Kentuck Lake

Forest & Vilas Counties, Wisconsin

Advanced Water Quality Study & AIS Monitoring: Final Report

March 2018

Sponsored by:

Kentuck Lake Protection & Rehabilitation District

WDNR Grant Program

LPT-4480-15

Onterra, LLC
815 Prosper Road
De Pere, WI 54115
920.338.8860
www.onterra-eco.com

Onterra LLC
Lake Management Planning

Kentuck Lake

Vilas and Forest Counties, Wisconsin

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Created by: Brenton Butterfield, Tim Hoyman, Paul Garrison, and Eddie Heath
Onterra, LLC
De Pere, WI

Funded by: Kentuck Lake P & R District
Wisconsin Dept. of Natural Resources
(LPT-480-15)

Acknowledgements

This study was truly a team-based project and could not have been completed without the volunteer efforts and input from the following individuals:

Kentuck Lake Protection and Rehabilitation District

Scott Blankmann
Jane Bonkoski
Jack Brah
Kathy Brah
Tom Carlstedt
Joel DeAngelo
Nils Holmgren
Janet Long
Jim Park
Maribeth Park
Kathy Peterson
Tom Peterson
Candy Purdy
Nancy Steenport

Wisconsin Dept. of Natural Resources

Kevin Gauthier
Jim Kreitlow
Scott Van Egeren
Cory McDonald
Greg Sass

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1.0 INTRODUCTION

Kentuck Lake (Forest and Vilas counties) is an approximate 957-acre, eutrophic, polymictic spring lake with a maximum depth of 40 feet and a mean depth of 13 feet (Figure 1 and Map 1). A headwater lake within the Menominee River Basin, the lake possesses a perennial outlet tributary (Kentuck Creek) and a number of intermittent inlet tributaries. While this lake in the past has been classified as a drainage lake, it was found during this study that the inflowing stream is intermittent while the outflowing stream is perennial indicating that groundwater and precipitation are the primary water sources. When the inflowing stream is flowing, it rarely directly enters the lake but instead sheet flows through an adjacent wetland along the lake's northeast side. Kentuck Lake's surficial watershed encompasses approximately 4,842 acres, the majority of which is comprised of intact forests and wetlands. While its watershed includes the Spectacle Creek subwatershed, water from this subwatershed was only observed to flow into Kentuck Lake during the spring snowmelt when water levels in the adjacent wetland to the north were high. Kentuck Lake is currently on the 303(d) list of impaired waters under the Clean Water Act for chlorophyll-*a* concentrations which exceed thresholds for recreational use.

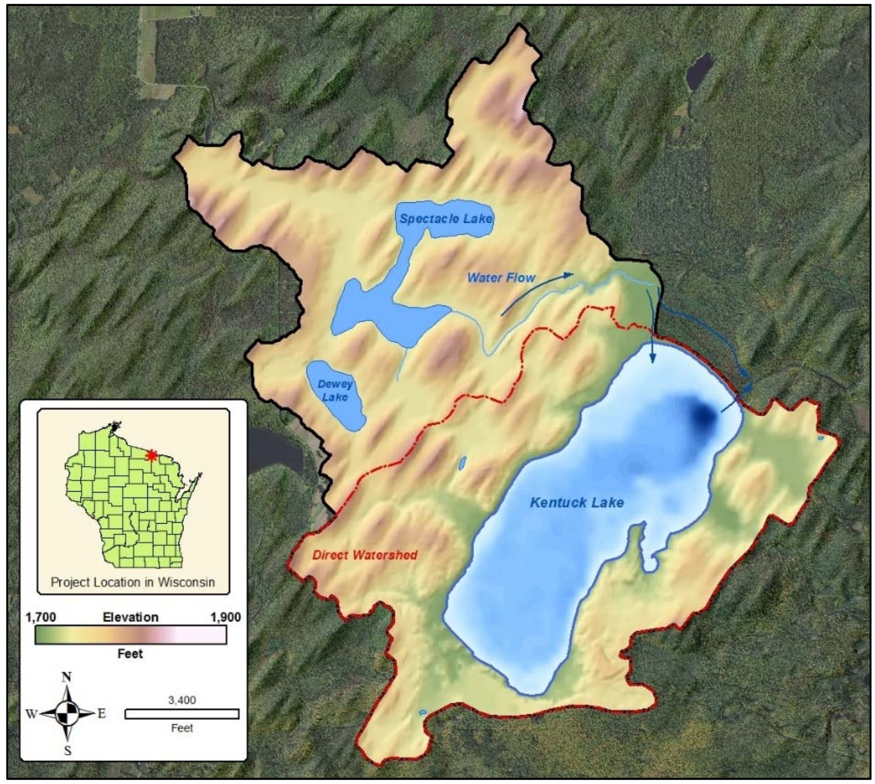

Figure 1. Kentuck Lake and watershed. Black solid line indicates entire drainage area and the red dashed line indicates direct drainage area.

In 2011, average summer near-surface total phosphorus concentrations in Kentuck Lake were found to have increased to 64 µg/L from the 1989-2010 average of 30 µg/L, an increase of 113%. The increase in phosphorus concentrations in 2011 spurred severe cyanobacterial (blue-green algae) blooms, with chlorophyll-*a* concentrations approaching 170 µg/L in late-summer. Because cyanobacteria have the potential to produce harmful cyanotoxins (Sorichetti et al. 2014) and also reduce the lake's aesthetic appeal, members of the Kentuck Lake Protection and Rehabilitation District (KLPRD) expressed great concern about what these blooms signified in terms of Kentuck Lake's ecology and also the threat this change in water quality would pose on the ecosystem services the lake provides. With concerns regarding the sudden change in Kentuck Lake's water quality in addition to a recent discovery of a population of the non-native aquatic plant Eurasian watermilfoil, the KLPRD quickly elected to move forward with the creation of a comprehensive lake management plan.

With the assistance of Onterra, the KLPRD was awarded a Wisconsin Department of Natural Resources (WDNR) Lake Management Planning Grant in 2012 to aid in funding the development of a lake management plan. In 2013 and 2014, Onterra ecologists conducted baseline studies of Kentuck Lake's water quality, watershed, immediate shoreline, and aquatic plant community. Examination of historical annual water quality data available since 1988 revealed that while total phosphorus concentrations in 2011 were significantly higher than the previous 22 years, concentrations were similar to those measured in 1988. These historical data indicated that the conditions that led to the sudden and rapid increase in phosphorus concentrations in 2011 may have occurred in the past.

Near-surface total phosphorus concentrations measured in 2013 were not as high as those measured in 2011, but were still 36% higher than average concentrations measured from 1989-2010 and were high enough to again spur cyanobacterial blooms. Wisconsin Lakes Modeling Suite (WiLMS) was used to predict phosphorus concentrations based upon the size and composition of Kentuck Lake's watershed. This modeling predicted an estimated 484 pounds of phosphorus are externally loaded to Kentuck Lake annually with a predicted in-lake total phosphorus concentration of 16 μg/L. This predicted concentration was half the 1988-2013 measured average total phosphorus concentration of 32 μg/L, indicating a significant amount of phosphorus was being loaded to Kentuck Lake from an unaccounted source.

The historical data in combination with the data collected in 2013 provided indications that this excess phosphorus was likely originating from *internal nutrient loading*, a process where phosphorus and other nutrients are released from bottom sediments into the overlying water under anoxic (devoid of oxygen) conditions. The available data from 1988-2013 revealed that monthly near-surface total phosphorus concentrations in Kentuck Lake tended to increase over the course of the growing season, with increases typically beginning between June and July and increasing into September. It was hypothesized that with the onset of thermal stratification/anoxia, phosphorus that was released from bottom sediments into bottom waters was mobilized to the surface either through vertical entrainment and/or complete mixing events. This internal "nutrient pump" where sediment-released phosphorus and other nutrients are delivered from bottom to surface waters has been documented in shallow lakes world-wide (Orihel et al. 2015).

However, nutrient concentrations from near-bottom waters in Kentuck Lake were lacking, and Onterra ecologists did not have an understanding as to why the lake experienced a rapid shift to a higher nutrient state in 2011. In order to obtain a more detailed understanding of Kentuck Lake's water quality dynamics and to be able move forward with potential mitigation strategies, Onterra and WDNR scientists agreed that additional data were needed to confirm the hypothesis that internal nutrient loading was the primary source of the excess phosphorus in Kentuck Lake.

Understanding that the development of the Kentuck Lake Comprehensive Management Plan (Onterra 2015) was a baseline assessment of Kentuck Lake, the KLPRD elected to proceed with a two-year, advanced assessment of Kentuck Lake's water quality beginning in 2015. The KLPRD was awarded a WDNR Lake Protection Grant in 2015 to aid in funding this project. Onterra ecologists partnered with WDNR research scientists to design a study that would have a primary goal of determining where the excess phosphorus in Kentuck Lake was originating and if any management strategies could be implemented to reduce the frequency and/or lessen the severity of cyanobacterial blooms. The primary objectives of this study were as follows:

1. Determine if phosphorus is the primary nutrient regulating phytoplankton production in Kentuck Lake.
2. Obtain empirical evidence that release of phosphorus and other nutrients from anoxic bottom sediments is occurring and quantify the annual load.
3. Collect data to aid in understanding the role of ferrous iron in cyanobacterial bloom formation.
4. Determine dominant phytoplankton taxa over the course of the growing season and provide evidence of cyanobacteria in bottom waters.
5. Determine the drivers of interannual variability in nutrient concentration and phytoplankton abundance.
6. Determine how Kentuck Lake's water quality has changed since Euro-American settlement through sediment core analysis.
7. Determine if groundwater is a significant contributor of phosphorus to the lake.
8. Determine if mitigation strategies can be implemented to reduce the frequency and/or lessen the severity of cyanobacterial blooms.
9. Map non-native aquatic plants in Kentuck Lake and determine appropriate monitoring and /or control strategies.

These primary study objectives were fulfilled during this project, and greater understanding of Kentuck Lake's water quality dynamics was attained. The data collected show that Kentuck Lake's water quality has changed significantly since Euro-American development began in the early twentieth century, and nutrient levels and biological production are presently the highest they have been within the past 200 years. Internal loading became a significant source of phosphorus to the lake in the late-1970s and is currently the largest source of phosphorus to the lake accounting for approximately 70% of the annual load since 2011. Phytoplankton production is largely dependent upon phosphorus availability, and annual phosphorus concentrations are highly correlated with the strength of thermal stratification and the duration of anoxia. This report serves as the final deliverable project under the 2015 Lake Protection Grant (LPT-480-15) and discusses the results of the studies completed as well as management strategies developed with the KLPRD to enhance the lake's water quality, overall ecology, and ecosystem services.

1.1 Background on Kentuck Lake Water Quality

Near-surface total phosphorus and chlorophyll-*a* concentrations have been collected annually on Kentuck Lake since 1988 through a combination of the WDNR Long-Term Trend Monitoring and Citizens Lake Monitoring Network Programs. From 1988-2016, average summer (June-August) near-surface total phosphorus concentrations have ranged from 78 µg/L in 1988 to 18 µg/L in 2005, and have averaged 34 µg/L (median 31 µg/L) over this period (Figure 2). Summer average chlorophyll-*a* concentrations have ranged from 86 µg/L in 1988 to 7 µg/L in 2001, 2006, and 2008, with an average of 22 µg/L (median 15 µg/L) (Figure 2). These data show the Kentuck Lake has experienced a high degree of interannual variability in terms of near-surface phosphorus and chlorophyll-*a* concentrations since 1988.

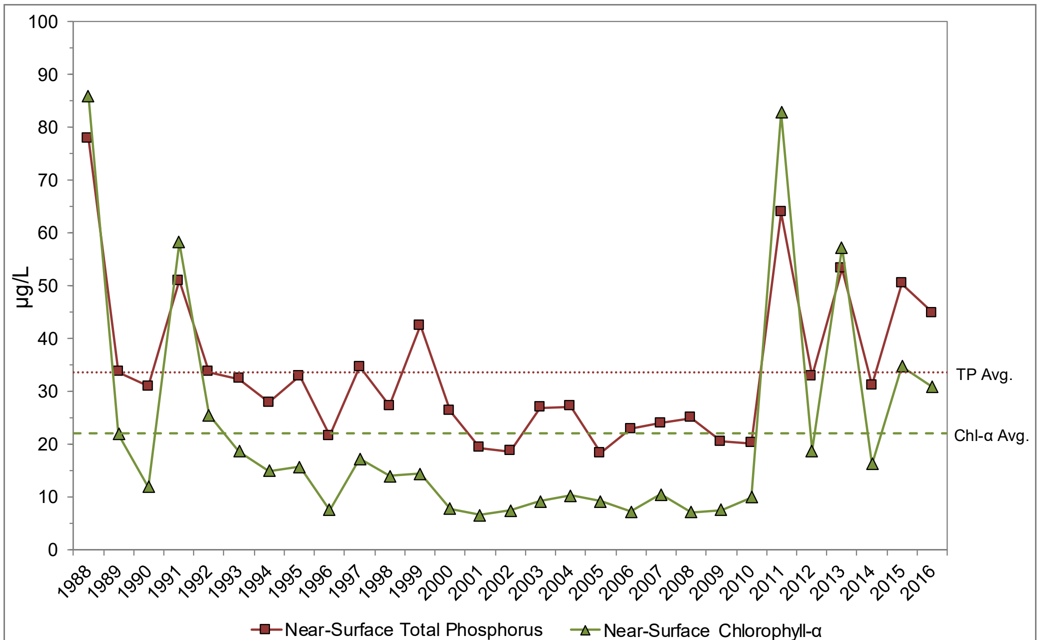

Figure 2. Kentuck Lake average summer near-surface total phosphorus and chlorophyll-α concentrations from 1988-2016. Data were collected from the Deep Hole Location – Site 1.

From 1988-2010, there had been a declining trend in near-surface total phosphorus and chlorophyll-*a* concentrations in Kentuck Lake, and from 2000-2010 phosphorus and chlorophyll-*a* concentrations were relatively stable (Figure 2). However, as discussed in the previous section, near-surface total phosphorus and chlorophyll-*a* concentrations increased markedly in 2011 in the absence of any significant watershed disturbance. Since 2011, near-surface total phosphorus and chlorophyll-*a* concentrations have been highly variable, but on average have increased by approximately 53% and 135%, respectively, when compared to average concentrations from 1988-2010.

Simple linear regression of summer near-surface total phosphorus and chlorophyll-*a* concentrations shows a strong, positive relationship indicating that phytoplankton production in Kentuck Lake is primarily regulated by the availability of phosphorus (Figure 3a). The WDNR has defined nuisance algal blooms as those exceeding a chlorophyll-*a* concentration of 20 µg/L, and in Kentuck Lake summer chlorophyll-*a* concentrations exceed this threshold when near-surface total phosphorus concentrations are >32 µg/L. Comparing summer Secchi disk depth, a measure of water clarity, against summer chlorophyll-*a* concentrations shows a strong, negative, non-linear relationship (Figure 3b). This non-linear relationship between Secchi disk depth and chlorophyll-*a* has been observed in other lakes, and is believed to be due to phytoplankton producing more chlorophyll as light availability declines (Carlson 1977). The relationship between chlorophyll-*a* and Secchi disk depth indicates that water clarity in Kentuck Lake is primarily dependent upon phytoplankton abundance. When chlorophyll-*a* concentrations reach the 20 µg/L threshold, Secchi disk depth declines to approximately 4.9 feet (Figure 3b).

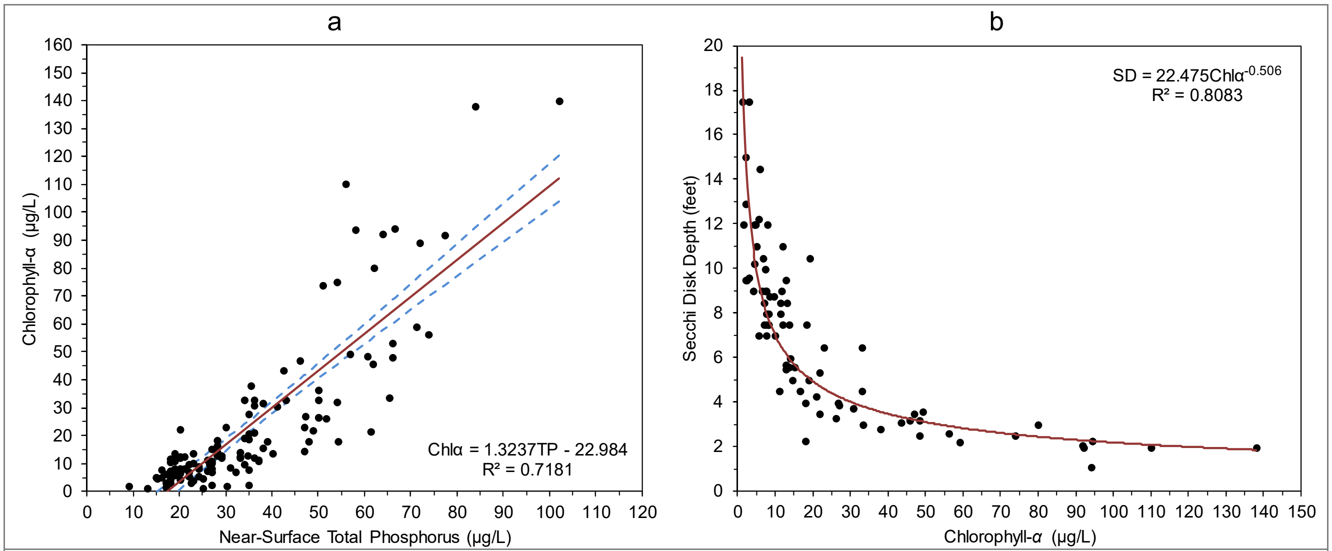

Figure 3. Kentuck Lake summer chlorophyll-α concentration as a function of summer near-surface total phosphorus concentration (a; N = 125) and summer Secchi disk depth as a function of summer chlorophyll-α concentration (b; N = 83). Dashed lines represent 95% confidence interval. Created using June-August data from 1986-2016. Data collected from Deep Hole Location – Site 1.

While there is a high degree of interannual variability among near-surface total phosphorus and chlorophyll-*a* concentrations in Kentuck Lake, a general pattern of increasing concentrations of phosphorus and chlorophyll-*a* over the course of the growing season has occurred each year since 1988. While the magnitude of this increase varies between years, total phosphorus and chlorophyll-*a* concentrations are lowest in spring followed by a continual increase beginning between June and July and continuing into September (Figure 4). This pattern of increasing phosphorus concentrations over the course of the growing season was an early indicator to Onterra and WDNR scientists that internal nutrient loading was likely occurring. In

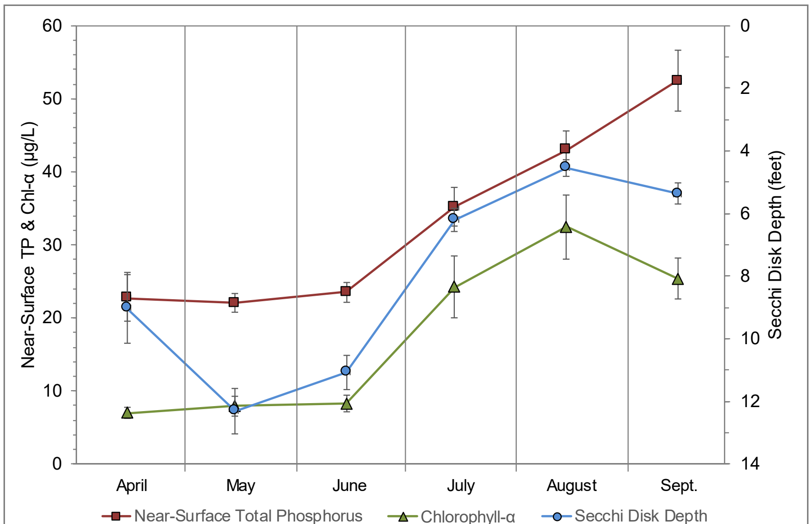

Figure 4. Kentuck Lake average monthly near-surface total phosphorus (N = 180) and chlorophyll-α (N = 167) concentrations and Secchi disk depth (N =289) from 1988-2016. Error bars indicate one standard error. Data were collected from the Deep Hole Location – Site 1.

general, in stratified lakes like Kentuck Lake, phosphorus concentrations tend to be higher early in the growing season following spring rains and snowmelt and then begin to decline over the course of the summer as precipitation decreases and phytoplankton consume phosphorus, die, and sink to the bottom. Come fall, phosphorus concentrations tend to increase again with fall mixing and increased precipitation. The atypical pattern of increasing phosphorus in Kentuck Lake has been observed in other shallow lakes which experience internal nutrient loading (Moss et al. 2013).

External phosphorus loading to Kentuck Lake was estimated using Wisconsin Lakes Modeling Suite (WiLMS) during the 2012 comprehensive lake management planning project and updated during the 2015 advanced water quality study. Kentuck Lake's surficial watershed encompasses approximately 4,842 acres. Approximately 34% of the lake's watershed is comprised of the Spectacle Lake watershed while the lake's direct watershed is comprised of forests (39%), the lake's surface (21%), wetlands (4%), and pasture/grass (2%). In addition to these land cover types, estimates were made for external phosphorus originating from riparian properties and riparian septic systems. To estimate the external phosphorus originating from riparian properties, the number of riparian parcels containing structures were totaled. Each parcel was assigned an area of 0.13 hectares. The total hectares of parcels containing structures was then multiplied by a phosphorus coefficient of 0.3 kg/year, the same rate used in a USGS study on nearby Anvil Lake (Panuska and Kreider 2003). The external phosphorus loading from septic systems was completed in WiLMS using data collected from an anonymous stakeholder survey distributed to lake riparians in 2013.

The watershed modeling indicated that Kentuck Lake receives an estimated 484 pounds of phosphorus from external sources annually (Figure 5). Forests, pasture/grass, and wetlands within the lake's direct watershed contribute 40% (194 lbs) of the annual phosphorus load, while 31% (150 lbs) is estimated to originate from atmospheric deposition onto the lake's surface. Total phosphorus data from Spectacle Lake were used in the modeling, and it is predicted that the Spectacle Lake watershed contributes 21% (103 lbs) of the annual phosphorus load to Kentuck Lake. Riparian properties and riparian septic systems were estimated to contribute 5% (23 lbs) and 3% (14 lbs) of the annual external load of phosphorus, respectively.

Based on the estimated external phosphorus load of 484 pounds, WiLMS predicted an in-lake growing season mean total phosphorus concentration of 16 µg/L, approximately 51% lower than the measured growing season mean of 33 µg/L. WiLMS estimated that a loading of an additional 1,950 pounds of phosphorus per year is required to achieve a growing season mean concentration of 33 µg/L, yielding a total estimated annual load of 2,435 pounds. This watershed modeling indicates that external anthropogenic

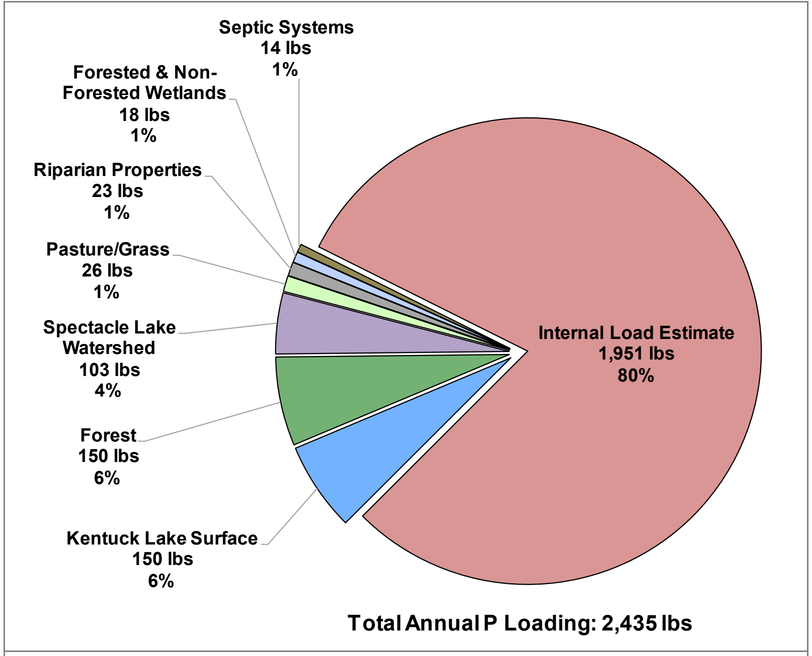

Figure 5. Kentuck Lake estimated external and internal annual loading of phosphorus. Created from model developed in WiLMS.

sources of phosphorus cannot account for the phosphorus required to attain the concentrations measured in Kentuck Lake. The lack of an external source for this excess phosphorus was another indicator that internal nutrient loading was likely the source of this unaccounted phosphorus in Kentuck Lake.

1.2 Primer on Internal Nutrient Loading

In general, lakes tend to act as phosphorus sinks, meaning they tend accumulate phosphorus over time and export less phosphorus than the amount that is loaded to the lake from its watershed. In most lakes, there is a net movement of phosphorus from the water to bottom sediments where it accumulates over time. The retention of this phosphorus within bottom sediments depends on a number of physical, chemical, and biological factors (Wetzel 2001). If this phosphorus remains bound within bottom sediments, it is largely unavailable for biological use. However, under certain conditions, this phosphorus can be released from bottom sediments into the overlying water where it may become biologically available. This release of phosphorus (and other nutrients) from bottom sediments into the overlying water is termed *internal nutrient loading*. The primary conditions which lead to phosphorus being released from bottom sediments are discussed below.

Anoxia at the Sediment-Water Interface

Lake stratification occurs when temperature and density gradients are developed with depth in a lake. During stratification, the lake can be broken into three layers: The ***epilimnion*** is the surface layer with the lowest density and has the warmest water in the summer months and the coolest water in the winter months. The ***hypolimnion*** is the bottom layer the highest density and has the coolest water in the summer months and the warmest water in the winter months. The ***metalimnion***, often called the thermocline, is the layer between the epilimnion and hypolimnion where temperature changes most rapidly with depth.

When water at the sediment-water interface contains oxygen, phosphorus largely remains bound to ferric iron within the sediment. When the water at the sediment-water interface becomes anoxic, or devoid of oxygen, ferric iron is reduced to ferrous iron and the bond between iron and phosphorus is broken. Under these conditions, iron and phosphorus are now soluble in water and are released from the sediments into the overlying water (Pettersson 1998). Anoxia at the sediment-water interface typically first develops following thermal stratification, or the formation of distinct layers of water based on temperature and density. As surface water warms in late-spring/early summer, it becomes less dense and floats atop the colder, denser layer of water below. The large density gradient between the upper, warm layer of water (*epilimnion*) and lower, cold layer of water (*hypolimnion*) prevents these layers from mixing together and eliminates atmospheric diffusion of oxygen into bottom waters. If there is a high rate of biological decomposition of organic matter in the bottom sediments, anoxic conditions within the hypolimnion can develop as oxygen is consumed and is not replaced through mixing. The loss of oxygen then results in the release of phosphorus from bottom sediments into the hypolimnion.

The development of an anoxic hypolimnion and subsequent release of phosphorus from bottom sediments occurs in many lakes in Wisconsin. However, in deeper, dimictic lakes which remain stratified during the summer, internal nutrient loading is often not problematic as the majority of the phosphorus released from bottom sediments is confined within the hypolimnion where it is largely inaccessible to phytoplankton. Dimictic lakes are those which remain stratified throughout the summer (and winter) and experience only two complete mixing events (turnover) per year, one in spring and one in fall. In dimictic lakes, phosphorus released from bottom sediments into the hypolimnion during stratification only becomes available to phytoplankton in surface waters during the spring and fall mixing events. While these spring and fall mixing events can stimulate diatom and golden-brown phytoplankton blooms, these mixing events generally to not stimulate cyanobacterial blooms because water temperatures are cooler.

Internal nutrient loading can become problematic in lakes when sediment-released phosphorus becomes accessible to phytoplankton during the summer months when surface temperatures are at their warmest. Sediment-released phosphorus can be mobilized to surface waters during the summer in polymictic lakes, or lakes which have the capacity to experience multiple stratification and mixing events over the course of the growing season. Some polymictic lakes tend to straddle the boundary between deep and shallow lakes, and have the capacity to break stratification in summer when sufficient wind energy is generated. Consequently, phosphorus which has accumulated in the anoxic hypolimnion during periods of stratification is mobilized to the surface during partial or full mixing events where it then can spur nuisance phytoplankton blooms at the surface.

Phosphorus from bottom waters can also be mobilized to the surface in polymictic lakes through entrainment, or the continual deepening of the epilimnion and erosion of the metalimnion below (Wetzel 2001). Wind-driven water generates turbulence across the thermal barrier between the epilimnion and the metalimnion and the metalimnion is eroded, mixing sediment-released nutrients into the epilimnion above. Both periodic mixing and entrainment act as “nutrient pumps” in polymictic lakes, delivering sediment-released nutrients in bottom waters to surface waters (Orihel et al. 2015). While a continuum exists between dimictic and polymictic lakes, the Osgood Index (Osgood 1988) is used to determine the probability that a lake will remain stratified during the summer. This probability is estimated using the ratio of the lake’s mean depth to its surface area. Lakes with an Osgood Index of less than 4.0 are deemed polymictic. As is discussed further in this report, Kentuck Lake has an Osgood Index value of 2.0, indicating Kentuck Lake is a polymictic system.

Elevated pH

The pH scale ranges from 0 to 14 and is an indicator of the concentration of hydrogen (H+) within the lake’s water and is an index of the lake’s acidity. Water with a pH value of 7.0 has equal amounts of hydrogen ions and hydroxide ions (OH-) and is considered neutral. Water with a pH of less than 7.0 has a higher concentration of hydrogen ions and is acidic, and water with a pH of greater than 7.0 has lower hydrogen ion concentrations and is considered basic or alkaline. The pH scale is logarithmic, meaning that for every 1.0 pH unit the hydrogen ion concentration changes tenfold. The normal range for lake water pH in Wisconsin is about 5.2 to 8.4, though values lower than 5.2 can be found in some acid bog lakes and higher than 8.4 in some marl lakes and highly productive softwater likes like Kentuck Lake (Shaw and Nimphius 1985).

Carbon dioxide dissolves in and reacts with lake water to form carbonic acid which lowers the water’s pH. However, during the day, photosynthesizing phytoplankton (and *macrophytes*) consume carbon dioxide and can raise water pH. When phytoplankton become overly abundant they have the capacity to raise a lake’s pH to 9.0 or greater during the day. When pH reaches these levels, the capacity of phosphorus to remain bound within the sediment is reduced, and phosphorus can be released from sediment under these conditions even in the presence of oxygen (Solim and Wanganeo 2009).

Macrophytes are larger aquatic plants that can be seen with the naked eye, and include flowering plants such as pondweeds and milfoils and macroalgae like muskgrasses among others.

Cyanobacterial Migration

While most species of phytoplankton are unable to obtain phosphorus bound within the sediment, certain species of cyanobacteria are able to access phosphorus within the sediment and make it available to other phytoplankton within the water column (Cottingham et al. 2015). These cyanobacteria spend a portion of their life cycle on the lake bottom (Barbiero and Welch 1992). Three genre that are found in Kentuck Lake, *Gloeotrichia, Anabaena*, and *Aphanizomenon* overwinter on the lake bottom and begin to uptake sediment-bound phosphorus in the spring. In fact, studies have shown that *Gloeotrichia* takes up more phosphorus from the sediment than it actually requires for growth and is termed *luxury uptake* (Carey et al. 2014). Once it has acquired sufficient amounts of nutrients, it migrates from the sediment into the water column to reproduce. While the mechanism is yet unknown, *Gloeotrichia* releases phosphorus that it obtained from the sediment into the water column which can stimulate the growth of other phytoplankton (Carey and Rengefors 2010). This uptake and delivery of phosphorus from the sediments into the water column by cyanobacteria has shown to be significant form of internal phosphorus loading in certain lakes (Barbiero and Welch 1992).

Recently, it has also been proposed that cyanobacteria may migrate from the surface down into the hypolimnion to access phosphorus, iron, and other nutrients released from the sediment under anoxic conditions (Molot et al. 2014). Under oxygenated conditions, iron is in a form (ferric) that is generally unavailable for use by most cyanobacteria, but under anoxic conditions iron is reduced to a form (ferrous) that is readily utilized by cyanobacteria. Because ferrous iron will only be found under anoxic conditions, it is believed cyanobacteria can migrate into the hypolimnion to access it. Cyanobacteria have a higher iron demand when compared to true-algae, and it enhances their capacity to fix nitrogen. Nitrogen fixation, a unique ability of cyanobacteria among phytoplankton, is the ability to utilize atmospheric nitrogen and convert it into a usable form. When nitrogen becomes limiting within the water, cyanobacteria can access atmospheric nitrogen which provides them with a competitive advantage over true-algae. The combination of nitrogen limitation and the availability of ferrous iron is believed to be the catalyst for the development of potentially harmful nitrogen-fixing cyanobacterial blooms. As mentioned earlier, one of the objectives of this study was to aid in the understanding the role of ferrous iron in cyanobacterial bloom formation.

2.0 PROJECT METHODS

2.1 Surface Water Sampling

In 2015, water quality samples were collected by KLPRD volunteers from five locations distributed across the lake (Table 1 and Map 1) every other week (biweekly) from late-April through mid-October, yielding a total of 14 sampling events. Similarly, water quality samples were collected by KLPRD volunteers in 2016 biweekly beginning in early-May through mid-October again yielding 14 sampling events. However, because parameters collected from near-surface samples were not statistically different across all five sampling locations in 2015, sampling was only conducted at sampling sites 1 and 5 in 2016.

Photo 1. KLPRD volunteers being trained to collect water quality data in Kentuck Lake by Onterra ecologist Brenton Butterfield. Photo credit Onterra.

A 3.0 L Van Dorn Sampler was used to collect the water samples, and all samples that required laboratory analysis were preserved appropriately and shipped on ice to the Wisconsin State Laboratory of Hygiene for analysis. Temperature and dissolved oxygen profiles were collected using a YSI ProODO® handheld optical dissolved oxygen meter. Water clarity was measured using a 20 cm Secchi disk. Onterra ecologists met with the KLPRD volunteers during the first sampling event of each year to provide training (Photo 1).

In 2015, at sampling sites 1 and 5, water quality samples were collected from the near-surface (3.0 feet), near-oxycline (1.0 foot below first dissolved oxygen reading of 2.0 mg/L), and near-bottom (3.0 feet above the bottom). If dissolved oxygen was above 2.0 mg/L throughout the entire water column, the near-oxycline sample was taken midway between the surface and the bottom. Water samples were collected from the near-surface and near-bottom at sites 2, 3, and 4. The water quality parameters collected from each water quality sampling location in 2015 can be found in Table 2. In the winter of 2016, samples were collected and analyzed for total phosphorus only, and samples were collected from the near-surface and near-bottom from site 1 and the near-surface of site 5.

In 2016, at sampling site 1, water quality samples were collected from the near-surface, near-oxycline, and near-bottom utilizing the same methodology as described for 2015. At sampling site 5, water quality samples were collected from the near-surface only. In the winter of 2017, water quality samples were collected and analyzed for total phosphorus only, and samples were collected from the near-surface and near-bottom from site 1 and the near-surface from site 5. The water quality parameters collected from each sampling site in 2016 can be found in Table 2.

In addition to the parameters listed in Table 2, samples were also collected from sites 1 and 5 biweekly throughout the growing seasons for 2015 and 2016 for *phytoplankton* taxa identification. Samples were collected from the near-surface, near-oxycline, and near-bottom from both sampling locations. These samples

> **Phytoplankton** are free-floating, microscopic photosynthesizing organisms and include algae and cyanobacteria among others.

were collected with a 3.0 L Van Dorn Sampler and were analyzed by Jim Kreitlow (WDNR). The phytoplankton were identified to at least the genus level, and if possible was identified to species. The dominant phytoplankton taxa within each sample were noted.

Table 1. Location and maximum depth of 2015 and 2016 surface water quality sampling locations in Kentuck Lake.

Site	Lat.	Long.	Max Depth (feet)
1	45.992833	-88.986182	40.9
2	45.988739	-88.994717	23.1
3	45.983702	-89.003975	18.0
4	45.978478	-89.002627	19.0
5	45.976590	-89.011548	15.4

Table 2. Water quality parameters collected from Kentuck Lake from 2015-2016.

	Site 1			Site 2			Site 3			Site 4			Site 5		
Parameter	NS	NO	NB	NS	NO	NB	NS	NO	NB	NS	NO	NB	NS	NO	NB
Total Phosphorus	●●●	●●	●●●	●		●	●		●	●		●	●●●	●	●
Dissolved Phosphorus	●●	●●	●●										●●	●	●
Nitrate-Nitrite Nitrogen	●●	●●	●●										●●	●	●
Ammonia Nitrogen	●●	●●	●●										●●	●	●
Total Iron	●●	●●	●●										●●	●	●
Chlorophyll-α	●●												●●		

NS = Near-Surface; NO = Near-Oxycline; NB = Near-Bottom

● = April-October 2015; ● = May-October 2016; ● = Winter 2016

2.2 Sediment Core Analyses

Two sediment cores were taken from Kentuck Lake on September 22, 2015. An 80-cm core was recovered from a site in the deep area in the northeast part of the lake (Site 1 on Map 2) in a water depth of 38 feet. A 94-cm core was recovered from a site in the southwestern part of the lake (Site 2 on Map 2) in a water depth of 16.8 feet using a piston corer. Each core was sectioned into 1 cm slices in the upper 20 cm and then 2 cm slices down to 70 or 72 cm. The samples were placed in plastic freezer bags and kept in a dark refrigerator.

The core at Site 1 was black/brown in color for the upper 12 cm and a lighter brown color with some sand from 12 to 24 cm. From 24 to 70 cm the core was composed almost entirely of sand. The very bottom of the core was dark gray clay. The core at Site 2 had a very different composition than the core from Site 1. The entire core was composed of organic muck with the upper 48 cm being dark brown in color and the deeper portion of the color being medium brown in color. It seems likely that the sediments at Site 1 had been significantly disturbed at some time in the past. The shallow depth of organic sediment which overlays sandy material below 12 cm indicates that there might have been dredging at this site; however, this has not been confirmed. It is likely the sand below 12 cm was deposited as the glacier receded 13,000 years ago. Because it is likely the sediment at Site 1 had been significantly disturbed, the paleoecological core analysis for this study was conducted on the core from Site 2. A description of the parameters analyzed within the sediment core are discussed in the Paleoecology Results Section (Section 3.4).

In February 24, 2016, Onterra ecologists also collected two small sediment cores through the ice, one in the southwest area of the lake and one near the deep hole (Map 2 and Photo 2). These sediment cores were sent to Dr. William James at UW-Stout where phosphorus release rates were measured under oxic and anoxic conditions.

Photo 2. Left: WDNR paleoecologist Paul Garrison (now with Onterra) (front) and aquatic ecologist Dan Cibulka (back) view one of two sediment cores collected from Kentuck Lake in September 2015. Right: Paul Garrison with one of six sediment cores collected for nutrient release analysis in February of 2016. Photo credit Onterra.

2.3 Groundwater Quality Assessment

While groundwater is typically not a significant source of phosphorus to lakes, an attempt to sample phosphorus concentrations within groundwater entering Kentuck Lake was made to confirm that groundwater was not the source of the unaccounted phosphorus. A mini-piezometer was constructed to determine the static water head level relative to the surface of the lake at 67 locations spread 500 feet apart around the perimeter of Kentuck Lake (Map 1). The goal was to collect groundwater samples at locations where groundwater was determined to be flowing into the lake and analyzed for total phosphorus.

Photo 3. Onterra limnologist Tim Hoyman installs a mini-piezometer in Kentuck Lake. Photo credit Onterra.

Two separate attempts were made to execute this groundwater study, once on September 22-23, 2015, and again on May 2, 2016 (Photo 3). However, both attempts were met with inconclusive results. Onterra ecologists visited areas of the lake that were observed to have open water near shore in winter, an indication of groundwater inflow. In some areas, the mini-piezometers were unable to penetrate beneath the rock/cobble present in many areas around the lake. And in areas of sand where the mini-piezometers were able to penetrate, the mini-piezometer did not provide conclusive indications of groundwater inflow. Because a definitive indication of groundwater inflow could not be made, no groundwater quality samples were collected for analysis. As is discussed within the results section, the pattern of phosphorus concentrations over the course of the growing season in Kentuck Lake as measured from surface water samples indicate that groundwater is not likely a significant source of the unaccounted phosphorus.

2.4 Climatic Data

An analysis was completed to investigate the relationship between Kentuck Lake's water column stability, or strength of thermal stratification, with local climatic conditions. Climatic data from 2001-2015 were obtained from a weather station at the Eagle River Union Airport in Eagle River, approximately 13 miles southwest of Kentuck Lake, while annual climatic data from 1988-2000 were obtained from a weather station in Rhinelander, approximately 30 miles southwest of Kentuck Lake (Weather Underground 2015) (Figure 6). Climatic data obtained included average daily air temperature and average daily wind speed.

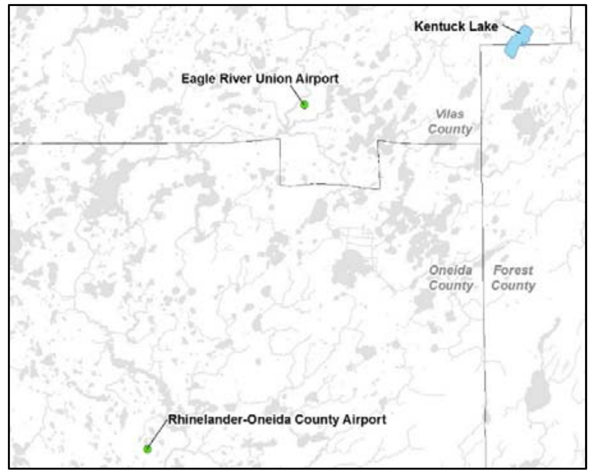

Figure 6. Location of Eagle River Union Airport and Rhinelander-Oneida County Airport weather stations.

2.5 Aquatic Invasive Species Monitoring

Two visual, meander-based surveys of Kentuck Lake's littoral zone were completed in 2015 and 2016 aimed at locating and mapping occurrences of the non-native plants curly-leaf pondweed (CLP) and Eurasian watermilfoil (EWM). The early-season aquatic invasive species (ESAIS) surveys were completed on June 30, 2015 and June 15-16, 2016. In 2015, the EWM peak-biomass survey was completed on September 16. This survey was attempted on August 17, 2016 but the water clarity was too poor and the survey was aborted. However, EWM was mapped during the 2016 ESAIS survey.

During these surveys, the CLP and EWM were mapped by using either 1) point-based or 2) area-based mapping methodologies. Large colonies, >40 feet in diameter, were mapped using polygons (areas) and were qualitatively attributed a density rating based upon a five-tiered scale ranging from *highly scattered* to *surface matting*. Point-based mapping techniques were applied to locations that were considered as *small plant colonies* (<40 feet in diameter), *clumps of plants*, or *single or few plants.*

As mentioned, the 2016 EWM peak-biomass survey was not completed due to low water clarity. Instead of postponing this survey until 2017, Onterra recommended that a whole-lake point-intercept survey be completed in 2017 to quantitatively assess the lake's EWM and native aquatic plant population. This quantitative survey would allow for a determination of EWM's frequency of occurrence within the lake as well as allow for comparisons of the lake's aquatic plant community to point-intercept surveys completed in 2007 and 2011. In addition, a point-intercept survey completed in 2017 would allow for a picture of the lake's plant prior to the implementation of a strategy to enhance the lake's water quality. The KLPRD agreed to move forward with completing a point-intercept survey in 2017 instead of an EWM peak-biomass survey. The point-intercept survey was completed by Onterra ecologists on August 16 and 17, 2017.

The point-intercept method as described in the Wisconsin Department of Natural Resource document, Recommended Baseline Monitoring of Aquatic Plants in Wisconsin: Sampling Design, Field and Laboratory Procedures, Data Entry, and Analysis, and Applications (WDNR PUB-SS-1068 2010) was used to complete this study. A point spacing of 86 meters was used resulting in 543 points.

3.0 PROJECT RESULTS & DISCUSSION

3.1 2014, 2015, & 2016 Temperature & Dissolved Oxygen Monitoring

Prior to the start of the advanced water quality monitoring project funded by the WDNR Lake Protection Grant, KLPRD volunteers collected temperature and dissolved oxygen profiles from seven locations distributed across Kentuck Lake biweekly from late-May through early-October 2014. While the Osgood Index indicated Kentuck Lake was likely polymictic, the data collected by the KLPRD volunteers in 2014 provided empirical evidence of polymixis.

> An **isopleth** is a graph of two variables (e.g. temperature and depth) with lines representing equal values.

Figure 7 displays temperature and dissolved oxygen *isopleths* collected by KLPRD volunteers in 2014 from the Deep Hole Location (Site 1) in Kentuck Lake. These figures illustrate that the lake was already thermally stratified during the first sampling event in late-May, and it remained stratified through most of June. Between late-June and early-July, strong, sustained south-southwest winds imparted sufficient energy across the lake to cause a complete mixing of the entire water column.

This event is evidenced by the relatively uniform water temperature throughout the water column of approximately 21°C in early July. This destabilization of the water column in early summer prevented the lake from developing a strong thermal gradient for the remainder of the growing season. Lacking a strong thermal gradient, the water column was easily mixed during subsequent wind events and water temperatures remained relatively uniform throughout the water column into fall. The ability of the water column to mix more readily allowed for the mixing of oxygen down to deeper depths, and only a small area of the lake deeper than 30 feet was found to experience anoxia following the early summer mixing event.

Following spring mixing in 2015, Kentuck Lake was thermally stratified for a brief period in mid-May before completely mixing again by the end of the month (Figure 8). In early- to mid-June, the lake again began to establish thermal stratification and the development of anoxia in bottom waters. Unlike 2014 where thermal stratification and anoxia were disrupted during a complete mixing event in late-June/early-July, in 2015 thermal stratification and anoxia persisted for approximately 11 weeks. However, this thermal gradient was relatively weak and the epilimnion grew deeper over the course of the summer. Because of the weak stratification, anoxia extended into a large part of the water column.

Anoxia persisted until late-August 2015 when a period of cooler weather and strong winds resulted in a complete mixing event and ended the 11-week period of anoxia. Approximately two weeks later, despite only a 3°C gradient between surface and bottom waters, anoxia developed in waters 20 feet and deeper during a period of hot, calm weather. This second period of anoxia was relatively short-lived as another period of cool, windy weather resulted in another complete mixing event by late-September and the lake remained mixed into October. Isopleths of temperature and dissolved oxygen collected in 2015 from sampling sites 2, 3, 4, and 5 can be found in Appendix A.

Temperature and dissolved oxygen monitoring from Kentuck Lake's Deep Hole sampling location in 2016 indicated a similar pattern to that observed in 2015 (Figure 9). The lake began to thermally stratify in late-May/early-June, forming an epilimnion and large metalimnion. As in 2015, the epilimnion was gradually grew deeper. Anoxia persisted for approximately 12-14 weeks, and again extended into depths around 19 feet by mid-summer. In late-August/early-September, a period of cool, windy weather resulted in a complete mixing event and oxygenation of bottom waters. The lake remained mixed and oxygenated throughout the remainder of the sampling period into October.

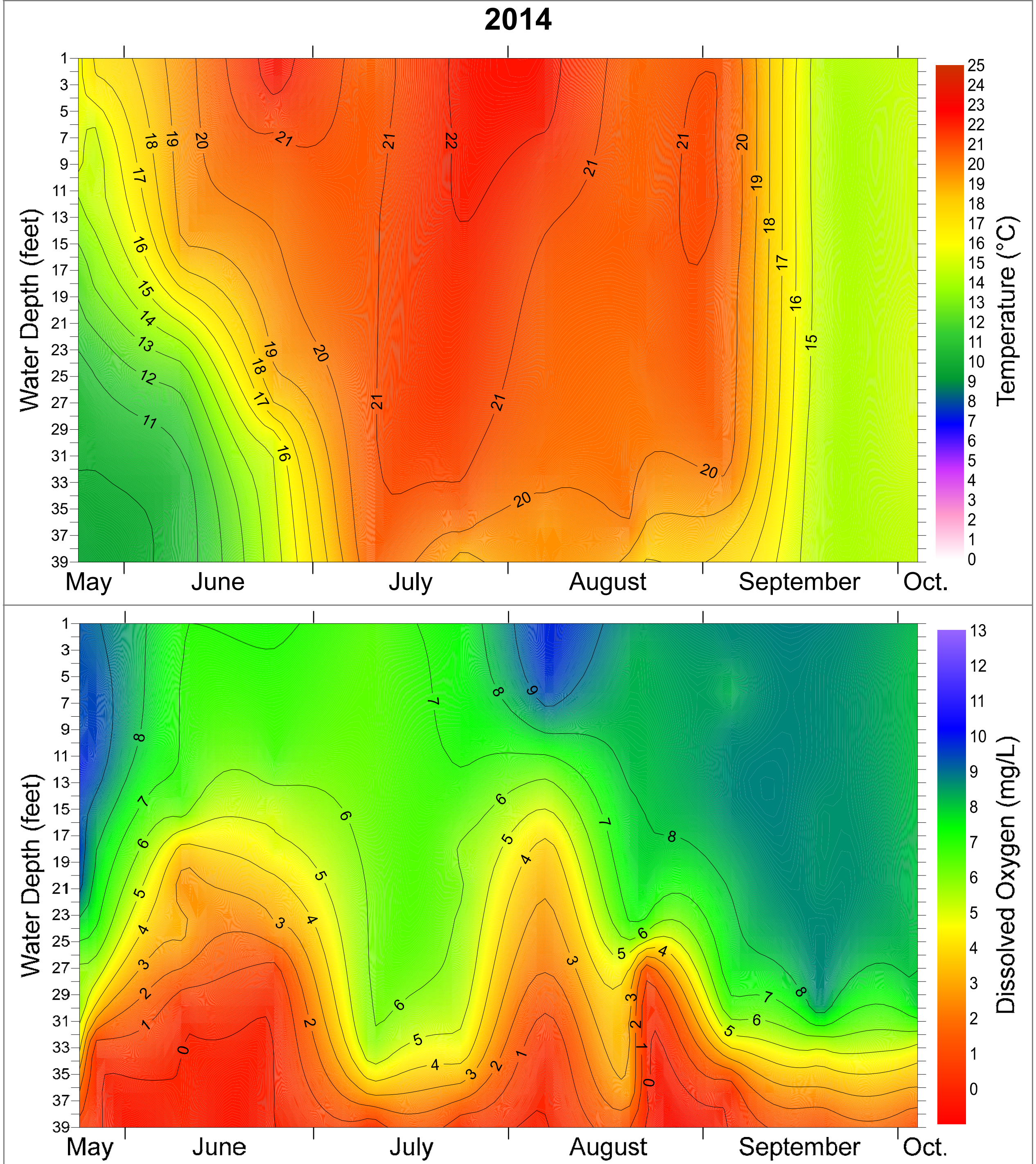

Figure 7. Kentuck Lake 2014 temperature (top) and dissolved oxygen (bottom) isopleths. Data collected from Deep Hole Location – Site 1.

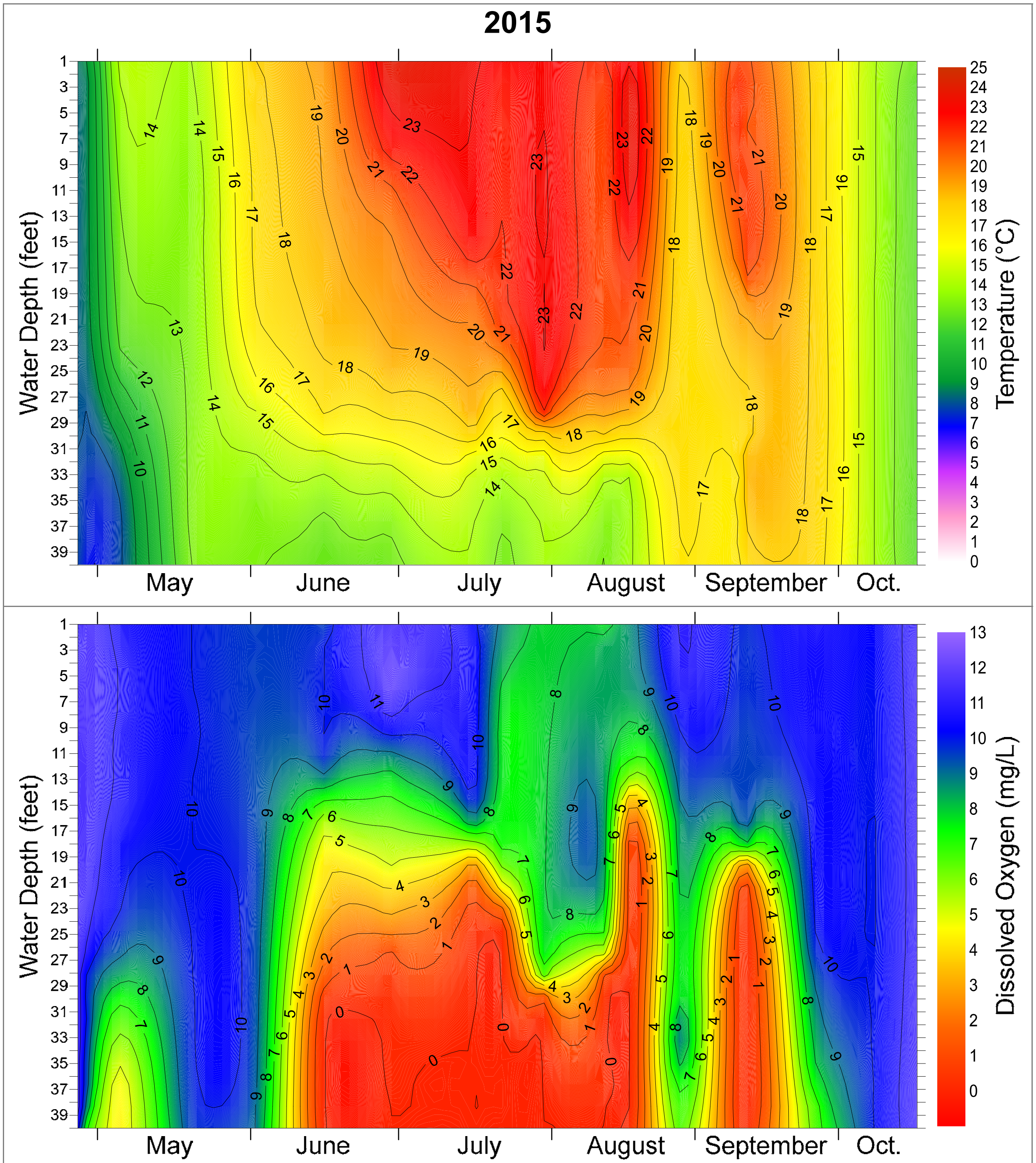

Figure 8. Kentuck Lake 2015 temperature (top) and dissolved oxygen (bottom) isopleths. Data collected from Deep Hole Location – Site 1.

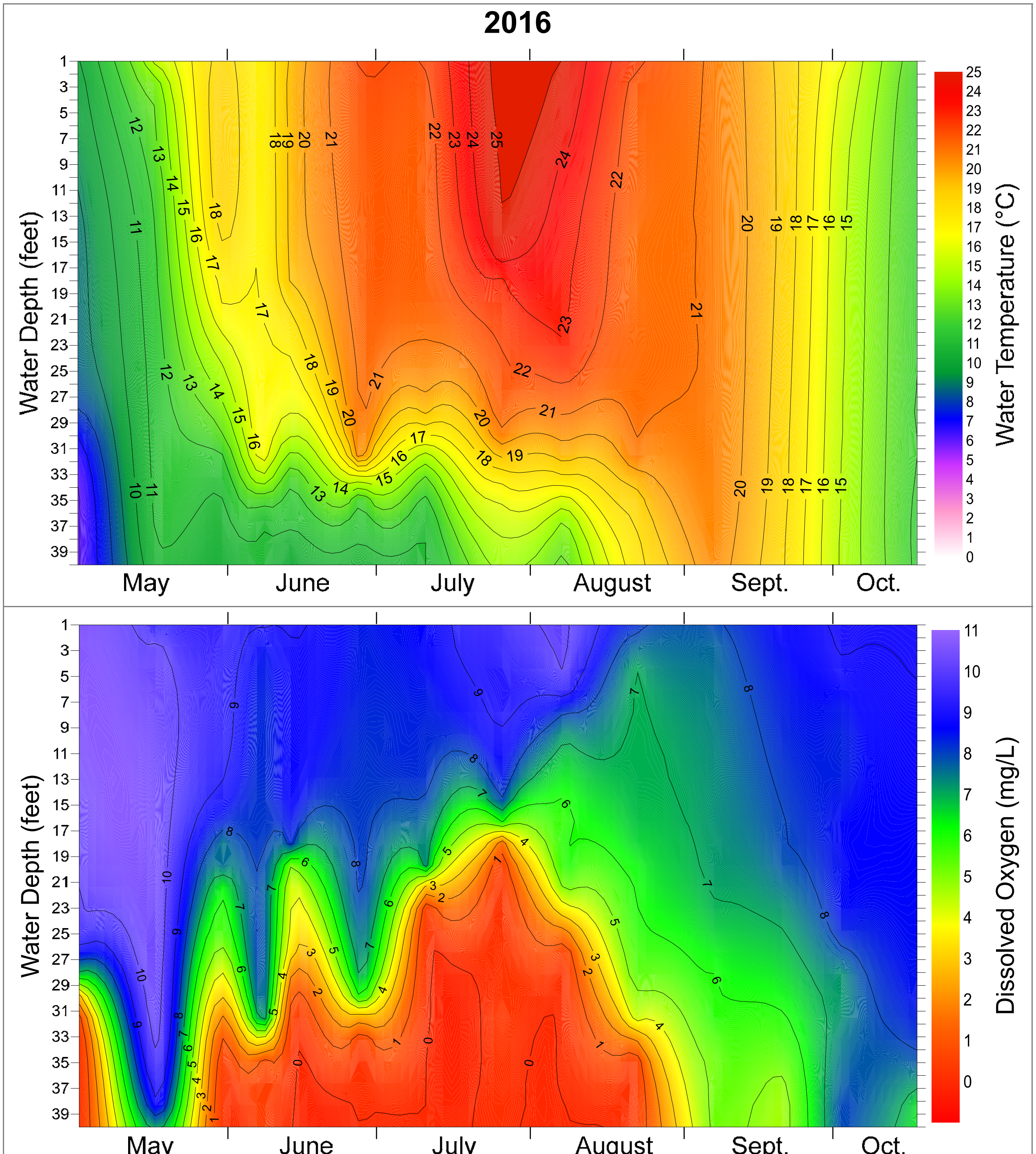

Figure 9. Kentuck Lake 2016 temperature (top) and dissolved oxygen (bottom) isopleths. Data collected from Deep Hole Location – Site 1.

3.2 2015 & 2016 Nutrients, Phytoplankton, & Water Clarity

Near-surface water chemistry data were not statistically different among the five sampling locations in 2015 and among the two sampling locations in 2016 (one-way ANOVA $\alpha = 0.05$). Water chemistry data collected from the near-bottom at sites 2, 3, 4 and 5 in 2015 and site 5 in 2016 were not statistically different from those at the surface. Site 1, the deepest location in Kentuck Lake, was the only site to experience prolonged periods of anoxia in 2015 and 2016 and was the only sampling location where near-bottom water chemistry values differed significantly from those at the surface. Because of this, near-bottom data discussed in this section refer to those collected at the Deep Hole location (Site 1). Near-bottom data collected from remaining sampling locations in 2015 and 2016 can be found in Appendix A.

Phosphorus

As is discussed in the Background on Kentuck Lake Water Quality Section (Section 1.1), phosphorus is the most important nutrient in terms of phytoplankton production in Kentuck Lake. In 2015 and 2016, both soluble reactive phosphorus (SRP) and total phosphorus (TP) were collected. Soluble reactive phosphorus (dissolved phosphorus) is a measure of orthophosphate, the soluble form that is most available for uptake by plant cells. Total phosphorus is a measure of all forms of phosphorus, including SRP and phosphorus bound to sediment particles or within plant and animal cells (particulate phosphorus).

Near-surface TP concentrations measured in Kentuck Lake in 2015 followed the same pattern of increasing concentration over the course of the growing season as was observed in previous years (Figure 9). In late-April through early-June when the water column was well mixed and oxygenated, near-surface TP concentrations were relatively stable with an average concentration of 23 µg/L. Near-surface TP concentrations began to increase in mid-June as thermal stratification strengthened and anoxia in bottom waters developed, and concentrations reached a maximum of 69 µg/L in late-July. Near-surface TP concentrations averaged 64 µg/L through early-September before declining to an average of 55 µg/L into October. The growing season average for near-surface TP concentrations in 2015 was 46 µg/L.

From late-April through early-June 2015, near-bottom TP concentrations varied widely and were similar to those measured at the surface when the lake was mixed and higher in mid-May when the lake briefly stratified (Figure 10). Average near-bottom TP concentrations from late-April through early-June were 53 µg/L. In mid-June, near-bottom TP concentrations began to increase rapidly as thermal stratification strengthened and anoxia developed. Near-bottom TP concentrations reached a maximum concentration of 660 µg/L in late-June. From mid-July through mid-August, near-bottom TP concentrations had declined to approximately 400 µg/L, likely a result of a deepening epilimnion. The near-bottom TP concentration declined to 70 µg/L in late-August when the lake experienced a complete mixing event; however, it quickly increased again to 460 µg/L in early-September with the onset of second period of brief anoxia. By mid-September in response to another mixing event, near-bottom TP concentrations declined and averaged 54 µg/L through October.

Water samples collected from the near-oxycline/mid-depth in 2015 indicated TP concentrations were similar to near-surface concentrations in late-April through early-June (Figure 10). Near-oxycline TP concentrations began to increase with near-bottom and near-surface concentrations in mid-June with the onset of thermal stratification and anoxia. However, near-oxycline TP concentrations from mid-June through late-July were approximately five times lower than near-bottom concentrations indicating an increasing TP concentration gradient from the oxycline to the bottom. The near-oxycline TP concentration continued to increase and was similar to the near-bottom concentration by mid-August. Following the late-August mixing event, near-oxycline TP concentrations were similar to near-surface concentrations for the remainder of the sampling period.

During a winter sampling event through the ice in February 2016, Kentuck Lake was found to be thermally stratified with anoxia at 20 feet and deeper. Near-surface TP concentrations had declined to 31 µg/L while the near-bottom TP concentration was 173 µg/L.

The concentrations and pattern over time of near-surface TP in Kentuck Lake during the growing season of 2016 were very similar to those observed in 2015 (Figure 10). However, initial near-surface TP concentrations from late-April through early-June in 2016 were on average 7.0 µg/L higher than those measured in 2015. Like in 2015, Kentuck Lake began to thermally stratify and develop anoxia in bottom waters in early-June 2016, and near-surface TP concentrations reached a maximum concentration of 64 µg/L in early-August. The rate of increase in phosphorus concentration was more gradual than what was observed in 2015 where maximum near-surface TP concentrations were reached in July. Near-surface TP concentrations averaged 56 µg/L through the end of September until they declined to an average of 41 µg/L in October. All growing season near-surface TP concentrations in 2016 averaged 44 µg/L.

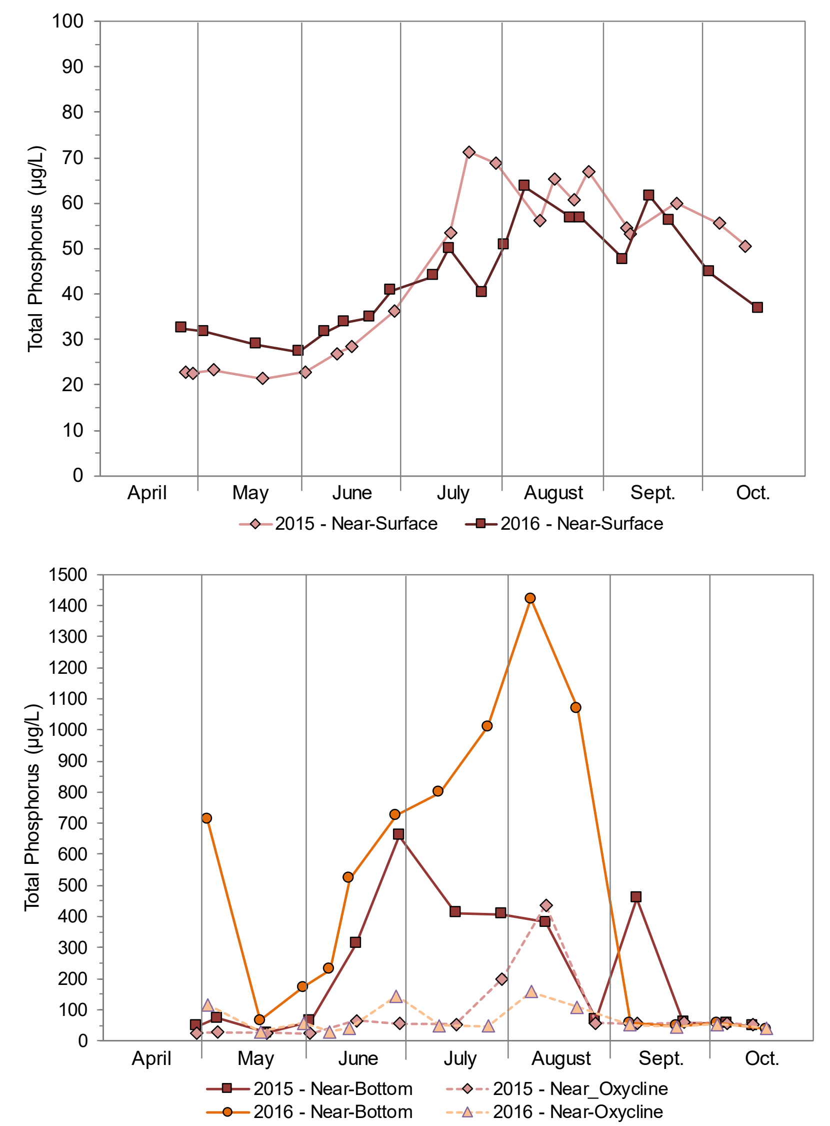

Figure 10. Kentuck Lake 2015 and 2016 growing season near-surface total phosphorus concentrations (top) and near-oxycline and near-bottom total phosphorus concentrations (bottom). Near-surface data represent mean concentrations collected from 2015 site 1, 2, 3, 4, & 5 and 2016 site 1 & 2. Near-oxycline and near-bottom concentrations collected from site 1.

Near-bottom TP concentrations measured in 2016 also followed a similar pattern to those measured in 2015 (Figure 10). The near-bottom TP concentration measured during the first growing season sampling event in early-May was high at 713 µg/L. It is possible that sediments may have contaminated this early spring sample as near-bottom concentrations took approximately four weeks to attain this concentration again in June. The near-bottom TP concentration declined to 66 µg/L during the next sampling event in mid-May, and subsequently began to increase through June and July and reached a maximum concentration of 1,420 µg/L in early-August. The near-bottom TP concentration declined slightly in late-August to 1,070 µg/L, and then declined to 58 µg/L in early-September following a complete mixing event and was similar to near-surface concentrations. Near-bottom TP concentrations were similar to near-surface concentrations for the remainder of the 2016 sampling season, averaging 48 µg/L.

Near-oxycline/mid-depth TP concentrations in 2016 were similar to those measured at the surface until mid-June when the lake began to thermally stratify and develop anoxia in bottom waters (Figure 10). While near-oxycline TP concentrations increased from mid-June into mid-August, concentrations were approximately ten times lower than near-bottom TP concentrations, indicating a strong gradient between the near-bottom and oxycline. Like near-bottom TP concentrations, near-oxycline TP concentrations reached a maximum in early-August 2016, and declined to concentrations similar to those measured at the surface following the complete mixing event in late August. Subsequent near-oxycline TP concentrations were similar to near-surface concentrations for the remainder of the sampling period.

Soluble reactive phosphorus within the epilimnion is rapidly incorporated into phytoplankton, meaning it is generally found in concentrations that are low (Wetzel 2001). Near-surface SRP concentrations collected in 2015 and 2016 were relatively low in April-July, with a number of concentrations falling below the limit of detection (1.7 µg/L) (Figure 11). Near-surface SRP concentrations were highest later in the growing seasons of 2015 and 2016, presumably due to the increased loading to the epilimnion during the complete mixing events and/or the reduced assimilation rate by phytoplankton.

Like near-bottom and near-oxycline TP concentrations, near-bottom and near-oxycline SRP concentrations were high under anoxic conditions and lower under oxic conditions (Figure

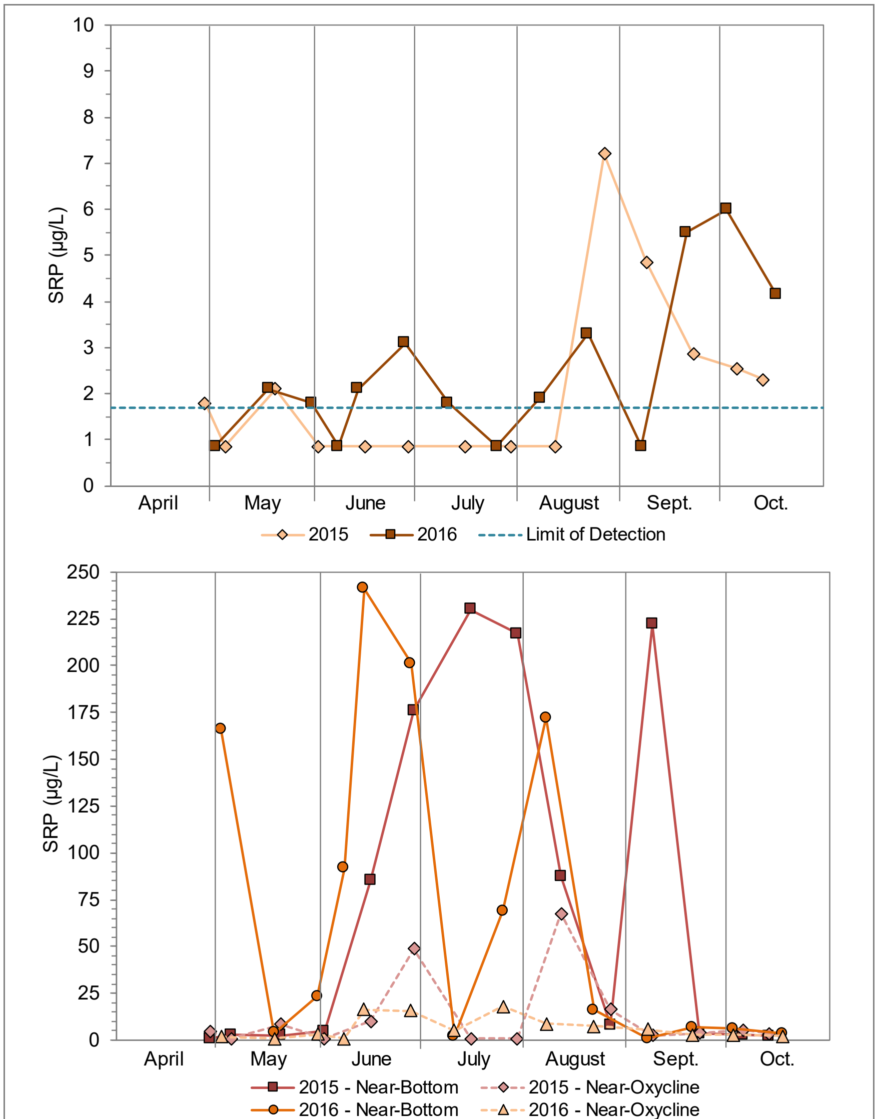

Figure 11. Kentuck Lake 2015 and 2016 growing season near-surface SRP concentrations (top) and near-oxycline and near-bottom SRP (bottom). Near-surface data represent mean concentrations collected from 2015 site 1, 2, 3, 4, & 5 and 2016 site 1 & 2. Near-oxycline and near-bottom concentrations collected from site 1.

11). In 2015 and 2016, near-bottom and near-oxycline SRP concentrations began to increase in June. In 2015, near-bottom SRP concentrations remained high through July and declined through August as the epilimnion grew deeper and the lake eventually experienced a complete mixing event. In 2016, near-bottom SRP concentrations declined to concentrations similar to near-surface concentrations in early-July despite a mixing event not occurring. Near-bottom concentrations then continued to increase into early-August before declining again for the remainder of the 2016 sampling period. Like near-oxycline TP concentrations, near-oxycline SRP concentrations in 2015 and 2016 were lower than near-bottom concentrations indicating a concentration gradient was present between the bottom and oxycline.

Nitrogen

In Kentuck Lake, nitrogen is the second-most important nutrient after phosphorus in terms of phytoplankton production. In 2015 and 2016, samples were collected to measure nitrate (NO_3^-) plus nitrite (NO_2^-) and ammonium (NH_4^-). Ammonia, like phosphorus, accumulates in the sediment under oxic conditions and can be released from the sediments under anoxic conditions. Studies have shown that cyanobacteria growth rates increased when their source of nitrogen was ammonia (Wetzel 2001 and McCarthy et al. 2013), and phytoplankton generally prefer to use ammonia over other forms of nitrogen (Beutel 2006).

In 2015, all measurements for nitrate plus nitrite were below the limit of detection with the exception of a near-bottom concentration collected in mid-June with a concentration of 21 µg/L. Similarly, nitrate plus nitrite concentrations measured in 2016 all fell below the limit of detection with the exception of detectable concentrations measured from the near-surface (90 µg/L), near-oxycline (44 µg/L), and near-bottom (171 µg/L) in early-August.

In 2015, near-surface measurements of ammonia remained below the limit of detection from late-April through late-July (Figure 12). In late-July 2015, near-surface ammonia concentration increased to 42 µg/L, an increase believed to be the result of a pulse from bottom waters as the epilimnion was driven approximately 10 feet deeper, cutting into the metalimnion. Near-surface ammonia concentrations increased further and reached a maximum in mid-August 2015 before fluctuating above and below the limit of detection into October.

Near-oxycline and near-bottom ammonia concentrations in 2015 began to increase with the onset of anoxia in bottom waters in late-May/early-June. Near-bottom ammonia concentrations reached a maximum concentration of 1,790 µg/L in late-June. From July through mid-August, near-bottom ammonia concentrations averaged approximately 1,200 µg/L before declining to 267 µg/L during the complete mixing event in late-August. Concentrations increased to 2,860 µg/L in early-September during a brief period of anoxia, and quickly declined in late-September with the return of oxic conditions. Like with TP and SRP, near-oxycline ammonia concentrations were lower than near-bottom concentrations indicating a concentration gradient between the oxycline and near-bottom.

In 2016, near-surface ammonia concentrations followed a similar pattern to those measured in 2015. However, near-surface ammonia concentrations were above the limit of detection during the first sampling event in early-May 2016 when the lake was briefly stratified (Figure 12). Near-surface ammonia concentrations were below the limit of detection from mid-May through late-June, and increased to just above the limit of detection in July through early-August with an average concentration of 17 µg/L. In late-August 2016, near-surface ammonia concentrations increased markedly to 182 µg/L corresponding with a complete mixing event, and declined to just above the limit of detection by early-

October. In mid-October, near-surface ammonia concentrations increased to 125 µg/L, presumably as the result of a brief period of stratification followed by mixing.

In 2016, near-bottom and near-oxycline ammonia concentrations were high when anoxic conditions were present (Figure 12). Near-bottom concentrations reached a maximum concentration of 3,930 µg/L in early-August. Near-bottom concentrations declined to 286 µg/L by early-September corresponding with the complete mixing event, and concentrations averaged 154 µg/L into October. Near-oxycline ammonia concentrations were on average six times lower than near-bottom concentrations, again indicating a concentration gradient between the near-bottom and oxycline.

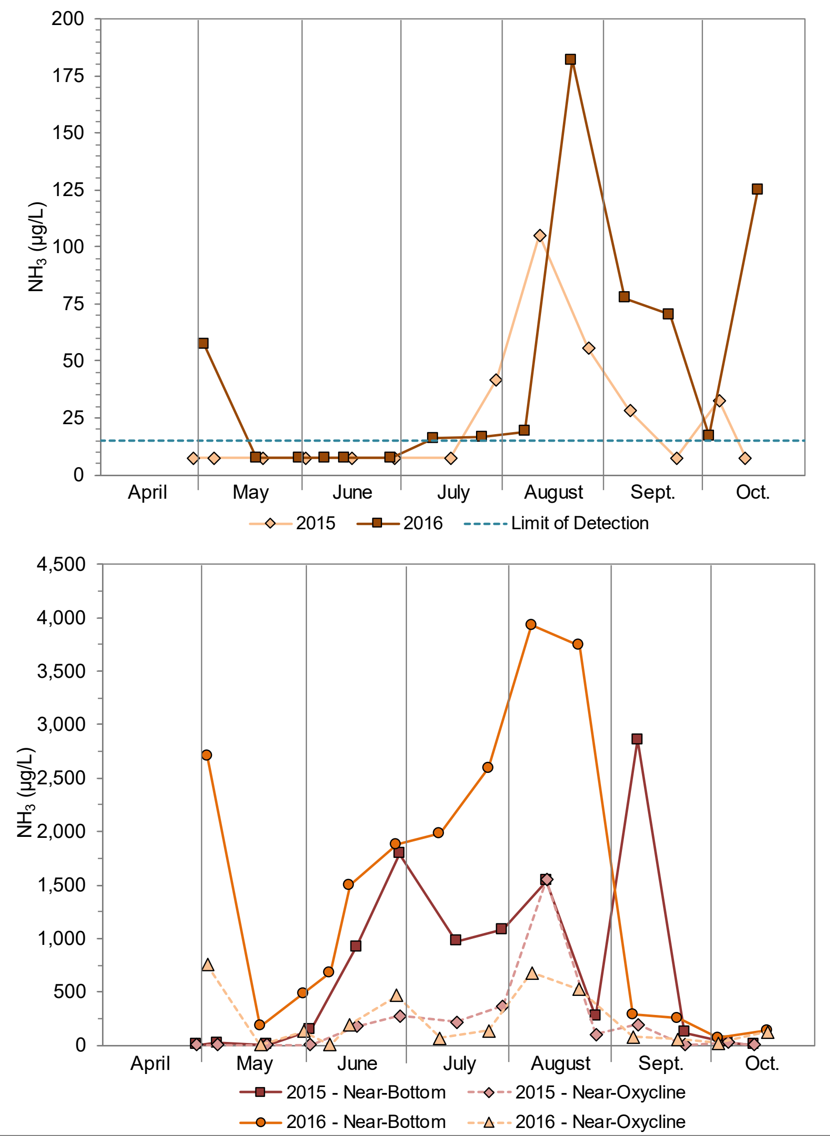

Figure 12. Kentuck Lake 2015 and 2016 growing season near-surface ammonia nitrogen concentrations (top) and near-oxycline and near-bottom ammonia nitrogen concentrations (bottom). Near-surface data represent mean concentrations collected from 2015 site 1, 2, 3, 4, & 5 and 2016 site 1 & 2. Near-oxycline and near-bottom concentrations collected from site 1.

Iron

As is discussed within the Primer on Internal Nutrient Loading Section (section 1.2), under anoxic conditions ferrous iron is released from bottom sediments into the overlying water after dissociating from phosphorus. Under oxic conditions, iron is relatively scarce as most of it exists as ferric iron, an insoluble form which rapidly precipitates to bottom sediments (Wetzel 2001). While it has been well documented that phosphorus and nitrogen are the primary nutrients which regulate phytoplankton biomass, ongoing research is indicating that ferrous iron produced under anoxic conditions and made available to phytoplankton at the surface is favoring cyanobacterial bloom production.

Wisconsin Department of Natural Resources scientists have identified a set of lakes in Wisconsin which produce a higher amount of chlorophyll-*a* relative to the concentration of phosphorus when compared to other lakes with similar phosphorus concentrations. Analyses of the lakes found that in the lakes with higher chlorophyll-*a* to phosphorus ratios had phytoplankton communities that were dominated by *nitrogen-*

Nitrogen fixation is the process in which nitrogen gas (N_2) in the atmosphere is converted into a usable form such as ammonium (NH_4^+) or nitrogen dioxide (NO_2).

fixing cyanobacteria. It is theorized that when nitrogen availability is limiting within the water, nitrogen-fixing cyanobacteria, able to access atmospheric nitrogen, outcompete other phytoplankton and become dominant. However, WDNR scientists found that nitrogen limitation occurred in both sets of lakes, those that were dominated by nitrogen-fixing cyanobacteria and those that were not. This was an indication that a factor other than nitrogen limitation was favoring the dominance of nitrogen-fixing cyanobacteria in these lakes.

Recent studies are indicating that the availability of ferrous iron provides nitrogen-fixing cyanobacteria with a competitive advantage over other phytoplankton (Molot et al. 2014). The nitrogen-fixing cyanobacteria have a higher iron requirement compared to other phytoplankton, and it increases their nitrogen fixing capacity. Laboratory studies have shown that additions of iron significantly increase the growth and production of cyanobacteria (Molot et al. 2010 and Sorichetti et al. 2014). The WDNR scientists found that the lakes which were dominated by nitrogen-fixing cyanobacteria experienced anoxia in bottom waters during the growing season and were polymictic, while the lakes which did not produce nitrogen-fixing cyanobacteria did not experience anoxia. They hypothesized that ferrous iron released into bottom waters was made available to nitrogen-fixing cyanobacteria at the surface through physical mixing and/or migration of cyanobacteria down to anoxic waters.

When phosphorus loading becomes high in productive lakes, the uptake of nitrogen by phytoplankton increases and they can deplete nitrogen sources within days or weeks (Wetzel 2001; Cole and Weihe 2016). Nitrogen limitation in combination with the availability of ferrous iron under anoxic conditions is currently the working hypothesis for the conditions required for nitrogen-fixing cyanobacterial bloom formation. To gain further insight into the role ferrous iron plays in cyanobacterial bloom formation, total iron concentrations were measured in Kentuck Lake in 2015 and 2016.

In 2015 and 2016, near-surface total iron concentrations were below the limit of detection at the start of sampling and reached detectable concentrations in June with the onset of anoxia (Figure 13). In 2015, near-surface total iron concentrations continued to increase through August before declining in September into October. In 2016, concentrations increased through July, declined in August, increased again in September, and declined in October. As mentioned earlier, iron is generally found in low concentrations in oxygenated water as ferric iron is insoluble and is quickly lost through sedimentation. The iron measured in oxygenated near-surface waters in Kentuck Lake beginning in June in 2015 and 2016 is believed to be iron which was mobilized from anoxic bottom waters and incorporated into phytoplankton at the surface, keeping it suspended within the epilimnion.

Near-bottom and near-oxycline concentrations of total iron in 2015 and 2016 followed a similar pattern to those of total phosphorus, soluble reactive phosphorus, and ammonia nitrogen (Figure 13). Near-bottom total iron concentrations increased rapidly with the onset of anoxic conditions, reaching a maximum of 8.3 mg/L in late-June 2015 and 16.6 mg/L in early-August 2016. Near-bottom total iron concentrations declined rapidly following the complete mixing events which occurred in late-summer of 2015 and 2016. Near-oxycline total iron concentrations were lower than those measured near the bottom, and like the other parameters measured, indicate a concentration gradient from the oxycline to the bottom.

Phytoplankton & Water Clarity

Chlorophyll-*a* concentrations in Kentuck Lake followed a similar pattern in 2015 and 2016 (Figure 13), and were most closely correlated with near-surface total phosphorus concentrations. Chlorophyll-*a* concentrations were low early in the growing season from late-April through early-June, averaging approximately 4 µg/L in both 2015 and 2016. Concentrations began to increase in early-June 2015 and in mid- to late-June 2016 with the onset of thermal stratification and anoxia. In 2015, concentrations reached a maximum concentration of 74 µg/L in late-July, and declined to an average of 40 µg/L through October. In 2016, reached a maximum of 84 µg/L in early-August, and declined to an average of 31 µg/L through October.

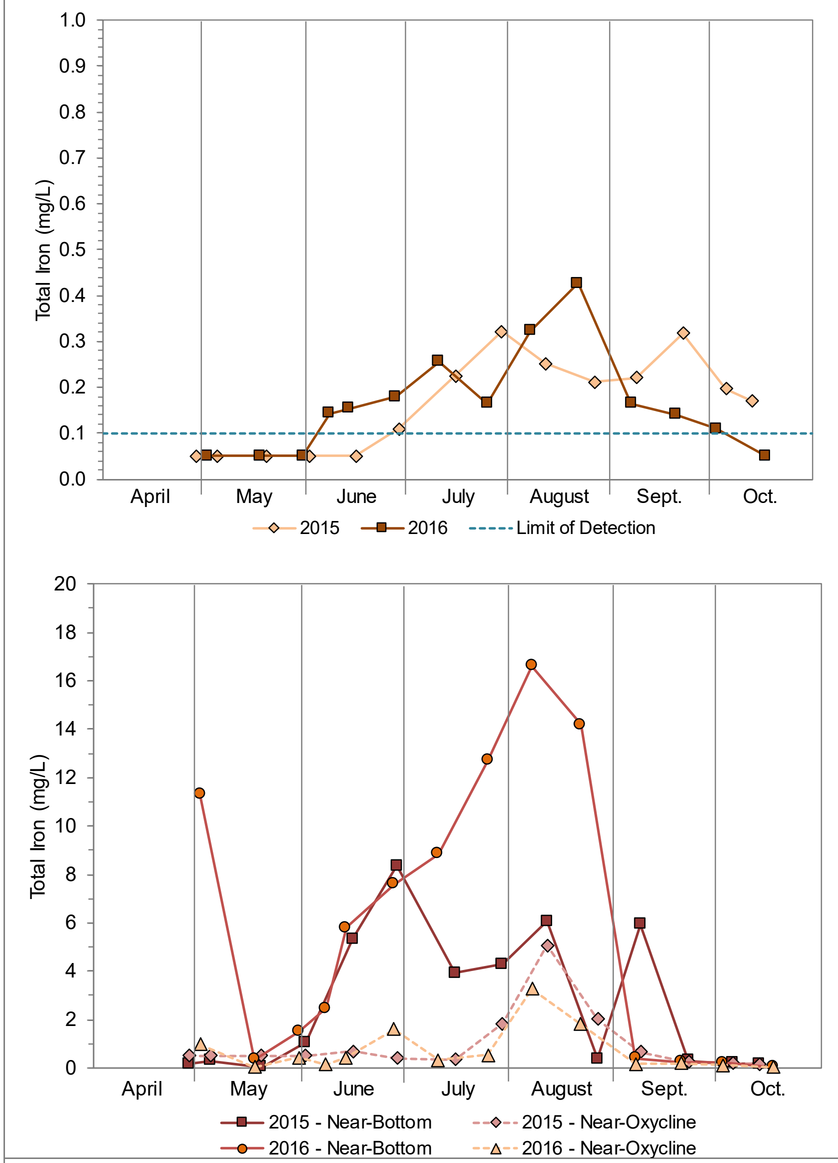

Figure 13. Kentuck Lake 2015 and 2016 growing season total iron concentrations (top) and near-oxycline and near-bottom total iron concentrations (bottom). Near-surface data represent mean concentrations collected from 2015 site 1, 2, 3, 4, & 5 and 2016 site 1 & 2. Near-oxycline and near-bottom concentrations collected from site 1.

Phytoplankton taxa analysis revealed compositional changes over the course of the growing season in Kentuck Lake's phytoplankton community in 2015 and 2016 (Table 3). In both years, the phytoplankton community was dominated by eukaryotic algae (not cyanobacteria) across all sample depths from the start of sampling through mid-June. In 2015, cyanobacteria began to dominate near-surface samples in mid-June soon after the onset of thermal stratification and anoxia. However, cyanobacteria did not become codominant in mid-depth and near-bottom samples until the end of July, and they remained codominant with eukaryotic algae for the remainder of the sample period.

In 2016, cyanobacteria became dominant or codominant with eukaryotic algae across all sample depths in mid-June, and remained dominant taxa within all subsequent samples. A total of 12 genera of cyanobacteria were identified from the samples collected in 2015 and 2016, but *Anabaena* sp., *Aphanizomenon flos-aqaue*, and *Gloeotrichia echinulata* were most often identified as the dominant taxa when cyanobacteria were dominant. All three of these genera are nitrogen-fixing cyanobacteria belonging to the order Nostocales and have the capacity to produce cyanotoxins.

Secchi disk depth, a measure of water clarity, in 2015 and 2016 was closely correlated (negatively) with chlorophyll-*a* concentrations (Figure 14). In both years, water clarity was highest in May, declining to minimum values in July and August, and increasing slightly in September and October. Average of all measurements within each year indicate 2016 Secchi disk depths were 1.6 feet deeper than those recorded in 2015, with the greatest differences in clarity occurring in May and June. Chlorophyll-*a* concentrations increased earlier in 2015 when compared to 2016, and this may account for greater clarity conditions persisting longer in 2016.

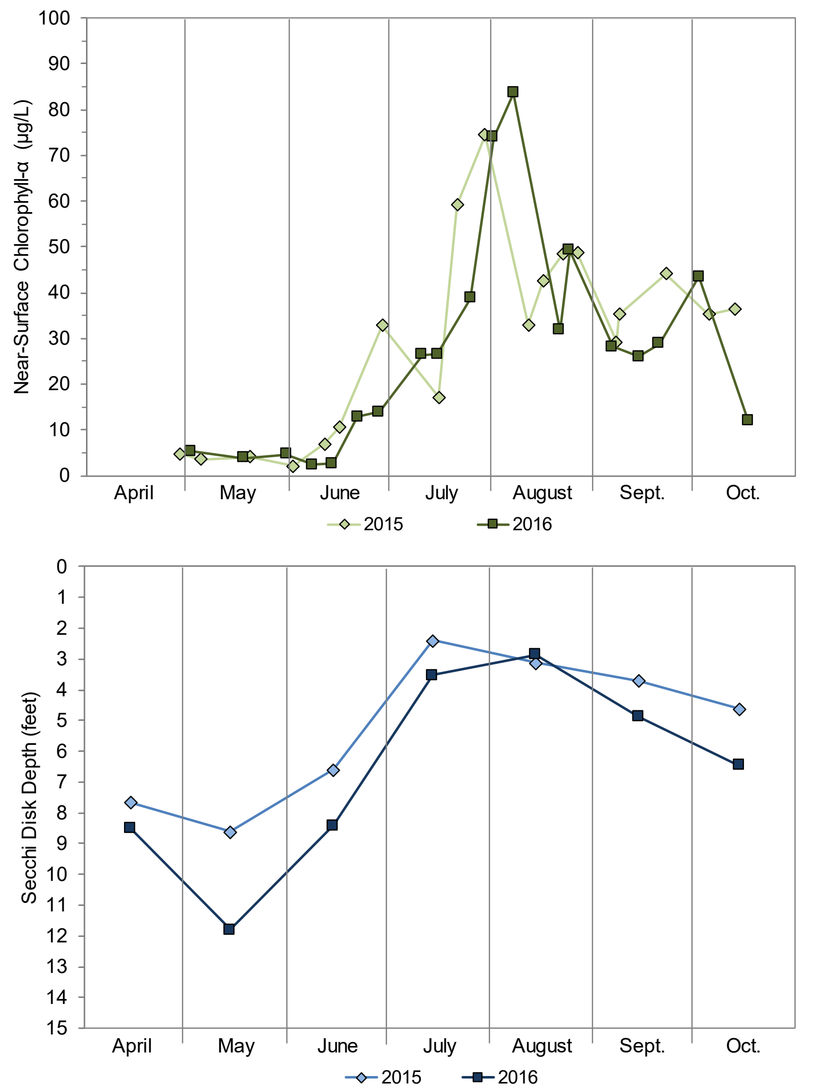

Figure 14. Kentuck Lake 2015 and 2016 growing season near-surface chlorophyll-α (top) and average monthly Secchi disk depths (bottom). Chlorophyll-α data represent mean concentrations collected from 2015 and 2016 site 1 & 5. Secchi disk depth data represent monthly means calculated from 2015 site 1, 2, 3, 4, & 5, and 2016 site 1 & 5.

Table 3. Dominant phytoplankton groups (eukaryotes and cyanobacteria) from Kentuck Lake over the 2015 and 2016 growing season. Samples collected from Deep Hole Location – Site 1. Analysis conducted by Kim Kreitlow (WDNR).

	Sample Location	April				May				June				July				August				September				October			
2015	Near-Surface (3 ft)				E	E	E	E	E	E	E	CE	C	C	C	C	CE	C	C	CE	CE	CE	CE	CE	CE	CE	CE		
	Near-Oxycline (depth varies)				E	E	E	E	E	E	E	E	E	E	E	E	CE	CE	CE	CE	CE	CE	CE	CE	CE	CE	CE		
	Near-Bottom (37 ft)				E	E	E	E	E	E	E	E	E	E	E	E	CE	CE	CE	CE	CE	E	CE	CE	CE	CE	CE		
2016	Near-Surface (3 ft)					E	E	E	E	E	E	CE	CE	C	C	CE	CE	CE	CE	CE	CE	C	C	CE	CE				
	Near-Oxycline (depth varies)					E	E	E	E	E	E	E	CE	C	C	CE	CE	CE	CE	CE	CE	CE	CE	CE	CE				
	Near-Bottom (37 ft)					E	E	E	E	E	E	CE	CE	CE	CE	CE	CE	CE	CE	CE	CE	CE	CE	CE	CE				

C = Cyanobacteria Dominant; E = Eukaryotic Algae Dominant; CE = Cyanobacteria & Eukaryotic Algae Codominant

3.3 Evidence for Internal Nutrient Loading

The near-bottom water quality data collected from Kentuck Lake's deep hole in 2015 and 2016 provide empirical evidence that the internal loading of nutrients from bottom sediments occurs when anoxic conditions are present. The concentrations of total phosphorus, soluble reactive phosphorus, total iron, and ammonia nitrogen, all of which increased markedly in near-bottom waters during anoxia indicate these nutrients are being released from bottom sediments into the overlying water. The mobilization of these nutrients from bottom sediments into the overlying water under anoxic conditions has been well documented (Beutel 2006, Özkundakei et al 2011, Orihel et al. 2015). The simultaneous increase of total phosphorus in near-surface waters with the onset of anoxia indicates these nutrients are being mobilized from bottom waters to the surface.

However, despite increasing nutrients in near-surface waters in mid-June of 2015 and 2016, temperature and dissolved oxygen data indicate that complete mixing events did not occur in these years until late-August. It is believed that sediment-released nutrients were mobilized to surface waters in 2015 and 2016 through a combination of entrainment and/or cyanobacterial migration. Entrainment is the continual erosion of the metalimnion and deepening of the epilimnion (Wetzel 2001). For example, southwesterly winds blowing across Kentuck Lake will cause epilimnetic water to pile up on the northeast end, physically raising the lake's surface elevation on this end of the lake. As the water builds up in the northeastern part of the lake, the weight of the water causes the metalimnetic water to push up at the southeastern part of the lake. This "sloshing" of the layers of water results in the nutrient rich deeper waters being translocated to the surface. Furthermore, the water moving in the metalimnion will generate turbulence across the barrier between the oxic and anoxic, further mixing sediment-released nutrients into the epilimnion. In other words, even if complete destratification does not occur, sediment released nutrients are transported into the surface waters during significant wind events. In deeper lakes this action is minimal as there is sufficient depth so that the high nutrient bottom waters do not come in contact with the turbulent zone in the upper part of the water column.

In 2015 and 2016, Kentuck Lake was weakly stratified, with an epilimnion, large metalimnion, and little to no hypolimnion. The data show a gradual deepening of the epilimnion over the course of the growing season with anoxia eventually passing across the barrier between the epilimnion and metalimnion. The entrainment of nutrients from bottom waters to the surface can be a significant source of nutrient loading in polymictic lakes (Kamarainen et al. 2009 and James et al. 2015). Rather than receiving a pulse of nutrients at the surface from complete mixing events, entrainment acts as a pump, more or less continuously transporting nutrients to the surface across a weak thermal gradient.

In addition to entrainment, it is possible a portion of the increase in nutrients measured at the surface is the result of cyanobacterial migration and the translocation of nutrients from the sediment to the water. *Gleotrichia*, *Anabaena*, and *Aphanizomenon*, dominant genera of cyanobacteria in Kentuck Lake, are known to be able to cause a significant increase in epilimnetic phosphorus through uptake of sediment phosphorus in spring and migration into the water column in summer (Barbiero and Welch 1992; Cottingham et al. 2015). *Gleotrichia* has been documented to stimulate the growth of other phytoplankton by releasing sediment-obtained nutrients into the water (Carey and Rengefors 2010).

In addition, it is believed these cyanobacteria have the ability to actively migrate to anoxic bottom waters, obtain sediment-released nutrients like phosphorus, iron, and nitrogen, and migrate back to the surface. While apparently alive cyanobacteria were observed in near-oxycline and near-bottom samples collected in Kentuck Lake in 2015 and 2016, it cannot be said if this was the result of active migration or the result of these cyanobacteria being passively mixed down to these depths by wind/water movement. The translocation of phosphorus by cyanobacteria was not quantified during this study, and it is not known if this process represents a significant portion of the internal phosphorus loading.

In February 2016, Onterra ecologists collected two small sediment cores through the ice, one in the southwest area of the lake and one near the deep hole (Map 1). These sediment cores were sent to Dr. William James at UW-Stout where phosphorus release rates were measured under oxic and anoxic conditions. Under oxic conditions, the phosphorus release rate was low with an average of 0.275 $mg/m^2/day$, while the phosphorus release rate under anoxic conditions was higher with an average of 3.13 $mg/m^2/day$. Using the phosphorus release rate under anoxic conditions and the estimated duration and area of anoxia as determined from dissolved oxygen profiles, it was expected that approximately 370 and 340 pounds of phosphorus would have been internally loaded from bottom sediments in 2015 and 2016, respectively.

Calculations of the net internal phosphorus load for 2015 and 2016 using total phosphorus concentrations measured during spring mixing and immediately following the first complete mixing event indicated the estimated internal phosphorus loads based on the sediment release rate were underestimated. The estimated net internal load ($P_{internal}$) of phosphorus in pounds in Kentuck Lake for 2015 and 2016 were calculated using the following equation from James et al. 2015:

$$\Delta P_{lake\ storage} = (P_{external\ load} - P_{outflow}) + P_{internal},$$

Where:

$\Delta P_{lake\ storage}$ is the change in total phosphorus mass in pounds within the lake between spring overturn and the late-summer mixing events.

$P_{external\ load}$ is the external load of phosphorus in pounds from the watershed between spring overturn and the late-summer mixing events as estimated from WiLMS.

$P_{outflow}$ is the estimated outflow of phosphorus in pounds from the lake between the spring overturn and late-summer mixing events as estimated from WiLMS.

The annual external load based upon land cover composition within Kentuck Lake's watershed indicated that approximately 484 pounds of phosphorus are loaded to the lake annually from external sources. WiLMS was also used to estimate the annual outflow of phosphorus from Kentuck Lake, which was estimated at approximately 629 pounds per year. The net internal phosphorus load for 2015 using measured total phosphorus concentrations was estimated at approximately 1,600 pounds, over four times higher than predicted based on the measured sediment release rate. Similarly, in 2016, the net internal phosphorus load was estimated at approximately 1,000 pounds, nearly three times higher than predicted based on the laboratory measured sediment release rate. Using these estimated net internal phosphorus loads, the sediment release rate of phosphorus within anoxic areas in 2015 and 2016 was estimated to be 13.0 and 9.0 $mg/m^2/day$. While these sediment release rate estimations are higher than what was measured within collected sediment cores, they are similar to the release rate measured in Cedar Lake of 12 $mg/m^2/day$ (James et al. 2015), a polymictic lake in Polk and St. Croix Counties with similar morphometry to Kentuck Lake. These estimated release rates are also similar to the median release rate of 10 $mg/m^2/day$ for eutrophic lakes reported by Nürnberg (1988) in a review of phosphorus release rates from anoxic sediments worldwide.

The estimated sediment release rate of phosphorus under anoxic conditions in Kentuck Lake is believed to be higher than those measured within the sediment cores under laboratory conditions for two primary reasons. One, Kentuck Lake's shallow nature allows for the continual entrainment of sediment-released phosphorus from near-bottom to surface waters. This continual draw or siphoning of sediment-released phosphorus increases the concentration gradient between the sediment and overlying water causing more phosphorus to be released from the sediment. Second, near-surface summer pH in 2015 and 2016 was exceeded 9.0, indicating phosphorus was likely also being released from oxic sediments in shallower areas around the lake. It is possible that a portion of the internal phosphorus load is the result of translocation by cyanobacteria, but as discussed previously, this was not quantified. The actual release rate of phosphorus from anoxic sediments in Kentuck Lake are believed to be higher than those measured in the laboratory due to entrainment but lower than those estimated from changes in the lake's phosphorus mass given phosphorus is likely also released from oxic sediments under high pH conditions. The estimated annual internal load of phosphorus indicates that approximately 80% of Kentuck Lake's annual phosphorus budget since 2011 originates from internal nutrient loading.

3.4 Drivers of Interannual Variability in Water Quality

As discussed earlier, data from 1988-2016 indicate that water quality in terms of nutrient concentrations, algal abundance, and water clarity is highly variable from year to year in Kentuck Lake. The data collected in 2015 and 2016 provided empirical evidence that nutrients are released from bottom sediments under anoxic conditions and subsequently mobilized to the surface. Spring and fall turnover phosphorus data are available for 24 of the 28 years between 1989 and 2016, and these data were used to estimate the amount phosphorus internally loaded within the lake each year using the method from James et al. (2015) as described previously. These estimates indicate a wide degree of interannual variability in the amount of internally-released phosphorus (Figure 15). The estimated annual internal release of phosphorus was positively correlated with summer average total phosphorus concentration ($R^2 = 0.659$) (Figure 15).

Given stored nutrients are released from bottom sediments under anoxic conditions with the onset of thermal stratification, it was hypothesized that annual differences in climatic conditions (primarily temperature and wind) which dictate the duration and strength of thermal stratification and thus anoxia would explain annual differences in nutrient concentrations. In other words, the longer the lake remains thermally stratified and anoxic the more nutrients will be released from bottom sediments into the overlying water. To test this hypothesis, annual water temperature and dissolved oxygen data, climatic data, and total phosphorus data were analyzed to determine if relationships existed between annual weather, thermal stratification, and phosphorus concentrations.

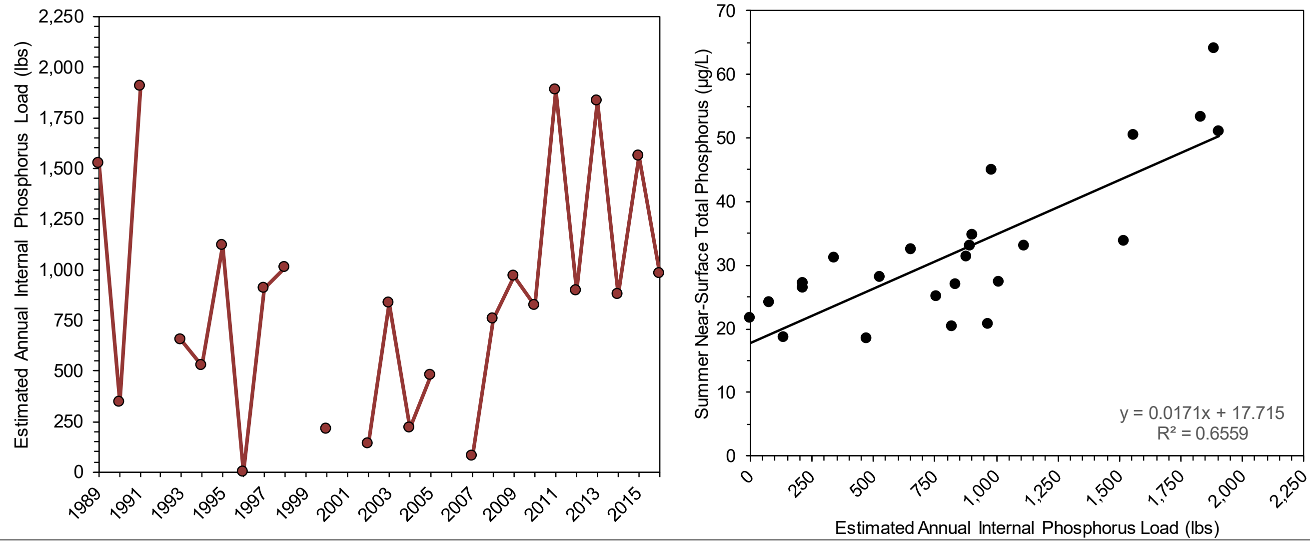

Figure 15. Kentuck Lake estimated annual internal phosphorus load (left) and average summer near-surface total phosphorus plotted against the estimated internal phosphorus load from 1989-2016.

To quantify the strength of thermal stratification, water column stability, S (g/cm) was calculated using the Schmidt Stability index formula where:

$$S = \frac{1}{A_0} \sum_{z=0}^{z=m} (p_z - p^*)(z - z_p)(A_z)(\Delta z)$$

Where:
A_0 = the area of the water body (cm^2)
A_z = lake area at depth z (cm)
p_z = density as calculated from temperature (g/cm^2)
p^* = lake's mean density (g/cm^2)
z_p = depth at which mean density is found (cm)
m = maximum depth of the lake (cm)
Δz = depth interval of measurements (cm)

Schmidt Stability calculates how much energy (wind) would be needed (g/cm) to cause a complete mixing of water column without the addition or subtraction of heat energy. A Schmidt Stability value of 0 indicates the entire water column is of uniform temperature (and density) and no energy is required to mix it. As thermal stratification strengthens and the temperature/density gradient increases between surface and bottom waters, the Schmidt Stability value increases indicating the lake is more resistant to mixing and more energy is required to mix the water column. Kentuck Lake's water column stability was calculated using temperature profile data collected prior to mid-summer mixing events and plotted against the minimum depth of measured anoxia (dissolved oxygen ≤ 2.0 mg/L) and shows that as water column stability increases, the minimum depth of anoxia declines (Figure 16). As the lake's resistance to mixing increases, the area of anoxia within the lake increases.

Given that the area of anoxia within the lake increases with water column stability, the relationship between water column stability and local weather conditions was examined. The onset, strength, and duration of stratification is going to be determined primarily by a combination of air temperature and wind. A Daily Climate Index (DCI) was developed to incorporate daily mean air temperature and daily mean wind speed into a single daily value where:

$$DCI = \frac{T_z}{\sqrt{W_z}}$$

Where
T_z = Daily average air temperature (°C)
W_z = Daily average wind speed (kph)

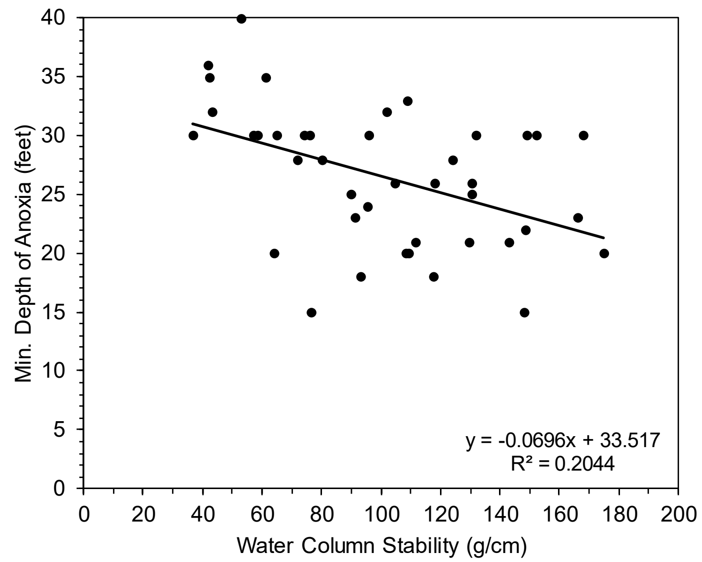

Figure 16. Kentuck Lake measured minimum depth of anoxia plotted against water column stability. Determined using data collected from the Deep Hole Sampling Location – Site 1 from 1998-2016.

Higher DCI values indicate days with higher average air temperatures and lower average wind speeds, conditions which favor higher thermal stability. Lower values indicate days with lower average temperatures and/or higher wind speeds, conditions which favor lower thermal stability. The mean DCI was calculated from 1 to 10 days prior to the collection of a temperature profile in Kentuck Lake. Simple linear regression indicated the best predictor of water column stability in Kentuck Lake prior to a complete mixing event is the 8-day mean DCI (R^2 = 0.753) (Figure 17). This analysis indicates that local climatic conditions of higher temperatures and lower wind speeds facilitate stronger water column stability in Kentuck Lake. As discussed previously, measured near-surface total phosphorus concentrations in 2015 and 2016 began to increase with the onset of thermal stratification and anoxia in near-bottom waters. Simple linear regression of near-surface total phosphorus and water column stability from 2011-2016 indicate that near-surface total phosphorus increases with increasing water column stability (R^2 = 0.601) (Figure 18).

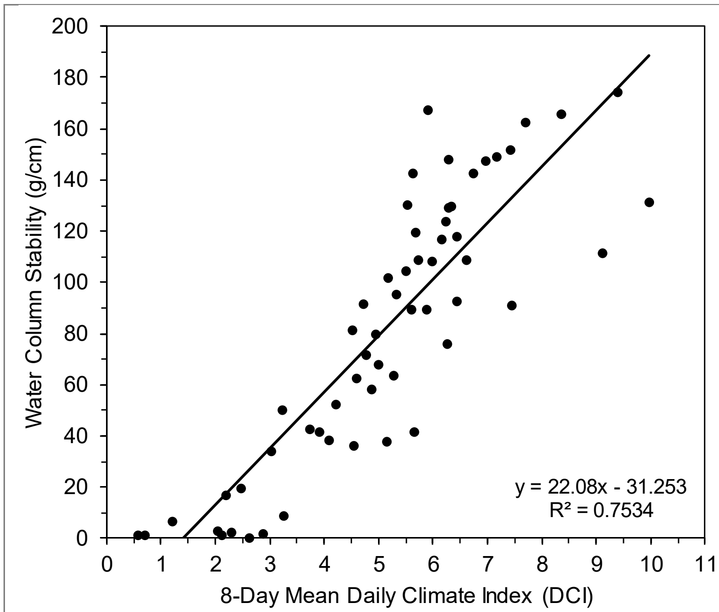

Figure 17. Kentuck Lake measured water column stability plotted against the calculated 8-day mean daily climate index. Created using available data from 2002-2016.

Plotting measured summer chlorophyll-*a* against summer water column stability shows an exponential relationship (R^2 = 0.722) (Figure 19). As mentioned previously, nuisance phytoplankton blooms are defined as those with chlorophyll-*a* concentration of >20 µg/L, and chlorophyll-*a* concentrations exceed 20 µg/L when summer water column stability exceeds 94 g/cm. In the high phosphorus/chlorophyll-*a* years since 2011 (2011, 2013, 2015, and 2016) measured water column stability in Kentuck Lake was maintained above 94 g/cm starting in June and persisting through late-August indicating prolonged thermal stratification and anoxia in near-bottom waters for most of the summer. In the lower phosphorus/chlorophyll-*a* years since 2011 (2012 and 2014), weather conditions facilitated early-season mixing events where thermal stability did not exceed or only briefly exceeded 100 g/cm just prior to mixing in June. In these years, early-season mixing allowed oxygenated water to mix deeper into the water column, reducing the area of anoxia and thus sediment nutrient release.

While water column stability was found to be a good predictor of near-surface total phosphorus and chlorophyll-*a* concentrations in Kentuck Lake from 2011-2016, there was no apparent relationship between these variables among available data from 1998-2010. Using the predictive equations generated from plotting water column stability against near-surface total phosphorus and chlorophyll-*a* from 2011-2016 (Figure 17 and 18), predicted near-surface total phosphorus and chlorophyll-*a* were calculated using water column stability from 1998-2010 and compared against concentrations. Plotting measured versus predicted near-surface total phosphorus concentrations from 1998-2010 indicates that phosphorus concentrations were significantly lower than predicted in a number of years with higher water column stability (Figure 20). Similarly, comparison of measured versus predicted chlorophyll-*a* concentrations from 1998-2010 indicated chlorophyll-*a* concentrations in multiple years with high water column stability were significantly lower than predicted (Figure 21).

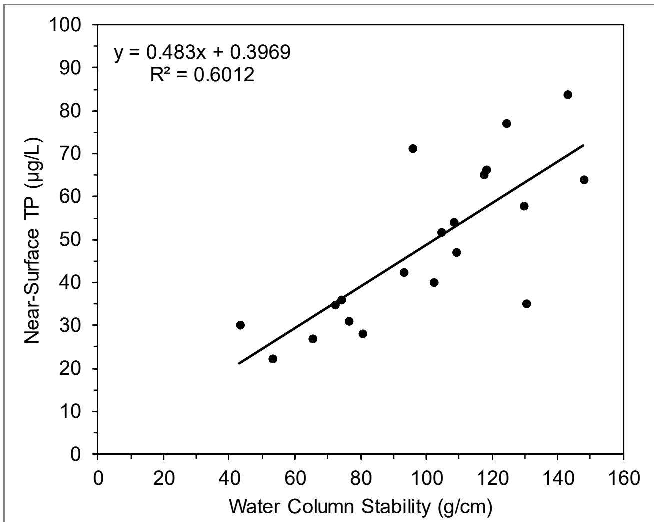

Figure 18. Kentuck Lake 2011-2016 summer near-surface total phosphorus plotted against measured water column stability.

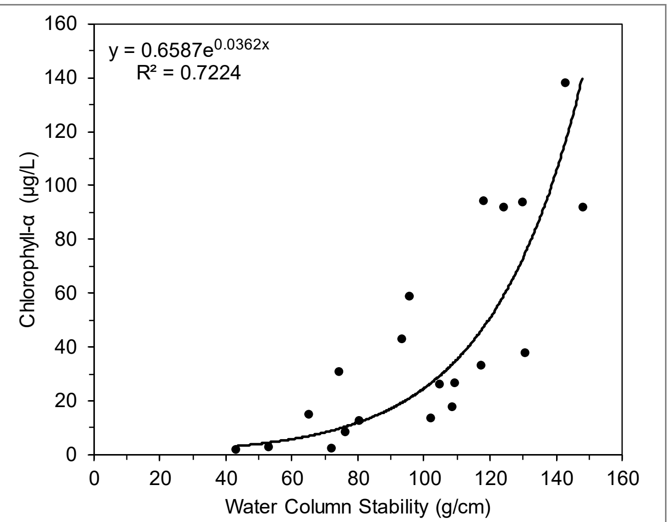

Figure 19. Kentuck Lake 2011-2016 summer chlorophyll-α plotted against measured water column stability.

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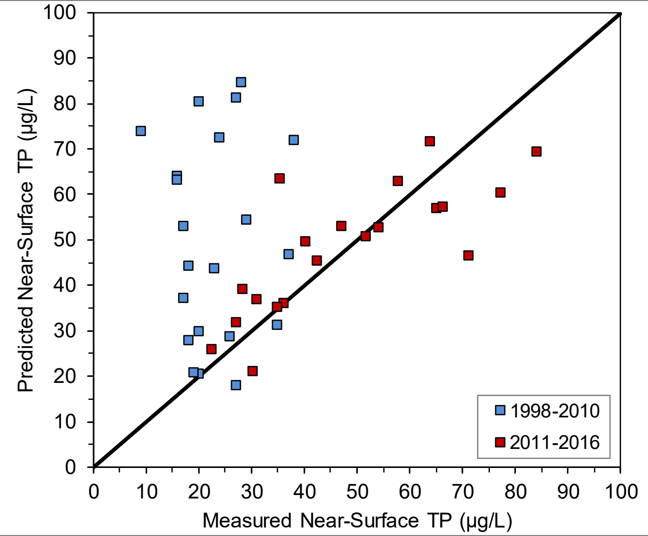


Figure 20. Kentuck Lake 1998-2010 and 2011-2016 measured versus predicted summer near-surface total phosphorus using water column stability.

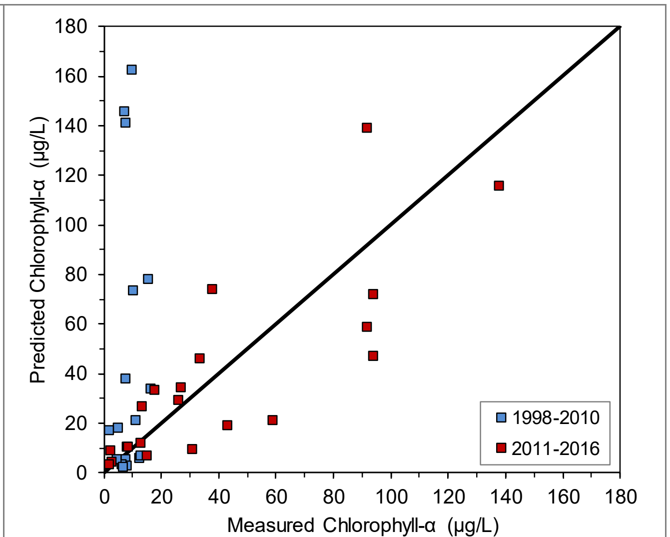


Figure 21. Kentuck Lake 1998-2010 and 2011-2016 measured versus predicted summer chlorophyll-α using water column stability.

Despite high water column stability and the development of anoxia in bottom waters in multiple years from 1998-2010, measured near-surface total phosphorus and chlorophyll-*a* concentrations were lower than predicted based on observations from 2011-2016. These results suggest prior to 2011 there was another process regulating phosphorus and phytoplankton abundance within the lake other than sediment released phosphorus. Numerous studies have documented the effects of trophic interactions on water quality (Carpenter et al. 2001, Benndorf et al. 2002, Søndergaard et al. 2007, Jeppesen et al. 2012), and it is hypothesized that changes in piscivore/planktivore abundance in Kentuck Lake may contribute the abrupt change in phosphorus and phytoplankton abundance between 2010 and 2011.

Piscivore is a term used to describe an animal which eats primarily fish.
Zooplanktivore is a term used to describe an animal which eats primarily zooplankton.
Zooplankton are plankton (small, floating organisms) consisting of small animals (e.g. crustaceans).

At a basic level, when *piscivores* (e.g. walleye) are abundant, predation reduces *zooplanktivore* (e.g. panfish) abundance and/or shifts zooplanktivore foraging behavior from open water to sheltered littoral areas with aquatic plants (Figure 22). The reduction in abundance and/or shift in foraging behavior of zooplanktivores selects for a *zooplankton* community with larger-bodied individuals (e.g. *Daphnia* spp.). Larger-bodied zooplankton eat phytoplankton at higher rates and have a wider range of the types of phytoplankton they consume when compared to smaller-bodied zooplankton. In addition, these large-bodied zooplankton sequester phosphorus and increase its sedimentation (fecal pellets) reducing its availability to phytoplankton. Under high piscivore/low zooplanktivore conditions, phytoplankton abundance and phosphorus concentrations are lower due to increased grazing upon phytoplankton and sequestration of phosphorus (Carpenter et al. 1987, Carpenter et al. 2001, Jeppesen et al. 2012).

In contrast, when piscivore abundance is low, zooplanktivore abundance increases and/or their foraging behavior shifts to open water. Increased predation upon zooplankton selects for a community comprised of smaller-bodied individuals which have lower feeding rates and a narrower range of phytoplankton types that they will consume. Under low piscivore/high zooplanktivore abundance, phytoplankton abundance and total phosphorus concentrations tend to be higher.

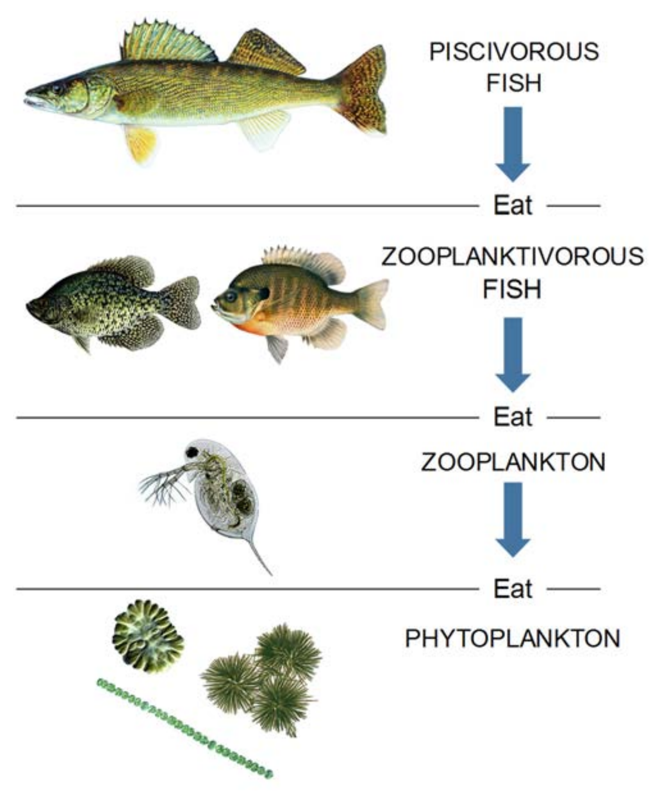


Figure 22. Simplified lake food web structure.

Annual walleye abundance data from Kentuck Lake collected by the Great Lakes Indian Fish and Wildlife Commission (GLIFWC) and the WDNR are available from 1989-2016. Annual panfish abundance data collected by the WDNR are available intermittently from 1999-2015. These data indicate that adult walleye density in Kentuck Lake varied widely from 1989-2016, ranging from 0.6 adult fish/acre in 2000 to 13.9 adult fish/acre in 2008 (Figure 23). Panfish abundance has also been highly variable, ranging from 6.4 catch per unit effort (CPUE) in 2004 to 239.1 CPUE in 2014 (Figure 23).

Given that historical or current data regarding the zooplankton community composition are not available, the food-wed driven effects on Kentuck Lake's water quality remain hypothetical. However, numerous studies have documented the reduction in phosphorus and chlorophyll-*a* concentrations in lakes where piscivore and zooplanktivore populations were artificially manipulated, particularly in shallow mesotrophic lakes. During the period of high walleye/low panfish abundance in Kentuck Lake, the estimated annual internal load of phosphorus was approximately 65% lower. While dissolved oxygen measurements indicated anoxia still developed during this period, it is thought the food web structure facilitated the redistribution of this phosphorus to the zooplankton community and increased is sedimentation back to the lake bottom. In addition, with reduced phytoplankton abundance during this period, the resulting higher water clarity likely allowed for deeper and more abundant aquatic plant growth. Increased aquatic plant growth and periphyton growing on the plants would likely have sequestered additional nutrients making them unavailable to phytoplankton.

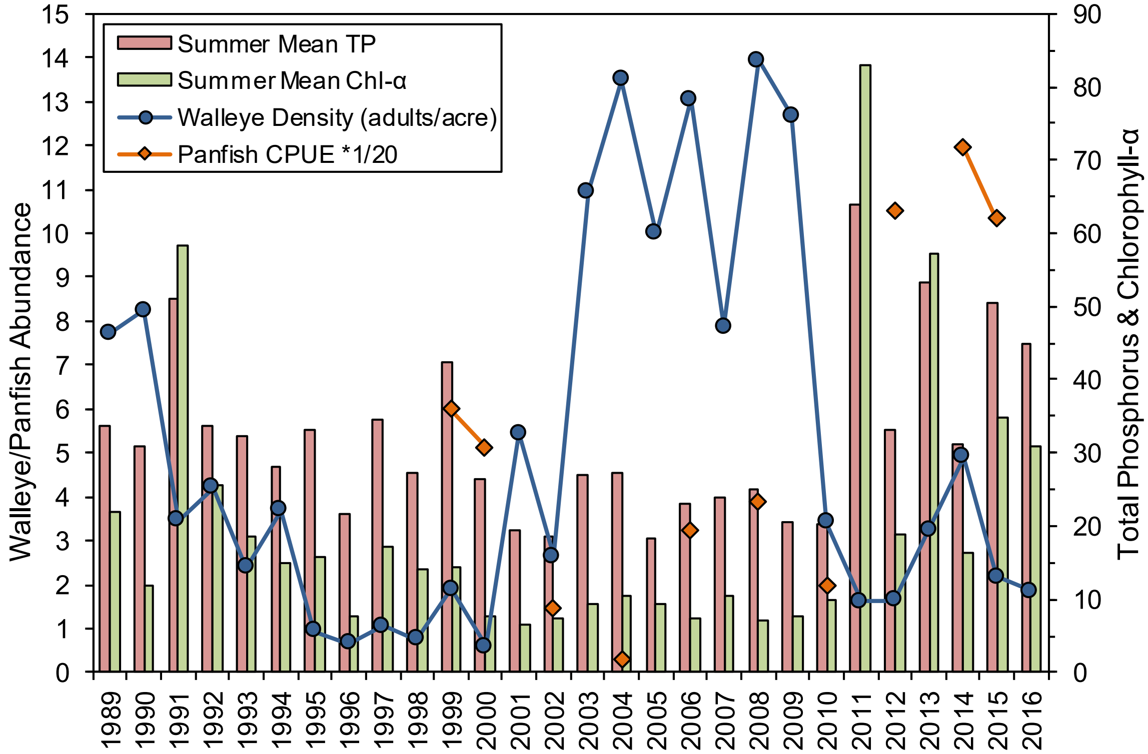


Figure 23. Kentuck Lake 1989-2016 walleye and panfish abundance and mean summer total phosphorus and chlorophyll-α concentrations. Walleye population data provided by GLIFWC. Panfish data provided by WDNR.

3.5 Paleoecology Results

Primer on Paleoecology and Interpretation

Questions often arise concerning how a lake's water quality has changed through time as a result of watershed disturbances. In most cases, there is little or no reliable long-term data. They also want to understand when the changes occurred and what the lake was like before the transformations began. Paleoecology, the examination of fossilized animals and plants and geochemical parameters to determine ecological conditions in the past, offers a way to address these issues. The paleoecological approach depends upon the fact that lakes act as partial sediment traps for particles that are created within the lake or delivered from the watershed.

The sediments of the lake entomb a selection of fossil remains that are more or less resistant to bacterial decay or chemical dissolution. These remains include frustules (silica-based cell walls) of a specific algal group called diatoms, cell walls of certain algal species, and subfossils from aquatic plants. The chemical composition of the sediments may indicate the composition of particles entering the lake as well as the past chemical environment of the lake itself. By collecting an intact sediment core, sectioning it off into layers, and utilizing all of the information described above, paleoecologists can reconstruct changes in the lake ecosystem over any period of time since the establishment of the lake.

Nearly all natural lakes in Wisconsin were created as a result of the last glaciation period. Most Wisconsin lakes are 12,000 to 14,000 years old as this is when the glacial ice sheets melted and receded from the state. The exception to this are lakes along Lake Michigan like those found in Door County.

These lakes are much younger, having been formed when the lake level of Lake Michigan dramatically dropped about 2,800 years ago. Although the newly formed lakes underwent significant ecological changes immediately after the recession of the glaciers, as the climate became warmer and drier, the last 150 years have generally seen the most dramatic changes to the lake's ecology due to the impacts from human settlement within their watersheds and along their shorelines.

Generally, Europeans began settling in Wisconsin after the 1830s in the southern part of the state and later in the northern part of the state. Early settlement largely consisted of subsistence farming in the lakes' watersheds which had minor but noticeable impacts on lake ecology. The greatest impact that settlement has caused to lakes occurred during the twentieth and twenty-first centuries. Often lakes with agriculture in their watersheds experienced significant degradation beginning in the 1940s and 1950s. Following World War II, mechanization improved allowing more land to be tilled. There was also an increased use of synthetic fertilizers to enhance production. Many of the factories that were used to produce ammunition for the war effort were converted to producing this fertilizer. The increased mechanization and use of fertilizers resulted in increased soil erosion from the land to the lakes as well as a large input of nutrients, e.g. phosphorus, that are attached to soil particles as well as associated with the fertilizer. Also, cow herd sizes increased, resulting in additional nutrients from manure. Since the 1970s, many parts of the state have experienced increased urbanization which has resulted in increased runoff from homes and streets into the lakes as well.

In northern Wisconsin, the earliest impacts to the lakes were from wide-spread logging operations. This activity generally had a short-term impact upon the lakes' ecology. With the failure of the agricultural experiment following the early logging in the late nineteenth and early twentieth centuries, tourism increased resulting in the addition of cottages around many lakes after the late 1920s (Davis 1995). Beginning around the 1970s, lake shore homes began to become larger and lawn maintenance more common. This increased urbanization resulted in increased delivery of sediment and nutrients to the lakes. This often resulted in large impacts on shoreland habitat as well as nearshore habitat.

Parameters Analyzed

There are many parameters that can be measured in a sediment core to reconstruct changes that have occurred in the lake throughout the time period covered by the core. The most frequently utilized are:

Sedimentation rate and dating is usually measured by the constant rate of supply model (Appleby 1998, 2001; Appleby and Oldfield 1978). The radionuclides lead-210 (^{210}Pb) and cesium-137 (^{137}Cs) are measured in samples throughout the core. Lead-210 is a naturally occurring radionuclide and is the result of natural decay of uranium-238 to radium-226 to radon-222. Since radon-222 is a gas (that is why it is sometimes found in high levels in basements) it moves into the atmosphere where it decays to ^{210}Pb. The ^{210}Pb is deposited on the lake during precipitation and with dust particles. After it enters the lake sediments, it slowly decays through the radionuclides described above. The half-life of ^{210}Pb is 22.26 years (time it takes to lose one half of the concentration of ^{210}Pb) which means that it can be detected and used for dating on samples that are about 130-150 years old.

Cesium-137 is a byproduct of atmospheric nuclear testing. This testing began in 1954 by the USA. Later the USSR also did testing. Atmospheric testing was banned in 1963 with the signing of the Atmospheric Test-Ban Treaty. Since the testing conducted by the USSR was much dirtier than the USA or UK, the peak deposition of ^{137}Cs was in 1963. Therefore, the peak concentration of ^{137}Cs in the core represents a date of 1963 (Krishnaswami and Lal 1978).

Another elemental profile that can be used to verify the dating model is that of stable lead. Stable lead has a historical pattern of deposition that is very consistent among lakes, with lead concentrations increasing from around 1880 to the mid-1970s, and decreasing to the present. The decline of lead is largely the result of the discontinued use of bonded leaded gasoline in the mid-1970s (Gobeil et al. 1995; Callender and Van Metre 1997).

Another useful parameter to estimate when early settlement occurred in the lake's watershed is changes in the loss on ignition (organic matter). Studies have shown that this decline is the result of watershed activities which result in an increase in the soil erosion (Engstrom et al., 1985; Garrison, 2000a,b; Garrison, 2003; Garrison and Wakeman, 2000). This erosion is largely composed of inorganic material which dilutes the organic matter and thus organic matter concentrations decline. The increased delivery of inorganic material in southern lakes occurred in the mid-1800s. In northern Wisconsin, the increase tended to occur around 1900.

Geochemical parameters are various chemicals deposited in the sediments. Some of them are useful for determining changes that have occurred in the lake and what watershed activities have caused the changes. Examples of chemicals analyzed are: phosphorus, nitrogen, carbon, aluminum, titanium, iron, manganese, uranium, zinc, and calcium. While some of these chemicals directly indicate changes in the lake's ecology, others act as surrogates for activities in the lake or watershed. Aluminum and titanium are surrogates for soil erosion as they are common components of clay particles in soils. Calcium, in the form of lime, is often used as a soil amendment in lawn maintenance, especially in sandy soils. While phosphorus and nitrogen are common components of synthetic fertilizers, their concentrations are easily affected by lake processes and thus not good indicators of fertilizer application on the landscape.

Instead, other components of the fertilizer are used to track changes in fertilizer use. Two components that are more conservative are potassium and uranium. Uranium is found in many phosphate ore bodies. Changes in the nutrients phosphorus, nitrogen, and carbon are often useful in determining changes in the lake's productivity. Changes in the concentrations of iron and manganese are used to estimate changes in oxygen content in the bottom waters of stratified lakes. As the bottom waters lose oxygen with increased lake productivity, more manganese is released from the sediments than iron. Therefore, a decline in the ratio of Fe:Mn is an indication of declining oxygen levels in the sediments. While changes in the concentration of these chemicals is often useful, changes in the accumulation rate provide a more accurate picture of changes within the lake. The accumulation rate is determined by multiplying the bulk sedimentation rate times the concentration.

The **diatom community** is one of the most useful fossils for reconstructing changes in the lake over time. Diatoms are a type of alga which possess siliceous cell walls and are usually abundant, diverse, and well preserved in sediments (Photo 4). They are especially useful as they are ecologically diverse and their ecological optima and tolerances can be quantified. In other words, certain species thrive in certain conditions, so the past ecological conditions are able to be reconstructed based on the species that were present. For example, some species prefer lower phosphorus concentrations while other species grow attached to benthic substrates such as aquatic plants.

Diatom assemblages have been used as indicators of trophic changes in a qualitative way (Bradbury 1975, Carney 1982, Anderson et al. 1990), but quantitative analytical methods exist as well. Ecologically relevant statistical methods have been developed to infer environmental conditions from diatom assemblages. These methods are based on multivariate ordination and weighted averaging regression and calibration (Birks et al. 1990). Ecological preferences of diatom species are determined by relating modern limnological variables to surface sediment diatom assemblages. The species-environment relationships are then used to infer environmental conditions from fossil diatom assemblages found in the sediment core. There are other types of analyses that are less frequently performed in sediment cores. These generally are not as universally useful as the more frequent analyses, but can help explain changes in lake's ecosystem in specific cases. Examples of these type of analyses are:

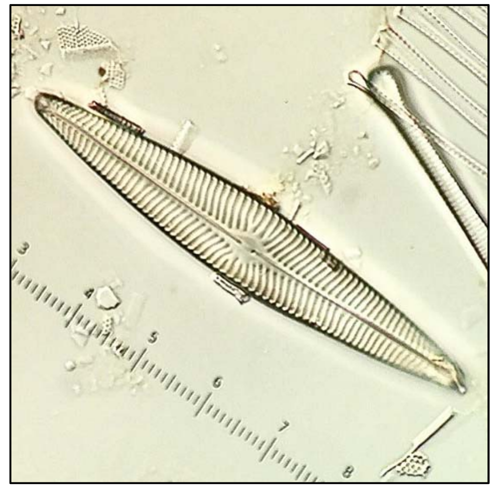


Photo 4. Fossilized diatom (*Navicula radiosa*) extracted from a lake sediment core. Photo credit Onterra.

Cyanobacteria (blue-green algae) are more common in eutrophic lakes and changes in their abundance can be an indication of increased algal blooms. Only a few species commonly leave fossils, but fortunately two of the three most problem causing taxa do leave fossils. These are *Anabaena* and *Aphanizomenon*. A third problem taxon, *Microcystis*, does not leave fossils which can confuse the interpretation of changes in algal blooms throughout the time period covered by the core. Cyanobacteria fossils are useful for demonstrating that a lake possessed these bloom-forming algae prior to the arrival of European settlers.

Zooplankton are microscopic animals that often feed on algae. While larger zooplankters consume more algae compared with smaller taxa, the larger animals are also more susceptible to fish predation. Unfortunately, not all zooplankton leave fossils. The only group that typically leaves enough fossils to be useful are cladocerans. Examples of these types of zooplankters are *Daphnia, Bosmina*, and *Chydorus*. The information from the zooplankton fossils can be used to estimate changes in fish predation and thus the type of fish present (fish eating vs plankton eating). These fossils can also be useful for documenting changes in the coverage of macrophytes as certain taxa are more common in plant beds. Zooplanktors were examined from the sediment core collected in Kentuck Lake; however, the results were inconclusive.

Core Types

There are two types of lake sediment cores that are usually collected for paleoecological analysis. The top/bottom core only analyzes the top (usually 1 cm) and bottom sections. The top section represents present day conditions and the bottom section is expected to represent pre-settlement conditions by having been deposited at least 100 years ago. While it is not possible to determine the actual date of deposition of bottom samples, a determination of the radionuclide lead-210 estimates if the sample was deposited at least 100 years ago. The primary analysis conducted on this type of core is the diatom community leading to an understanding of past nutrients, pH, and general macrophyte coverage.

A full core study, the method used in Kentuck Lake, retains the entire lake sediment core, usually in 1-2 cm sections. Typically, 15 to 20 sections are analyzed throughout the core. A much more detailed analysis of the sections is performed which results in much more detailed picture of the changes that have occurred throughout the time period encompassed by the core. Not only are ecological changes described, but often the cause of the changes is determined. Examples of analyses performed on full cores are changes in sedimentation rate, estimating dates when specific sections were deposited, geochemistry, and the diatom community. Occasionally, additional analyses are performed, e.g. zooplankton fossils, cyanobacterial fossils, and macrophyte remains.

Kentuck Lake Paleoecological Results

Analysis of the full sediment core collected from Kentuck Lake in 2015 indicates that during the last 200 years, the lake and its watershed have experienced a number of changes. In the early 1900s, the area was initially logged and there was at least one sawmill located on the northeastern lakeshore. When the area around Crandon was first being logged around the beginning of the twentieth century, many of the loggers were brought in from Kentucky (this may be where Kentuck Lake got its name). During the first half of the twentieth century, there was limited development on the lake shore but a couple of small resorts were present as well as a prison farm. By 1970 only about 40 residences were present around the lake (Table 4). By today's standards these cottages and resorts were primitive, used outhouses, and the buildings were small. Starting in the 1970s, many cottages were upgraded and additional homes built. These new structures were larger and greatly expanded the amount of impervious surface around the lake. This development occurred later around Kentuck Lake than many other northern Wisconsin lakes. By 1980, the number of homes had nearly doubled to 76 and by 2000 the number of residences was 135.

Table 4. Residential structures on Kentuck Lake from 1970-2000. Adapted from *Recreational Homes and Regional Development* by D. Marcouiller, University of Wisconsin Dept. of Urban/Regional Planning).

Year	Number of Residential Structures
1970	40 est.
1980	76
1990	105
2000	135

Sedimentation Rate

Sediment age for the various depths of sediment were determined by constant rate of supply (CRS) model (Appleby and Oldfield 1978). Bulk sediment accumulation rates ($g/cm^2/yr$) were calculated from output of the CRS model. The mean mass sedimentation rate for the last 200 years was 0.0084 $g/cm^2/yr$ which is lower when compared to 58 other Wisconsin lakes where the sedimentation rate has been measured (Figure 24). The relatively low rate is due to a combination of factors. First, Kentuck Lake is a softwater lake meaning there is little calcium carbonate deposition to the lake bottom. Second, being a shallow lake, sediment deposition is more widespread throughout the lake as opposed to a deep lake where the highest sediment deposition rate occurs in the deepest area of the lake. And third, Kentuck Lake has a relatively small watershed to lake area ratio, meaning the lake likely receives a lower rate if sediment input. The average linear rate is 0.29 cm/yr which equates to a sediment deposition rate of 0.1 inch of sediment per year.

To account for sediment compaction and to interpret past patterns of sediment accumulation, the dry sediment accumulation rate was calculated. The historical sedimentation rate was about 0.004 $g/cm^2/yr^-$ but the rate increased after 1900 (Figure 23). The highest sedimentation rate occurred at the top of the core where the rate was almost 0.02 cm^2/yr, five times greater than the historical rate. The increase in the sedimentation rate in the early part of the twentieth century was the result of early logging activity.

The rate further increased with early shoreland development prior to the 1970s. With more intensive development in the last 40 years the rate further increased and at the present time the rate is the highest measured in the last 200 years. The sedimentation rate profile of Kentuck Lake is similar to some other northern Wisconsin lakes that are relatively shallow and do not have significant inflowing streams (Figure 25). Nearby Anvil Lake has a much lower sedimentation rate but this lake has the lowest rate measured in Wisconsin lakes. In Figure 25, Sand Lake is the only drainage lake shown. Because of the sediment delivered from the stream, drainage lakes naturally have a higher sedimentation rate.

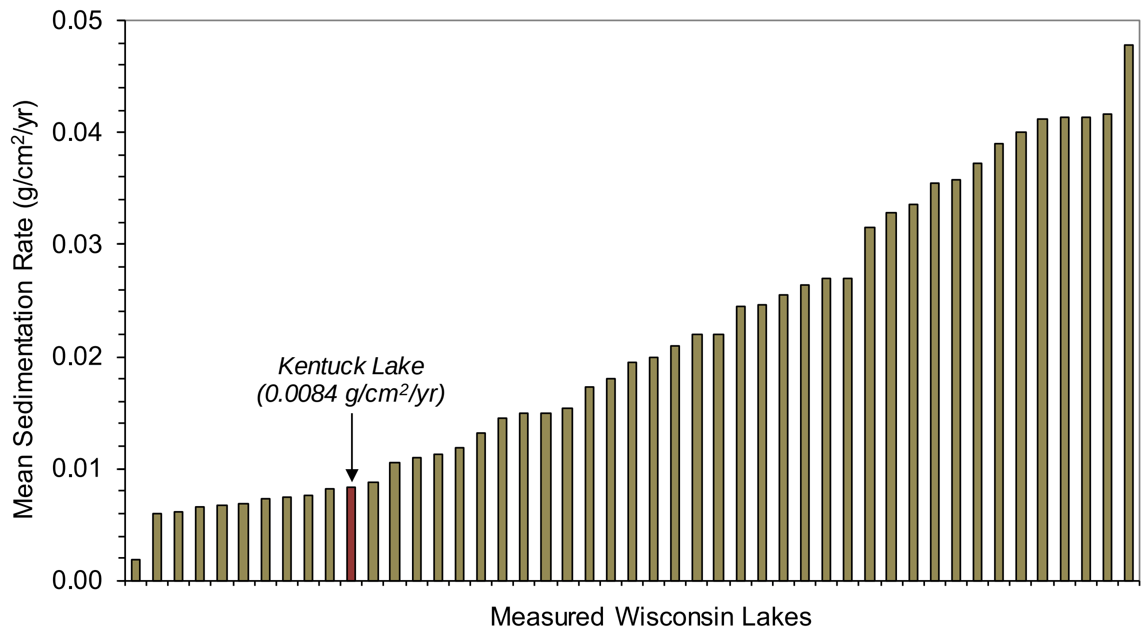


Figure 24. Kentuck Lake sedimentation rate compared to sedimentation rates measured from 58 other Wisconsin Lakes. Kentuck Lake is near the bottom third due to a combination of its shallow depth, low calcium concentration (softwater), and low watershed to lake area ratio.

Sediment Geochemistry

Historically, the sediments in Kentuck Lake contained a higher proportion of organic material when compared to inorganic material (Figure 25). With the onset of early logging activity around 1900, more inorganic sediment was delivered to the lake from the watershed and the composition of the sediment changed to nearly equal parts of organic to inorganic material. The change in the color from medium to dark brown occurred during this time period of early logging. With increased anthropogenic development and the removal of vegetation within the lake's watershed, more inorganic sediment was delivered to the lake, and between the period 1900-1980 inorganic material increased to comprise the majority of the sediment composition. During the last approximately 15 years, deposition of inorganic sediment has declined indicating reduced erosion from the watershed, but the amount of organic sediment has increased and the highest level in the core was in the surface sediments. Part of the reason the surface sediments contain a higher amount of organic sediment is that the organic matter has not had time to degrade. However, more importantly, it is likely the higher amount of organic sediment at the top of the core is the result of increased the productivity the lake has experienced since 2000.

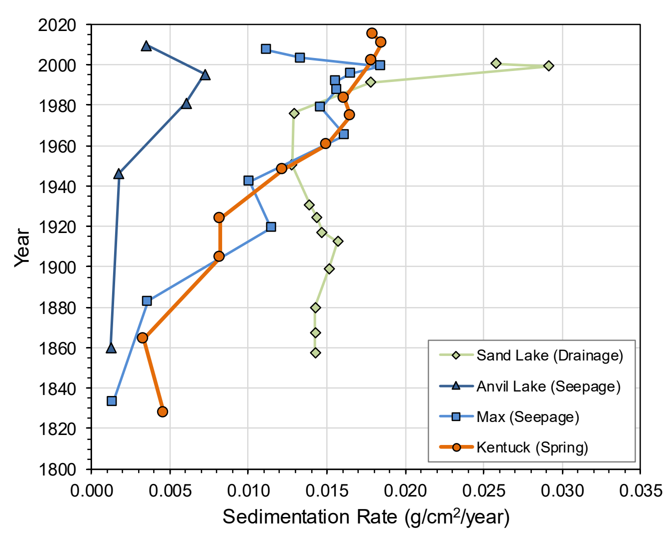


Figure 25. Kentuck Lake sedimentation rate compared to sedimentation rates from select northern Wisconsin lakes. Nearby Anvil Lake has the lowest rate measured in Wisconsin lakes. Kentuck Lake is similar to other northern Wisconsin lakes that do not have a significant inflowing stream. Sand Lake in Sawyer County is a drainage lake with an inflowing stream which results in having a higher historical rate.

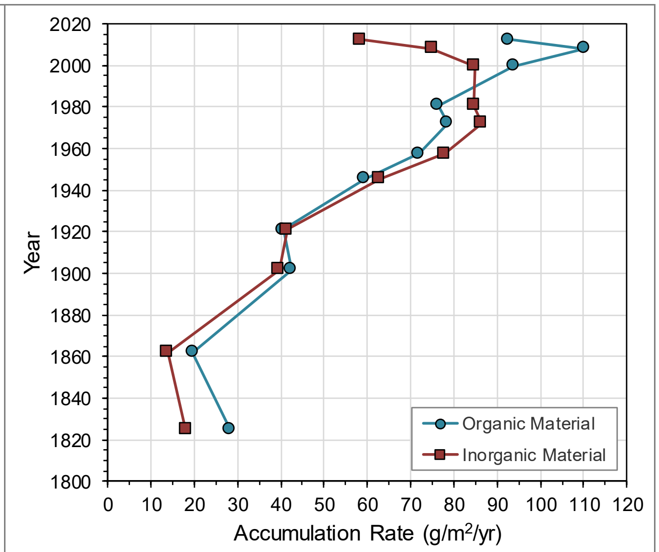


Figure 26. Kentuck Lake accumulation rate of inorganic and organic components within the sediment core. Historically, organic sediment was the main component. With early watershed activities, more inorganic material was delivered to the lake. During the last 15 years, soil erosion has declined but the lake's productivity has increased so that the organic component is greater than the inorganic component.

Selected geochemical parameters were analyzed in the core to better understand how activities in the watershed have impacted the lake. Other parameters are used to determine changes that have occurred within the lake itself. Aluminum is largely found in soil particles and its profile indicates the history of soil erosion around the lake. Prior to Euro-American settlement, aluminum deposition rates were very low (Figure 27). Deposition rates increased in the early 1900s as a result of logging activities but the rate increased even higher after 1945 as a result of human development around the lake. Even though agricultural activity was limited during the twentieth century and lake shore development was limited before the 1970s, the steep topography found in areas of the lake's watershed resulted in significant soil erosion to the lake until 2000. Since 2000 soil erosion has declined but remains much higher than pre-Euro-American settlement rates.

Even though the proportion of inorganic to organic material and aluminum measured within the core indicate that soil erosion has been declining since 2000, calcium, phosphorus, and nitrogen deposition rates have continued to increase (Figure 27). While these elements are found in soil particles, if soil erosion from the watershed was the source they would have been expected to decline similar to that of aluminum. The fact that aluminum deposition has declined since 2000 while calcium, phosphorus, and nitrogen have continued to increase indicates there is another source of these elements to the lake. The continued increase of calcium deposition since 2000 is most likely from soil amendments applied to properties within the watershed for lawn maintenance. Calcium is often used in the form of lime and is applied to lawns to increase soil pH. Prior to 2000, the increase in calcium deposition is most likely from soil erosion.

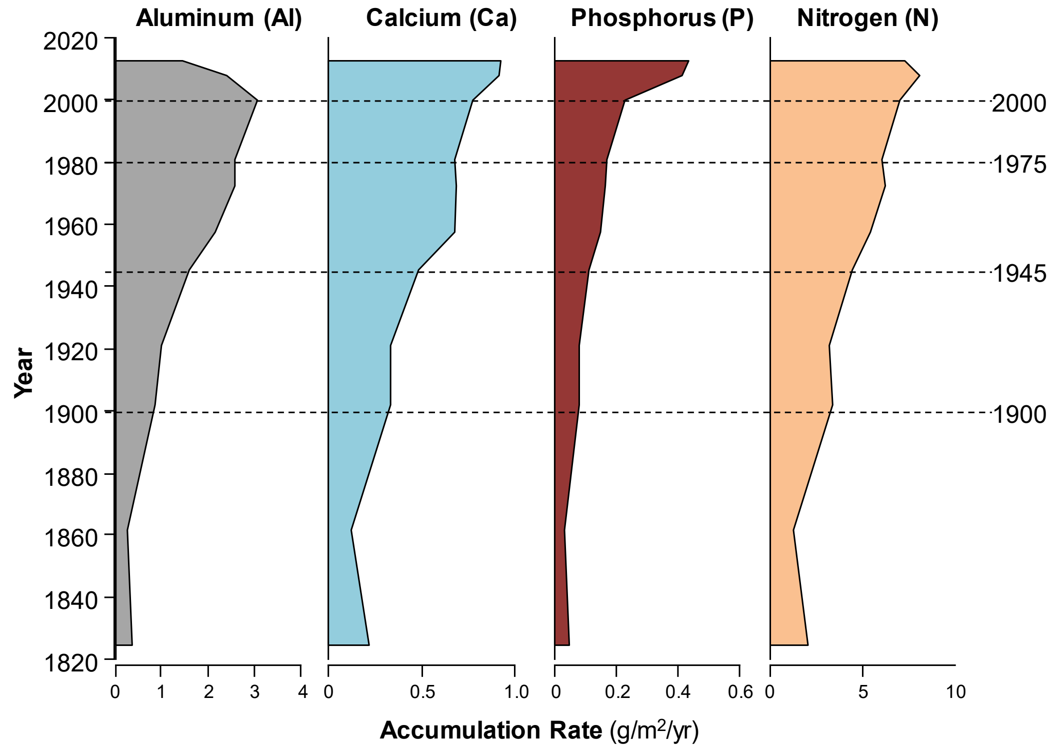


Figure 27. Sediment deposition rates of select geochemical elements within the Kentuck Lake sediment core. Aluminum is found in soil particles and is an indication of soil erosion. Calcium enters the lake from soil erosion but also is applied to lawns in the form of lime. Phosphorus and nitrogen are essential nutrients for plant growth and enter the lake from soil erosion, commercial fertilizer, and release from anoxic lake sediments.

As mentioned, like aluminum and calcium, both phosphorus and nitrogen also enter the lake attached to soil particles. Prior to 2000, both these elements follow a similar trend as aluminum and calcium with initial increases in deposition beginning around 1900 with a larger deposition rate after 1945. This is similar to the aluminum and calcium profile indicating that soil erosion is the primary source of these nutrients during this period. However, like calcium and unlike aluminum, both phosphorus and nitrogen have continued to increase since 2000, especially phosphorus. While the application of lawn fertilizers is likely a contributing source of these nutrients, phosphorus has been banned from lawn fertilizers since 2009 in Wisconsin, and the large increase in phosphorus deposition indicates that there is likely another source to the lake.

The ratios between geochemical elements are also examined to better understand the sources and changes in the deposition rates over time (Figure 28). The decline in the ratio of aluminum to calcium (Al:Ca) after 2000 provides further evidence that calcium deposition within the lake since 2000 is most likely originating from the application of lime and not from soil erosion within the watershed. Similarly, the ratio of aluminum to potassium (Al:K) also declines after 2000. Potassium is an important component of lawn fertilizer, and the decline in the Al:K ratio after 2000 is another indication of fertilizers making their way into the lake. The ratio of calcium to phosphorus (Ca:P) declined after 1975, indicating that phosphorus likely began originating from a source other than soil erosion and the application of lawn fertilizers. If lawn fertilizers were largely responsible for the increase in phosphorus since 2000, it would be expected that the Ca:P ratio should have remained the same rather than declining given calcium is often a common component of lawn fertilizers. While phosphorus has been banned from personal lawn fertilizers in Wisconsin since 2010, lake property owners should forgo the use of lawn care fertilizers if possible in an effort to enhance water quality and refer to best management practices for shoreland properties outlined within the Kentuck Lake Comprehensive Management Plan.

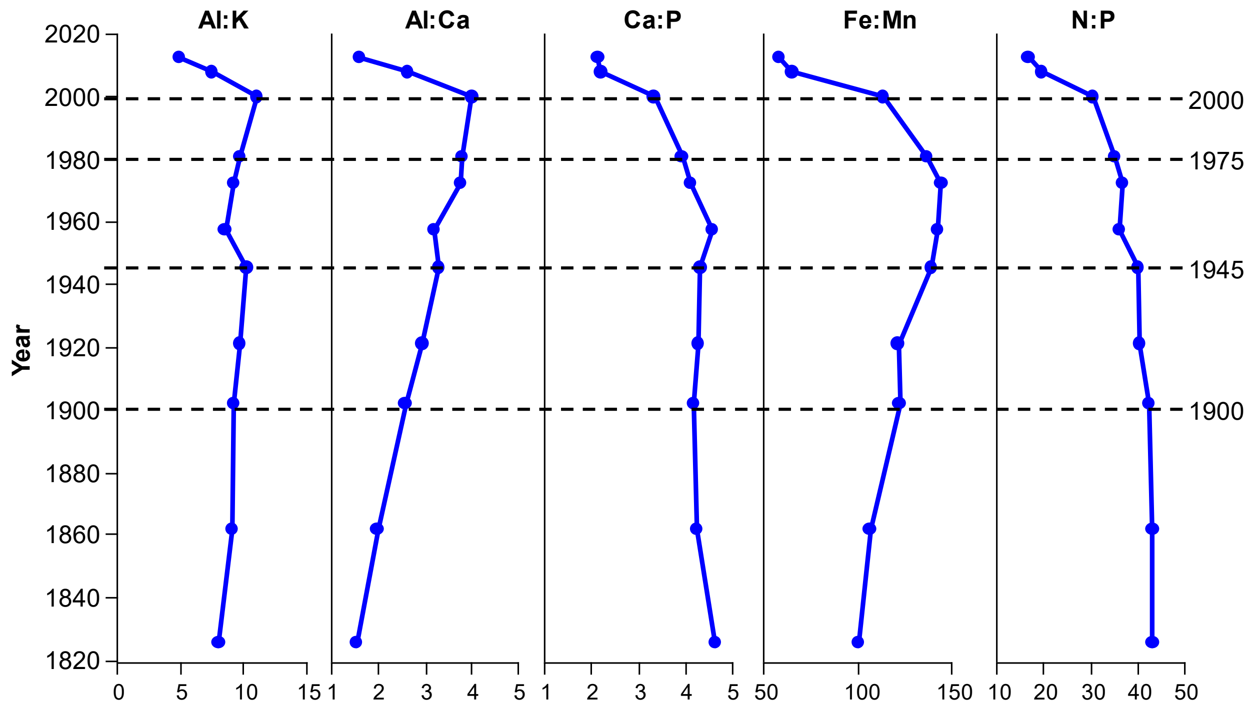


Figure 28. Select geochemical element ratios within the Kentuck Lake sediment core. The decline in Al:K and Al:Ca since 2000 indicates that soil amendments are being applied to lawns. The decline in Fe:Mn which begins in the mid-1970s and is accelerated after 2000 indicates a loss of oxygen in the bottom waters of the lake. This results in release of sediment bound phosphorus and nitrogen and exacerbates algal blooms.

The ratio of iron to manganese (Fe:Mn) is used to assess changes in dissolved oxygen concentrations in near-bottom waters over time. As bottom waters become increasing devoid of oxygen, manganese is mobilized from the sediments. This manganese then moves into the deeper waters resulting in enrichment of manganese in the sediments of the deeper waters. While this also occurs with iron, it happens sooner with manganese as it goes into solution sooner (Jones and Bowser 1978). Therefore, as the bottom waters lose oxygen, manganese is preferentially moved with respect to iron (Engstrom et al. 1985). The result is that with the loss of oxygen, the ratio of iron to manganese (Fe:Mn) declines (Mn increases).

In Kentuck Lake, the Fe:Mn ratio increases during the period 1820 to 1975 primarily as the result of soil erosion (Figure 27) indicating little change in the oxygen content in the bottom waters of the lake. However, in the late-1970s, the Fe:Mn ratio begins to decline, and a rapid decline is observed after 2000. This decline is an indication that the bottom waters in the lake began to lose their oxygen in the 1970s, but this loss rate accelerated after 2000. With this loss of oxygen, phosphorus is released from the sediments and mobilized to the surface causing increased phytoplankton growth. While internal nutrient loading likely started in the late-1970s, it has increased significantly since 2000. This internal loading explains the increase in phosphorus deposition rates since 2000 despite a reduction in soil erosion. Even though nitrogen in the form of ammonium is also released from the sediments during anoxic conditions, the ratio of nitrogen to phosphorus (N:P) has declined since 2000. The majority of the nitrogen released from bottom sediments is denitrified and escapes from the lake into the atmosphere rather than being deposited back to the sediment.

The onset of anoxia in bottom waters beginning in the late-1970s as indicated by the ratio of Fe:Mn within the sediment core corresponds to anecdotal ecological changes within the lake during this period. Historical accounts as recent as the early-1970s by both Kentuck Lake users and WDNR biologists indicate cisco (*Coregonus artedi*), a coldwater whitefish species, was once abundant in Kentuck Lake. Cisco, like other coldwater fish species, require cooler, oxygenated water to survive during the summer. Therefore, cisco are restricted to inland waters which can maintain oxygenated conditions in bottom waters during the summer, and these lakes are usually deeper with good water quality. Cisco are able to tolerate water temperatures up to 20°C (68°F) and dissolved oxygen concentrations as low as 3.0 mg/L (Frey 1955). Temperatures exceeding 20°C or dissolved oxygen concentrations lower than 3.0 mg/L are considered to the lethal conditions for cisco. As the cooler bottom waters in Kentuck Lake became increasingly devoid of oxygen in the late-1970s, the cisco, unable to tolerate both the low-oxygen conditions in deeper waters and the warmer temperatures found in shallower waters found themselves with ever-diminishing habitat. The elimination of the cool, oxygenated bottom water habitat in Kentuck Lake has resulted in the elimination or near elimination of cisco from this lake (Steve Gilbert, personal comm. 2015).

The geochemical analyses of the sediment core collected from Kentuck Lake indicate that internal nutrient loading increased significantly around 2000. However, as is discussed in Section 3.4, the period from 2000-2010 saw the lowest measured near-surface total phosphorus concentrations (average 23 μg/L) within the available historical dataset from 1988-2016. This seemingly contradictory information between geochemical elements within the sediment core and total phosphorus concentrations measured within the water strengthens the hypothesis that food web dynamics discussed in Section 3.4 may play a role in Kentuck Lake's water quality.

Diatom Community

Analysis of fossilized diatoms within the Kentuck Lake sediment core indicate the diatom community was historically composed of near equal amounts of planktonic (free-floating) and benthic (growing on the bottom or attached to plants) diatoms (Figure 29). This is not surprising as this lake is relatively shallow and it would be expected that benthic taxa would be an important part of the diatom community. The first detectable change in the diatom community and indication of a change in the lake's water quality occurred around 1900 with the onset of logging. Planktonic diatoms were still an important component of the diatom community but there was a decline in *Fragilaria tenera* and *F. capucina* var. *gracilis* and an increase in *Stephanodiscus niagarae* and *Cyclotella michiganiana*. The former taxa prefer lower phosphorus concentrations than do their replacements. Around the mid-1940s, benthic diatoms became more common with the increase in *F. pinnata* var. *acuminata* and *Staurosira construens*. The diatom community indicated a further change in the lake's water quality after 2000 with the near disappearance of planktonic taxa and an increase in small *Fragilaria* and other benthic *Fragilaria*.

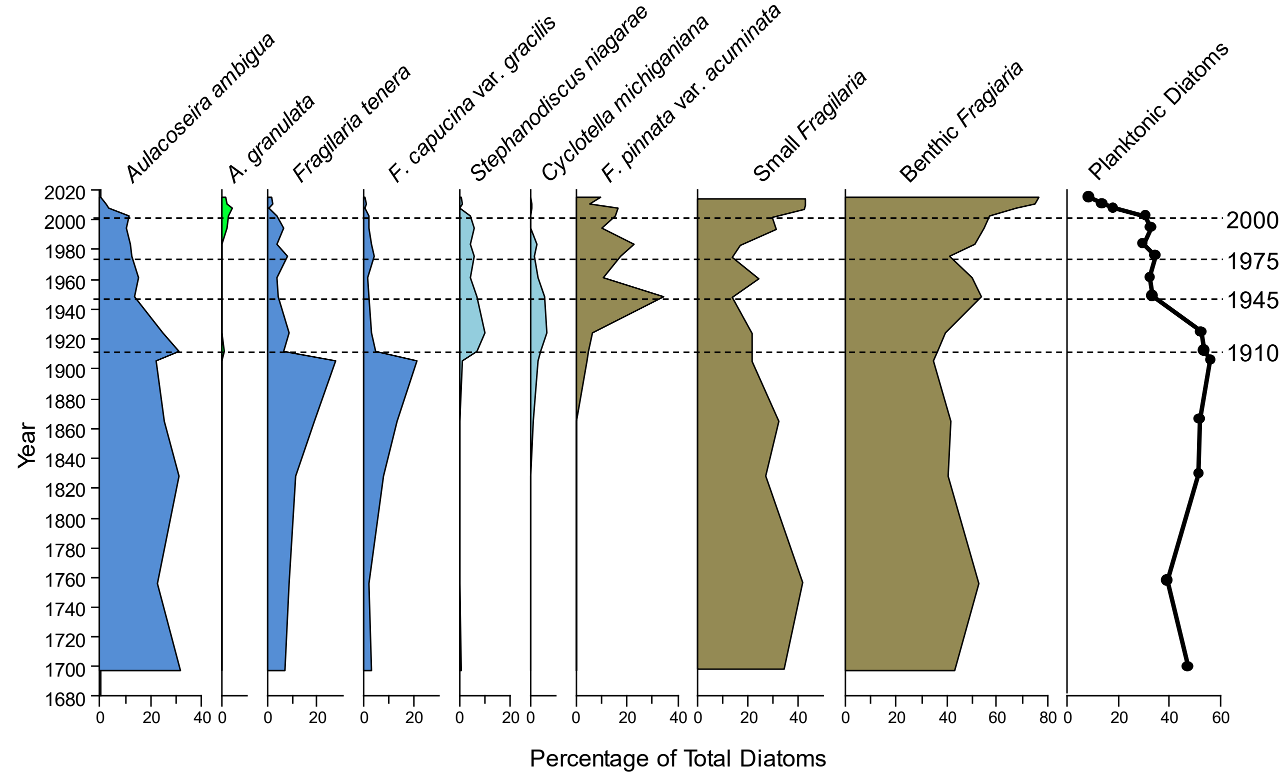


Figure 29. Abundance of common diatom taxa within the Kentuck Lake sediment core. Diatoms colored blue/green are planktonic diatoms which are found floating in the water column. Dark blue represents low phosphorus levels, lighter blue indicates moderate phosphorus levels, and green represents higher phosphorus levels. The diatoms colored brown are benthic taxa and grow attached to substrates such as plants and bottom sediments.

Relationships among diatom communities within the sediment core were explored using Principal Components Analysis (PCA), using CANOCO 5 software (ter Braak and Smilauer 2012). Core depths/dates were plotted in ordinate space and their relationships and variability used to identify periods of change, sample groups, and ecological variability among core samples. A general rule for interpreting a PCA is that samples that plot closer to one another have more similar assemblages.

Based upon a principal components analysis, four biostratigraphic zones were identified in the sediment core (Figure 30): Zone 1 (1697-1865), Zone 2 (1904-1924), Zone 3 (1948-1983), and Zone 4 (1994-2016). Zone 1 represents the time period prior to the arrival of Euro-Americans. Zone 2 represents changes in water quality brought about by increase erosion from early logging activity in the watershed. Zone 3 represents further changes to water quality as a result of early residential structure construction into the period when more extensive shoreland development was beginning to occur. Zone 4 represents the changes that have occurred over the last 20 years when shoreland development became more extensive and the geochemistry analysis indicated that Kentuck Lake's water quality has undergone its most significant change.

Weighted averaging calibration and reconstruction (Birks et al. 1990) was used to infer historical water column summer average phosphorus concentrations in the sediment core based on the diatom community. A training set was developed from 89 shallow Wisconsin lakes. Training set species and environmental data were analyzed using weighted average regression software (C2 ver. 1.7.6; Juggins 2014). The resulting transfer functions were subsequently applied with weighted averaging calibration to the fossil diatom assemblages. Initial phosphorus estimates from weighted averaging regression were corrected using classical deshrinking. Bootstrapped error estimates are based on initial log transformed data with the phosphorus log error being 0.38.

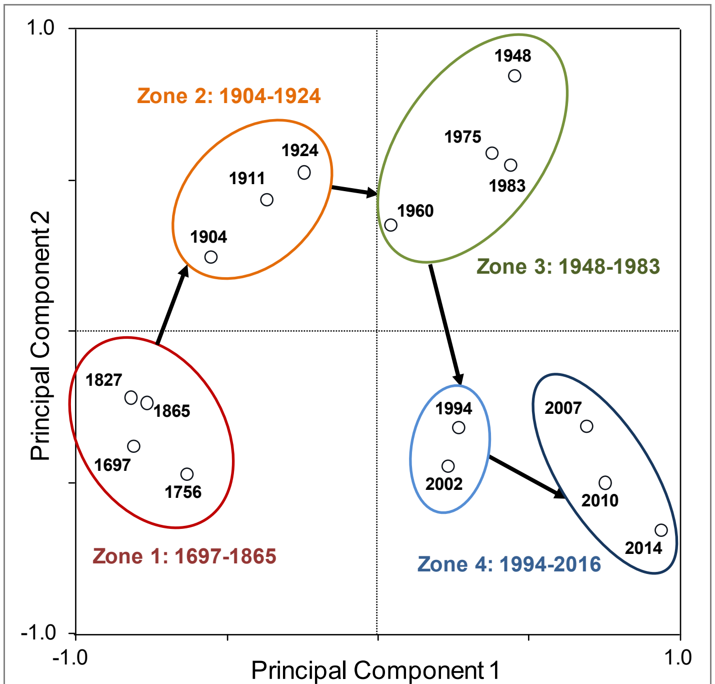


Figure 30. Principal components analysis of diatom communities within the Kentuck Lake sediment core. Four stratigraphic zones were identified. Zones 2-4 occurred after Euro-American development within the watershed and indicate the induced changes to the lake.

Prior to the arrival of Euro-Americans, inferred phosphorus concentrations were nearly constant at about 34 µg/L (Figure 31). With early logging, phosphorus concentrations increased to over 40 µg/L but then appeared to decline after the mid-1940s. Phosphorus appeared to decline further after 2000 to values below estimated historical levels. It is very unlikely that this is true. Recent papers have pointed out that using the diatom community to estimate historical phosphorus concentrations can be problematic. Quantitative phosphorus reconstructions using diatoms have two fundamental assumptions (Juggins and Birks 2012). The first is that phosphorus is linearly related to an ecologically important determinant of species distribution and abundance. In most ecological systems, species distribution is a complex function of multiple, often inter-correlated environmental factors. The second assumption requires that the effect of these secondary environmental variables is either negligible or that their joint distribution with phosphorus is constant in space and time.

Juggins et al. (2013) reported that most benthic taxa do not show a significant response to phosphorus. In Kentuck Lake, benthic diatoms are an important part of the diatom community and they became a more important component after the mid-1940s. It is likely that with increased nutrient levels in the 1940s indicated by the geochemical profiles, that cyanobacteria became more common. These phytoplankton would out compete diatoms for nutrients and light during much of the summer meaning that planktonic diatoms would become less common. As phosphorus levels increase further, especially after 2000, planktonic diatoms would only be present during the spring and early summer. In recent years, measured phosphorus concentrations during this time period is about 20-25 ug/L, which is the concentration inferred by the diatom community. With the increased phosphorus concentrations, benthic diatoms become more common as they would proliferate in the shallow waters with increased phosphorus levels. It is most likely that the inferred phosphorus concentrations in the sediment core prior to 1940 are reasonably accurately estimating summer phosphorus levels, but after 1940 the inferred phosphorus concentration more closely estimates early growing season phosphorus levels. This is most true since 2000 when internal phosphorus loading accelerated.

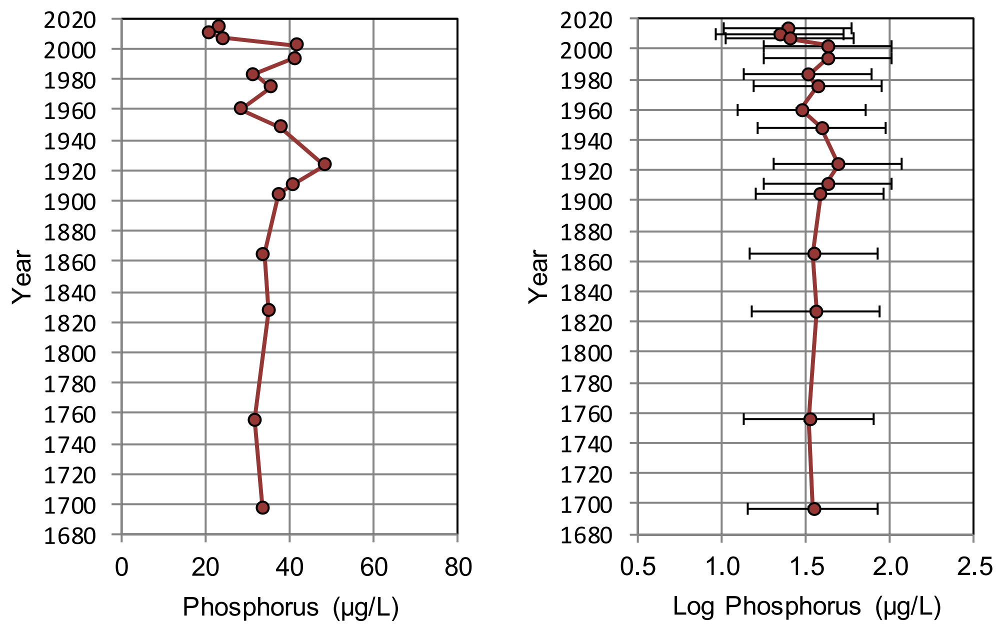


Figure 31. Diatom inferred total phosphorus reconstructions 1697-2015. Phosphorus is reconstructed as log total phosphorus; error bars represent the bootstrapped root mean square error of prediction for the Wisconsin shallow lakes diatom calibration model. After 1940 the model does not accurately predict summer phosphorus because of blue-green algal proliferation. Instead, the model more accurately estimates spring and early summer concentrations. This means the estimates don't reflect increased summer phosphorus from internal loading, especially after 2000.

Summary of Paleoecological Results

Both the geochemical element and diatom community analysis of the full sediment core collected from Kentuck Lake indicate that the lake has seen significant changes in water quality and the lake has become more productive since the settlement of Euro-Americans within its watershed. The geochemical element analysis was more informative of these changes than the diatom community for the reasons discussed in the previous section. However, the combination of these analyses indicate that increased soil erosion initially induced by logging in the early 20th century followed by the subsequent construction of residential structures along the lakeshore resulted in a slow but steady increase in the external nutrient input to the lake.

The increase in external nutrient input and corresponding increase in biological production reached a level in the late-1970s where anoxic conditions began to develop in bottom waters. With the loss of oxygen in bottom waters, sediment-bound nutrients which had been largely unavailable for biological production were released. The availability of sediment-bound nutrients likely increased biological production further, consuming more oxygen and causing the release of even more sediment-bound nutrients. Around 2000, this positive feedback cycle reached a threshold where the internal release of nutrients increased significantly. At present, the internal release of phosphorus is currently the largest source to the lake.

While other lakes in northern Wisconsin have experienced similar increases in nutrient inputs from their watersheds as the result of logging and the construction of residential structures, few have exhibited similar responses in their water quality like those in Kentuck Lake. Logging within the relatively steep topography of Kentuck Lake's watershed likely resulted in a higher soil erosion rate, evidenced by the visual color change within the sediment core that is often not evident in cores collected from northern Wisconsin lakes. In addition, Kentuck Lake's morphometry which straddles the boundary between a deep and shallow lake allows for the mobilization of sediment-released nutrients to the surface where they become available to phytoplankton. The combination of Kentuck Lake's steep topography within its watershed and its polymictic morphometry likely made this lake more sensitive to increasing nutrient inputs from the watershed. The sediment core indicates that soil erosion from the watershed has declined since 2000, likely a reflection of reduced construction of cottages and homes along the shoreline. However, the core indicates that soil amendments for lawn maintenance, like lime and fertilizer, are still contributing nutrients to the lake.

3.6 Aquatic Plant Survey Results

Non-Native Aquatic Plant Survey Results

Curly-leaf pondweed

The non-native, invasive plant curly-leaf pondweed (*Potamogeton crispus*; CLP) was first documented in Kentuck Lake in 1999, but plants were not observed in the lake again by KLPRD members until the summer of 2012. While subsequent surveys since 2012 have found that for unknown reasons the CLP population in Kentuck Lake remains very small, the KLPRD understands that this plant has the capacity to expand rapidly and has elected to continue to monitor the lake's population on an annual basis. In 2015 and 2016, Onterra ecologists visited Kentuck Lake to complete the Early-Season AIS Survey in early summer to locate and map occurrences of CLP when these plants are at or near their peak growth.

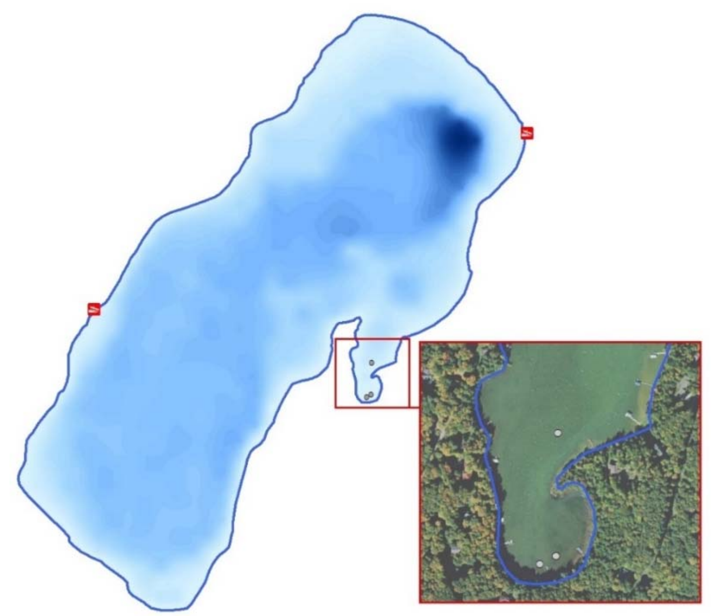


Figure 32. Kentuck Lake June 2016 curly-leaf pondweed locations. Gray circles represent single or few plants.

During both the 2015 survey, a few fragments of CLP were observed floating of the surface were observed but no rooted plants were able to be located. In 2016, a few single plant occurrences of CLP were located in shallow water within the small bay on the eastern side of the lake (Figure 32). The few occurrences of CLP observed indicate that while CLP is still present in Kentuck Lake, its population remains at a level which is not causing adverse ecological or recreational impacts. However, as is discussed in the Updated Water Quality and Aquatic Plant Management Section (Section 5.0), it will be important that CLP population continued to be monitored in the event that it does begin to expand.

Eurasian watermilfoil

Photo 5. Colony of flowering Eurasian watermilfoil in Kentuck Lake. Photo credit Onterra, June 2015.

The non-native, invasive plant Eurasian watermilfoil (*Myriophyllum spicatum*; EWM) was discovered recently in Kentuck Lake in 2011 (Photo 5). Onterra was contracted by the KLPRD in 2012 to conduct a Late-Summer EWM Peak-Biomass Survey to map areas of EWM and to develop a management strategy. During the 2012 survey, approximately 13.4 acres of colonized EWM (polygons) were located. Following discussions between the KLPRD, Onterra, and the WDNR over the winter of 2012/2013, an 18.6-acre herbicide spot treatment was conducted in the spring of 2013. Post-treatment surveys conducted in the summer of 2013 and 2014 found that the spring 2013 herbicide treatment was mostly successful, with EWM density being reduced within the two of the three application areas one year following the treatment. In 2014, 3.7 acres of colonized EWM mainly delineated as *highly scattered* was located, and given this low level of EWM no herbicide control strategy was proposed for 2015.

Eurasian watermilfoil is typically mapped in mid- to late-summer as this is when this plant is at or near its peak growth. The EWM population in Kentuck Lake was mapped by Onterra ecologists in late-summer of 2015. In 2016, an attempt was made to map the EWM in late-summer, but the water clarity was very low and it was difficult to see plants from the surface and the survey was aborted. However, EWM was mapped during the 2016 Early-Season AIS Survey and the results from that survey are discussed within this section. Figure 33 displays the annual acreage of mapped EWM polygons since Onterra began mapping in 2012. Please note that the acreages displayed in this figure represent the acreage of mapped EWM polygons, not EWM mapped with point-based techniques (*single or few plants, clumps of plants,* or *small plant colonies*). Taken out of context, this figure can be misleading as it relates to the EWM population changes. For instance, large increases in colonized acreage between 2014 and 2015 may seem like a drastic change, but they actually represent a collection of point-based EWM occurrences that increased in density to the point they require delineation with polygons.

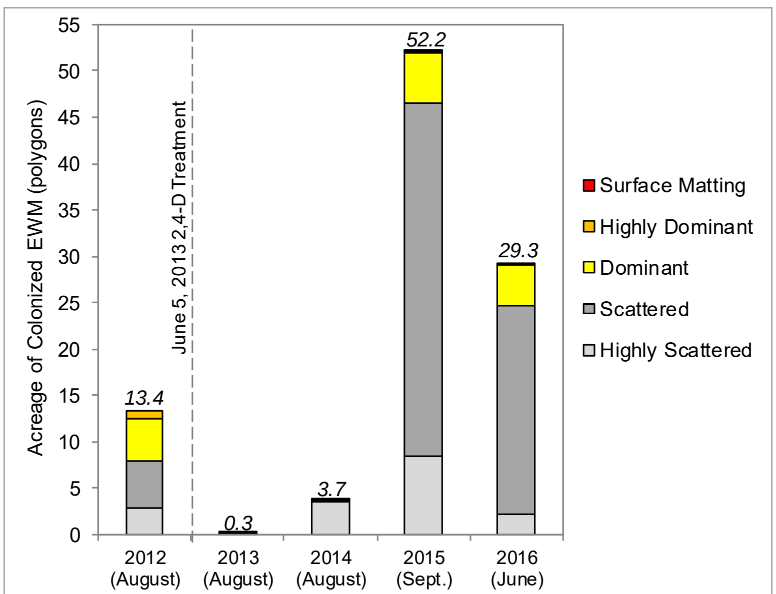


Figure 33. Kentuck Lake acreage of colonized Eurasian watermilfoil from 2012-2016. Please note the acreage in 2016 was determined from a June survey, which may have underestimated the full extent of the population.

The September 2015 Late-Summer EWM Peak-Biomass Survey found that the EWM population in Kentuck Lake has increased in area and density from previous surveys. Acreage of colonized EWM increased from approximately 3.7 acres in 2014 to 52.2 acres in 2015 (Figure 33 and Figure 34). However, 88% of the colonized EWM located in 2015 (46.4 acres) was comprised of low-density EWM, classified as either *scattered* or *highly scattered*. The increase in EWM acreage in one year's time Kentuck Lake is not unprecedented, and has been observed in a number of northern lakes that contain EWM. Long-term aquatic plant data collected by WDNR and Onterra ecologists indicate that EWM not only has the capacity to rapidly increase in occurrence, but that populations can also rapidly decline despite no management action occurring.

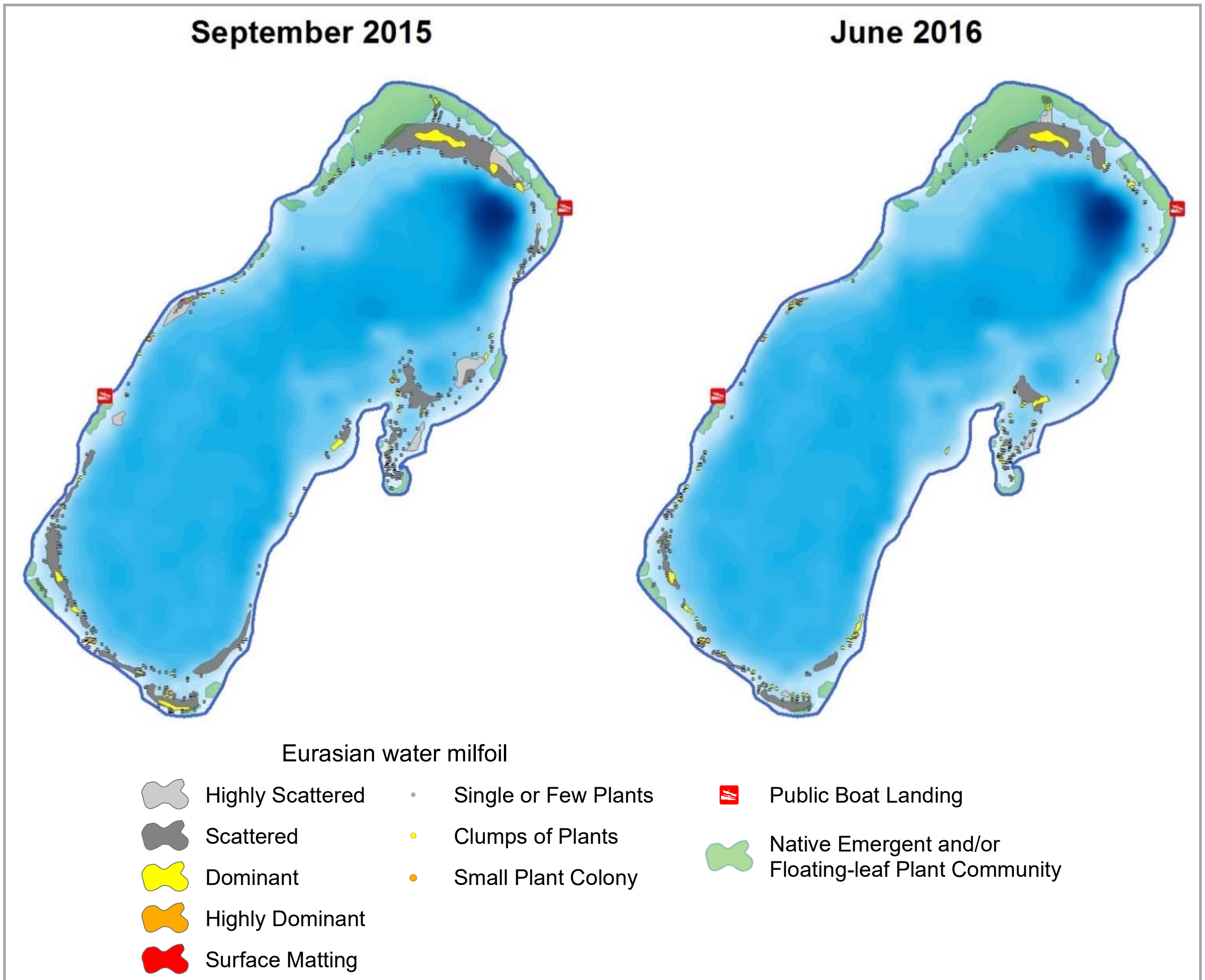


Figure 34. Kentuck Lake September 2015 and June 2016 Eurasian watermilfoil locations.

As mentioned earlier, the 2016 late-summer EWM survey was aborted after water clarity conditions were too poor to complete the survey. However, EWM was mapped during the June 2016 Early-Season AIS Survey. This survey indicated the acreage of colonized EWM had declined by approximately 23 acres. While this survey likely underestimated the full extent of the EWM population as it was conducted in June, it is likely the EWM population was slightly smaller in 2016 when compared to 2015. These natural fluctuations in EWM populations have been observed in lakes in the absence of any management actions, and this is discussed further in the next section. Future monitoring and potential control strategies for Kentuck Lake's EWM population are discussed within the Updated Water Quality and

Aquatic Plant Management Section (Section 5.0). The following section provides updated information on the current status of EWM research and control in Wisconsin's lakes.

As discussed in the Project Methods Section (Section 2.0), it was recommended that a whole-lake point-intercept survey be completed in 2017 instead of the originally proposed EWM peak-biomass survey. While the majority of the data collected during this survey is discussed later in this section, this survey revealed that the littoral occurrence of EWM increased to 4.8% in 2017 compared to 0.2% in 2011 and 0.0% in 2007 (Figure 35). While EWM has increased within the lake over this 10-year period, as is discussed further in the following subsection, its current littoral occurrence of 4.8% is relatively low indicating that the EWM population is likely not having detectable impacts to Kentuck Lake's ecology and that lake-wide control strategies are not warranted on Kentuck Lake at this time.

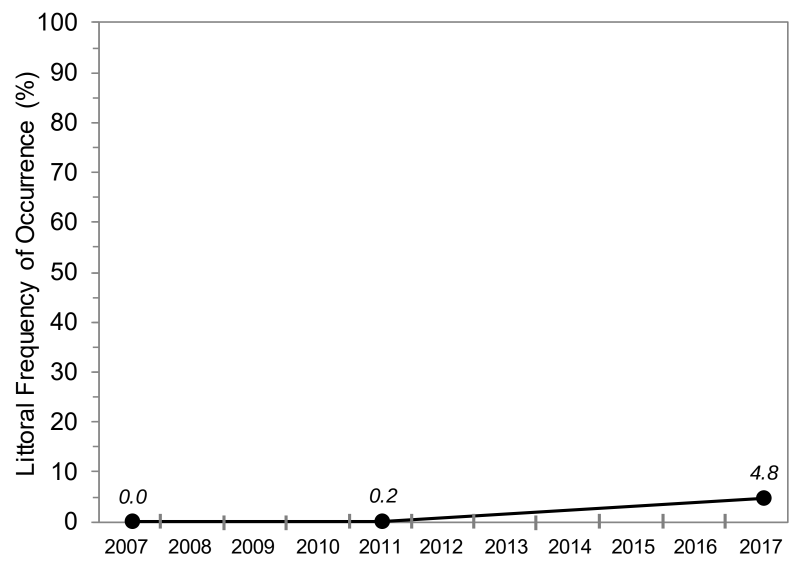


Figure 35. Littoral frequency of occurrence of Eurasian watermilfoil in Kentuck Lake from 2007, 2011, and 2017. Created using data from 2007 Northern Environmental (N = 471), 2011 WDNR (N = 518), and 2017 Onterra (N = 439) point-intercept surveys.

WDNR Long-Term Eurasian watermilfoil Monitoring Research Project

Starting in 2005, WDNR Science Services began conducting annual point-intercept aquatic plant surveys on a set of lakes to understand how EWM populations vary over time. This was in response to commonly held beliefs of the time that once EWM becomes established in a lake, its population would continue to increase over time. As outlined in *The Science Behind the "So-Called" Super Weed* (Nault 2016), EWM population dynamics on lakes are not that simplistic.

Like other aquatic plants, EWM populations are dynamic and annual changes in EWM frequency of occurrence have been documented in many lakes, including those that are not being actively managed for EWM control (no herbicide treatment or hand-harvesting program). The data are most intuitive for unmanaged lakes within the Northern Lakes and Forests Ecoregion (Figure 36). Some lakes, such as Hancock Lake, maintained low EWM populations over the study averaging a littoral frequency of occurrence of 2.3% between 2008 and 2015. At these low levels, there are likely no detectable ecological impacts to the lake and there are no reductions in ecosystem services to lake users. The EWM population of Hancock Lake has increased in recent years to 5.2% in 2015 and over 10% in 2016 (preliminary data not shown in Figure 36).

The littoral frequency of occurrence of EWM populations in other lakes, such as Bear Paw Lake and Little Bearskin Lake, trended to almost 25% only to decline to approximately 5% by the end of the study period (Figure 36). There are many factors that could contribute to the natural decline of the EWM population of these lakes, including climactic conditions and water quality parameters. Little Bearskin Lake is known to contain a robust population of milfoil weevils, and this native insect may be having an impact on the EWM population within the lake. Boot Lake is a eutrophic system with low water clarity (approx. 3-ft Secchi disk depth) due to naturally high phosphorus concentrations. It is hypothesized that water clarity conditions in some years may favor EWM growth whereas keep the population suppressed in other years. Extreme changes in EWM populations like those observed on Weber Lake have also been documented. The EWM population in 2010-2011 had a littoral frequency of occurrence of 20% before spiking above 50% in 2012. Then the population declined back to approximately 15% in 2014 and 2015.

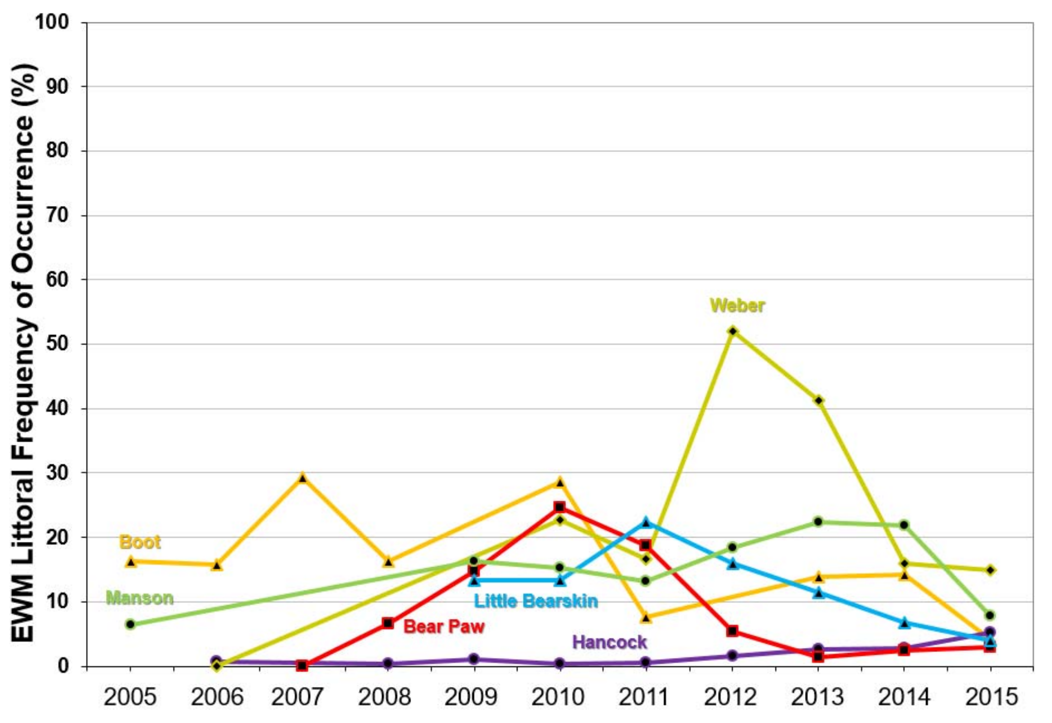


Figure 36. Littoral frequency of occurrence of EWM in the Northern Lakes and Forests Ecoregion without management. Data provided by and used with permission from the WDNR Bureau of Science Services.

The results of the study clearly indicate that EWM populations in unmanaged lakes can fluctuate greatly between years. Following initial infestation, EWM expansion was rapid on some lakes, but overall was variable and unpredictable (Nault 2016). On some lakes, the EWM populations reached a relatively stable equilibrium whereas other lakes had more moderate year-to-year variation. Some lake managers interpret these data to suggest that in some circumstances, it is not appropriate to manage the EWM population as in some years the population may become less. However, even a lowered EWM population of approximately 10% exceeds the comfort level of many riparians because it is potentially approaching a level than can be impactful to the function of the lake as well as not allowing the lake to be enjoyed by riparians as it had been historically. As discussed in the previous sub-section, the 2017 point-intercept survey conducted on Kentuck Lake found that the littoral occurrence of EWM was relatively low at 4.8%.

Some lake groups choose to manage the EWM population to keep it at an artificially lowered level. Following detection of an EWM population within a lake, it is common for a lake group to initiate management activities and not wait to see if the EWM population will become a problem in their lake. In other instances, the management strategy is simply to maintain a lower level population of EWM for the purposes of allowing the ecosystem to function as it had before the exotic was introduced to the lake. And yet other lakes are managed simply to alleviate the lost ecosystem services, most notably to manage for multiple human uses. There are a number of different management techniques used for controlling EWM with the most commonly implemented being hand-harvesting and herbicide control.

Following the discovery of EWM in Kentuck Lake, it was determined that localized spot treatments were the most appropriate initial control strategy. In many lakes, this method is able to slow the spread and population of EWM throughout the lake and may even be able to cause a decline in the EWM population where the activities were conducted. But in other lakes, the EWM population progression is too great for the method to provide effective lake-wide control. Continuing localized spot treatments on these lakes may be able to provide localized EWM reductions where the control strategy is applied and reduce that specific colony from contributing to the overall population increase to the lake. These efforts may also reduce recreational impediments that are caused by dense EWM colonies.

Background on Herbicide Application Strategy

Herbicides that target submersed plant species are directly applied to the water, either as a liquid or an encapsulated granular formulation. Factors such as water depth, water flow, treatment area size, and plant density work to dilute herbicide concentration within aquatic systems. Understanding concentration-exposure times (often referred to as CETs) is an important consideration for the use of aquatic herbicides. Successful control of the target plant is achieved when it is exposed to a lethal concentration of the herbicide for a specific duration of time.

A Cooperative Research and Development Agreement between the Wisconsin Department of Natural Resources and U.S. Army Corps of Engineers Research and Development Center in conjunction with significant participation by private lake management consultants have coupled quantitative aquatic plant monitoring with in-lake herbicide concentration data to evaluate efficacy, selectivity, and longevity of chemical control strategies implemented on a subset of Wisconsin waterbodies. Based on the preliminary findings from this research, lake managers have adopted two main treatment strategies: 1) spot treatments, and 2) large-scale (whole-lake) treatments.

Spot treatments (like those conducted in 2013) are a type of control strategy where the herbicide is applied to a specific area (treatment site) such that when it dilutes from that area, its concentrations are insufficient to cause significant effects outside of that area. Herbicide application rates for spot treatment are formulated volumetrically, typically targeting EWM with 2,4-D at 3.0-4.0 ppm acid equivalent (ae). This means that sufficient 2,4-D is applied within the *Application Area* such that if it mixed evenly with the *Treatment Volume*, it would equal 3.0-4.0 ppm ae. This standard method for determining spot treatment use rates is not without flaw, as no physical barrier keeps the herbicide within the *Treatment*

Volume and herbicide dissipates horizontally out of the area before reaching equilibrium (Figure 37). While lake managers may propose that a particular volumetric dose be used, such as 3.0-4.0 ppm ae, it is understood that actually achieving 3.0-4.0 ppm ae within the water column is not likely due to dissipation and other factors.

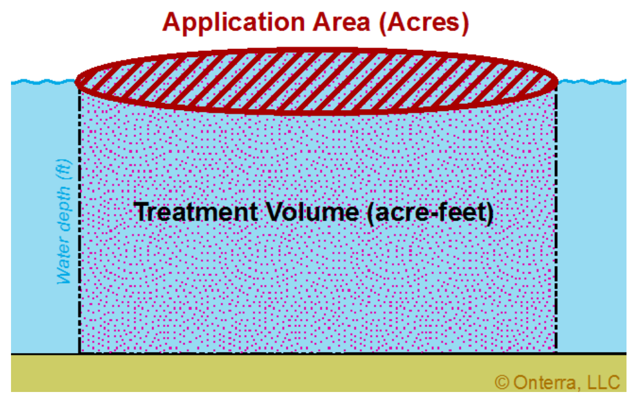


Figure 37. Herbicide spot treatment diagram.

Ongoing research clearly indicates that the herbicide concentrations and exposure times of large (> 5 acres each) treatment sites are higher and longer than for small sites (Nault 2015). Research also indicates that higher herbicide concentrations and exposure times are observed in protected parts of a lake compared with open and exposed parts of the lake. Areas targeted containing higher water exchange (i.e. flow) are often not able to meet herbicide concentration-exposure time (CET) requirements for control.

Wisconsin Department of Natural Resources administrative code defines large-scale treatments as those that exceed 10% of the littoral zone (NR 107.04[3]). From an ecological perspective, large-scale (whole-lake) treatments are those where the herbicide is applied to specific sites, but when the herbicide reaches equilibrium within the entire volume of water (of the lake, lake basin, or within the epilimnion of the lake or lake basin) it is at a concentration that is sufficient to cause mortality to the target plant within that entire treated volume. In regards to the WDNR's large-scale treatment definition of a treatment exceeding 10% of the littoral zone, there is ecological basis in this standard. In general, if 10% of a lake was targeted with 2,4-D at 4.0 ppm ae, the whole-lake equilibrium concentration would be approximately 10% of that rate or 0.4 ppm ae. The target 2,4-D concentration for large-scale EWM treatments is typically between 0.250 and 0.400 ppm ae understanding that the exposure time would be dictated by herbicide degradation and be maintained for 7-14 days or longer. Therefore, spot treatments that approach 10% of a lake's area will become large-scale treatments.

Large-scale treatments have become more widely utilized by many lake managers (and public sector regulatory partners) as they impact the entire EWM population at once. This minimizes the repeated need for exposing the lake to herbicides as is required when engaged in an annual spot treatment program. Properly implemented large-scale herbicide treatments can be highly effective, with minimal EWM, often 0.0%, being detected for a year or two following the treatment (Figure 38). Some large-scale treatments have been effective at reducing EWM populations for five to six years following the application.

Predicting success (EWM control) and native plant impacts from whole-lake treatments is also better understood than for spot treatments. Some native plants are quite resilient to this herbicide use pattern, either because they are inherently tolerant of the herbicide or they emerge later in the year than when the herbicide was active in the lake. Other species, particularly dicots, some narrow-leaved pondweeds (*Potamogeton* spp.), and naiad species (*Najas* spp.), can be impacted and take a number of years to recover. Often during the year of treatment, overall native plant biomass can be lessened but typically (not always) rebounds the following year.

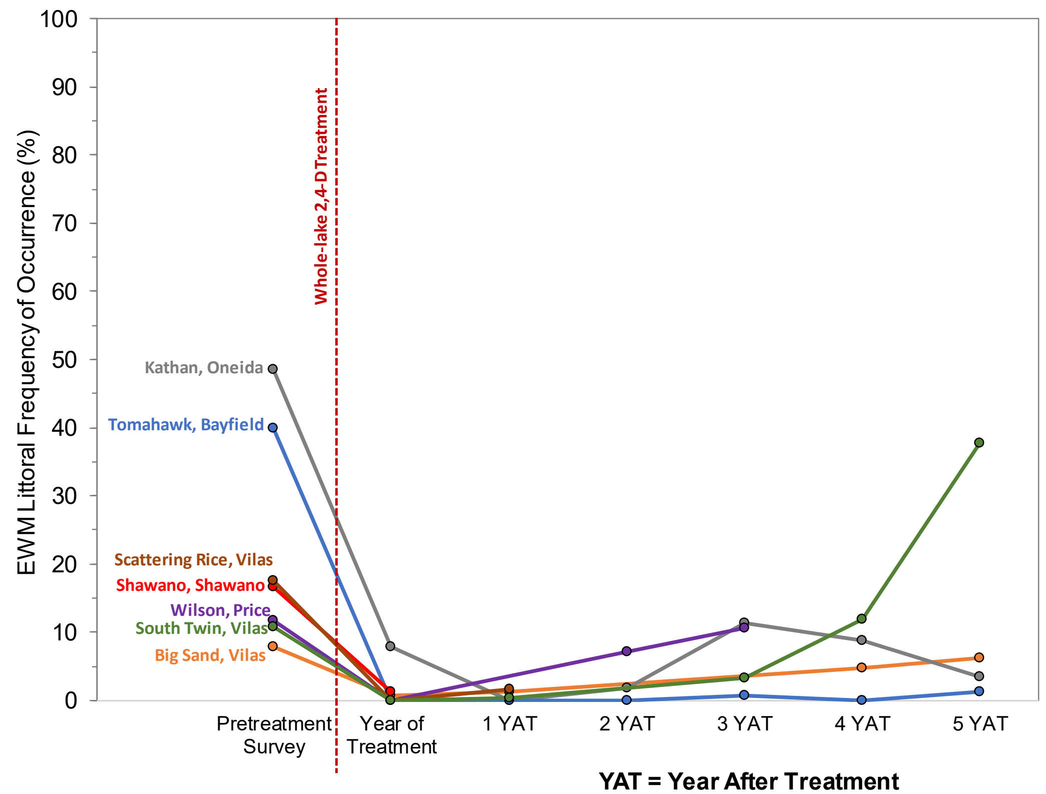


Figure 38. Littoral frequency of occurrence of EWM in lakes managed with large-scale 2,4-D treatments.

It is also important to note that US EPA registration of aquatic herbicides typically requires organismal toxicity studies to be conducted using concentrations and exposure times consistent with spot-treatment use patterns (high concentrations, short exposure times). Therefore, only limited organismal toxicity data is available for concentrations and exposure times consistent with whole-lake treatment use patterns (low concentrations, long exposure times).

Because of their durability as a laboratory species, fathead minnows are often the subject of organismal toxicity studies. The LC50 (lethal concentration when half die) for fathead minnow exposure to 2,4-D (amine salt) has been determined to be 263 ppm ae sustained for 96 hours, a thousand times higher than fish would be exposed to in a large-scale treatment (target of approximately 0.3 ppm ae). With the assistance of a WDNR AIS-Research Grant, DeQuattro and Karasov (2015) investigated the impacts on fathead minnow of 2,4-D concentrations more relevant to what would be observed in large-scale treatments. The focus of their investigations was on reproductive toxicity and/or possible endocrine disruption potential from the herbicide. The study revealed morphological changes in reproducing male fathead minnows, such that they had lower tubercle scores (analogous to smaller antlers on a male white-tail deer) with some 2,4-D products/use-rates and not with others. This may suggest that the “inert” carrier may be the cause, not the 2,4-D itself.

At a static exposure of 0.05 ppm ae for 58 days (adult fish exposed for 28 days then larval fish from eggs they laid were continued to be exposed for 30 more days post hatching) uncovered a reduction in larval fathead survival from 97% to 83% at the lowest dose of the 2,4-D (amine salt) fromulation that was tested (no reduction at higher doses).

As discussed above, large-scale treatments can have potential secondary impacts to the lake in addition to the financial costs to the lake group. Therefore, large-scale EWM treatments are typically postponed until the population exceeds a pre-defined threshold in an attempt to balance these factors. While EWM is widespread throughout littoral areas of Kentuck Lake, its relatively low littoral occurrence of 4.8% in 2017 indicates a whole-lake treatment strategy is not warranted at this time. A management strategy for EWM in Kentuck Lake is discussed within the Updated Water Quality and Aquatic Plant Management Strategy Section (Section 5.0).

2017 Whole-Lake Point-Intercept Survey

The 2017 point-intercept survey conducted on Kentuck Lake was completed to gather quantitative data on the lake's aquatic plant community to serve as reference or baseline data prior to the potential initiation of a management strategy to improve the lake's water quality. While the point-intercept survey provided a quantitative assessment of the level of EWM within the lake, this survey also assessed the lake's native aquatic plant community. In addition to the data collected in 2017, a point-intercept survey was completed in 2007 by Northern Environmental and in 2011 by the WDNR. Because these surveys were completed using the same sampling locations and methodology, the data can be statistically compared to see how the plant community has changed over this 10-year period.

The data collected from these three surveys show that Kentuck Lake's aquatic plant community has undergone significant changes over this period. The overall occurrence of aquatic plants within Kentuck Lake's littoral zone has declined from 85% in 2007 to 46% in 2017, a decrease in occurrence of 46% (Figure 39). Most of this decline can be attributed to the reduction of aquatic plants in deeper areas of the littoral zone, mainly between 11 and 24 feet of water (Figure 40). In 2007, 70% of the sampling locations within 11-24 feet of water contained aquatic plants compared to 44% in 2011 and 24% in 2017, representing a decrease in occurrence of 66% between 2007 and 2017. The littoral occurrence of vegetation in shallower water from 1-10 feet remained unchanged at 87%, 86%, and 89% in 2007, 2011, 2017, respectively.

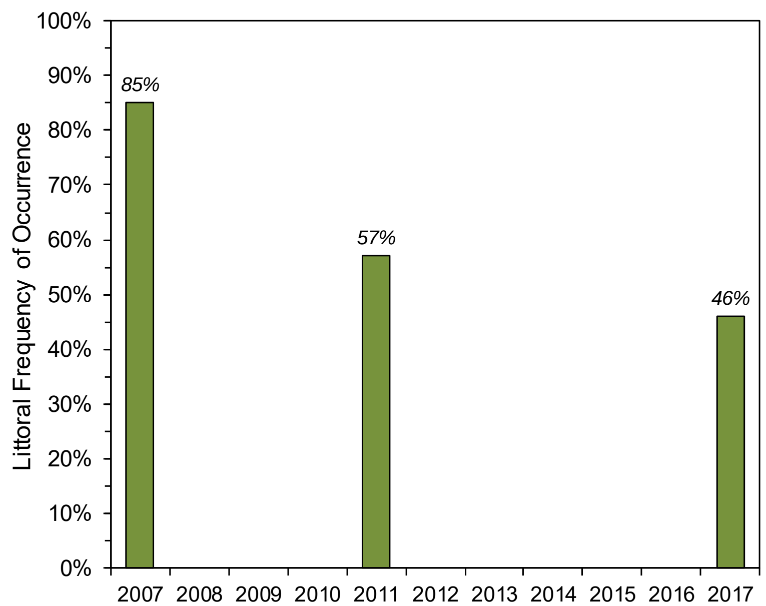


Figure 39. Littoral frequency of occurrence of aquatic vegetation in Kentuck Lake from 2007, 2011, and 2017. Created using data from 2007 Northern Environmental (N = 471), 2011 WDNR (N = 518), and 2017 Onterra (N = 439) point-intercept surveys.

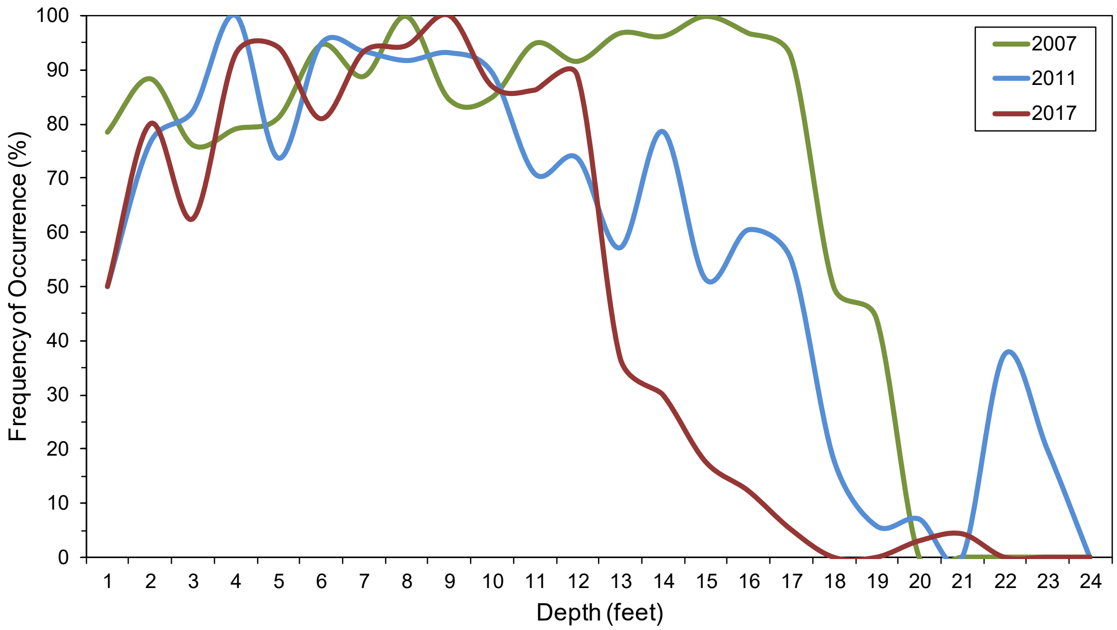


Figure 40. Occurrence of aquatic plants across littoral depths in Kentuck Lake from 2007, 2011, and 2017 showing reduction in the occurrence of aquatic plants in deeper waters. Created using data from 2007 Northern Environmental (N = 471), 2011 WDNR (N = 518), and 2017 Onterra (N = 439) point-intercept surveys.

In terms of area, Kentuck Lake's littoral zone (based on maximum depth of plants) encompasses approximately 929 acres of the lake. The area of water 11-24 feet in depth encompasses approximately 615 acres or 66% of the littoral zone, meaning that two-thirds of Kentuck Lake's littoral zone has seen a 66% reduction in the occurrence of vegetation between 2007 and 2017. This represents a significant change to the lake's aquatic plant community over a relatively short period of time.

The reduction in the occurrence of aquatic plants in Kentuck Lake is most certainly the result of light limitation from the rapid decline in water clarity due to increased phytoplankton abundance which began in 2011. As discussed earlier in the report, mean summer Secchi disk depth declined from an average of 8.3 feet from 1986-2010 to an average of 5.0 feet from 2011-2016. While average summer Secchi disk depth increased in 2017 to an average of 8.3 feet, it will likely take a number of consecutive years with consistent higher clarity to allow plants to reestablish deeper areas of the littoral zone.

Figure 41 displays the littoral occurrences of the eight-most frequently encountered native aquatic plant species along with EWM from the 2007, 2011, and 2017 point-intercept surveys. The occurrences of large-leaf pondweed, white-stem pondweed, and a suspected hybrid between these two species were combined for this analysis. Five of these seven native species exhibited statistically valid reductions in their littoral occurrence between 2007 and 2017 (Chi-Square $\alpha = 0.05$). Small pondweed declined in occurrence by 44%, common waterweed declined by 90%, coontail declined by 93%, northern watermilfoil declined by 94%, and the large-leaf/white-stem hybrid declined by 90%. Slender naiad exhibited a statistically valid increase in occurrence of 73% while the occurrence of wild celery was not statistically different. The large reductions in the occurrence of these dominant plants in Kentuck Lake between 2007 and 2017 is likely due to the reduction in light availability over this time period.

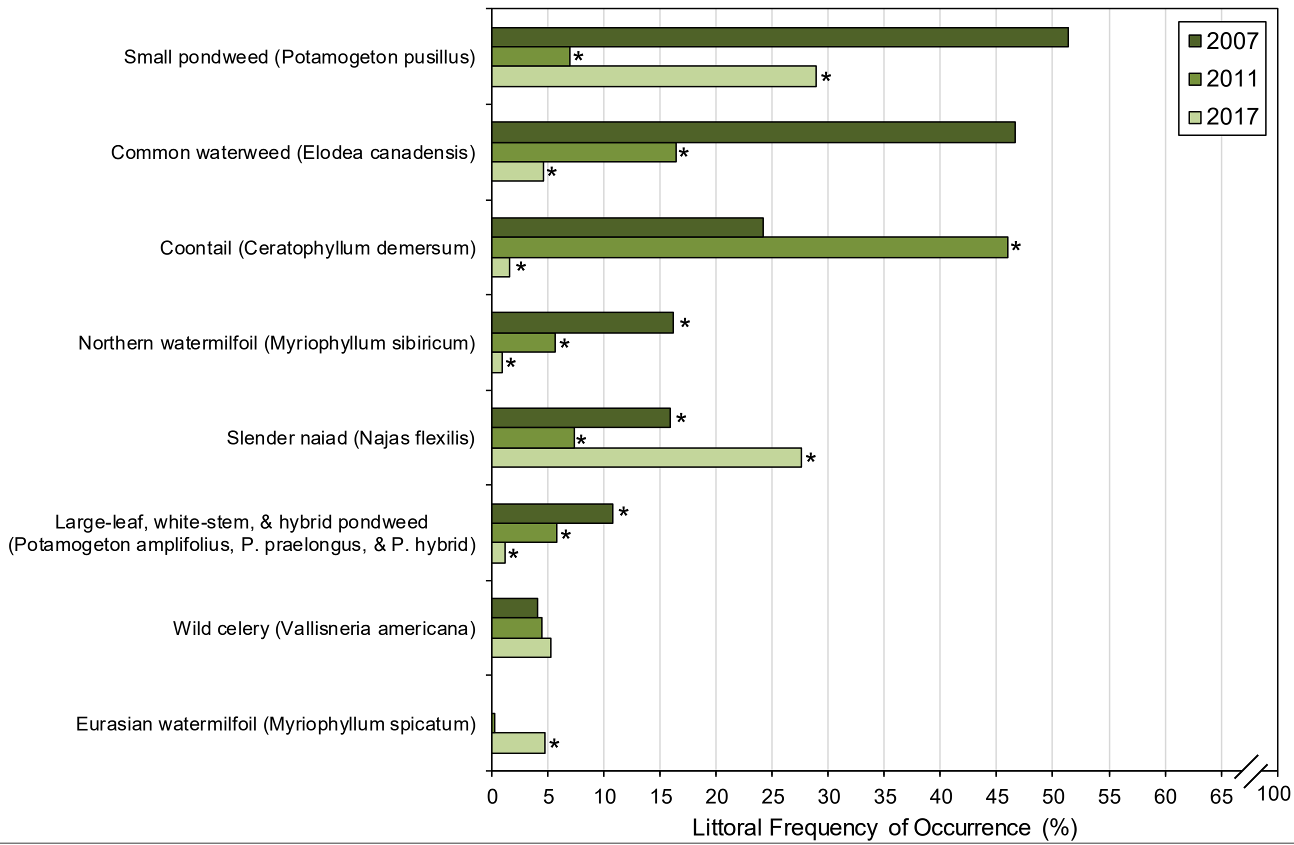


Figure 41. Littoral frequency of occurrence of seven-most frequently encountered native aquatic plant species and EWM from 2007, 2011, and 2017 in Kentuck Lake.
* = Statistically valid change in occurrence from previous survey (Chi-Square α = 0.05). Created using data from 2007 Northern Environmental (N = 471), 2011 WDNR (N = 518), and 2017 Onterra (N = 439) point-intercept surveys.

Floristic Quality Assessment (FQA) is a tool used for tracking the integrity of a lake's aquatic plant community over time and can also be used to compare the aquatic plant communities of multiple lakes to one another. The floristic quality of a lake is calculated using its native plant species richness and average native species conservatism. Native species richness is simply the number of native species that were recorded on the rake during the point-intercept surveys. Average species conservatism utilizes the coefficient of conservatism values for each of those species in its calculation. A species coefficient of conservatism value indicates that species likelihood of being found in an undisturbed (pristine) system. The values range from 1 to 10. Species that can tolerate living in disturbed systems have lower coefficients, while species which have specific habitat requirements and are more sensitive to changes in their environment have higher conservatism values.

For example, cattails, which can be found growing in disturbed conditions such as roadside ditches, have a conservatism value of 1. In contrast, algal-leaf pondweed, which is only found in acid lakes with good water quality and is sensitive to changes in that environment has a conservatism value of 10. Both the native species richness and the average native plant conservatism value are used to calculate the lake's floristic quality using the equation below:

$$\text{FQI} = \text{Average Coefficient of Conservatism} * \sqrt{\text{Number of Native Species}}$$

The Floristic Quality components for Kentuck Lake are displayed in Figure 42 and indicate that despite the changes observed in water quality and the large reductions in the occurrence of dominant plant species within the lake, Floristic Quality has increased slightly from 2007 to 2017. This is an indication that there have not been significant changes in terms of species composition of Kentuck Lake's aquatic plant community between 2007 to 2017. While the species composition of the lake's plant community has changed little over this time period, the reductions observed in dominant plant species discussed previously indicate degradation of the lake's plant community has occurred.

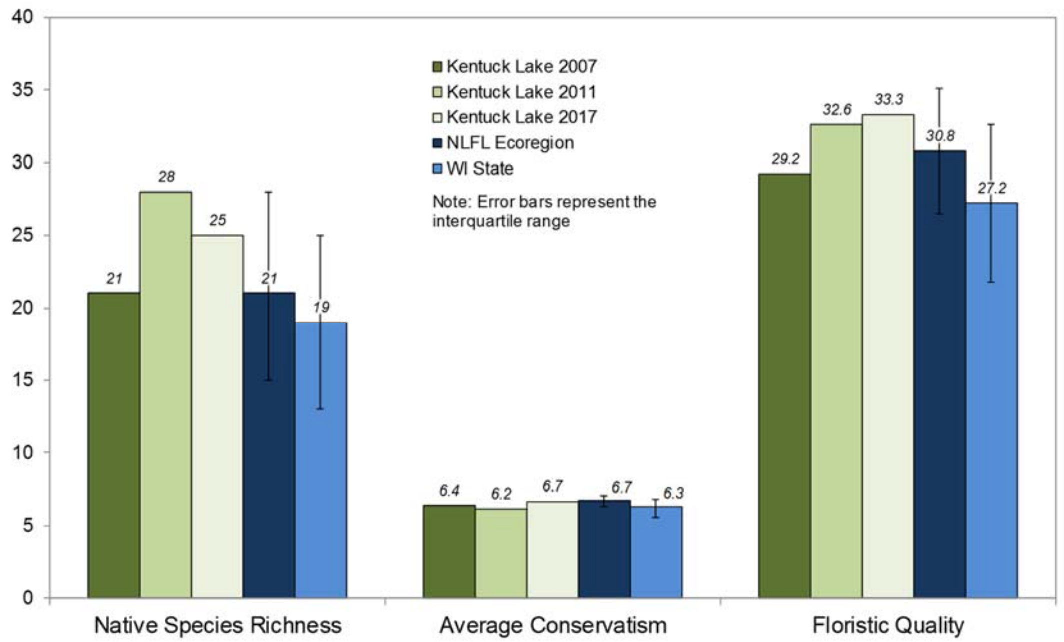


Figure 42. Kentuck Lake Floristic Quality Assessment. Created using data from 2007 Northern Environmental, 2011 WDNR, and 2017 Onterra point-intercept surveys. Analysis follows Nichols (1999).

Lakes with diverse aquatic plant communities have higher resilience to environmental disturbances and greater resistance to invasion by non-native plants. Species diversity differs from species richness in that it also takes into account how evenly the species occur within the community. The diversity of Kentuck Lake's aquatic plant community was calculated using the Simpson's Diversity Index. This analysis shows that the diversity of Kentuck Lake's aquatic plant community has declined slightly with index values of 0.82, 0.81, and 0.79 in 2007, 2011, and 2017, respectively (Figure 43). In other words, if two locations were sampled for aquatic plants in Kentuck Lake in 2017, there would be a 79% probability that the two locations would contain different species. These diversity values fall below the median value for lakes in the Northern Lakes and Forests ecoregion (0.88) and indicate that the aquatic plant species in Kentuck Lake's plant community are unevenly distributed. In 2017, 63% of the plant community was comprised of just two species (small pondweed and slender naiad) while the remaining 37% was comprised of 25 species.

The 2017 point-intercept data indicate there has been a decreasing trend in the occurrence of aquatic vegetation in Kentuck Lake from 2007-2017. The reduction of aquatic vegetation has mainly occurred in deeper areas of Kentuck Lake's littoral zone (>11 feet), and is an indication that light limitation due to intense algal blooms in four of the seven years since 2011 is likely the cause. This represents a loss in valuable structural habitat in deeper waters of the lake. While the species composition of Kentuck Lake's aquatic plant community has changed little over this time period, species diversity is also trending downward with greater than half of the plan community comprised of just two native species. Overall, the 2017 data indicate that Kentuck Lake's native aquatic plant community has degraded since 2007.

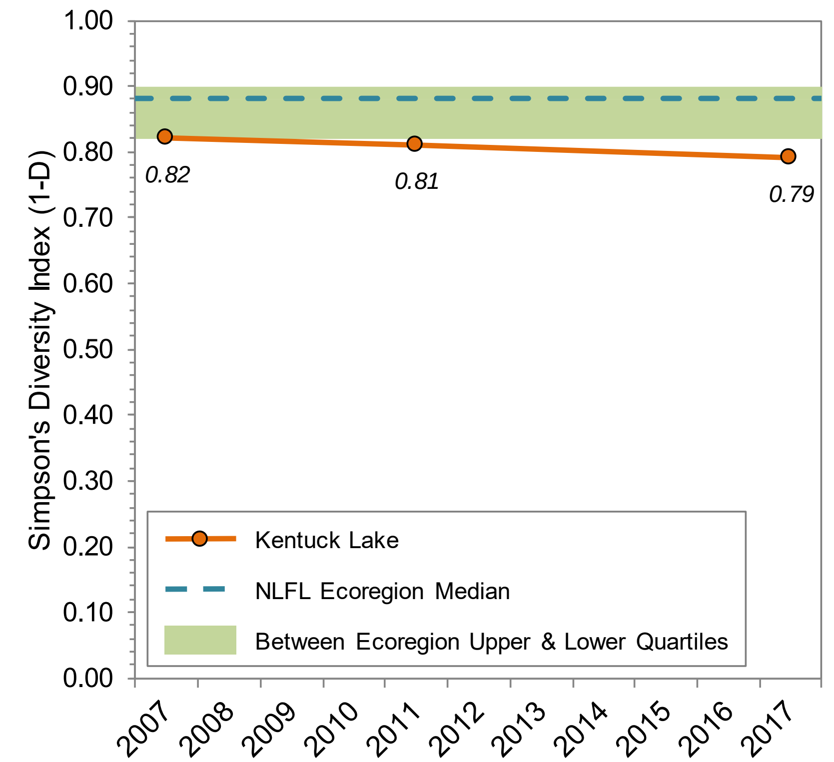


Figure 43. Kentuck Lake 2007, 2011, and 2017 aquatic plant community Simpson's Diversity Index. Created using data from 2007 Northern Environmental, 2011 WDNR, and 2017 Onterra point-intercept surveys. NLFL = Northern Lakes and Forests Lakes ecoregion.

4.0 SUMMARY & CONCLUSIONS

While Kentuck Lake's watershed is primarily comprised of forests, historical disturbances such as logging and ongoing development within the lake's immediate shoreland zone have resulted in conditions which facilitate the periodic development of potentially harmful cyanobacterial blooms. These blooms do not develop every year, but their frequency has increased in recent years. Four years within 2011-2016 saw the development of nuisance cyanobacterial blooms (average summer chlorophyll-*a* at 52 µg/L) following a 17-year period where nuisance blooms did not occur. With the increased frequency of cyanobacteria blooms since 2011, Kentuck Lake is currently on the Environmental Protection Agency list of impaired waterbodies for chlorophyll-*a* concentrations which exceeded the threshold for recreational use.

The analyses completed on a sediment core collected from Kentuck Lake indicate that anthropogenic development within the lake's watershed has led to significant changes in water quality. Soil erosion and external nutrient input from the watershed began to increase in Kentuck Lake in the early twentieth century with the advent of logging and the removal of vegetation. The construction of residential structures along the lake's shoreline was relatively minimal in the first half of the twentieth century. However, by 1970 there was an estimated 40 residential structures along the shores of Kentuck Lake and this increased to 135 by 2000. This rapid increase in shoreland development corresponded with measured increases in soil erosion and external nutrient input within the sediment core.

The increase in external nutrient input and corresponding increase in biological production reached a tipping-point in the late-1970s where anoxic conditions began to develop in bottom waters. With the loss of oxygen in bottom waters, sediment-bound nutrients which had been largely unavailable for biological production were released into the overlying water. Kentuck Lake's long fetch in combination with relatively shallow water allowed these sediment-released nutrients to be mobilized to surface waters through entrainment and/or mixing events. The availability of sediment-bound nutrients likely increased biological production further, consuming more oxygen and causing the release of even more sediment-bound nutrients. Around 2000, this positive feedback cycle reached a threshold where the internal release of nutrients increased significantly. The sediment core indicated that nutrient concentrations and biological production are at present the highest the lake has seen in the past 200 years. Soil erosion has declined since 2000, but the internal loading of phosphorus is currently the largest source of phosphorus to Kentuck Lake, accounting for an estimated 80% of the annual phosphorus load since 2011.

In 2011, mean summer total phosphorus and chlorophyll-*a* concentrations increased to 64 and 83 µg/L, respectively, from mean concentrations of 23 and 9 µg/L from 2000-2010. Since 2011, near-surface total phosphorus concentrations have been positively correlated with water column stability, or the strength of thermal stratification. Strong thermal stratification in four of the six years from 2011-2016 allowed for the prolongment of anoxia and a continued release of phosphorus from bottom sediments. As a result, these four years saw the development of severe, nuisance cyanobacterial blooms. However, in contrast, two of these six years exhibited conditions which resulted in complete mixing events early in the growing season. It is believed the destabilization of the water column early in the season in these two years made the lake more prone to mixing during subsequent wind events, and the duration of anoxia as well as the area of anoxia were reduced. As a result, less phosphorus was released from bottom sediments, summer near-surface total phosphorus and chlorophyll-a concentrations were lower, and nuisance algal blooms were not reported.

Given the analyses of historical and current data, it is believed that artificial water column destabilization using an aeration system deployed in the deeper area of the lake would reduce the duration of anoxia and the amount of phosphorus released from anoxic bottom sediments. It is believed that if the early-season mixing conditions observed in the two years (2012 and 2014) with lower phosphorus and chlorophyll-*a* concentrations can be created artificially each year, chlorophyll-a concentrations could be lowered below the impairment threshold. If the average annual internal loading of phosphorus can be reduced by 90%, the WiLMS modeling predicts that average growing season phosphorus concentrations would be reduced to around 20 μg/L or approximately 53% from the 2011-2016 average.

The artificial destabilization of the water column through aeration has been attempted on lakes which experience internal loading, such as Cedar Lake (Polk and St. Croix counties). However, investigations (James et al. 2015) on Cedar Lake found that aeration intensified the internal loading of phosphorus, and the investigators attributed this to a low iron to phosphorus ratio (Fe:P). The authors indicated that a Fe:P ratio greater than 3.6:1 on a mass basis is needed to have near-complete binding of phosphorus by iron. In Kentuck Lake, the Fe:P ratio is higher at approximately 13:1. With this higher Fe:P ratio, it is believed artificial water column destabilization using an aeration system will have a higher probability of success in Kentuck Lake. The steps the KLPRD will take to implement such a system on Kentuck Lake is detailed within the subsequent Updated Water Quality and Aquatic Plant Management Strategy Section (Section 5.0). Alternative management strategies that were considered, but deemed unlikely to be effective and/or unfeasible in Kentuck Lake are also discussed in the next section.

The aquatic plant surveys completed as part of this project found the lake’s curly-leaf pondweed population remains small and is not having detectable impacts to lake ecology or ecosystem services. Kentuck Lake’s EWM population has become more widespread but the population remains relatively small with a 2017 littoral frequency of occurrence of 5%. However, the data collected in 2017 indicate that the occurrence of aquatic vegetation in Kentuck Lake’s littoral zone has declined by 46% since 2007, with most of this decline occurring within deeper areas of the littoral zone (66% reduction in water ≥11 feet). The reduction in the occurrence of aquatic vegetation in deeper areas of the littoral zone is believed to be due to decreased light availability caused by the intense summer algal blooms observed in 2011, 2013, 2015, and 2016. Five of the seven most frequently-encountered native aquatic plant species have seen large reductions in their occurrence between 2007 and 2017, with most seeing 90% or greater declines in occurrence. These large reductions in occurrence have also resulted in lower species diversity, with greater than 50% of the plant community comprised of just two species. The degradation of Kentuck Lake’s water quality since 2011 has led to the loss of aquatic vegetation and valuable aquatic habitat.

5.0 UPDATED WATER QUALITY & AQUATIC PLANT MANAGEMENT STRATEGY

The water quality and aquatic plant management goal updates provided within the section represent a revision to Management Goal 1 and 2 within the Kentuck Lake Comprehensive Management Plan finalized in 2015. These updated management goals were created through the collaborative efforts of the KLPRD, Onterra ecologists, WDNR, GLIFWC, and the Mole Lake Tribe. It represents the path the KLPRD will follow for improving Kentuck Lake's water quality and maintaining/enhancing the integrity of the lake's native aquatic plant community and the ecosystem services the lake provides. These revised goals represent a living document that will be under constant review and adjusted depending on how the lake's water quality responds to the implementation of management actions, how CLP/EWM populations change with time, the availability of funds, level of volunteer involvement, and the needs of the KLPRD members.

Revised Management Goal 1: Improve Kentuck Lake's water quality, ecological integrity, ecosystem services, and remove the lake's 303(d) impairment listing

Management Action: Investigate feasibility of utilizing aeration system to artificially destratify deeper areas of Kentuck Lake to reduce water column stability and internal phosphorus loading.

Funding Source(s): WDNR Lake Protection Grant

Timeframe: Continuation of current effort

Facilitator: KLPRD Board of Directors

Description: In 2016, Kentuck Lake was listed on the US Environmental Protection Agency's 303(d) list of impaired waters for chlorophyll-*a* concentrations which exceed recreational use thresholds (> 20 µg/L for >5% of days between July 15 – September 15). This listing was determined based on chlorophyll-*a* concentrations measured from 2011-2016. Total phosphorus concentrations assessed over this same time period fell just below the threshold for impairment for recreational use. Neither total phosphorus nor chlorophyll-*a* concentrations exceeded impairment thresholds for fish and aquatic life.

The primary goal of this management action is to lower Kentuck Lake's chlorophyll-*a* concentrations below exceedance thresholds for recreational use by reducing the amount of phosphorus internally loaded from anoxic bottom sediments. Reducing Kentuck Lake's phosphorus and chlorophyll-*a* concentrations will enhance the lake's ecological integrity and improve ecosystem services. As is discussed within the Project Results and Discussion Section (Section 3.0) and Summary and Conclusions Section (Section 4.0), it is believed that artificial destratification through aeration of the water column in deeper areas of the lake is likely the most feasible and realistic option for reducing the internal loading phosphorus in Kentuck Lake. The alternative management strategies of biomanipulation and aluminum sulfate application that have been implemented on other lakes with internal loading were also considered; however, it is not believed these strategies are feasible in Kentuck Lake for the reasons discussed below:

Biomanipulation

Biomanipulation in terms of piscivore stocking and/or zooplanktivore removal to improve water quality has been shown to be successful in a number of shallow lakes; however, the results do not often last more than 10 years and the stocking and/or removal has to be conducted periodically to maintain the improved conditions (Jeppesen et al. 2012). The lower phosphorus and chlorophyll conditions observed from 2000-2010 in Kentuck Lake are believed to be largely the result of food web-driven effects; however, data regarding zooplankton community composition and/or body size during this period are not available. Because of this, the food web effects on water quality remain a hypothesis.

Additionally, WDNR fisheries biologists consider the walleye densities measured from 2000-2010 to be unusually high and unsustainable, and the large fluctuations in the walleye population are not well understood (Steve Gilbert personal comm. 2015). Approximately 34,000 and 35,000 small walleye fingerlings were stocked in Kentuck Lake in 2014 and 2016, respectively, as part of the Wisconsin Walleye Initiative. Updates provided by WDNR fisheries biologists in November 2017 indicate that natural recruitment of walleye was observed in 2017 for the first time since 2006. The stocking of walleye is scheduled to occur again in 2018. The WDNR fisheries biologists are uncertain as to what extent the walleye population will increase and how long higher population densities will last. Given these uncertainties, biomanipulation through piscivore stocking is not believed to be a reliable long-term strategy for improving Kentuck Lake's water quality.

Aluminum Sulfate (Alum) Application

The application of aluminum sulfate (alum: $Al_2(SO_4)_3$) to deeper areas of Kentuck Lake was also considered. The use of alum to inactivate sediment phosphorus involves the application of alum over areas of lake bottom determined to be releasing significant amounts of phosphorus. In Kentuck Lake, this would be conducted over deeper areas that frequently experience anoxia to prevent the release of phosphorus. When applied, the alum forms a floc within the water which sinks to the bottom. This layer of alum on the bottom binds sediment-released phosphorus, even under anoxic conditions. Given Kentuck Lake's lower alkalinity, a buffering agent would also have to be applied to prevent the pH from falling. It is estimated that an alum treatment of approximately 100 acres over areas of 20 feet and deeper in Kentuck Lake would cost nearly $300,000. The longevity of such a treatment in a shallow lake with flocculent sediments like Kentuck Lake is uncertain as the alum could be buried and/or disturbed relatively quickly. Because of the high cost and uncertainty regarding the longevity of an alum treatment, this is not a proposed strategy for Kentuck Lake.

Destratification Through Aeration

While 2011, 2013, 2015, and 2016 had summer average chlorophyll-*a* concentrations of 52 µg/L, average concentrations were 17.5 µg/L in 2012 and 2014. The lack of cyanobacterial bloom formation in these two years is believed to be the result of climatic conditions which resulted in early-season mixing events. This complete mixing and destabilization of the water column early in the growing season prevented strong thermal stratification or high water column stability from developing during the remainder of the growing season (Figure 37). The destabilization of the water column early in the growing season made the lake more prone to mixing with subsequent wind events and water column stability remained lower throughout the remainder of the growing season. As a result, the duration and the area of anoxia were reduced and less phosphorus was released from bottom sediments and phytoplankton production was lower. In contrast, 2011, 2013, 2015, and 2016 saw high water column stability maintained through most of the summer, resulting in the prolongment of anoxia and increased amount of phosphorus released from bottom sediments (Figure 37).

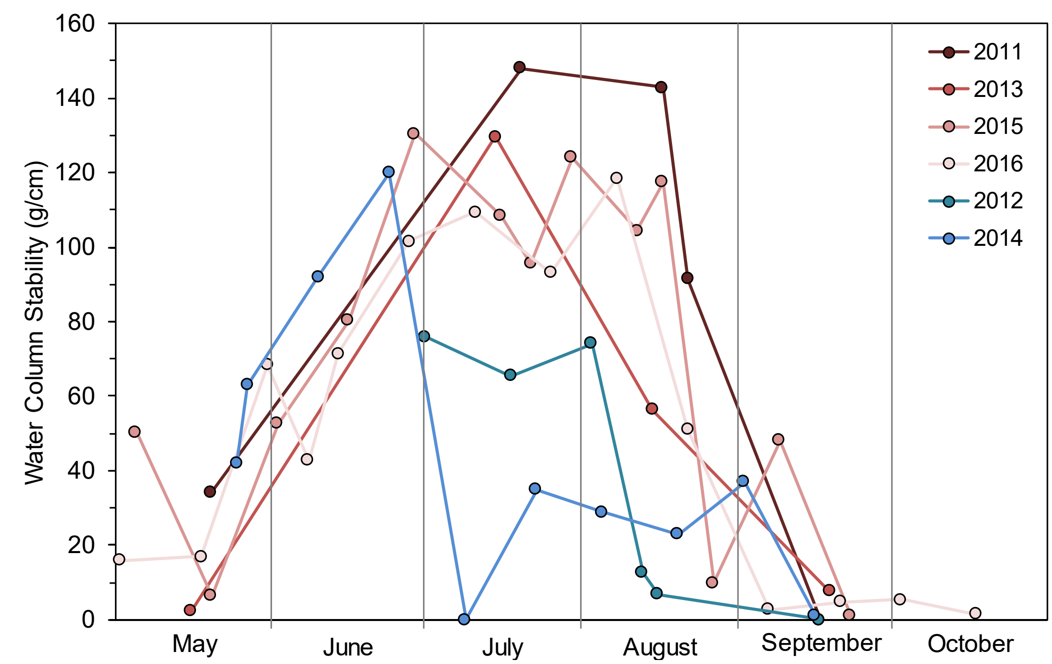


Figure 37. Kentuck Lake measured growing season water column stability from 2011-2016. Red lines (2011, 2013, 2015, 2016) indicate years where high water column stability was maintained through most of the summer and the lake experienced high phosphorus and chlorophyll-α concentrations. Blue lines (2012 and 2014) indicate years where the water column was destabilized early in the growing season and resulting lower water column stability through the remainder of the growing season. These years had lower phosphorus and chlorophyll-α concentrations.

Given what was observed in 2012 and 2014, it is believed that if the destabilization of Kentuck Lake's water column can be completed early in the growing season, phosphorus release from bottom sediments can be reduced. With less phosphorus released from bottom sediments, it would be expected that the occurrence and severity of cyanobacterial blooms would also decline. The data indicate that the water column of Kentuck Lake can become destabilized early in growing season due to natural climatic conditions; however, the available data also indicate that this does not happen every year. Therefore, it is recommended that the KLPRD pursue the implementation of a system (e.g. aeration) which would be designed to increased water movement and decrease water column stability, facilitate deeper mixing of oxygenated water, and reduce the duration and size of the area of anoxia. The dissolved oxygen data indicate that high-phosphorus years occur when areas shallower than 20 feet become anoxic. Figure 38 illustrates that the acreage of bottom sediments exposed to anoxia increases exponentially at depths shallower than 20 feet.

Part of this management goal is to reduce chlorophyll-*a* concentrations below the threshold for recreational impairment so that Kentuck Lake can be removed from the 303(d) list of impaired waters. If the internal loading of phosphorus can be reduced by 75% from what was measured in high-phosphorus years, modeling predicts that phosphorus concentrations would be reduced to approximately 24 µg/L and average summer chlorophyll-*a* concentrations would be less than 10 µg/L. In 2017, naturally-induced early-season mixing resulted in approximately 80% less internally loaded compared to 2015 and growing season phosphorus and chlorophyll concentrations were reduced to 27 µg/L and 8 µg/L, respectively. The modeling indicates that internal phosphorus loading in high-phosphorus years needs to be reduced by at least 50% to bring chlorophyll-*a* concentrations below the nuisance algal bloom threshold of 20 µg/L. This reduction would place Kentuck Lake well below the impairment threshold and could lead to the lake being delisted.

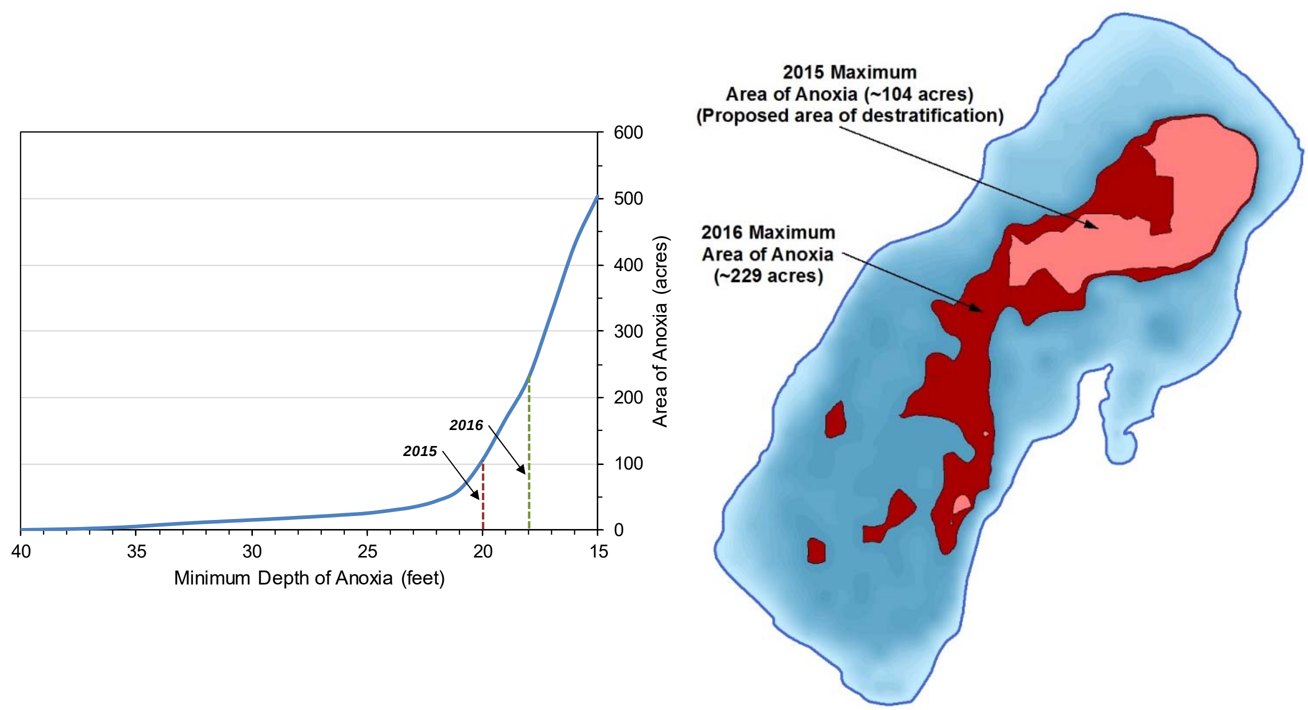


Figure 38. Kentuck Lake 2015 and 2016 maximum measured area of anoxia graph (left) and lake map (right).

An aeration system was employed in Cedar Lake, a lake with similar morphometry to Kentuck Lake, in 1991 in an effort to reduce internal loading of phosphorus and the occurrence of cyanobacterial blooms. Investigations by James et al. (2015) found that the aeration system employed in Cedar Lake actually exacerbated the problem, doubling internal phosphorus loading. They concluded that there was an insufficient amount of iron to rebind to the sediment-released phosphorus when it was mixed up into oxygenated waters. Exacerbating internal loading in Kentuck Lake is certainly not the intent of this action, but it is a risk that has to be considered. James et al. (2015) indicate that a mass ratio of iron to phosphorus (Fe:P) of > 3.6:1 is required for complete rebinding of phosphorus by iron. Mass Fe:P ratios were calculated using near-bottom samples collected in anoxic waters in Kentuck Lake and show higher average Fe:P ratios of 14:1 in 2015 and 12:1 in 2016. With these higher Fe:P ratios in Kentuck Lake, it is believed that artificial water column destabilization will not result in intensified internal nutrient loading.

It is likely that an aeration system will be the most feasible method for achieving artificial water column destabilization in Kentuck Lake. Because the deepest areas of the lake are located in the northern portion of the lake (Figure 38), the onshore housing will likely need to be placed somewhere along the northeast shoreline on national forest property. The KLPRD has elected to move forward with the implementation of an aeration system in Kentuck Lake and they have begun soliciting aeration designs and quotes to gain a better understanding of the costs associated with installing and maintenance. The KLPRD has also initiated conversations with the US Forest Service to inquire about placing the housing on national forest property. It is understood that the specifics of an aerator unit will need to be determined based upon its placement location, and ongoing conversations with aerator manufacturers will help the KLPRD understand a reasonable estimate of cost, maintenance, etc. The KLPRD will need to get guaranteed assurance from whichever company they choose that the aeration system will destabilize the water column as requested.

Once the KLPRD has taken the steps in terms of finalizing an aeration design, cost, and have worked with the appropriate representatives for installing housing on shore and the studies discussed in the previous management action for 2018 have been completed, the next step for the KLPRD would be to submit a grant application to the WDNR Surface Water Grants program in February of 2019. This grant, like the one used to fund this project, would be under the Lake Protection – Lake Management Plan Implementation category of this program (http://dnr.wi.gov/lakes/grants/).

The application would propose a three-year project that would contain partial costs for the aeration system equipment, a water quality monitoring plan, aquatic plant monitoring plan, an education program to share results of the aeration project with KLPRD members and the public, and in-kind volunteer time for monitoring and implementation of the program. The KLPRD would ask support from other management entities (WDNR, County, Township, etc.) to display to WDNR administrators that the project has the support of these groups. Notification of the grant award would likely occur by March or early April of 2019, meaning the aeration system would be installed and activated in the spring of 2019 and water quality monitoring would take place in 2019, 2020, and 2021.

The water quality monitoring would involve the collection of temperature and dissolved oxygen profiles and water quality samples from the deep hole sampling location every other week (biweekly) approximately two weeks after ice out through early October in 2019, 2020, and 2021. Water quality samples would be collected from the near-surface (3 feet) and near-bottom (approximately 3 feet above the bottom). Near-surface samples would be analyzed for nutrients (phosphorus and nitrogen, both organic and inorganic), chlorophyll-*a*, and total iron. Near-bottom samples would be analyzed for these same parameters with the exception of chlorophyll-*a*. Near-surface samples would be collected for zooplankton which would be analyzed for taxa identification, abundance, and mean body length. Similarly, samples would be taken to collect phytoplankton which would be analyzed for taxa identification, abundance, and biovolume.

The aeration project would be considered successful if the following criteria are met:

1. Water column temperature profile data indicate that the aeration system was successful at reducing water columns stability over the course of the growing season in Kentuck Lake. Water column stability (S) would be maintained below 100 g/cm over the course of the growing season, ideally around 50 g/cm.
2. Dissolved oxygen data indicate the duration and/size of the area of summer anoxia in bottom waters was reduced to at least the area of 30 feet and deeper (~11 acres).
3. The internal loading of phosphorus is reduced to a level at which chlorophyll-*a* concentrations are < 20 µg/L for 95% of the days (59 days) between July 15 and September 15, the threshold at chlorophyll-*a* concentrations will remain below the threshold for recreational impairment. Data from Kentuck Lake indicate near-surface total phosphorus concentrations need to remain below 32 µg/L and growing season

internal load of phosphorus needs to remain below 1,000 pounds to achieve chlorophyll-*a* concentrations below 20 µg/L.

Action Steps:

1. KLPRD Board of Directors solicit quotes for aerator designs and costs for destabilizing the water column within areas 20 feet and deeper in Kentuck Lake.

2. KLPRD Board of Directors coordinate onshore housing site evaluation with US Forest Service, WDNR representatives, and nearby private riparian property owners.

3. Consultant solidifies water quality monitoring design with assistance from WDNR and other agencies as applicable.

4. Consultant creates preliminary cost estimate with costs of aeration system provided by KLPRD.

5. KLPRD applies for a WDNR Lake Protection Grant for February 1, 2019 deadline to aid in funding costs of aeration purchase and installation and water quality monitoring.

Revised Management Goal 2: Continue Monitoring and Management of Existing Aquatic Invasive Species Populations in Kentuck Lake

Management Action: Continue KLPRD volunteer monitoring of Kentuck Lake's curly-leaf pondweed (CLP) population.

Timeframe: Continuation of current effort

Facilitator: KLPRD Board of Directors (suggested)

Description: As is discussed within the Non-Native Aquatic Plant Survey Results Section (Section 3.6), Kentuck Lake's CLP population is currently very small, indicated by the fact that only three single plant occurrences were able to be located in 2016. Given the CLP population has remained small since its presence was detected 17 years ago, annual professional monitoring of the CLP population in Kentuck lake is likely not warranted. The KLPRD has been conducting annual volunteer monitoring for non-native plants annually, and the small CLP population lends itself well for volunteer monitoring. The KLPRD will survey littoral areas of the lake in June and use a district-owned GPS unit to record locations of CLP. Their findings will be reported to resource managers on an annual basis, and their data will yield an understanding of how the CLP population is changing over time. If the KLPRD volunteers find that the CLP population is expanding and/or forming larger, monotypic colonies, they can consider having professionals complete and Early-Season AIS Survey on the lake to assess the population and determine if control strategies are warranted.

Action Steps:

1. KLPRD volunteers continue to search littoral areas of Kentuck Lake for occurrences of CLP and report annual findings to professionals (i.e. WDNR and consultants).

2. KLPRD volunteer monitors update their skills by attending AIS identification training sessions conducted by the Vilas County AIS Coordinator Catherine Higley (715-479-3738).

3. Trained volunteers recruit and train additional district members.

Management Action: Monitor and control Eurasian watermilfoil (EWM) population in Kentuck Lake.

Timeframe: Continuation of current effort

Facilitator: KLPRD Board of Directors (suggested)

Description: As is discussed within the Non-Native Aquatic Plant Survey Results Section (Section 3.6), assessments of Kentuck Lake's EWM population in 2015-2017 indicate that the population is widespread throughout littoral areas of the lake but had a relatively low littoral occurrence of 5% in 2017. As of 2017, an herbicide treatment strategy was not believed to be warranted and it was recommended that the EWM population continue to be monitored to track its changes over time. At a meeting with the KLPRD in November of 2016, Onterra ecologists presented that one of the most feasible methods for controlling a widespread population of EWM like that found in Kentuck Lake was through the use of herbicide applications – specifically, and early-spring whole-lake treatment. In a stakeholder survey sent to KLPRD members in 2013, 61% of 90 respondents indicated support for the use of herbicides in Kentuck Lake to control non-native aquatic plants. In addition, 80% indicated *definitely* or *probably yes* when asked if they believed aquatic plant control was needed in Kentuck Lake.

While the localized spot treatments completed in 2013 were considered successful, continued use of this strategy in Kentuck Lake given the more widespread nature of EWM would not target the population at the lake-wide level and would only achieve nuisance control. The KLPRD agrees that use of herbicides to control EWM needs to have more favorable and predictable results for the control action to be worth the risk of using herbicides. It is also understood that targeting the EWM on a lake-wide basis will produce more predictable results.

The KLPRD had many discussions regarding the EWM Long Term Trends data and how even in unmanaged EWM populations, EWM frequencies can vary up and down from year to year. Because of that knowledge and the risk assessment of herbicide use, the KLPRD chose to tolerate the reduced ecosystem services the lake provides (areas of reduced navigation and aesthetics) and select a higher threshold for when a whole-lake treatment will take place.

The KLPRD will continue to visually monitor the EWM population, and if they feel EWM is increasing within the lake, they will contact a qualified consultant to conduct a whole-lake point-intercept survey. If the EWM littoral frequency of occurrences is greater or equal to 15%, the KLPRD will consider initiating the planning and pretreatment steps necessary to conduct a whole-lake treatment on Kentuck Lake, likely utilizing 2,4-D at a standard epilimnetic dosing strategy (0.275 – 0.375 ppm ae). Continued investigations of whole-lake 2,4-D treatments prior to initiation of one on Kentuck Lake may alter the accepted dosing range.

Active Management Monitoring Strategy:
A cyclic series of steps will be used to plan and implement the control efforts. The series includes conducting the following surveys during the year prior to the treatment, year of the treatment, and year following the treatment:

- A lake-wide mapping assessment of EWM completed while the plant is at peak growth stage (peak biomass).
- A detailed assessment of bathymetric data from the lake, potentially augmenting with an acoustic survey of the lake.
- Quantitative assessments of the native and non-native aquatic plant community of the lake utilizing point-intercept survey methodology.

During the year of the treatment, the project would include verification and refinement of the treatment plan immediately before control strategies are implemented. This potentially would include refinements of herbicide application areas, assessments of growth stage of aquatic plants, and documentation of thermal stratification parameters that influence the final dosing strategy.

Volunteer-based monitoring of temperature profiles would also be coordinated surrounding the treatment, as well as collection of post treatment herbicide concentration sample at multiple locations and sampling intervals. The success criteria of a whole-lake treatment would be a 70% reduction in EWM littoral frequency of occurrence comparing point-intercept surveys from the year prior to the treatment to the year after the treatment. This means if the treatment occurs in 2020, the year before treatment would be 2019 and the year after treatment would be 2021. Regardless of treatment efficacy, a whole-lake treatment would not be conducted during the year following the treatment.

As shown in Figure 36, the EWM population is often greatly impacted the year of large-scale treatment and low EWM levels are maintained for 4-5 years following the control action. Many lake groups initiate a whole-lake herbicide strategy with the intention of implementing smaller-scale control measures (herbicide spot treatments, hand-removal) when EWM begins rebounding. Occasionally, the EWM rebounds in a fashion that does not lend well to these methods and the lake groups then tolerates the EWM to until it again exceeds the predefined threshold to trigger another whole-lake treatment.

If the KLPRD initiates a large-scale treatment that does not meet the success criteria, they will consider alternative chemical control use patterns for ecosystem restoration or possibly chemical or non-chemical control techniques for restoring localized areas to deliver higher ecosystem services. Appropriate implementation triggers and success criteria for these actions would need to be developed through an updated lake management planning effort. Funds from the Wisconsin Department of Natural Resources Aquatic Invasive Grant Program will be sought to partially fund this control program. Specifically, funds would be applied for under the Established Population Control classification, currently on February 1 of each year.

Based on the data collected over the three-year project, the KLPRD would revisit their management plan as it applies to EWM control and monitoring. Based upon the information gained during the multi-year control project, the KLPRD would update their management plan as appropriate. This may include targeting low-level EWM populations through coordinated volunteer and professional hand-harvesting efforts.

Action Steps:

1. KLPRD retains and contacts qualified professional to conduct a whole-lake point-intercept survey when they feel EWM has increased markedly within the lake.

2. KLPRD retains qualified professional assistance to develop a specific project design utilizing the methods discussed above.

3. Apply for a WDNR Aquatic Invasive Species Grant based on developed project design when a whole-lake treatment is proposed to occur.

4. Initiate control and monitoring plan.

5. Update management plan to reflect changes in control needs and those of the lake ecosystem.

LITERATURE CITED

Anderson, N.J., B. Rippey, & A.C. Stevenson, 1990. Diatom assemblage changes in a eutrophic lake following point source nutrient re-direction: a palaeolimnological approach. Freshwat. Biol. 23:205-217.

Appleby P.G. 1998. Dating recent sediments by ^{210}Pb: Problems and solutions. Proc. 2nd MKS/EKO-1 Seminar. Helsinki. 2-3 April 1997. STUK. Helsinki: 7-24.

Appleby P.G. 2001. Chronostratigraphic techniques in recent sediments. In: Last, W.M. and Smol, J.P. (eds), Tracking Environmental Change Using Lake Sediments. Volume 1: Basin Analysis, Coring, and Chronological Techniques. Kluwer Academic Publishers, Dordrecht, The Netherlands. pp. 171-203.

Appleby, P.G., and F. Oldfield, 1978. The calculation of lead-210 dates assuming a constant rate of supply of unsupported ^{210}Pb to the sediment. Catena. 5:1-8.

Barbiero RP and Welch EB (1992) Contribution of benthic blue-green algal recruitment to lake populations and phosphorus translocation. Freshwater biology 27:249–260.

Benndorf J, Böing W, Koop J, and Neubauer I (2002) Top-down control of phytoplankton: the role of time scale, lake depth and trophic state. Freshwater Biology 47:2282–2295.

Beutel MW (2006) Inhibition of ammonia release from anoxia profundal sediments in lakes using hypolimnetic oxygenation. Ecological Engineering 28:271–279.

Birks, H.J.B., J.M. Line, S. Juggins, A.C. Stevenson, & C.J.F. ter Braak, 1990. Diatoms and pH reconstruction. Phil. Trans. R. Soc., Lond., series B 327:263-278.

Bradbury, J.P., 1975. Diatom stratigraphy and human settlement in Minnesota. Geol. Soc. America Spec. Paper. 171:1-74.

Callender E. and P.C. Van Metre. 1997. Reservoir sediment cores show U.S. lead declines: Environ. Sci. and Tech. 31:424-428.

Carey CC and Rengefors K (2010) The cyanobacterium *Gloeotrichia echinulata* stimulates the growth of other phytoplankton. Journal of Plankton Research 32(9):1349–1354.

Carey CC, Weathers KC, Ewing HA, Greer ML, and Cottingham KL (2014) Spatial and temporal variability in recruitment of the cyanobacterium *Gloeotrichia echinulata* in an oligotrophic lake. Freshwater Science 33(2):577–592.

Carney, H.J. 1982. Algal dynamic and trphic interactions in teh recent history of Frains Lake, Michigan. Ecology. 63:1814-1826.

Carpenter SR, Kitchell JF, Hodgson JR, Cochran PA, Elser JJ, Elser MM, Lodge DM, Kretchmer D., and He X (1987) Regulation of lake primary productivity by food web structure. Ecology 68(6)1863–1876

Cottingham KL, Ewing HA, Greer ML, Carey CC, and Weathers KC (2015) Cyanobacteria as biological drivers of lake nitrogen and phosphorus cycling. Ecosphere 6(1):1–19.

Davis, M. 1995. Northern Choices: Rural Forest County in the 1920's, Part I. Wisconsin Magazine of History. 79:2-31.

Engstrom, D.R., E.B. Swain, and J.C. Kingston, 1985. A paleolimnological record of human disturbance from Harvey's Lake, Vermont: geochemistry, pigments, and diatoms. Freshwat. Biol. 15:261-288.

Frey, D.G. 1955. Distributional ecology of the cisco (*Coregonus artedi*) in Indiana. Invest. Indiana Lakes & Streams. (4)7:177-228.

Garrison, P.J. 2000a. Paleoecological Study of Beulah Lake, Walworth County. Wisconsin Department of Natural Resources. PUB-SS-950 2000. 18p.

Garrison, P.J. 2000b. Paleoecological Study of Geneva Lake, Walworth County. Wisconsin Department of Natural Resources. PUB-SS-952 2000. 25p.

Garrison, P.J. 2003. Paleoecological Study of Big Cedar Lake, Washington County. Wisconsin Department of Natural Resources. PUB-SS-984 2003. 15p.

Garrison P.J. and R.S. Wakeman. 2000. Use of paleolimnology to document the effect of lake shoreland development on water quality. J. Paleolim. 24:369-393.

Gobeil C., W.K. Johnson, R.W. MacDonald, and C.S. Wong. 1995. Sources and burdens of lead in the St. Lawrence estuary sediments—isotopic evidence: Environ. Sci. and Tech. 29:193-201.

James WF, Sorge PW, and Garrison PJ (2015) Managing internal phosphorus loading and vertical entrainment in a weakly stratified eutrophic lake. Lake and Reservoir Management 31:292– 305.

Juggins, S. 2014. C2 User guide. Software for ecological and palaeoecological data analysis and visualisation (version 1.7.6). University of Newcastle. Newcastle upon Tyne. 69 pp.

Juggins, S. and H.J.B. Birks. 2012. Quantitative environmental reconstructions from biological data. In: H.J.B. Birks, A.F. Lotter, S. Juggins, J.P. Smol (eds). Tracking Environmental Chagne using Lake Sediments: Data Handling and numerical Techniques. Springer. Dordrecht. p, 431-494.

Juggins, S. N.J. Anderson, J.M. Ramstack, A.J. Heathcote. 2013. Reconstructing epilimnetic total phosphorus using diatoms: statistical and ecological constraints. J. Paleolimnol. 49:373-390.

Kamarainen AM, Yuan H, Wu CH, and Carpenter SR (2009) Estimates of phosphorus entrainment in Lake Mendota: a comparison of one-dimensional and three-dimensional approaches. Limnol. Oceanogr.:Methods 7:553–567.

Krishnaswami, S. and D. Lal. 1978. Radionuclide limnochronology. In: Lerman, A. (ed.), Lakes: Chemistry, Geology, Physics. Springer-Verlag, NY: 153-177.

McCarthy MJ, Gardner WS, Lehmann MF, and Bird DF (2013) Implications of water column ammonium uptake and regeneration for the nitrogen budget in temperate, eutrophic Missisquoi Bay, Lake Champlain (Canada/USA). Hydrobiologia 718:173–188.

Molot LA, LI G, Findlay DL, and Watson SB (2010) Iron-mediated suppression of bloom-forming cyanobacteria by oxine in a eutrophic lake. Freshwater Biology 55:1102–1117.

Molot LA, Watson SB, Creed IF, Trick CG, McCabe SK, Verschoor MJ, Sorichetti RJ, Powe C, Venkiteswaran JJ, and Schiff SL (2014) A novel model for cyanobacteria bloom formation: the critical role of anoxia and ferrous iron. Freshwater Bioology 59:1323–1340.

Nichols, S.A. 1999. Floristic quality assessment of Wisconsin lake plant communities with example applications. Journal of Lake and Reservoir Management 15(2): 133-141

Nürnberg GK (1988) Prediction of phosphorus release rates from total and reductant-soluble phosphorus in anoxic lake sediments. Can. J. Fish. Aquat. Sci. 45:453–462.

Onterra, LLC (2015) Kentuck Lake Comprehensive Management Plan. LPL-1485-13.

Orihel D.M., D.W. Schindler, N.C. Ballard, M.D. Graham, D.W. O'Connell, L.R. Wilson, and R.D. Vinebrook. 2015. The "nutrient-pump:" Iron-poor sediments fuel low-nitrogen-to-phosphorus ratios and cyanobacterial blooms in polymictic lakes. Limnol. Oceanogr. 60:856-871.

Osgood RA (1988) Lake mixis and internal phosphorus dynamics. Arch. Hydrobiol. 113(4)629–638

Panuska, J.C. and Kreider, J.C. 2003. Wisconsin lake modeling suite program documentation and user's manual, version 3.3 fir Windows: Wisconsin Department of Natural Resources PUBL-WR-363-94.

Pettersson K (1998) Mechanisms for internal loading of phosphorus in lakes. Hydrobiologia 373(0):21–25.

Shaw BH and Nimphius N (1985) Acid rain in Wisconsin: understanding measurements in acid rain research (#2). UW-Extension, Madison. 4pp.

Solim SU and Wanganeo A (2009) Factors influencing release of phosphorus from sediments in a highly productive polymictic lake system. Water Science and Technology 60(4):1013–1023.

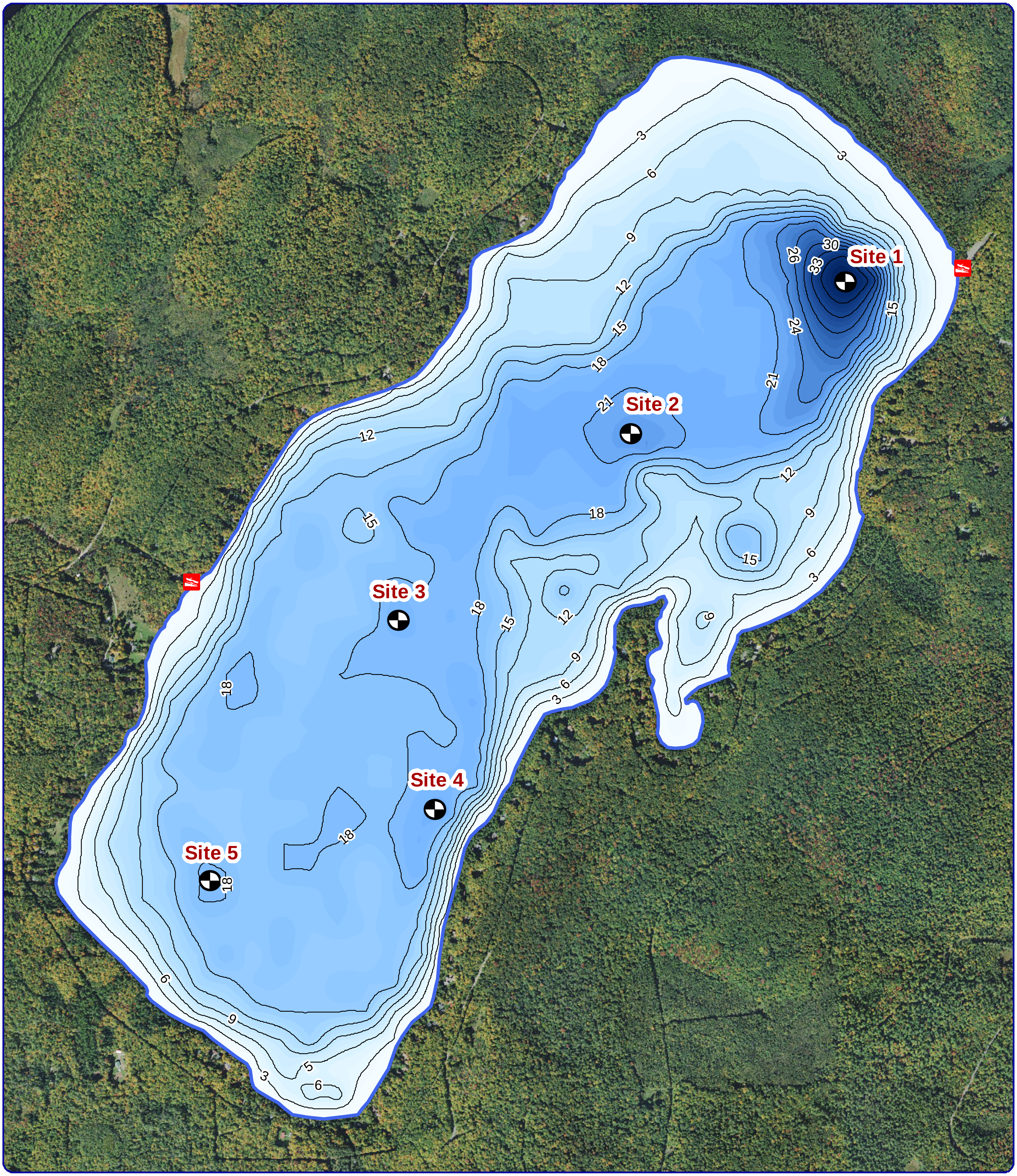
Sorichetti RJ, Creed IF, Trick CG (2014) Evidence for iron-regulated cyanobacterial predominance in oligotrophic lakes. Freshwater Biology 59:679–691.

ter Braak, C.J.F. and P. Smilauer. 2012. CANOCO Reference Manual and User's Guide: Software for Ordination (version 5.0). Microcomputer Power (Ithaca, NY, USA). 496 pp.

Weather Underground – Weather History and Data Archive. www.wunderground.com/history. Accessed January 2016.

Welch EB and Cooke GD (1999) Effectiveness and longevity of phosphorus inactivation with alum. Lake Reverv. Manage. 15:324–331.

Wetzel RG (2001) Limnology: lake and river ecosystems. 3rd Edition. Academic Press, San Diego, CA.



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Onterra LLC
Lake Management Planning
815 Prosper Road
De Pere, WI 54115
920.338.8860
www.onterra-eco.com

Sources:
Roads and Hydro: WDNR
Bathymetry: WDNR 2011; digitized by Onterra
Orthophotography: NAIP, 2015
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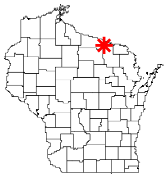

Project Location in Wisconsin

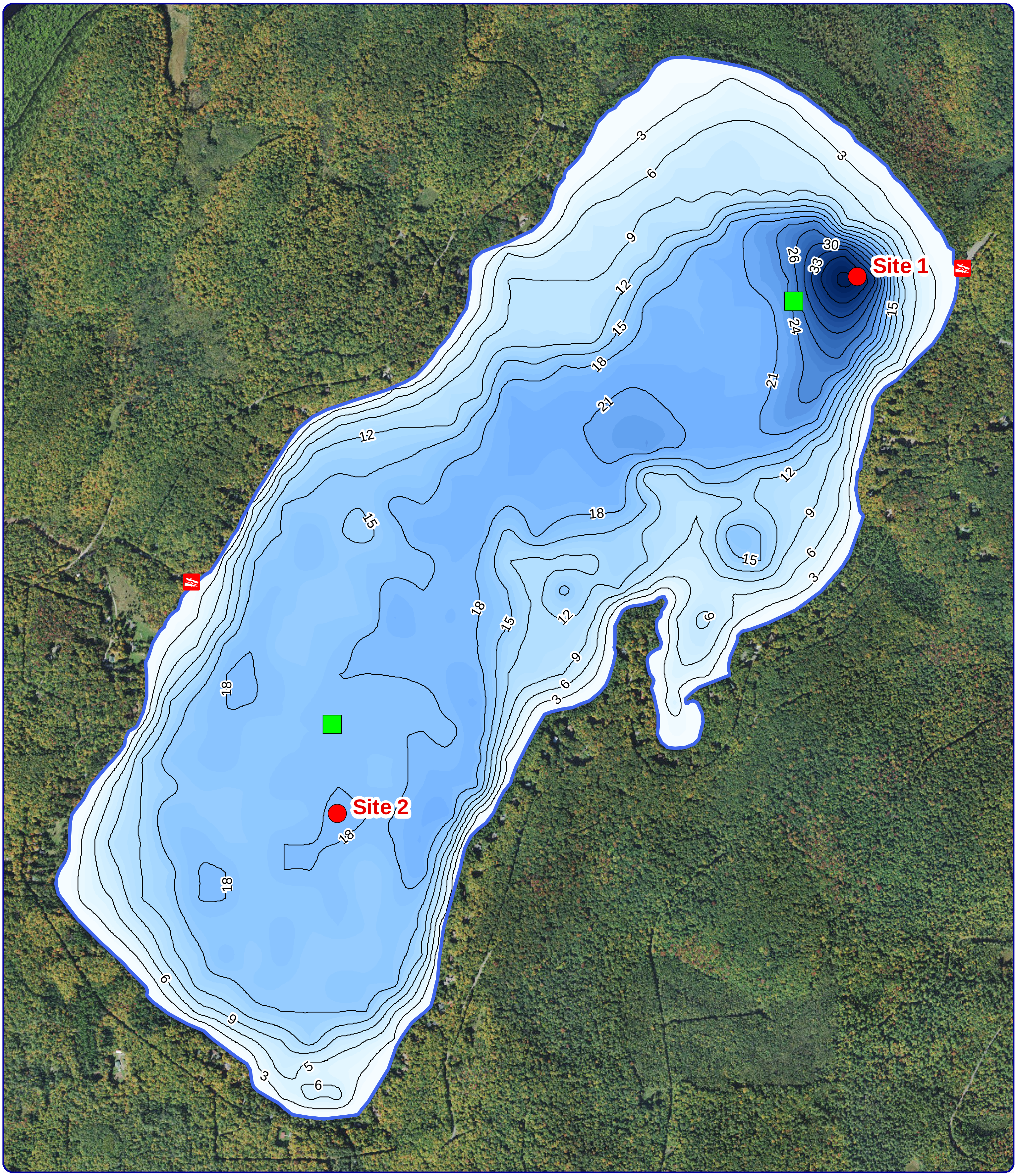
Legend

Surface Water Sampling Location

Public Access

Map 1
Kentuck Lake
Forest & Vilas Counties, Wisconsin

Surface Water Sampling Locations



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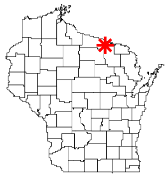

Project Location in Wisconsin

Legend

Full Sediment Core Sampling Location

Nutrient Release Sediment Core Sampling Location

Map 2

Kentuck Lake

Forest & Vilas Counties, Wisconsin

Sediment Core Sampling Locations