level indicating that these parameters are higher with higher water levels during the growing season (May – Sept). Conductivity, chloride and Secchi depth were negatively correlated with mean daily water level indicating that concentrations are typically lower with higher water levels. Precipitation was not significantly correlated to any parameter listed in Table 5, but this may be because growing season conditions were compared and the test would not capture early spring precipitation and associated flushes prior to May. Water level, which is regulated by precipitation, clearly impacts water quality variables. Cumulatively, these significant correlations indicate that the hydrology and hydraulics are drivers of water quality in Como Lake (Table 6).
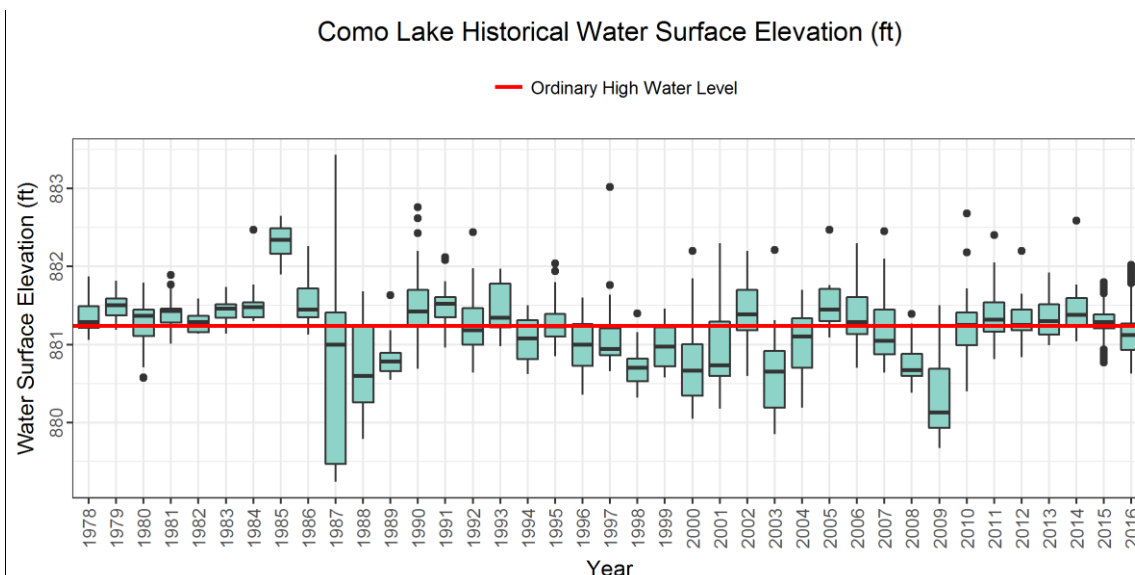
Figure B4. Inter-annual fluctuation in water surface elevation during the growing season (May – Sept). Red horizontal line represents Ordinary High Water level (OHW: 881.24 ft.)

Cumulative annual precipitation measured at the University of Minnesota showed inter-annual variability for the available data record (2005-2016) with precipitation for most years above the 30-year average (Figure B5). The red horizontal line represents the 30-year precipitation average (30.61 inches) for this region. As stated above, no significant correlation was found between cumulative daily precipitation and environmental variables for the growing season during the period of record (Table 6).
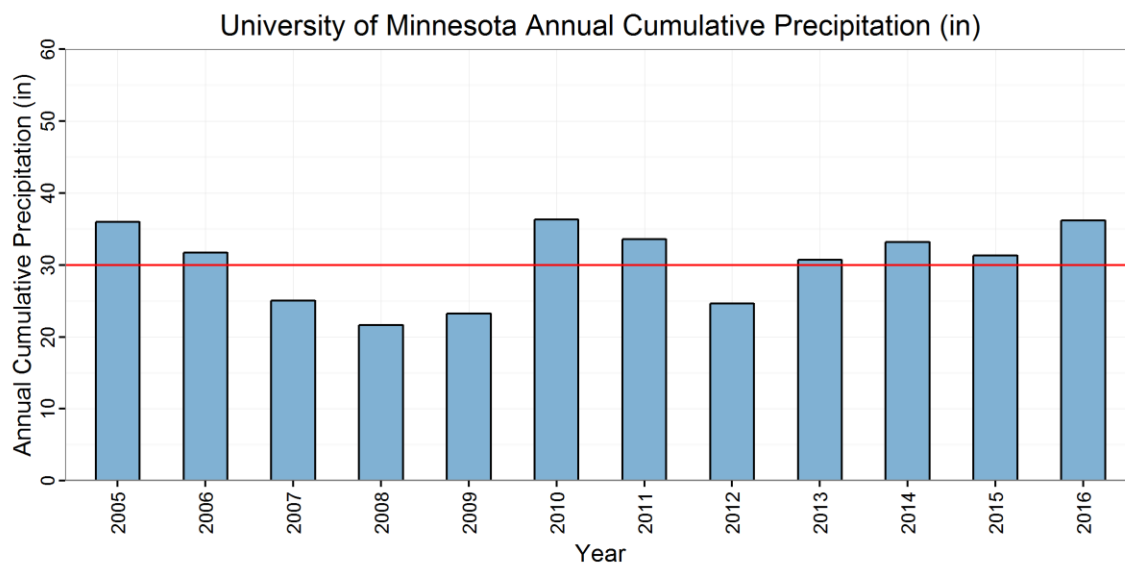
Figure B5. Annual cumulative precipitation from the University of Minnesota 2005-2016. Red horizontal line represents 30-year normal (1981-2010) as reported by NOAA (2017).

Appendix C: Growing Season (May-Sept) abundance of major phytoplankton and zooplankton taxa in Como Lake

Table C1. Annual abundance of phytoplankton taxa.

Year	Site	Phytoplankton Abundance						
		Blue-Green	Cryptomonad	Diatom	Dinoflagellate	Euglenoid	Green	Yellow-Green
1984	201	72.29	0	12.91	0.04	0.12	14.64	NA
1985	201	71.66	0.49	6.41	NA	0.02	21.25	0.17
1986	201	77.87	5.8	0.96	NA	0.1	15.27	NA
1987	201	13.84	35.77	1.59	0.18	0.07	48.41	0.13
1988	201	83.66	4.91	0.12	0.04	0.04	11.23	NA
1989	201	68.85	8.97	6.21	0.4	0.08	8.25	7.24
1990	201	49.29	27.73	0.3	0.11	0.16	22.41	NA
1991	201	7.63	20.18	10.52	NA	0.05	61.62	NA
1992	201	21.79	12.18	4.48	1.28	0.28	59.99	NA
1993	201	50.35	21.29	16.17	1.05	0.19	10.95	NA
1994	201	78.51	9.34	NA	0.16	NA	12	NA
1995	201	97.84	1.81	0.05	0.04	NA	0.26	0.01
1996	201	74.17	3.78	1.09	0.67	NA	20.29	NA
1997	201	78.08	13.73	2.22	0.1	NA	5.87	NA
1998	201	47.33	47.06	0.5	0.06	0.21	4.81	0.03
1999	201	24.18	68.89	0.08	NA	NA	6.85	NA
2000	201	64.21	24.07	0.3	0.01	0.16	10.62	0.64
2001	201	70.33	9.68	0.77	NA	0.44	18.63	0.15
2002	201	72.93	12.38	4.91	NA	0.34	9.44	NA
2003	201	23.33	25.03	0.49	NA	1.72	49.2	0.23
2004	201	50.78	40.54	0.2	NA	0.06	8.41	NA
2005	201	56.33	24.63	2.17	NA	NA	16.87	NA
2006	201	80.69	5.33	0.55	NA	0.24	13.19	NA
2007	201	47.08	7.94	1.63	NA	NA	43.27	0.08
2008	201	34.49	12.37	1.32	0.22	NA	51.46	0.13
2009	201	45.81	25.56	1.81	0.08	5.26	21.26	0.23
2010	201	58.88	7.87	2.33	NA	18.41	12.03	0.47
2011	201	87.5	8.62	1.95	NA	0.04	1.89	NA
2012	201	47.68	3.74	2.77	NA	NA	45.46	0.34
2013	201	26.41	30.54	8.6	NA	NA	32.41	2.04
2014	201	54.02	3.7	0.22	NA	0.21	41.86	NA
2015	201	62.81	16.75	5.29	0.62	NA	14.34	0.19
2016	201	46.25	2.67	0.82	NA	NA	48.06	2.19

Table C2. Annual abundance of zooplankton taxa

Year	Site	Zooplankton Relative Abundance										
		Bosminae	Calanoid	Ceriodaphnia	Chydorus	Cyclopoid	Daphnia	Diaphanosoma	Leptodora	Naupli	Rotifer	Unknown
1984	201	11.95	1.33	NA	NA	3.89	0.13	0.63	NA	16.74	65.32	NA
1985	201	4.65	4.53	NA	0.44	4.84	0.26	1.39	NA	26.6	57.26	0.03
1986	201	25.98	0.47	NA	0.09	8.45	10.92	5.14	NA	27.5	21.47	NA
1987	201	29.25	7.7	NA	0.14	12.11	12.54	7.39	NA	27.72	3.15	NA
1988	201	9.4	2.58	NA	0.57	10.95	20.98	1.12	NA	29.35	25.04	NA
1989	201	9.58	10.08	0.66	2.27	8.74	3.25	7.98	NA	25.73	31.71	NA
1990	201	21.37	0.98	2.62	16.28	15.87	2.81	0.95	NA	17.13	22	NA
1991	201	35.29	0.55	4.3	11.3	9.34	1.54	1.44	0.03	13.86	22.32	0.03
1992	201	31.66	1.8	0.81	10.25	7.56	4.32	2.45	NA	28.15	13	NA
1993	201	8.22	3.47	0.19	13.26	10.22	19.71	0.04	NA	31.47	13.42	NA
1994	201	8.85	3.39	3.99	12.03	17.18	17.15	0.92	NA	30.39	6.1	NA
1995	201	14.65	1.14	0.83	18.51	13.72	6.73	0.33	NA	38.13	5.96	NA
1996	201	6.33	0.36	4.63	11	15.94	9.81	NA	NA	19.95	31.98	NA
1997	201	40.24	1.09	2.61	0.08	25.8	9.04	0.22	NA	12.98	7.95	NA
1998	201	10.42	7.33	0.4	0.08	17.9	13.99	4.33	NA	36.96	8.59	NA
1999	201	31.49	3.51	7.56	0.02	8.55	2.48	3.72	NA	23.96	18.69	NA
2000	201	31.15	9.82	1.48	0.93	7.79	3.52	0.8	NA	18.52	25.99	NA
2001	201	33.51	0.58	0.16	5.73	9.8	8.6	0.02	NA	25.35	16.25	NA
2002	201	22.42	1.4	0.48	4.1	7.75	7.71	0.49	NA	23.97	31.69	NA
2003	201	47.12	0.21	1.51	0.69	5.02	0.64	1.27	NA	18.53	25	NA
2004	201	22.87	0.08	6.53	0.46	8.31	1.42	1.11	NA	49.24	9.97	NA
2005	201	39.06	3	3.26	4.82	8.41	2.06	1.87	NA	28.08	9.42	NA
2006	201	32.4	1.97	3.08	11.12	5.28	3.36	1.5	0.04	24.38	16.88	NA
2007	201	27.65	3.14	11.88	3.77	5.74	4.17	0.57	NA	25.72	17.37	NA
2008	201	46.28	4.16	4.35	0.3	2.22	6	0.48	NA	36.21	NA	NA
2009	201	30.21	11.32	6.78	3.99	8.82	1.59	1.15	NA	22.08	14.06	NA
2010	201	35.16	1.2	5.88	10.74	13.59	3.24	2.68	NA	19.42	8.08	NA
2011	201	20.78	1.2	1.5	27.37	9.86	22.7	1.01	NA	10.83	4.77	NA
2012	201	34.13	0.96	2.27	31.19	5.97	4.35	4.35	NA	5.2	11.59	NA
2013	201	35.6	1.5	1.98	20.28	3.91	0.8	5.84	NA	6.25	23.84	NA
2014	201	18.54	6.51	1.67	2.93	10.45	19.03	5.23	NA	15.04	20.6	NA
2015	201	24.56	1.41	0.86	37.98	7.09	11.08	1.65	NA	10.83	4.53	NA
2016	201	25.63	0.56	0.86	30.91	6.3	15.38	3.47	NA	7.82	9.09	NA

Appendix D: Implicit Calculation of Anoxic Flux Rate

As described in Chapter 6 of this report, the steps taken to implicitly estimate the anoxic flux rate consisted of:

1. Examining the entire time series of data at station 5401-201
2. Finding the first date that has stratification and bottom DO <= 1, that persists for more than one date
3. Calculating the average hypolimnetic TP for that date
4. Find the last date of stratification corresponding to the event identified in Step 1
5. Calculating the average hypolimnetic TP for that date
6. Calculating the rate of rate of change in TP concentration over time (e.g. [TP2 -TP1]/ΔT)
7. Calculating the volume and bottom surface area of hypolimnion, based upon average reported thermocline depth during stratification period
8. Calculating flux rate as Rate of concentration change x volume ÷ surface area

Table D1 provides the specific calculations for the 19 anoxic stratification events identified. The first column lists the dates of each stratification event, with the second column indicating the duration. The third column lists the average depth of thermocline calculated for the period of stratification, while the fourth column presents the observed rate of rate of change in TP concentration over time. The fourth and fifth columns represent the volume and surface area of the hypolimnion, based upon the depth of thermocline and regressions between volume/area/depth calculated from the digital elevation map of lake bathymetry. The final column provides phosphorus flux rate calculated for each stratification period.

Table D1. Implicit Calculation of Flux Rate for Each Period of Stratification

Period	No. of Days	Thermocline Depth (m)	TP Slope (mg/l/day)	Volume (m³)	Area (m²)	TP Flux Rate (mg/m²/day)
1984-06-14 to 1984-06-29	15	2.5	0.008	43810.125	71248.13	4.92
1987-07-28 to 1987-08-11	14	2.6	0.0293571	37044.79	63801.4	17.05
1989-07-10 to 1989-07-24	14	2.5	0.0005	43810.125	71248.13	0.31
1990-05-15 to 1990-08-02	79	3.5	0.0016709	3912.275	15509.96	0.42
1991-05-21 to 1991-09-03	105	3	0.0006476	16802.8	38130.96	0.29
1992-05-29 to 1992-07-29	61	2.7	0.0024426	31000.478	56763.58	1.33
1994-05-26 to 1994-06-21	26	3	0.006	16802.8	38130.96	2.64
1995-06-20 to 1995-08-17	58	2.05	0.0145172	84224.819	109745.4	11.14
1998-05-05 to 1998-08-14	101	2.5	0.0013267	43810.125	71248.13	0.82
2001-06-15 to 2001-07-26	41	2.5	0.0404146	43810.125	71248.13	24.85
2005-05-26 to 2005-08-01	67	2.5	0.0026418	43810.125	71248.13	1.62
2007-05-03 to 2007-07-30	88	3	0.0018295	16802.8	38130.96	0.81
2008-05-29 to 2008-07-28	60	3	0.0013833	16802.8	38130.96	0.61
2010-06-08 to 2010-08-24	77	2.3	0.0026623	59660.212	87357.4	1.82
2011-05-19 to 2011-08-24	97	2.3	0.0038247	59660.212	87357.4	2.61
2013-05-17 to 2013-08-26	101	2	0.0037574	89820.2	114519.8	2.95
2014-06-10 to 2014-08-25	76	3	0.0057566	16802.8	38130.96	2.54
2015-06-12 to 2015-08-10	59	3	0.0250678	16802.8	38130.96	11.05
2016-06-10 to 2016-08-04	55	2	0.0232909	89820.2	114519.8	18.27
Average						**5.6**

Appendix E: Calculation of Sediment Diffusive Flux Loading

Table E1 provides the background detail of the calculations provided in Table 12 of the report. Annual load (lbs/year) for each station and oxygen status were calculated by multiplying:

No. of days x Flux Rate (mg/m2/d) x Area (m2) x Unit Conversion Factor (2.20E^{-06} lb/mg)

Table E1. Background Detail on Annual Flux Calculations

Station	No. of Days		Flux Rate (mg/m²/d)	Area (m²)	Phosphorus Load (pounds/year)	
	DO < 1 at deepest measurement	DO < 2 at next-to-deepest measurement			DO < 1 at deepest measurement	DO < 2 at next-to-deepest measurement
5401-201 Oxic	144	227	0.2	64818	4	6
5401-201 Anoxic	221	138	6.5	64818	205	128
5404-102 Oxic	170	323	0.2	212482	16	30
5404-102 Anoxic	195	42	6.5	212482	594	128
Total					**819**	**293**