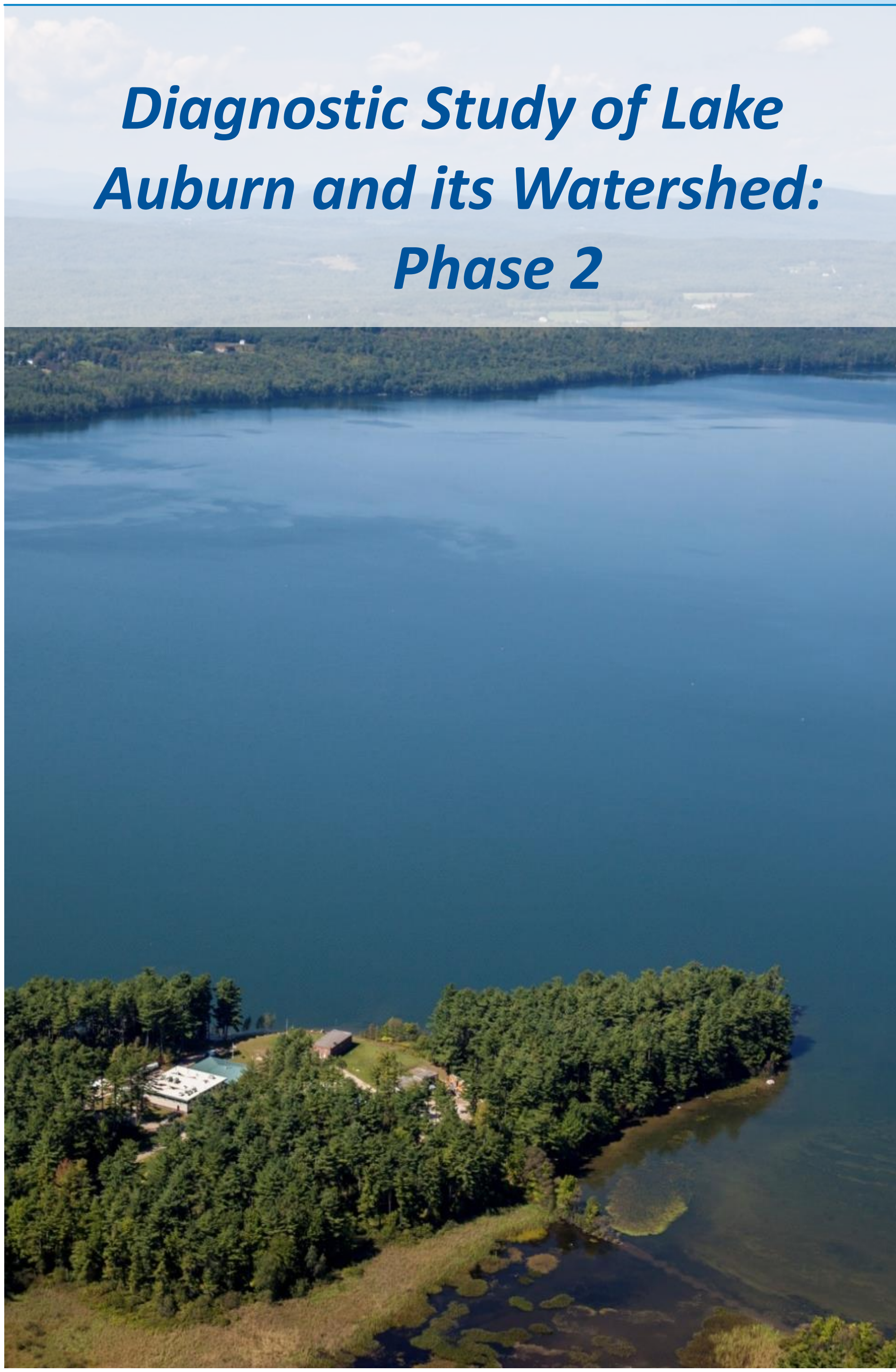
LAKE AUBURN
WATERSHED PROTECTION COMMISSION

REPORT

Diagnostic Study of Lake Auburn and its Watershed: Phase 2



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WRS

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Executive Summary

1. Project Background

Lake Auburn, the principal drinking water supply for the communities of Lewiston and Auburn, Maine, has been known for its excellent water quality as a result of a strong watershed protection program implemented by the two communities. In 2011 and 2012, however, water quality was degraded due to a combination of factors that raised turbidity in the lake to near the limit allowed under the filtration avoidance waiver granted to the Auburn Water District and Lewiston Water Division (AWD/LWD). In 2012, dissolved oxygen was severely reduced throughout the bottom waters of the lake, compromising the cold water fishery habitat and resulting in the death of some lake trout (Togue) in September 2012.

While relationships between drivers of decreased water quality are not completely known, it is clear that if the quality in the lake continues to degrade construction of more advanced water treatment facilities may be required. Also, if levels of algae in the fall were to increase further, there could be taste and odor issues in the distribution system and threats to the fishery in the lake.

This study was divided into two phases:

- **Phase 1** – Examined available data to investigate the causes of the recent degradation in water quality, and made recommendations on short-term actions that could be taken if poor water quality recurred
- **Phase 2** – Used additional data from 2013 to re-assess short-term actions and recommend long-term actions to reduce the sources and mitigate the adverse impacts of excess phosphorus on the water quality of Lake Auburn.

Water quality data from AWD/LWD and Bates College[1] underpins this study. The study team includes Comprehensive Environmental, Inc. (CEI) to address watershed changes and management actions, and Dr. Ken Wagner of Water Resource Services Inc. to advise on lake water quality and in-lake management options.

2. Phase 1 Summary

The March 2013 Phase 1 report concluded that while the degraded water quality was a serious concern, and watershed management programs needed to be continued and strengthened, an immediate in-lake management action did not need to be implemented. Instead, the following short-term management actions were recommended:

- Implement an enhanced monitoring program in the watershed and lake, and

[1] Dr. Holly Ewing and co-Principal Investigators (see Section 2) provided valuable water quality data used in this study. Their data were collected, analyzed and compiled under grants from the National Science Foundation (NSF): (1) NSF DEB-0749022, (2) NSF EF-0842112, (3) NSF EF-0842125, and (4) NSF EF-0842267.

- Obtain permits to allow for application of algicide to control nuisance algal blooms as a short-term contingency plan if turbidity levels were elevated as in 2011 and 2012.

In 2013, AWD/LWD and LAWPC implemented these recommendations, and also implemented several measures to further strengthen the watershed management program.

3. Key Findings from 2013 Water Quality Data

What was Measured

Observations from the 2013 water quality monitoring program are provided below along with a discussion of trend in the data compared to 2011/2012.

- **Surface phosphorus levels were similar to 2012; Figure 1** – Monthly average surface phosphorus concentrations in 2013 ranged from 9 to 17 µg/l, similar to 2012. These values are 10 to 50 percent higher than the limited data in 2005 and 2010, which ranged from 6 to 14 µg/l. Surface phosphorus concentrations fuel algal blooms; values below 10 µg/l are considered desirable, while values above 25 µg/l are undesirable.
- **Low dissolved oxygen in the bottom waters improved; Figure 2** – The area of bottom layer of the lake exposed to very low oxygen was similar to 2011, but smaller than 2012. On the other hand, oxygen in the bottom waters did not reach low levels until early fall 2013 about a month later than in 2012. The anoxic factor, which combines duration and extent of low oxygen, showed significant improvement in 2013. It was 9.9 in 2013 and greater than 18 in 2011/2012; prior to 2011 the anoxic factor was zero or less than one except in 2002. Low oxygen can be harmful or fatal to fish and other life, and cause phosphorus to be released from sediment, which can fuel algal blooms.
- **Shallow Secchi depth occurred in late Summer/Fall; Figure 3** -- Secchi depth is another measure of transparency, which is a surrogate for algae in Lake Auburn. Late summer/fall of 2011-12 produced the shallowest Secchi depth readings recorded since 1978, with record low values from mid-September through October. In 2013 Secchi depth was close to the long-term average until August, and then they became shallower relative to the long-term average. Except for a couple of weeks, the 2013 values were deeper than 2011 and 2012.
- **Number of days of elevated turbidity improved; Figure 4** -- Turbidity is a measure of water transparency; high turbidity in Lake Auburn indicates increases in algae[2]. In 2013, the number of days with turbidity above 2 NTU dropped significantly, while the number of days with turbidity above 1 NTU was similar. Turbidity has historically been far below 1 NTU, with only rare excursions above 2 NTU. To maintain the filtration waiver turbidity must not exceed 5 NTU for more than two events per year and not more than 5 events in ten years. The reportable turbidity at the Lake Auburn intake has never exceeded 5 NTU, with the highest level occurring in 2011.
- **No significant blue-green algal bloom in 2013** – The algal community differed from 2011/2012 by not having a large fall blue-green algal bloom. Instead the dominant algae were diatoms and

[2] Particle size distribution matters as much as the overall quantity of particles to turbidity, so there can be substantial variation in turbidity with changing algal communities

golden browns. Bates College data show *Gloeotrichia* levels in 2013 were lower than 2011/2012.

- **Tributary phosphorus data** – 2013 monitoring data was used to compare phosphorus concentrations and estimate annual loads at Townsend Brook and the North Auburn Dam. The concentrations at Townsend Brook are nearly double those measured at the North Auburn Dam, but the much higher flow at the North Auburn Dam results in an estimated annual phosphorus load of 180 kg vs. 120-130 kg from Townsend Brook.

- **Rainfall was below normal; temperature was typical; no fish kill occurred; Figure 5** – Annual rainfall in 2013 was about 8 inches below average, but the distribution of monthly rainfall in 2012 differed significantly with a dry winter and fall and a very wet June. Water temperatures at the intake were average, and the combination of sufficient oxygen in a portion of the bottom waters combined with a return to typical water temperatures supported the coldwater fishery throughout the late summer and fall.

What This Means

Overall many indicators showed significant improvement in 2013 compared to 2011/2012, but the water quality was still notably degraded from previous years. The reduction in maximum turbidity values and lack of large blue-green algal blooms meant that an algicide application was not necessary in 2013.

Watershed loads could be estimated from a robust set of dry and wet weather measurement of total phosphorus levels at Townsend Brook and the North Auburn Dam. These data suggest watershed loads alone would support excellent lake water quality.

The 2013 data support the Phase 1 analysis results that both internal and external loads can contribute to explaining the increase in phosphorus concentrations in the surface water, though the most likely explanation is that increased internal loads are largely responsible. Because the 2013 phosphorus measurements in the two main tributaries to Lake Auburn did not suggest significant uncontrollable loads discharging to the lake, it will likely be difficult to make a large reduction in watershed load (in part because of the strong watershed management program that exists today, which must be maintained and strengthened). Thus, if future conditions require implementation of a scheme to reduce phosphorus to control algal growth, reduce turbidity, and protect the coldwater fishery, the most effective approach would be to implement a measure to control the internal load from the sediments.

The 2013 data were also used to revisit the question of whether water quality changes in Lake Auburn since 2010 are part of long-term trend or result from single year events. Overall, the factors that would suggest a long-term trend improved in 2013, and there were no unusual meteorological events (as in 2011/2012) to drive further degradation of water quality in 2013.

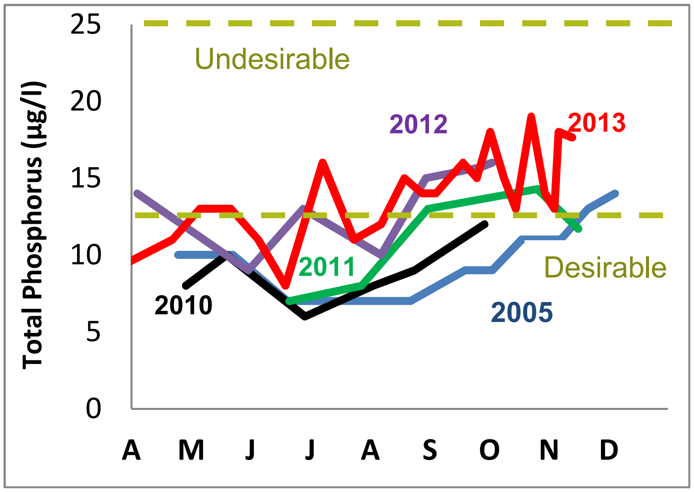


Figure 1: Concentration of Total Phosphorus in Surface Water at Deep Hole (0 to 5 meter depth); ***10 and 25 µg/l approximate the range of desirable (low algal population) and undesirable (high algal population) phosphorus levels.***

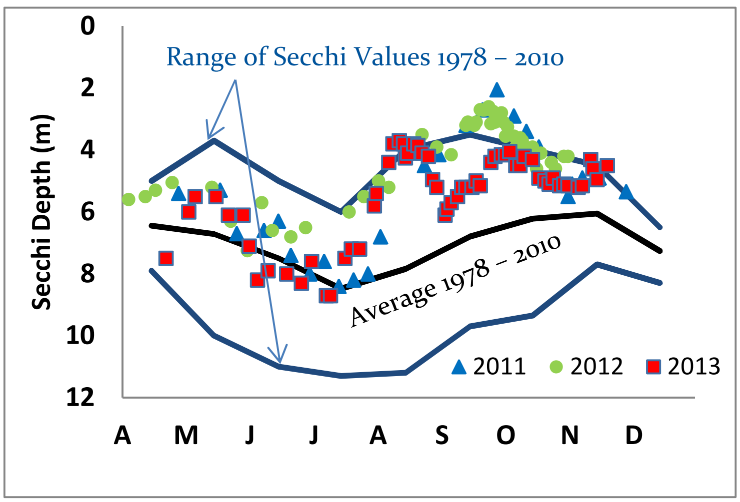


Figure 3: 2011-12 Secchi Depths Compared to 1978-2010 Average and Long-term Range

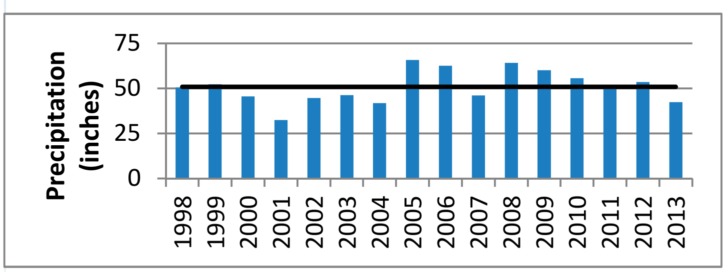


Figure 5: Annual Precipitation, Poland, Maine

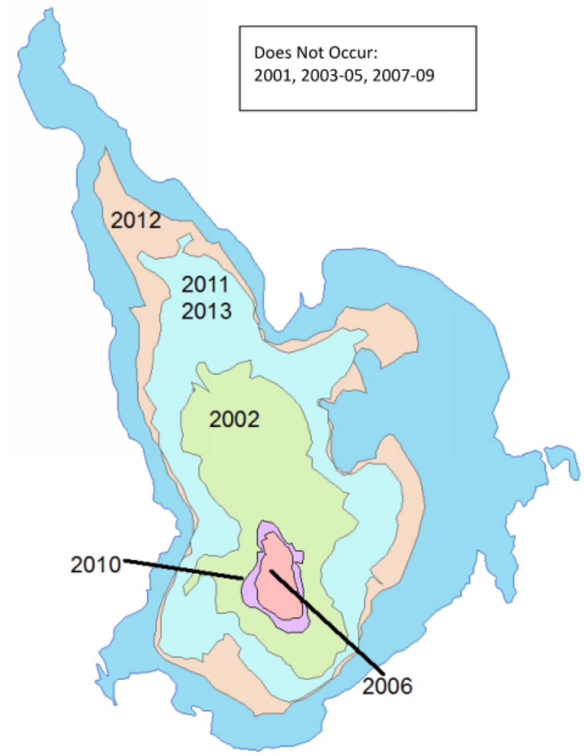


Figure 2: Bottom Area of Lake Auburn with Dissolved Oxygen < 2 mg/l

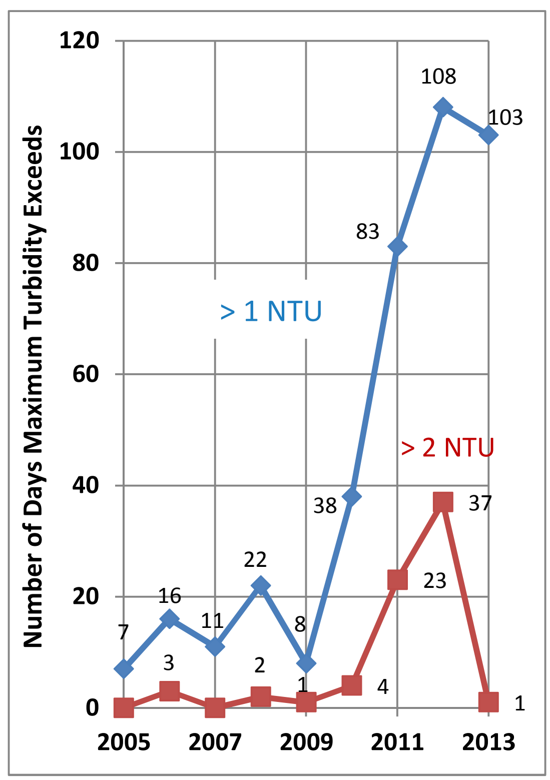


Figure 4: Days Maximum Turbidity at Raw Water Intake Exceeds 1 or 2 NTU

Is Degraded Water Quality Likely to Continue?

Factors that would indicate if degraded conditions are likely to continue are:

- **High spring total phosphorus in the surface water** -- fuels early algal growth and results in early high turbidity values;
- **Unusual warm temperatures** -- strengthens stratification and possibly the magnitude and extent of oxygen depletion in the bottom water, and promotes blue-green algal blooms; and
- **Large intense storms** – increases phosphorus load to the lake spurring algal growth.

Of these factors, there was only evidence of the first one in 2013. Spring total phosphorus concentrations were similar to 2012, and this resulted in a bloom of golden brown algae and 10 occurrences of maximum daily turbidity above 1 NTU in May, which was much higher than the previous record of two in May 2007.

While 2013 saw improvements in many water quality indicators, it would take a long time (in absence of unusual meteorological events) for the lake to return to pre-2010 phosphorus concentrations (closer to the desirable concentration of 10 µg/l). Lake Auburn has a 4.8-year residence time (USGS, 2004) and additional total phosphorus mass that became present in the lake in 2011 and 2012 would require nearly a decade to be reduced in concentration to levels that result from watershed loads alone. Enhanced control of watershed loads would aid this process.

Looking to the future, AWD/LWD need to watch for signs of degraded water quality and consider the need to implement additional in-lake management measures. Two key questions to be addressed are (1) will the lake's phosphorus levels in the surface water increase, stabilize or decrease from where they are today, and (2) is the algal community that develops each year from these phosphorus levels acceptable from the point of view of treatability, aesthetics, and risk for losing the filtration avoidance waiver?

4. Long-term Management Recommendations

The recommended long-term management plan for Lake Auburn consists of three elements:

- Continued monitoring of the watershed and lake water quality
- Watershed management activities
- In-lake management activities

Continued Monitoring

The 2013 monitoring program for the watershed and the lake should be continued with a few changes that are documented in the Phase 2 report. Data from the monitoring program should be evaluated on both an on-going and annual basis. On-going evaluation allows for identification of trends that need to be addressed immediately. These could include increases in turbidity in the lake, atypical results from watershed monitoring, or occurrence of nuisance algae. In addition, an annual data summary should be prepared at the end of each year.

Watershed Management

AWD/LWD and the LAWPC should continue and strengthen their on-going watershed management program that serves as the first barrier in their multiple barrier approach to providing safe drinking water.

Estimates of the external load of phosphorus to Lake Auburn show that it has been controlled to a substantial extent through existing watershed programs to control non-point sources of phosphorus. However, recent investigations (CEI, 2013a and Summit Environmental, 2014) have suggested some areas of concern within the watershed, particularly the occurrence of many sediment deltas of recent origin around the lake's perimeter and in the watershed and in the Townsend Brook watershed. Both concerns focus on erosion which is of concern because sediment adds phosphorus load to the lake and most likely oxygen demanding substances as well. Ultimately, the reserves of phosphorus in Lake Auburn's sediments have their origin in the watershed and minimizing watershed loading is in the best interest of the lake and its users.

One of the most important controls for sediment load that exists in the watershed today are the impoundment behind the North Auburn Dam (the Basin) and other upstream ponds that exist as part of a linked stream pond system. These ponds serve as natural sedimentation basins and monitoring sediment accumulation and maintaining their capacity to store sediment (e.g., through dredging if needed) should be a high priority.

CEI (2013b) prepared a Phase 2 report that included recommendations for structural and non-structural BMPs for the watershed. The structural BMPs focused on controls of sediment from problem areas noted in the lake and watershed or control of potential spills that could occur from the roads that surround the lake. AWD/LWD should continue to take advantage of opportunities to implement these BMPs as they arise (e.g., through use of dedicated watershed funds and available grant funding), focusing initially on controlling erosion at locations that discharge directly to the lake.

The CEI Phase 2 report also included recommendations on non-structural BMPs. The recommendations from that report are summarized below. AWD/LWD has already started to implement many of these and we recommend full implementation of these recommendations.

- Amending regulations to promote low impact development, amend the zoning ordinance to strengthen subsurface watershed disposal requirement, update the phosphorus control ordinance, amend the ordinance to prohibit certain land uses within the watershed, and adding a steep slope ordinance.
- Developing a public education program, including an enhanced website and educational materials, and education programs for schools, farmers, residents on the value of buffers and gravel road maintenance.
- Continuing watershed monitoring activities including tributary water quality and forestry activity.
- Setting aside funds for maintaining watershed lands.
- Continuing with key parcel land acquisition and conservation efforts of environmentally sensitive lands.

- Developing trail networks on LAWPC properties.
- Continuing to control invasive species.
- Continuing with the gull management program.

In addition, we recommend that AWD/LWD continue with identification and control of sources of phosphorus in the watershed including working with the golf course to reduce use of fertilizers, working with farmers in the surrounding area to encourage best practices, and modeling best management practices concerning the timber harvesting plan that was developed in 2013 and encouraging their use throughout the watershed.

In-lake Management

During Phase 2, a wide range of in-lake management options were assessed with respect to their ability to reduce or prevent future algal blooms, episodes of high turbidity and to protect the coldwater fishery in Lake Auburn. The short list of feasible and applicable in-lake management measures are provided below along with the selection of specific treatment techniques for Lake Auburn.

- **Dredging** – this option was subsequently eliminated due to its much higher cost
- **Hypolimnetic oxygenation** – Among several types of hypolimnetic oxygenation systems, a diffused oxygen distributor system was selected for Lake Auburn based on its lower installation cost, simpler operation (this oxygen system requires no power or moving parts), and similar operational costs (oxygen vs. power requirements for other oxygenation systems)
- **Phosphorus inactivation –** Among the techniques for inactivating phosphorus an aluminum sulfate (alum) application targeting the surficial sediments was selected for Lake Auburn.

Table 1 provides a comparison of the diffused oxygen distributor system for hypolimnetic oxygenation and alum application for phosphorus inactivation.

Implementation of an in-lake management system could be needed within the next several years to drive a significant improvement in water quality in Lake Auburn. It is not needed immediately, however, due to the contingency plan AWD/LWD has to apply algicide if algal blooms similar to 2011/2012 recur. While the lake's water quality could continue to improve, as was seen in many indicators in 2013, changing climatic factors (e.g., more frequent intense rain events and the increase in stratification stability) will pose hurdles to continued water quality improvement. Nonetheless, because recurrence of degraded water quality similar to 2011 or 2012 is heavily dependent on climate and other factors outside of our control, it is sensible to begin planning now to implement an in-lake management system.

From the perspective of effectiveness of phosphorus control, there is no technical reason to prefer one of these treatments over the other. Planning level costs for these two systems are similar, with the higher capital cost of alum being balanced with the operation and maintenance cost of the oxygenation system. We recommend AWD/LWD conduct the necessary preparations to be able to implement either system should an in-lake system be required. Ultimately the selection of a system will require consideration of the tradeoffs related to operating an aeration system on an on-going basis, stakeholder acceptance, regulatory approval, potential for benefits to the lake's aquatic life, and flexibility provided by the management action.

Table 1
Comparison of Hypolimnetic Oxygenation with Phosphorus Inactivation

Factor	Hypolimnetic Oxygenation	Phosphorus Inactivation
Capital cost[1]	$2.6 to 4.2 million (best est. $3.4 million)	$1.8 to 6.2 million (best est. $4.1 million)
Annual operational cost[1]	$68,000 to $310,000 (best est. $155,000)	None
Mode of algal control	Suppresses phosphorus release from sediment; also reduces iron release, which may aid in control of blue-green algae	Suppresses phosphorus release from sediment, removes some phosphorus from water column during treatment
Additional benefits	Enhances deep-water habitat for coldwater fishery through increased oxygen levels; may reclaim entire bottom layer or any targeted portion thereof	May reduce oxygen demand from settling algae; tends to reduce oxygen demand in the middle layer and may extend oxygen about 10 feet deeper into bottom layer than might otherwise occur
Longevity of benefits	Indefinite while system is running; benefits lost quickly (days to weeks) when system is not running. System would likely run about 2 months each year.	Typically at least 10 years for deeper lakes, usually close to 20 years, not expected beyond 30 years. Control of watershed load needed to extend longevity.
Longevity of equipment	Most systems have maintenance schedule to replace in-lake portion of system about every 10 years	No equipment to replace, but may require repeat application any time after 10 years
Toxic impacts	None	Risk during actual treatment depends on pH and dose.
Sediment Impacts	Will encourage decomposition of organic sediment and lowering of oxygen demand; may have decreasing (but not eliminated) need for oxygen addition with time.	Adds minor amount of material to sediment. Some smothering of some benthic organisms expected; recovery expected within 2 years, possibly with better quality assemblage

[1] *The capital and O&M costs presented in this table were marked up to include 35% for contingencies given the planning level basis of the cost estimates.*

5. Proposed Action Plan

The proposed plan of action is as follows.

1. Continue to monitor the lake water quality with the modified monitoring program, and prepare an annual summary of the monitoring data.

2. Continue to maintain and strengthen the watershed management program to control external loads of phosphorus through both structural and non-structural BMPs as both a best practice as part of a multiple barrier system for providing safe drinking water, and also actions that could potentially delay the need for and/or improve the success of any large scale in-lake management system.

3. Maintain the ability to apply algicide (by renewing the permit as necessary) as a short-term measure to control a significant algal bloom before it develops. The timing of treatment would be triggered by an increase in turbidity above 1.5 NTU as a rolling two-day average with collaboration that algae are responsible for the elevated turbidity.

4. Plan for the implementation of either a diffused oxygen distributor system to oxygenate a portion or all of the hypolimnion in Lake Auburn or an alum treatment to bind phosphorus in the sediment in the lake, including:

    a. initiating outreach to the regulatory agencies and stakeholders

    b. establishing the ability to fund the capital costs to implement either management approach and to continue to fund operation and maintenance of an oxygenation system,

    c. identifying the location for land facilities for an oxygenation system (secured fenced area for liquid oxygen storage tank on a concrete pad and vaporizer),

    d. collecting additional sediment samples to help refine the area for an alum treatment,

    e. identifying permitting requirements for both management approaches, and

    f. preparing a draft procurement document for both management approaches with performance specifications, which will need to be reviewed and validated with most recent information after the decision is made to implement one of the approaches.

5. If an algicide treatment is required, immediately review that current year's water quality data and climatic events to determine a likely cause(s), and unless the cause(s) is highly unusual (e.g., 2011's breakdown of stratification due to passage of Hurricane Irene) initiate design and permitting to implement either a diffused oxygen distributor system or alum application by July of the following year. The design of the oxygenation system would use all the temperature and dissolved oxygen profiles, including those collected in the year when the algicide was applied. The area to be treated with alum would be informed by current and additional data on sediment phosphorus levels.

Section 1

Introduction and Overview

Lake Auburn is the drinking water supply for the communities of Lewiston and Auburn, Maine. Historically, Lake Auburn has been known for its excellent water quality, and the Auburn Water District and Lewiston Water Division (AWD/LWD) were granted a filtration waiver for Lake Auburn from Maine Division of Environmental Health in 1991. Lake Auburn's excellent water quality results from its largely undeveloped watershed and the strong watershed protection program implemented by the utilities. The ongoing work to protect the watershed and the lake to maintain the filtration waiver has resulted in significant cost savings to AWD/LWD by avoiding filtration over the years.

In late summer/fall 2012, water quality in Lake Auburn was degraded due to a combination of factors that raised turbidity in the lake close to the limit allowed under the filtration avoidance waiver granted to AWD/LWD. Dissolved oxygen (DO) was severely reduced throughout the bottom waters of the lake (DO was <2 mg/l) creating anoxic conditions, compromising the cold water fishery habitat, and resulting in the death of some lake trout (Togue). Following the fish kill, Maine's Department of Inland Fisheries and Wildlife conducted a survey where they netted and identified the fish and found that some of the lake trout survived.

While the causal relationships in water quality are not completely known, it is clear that if the lake's water quality continues to degrade it would put AWD/LWD at risk for violation of the compliance parameters for filtration avoidance waiver, and thus, could require the need for more advanced water treatment facilities. Increased nutrient levels in the lake, especially phosphorus, led to an increase in late summer/early fall algal blooms that caused the increase in turbidity, which is one of the compliance parameters for maintaining the filtration waiver. This compliance criterion requires that turbidity at the point of withdrawal at Lake Auburn must not exceed 5 NTU for more than two events per year and not more than 5 events in ten years. The primary agency may waive an exceedance (a 'turbidity event') if they determine that the event or the circumstances leading to the event exceeding 5 NTU are unusual and unpredictable. If the turbidity levels are exceeded and not waived, filtration would be required to be added to the existing water treatment plant.

An additional concern related to an increase in algal activity is the potential for increased taste and odor issues throughout the distribution system and threats to Lake Auburn's cold water fishery. The dominant algae in late summer/fall blooms in 2011 and 2012 have the potential to generate unpleasant taste and odor; although not a health issue or specifically a regulatory compliance issue, it is an aesthetic issue that impacts water customers. When these algae die, decomposition consumes dissolved oxygen in the lower waters of the lake, jeopardizing the health of the fishery there.

The Lake Auburn Watershed Protection Commission contracted with CDM Smith Inc. to investigate the causes of recent degradation in water quality and to recommend short-term and long-term management options to mitigate the adverse impacts of excess phosphorus on the water quality of Lake Auburn. The study team includes Comprehensive Environmental, Inc. (CEI) to address on watershed changes and management actions, and Dr. Ken Wagner of Water Resource Services Inc. to provide advice on lake water quality and in-lake management options. Phase I of this study (CDM Smith, 2013) involved reviewing available water quality sampling data, analyzing the data to understand the causes of poor water quality in Lake Auburn, and recommending short-term lake and watershed management options; the watershed sub-report (CEI, 2013a) also recommended management options that would be implemented over several years.

Since the March 2013 Phase I report, AWD/LWD have implemented many of the recommendations in the report, including:

- **Revised monitoring program for both in-lake and watershed sampling –** The revised monitoring program was designed to collect data to enhance our understanding of the drivers behind the decline in water quality. Description and analysis of the data collected under the revised data collection program is in Section 2 of this report.

- **Obtained an algicide application permit from Maine DEP** – AWD/LWD received a permit to apply a copper sulfate algicide for use as a contingency stopgap measure should water quality conditions that could jeopardize the filtration waiver reoccur in 2013. An algicide application was not required in 2013. The permit is valid for five years after the effective date and allows for one application or a series of applications not to exceed six months in length; and two applications or series of applications over the 5-year term of the permit.

- **Implementation of several non-structural BMPs --** AWD/LWD hired an education and outreach manager to both perform outreach and education to watershed residents and oversee other parts of the watershed management program. Among the non-structural activities that have been started or completed are:

    - A school-based education program has been developed and delivered at schools in Turner, Hebron, Lewiston, and Auburn. The program consists of a series of hands-on lessons designed to provide an overview of water resources, management, and protection. Topics include the water cycle, properties of water, surface and groundwater dynamics, stormwater issues, water wildlife, FishKids (raising brook trout in the classroom) and other concepts as requested by teachers or community groups.

    - The Lake Auburn Watershed Protection Commission (LAWPC) is now Maine's sponsor for Project WET (Water Education for Teachers) - an award winning water-focused curriculum guide. Using Project WET, each year more than 100 Maine educators are trained to deliver hands-on water lessons. In addition, the LAWPC education and outreach manager collaborates with other state partners for educational programs delivered in the region, such as the Southern Maine Children's Water Festival.

    - LAWPC sponsored homeowner and watershed resident assistance programs include Septic Sense, providing septic system education and pump out assistance; Farm Friends to enable manure management plan development and support; and Rooting for Shorelines which provides funding for buffer planting projects. A grant from the Maine Drinking Water Program was obtained to provide these protection programs.

  - A review of the LAWPC properties management strategy is underway. The timber harvest plan was updated in 2013 and will be modified to reflect goals under development. Representatives from the LAWPC are partnering with the Lewiston Auburn Community Forest Board, Androscoggin Land Trust, Androscoggin Soil and Water Conservation District, Lake Auburn Community Center, and others to formulate goals for the lands that will protect water quality while allowing the possibility of increased public access and recreational opportunities.

- **Construction and implementation of several structural BMPs** – AWD/LWD has built or is planning to build several structural BMPs in areas recommended by CEI (Richard, 2014).
  - Worked with the Department of Transportation to remediate a site on Route 4 that was adding sediment to the lake.
  - Addressed two large sites on Whitman Spring Road and one on West Auburn Road. Both of these sites had severe erosion problems and were discharging to the lake.
  - Provided guidance to the City of Auburn for an upcoming project on Holbrook Road where flow will be diverted to a treatment area.

This report reviews and analyzes water quality data collected throughout 2013 and make long-term recommendations for both holistic watershed management and in-lake remediation activities. Section 2 describes and analyzes data collected throughout 2013. Section 3 provides information on in-lake management options applicable to Lake Auburn, and recommends holistic watershed and lake management measures to ensure AWD/LWD's compliance with the terms of its filtration avoidance waiver and to maintain Lake Auburn's historically high water quality.

Section 2

Summary of 2013 Data

The recent changes to water quality in Lake Auburn prompted the design of a revised water quality monitoring program for the 2013 sampling period. Goals of the revised monitoring program are to:

1. Collect data sufficient to improve the understanding of key drivers of water quality in the lake;
2. Build on existing sampling programs to maximize the usefulness of existing data; and
3. Design the sample collection to obtain sufficiently frequent data with both time and in vertical depth.

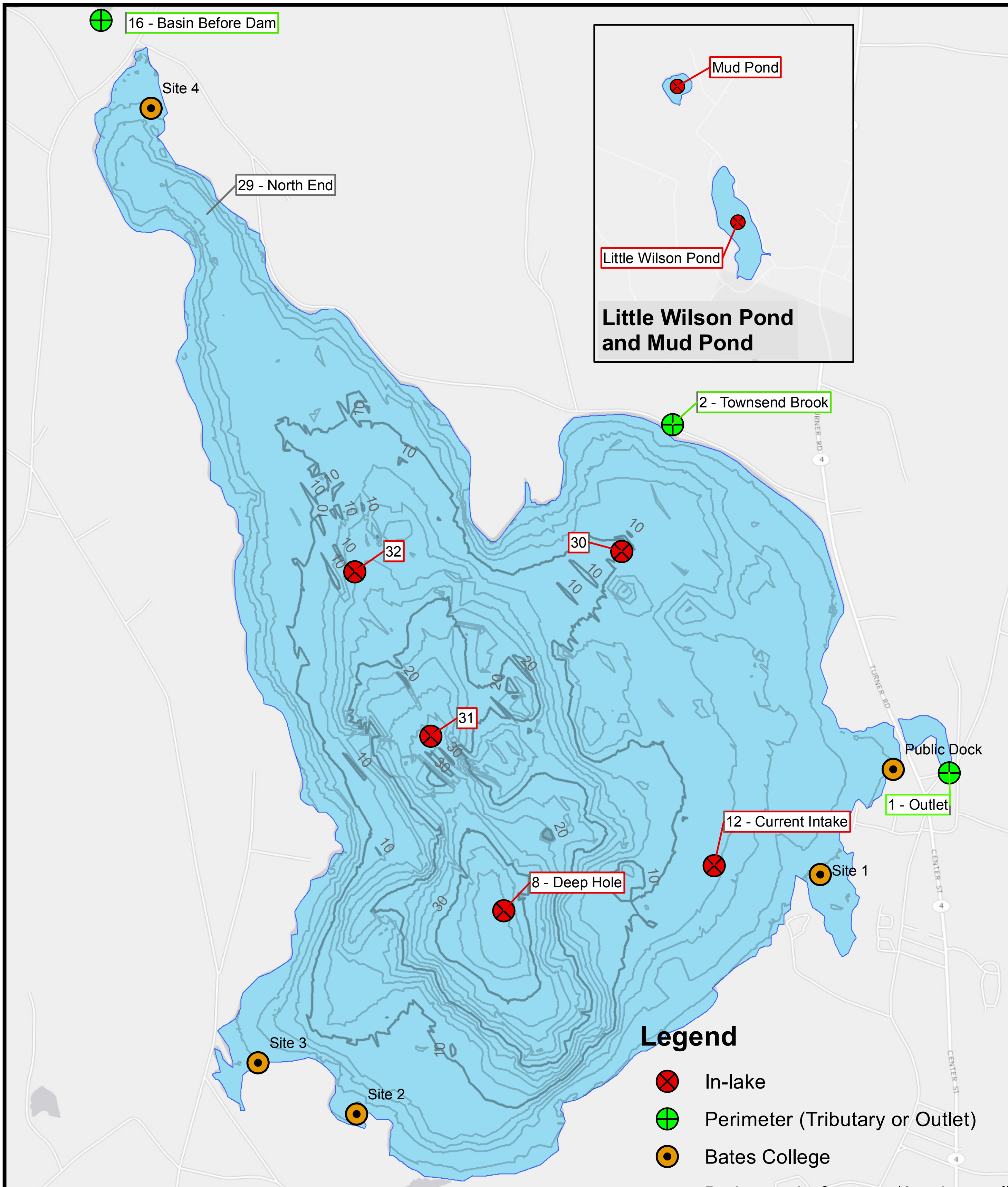
2.1 Brief Description of 2013 Monitoring Program

The 2013 monitoring program was an extensive effort. AWD/LWD collected over 6,000 data points throughout the year providing a comprehensive picture of Lake Auburn's water quality. Sampling locations were revised from AWD/LWD's historical sampling sites and include both in-lake and tributary locations. These sampling locations represent an overall reduction in the number of locations but an increase in the frequency of sampling. The 2013 monitoring program consisted of five open water lake sampling locations, five shoreline sampling stations to continue the Bates College *Gloeotrichia* monitoring, two principal tributary sampling locations (the Basin before the North Auburn Dam and Townsend Brook), the lake's outlet, and two sites in the contributing watershed, Mud Pond and Little Wilson Pond. Sampling locations are shown in Figure 2-1 and are described below.

Open Water Lake Sampling

1. The current intake (#12)
2. The deep hole (#8)
3. In the north toward the Basin inlet (#32)
4. In the northeastern lobe of the reservoir (#30)
5. In between the deep hole and the Basin where the water depth reaches 30 m (#31)

Vertical profile data for the lake's physical structure were collected at each station, as was Secchi depth. Analysis for nutrient parameters was collected from discrete samples, most often taken at near-surface and near-bottom locations. Profiles of nutrient parameters were taken on three dates for a total of six profiles: four profiles were taken at the deep hole plus sites 30, 31, and 32 on November 13, and two additional profiles were taken at the deep hole sampling location on March 7 and November 4. Epilimnetic cores (depth-integrated samples) of the water column (surface to 1 m below the thermocline) were taken approximately biweekly and analyzed for nutrients and chlorophyll; algae



Bathymetry data: University of Maine, Farmington
Sample Locations: G. Jalbert and M.J. Dillingham, AWD

Figure 2-1: 2013 Sampling Locations

N

0 250 500 1,000 Meters

Contour Interval: 2 m

in these samples were also identified and enumerated. A smaller number of cores were collected in a 3-m vertical region around the thermocline to identify algae there.

Gloeotrichia Sampling Sites

Bates College again sampled for *Gloeotrichia echinulata* at five sites around the perimeter of the lake. In 2013, Bates added samples at the five open water lake sites. The Bates College *Gloeotrichia* sites are also shown on the sampling map in Figure 2-1.

Throughout 2013 the perimeter locations were sampled weekly for total phosphorus, *Gloeotrichia,* and chlorophyll and the open water lake locations were sampled weekly for *Gloeotrichia*.

Outlet/Tributary Sampling Sites

1. Outlet (#1); only sampled when water is being released from the lake

2. Townsend Brook (#2)

3. The Basin before the North Auburn Dam (#16)

Dry weather sampling at these locations included weekly sampling for physical parameters, nutrients and TSS. Wet weather sampling was also conducted on 25 dates at the tributary stations.

In addition to collecting discrete water samples, AWD/LWD installed flow metering equipment at Townsend Brook and the North Auburn Dam to allow estimates of constituent loads into the lake to be developed.

This section describes data collected at each site and provides analysis and comparison of the data to the historic monitoring data discussed in Phase 1 of this study.

2.2 Weather and Climatological Data

Several sources of climatological data were used in this analysis. The principal data source of meteorological data is the National Climatic Data Center (NCDC), part of the National Oceanic and Atmospheric Administration (NOAA). Four National Weather Service Cooperative Observer Program (COOP) stations are active within a 15-mile radius of Auburn, Maine; they are located at Durham, Poland, Gray and Turner. Another COOP station was formerly located in Lewiston, but the completeness of its dataset dropped beginning in 2002, and recordkeeping ceased in 2010. Data from each station was compared in the Phase 1 report (CDM Smith, 2013) and the stations were found to be sufficiently similar such that Poland was selected for use in this analysis because it is the closest station to Lake Auburn. Total annual precipitation in 2013 was 42.33 inches which is below the average between 1998 and 2013 of 50.8 inches and below the 2011 and 2012 total precipitation values. Histogram plots comparing the annual precipitation for 2001 through 2013 and monthly precipitation for 2011 through 2013 against the long term average are shown in Figures 2-2 and 2-3.

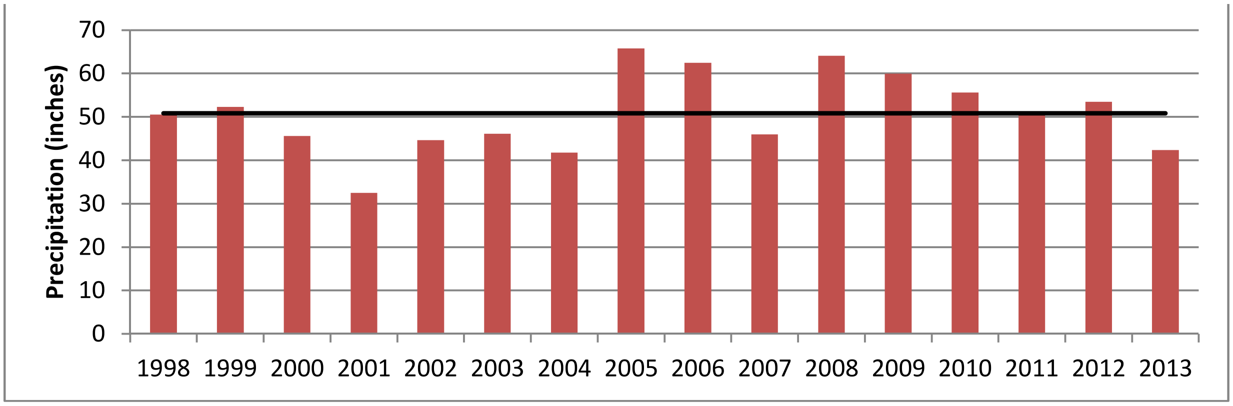


Figure 2-2
Annual Precipitation Totals Compared to the 1998-2013 Average Precipitation at the NWS COOP Station 176856 in Poland, ME

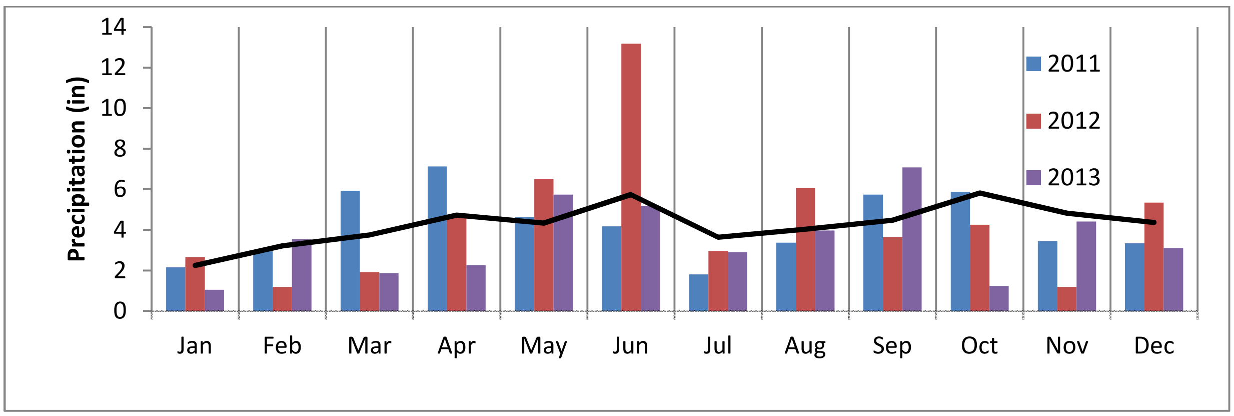


Figure 2-3
Monthly Precipitation Totals for 2011-2013 Compared with the 1998-2013 Average Monthly Precipitation at the NWS COOP Station 176856 in Poland, ME

These plots show that 2013 was a below-average year in terms of total precipitation. January through April were drier than average, as were July, October, and December. June, August, and November were just about average, and May and September were slightly above average.

In addition to official records from the NCDC, AWD/LWD maintains a weather station at the water treatment plant that measures and reports hourly air temperature, wind speed, and wind direction. Hourly precipitation is measured at the Main Street pump station in Lewiston. While the Lewiston precipitation record has not recently tracked well with surrounding gages, the Lewiston gage measured 41.84 inches of precipitation in 2013. It should be noted, however, that the engineer in Lewiston responsible for maintaining the station states that the heater does not work so the snow water equivalent is estimated. A comparison of 2013 monthly precipitation totals between the Poland COOP and the Lewiston precipitation gage is shown in Figure 2-4. This figure shows that the two gages agree well with each other in most months, suggesting that the hourly Lewiston gage could be used to estimate storm intensity from the daily Poland COOP gage. For consistency this analysis continues to use the Poland COOP gage.

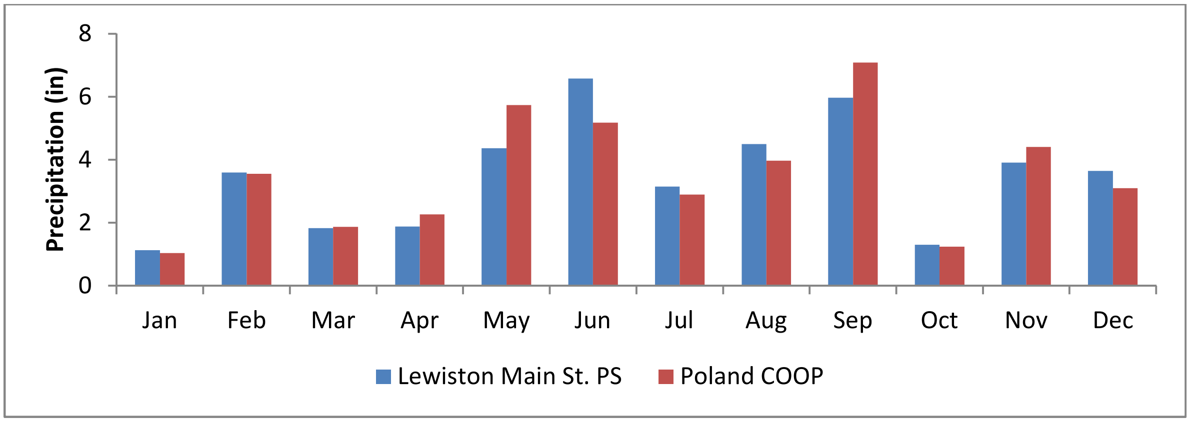


Figure 2-4
Comparison of 2013 Monthly Precipitation at the NWS Poland COOP Gage and the Lewiston Hourly Precipitation Gage at the Main Street Pump Station

In addition, the wind speed and direction measurements at the water treatment plant in Auburn were analyzed to determine the primary direction of wind. This information is commonly presented in a wind rose, a plot that shows the frequency and wind velocity in each cardinal direction. A wind rose depicting wind data collected in 2013 superimposed on Lake Auburn is shown in Figure 2-5.

In this figure the wind rose elements show the direction from which wind comes. The size of each slice represents the frequency of wind occurring in each direction. This shows that the principal wind directions are along the long axis of the lake, blowing towards the intake, and along the short axis of the lake, blowing towards the outlet. The direction and relative strength of the wind is important for understanding the hydrodynamics and potential for mixing within the lake.

2.3 Physical Lake Parameters

Physical lake parameters describe the overall structure of the water body, such as the strength and duration of stratification, the amount of dissolved oxygen, and the water clarity. This section describes measurements taken in 2013 and compares them to values collected prior to this year's sampling program.

Vertical data for physical lake parameters was collected using two different methodologies: discrete profiles and a temperature-dissolved oxygen meter array. As in the past, discrete profiles were taken weekly at 1-meter intervals at the deep hole site (#8). In 2013, four additional profile sites were added at sites 12, 30, 31, and 32 (see Figure 2-1). Each of these profiles included conductivity, pH, temperature, and dissolved oxygen measurements. Secchi depth was also collected at each open water station.

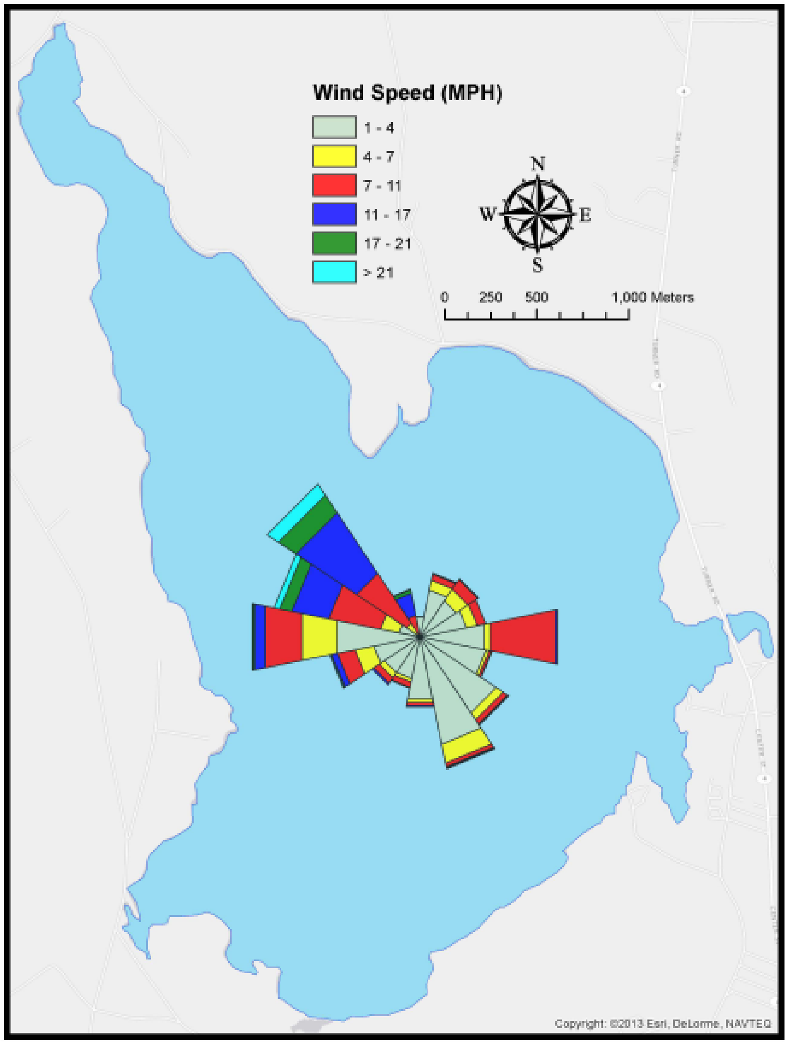


Figure 2-5
Wind Rose Computed from 2013 Wind Data Collected at the Auburn Water Treatment Plant

2.3.1 Discrete Profiles

Examination of the discrete profiles shows that the lake structure is generally uniform, meaning that the depth of the thermocline and the relative extent of anoxic conditions is similar across all five sites. This is best observed by comparing isopleths of temperature and dissolved oxygen at the three deepest sites (the Deep Hole, site 31, and site 32), presented in Figures 2-6 through 2-11. The vertical profile data for temperature are typical of a stratified northern temperate lake. The vertical profile data for oxygen are further analyzed for the onset and extent of anoxia, which is a driver for phosphorus release from sediment.

In addition to the general consistency among the five sites, the dissolved oxygen data show how the temporal and spatial extent of anoxia differs from prior years. Beginning in late July/early August 2013, a region of low oxygen occurs at a mid-depth below the thermocline at all locations in the lake; this region continues to lose oxygen until anoxic (<2 mg/l) values occur in early September. This is seen in the dissolved oxygen isopleth figures as an overhanging lens of low dissolved oxygen water and can be attributed to decay of organic matter at the thermocline. This lens, called a dissolved oxygen minima, results in a very sharp vertical gradient of dissolved oxygen at the top of the thermocline. In the hypolimnion, contemporary dissolved oxygen values are 1 to 2 mg/l higher. Oxygen in the thermocline drops below 2 mg/l on September 9, several weeks before the dissolved oxygen in the hypolimnion uniformly drops below 2 mg/l at the end of September.

This set up of low oxygen values in Lake Auburn is not unique in pattern, as vertically discontinuous regions of lower dissolved oxygen levels have happened in the past (though not in 2011 or 2012). 2013, however, is unique in that it is the first time on record that the minima in the thermocline reached anoxic levels.

Section 2 of the Phase 1 report (CDM Smith, 2013) included a description of Nürnberg's (1995) anoxic factor (AF), which is a method for understanding the potential impact of varying anoxic conditions on water quality. The anoxic factor is expressed in units of days per year and represents the total number of days that sediment area equal to the surface area of the lake is exposed to low oxygen water. The threshold for anoxia at 2 mg/l dissolved oxygen was used in the calculations below. In general, a healthy, oligotrophic lake will have an AF of less than 10 days per year. If the AF exceeds 20, lake health is in danger, but can be recovered with appropriate management. AF over 100 indicates a hypereutrophic lake with widespread anoxia throughout the bottom waters. The anoxic factor is presented in Table 2-1 for 2001 through 2013.

The trend in anoxic factor alone shows a dramatic and significant increase in the potential for internal nutrient load in 2011. In 2011 and 2012, the potential for anoxic release of phosphorus increased by a factor of 100 when compared with most other years with a nonzero anoxic factor. The 2013 anoxic factor is significantly less than 2011 and 2012, mainly due to the delayed onset of anoxia during 2013. While just on the border of a healthy oligotrophic lake, the value is still significant, particularly as it is about double the highest value calculated from data prior to 2011.

In general, improved oxygen conditions were seen in Lake Auburn in 2013. The duration that bottom sediments were exposed to anoxic water was about half of the duration in 2011 and 2012. The onset date for anoxia in the reservoir was a couple of weeks later for any point in the lake, and about 5 weeks later in the hypolimnion than in the prior two years. Unlike 2012, when the entire hypolimnion was devoid of oxygen from early September to turnover in November, there was a region between about 15 and 20 m depth with oxygen concentrations sufficient for coldwater fish, thus avoiding a repeat of the fish kill in 2012.

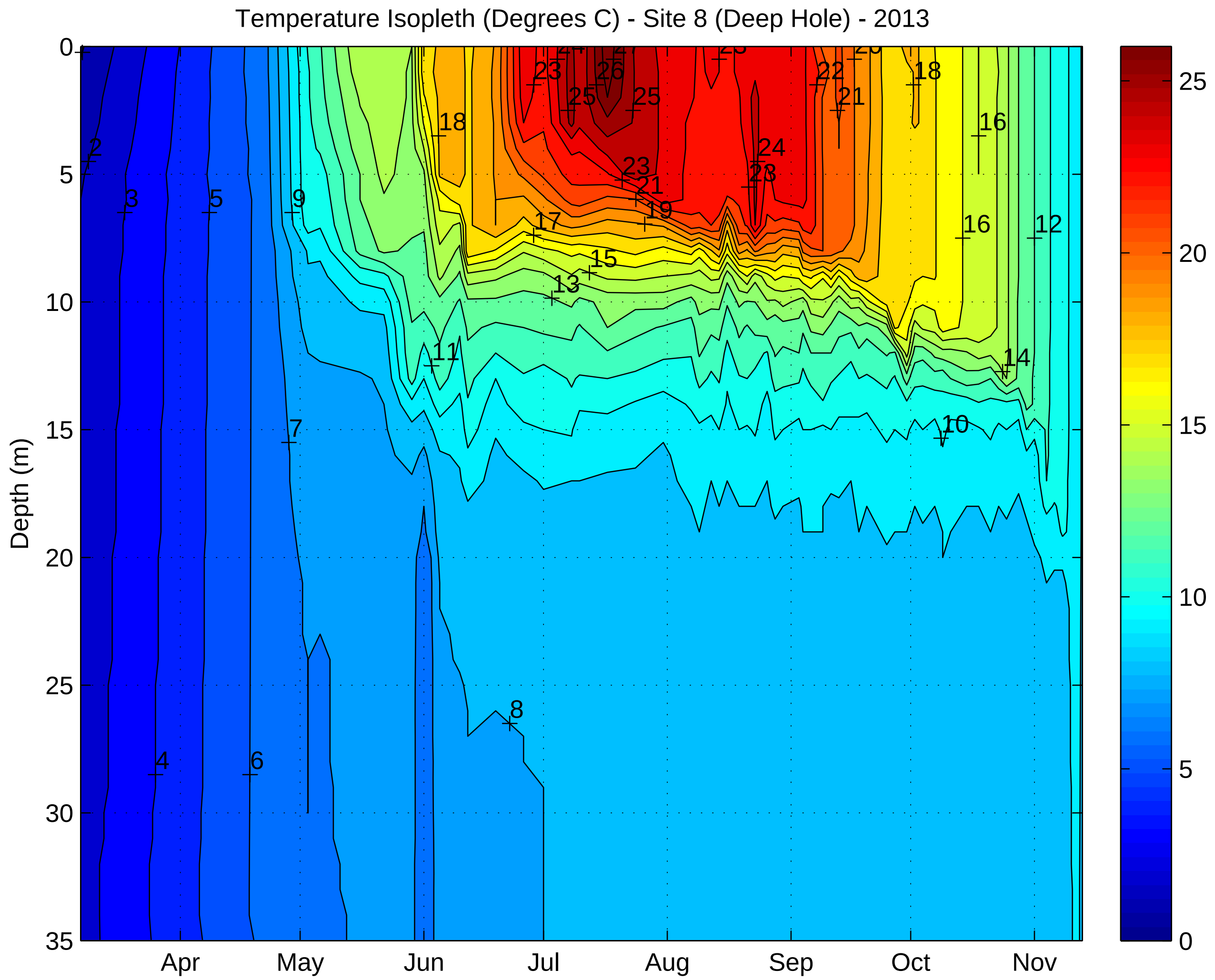


Figure 2-6
2013 Site 8 (Deep Hole) Temperature Isopleth

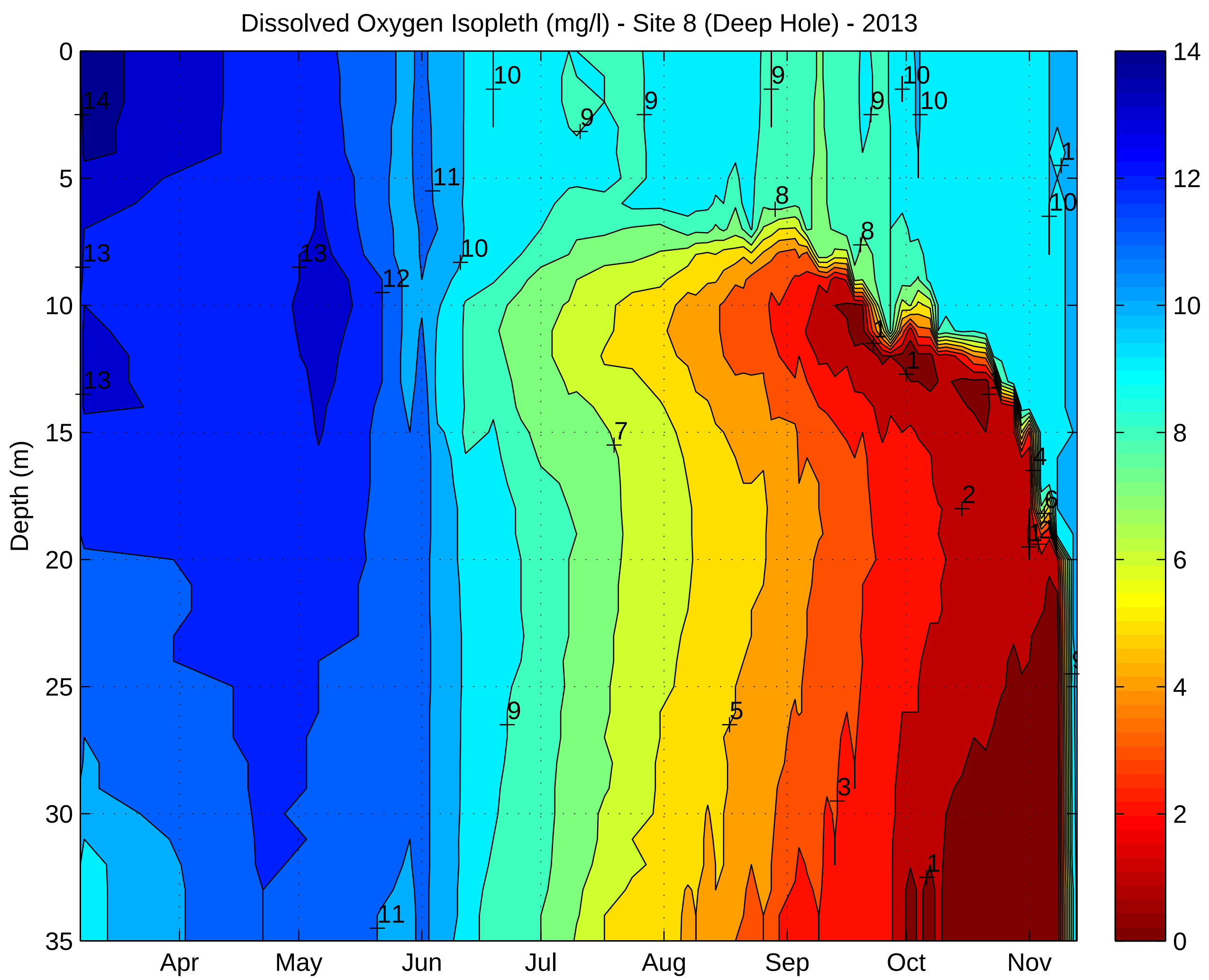


Figure 2-7
2013 Site 8 (Deep Hole) Dissolved Oxygen Isopleth

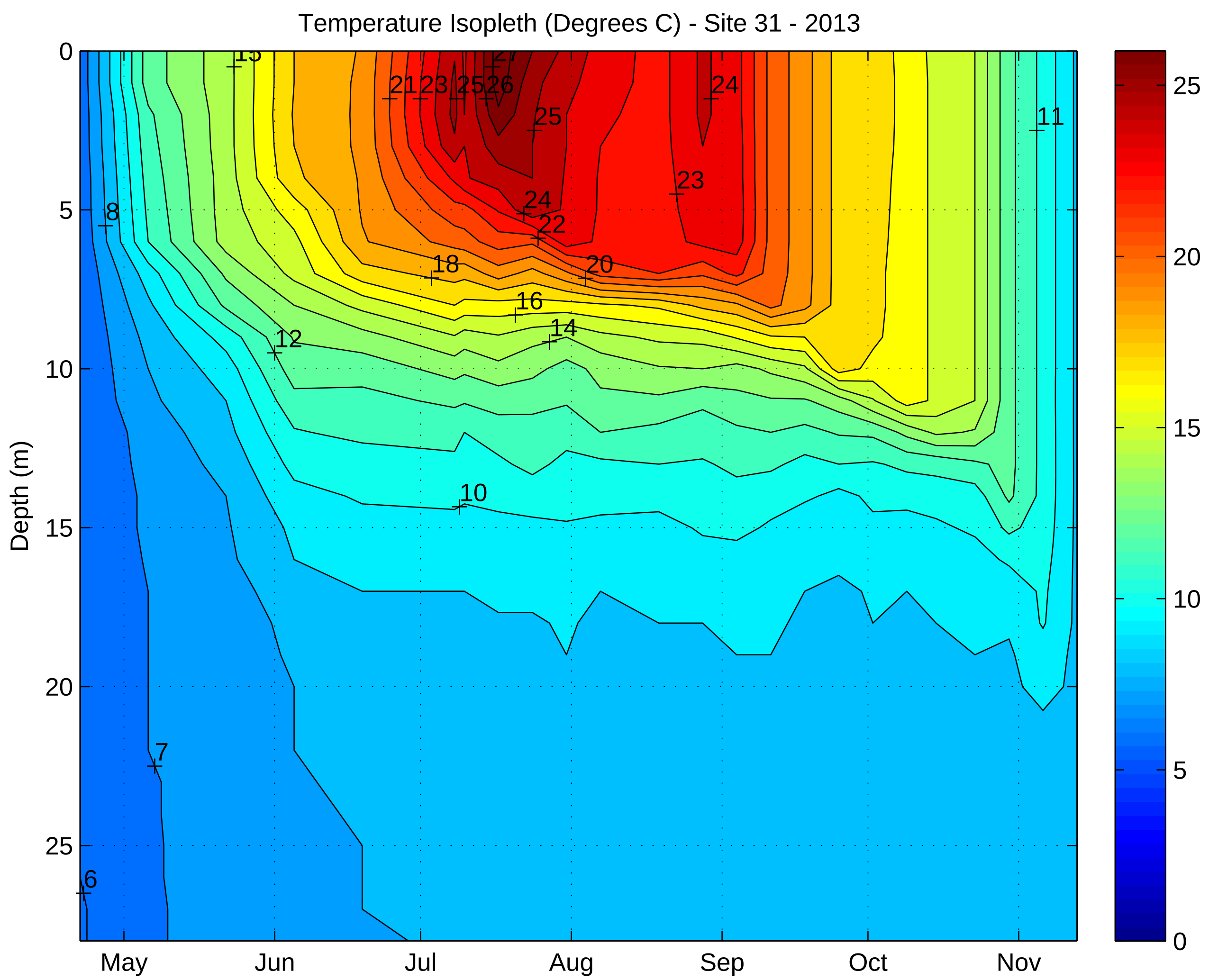


Figure 2-8
2013 Site 31 Temperature Isopleth

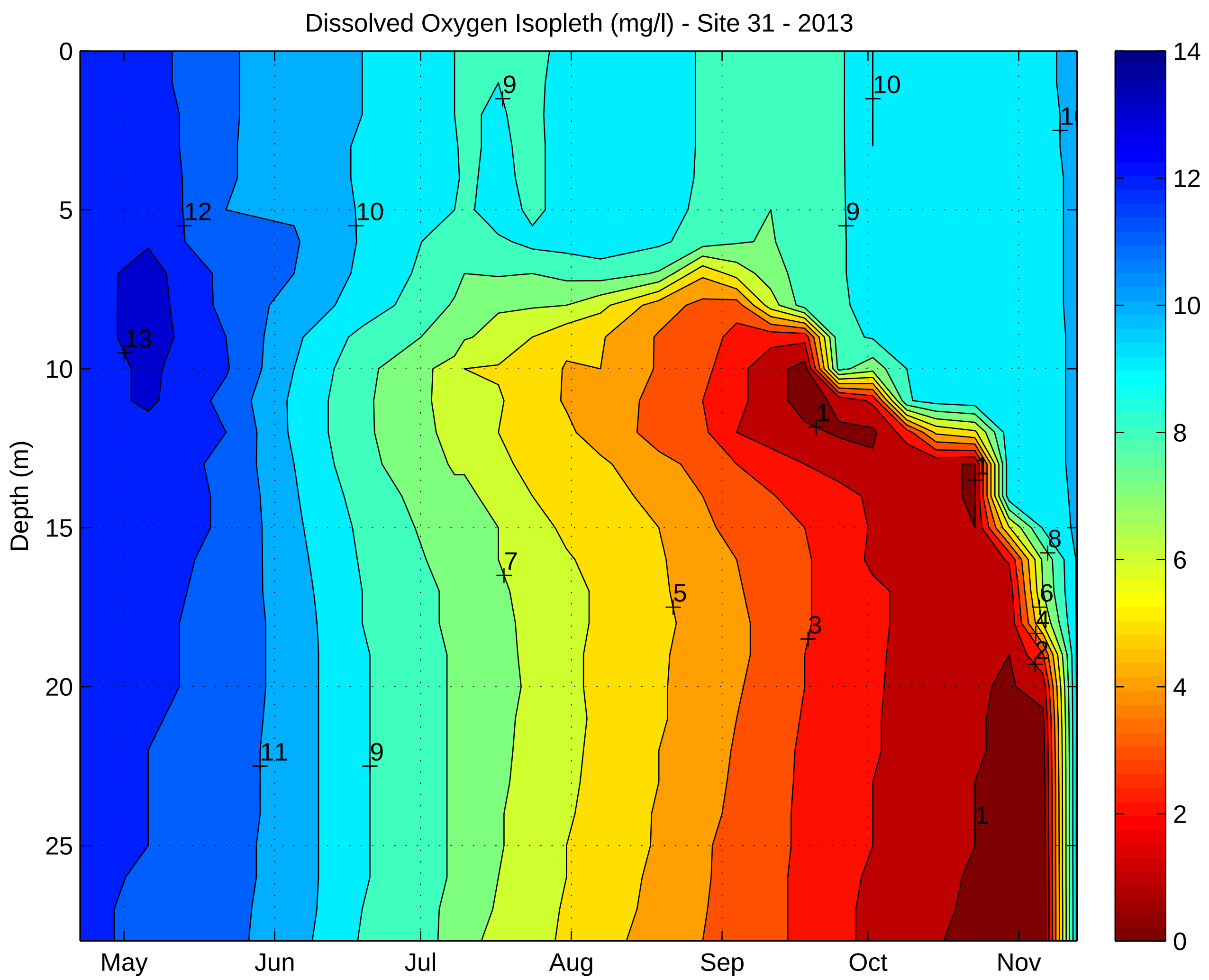


Figure 2-9
2013 Site 31 Dissolved Oyxgen Isopleth

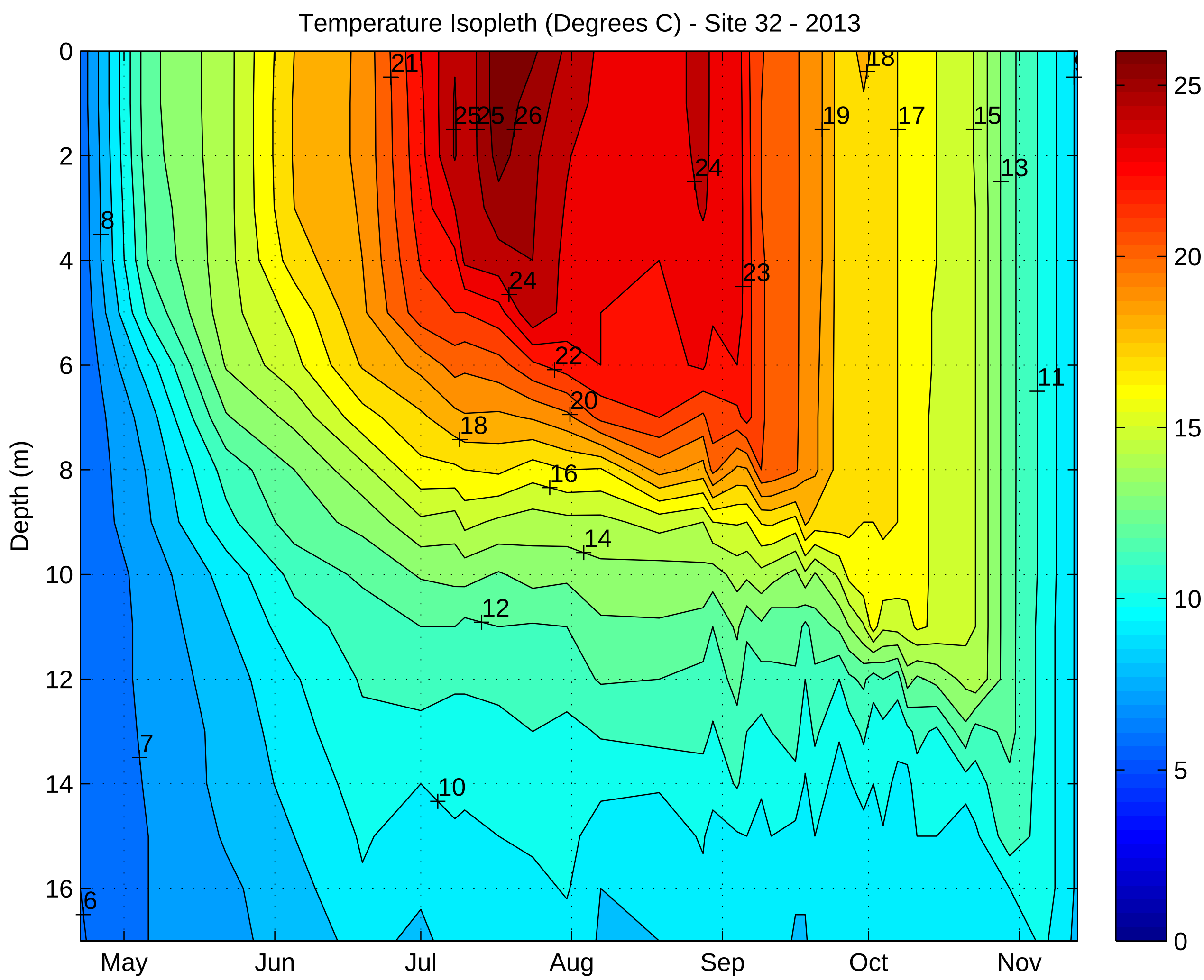


Figure 2-10
2013 Site 32 Temperature Isopleth

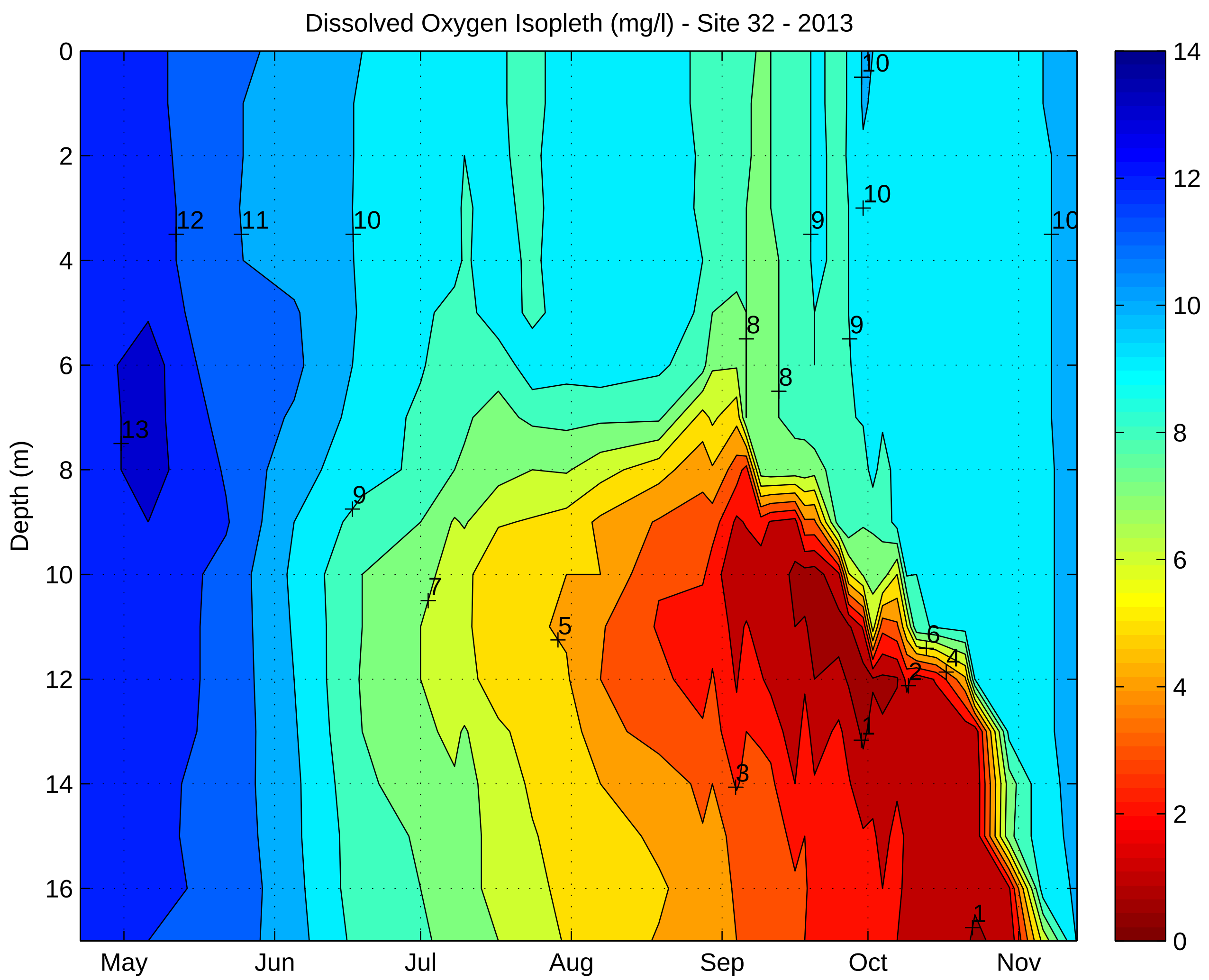


Figure 2-11
2013 Site 32 Dissolved Oxygen Isopleth

Table 2-1
2001 through 2013 Anoxic Factor (Nürnberg, 1995) for Lake Auburn

Year	Anoxic Factor (days/year)
2001	0
2002	5.7
2003	0
2004	0
2005	0
2006	0.26
2007	0
2008	0
2009	0
2010	0.28
2011	18.1
2012	18.9
2013	9.9

2.3.2 Bates College Temperature-Oxygen Meter Array

The Bates College meter array was located at the deep hole site and was active from July 25 through November 17, 2013. Temperature was measured via a thermistor chain with thermistors placed at 0.5, 1, 2, 4, 6, 8, 10, 12, 14.5, 16, 22, 30, and 32 meters; dissolved oxygen was measured via probes located at 1, 14.5, and 32 meters. Both the thermistors and the oxygen probes record at 10-minute intervals.

Figure 2-12 shows isotherms from array measurements over the entire period of record. These isotherms show a strong and typical temperature gradient at the start of the record in late July, with the upper waters above 20°C and the bottom water below 10°C. Throughout August the stratification is relatively stable. In mid to late September the upper waters begin to cool below 20°C while the bottom water remains around 10°C. In late September the thermocline suddenly drops coinciding with cooler weather. Through October the thermocline gradually deepens and becomes weaker, with the upper waters becoming steadily cooler coinciding with the cooler air temperatures. Turnover occurs on November 12, where the entire water column is a well mixed 10°C. This pattern is as expected, and is similar to the evolution of the lake's structure in previous years.

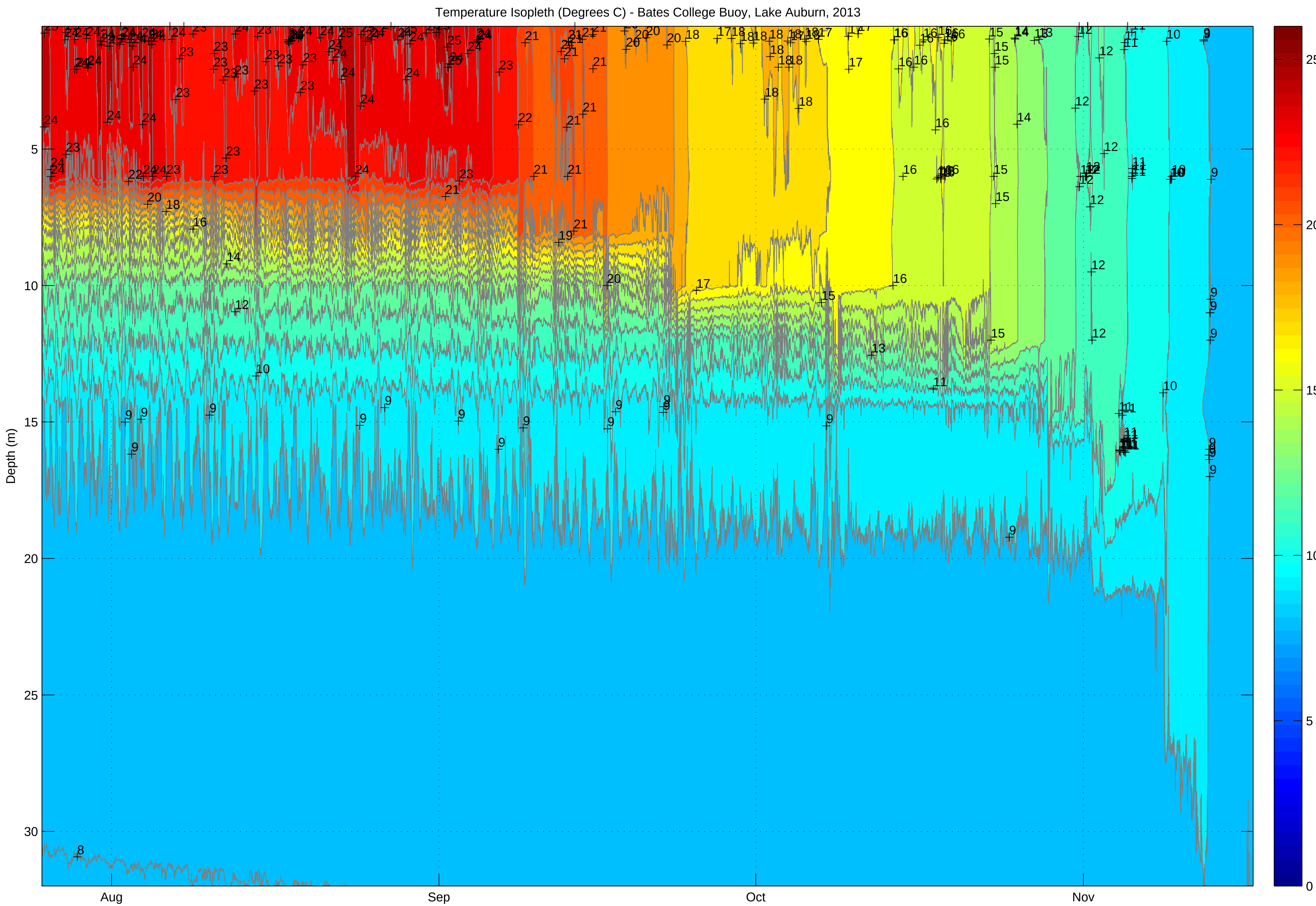


Figure 2-12
2013 Deep Hole Isotherm
Bates College Buoy

The principal advantage of the fine temporal resolution of the thermistor chain is the ability to observe changes in the lake structure on a fine time scale. This allows us to evaluate the effect of climatological forcing on the thermocline on a sub-daily scale. In the surface layer, the record permits observation of cyclical daily heating and cooling. Observations throughout the thermocline indicate high frequency variations in temperature and occasional intrusions of warmer water from the epilimnion that are indicative of mixing events forced by climatic factors. The more significant intrusions appear to have occurred every week or so, and notably the intrusions appear largely confined to the thermocline and, at most, are significantly dampened in the hypolimnion. The 2013 record does not include any whole water column mixing events similar to the one that is expected to have happened in 2011 with the passage of Hurricane Irene.

Examining the High Frequency Excursions

In examining the high frequency temperature variations, the first objective was to understand whether the physical layout of the meter array could contribute to the observed temperature variation. Ewing (2014) provided the following description. The meter array is attached to two floating mooring buoys off to the side of it. It also weighted on the bottom to keep it upright and pretty stable in the water. All of the probes hang vertically, and the whole unit will bob up and down if it is really windy. The lines with the probes do have very heavy chains at the bottom of them, so they should stay basically vertical, but it is possible for them to move some vertically in the water column. Therefore, it is possible that the entire thermistor chain moving up and down with surface wave action could cause some of the oscillation observed in the record.

Sustained high wind along either axis will ultimately cause the water surface to "set up," where the downwind side is pushed up and the upwind side pushed down by the wind stress on the surface. The potential energy from the elevated water surface elevation causes large-scale basin wide circulation that can transport pollutant mass within the epilimnion and hypolimnion, and can induce surface waves that propagate along the air-water interface and internal waves that propagate along the interfaces between the warm epilimnion and the cold hypolimnion. Surface and internal waves are theoretically possible in any stratified flow, but the strength and period of oscillation depend on the basin length and depth (Fischer *et al.,* 1979). Both types of waves must be considered when examining the apparent high frequency oscillation of the isotherms depicted in Figure 2-12.

The overall physical characteristics of the basin along with the measured predominant wind direction allow for computation of the expected theoretical period for surface and internal wave oscillation. For both surface and internal waves, the longest wavelength is on the order of the entire basin size; this wave is called a seiche. In general, the period and amplitude of internal waves are much greater than the period and amplitude of surface waves (Wetzel, 2001).

The period of the surface seiche, which is induced by the set up of the water surface caused by wind stress, can be approximated by the equation

$$t = \frac{2l}{\sqrt{g\bar{z}}}$$

where t is the surface seiche period in seconds, l is the basin length in meters, g is acceleration due to gravity (9.81 m s^{-2}), and $\overline{z}$ the average depth in meters (Wetzel, 2001). Assuming an average depth of 15.8 meters and a length of 5,200 meters along the long axis, this yields a period of approximately 13 minutes.

Similarly, the period of the dominant mode of the internal seiche can be approximated by the equation

$$t = \frac{2l}{\sqrt{\frac{g(\rho_h - \rho_e)}{\frac{\rho_h}{z_h} + \frac{\rho_e}{z_e}}}}$$

where $t, l,$ and g are as defined above, ρ_h and ρ_e are the hypolimnetic and epilimnetic density in kg/m^3, respectively, and z_h and z_e are the hypolimnetic and epilimnetic depth, respectively (Wetzel, 2001). Assuming an epilimnetic temperature of 23°C, depth of 10 m, and density of 997.7735 kg/m^3, and a hypolimnetic temperature of 10°C, depth of 6 m, and density of 998.228 kg/m^3, yields a period of 22 hours. Increasing the depth of the hypolimnion causes the period to decrease; a hypolimnetic depth equal to 20 m yields a period of 17 hours.

This information can be used to provide context for interpreting the isotherms in Figure 2-12 and a more detailed view of isotherms between August 14 and September 3 shown in Figure 2-13.

The longest theoretical period of oscillation for surface waves is under 15 minutes and very close to the 10-minute sampling frequency of the thermistors. Therefore, it is very unlikely that surface waves would manifest as a significant variation in temperature at a point because most of this variation is likely dampened by the 10-minute sampling frequency (the 10-minute sampling frequency would randomly measure temperature at the different points in the 15-minutes oscillation (not consistently measuring, say, peak or mid-point temperatures). Therefore, the low-frequency oscillation centered around the thermocline seems much more likely a product of internal waves. This is supported by looking at the overall temperature fluctuations at 8, 10, and 12 meters (Figure 2-14).

Each of these plots of the temperature fluctuations at these depths show a large-scale periodic temperature fluctuation occurring on the scale of hours (as opposed to minutes), which is consistent with the order of magnitude estimate for the period of the internal seiche. Since the seiche period is the longest, and it is possible that shorter period waves exist in addition to the long standing wave caused by the seiche, it is reasonable to assume that waves of periods smaller than the actual seiche period will exist in the record. Other features of the temperature record from the thermistor string include observable diurnal heating/cooling in the surface water, wind-induced mixing, and erosion of the thermocline.

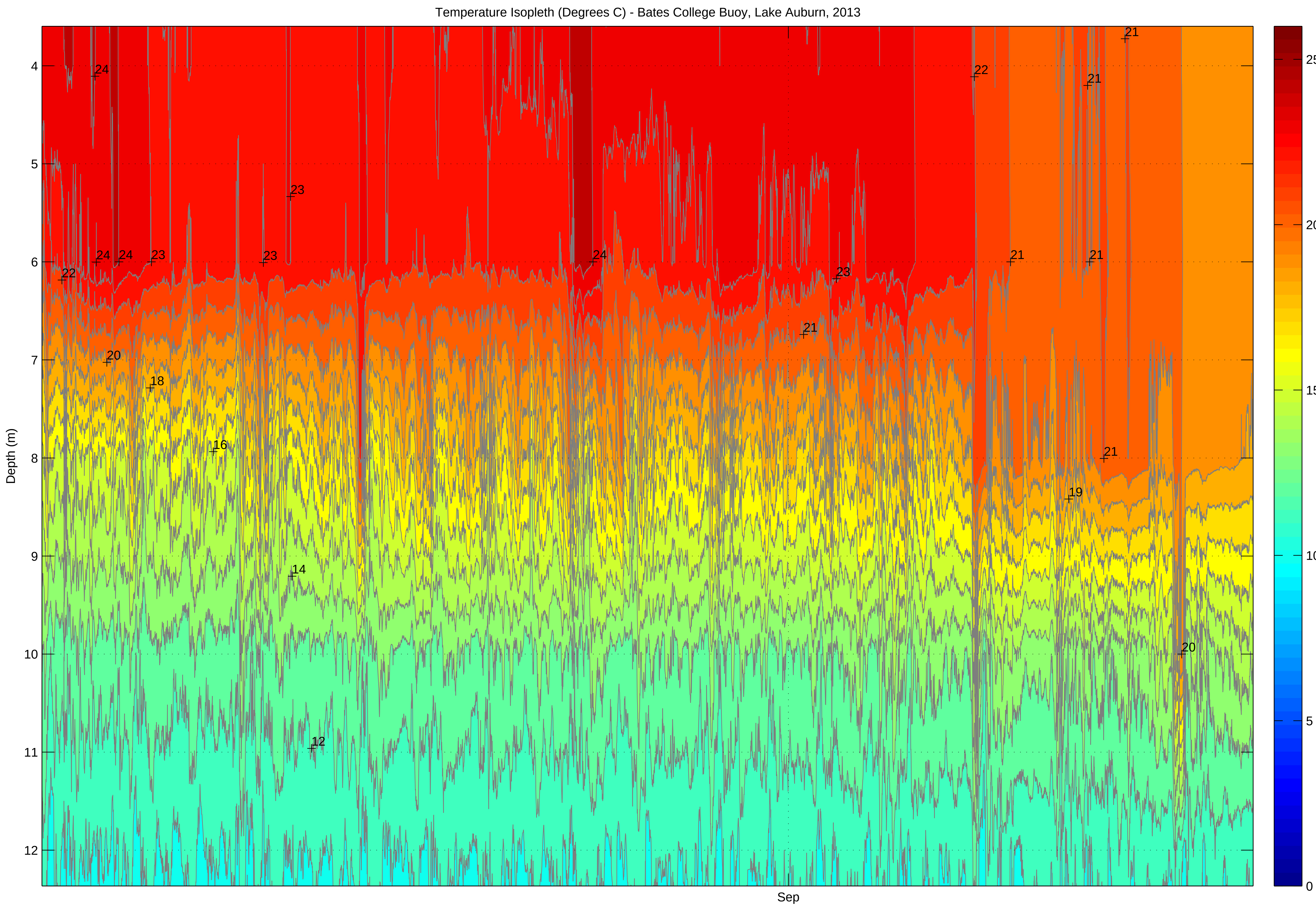


Figure 2-13
2013 Deep Hole Isotherm
Bates College Buoy
Metalimnion Zoom

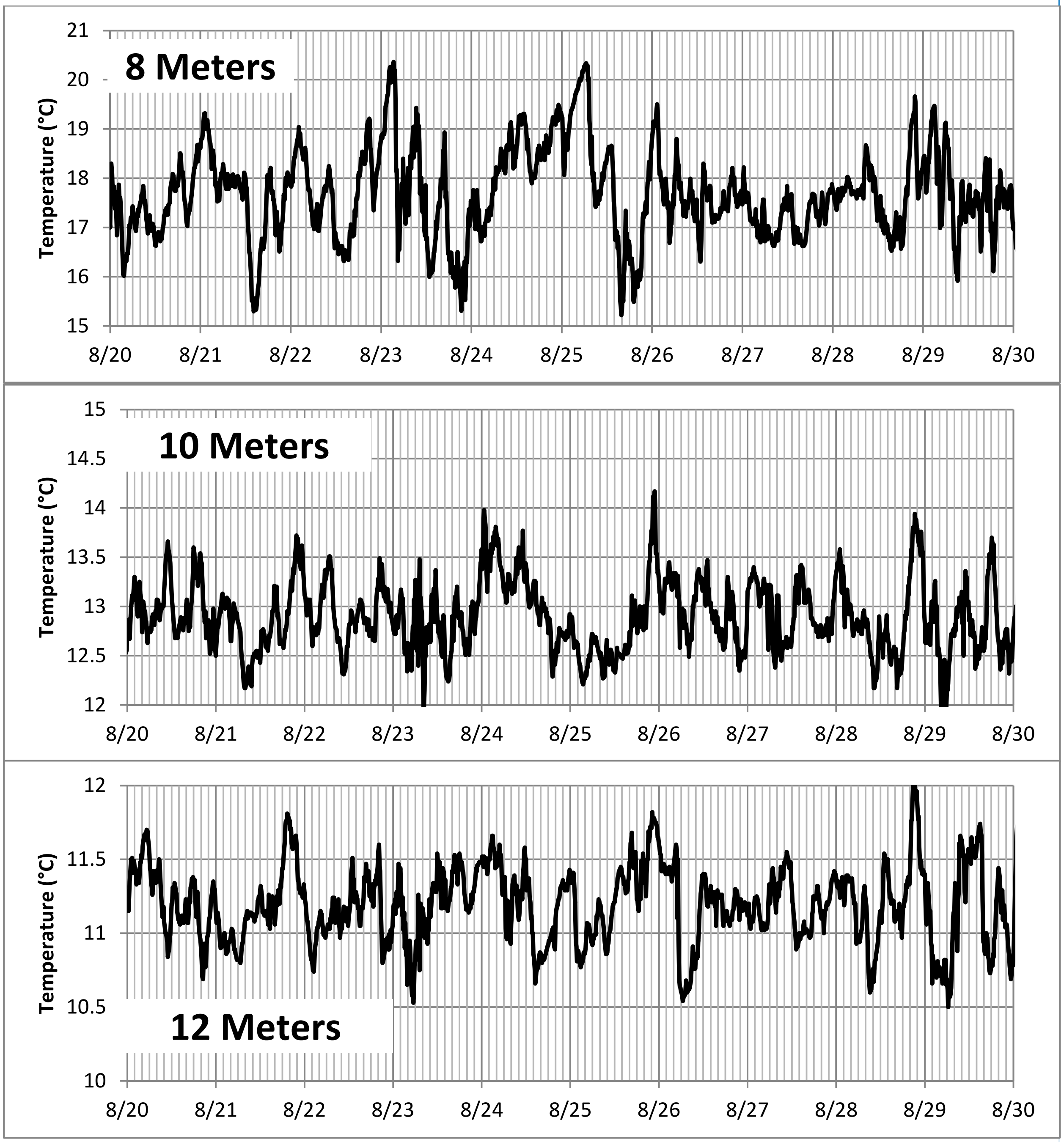


Figure 2-14
Temperature Fluctuations at 8 (top), 10 (middle) and 12 (bottom) Meter Depths in August

In addition to the sporadic and periodic temperature fluctuations about the thermocline, the temperature record in the upper water also shows significant variation that is likely attributable mixing events that were influenced by wind and changes in air temperature. One immediately apparent feature is numerous occurrences of warmer water that extend from the water surface down through the water column within the epilimnion. Examples of this occur around the first of August, the first of September, mid-September, and early October. These are diurnal and occur during periods of relatively low wind speed. For example, in early August (Figure 2-15) the measured wind speed is often below 5 mph, with only a few excursions to around 15 mph. August 5 is a windier day, with sustained wind above 10 mph. The diurnal pattern of heating and cooling stops briefly after this windy day, perhaps because the wind induced mixing introduced enough energy to mix the upper waters a little more uniformly. These diurnal fluctuations are likely caused by penetrative convection that induces an ephemeral and weak temperature gradient within the upper waters. This mild stratification is not very strong and is easily eroded by the mixing that occurs when the lake surface cools at night and by other circulation induced within the upper waters by the diurnal heating and cooling cycle (Fischer *et al.,* 1979).

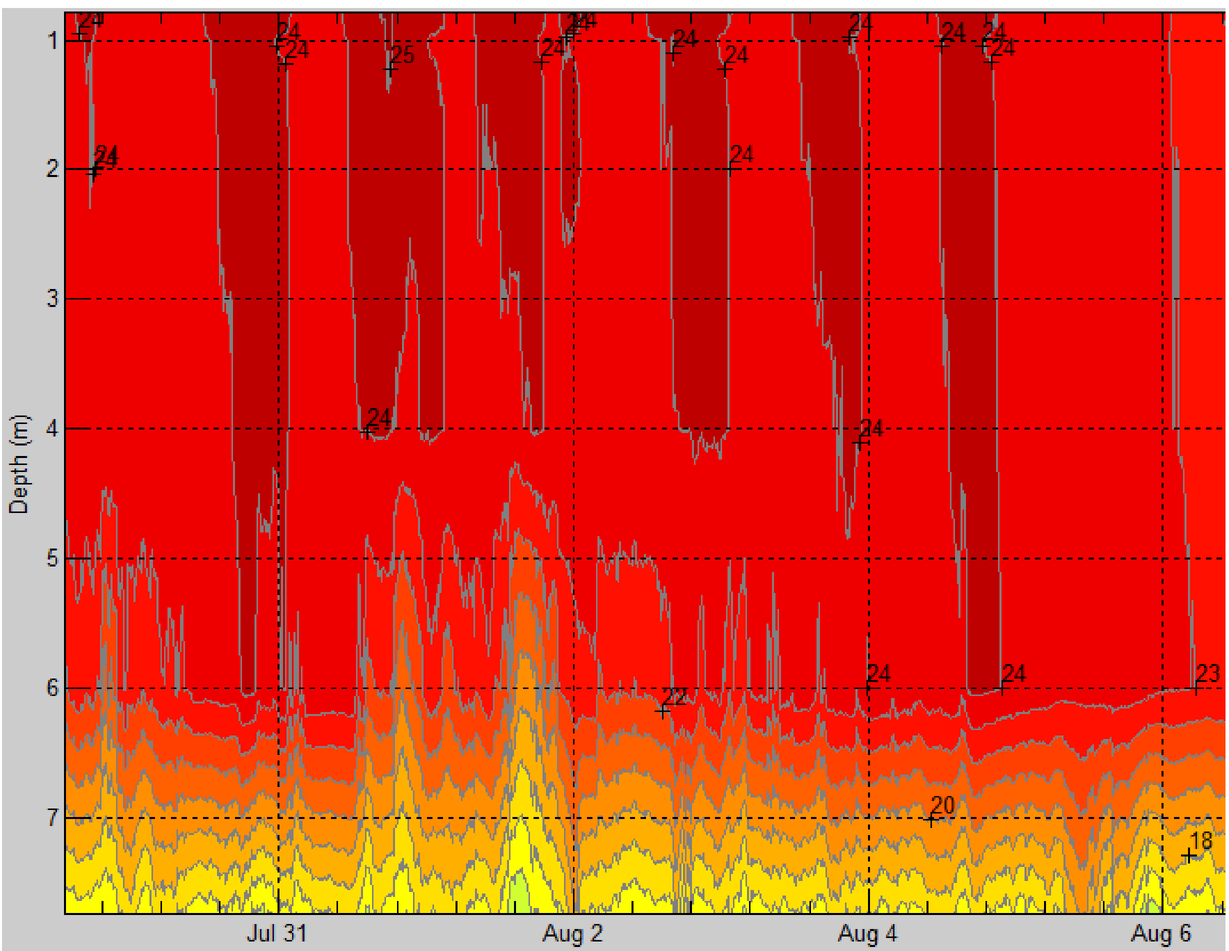


Figure 2-15
Example of Diurnal Heating and Cooling Caused by Penetrative Convection

Another interesting phenomenon observed in the data is periodic intrusions of warmer water into the cooler metalimnion. An example of this is on August 14, where warm water plunges 3 to 4 m into the metalimnion, as shown in Figure 2-16.

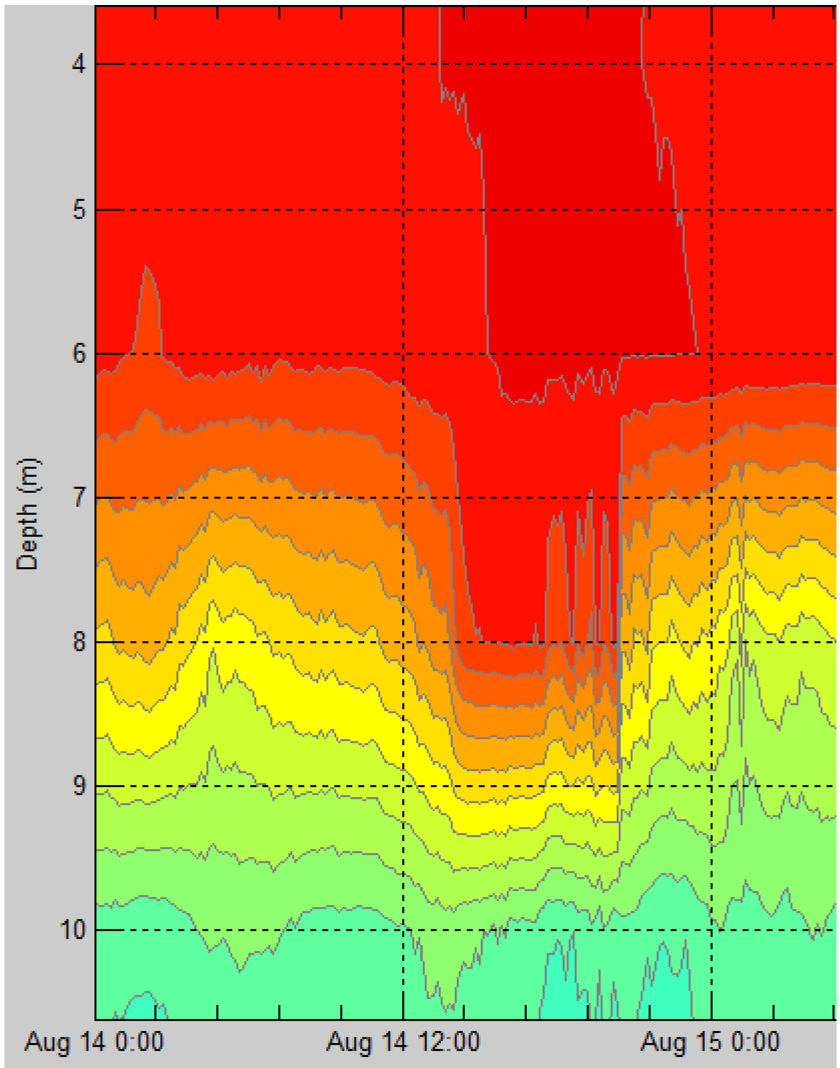


Figure 2-16
Example of 3-4 m Warm Water Intrusion into the Metalimnion on August 14, 2013.

August 14 was fairly windy, with a peak wind speed of nearly 20 mph occurring around 4 pm. The 3-4 m intrusion shown in Figure 2-16 appears to coincide with 4 pm, suggesting that the higher wind had sufficient energy to induce enough mixing and circulation in the upper layer to cause warm water to intrude briefly into the metalimnion.

On September 23, the thermocline suddenly appears to drop several meters, as shown in Figure 2-17.

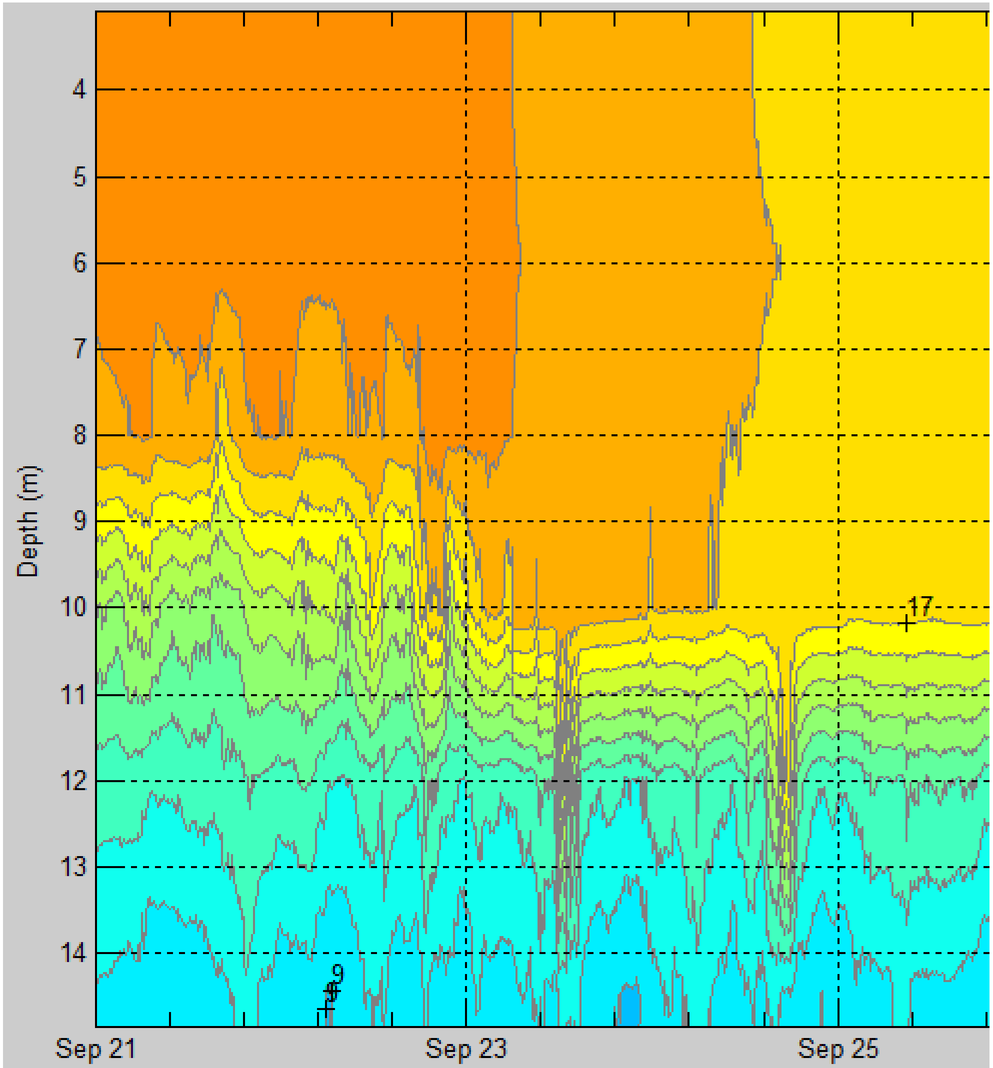


Figure 2-17
Thermocline Eroding and Dropping Several Meters on September 23

This occurs during a period of cooler air temperatures during which the upper layer of the lake cools, becoming denser. This results in density driven mixing and erosion of the thermocline. The extent of mixing is a function of the temperature (density) difference, the thickness of the thermal layers in the lake, and the wind, which supplies energy for mixing. This type of action occurs throughout the rest of the fall through turnover on November 12, where the entire water column gradually cools, the thermocline erodes, and the water column eventually becomes fully mixed.

2.3.3 Secchi Depth and Turbidity

Secchi Depth

Secchi depth has been collected at the deep hole monitoring location from 1978 to present, and a continuous record of turbidity at the raw water intake is available from 1997 to present.

Secchi disk measurements (and turbidity) are used as an indicator of water clarity. A high Secchi depth implies clearer, more transparent water. In general, Secchi depth is correlated with turbidity such that a shallow Secchi disk measurement implies a high turbidity measurement. Low Secchi depth and high turbidity measurements are indicative of particles in the water column such as sediment, bacteria or algae.

Figure 2-18 displays Secchi depth data for 2011, 2012, and 2013 along with the long-term monthly average from 1978 to present (black line), and an envelope of the minimum and maximum Secchi depths measured prior to 2011.

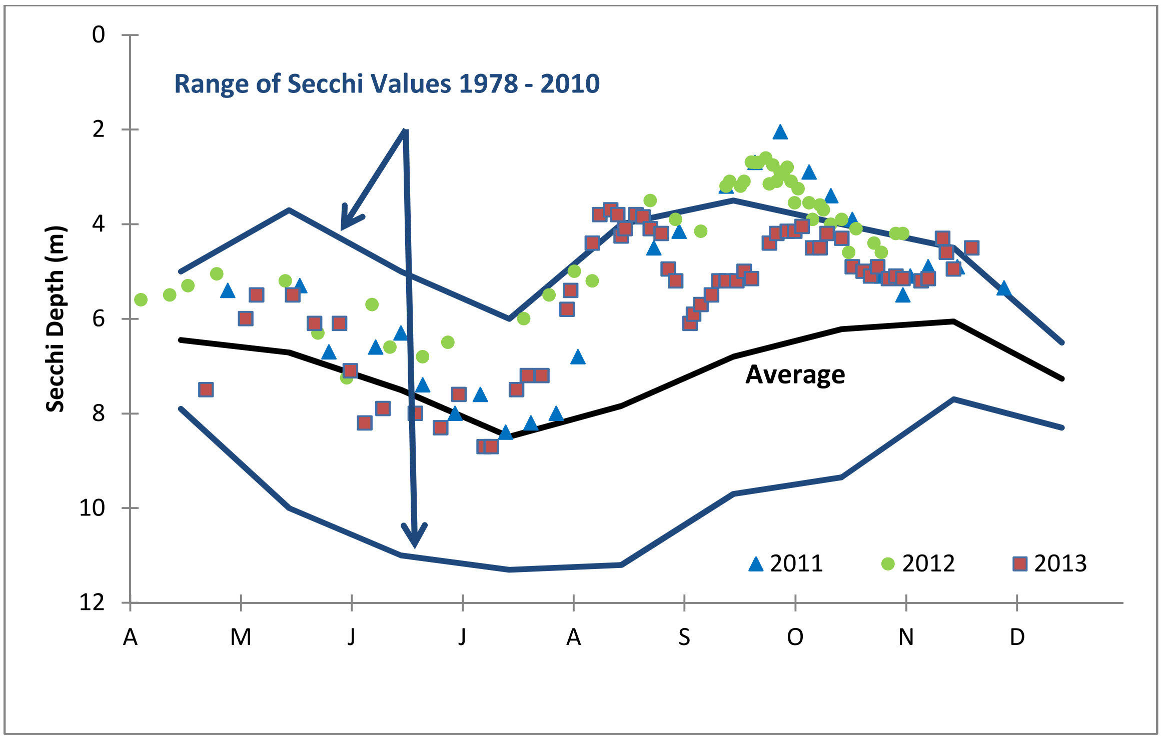


Figure 2-18
Comparison of 2013 Secchi Depths with 2011, 2012, and Long-term Statistics from 1978 to 2010
Average Monthly (black line); Maximum, and Minimum Values for the Record (blue lines)

The values and trend of the monthly average Secchi depths are typical for a high quality lake, where Secchi depths tend to be somewhat shallower in the spring and fall due to algal blooms and deeper in the summer when phosphorus, and thus, algal levels are lower. Compared with 2011 and 2012, the 2013 Secchi depth values were fairly similar through the beginning of August. The 2013 Secchi depths were slightly below average to average for April through July. In all three records, Secchi depths decrease from late July to early August reaching record shallow values. 2013 may have record low

Secchi depths through the first couple of weeks of August, though comparison of the 2013 record from early August to mid-September is complicated by the paucity of 2011-12 values.

From late August through early October, the pattern of Secchi depth in 2013 differs from 2011 and 2012, when values were consistently lower than the lowest depths recorded. In contrast, the 2013 Secchi depths decrease through late August, and then rise again through September, and only touch historic minimum levels in early October.

Overall, the trend in the Secchi depth mirrors that of dissolved oxygen: instances of poor water quality (record low Secchi depths in early August) not previously seen in the record, but overall improved conditions from 2011 and 2012.

Turbidity at the Intake

Intake turbidity data are also available from 1997 to present. Data prior to 2005 are monthly average grab samples from the raw water intake; data after 2005 are a daily average taken directly from the SCADA system. Figure 2-19 shows the average turbidity and the maximum turbidity measurement over 1 NTU for 1997 through 2013.

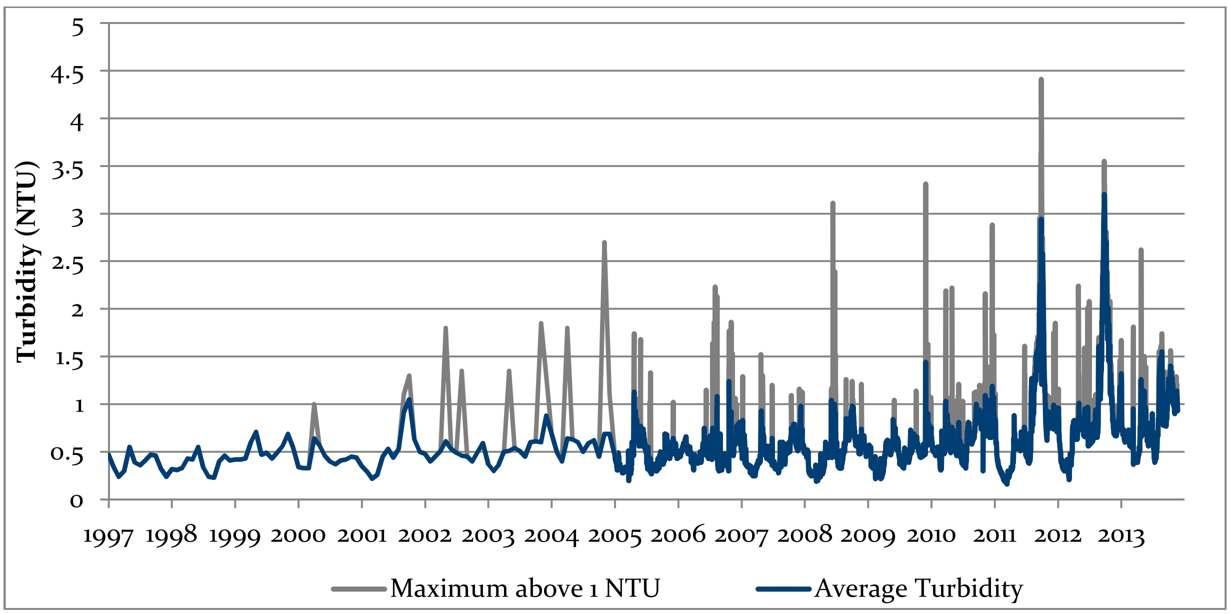


Figure 2-19
Average and Maximum when Above 1 NTU Raw Water Intake Turbidity for 1997 to Present
Average Turbidity is Monthly Average until 2005 and Daily Average from 2005

The turbidity record indicates that prior to 2011 the average turbidity was generally well below 1 NTU; only the peak value had rare excursions above 2 NTU. In 2011 and 2012 peak values nearly reached 4.5 NTU in 2011 and 3.5 NTU in 2012. The maximum turbidity value in 2013 was just over 2.5 NTU, much less than in 2011 and 2012 and similar to peak values seen annually prior to 2011 since the data record changed from monthly average to daily average.

Another metric for turbidity in the intake water is the number of days that maximum turbidity value was above 1 or 2 NTU. Figure 2-20 shows that the number of days with turbidity above 2 NTU dropped significantly, while the number of days with turbidity above 1 NTU was similar.

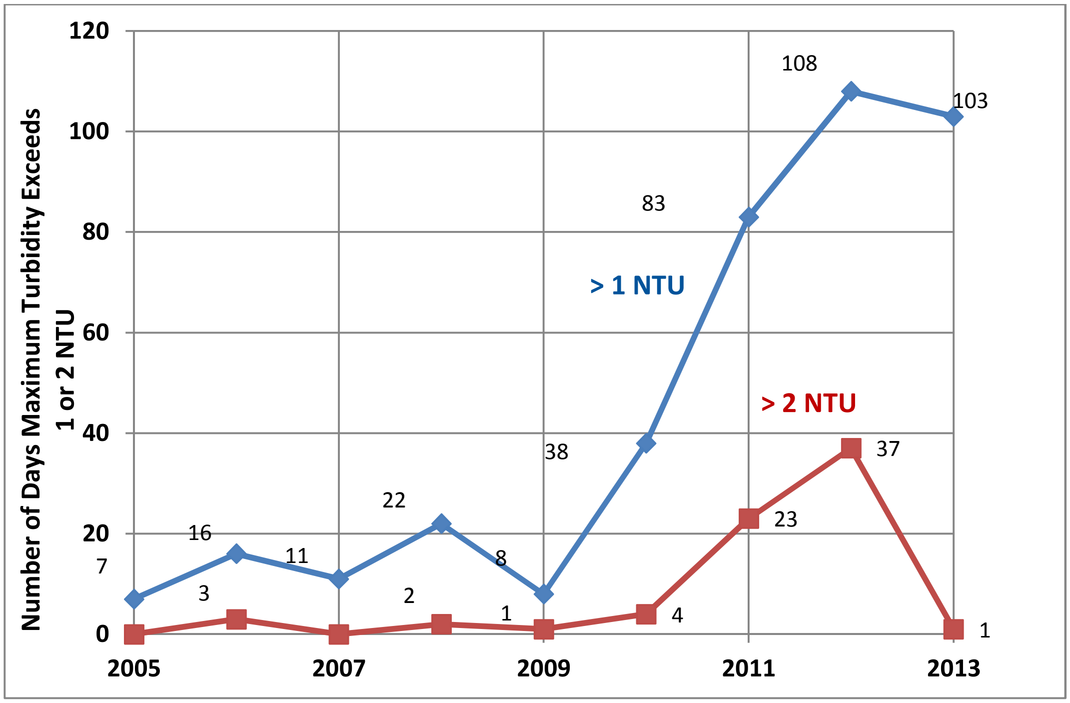


Figure 2-20
Annual Number of Days Raw Water Daily Maximum Turbidity Greater than 1 or 2 NTU

This turbidity record indicates that the water quality was improved over 2012 conditions as measured by both average and peak turbidity values. However, the metric of daily maximum turbidity surpassing 1 NTU was still elevated to around 25 percent of the days in a year, as was the case in 2012 and much higher than the period between 2005 and 2009 when only 5 percent of the days exceeded 1 NTU.

The number of days over which the daily maximum turbidity exceeded 1 NTU was also analyzed on a monthly basis. This analysis, presented in Table 2-2, indicates that most exceedances in 2013 occurred during the months of August through October, which is comparable to the pattern observed in 2011 and 2012. In addition, in 2013 there were a number of days in May where the daily maximum turbidity exceeded 1 NTU (10 days). This appears to be associated with a golden-brown algal bloom that occurred during May.

Table 2-2
Monthly Total of Days where the Daily Maximum Turbidity Exceeds 1 NTU, 2011 through 2013

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
2011	1	0	0	0	0	2	3	20	30	20	4	3	83
2012	3	0	0	3	1	7	1	13	30	31	9	10	108
2013	3	0	2	4	10	0	0	25	12	31	16	0	103

2.4 Nutrient Sampling Data

Limited samples were collected for in-lake nutrient levels prior to 2011. The tables in Appendix B of the Phase 1 report (CDM Smith, 2013) summarize the number of individual analyses by year through 2012, and their locations and depths. In total, the number of analyses is: ammonia (147), nitrate (28), nitrite (28), nitrate+nitrite (197), orthophosphate (110) and total phosphorus (412, over half of which were collected in 2011-12).

In 2013, samples were collected across all five open water lake sampling stations. These samples were generally taken at the surface and at depth, although certain dates also have core (surface to one foot below the thermocline) and intermediate depth samples in addition to measurements taken at the surface and at depth.

2.4.1 Total Phosphorus

The most comprehensive dataset of total phosphorus is from the deep hole sample location, but periodic phosphorus samples have been collected across 13 in-lake sample locations since 1990.

Upper Water (Epilimnetic) Total Phosphorus

Total phosphorus levels prior to 2013 in the epilimnion (to 5 meters depth) range from 6 to 18 µg/l (discounting samples in the early 1990s where detection limits appear to be at issue and an anomalous 1999 value of 62 µg/l), while deep samples (30-35 meters) range from 7 to 58 µg/l.

Table 2-3 provides the average monthly total phosphorus concentration in the epilimnion (to 5 meters depth) and Figure 2-21 provides individual total phosphorus measurements for 2005, 2010, 2011, 2012 and 2013. In general, total phosphorus levels around 10 µg/l are considered desirable as they are indicative of lakes with low algal productivity, while values above 25 µg/l are considered undesirable with high levels of algal productivity.

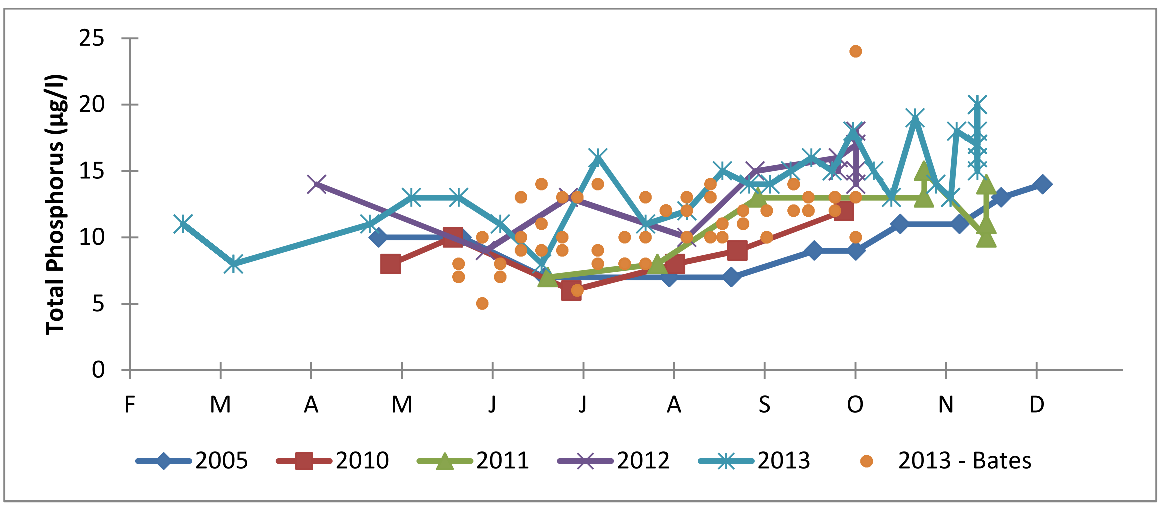


Figure 2-21
Total Phosphorus Measurements in the Upper Waters (0 to 5 meters); All Data from the AWD/LWD Sampling Program except the 2013 Bates College Samples (as noted)
Bates phosphorus data from Drs. Ewing, Weathers, and Cottingham under NSF grants NSF DEB-0749022 NSF EF-0842112, NSF EF-0842125, NSF EF-0842267

Table 2-3
Average Monthly Surface Water Total Phosphorus Concentration – AWD/LWD Data Only
2005 and 2010 – 2013

Month	Average Total Phosphorus (µg/l)				
	2005	2010	2011	2012	2013
April	10	8	-	**14**	11
May	10	10	-	9	**13**
June	7	6	7	**13**	9.5
July	-	-	8	-	**13.5**
August	7	8.5	13	12.5	**13.7**
September	9	12	-	**15.7**	15
October	10	-	14.3	**16**	15.8
November	12	-	11.7	-	**17.1**
December	14	-	-	-	

Compared with prior years, 2013 total phosphorus concentrations in the surface layer are generally similar to those in measured 2012. There is some variation in the spring record where less frequent samples in 2012 can affect the assessment of the change in magnitude of the concentrations, especially when looking at monthly average values such as those shown in Table 2-3. Based on available data, both 2012 and 2013 have higher spring concentrations than previous years, suggesting a greater mass of phosphorus was present in the lake to fuel spring algal blooms. In 2013, the clear water period concentration in June is also higher than previous years. Throughout the remainder of the growing season, 2013 levels are similar to 2012 and show a general increase with time, consistent with the trend in previous years, where each year exhibits a steady increase in late summer-fall total phosphorus concentrations. This rise in concentrations is consistent with an internal source of phosphorus.

Total Phosphorus at the Bottom of the Lake

Figure 2-22 presents total phosphorus measurements at the deepest point in the lake since 2000. Although sampling was relatively infrequent prior to 2010, the data indicate that deep-water phosphorus concentrations increase throughout each year until the lake fully mixes in November.

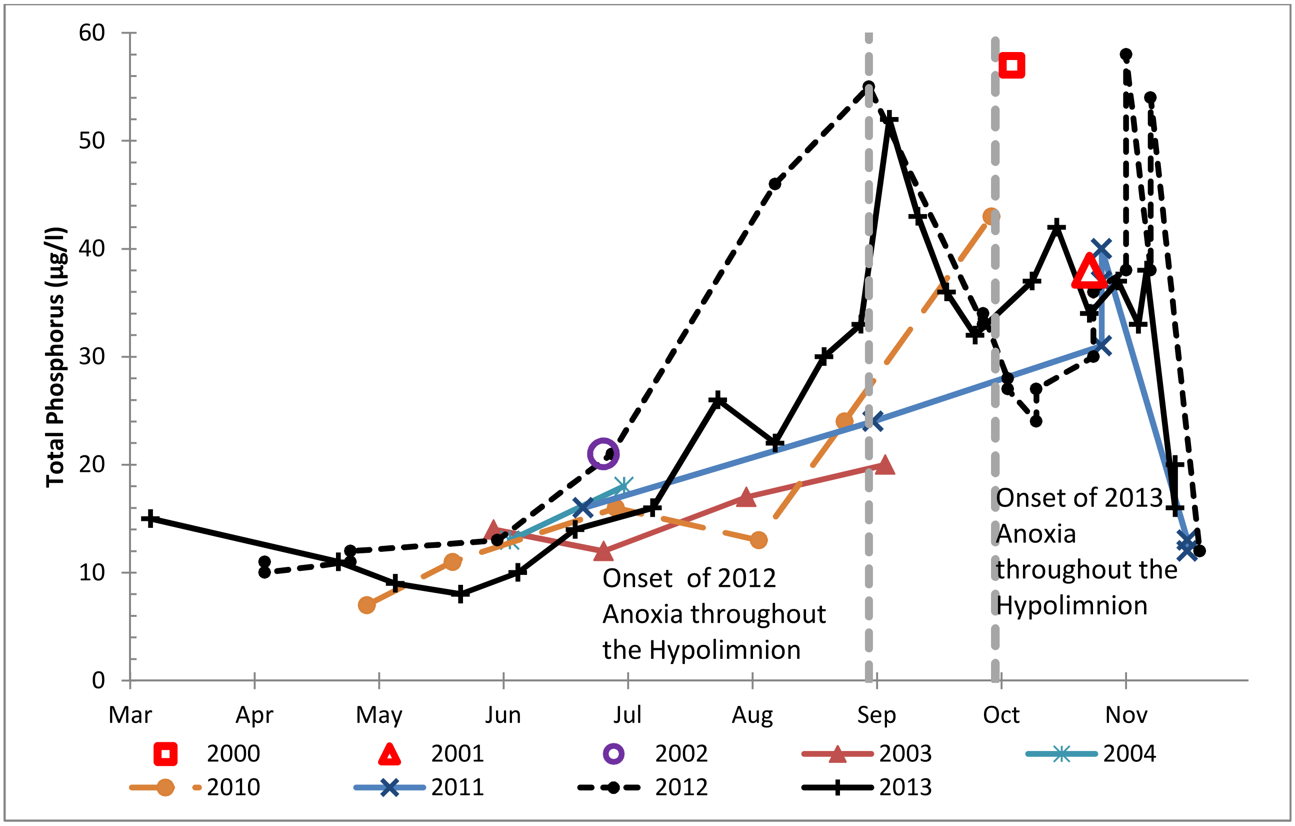


Figure 2-22
Bottom Water (30 – 35 m) Total Phosphorus Measurements, 2001 – Present
Note: this plot omits a total phosphorus concentration of 79 µg/l at the deep hole sampling site on October 2, 2013

In all years when samples were collected in the fall (2001, 2002, 2010, 2011, 2012, and 2013), total phosphorus values measured between 30 and 35 meters exceeded 35 µg/l. The total phosphorus in 2013 (solid black line with crosses in Figure 2-22) was lower than 2012 (dotted purple line) June through August, but increased substantially in September, matching the observed deep-water phosphorus concentrations in 2012 during this time period. There was one 2013 measurement of nearly 80 µg/l in early October that appears to be an anomaly as it represents the highest total phosphorus measurement on record and the rest of the October samples more closely resemble the concentrations observed in September[1]. The October and November 2013 phosphorus samples were similar in magnitude to those in 2011 and 2012 and appear to match conditions observed in the sparse historic dataset (e.g., 2000 and 2001).

In 2013, bottom phosphorus samples were collected at three other sites in addition to the deep hole: site 30 at 9 m, site 31 between 28 and 29 m, and site 32 between 17 and 18 m. A time history graph of the bottom water total phosphorus at each of these sites including the deep hole is shown in Figure 2-23.

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[1] This measurement was from 34 meters at the deep hole, so it is conceivable that the sample contained disturbed organic matter from the sediment bed and is thus not representative of conditions in the actual water column.

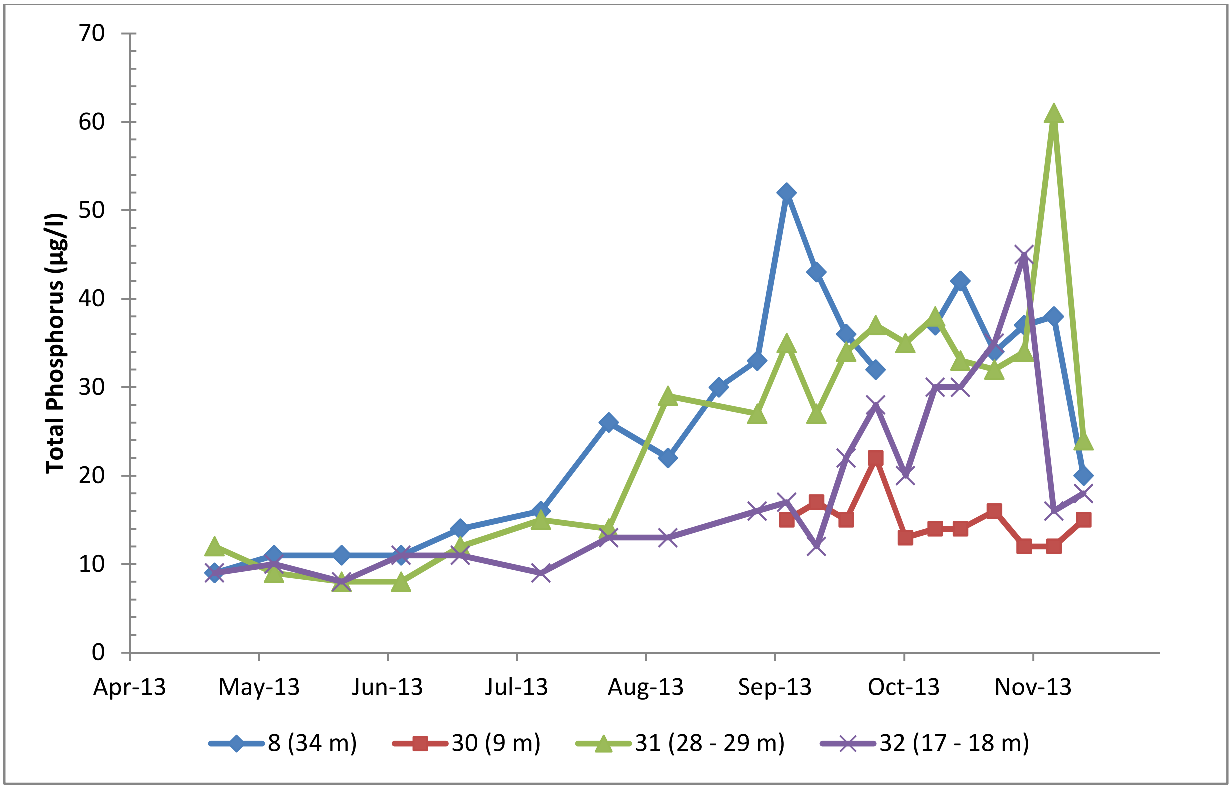

Figure 2-23
Bottom Total Phosphorus Concentration at Four Open Water Lake Sampling Sites in 2013
Note: this plot omits two outliers: a value of 79 µg/l at the deep hole sampling site on October 2 and a value of 150 µg/l at site 32 on September 11.

Measured bottom total phosphorus concentrations across the lake show that the concentration typically increases with increasing depth. Like the deep hole, Site 31 is located in an isolated deep depression in the lake's bottom; its total phosphorus levels are similar in pattern and magnitude to those at the deep hole. Site 32, a shallower site located close to the top of the hypolimnion, has an overall lower bottom total phosphorus concentration until mid-October when concentrations match those at the deeper sites. Site 30, whose bottom is located at 9 m, has much lower total phosphorus levels though the concentrations are still elevated relative to the average upper water concentration. This indicates that some amount of internal release of phosphorus is occurring in the shallower areas of the lake.

2.4.2 Nitrogen

Limited nitrogen data analyzed for nitrate, nitrite and ammonia was collected sporadically between 2005 and present day. The available data have values that are generally below the detection limit. Nitrogen sampling increased in 2013: 172 nitrate or nitrate + nitrite samples in 2013, compared with 148 samples total between 2001 and 2012, and 33 in 2012 alone.

Nitrate was measured between 2000 and 2004, was often not detected, and never exceeded 0.2 mg/l. Nitrite was never detected in the same time period. Starting in 2005 a combined parameter nitrate+

nitrite was measured. This was rarely detected in the lake (and then never exceeded 0.02 mg/l) until October 2012. Using only the discrete samples (and not the cores), the nitrate + nitrite was typically 0.03 or 0.04 mg/l, with the highest concentrations of 0.07 and 0.09 mg/l at 20 and 25 meters depth, respectively on November 1, 2012.

In 2013, nitrate + nitrite was measured. As observed in prior years, nitrate + nitrite concentrations were at times below the detection limit. The highest measured nitrate + nitrite concentrations (again using only the discrete samples) were above 0.2 mg/l, all occurring in deep samples after late August. Shallow nitrate + nitrite values are low – at the detection limit to 0.03 mg/l – which is consistent with prior years.

Although ammonia has historically been measured, it was very rarely detected; ammonia was only detected in 18 samples between 2005 and May 2012, and was typically found to be 0.01 mg/l and only once at 0.3 mg/l. In 2012, 44 samples were collected in October-November at various locations and depths in the lake that had typical ammonia concentrations ranging between 0.03 and 0.04 mg/l in October and 0.05 and 0.06 mg/l in November. Ammonia was not sampled in 2013, but total Kjeldahl nitrogen (TKN) was sampled, which is a measure of the sum of ammonia and organic nitrogen. TKN ranged from 0.1 mg/l in the surface waters to a maximum of 0.5 mg/l at depth in the deep hole. TKN was not measured prior to 2013. However, assuming that ammonia concentrations remain low, this result indicates that organic nitrogen is likely the dominant form of nitrogen in the lake.

2.4.3 Tributary Phosphorus Loads

Water quality data have historically been collected at several tributaries. In 2013, the tributary sampling program was altered to focus primarily on the North Auburn Dam and Townsend Brook and to include both wet and dry weather samples. These two tributaries represent over 60% of the total surface water inflow to the lake (Dudley, 2004), and no other single tributary has significant contributory area. Thus, the contribution from the Basin and Townsend Brook should represent a significant portion of the total surface water load into the lake.

Samples were differentiated between wet and dry using daily precipitation records from the Lewiston, ME rain gage. Table 2-4 lists all of the 59 tributary sampling dates in 2013 with wet and dry sampling periods identified. Note that all sites were not necessarily sampled on all days; an entry in this table simply indicates that a tributary sample was taken during that event. Wet periods are further differentiated by daily precipitation volume, separated at a volume of 0.5 inches.

Table 2-4
2013 Tributary Sampling Dates

Event #	Date	Rain (in)[1]	Wet/Dry	Event #	Date	Rain (in)[1]	Wet/Dry
1	4/11/2013	0.02	Wet < 0.5"	31	8/20/2013	0	Dry
2	4/16/2013	0	Dry	32	8/22/2013	0	Dry
3	4/23/2013	0	Dry	33	8/27/2013	0	Dry
4	4/30/2013	0	Dry	34	8/28/2013	0.54	Wet > 0.5
5	5/7/2013	0	Dry	35	8/29/2013	0	Dry
6	5/13/2013	0	Dry	36	8/30/2013	0	Dry
7	5/15/2013	0.2	Wet < 0.5"	37	8/31/2013	0.1	Wet < 0.5"
8	5/21/2013	0.28	Wet < 0.5"	38	9/1/2013	0	Dry
9	5/24/2013	1.39	Wet > 0.5	39	9/2/2013	2.98	Wet > 0.5
10	5/25/2013	0.59	Wet > 0.5	40	9/3/2013	0	Dry
11	5/26/2013	0.14	Wet < 0.5"	41	9/4/2013	0.05	Wet < 0.5"
12	5/27/2013	0	Dry	42	9/5/2013	0.18	Wet < 0.5"
13	6/4/2013	0	Dry	43	9/10/2013	0	Dry
14	6/7/2013	0.4	Wet < 0.5"	44	9/13/2013	1.09	Wet > 0.5
15	6/8/2013	0.57	Wet > 0.5	45	9/14/2013	0	Dry
16	6/11/2013	0.66	Wet > 0.5	46	9/15/2013	0	Dry
17	6/12/2013	0.1	Wet < 0.5"	47	9/24/2013	0	Dry
18	6/18/2013	0	Dry	48	10/1/2013	0	Dry
19	6/25/2013	1.27	Wet > 0.5	49	10/8/2013	0	Dry
20	6/28/2013	1.77	Wet > 0.5	50	10/17/2013	0	Dry
21	6/29/2013	0	Dry	51	10/22/2013	0	Dry
22	7/2/2013	0.34	Wet < 0.5"	52	10/29/2013	0	Dry
23	7/11/2013	0.03	Wet < 0.5"	53	10/31/2013	0.32	Wet < 0.5"
24	7/16/2013	0.09	Wet < 0.5"	54	11/1/2013	0.43	Wet < 0.5"
25	7/23/2013	1.17	Wet > 0.5	55	11/6/2013	0	Dry
26	7/30/2013	0	Dry	56	11/19/2013	0	Dry
27	8/6/2013	0	Dry	57	11/26/2013	0.06	Wet < 0.5"
28	8/10/2013	0	Dry	58	11/27/2013	2.64	Wet > 0.5
29	8/11/2013	0	Dry	59	11/28/2013	0	Dry
30	8/13/2013	0.13	Wet < 0.5"				

Notes: 1. Precipitation data derived from hourly data collected at the Lewiston Main St. pump station.

Average 2013 dry and wet weather concentrations were computed from the observed record using these classifications. For this analysis wet weather is considered any precipitation event with over 0.5 inches of rain, and dry weather is considered any dry day. Table 2-5 shows average dry and wet weather concentrations for the North Auburn Dam and Townsend Brook.

Table 2-5
Average 2013 Tributary Total Phosphorus Concentration during Dry and Wet Weather

Sample Type	Townsend Brook Phosphorus (µg/l)	North Auburn Dam Phosphorus (µg/l)
Dry	22.1	11.0
Wet (> 0.5”/day)	25.9	12.9

The data indicate that concentrations from Townsend Brook are about double those from the North Auburn Dam. These differences could occur for several reasons. The most obvious is that there is an additional source of phosphorus in the Townsend Brook watershed that is not ‘captured’ in land use based watershed loading estimates. AWD/LWD have already begun sampling along Townsend Brook to determine if an area with a high load can be identified, and then controlled. Another important factor is the multiple in-line ponds that occur in the Basin watershed. These ponds, and in particular the Basin itself (behind the North Auburn Dam), act as sedimentation ponds and improve water quality by removing particulate based pollutants. They are effective until their capacities are compromised by sediment accumulation, and thus, it is important to monitor sediment accumulation and take proactive steps to remove the sediment before scouring floodwaters transport the previously captured sediment to Lake Auburn. Finally, while these differences in concentration at the two tributaries is significant, their impact on the lake is mitigated by the much larger load coming from the North Auburn Dam due to higher contributory flows.

In addition, to help calculate loads, flow metering equipment was installed at the North Auburn Dam in 2012 and at Townsend Brook in 2013. The quality of the flow record at Townsend Brook appears to be acceptable during wet weather but suffers from data quality issues during dry weather. This is caused by limitations of the velocity probe where it is unable to accurately measure the lower velocities associated with dry weather flow. The corresponding depth record indicates that Townsend Brook always had flow, even during dry weather conditions. The North Auburn Dam flow is computed using a weir equation based on depth above the weir; this means that the flow measurement is more reliable during dry weather conditions because it is not relying on a velocity sensor to measure the lower dry weather velocities.

Samples were not taken during every wet and dry period in 2013. Therefore, to estimate the load at each tributary, the measured daily average streamflow was first separated into its baseflow and wet weather flow components. Wet weather was defined as any period when total flow exceeded the baseflow. Then, the load was computed by applying the dry and wet average measured phosphorus concentrations to the demarcated dry and wet weather periods, respectively.

At the North Auburn Dam the average load over the period of record (October 25, 2012 to December 26, 2013) was 0.45 kg/day; in 2013 the total computed load was 180 kg with an average load of 0.5 kg/day.

The Townsend Brook gage was only active between July and October 2013, so it is not possible to directly compute an annual phosphorus load. Using the same approach as at the North Auburn Dam for the period of record of the Townsend Brook gage, the average daily load at Townsend Brook was estimated as 0.35 kg/day; an annual load approximated for 2013 would be 130 kg. Annual Townsend Brook loads were also estimated from a correlation between baseflow with the North Auburn Dam flows and fitting a power law regression between North Auburn Dam and Townsend Brook baseflow. This results in a significant and reasonably strong correlation (Figure 2-24) that was used to create a synthetic estimated streamflow for 2013. Computing the Townsend Brook load using this methodology yields an estimated load of 120 kg. These estimated loads are summarized in Table 2-6.

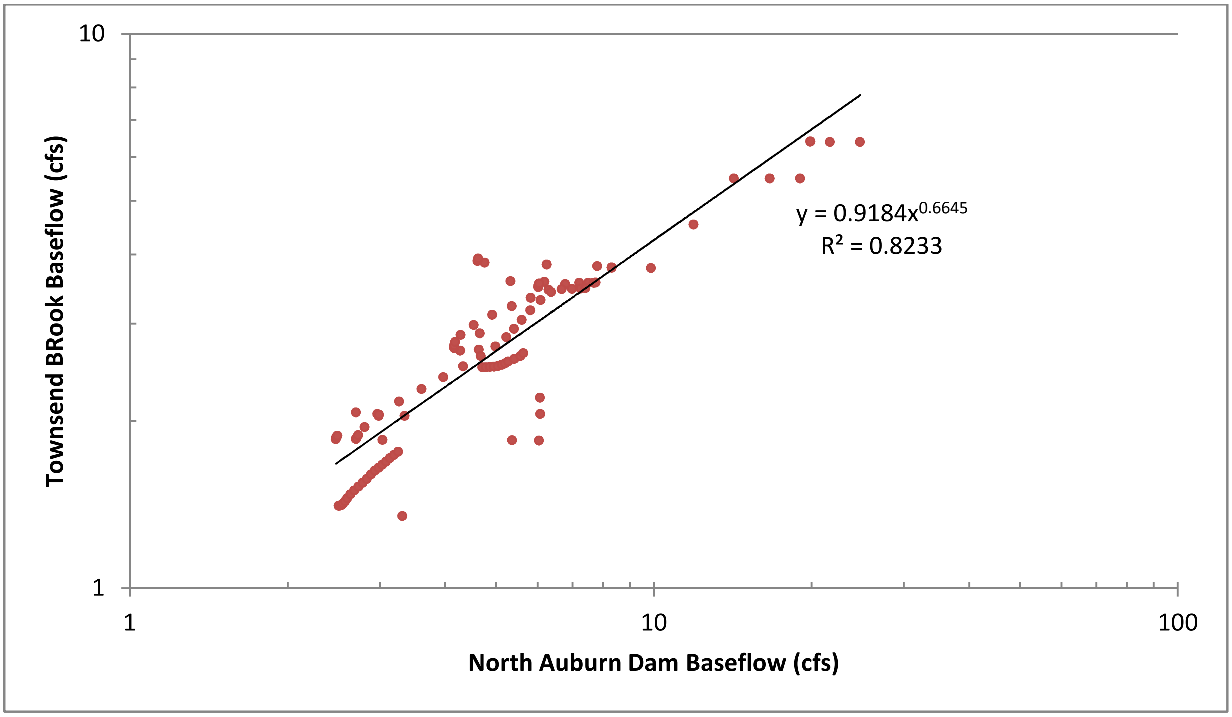

Figure 2-24
Correlation between Townsend Brook and North Auburn Dam Baseflow

Table 2-6
Estimated 2013 Average Daily and Annual Total Phosphorus Load, North Auburn Dam and Townsend Brook

Tributary	Estimated Average Daily Load (kg/day)	Estimated Annual Total Load (kg/yr)
North Auburn Dam	0.5	180
Townsend Brook	0.35[1]	120 – 130[2]

Notes: 1. The Townsend Brook flow record spans July 25 to October 23, 2013. Therefore, the estimated daily load is derived from this period of record only and not all of 2013.
2. The upper bound is estimated by extrapolating the estimated daily average load to the entire year. The lower bound is estimated by generating a synthetic streamflow record based on North Auburn Dam records.

These computed loads can be used to compare to the estimated nutrient loads presented in CEI's nutrient budget (CEI, 2013a). The estimated loads were derived from land use area and typical phosphorus concentrations in runoff by land use; tributary loads were estimated using the land use proportions presented in Table 3-2 of CEI's *Lake Auburn Diagnostic Study* (CEI, 2013a). Land areas and percentages by land use are presented in Table 2-7 for the area tributary to the North Auburn Dam and Townsend Brook.

Table 2-7
Percentage of North Auburn Dam and Townsend Brook Occupied by Each Land Use Category[1]

Land Use	Total Area (ac)	N. Auburn Dam Area (ac)	N. Auburn Dam Land Use Percentage	Townsend Brook Area (ac)	Townsend Brook Land Use Percentage
hay/pasture	960	342	36%	180	19%
cropland	179	55	31%	0	0%
forested	7,324	3,799	52%	1,096	15%
wetland	296	185	63%	54	18%
low intensity	299	84	28%	82	27%
high intensity	356	144	40%	58	16%
TOTAL	**9,414**	**4,609**	**-**	**1,470**	**-**

Note: 1. Modified from Tables 3-1 and 3-2 in CEI's *Lake Auburn Diagnostic Study* (CEI, 2013a)

These land use percentages were then used to scale the reported loads for 2010, 2011, and 2012. These loads are presented on an annual basis and an average of those years for both watersheds in Table 2-8.

The average estimated load from the CEI nutrient budget is close to the average computed from the 2013 flow and water quality monitoring data. The land use-based average (2010-2012) North Auburn Dam phosphorus load of 210 kg/yr compares favorably with the 2013 load of 180 kg/yr estimated from sampling data; while, the land use-based average Townsend Brook phosphorus load of 85 kg/yr is about 70% of the 2013 load of 120 – 130 kg/yr estimated from water quality data and streamflow transposition. The difference between the closer agreement at the North Auburn Dam and the greater difference in agreement at Townsend Brook may be attributable to the higher total phosphorus concentrations measured at Townsend Brook. This suggests there could be a phosphorus load in the Townsend Brook watershed that is not represented by the land use-based approach. Summit Environmental recently performed a study of phosphorus concentrations along Townsend Brook. This study found several potential sources of phosphorus, particularly along the tributaries to Townsend Brook.

Table 2-8
Estimated North Auburn Dam and Townsend Brook Loads Derived from the Nutrient Budget Presented in *Lake Auburn Diagnostic Study* (CEI, 2013a)

	Total Load to Lake Auburn (kg/yr)			North Auburn Dam (kg/yr)			
Land Use	**2010**	**2011**	**2012**	**2010**	**2011**	**2012**	**Average**
hay/pasture	367	264	236	131	94	84	103
cropland	106	86	86	33	26	26	29
forested	29	14	25	15	7	13	12
wetland	4	3	4	3	2	2	2
low intensity	13	14	15	4	4	4	4
high intensity	136	151	145	55	61	59	58
TOTAL	**656**	**532**	**510**	**240**	**195**	**189**	**208**
	Total Load to Lake Auburn (kg/yr)			**Townsend Brook (kg/yr)**			
Land Use	**2010**	**2011**	**2012**	**2010**	**2011**	**2012**	**Average**
hay/pasture	367	264	236	69	50	44	54
cropland	106	86	86	0	0	0	0
forested	29	14	25	5	2	4	3
wetland	4	3	4	1	0.5	0.5	0.5
low intensity	13	14	15	4	4	4	4
high intensity	136	151	145	22	25	24	24
TOTAL	**656**	**532**	**510**	**100**	**80**	**76**	**85**

2.5 Algae and Chlorophyll Data

2.5.1 Chlorophyll *a*

Chlorophyll *a* has been collected sporadically across 13 in-lake sample locations since 1999, with most samples collected at the deep hole location. Most chlorophyll *a* samples are epilimnetic core measurements, which capture a depth-integrated concentration throughout the epilimnion. Chlorophyll data are usually evaluated as summer average concentrations because the occurrence of a short-duration bloom is not usually a cause for concern; summer average chlorophyll levels below 3 µg/l would represent a high quality lake with low algal productivity. For Lake Auburn, the individual sample points are used instead because samples were not consistently collected.

In 2013, a more comprehensive chlorophyll *a* sampling program was implemented. Chlorophyll samples were collected and analyzed monthly at the five open water lake sampling locations and at Bates College sites 1, 2 and 3 around the perimeter of the lake. A time series plot of all chlorophyll data collected to date from open water in lake sampling locations is in Figure 2-25; Figure 2-26 presents chlorophyll data collected during 2013 from both the open water sampling locations and the Bates perimeter sampling locations.

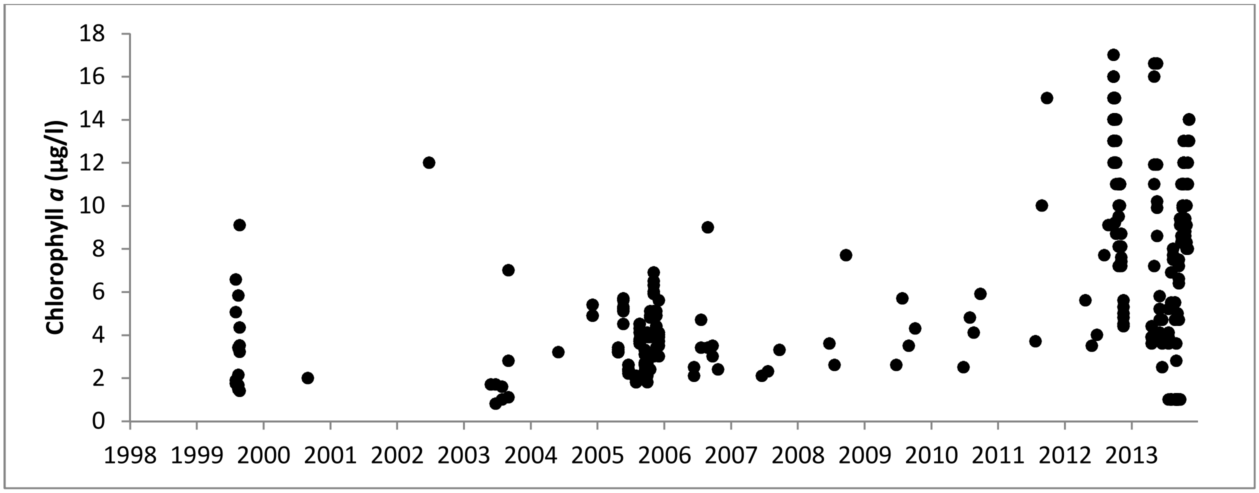

Figure 2-25
Time Series Plot of Chlorophyll *a*, 1999-Present

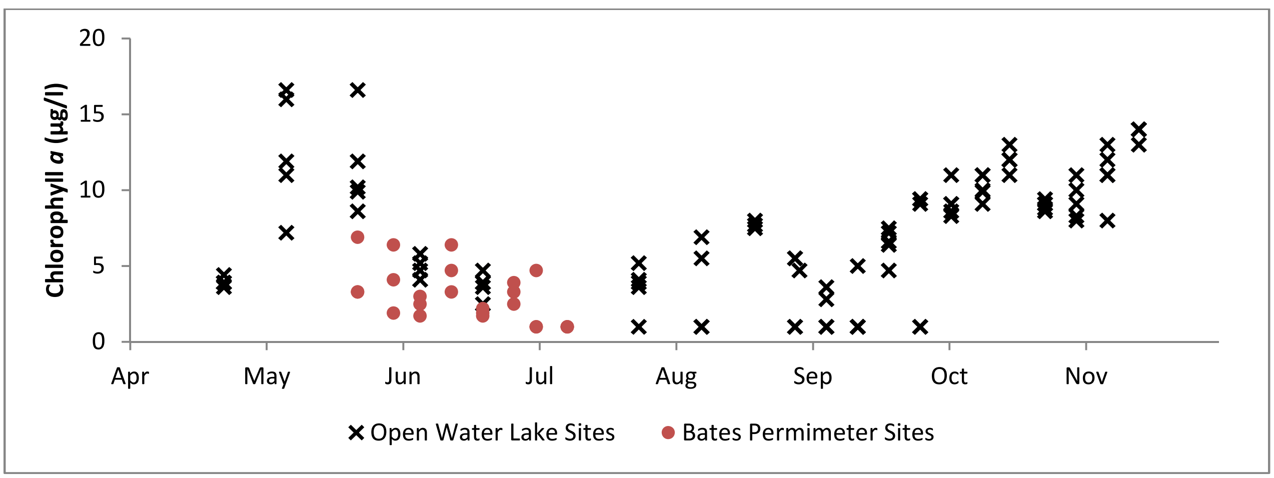

Figure 2-26
2013 Chlorophyll *a* Data for Open Water Lake Sites and Bates College Perimeter Sites
Bates chlorophyll data from Drs. Ewing, Weathers, and Cottingham under NSF grants NSF DEB-0749022, NSF EF-0842112, NSF EF-0842125, NSF EF-0842267

Figures 2-25 and 2-26 indicate that the early spring and late fall concentrations were higher than those typically observed during previous years but that the overall peak concentrations were lower than those observed in 2012. Even though chlorophyll is not a measure of the total algal community, there is agreement between the chlorophyll record and the Secchi disk and turbidity records, indicating that there was less algal growth in 2013 than in 2012. Figure 2-26 also shows the increase in chlorophyll associated with the record low Secchi depths in early August 2013.

2.5.2 Bates College *Gloeotrichia* Sampling

Bates College again sampled for *Gloeotrichia echinulata*. *Gloeotrichia* samples were collected from the Bates College perimeter sites as well as from the open water lake sampling sites. This was the fourth year that the perimeter was sampled and the first year that the open water lake sites were sampled. A time history of perimeter *Gloeotrichia* concentration from 2011, 2012, and 2013 is presented in Figure 2-27. 2013 *Gloeotrichia* measurements from both the open water lake samples and the perimeter samples are presented in Figure 2-28. The minimum and maximum measured *Gloeotrichia* concentrations on each date are designated by the error bars on both plots.

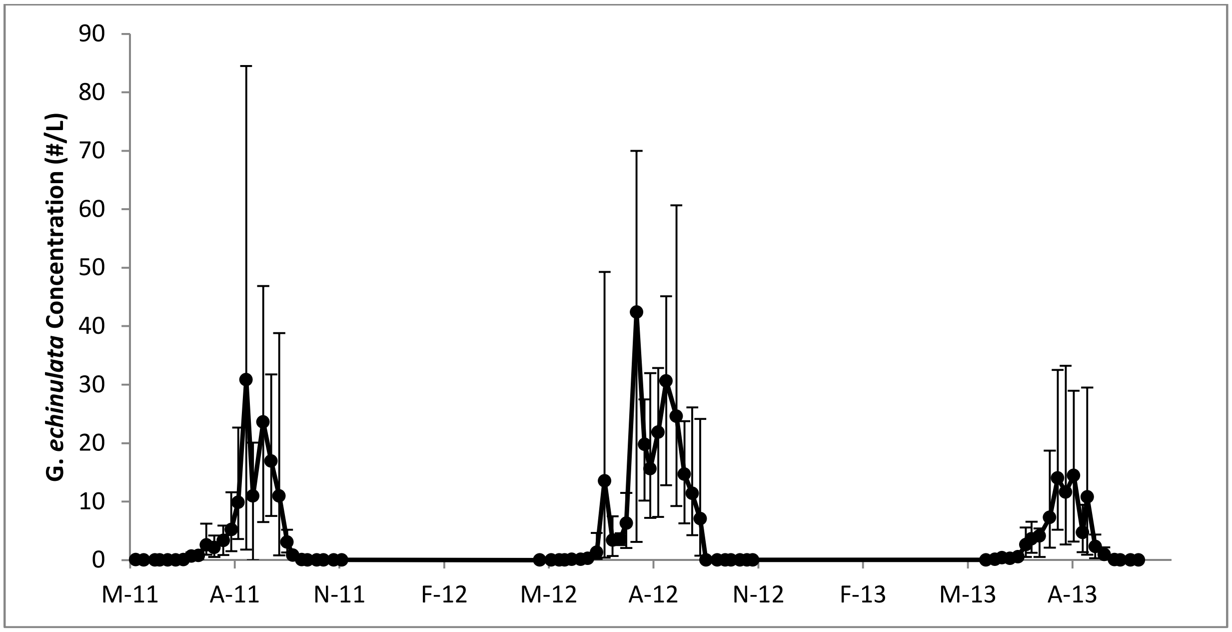

Figure 2-27
Time History of *Gloeotrichia* Concentration from the Bates College Perimeter Sampling Sites
Gloeotrichia Data from Drs. Ewing, Weathers, and Cottingham under NSF grants NSF DEB-0749022, NSF EF-0842112, NSF EF-0842125, NSF EF-0842267

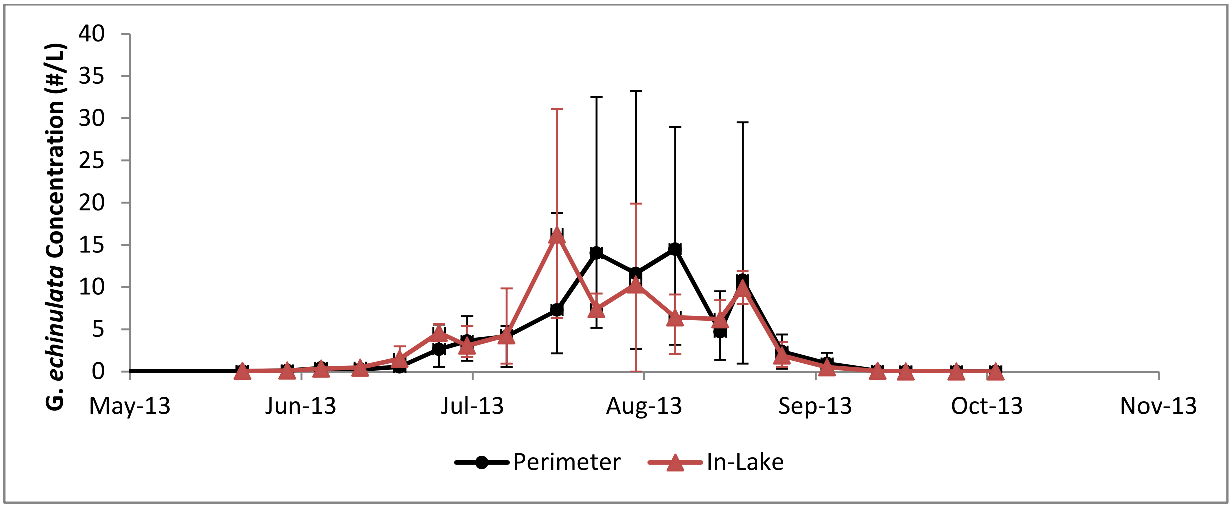

Figure 2-28
2013 Open Water Lake and Perimeter *Gloeotrichia* Concentration
Gloeotrichia Data from Drs. Ewing, Weathers, and Cottingham under NSF grants NSF DEB-0749022, NSF EF-0842112, NSF EF-0842125, NSF EF-0842267

Order of magnitude estimates described in Section 3 of the Phase 1 report (CDM Smith, 2013) indicate that *Gloeotrichia* could be one cause for increased phosphorus in the epilimnion.

2.5.3 AWD/LWD Algal Enumeration

AWD/LWD collected and enumerated phytoplankton at the five open water lake sampling sites. Most of the measurements were collected as a core sample, which is a depth integrated sample covering all water from the surface to 1 meter below the thermocline. A small subset of samples were collected from a 3 m band around the thermocline at the deep hole to understand if there are high concentrations of algae at the thermocline.

The 2013 algae data can be compared with a 2011 full algae enumeration performed under the direction of Dr. Holly Ewing at Bates College to understand differences in algae taxa between the sampling years. Unique to 2013 is an overall lower concentration of cyanobacteria than were present in 2011 and 2012. This manifests itself in the significantly lower turbidity measurements and is part of the reason why an algicide application was not required in 2013. Instead, the dominant phyla were golden brown algae – chrysophyta – in May and November (principally *Dinobryon* in May and *Synura* in November), diatoms (principally *Asterionella and Fragillaria)* in the fall, and a mix of greens, blue-greens, and diatoms during the summer. The *Synura* bloom in late November caused a number of taste and odor complaints, as large concentrations can impart a cucumber or fishy taste to the water. Histograms showing algae cell count by phyla at each of the five lake sampling sites are presented in Figures 2-29 through 2-33.

In addition to the core samples collected throughout the lake, a limited subset of samples was collected within 3 meters around the thermocline at the deep hole station. A histogram, shown in Figure 2-34, compares coincidental samples taken at the thermocline (designated with a "T" after the date) alongside the core samples (no designation after the date).

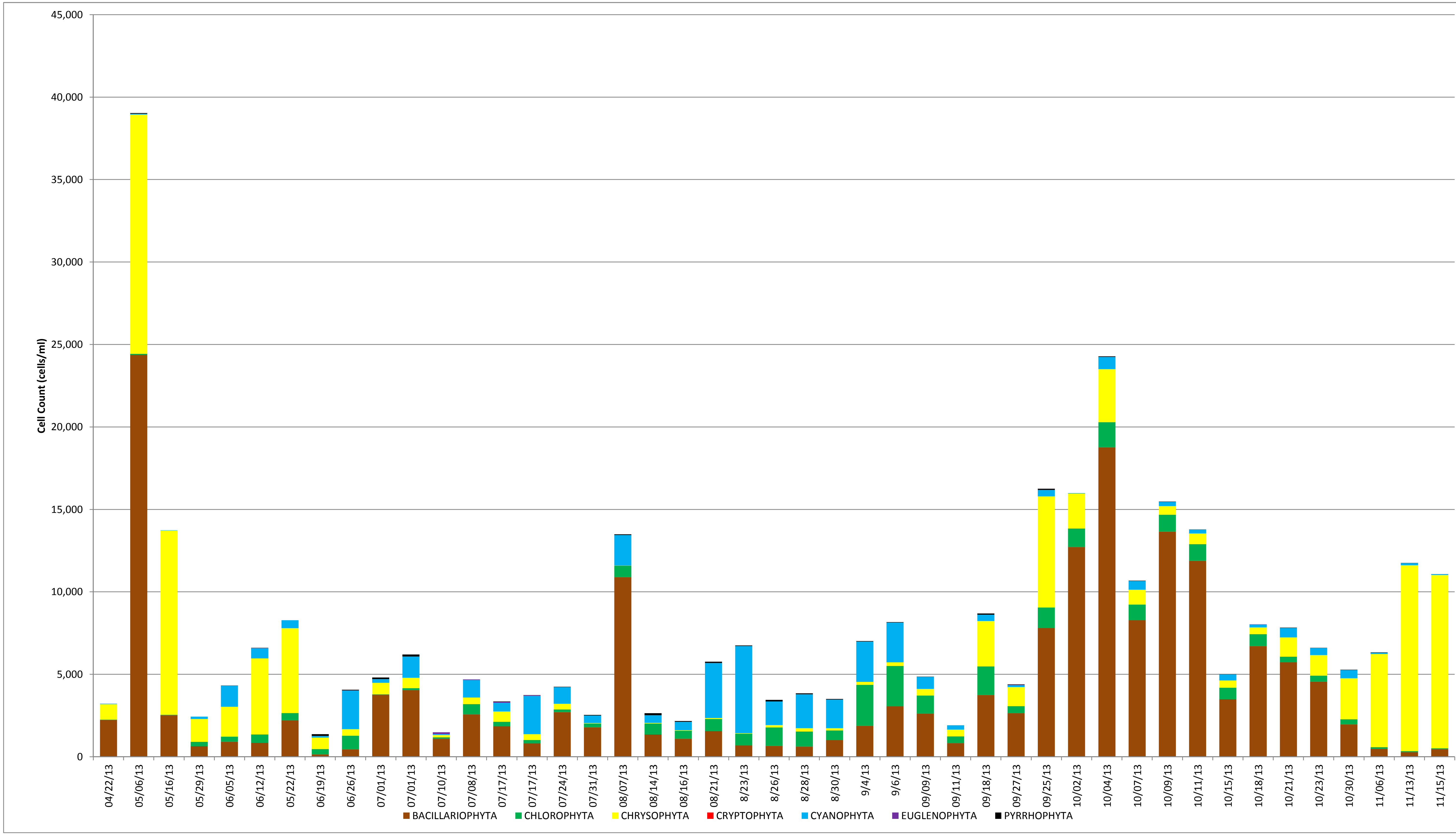

Figure 2-29
Deep Hole Core Algae Cell Count

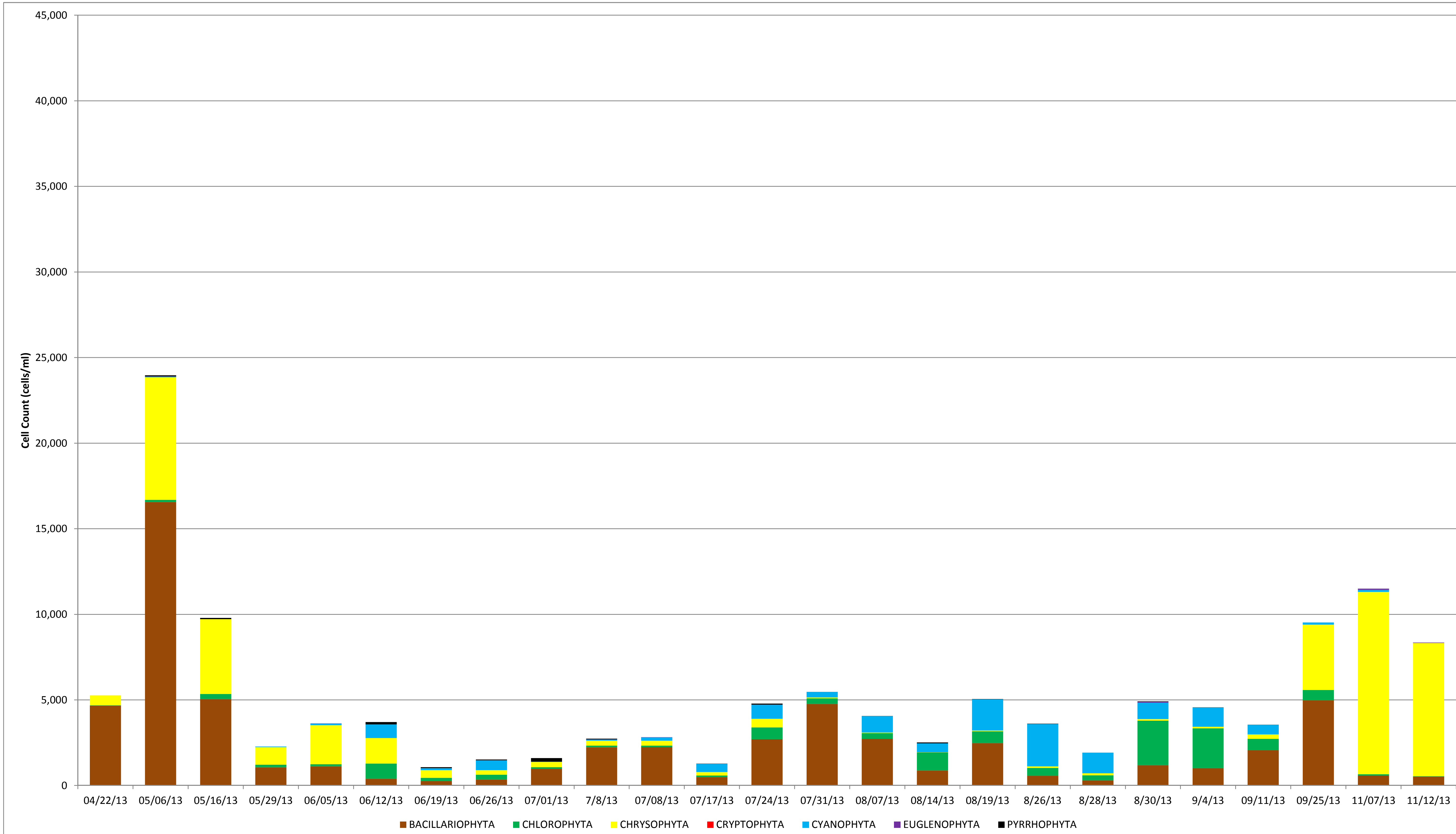

Figure 2-30
Site 12 (Intake) Core Algae Cell Count

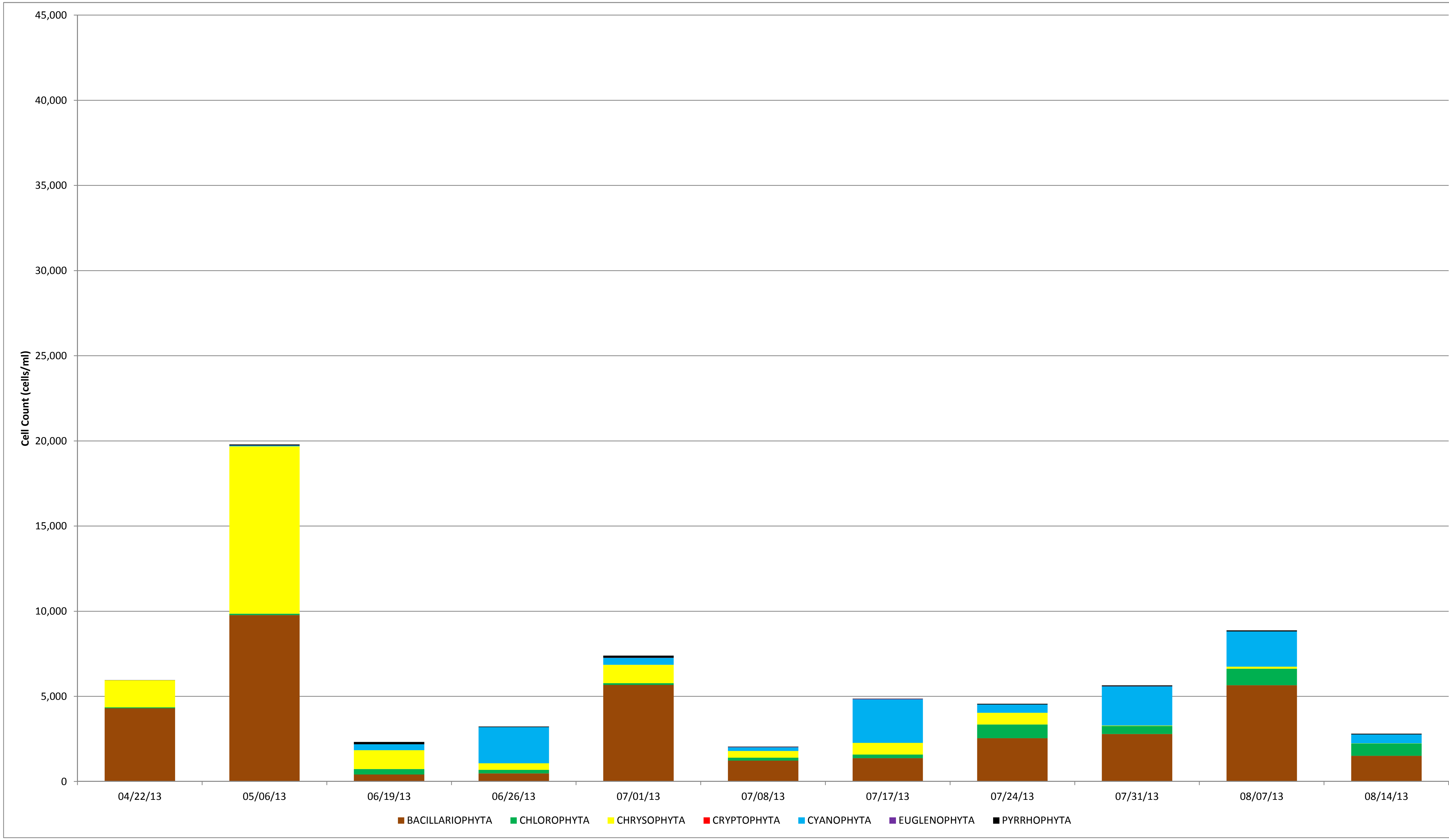

Figure 2-31
Site 30 Core Algae Cell Count

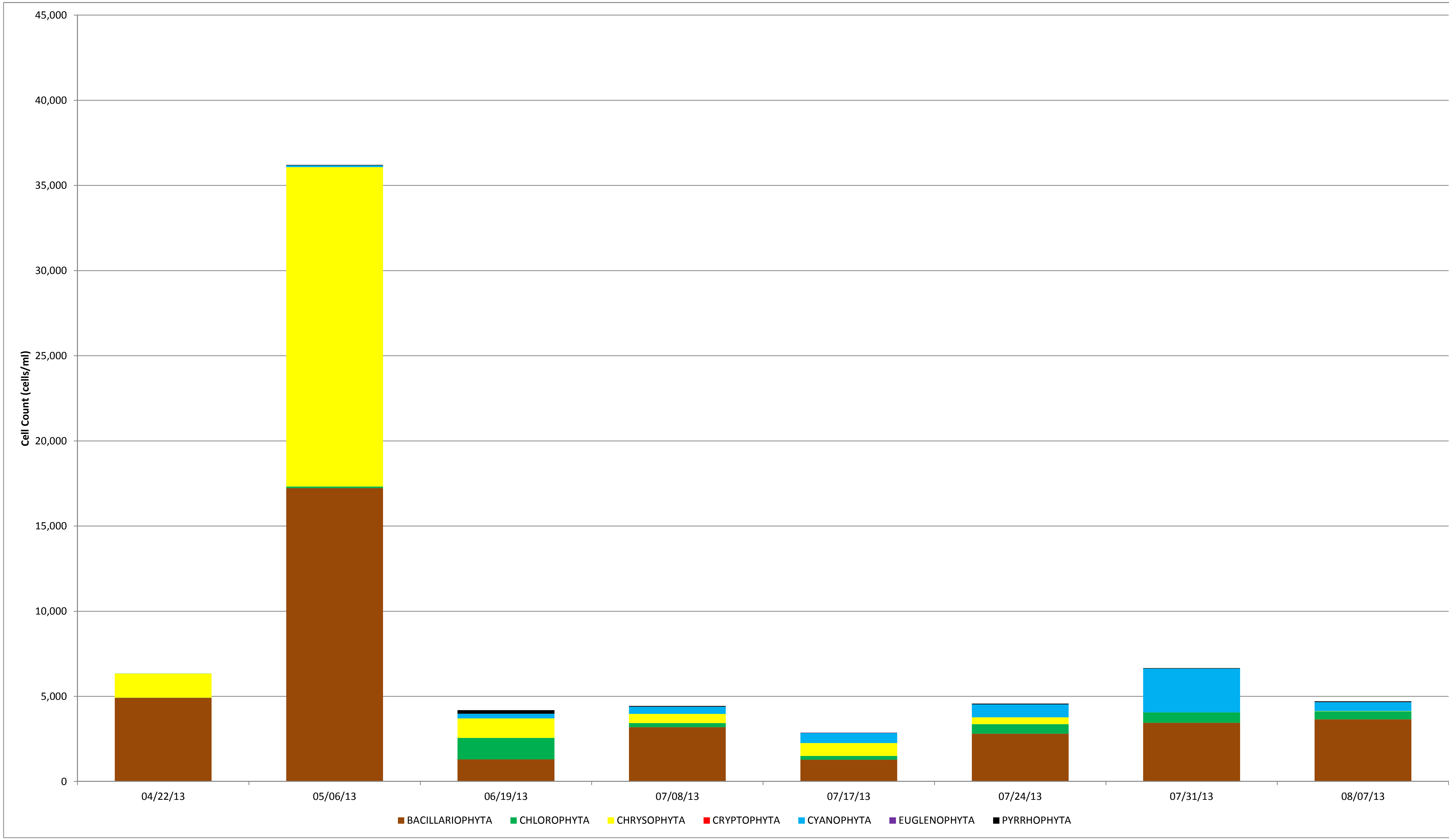

Figure 2-32
Site 31 Core Algae Cell Count

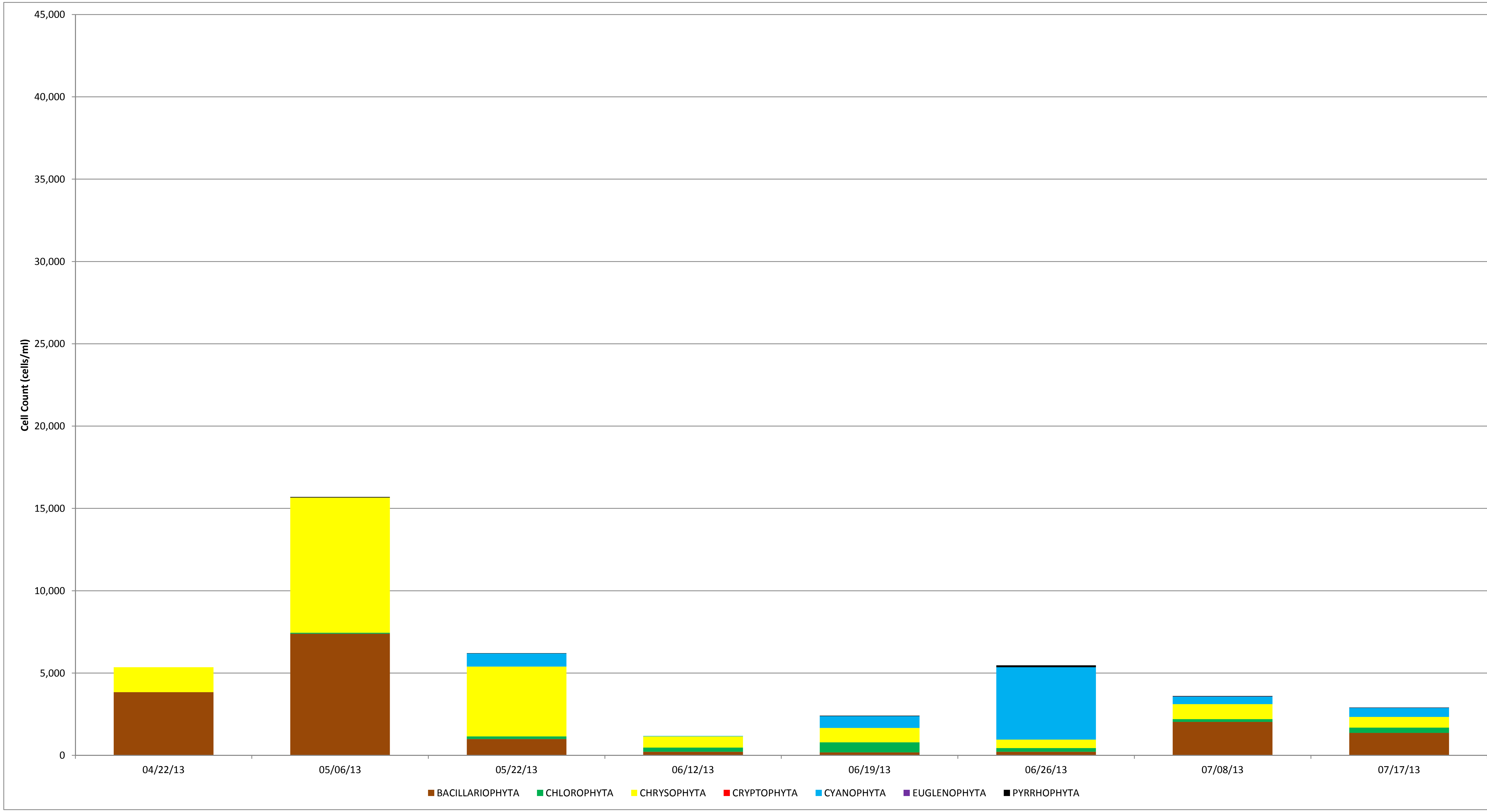

Figure 2-33
Site 32 Core Algae Cell Count

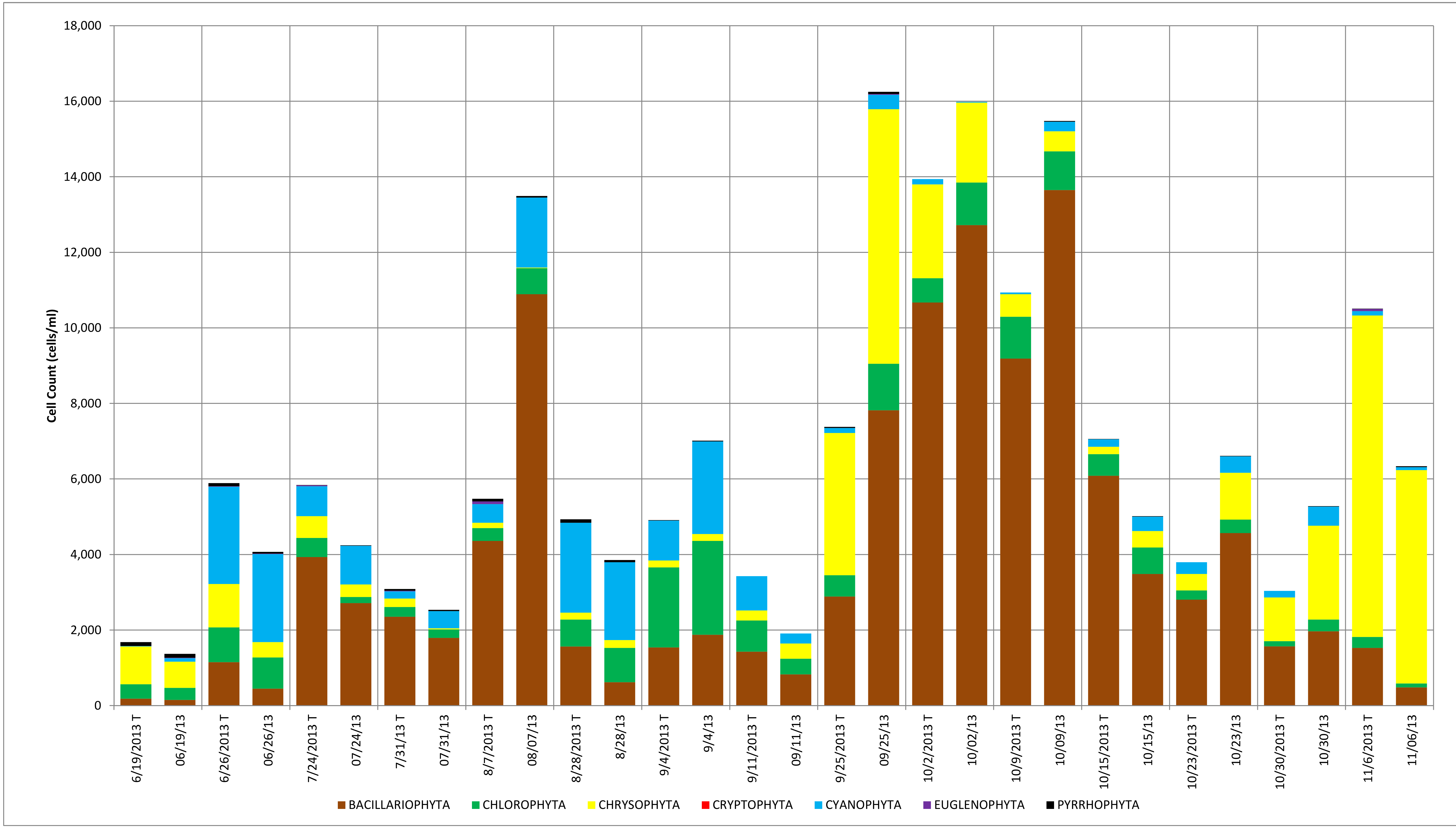

Figure 2-34
Deep Hole Core & Thermocline Cell Count Comparison

A layer of algae sitting at the thermocline would be illustrated by significantly higher cell counts in the "T" samples when compared with core samples taken on the same date. The data presented in Figure 2-34 indicate that there is almost never a significantly higher mass of algae in the thermocline core. There are a few instances when this occurs with any significance. One is in November, where the golden-brown (*Synura)* and diatoms appear to be at a higher concentration in the thermocline core sample, and another in late June with an apparent concentration of diatoms that is higher in the thermocline core sample. Often, however, this trend is reversed, with a higher concentration of cells contained within the core sample.

2.6 Discussion of 2013 Data

This section discusses the data collected in 2013 in relation to data collected in prior years. First, it is appropriate to re-visit the analysis in the Phase 1 report on the relative magnitude of internal and external loads, and then present a new analysis on the relative strength of stratification.

2.6.1 Internal/External Loads

Phosphorus in Lake Auburn ultimately originates as an external load – either from the watershed or the atmosphere (the latter is usually a small portion of the total load to a lake). Once in the lake, phosphorus remains in the water column, settles to the bottom of the lake as dead organisms, or exits through discharges at the outlets or withdrawals at the water treatment plant. Internal loads arise primarily when phosphorus in the sediment is released when the sediment-water interface is devoid of oxygen.

Analysis of the relative magnitude of internal and external loads to determine their contributions to the current phosphorus concentration in Lake Auburn informs decisions about future lake management. Order of magnitude estimates of internal load were included in the Phase 1 report (CDM Smith, 2013), which also used CEI's estimates (CEI, 2013a) of watershed load to make relative comparisons. The Phase 1 report concluded that external loads contribute most of the phosphorus to the lake, but that the increase in phosphorus in the surface layer of the lake could be explained by regeneration of phosphorus from sediments under anoxia or transport of phosphorus to the surface waters with *Gloeotrichia* and was not likely due to interannual variation in watershed load.

Next, an update of the Phase 1 analysis is presented with additional estimates of external loads from measured tributary data in 2013, and internal loads with values to represent the anoxic conditions seen in 2013. The updated analysis has a different construct than in Phase 1:

- Define a maximum acceptable baseline load to the lake based on a growing season average total phosphorus concentration. For this analysis, we use 10 µg/l, which is an accepted upper threshold for high quality lakes.
- Determine the current growing season average total phosphorus concentration from the lake's surface water.
- Refine estimates of internal loads using 2013 data.

- Compare the current epilimnetic total phosphorus levels to those defined as acceptable, commenting on the source of the loads and the ability to control them.

2.6.2 Permissible Watershed Load

A maximum permissible load that will maintain an average lake water phosphorus concentration of 10 µg/l can be estimated using empirical loading models, which do not differentiate between internal and external load sources. Perhaps notable[2] is that 10 µg/l is also the growing season average epilimnetic total phosphorus value for 2005, the only historic year with a detailed set of phosphorus data.

Empirical load models used for this analysis are described in Section 3.5.1 of the Phase 1 report, and consist of the following: Reckhow 1977, Vollenweider 1975, Larsen-Mercier 1976, and Jones-Bachmann 1976. Table 2-9 provides load estimates from the Phase 1 report that were recomputed to incorporate additional data and information collected in 2013.

Table 2-9
Estimated Watershed Loads Required for Average Epilimnetic Total Phosphorus Concentrations of 10 µg/l and 13 µg/l

Empirical Model	Estimated Permissible Watershed Load at 10 µg/l (kg)	[redacted]	Load Required for 13 µg/l (kg)[1]	Difference over Permissible (kg)
Kirchner-Dillon 1975	1,057	[redacted]	1,460	404
Vollenweider 1975	675	[redacted]	1,051	376
Reckhow 1977 (General)	1,338	[redacted]	1,740	401
Larsen-Mercier 1976	738	[redacted]	959	221
Jones-Bachmann 1976	1,134	[redacted]	1,474	340
Model Average	*988*	[redacted]	*1,337*	*348*

Note: 1. 13 µg/l is the 2013 average epilimnetic total phosphorus concentration

This estimate indicates that an average of 988 kg/yr (or about 1,000 kg/yr) of phosphorus is the maximum permissible load from all sources (external and internal) where the lake concentration will remain at 10 µg/l, and that the estimates range from about +/- 300 kg/yr.

[2] 2005, however, may not be a typical year. It had the highest total precipitation since 2001, nearly 17% larger than the average precipitation in that period

2.6.3 Epilimnetic Phosphorus Concentration in Recent Years

Average total phosphorus in the epilimnion in 2012 and 2013 is approximately 13 µg/l.

Table 2-9 provides also provides the estimate of the load that would be needed to achieve this concentration in the lake.

2.6.4 Estimate of Potential Internal Load

In the Phase 1 report, two methods were used to estimate the internal load of phosphorus in 2012 from sediment release under anoxia: the anoxic factor and use of the data on iron-bound phosphorus from sediment analysis. For 2012, these methods yielded 200 to 800 kg of phosphorus released based on the anoxic factor calculation and 300 to 1,200 kg based on the use of iron-bound phosphorus data.

Estimates of the potential load from anoxic release of phosphorus can be calculated using the anoxic factor method (Nürnberg, 1995):

$$P = [\text{Release Rate}]{*}AF * A_o$$

where P is the phosphorus load in mg/yr, the release rate is the phosphorus release rate under anoxic conditions in mg/m^2/d, AF is the anoxic factor, and A_o is the lake surface area in m^2.

A commonly applied load estimate for phosphorus release rate from sediments under anoxia is 12 mg/m^2/d (Wagner, 2014). Furthermore, data from other lakes show that 10 – 40% of the bottom water phosphorus becomes mixed into the upper waters. For this analysis it was assumed that 10% of phosphorus was released due anoxia in the deep waters of the hypolimnion and 20% was released for the region where the discontinuous anoxic region in the thermocline observed in 2013 contacted the lake's bottom. The higher value reflects the greater potential for phosphorus regenerated in the thermocline to be mixed into the upper waters.

We computed the potential phosphorus load using the anoxic factor for each year (see Section 2.3.1). Table 2-10 shows computed internal phosphorus load estimates for years with observed anoxia. These computed loads include both the estimated total mass able to be released from the sediment and the estimated mass to be mixed into the upper waters.

Table 2-10
Estimated Potential Phosphorus Load due to Internal Recycling

Year	Date of Onset	Total Days of Anoxia	AF (days/yr)	Potential Phosphorus Release to Bottom Water (kg)	Potential Phosphorus Mixed into Upper Water (kg)
1993	10/12	8	0.1	12	1
1994	8/5	83	2.3	248	25
2000	10/2	25	1.2	130	13
2002	9/20	35	5.7	623	62
2006	10/4	13	0.3	29	3
2010	10/14	6	0.3	31	3
2011	8/31	77	18.1	1,984	198
2012	8/23	70	18.9	2,077	208
2013 Total		59	9.9	1,090	140[1]
Metalimnion	*9/9*	*20*	*2.8*	*310*	*60*
Hypolimnion	*9/30*	*20*	*7.1*	*780*	*80*

Note: 1. Potential phosphorus mixed into upper waters is computed using a transfer rate of 20% during the metalimnetic minima that occurred from 9/9 to 10/9 and 10% during the period of hypolimnetic anoxia from 10/10 through 11/8.

The results in Table 2-10 indicate that regeneration of phosphorus from sediments could have resulted in 200 – 300 kg/yr reaching the upper layers of the lake in the last several years. These estimates are at the lower end of the range presented in the Phase 1 report.

In addition to the AF, another way to look at the relative extent and severity of anoxic conditions is to compute the total sediment area exposed to low oxygen conditions. This sediment area was computed for 2006 through present day and is presented in Figure 2-35.

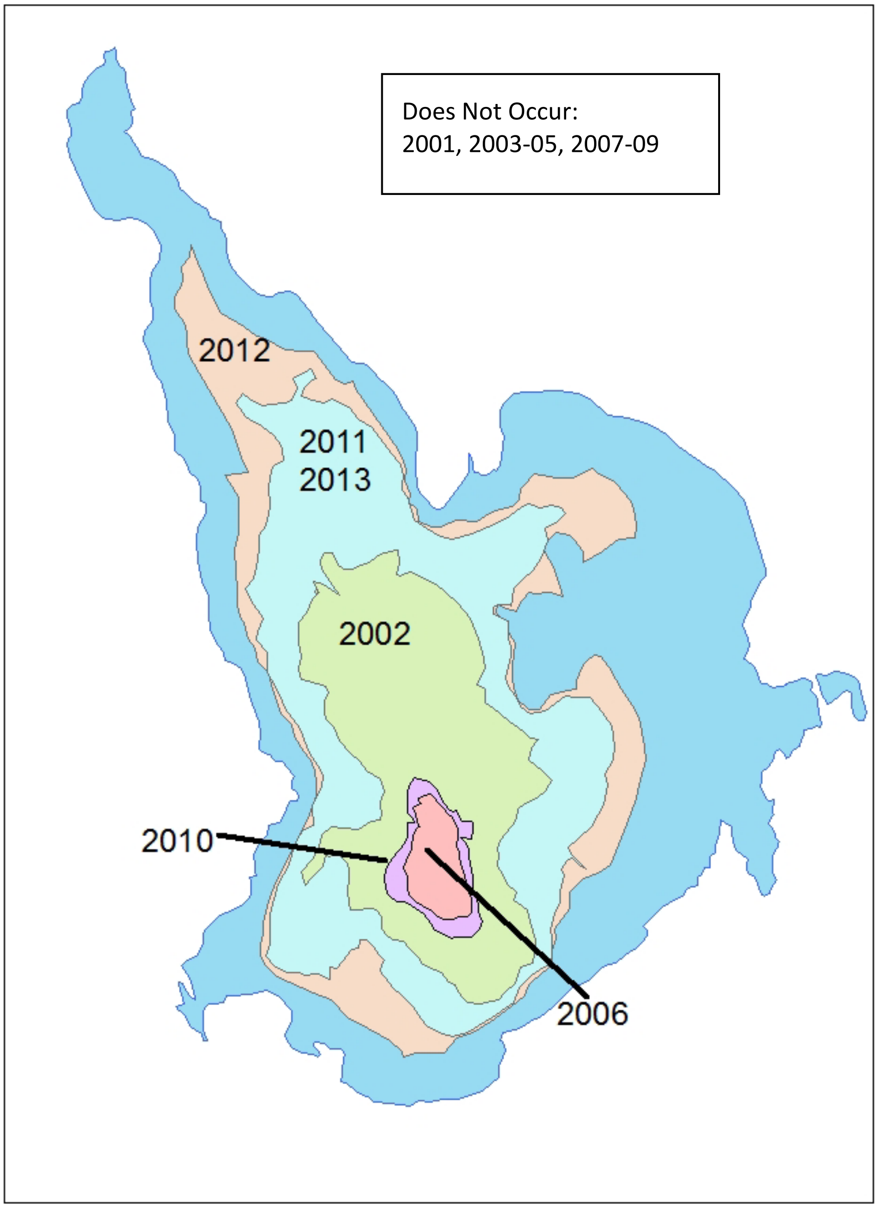

Figure 2-35
Areal Extent of Anoxic Conditions, 2006 through 2013

The surface area in Figure 2-35 indicates that the sediment area underlain by anoxic waters in 2013 was equivalent to 2011 and less than 2012. While the 2013 AF was significantly less than the 2011 AF due to the delayed onset of anoxic conditions throughout the entire hypolimnion, the relative area covered by anoxic conditions was similar. This figure also reiterates the result of the AF calculation, showing that the extent of anoxic conditions in 2013 was among the highest on record, and was greater than the largest extent observed prior to 2011.

2.6.5 Source of the Load

Average 2013 phosphorus levels in the surface water of Lake Auburn exceeded the selected permissible level (13 versus 10 µg/l). Based on empirical loading models, this additional 3 µg/l of total phosphorus, on average, represents an additional 348 kg of total phosphorus in the lake.

The order of magnitude estimators shown here and in the Phase 1 report indicate that both internal and external loads can contribute significantly together or solely to explaining this increase, though the most likely explanation is that increased internal loads are largely responsible. Using the refined calculations presented in Table 2-10 for 2013 and assuming that the watershed load remains relatively constant based on the average watershed loads (CEI, 2013a); the total load in Lake Auburn was estimated to be approximately 75 percent from watershed (external) loads and 25 percent from internal loads.

Because the 2013 phosphorus measurements in the two main tributaries to Lake Auburn did not suggest significant uncontrollable loads discharging to the lake, it will likely be difficult to make a large reduction in watershed load (though it is very important to maintain and strengthen the existing watershed program to decrease the external load of phosphorus). Thus, if future conditions in the lake require implementation of a scheme to reduce phosphorous levels to control algal growth and turbidity, the most effective – in terms of ease of achieving a significant reduction and immediacy of result – approach would be to implement a measure to control the internal load from the sediments.

2.6.6 Relative Strength of Stratification

Numerous recent studies in the literature have discussed the correlation between climate variation and upper water temperature variation within northern hemisphere temperate lakes. These studies have all noted an increase in overall air temperature since the 1980s that has driven an increase in upper water temperature and an increase in the strength of stratification (Stainsby *et al.,* 2011; Adrian *et al.,* 2009; North *et al.*, 2014). Studies have also found that phytoplankton community structure can be affected by increased strength of stratification. An example of this comes from a recent study conducted on Lake Tahoe that found that smaller and motile phytoplankton taxa were much better suited to the stronger stratification conditions that have been observed since the mid-nineties (Winder and Hunter, 2008).

Lake Auburn has an especially comprehensive temperature dataset in terms of breadth, with records of intake temperature dating back to 1951. It should be noted that temperature data prior to 2001 are of questionable accuracy as the quality of data collection improved significantly after this date. Nonetheless, general trends in intake temperature indicate an overall increase in average and late fall lake temperature. The intake is located within the well mixed upper waters, so it is reasonable to assume that this temperature is representative of the upper water temperature. Although the intake was moved to deeper waters in 1996, the intake was still located in the well mixed upper waters and should therefore be of a similar temperature to the new, deeper intake. Any difference due to the new intake location is likely to be outweighed by the annual variability in lake temperature, and even if the new intake were colder, the conclusion reached here would not change. Figure 2-36 shows a time history scatterplot of average annual temperature at the intake from 1951 to 2013.

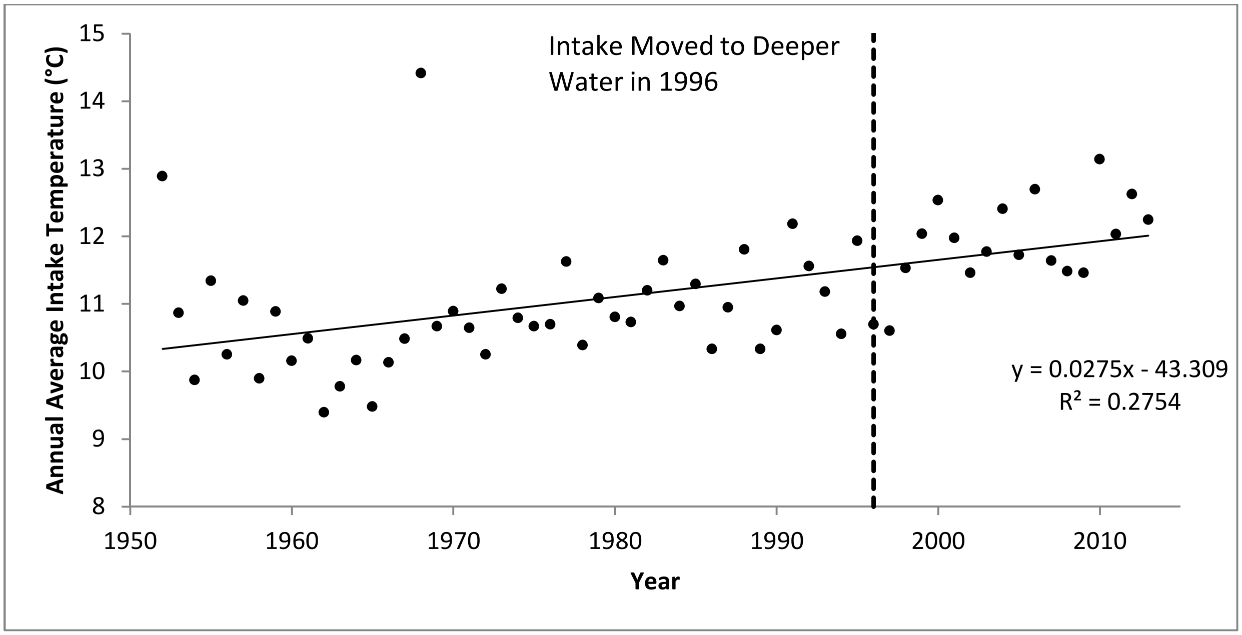

Figure 2-36
Annual Average Intake Temperature, 1951 – 2013

The relationship shown in Figure 2-36 is statistically significant and shows an overall trend towards warmer in lake temperatures. Note that some of the outliers in the 1950s and 1960s may result from questionable quality of the data. To determine whether certain months drive this general increase in average annual lake temperature, a linear regression was fit to the individual monthly averages for each year; the slope and correlation coefficient (R^2) for each month is shown in Figure 2-37.

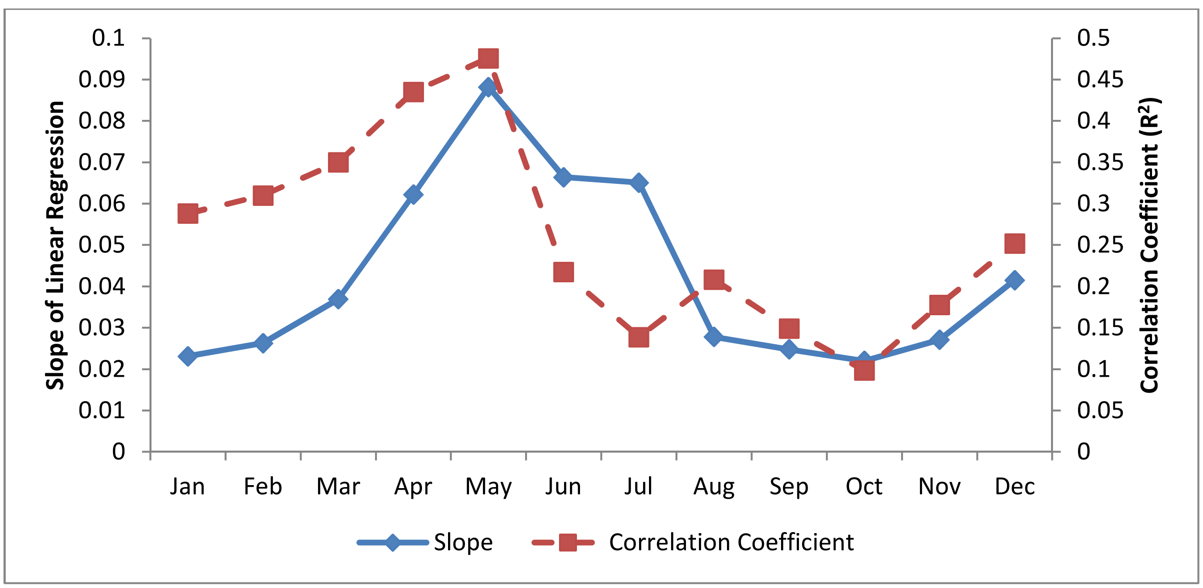

Figure 2-37
Linear Regression Parameters by Month for Long-term Intake Temperature, 1951 – 2013

The linear regression indicates an overall relative increase in monthly temperature across the period of record. A larger slope implies that the increase is more pronounced than it is during months with a smaller slope. Higher correlation coefficients imply a stronger relationship between year and lake temperature. Therefore, the data in Figure 2-37 indicate that the general year over year temperature increase occurs in all months and that the principal months driving the average annual temperature increase are April, May, June, and July. These months are also among those with the strongest correlation between year and temperature. This information is consistent with results reported in the literature describing temperature trends in northern hemisphere lakes.

Different temperatures within the upper and lower layers of a lake create a density driven stratification. As the temperature of the upper water warms, the density – or temperature – difference between the upper and lower layers increases. The cold bottom water is not warmed by the sun, so it remains cold and poorly mixed throughout the summer months. This causes the lake to be more resistant to mixing because it takes more energy to bring the cold, heavy bottom water up into the warmer upper water.

A commonly used metric to assess the overall strength of stratification is the Schmidt stability index, a number that represents the overall amount of energy required to mix the lake. The Schmidt stability index, *S,* is computed by summing the difference between the bottom water and upper water density per unit area. A large *S* implies that there is a large difference between the density of the top and bottom water and therefore a strongly stratified structure exists. Similarly, a small *S* implies that the density is close to uniform throughout the water column, implying weak stratification. The Schmidt stability will be zero following fall turnover when the lake is fully and uniformly mixed across the entire water column (Stainsby *et al.,* 2011; North *et al.,* 2014; Idso, 1973).

The Schmidt stability index was computed using Lake Analyzer (Read *et al.,* 2011) for each temperature profile collected from the deep hole site in Lake Auburn from 1981 to 2013. Peak stratification strength occurs in July and August when the upper waters are at their warmest. Therefore, we compared the monthly average Schmidt stability index for July, August, September, and October to see if any significant trends exist related to increased stratification strength. A scatterplot with monthly linear regression lines is presented in Figure 2-38.

Each of the monthly average Schmidt stability index computations shown in Figure 2-38 show a general positive year over year increase, suggesting that the strength of stratification has increased significantly since the early 1980s. This result agrees with published Schmidt stability index trends from the Lake of Zurich in Switzerland between 1972 – 2010 (North *et al.,* 2014) and Lake Simcoe near Toronto, Canada between 1980 and 2008 (Stainsby *et al.,* 2011).

The potential implications of stronger stratification for Lake Auburn are not well known but could include longer duration anoxia resulting in additional release of phosphorus from the lake's sediments and shifts in the algal population. This would also have to be considered during the design of an aeration/oxygenation system should this in-lake management option be pursued.

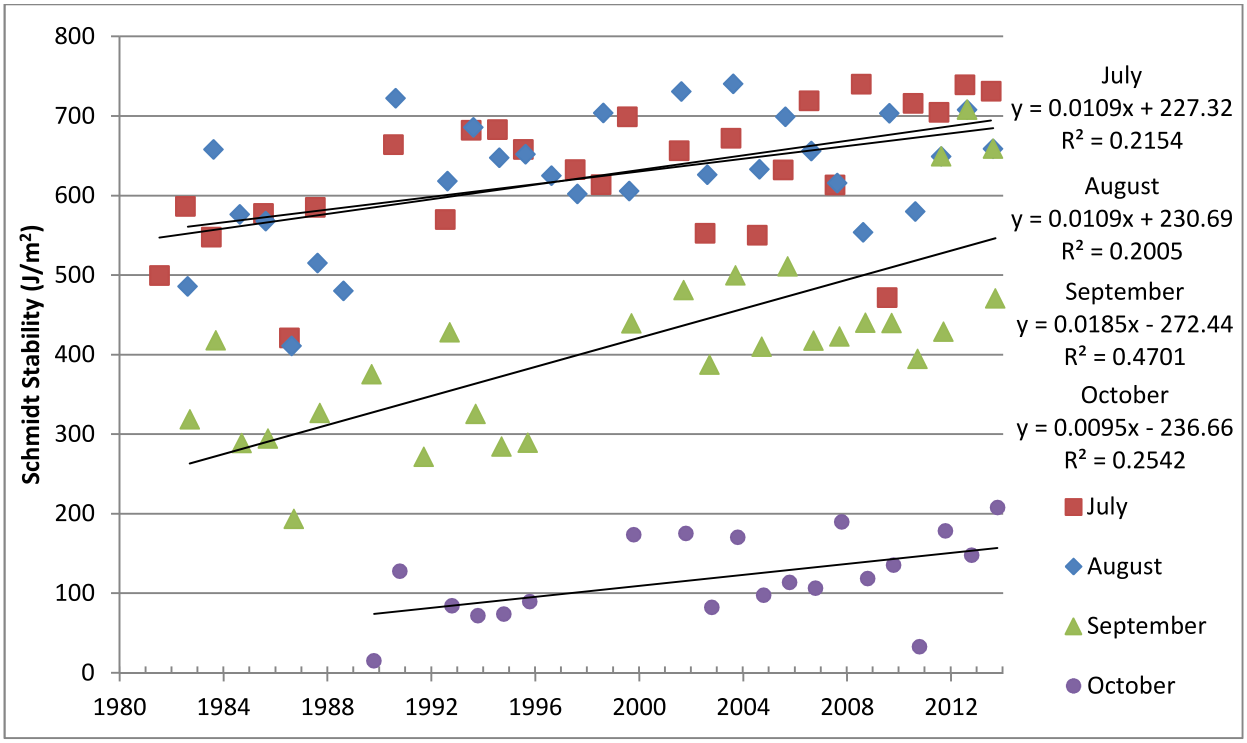

Figure 2-38
Monthly Average Schmidt Stability Index for July – October Computed from Temperature Profiles at the Deep Hole Sampling Site, 1981 – 2013

2.7 Overall Water Quality Assessment

The 2013 water quality demonstrated significant improvement compared with the long duration basin-wide anoxia, significant blue green algal blooms, and the fish kill observed in 2012. In 2013, anoxia set in later and at a deeper depth resulting in an anoxic factor of 9.9, compared with anoxic factors over 18 for both 2011 and 2012. Blooms of algae in the late summer and fall again affected the turbidity and Secchi depth readings, though not to the extent as in 2011 and 2012. The dominant algae in 2013 were diatoms and golden browns; there were no significant blooms of blue-green algae in 2013, as there were in 2011 and 2012. Thus, algal conditions did not warrant the application of algicide to control their impact on turbidity.

In 2013, watershed loads could be estimated from a robust set of dry and wet weather measured total phosphorus levels at Townsend Brook and the North Auburn Dam. The data suggest watershed loads alone would support excellent lake water quality. Further, the concentration data are not indicative of uncontrolled watershed loads.

While 2013 did suggest improving water quality for some parameters, others indicated less improvement. Epilimnetic phosphorus in 2013 was similar to levels found in 2012, as was the number of days when maximum turbidity exceeded 1 NTU. It is also worth noting that the improved water quality conditions in 2013 were still degraded when compared with conditions prior to 2011.

Key points describing and summarizing 2013 water quality are:

1. For the first time on record, a region of very low (anoxic) oxygen formed at mid-depth in the lake prior to similar low oxygen levels occurring in the deepest water in the lake. This mid-depth dissolved oxygen minima occurred earlier (by about a month) and was discontinuous from the layer of low oxygen conditions that form in the deep waters of the lake in some years. Note that the data record includes similar occurrences of discontinuous low dissolved oxygen concentrations in previous years, but these did not reach anoxic levels nor did they precede the development of low oxygen in the deepest waters at the bottom of the lake.

   This exposed an area of sediment between 9- and 15-m depth to low dissolved oxygen water. This means that some portion of the sediment, possibly in the photic zone, was in contact with low oxygen water while the deep hole still had oxygen. This region would also have a greater potential to mix with upper layers of the lake adding to enrichment there.

2. Daily average turbidity in 2013 remained elevated when compared with long-term average values prior to 2011. Daily average turbidity, however, occurred over a lower range with much lower peak values than in 2011 and 2012 (2013 peak daily average turbidity was 1.55 NTU compared to 2.94 NTU in 2011 and 3.2 NTU in 2012. The same is true for the maximum daily turbidity, which was 2.62 in 2013 compared to 4.41 NTU in 2011 and 3.55 NTU in 2012. In contrast, the number of days where maximum daily turbidity in 2013 was only slightly reduced from 2011 and 2012. This indicates that a larger population of algae was present in the lake in 2013 than in the pre-2011 period.

3. Upper water total phosphorus was elevated in the springs of 2012 and 2013 compared to previous years. While order of magnitude estimates have a broad range, it appears likely that this is mostly attributable to the increase of internal load of phosphorus to the lake (as opposed to increases or interannual variation in external loads). Furthermore, the year-over-year increase in upper water total phosphorus noted in the Phase 1 report (2005, 2010, 2011 and 2012) appears to have slowed and stabilized, with the overall average 2013 total phosphorus values not significantly higher than those measured in 2012. Based on flushing rate alone, it would take many years for the tributary loads to reduce to the sum of the current tributary and internal load to achieve upper water total phosphorus levels measured in 2005.

The Phase 1 report also considered the question of whether the water quality changes experienced in Lake Auburn since 2010 are part of a short- or long-term trend. In the tables below, the factors from the Phase 1 report are reconsidered in light of additional data obtained in the 2013. Together the tables indicate that the factors suggesting a long-term trend that will continue to degrade water quality showed improvement in 2013. There were also no unusual meteorological events to drive degraded water quality in 2013.

Evidence for Long-term Trend

Factor	Change in 2013 compared to 2011-12
Year-over-year increases in daily maximum values of turbidity above 1 NTU	Similar to 2012
Shallow Secchi depth readings in fall as a surrogate for large algal blooms	Mixed result. Early August brought new record shallow Secchi depths (algal counts indicate the community was principally associated with 5 types of diatoms, with smaller numbers of green and blue-green algae) but fall Secchi values were generally lower as the large blooms of blue green algae that occurred in 2011 and 2012 did not occur in 2013
Increase in extent and duration of anoxia	Improved; anoxic factor was half of 2011 and 2012 values. However, 2013 had new discontinuous mid-depth anoxic layer develop that in late August; this has the potential to deliver total phosphorus closer to the photic zone
Increase in total phosphorus concentrations in surface waters	Stabilized

Evidence of Near-term Drivers of Change

Factor	Change in 2013 compared to 2011-12
In-lake total phosphorus concentration resets each spring during clear water period	2013 also experienced a period of low total phosphorus in the epilimnion though the data suggest the concentration was somewhat higher than previous springs
Each year 2010-12 had an estimated large delivery of phosphorus load in the spring when algae was ready to use it	This did not occur in 2013 which had no unusual meteorological forcing factors
2012 presented ideal growing conditions for blue green algae (drivers were record warm lake water, clear skies, and large TP load in June)	Large blue-green algal blooms did not occur in 2013, perhaps because no similar drivers occurred
Early onset and ultimate extent of anoxia partly driven by warmer water temperatures that hold less oxygen	2013 was a typical year for water temperature
Anoxia reaches top of hypolimnion where mixing can better entrain increased bottom water phosphorus into the surface water	This occurred in 2013, but for a shorter amount of time. However, in 2013 an anoxic metalimnetic dissolved oxygen minima occurred which may deliver more internal load to the upper waters than is possible with the deeper anoxic phosphorus release

2.7.1 Are Degraded Conditions Likely to Continue?

Finally the factors identified in the Phase 1 report are considered as indicators of continued worsening conditions for Lake Auburn water quality to address whether degraded conditions are likely to continue.

1. High spring total phosphorus in the surface water, which would fuel early algal growth and result in early high turbidity values;
2. unusual warm temperatures, which would strengthen stratification and possibly the magnitude and extent of oxygen depletion in the bottom water, and promote blue-green algal blooms; and
3. large intense storms during algal growth periods increasing external loading of phosphorus to the lake.

Of these factors, there was only evidence of the first one in 2013. Spring total phosphorus concentrations were slightly elevated compared to previous years, and this did result in a bloom of golden brown algae and 10 occurrences of maximum daily turbidity above 1 NTU in May, which was much higher than the previous record of two in May 2007.

While there are improvements in the water quality as noted at the start of this section, it is too soon to know how they will play out. Lake Auburn has a 4.8-year residence time (Dudley, 2004) and additional total phosphorus mass that became present in the lake in 2011 and 2012 would require nearly a decade to be reduced in concentration to levels that result from watershed loads alone. In rough numbers, the average concentration in Lake Auburn in 2005 was about 10 µg/l, while in 2013 it was 13 µg/l. Assuming the watershed delivered a total phosphorus load that resulted in a 10 µg/l concentration in the lake's surface water, and using only dilution of the lake with the watershed (thus ignoring addition internal load), the fully mixed lake concentration after 10 years would be about 10.5 µg/l.

Looking to the future, AWD/LWD need to watch for signs of degraded water quality and consider the need to implement additional management measures. Two key questions to be addressed are (1) will the lake's phosphorus levels in the surface water increase, stabilize or decrease from where they are today, and (2) is the algal community that develops each year from these phosphorus levels acceptable from the point of view of treatability, aesthetics, and risk for losing the filtration avoidance waiver?

Section 3

Recommendations for Long-term Management

3.1 Review of Phase 1 Findings and Subsequent Actions

The Phase 1 report presented an analysis of available data to address what caused Lake Auburn – a lake with a long record of excellent water quality – to experience a rapid change in water quality in 2011 and 2012. The analysis found that:

- Excessive algal growth in late summer/early fall of blue green algae in 2011 and 2012 was responsible for the increases in turbidity that approached the 5 NTU criterion to maintain the filtration avoidance waiver. *Gloeotrichia*, which has only recently been found in Lake Auburn, did not directly contribute to turbidity increases.

- The 2012 fish kill was caused by a warmer than typical surface water temperatures and a bottom layer devoid of oxygen, which eliminated adequate habitat consisting of sufficiently cool water with enough oxygen to survive.

- Total phosphorus concentrations in the surface water of the lake have increased in each year since 2010 over a "baseline" dataset in collected in 2005[1]. Estimates were made of the source of additional phosphorus, which found that:

    - Interannual variation in watershed load was unlikely on its own to have been responsible for the increase in surface water phosphorus levels.

    - Increased occurrence of low oxygen in the lake's bottom waters resulting in release of phosphorus from the sediment could explain the increase.

    - The increased occurrence of the alga *Gloeotrichia*, whose life cycle starts in the sediments before rising to lake's surface water could enrich the surface waters with phosphorus; however, the increase in phosphorus follows the bloom of *Gloeotrichia* (it is not coincident with it).

    - The large *Anabaena* blooms of 2011 and 2012, whose life cycle also starts with growth from sediment dwelling cysts, could not alone explain the increase.

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[1] Two items are important to note about this "trend." First, 2005 may not represent a baseline year as its annual rainfall was the highest of the last decade. Thus the concentrations may be higher due to an expected much higher watershed load that would have been delivered that year. Second, insufficient total phosphorus data prior to 2005 or for 2006-09 exist to determine whether concentrations in the lake may have trended outside of the years discussed.

- An unusual weather event in each year was a notable contributor to the rise in phosphorus levels in the lake, which fueled the late summer-early fall algal blooms:
    - In 2011, the event was the likely mixing of the lake in late August due to the passage of Hurricane Irene which would have released phosphorus from the deep waters in the lake to the surface waters when conditions were excellent for algal blooms, and
    - In 2012, near-record or record-high lake water temperatures from late 2011 through early summer 2012 combined with 2.5 times average June rainfall created excellent conditions for algal growth throughout the remainder of the summer/fall in 2012.

The analysis in the Phase 1 report also looked to determine if the changes in water quality were part of a short- or long-term trend. The report concluded that the data were insufficient to support a firm conclusion.

Overall, the analysis in the Phase 1 report concluded that while the degraded water quality was a serious concern, and watershed management programs needed to be continued and strengthened, an immediate in-lake management measure did not need to be implemented. Instead, short-term management actions were recommended including implementing an enhanced monitoring program in the watershed and lake, continuing and strengthening watershed management, and obtaining permits to allow for application of algicide to control nuisance algal blooms as a short-term contingency plan if turbidity levels were elevated as in 2011 and 2012, which could, in turn, necessitate further in-lake management actions.

Following the Phase 1 report, AWD/LWD began implementing the recommendations including obtaining a permit to apply algicide (other actions are documented in Section 1). This permitting process included an alternatives analysis for wide range of watershed and in-lake management options with respect to their ability to reduce or prevent future algal blooms, episodes of high turbidity and to protect the coldwater fishery. The alternatives analysis evaluated their applicability to Lake Auburn (see Appendix A for a table of all the options considered including a more in-depth discussion of the mode of action, advantages, disadvantages, and applicability of each management approach considered) and resulted in:

- A confirmation of the need to continue with a strong watershed management program, and
- An evaluation of each action’s feasibility and applicability for Lake Auburn, shown in Table 3-1.

Table 3-1
Evaluation of In-lake Management Options for Lake Auburn

Alternative	Feasibility/Applicability
Circulation and destratification	Not Applicable
Drawdown	Not Applicable
Dredging	Applicable
Selective withdrawal	Potentially applicable
Hypolimnetic aeration or oxygenation	Applicable
Phosphorus inactivation	Applicable
Dilution and flushing	Not applicable
Algicides	Applicable
Enhanced grazing	Potentially applicable

3.2 Analysis of Long-term In-lake Management Alternatives

Building on the Phase 1 findings, the initial screening of in-lake management options, and the analysis of 2013 data in Section 2, this section presents the benefits and potential drawbacks and cost effectiveness of each of the most feasible and applicable options: dredging, hypolimnetic aeration or oxygenation, and phosphorus inactivation.

3.2.1 Dredging

Dredging physically removes sediment from the lake bottom. Although there are several different ways to dredge a lake, the only option applicable to large lakes like Lake Auburn is hydraulic removal. Hydraulic removal involves using suction or cutterhead dredges to create a slurry (mixture of water and sediment) that can be hydraulically pumped out of the lake. The slurry is typically dewatered, the water is returned to the lake, and the sediment is disposed on an upland site.

Dredging of Lake Auburn consists of two alternatives that may be applied singularly or in concert with one another. The first alternative involves a deep dredging operation to remove sediment high in iron-bound phosphorus; this would occur in water deeper than about 9 m (30 ft). The second alternative involves dredging shallow water to remove *Gloeotrichia* cysts resting on the benthos prior to recruitment; this would occur in water <9 m deep, with a focus on areas less than 4.6 m (15 ft) deep based on expected relative contribution of *Gloeotrichia* from depth contours based on light penetration.

Although expensive and invasive, dredging is a very effective lake management tool for both removing sediment phosphorus and *Gloeotrichia* cysts prior to recruitment into the water column. It can be truly restorative, allowing complete renovation of the aquatic ecosystem by physically removing the source of internal phosphorus load or the majority of the *Gloeotrichia* population. It can also reduce sediment oxygen demand, if all soft sediment is removed, further reducing the likelihood that anoxic conditions will develop.

Dredging in Lake Auburn is technically feasible, although dredging at the depths required is not typical and specialized equipment would be needed. The sediment to be removed is likely uncontaminated, so disposal costs would be reduced compared with the cost associated with handling contaminated sediments. In addition, the shallow treatment area specified is well within the limits of available technology. However, despite the many benefits, dredging is not a cost-effective method for controlling sediment phosphorus or *Gloeotrichia* in Lake Auburn because the costs for deep dredging (sediment phosphorus removal) or shallow dredging (*Gloeotrichia* removal) are extremely high. Therefore dredging is not a realistic option for Lake Auburn, and is not a recommended lake management approach.

The criteria used to assess the applicability of dredging to Lake Auburn are discussed below.

Benefits of Dredging

Dredging could have a net positive impact on water quality for both the deep and shallow alternatives. Although induced turbidity is minimal with hydraulic removal, it is possible that turbidity could be temporarily elevated during dredging operations.

The deep dredging alternative removes nutrient reserves from the sediments. This reduces the source of internal phosphorus load to the lake, which may result in an immediate decrease in phosphorus release rate during periods of anoxia.

The shallow dredging alternative removes *Gloeotrichia* cysts from the sediment-water interface. The removal of the *Gloeotrichia* cysts should significantly reduce or eliminate the peak *Gloeotrichia* concentration during the growing season following dredging operations.

Dredging can increase and improve spawning habitat for many fish species and remove pollutant reserves from the sediment, improving overall aquatic habitat quality.

Potential Drawbacks of Dredging

Impacts to biota, especially benthic invertebrates, are likely during dredging operations. A survey of benthic invertebrates in Lake Auburn has not been conducted, and would be needed prior to dredging to better understand potential impacts.

Three potential concerns for Lake Auburn could impact the allowable timing for dredging operations: presence of mussels, potential presence of the threatened spotted turtle, and fish reproduction. If mussels are found in the lake, it may be necessary to collect a portion of the population and re-introduce them in the lake following dredging operations. Maine Inland Fisheries and Wildlife's website (MEDIFW, 2013) on the endangered species program lists the spotted turtle (*Clemmys guttata*) as state threatened; protection of this species would need to be considered in planning the dredging operation. Regarding fish reproduction, the timing of dredging may be restricted to prevent removal of demersal eggs. Impacts will also occur to more abundant invertebrates, and while not ideal, these populations are typically characterized by fast reproduction rates so the population levels will not be depressed for a substantial amount of time.

Cost Effectiveness

Dredging is an expensive management technique, especially for a project on the scale of Lake Auburn. While dredging may yield a complete restoration of the aquatic ecosystem, it is typically only cost effective for smaller projects that are more easily managed, or where dredging would provide major additional benefits, such as increased volume and supply capacity where it is currently inadequate. There are many costs associated with dredging: planning and permitting, operation of the dredge, and disposal of the dredged material. Dredging costs cannot be reliably estimated on a per unit volume basis, but an approximate estimate for the unit cost of hydraulic dredging (including disposal costs) in Lake Auburn is $30 per cubic yard of sediment removed.

The deep dredging operation to remove sediment with high iron-bound phosphorus concentration would need to remove sediment between 30 and 50 feet at a minimum. This depth range is chosen because the shallowest depth of anoxia observed in 2012 was 9 meters (30 ft), and sediment iron-bound phosphorus measurements indicate that the peak concentrations occur around 15 meters (50 ft). Assuming that only the top 1.5 feet of sediment had to be removed, this corresponds to 2.1 million cubic yards of sediment, or $63.7 million at a unit cost of $30 per cubic yard ($86 million with 35% for contingencies).

The shallow dredging operation to remove sediment with *Gloeotrichia* cysts would need to remove sediment between 10 and 15 feet, which represents the expected zone from which most *Gloeotrichia* recruitment will occur. While *Gloeotrichia* is likely found in water shallower than 3 m (10 ft), these areas tend to be quite sandy and rocky, making dredging technically difficult, more expensive, and probably not necessary (viable resting stage density on sand and rock should be small). Again assuming that the top 1.5 feet of sediment is removed, this yields 3.9 million cubic yards of sediment or $117.9 million at a unit cost of $30 per cubic yard ($160 million with 35% for contingencies).

Both dredging operations have an exceptionally high cost, so they are not economically viable options for managing sediment phosphorus or the *Gloeotrichia* population in Lake Auburn.

3.2.2 Hypolimnetic (Bottom Water) Oxygenation

Hypolimnetic oxygenation is a technique for management of algae by introducing more oxygen into the water to limit internal recycling of phosphorus, thereby controlling algae. Putting pure oxygen or air into the aquatic system increases oxygen concentration by transfer from gas to liquid, and generates a controllable mixing force. The oxygen transfer function is used to prevent anoxia in the bottom water layer, the hypolimnion. By keeping the hypolimnion from becoming anoxic during stratification, the release of phosphorus, iron, manganese and sulfides from deep bottom sediments is minimized and the build-up of undecomposed organic matter and oxygen-demanding compounds (e.g., ammonium) is decreased. Hypolimnetic oxygenation can also increase the volume of water suitable for habitation by zooplankton and fish, especially coldwater forms.

The success of oxygenation for algal control is largely linked to reducing available phosphorus, although there is recent evidence that reducing the generation of ferrous iron can limit cyanobacteria (Molot *et al*., 2014). Maintaining oxygen at the sediment-water interface, usually with 2 mg/l as a minimum target, will minimize the release of multiple reduced compounds from the sediment and limit algal blooms in the overlying water. Short-term effectiveness may be achieved if oxygen levels near the bottom rise quickly and adequate phosphorus binders are present. Even then, a month or more of lag time might be expected for existing algae to suffer nutrient limitation or other stresses that reduce abundance. The control of phosphorus in surface waters may not be effective until the following year for hypolimnetic oxygenation if there has already been substantial release of phosphorus and other substances from the sediment when the system is turned on.

Types of Hypolimnetic Oxygenation Systems

Three types of hypolimnetic oxygenation systems that involve use of air and three types that use pure oxygen are commonly applied to lakes similar to Lake Auburn.

Of the air-driven approaches, one type is a full lift hypolimnetic oxygenation approach that moves hypolimnetic water to the surface, aerates it, and replaces it in the hypolimnion. Bringing the water to the surface can be accomplished with electric, solar or wind-powered pumps, but is most often driven by pneumatic force (compressed air). Return flow to the hypolimnion is generally directed through a pipe to maintain separation of the newly oxygenated waters from the epilimnion. To provide adequate oxygen, the hypolimnetic volume should be pumped and oxygenated at least every month and preferably every two weeks.

Another air-driven hypolimnetic oxygenation system is the partial lift system, in which air is pumped into a submerged chamber in which exchange of oxygen is made with the deeper waters. The newly oxygenated waters are released back into the hypolimnion without destratification. A shoreline site for a housed compressor is needed, but the oxygenation unit itself is submerged and does not interfere with lake use or aesthetics.

The third air-driven approach involves a process called layer oxygenation. Water can be oxygenated by full or partial lift technology, but by combining water from different (but carefully chosen) temperature (and therefore density) regimes, stable oxygenated layers can be formed anywhere between the thermocline and the bottom of the lake. Each layer acts as a barrier to the passage of phosphorus, reduced metals and related contaminants from the layer below. Each layer is stable as a consequence of thermally mediated differences in density. The whole hypolimnion may be oxygenated, or any part thereof, to whatever oxygen level is deemed appropriate for the designated use.

The first and simplest system using pure oxygen is a diffused oxygen distributor, which involves a source of liquid oxygen (storage tank or oxygen generator), a vaporizer (turning liquid oxygen into gas), distribution piping, and diffuser hose from which the oxygen is released as tiny bubbles. This method involves no power or moving parts, and depends on proper placement of diffuser hose to distribute the oxygen without destratifying the lake. The pure oxygen bubbles are absorbed as they rise, minimizing water movement at or above the thermocline.

The second pure oxygen approach is the Speece cone – a sealed, inverted funnel into which water is pumped from the top and oxygen is released near the bottom. Oxygen bubbles trying to rise are met by low oxygen water pumped in from the hypolimnion of the lake; with proper feed balance, all oxygen is absorbed into the water, which is then distributed by pipe to the target area. These systems require power and pumps, as well as a source of oxygen, but are fairly efficient. Speece cones are usually placed deep in the lake, but could be operated from shore with sufficient piping and pumping power.

The third oxygen-based approach is a sidestream supersaturation system, in which water is removed from the low oxygen target area, oxygenated under pressure to a state of supersaturation, and returned to the portion of the lake needing more oxygen. Like the Speece cone, sidestream saturation systems require pumps, power and an oxygen source, but with careful temperature management, a blanket of oxygen-rich water can be placed right on the bottom with substantial stability, putting the oxygen where it is needed most.

The mechanism of phosphorus control exercised through hypolimnetic oxygenation is the maintenance of high oxygen and limitation of phosphorus release from sediments. To successfully oxygenate a hypolimnion, the continuous oxygen demand of the sediments must be met; this translates into a need to add enough oxygen and distribute it properly, neither of which is an easy task. It is also essential that an adequate supply of phosphorus binder, usually iron, aluminum or calcium, be available to combine with phosphorus under oxic conditions. This is usually not an issue, but where longer term chemical reactions under anoxic conditions have limited binder availability, additional phosphorus binders may have to be added for oxygenation to have maximum effectiveness on phosphorus inactivation.

Benefits of Hypolimnetic Oxygenation

Hypolimnetic oxygenation has experienced varied success. Where oxygen is raised above 2 mg/l at the bottom, success has generally been attained, but achieving oxygenated bottom waters has not always been possible. Putting in adequate oxygen and distributing it throughout the target area can be both costly and an engineering challenge. Another complicating factor is metalimnetic anoxia (as occurred in Lake Auburn in 2013), where organic particles accumulating near the thermocline create an anoxic layer above the oxygenated hypolimnion. Using oxygenated water to create a stable layer instead of aerating the entire hypolimnion can eliminate the problem of metalimnetic anoxia that allows rapid phosphorus recycle and can act as a barrier to fish migration. This can be accomplished with layer aeration or any of the pure oxygen input techniques, as long as a thermally stable layer can be created. Creating a thermally stable layer near the thermocline is not an easy task, but has been accomplished in other lakes.

Any oxygenation system can make a marked improvement in lake conditions, but it should be noted that practical experience has demonstrated that effects are not uniform or consistent within and among aquatic systems. Zones of minimal interaction will often occur, possibly resulting in localized anoxia and possible phosphorus release. Any oxygenation system focusing on deep water or the sediment-water interfaces may allow a band of anoxic water to persist near the top of the metalimnion, allowing nutrient cycling and supply to the epilimnion and discouraging vertical migration by fish and zooplankton. This occurred in Lake Auburn for the first time in 2013, when a “doughnut” shaped impact area with low oxygen water at a depth of 10 to 15 m (30 to 50 feet) contacted the sediment at that depth range, creating a low oxygen ring within the lake. Also, phosphorus binders must be available for oxygenation to facilitate phosphorus inactivation. Uniformity of results should be achievable with careful design and operation, but probably also with increased cost.

Since oxygenation is an active treatment, the system must be kept running year after year, at least during the summer months. It seems plausible that effectiveness can be maintained over many years with this method, but there has been considerable variability in results. For example, the Fresh Pond destratification system in Cambridge, Massachusetts yielded positive results over a period approaching a decade, but has not performed as well in recent years, mainly due to operation only half of each day to save on power costs. As another example, Notch Reservoir in North Adams, Massachusetts also experienced improvement over about a decade with a hypolimnetic oxygenation system driven by air, but power failures allowed low oxygen zones to quickly develop at times. A long-term treatment of Lake Shenipsit, Connecticut with a layer oxygenation method revealed adequate oxygenation of the metalimnion in this 212 hectare (520 acre) lake with compressor systems totaling 60 HP that delivered 240 CFM of air. Total phosphorus was reduced marginally while blue-green algae decreased and the algal community shifted to green algae and diatoms. The Vadnais Reservoir hypolimnetic oxygenation system in St. Paul, Minnesota, improved water quality in that reservoir for over 20 years with air diffused into a chamber, but oxygen transfer never reached the design level and the level of improvement varied among years. That system was replaced in 2013 with a pure oxygen diffusion system.

A number of successful cases of pure oxygen use have surfaced in recent years, all reporting much improved habitat but not all documenting algal changes. In many cases, the watershed nutrient loads were not sufficiently controlled, and the oxygenation was supporting the fish community but not addressing the main sources of nutrients. Where internal recycling has been documented as the primary phosphorus source, these systems have been more effective at controlling algal blooms. The use of pure oxygen carries a higher material cost, but the lack of a power requirement in passive diffusion systems has offset the cost of oxygen.

The Tennessee Valley Authority has the longest history of pure oxygen use, dating to the early 1990s, and has gone through three generations of system refinement, mostly to improve habitat downstream of power producing reservoirs. Over ten diffused oxygen systems were placed in California water supply reservoirs over the last decade, and all are reported to have improved water quality, including a reduction in algal blooms overall and cyanobacteria in particular. Speece cones have produced the desired improvements in multiple documented cases, mostly in the western USA, and sidestream supersaturation has been used successfully in two western cases as well, but these techniques are applied less often than diffused oxygen systems. Lack of power requirement and minimal moving parts makes the diffused oxygen approach very attractive from an operational perspective.

Potential Drawbacks of Hypolimnetic Oxygenation

Very few negative impacts are expected from hypolimnetic oxygenation, but algal blooms may not be controlled if other sources of phosphorus are available. Since oxygen levels are increased in previously anoxic areas, many organisms that require oxygen such as fish, aquatic insects and zooplankton are expected to increase.

The greatest short-term risk from a hypolimnetic oxygenation is system failure after establishing an oxygenated zone. While cessation may not result in worse conditions than encountered before treatment, adjustment of system biota to a return to the low oxygen regime could be a problem. In several cases fish kills were reported in water supply reservoirs when oxygenation systems were shut off by power failures or mechanical difficulties. A return to low oxygen conditions and release of reduced compounds including manganese, iron and phosphorus is very rapid when systems are shut down. Continued operation over several years has been shown to reduce but not eliminate oxygen demand in subsequent years.

Long-term impacts to biota such as zooplankton and fish may occur following any changes in algal abundance or species composition. Usually these changes are beneficial, but not always, and they do represent a new set of conditions to which biota must adapt. Oxygen or nitrogen supersaturation could theoretically become a problem for fish in deep waters during oxygenation due to gas bubble disease, but formation of the harmful size bubbles from oxygenation is not expected. Gas bubble disease is most often a function of creation and entrapment of very fine air bubbles associated with hydropower facilities; oxygenation systems for drinking water reservoirs have not been observed to produce bubbles small enough to induce this disease. Nitrogen supersaturation represents a greater risk than oxygen levels, but no gas bubble disease has been detected in lakes with hypolimnetic oxygenation.

Cost Effectiveness

The cost of hypolimnetic oxygenation systems depends on the amount of oxygen to be delivered and how and where it is delivered. Based on the maximum extent of anoxia at the bottom of Lake Auburn (in 2012), about half the lake area could require oxygenation. The oxygen demand appears to be at the low end of the range where oxygenation is applied. Comparison of case studies for which cost data are available (Table 3-2) indicates that a diffused oxygen system is less costly to install than a hypolimnetic air system; air-driven systems require an in-lake chamber and compressor, while the diffused oxygen system requires neither. Operationally, costs are very similar, with power and oxygen costs roughly offsetting each other.

There is certainly variability among applications on a per acre basis, but a diffused oxygen distributor system would be applicable to Lake Auburn and capital cost would be projected to be about $2,100/acre, or approximately $2.5 million ($3.4 million with 35% for contingencies) for an area of 1,150 acres. The range is substantially higher, however, and for this application a range of +/- 25% would be reasonable.

Operational costs for a pure oxygen diffusion system average slightly less than $400 per acre per year, which would translate into an annual cost of $460,000 ($620,000 with 35% for contingencies), but the cases from which the costs are drawn operate from June into October and have high oxygen demands, while at Lake Auburn it is not expected that more than 60 days of operation would be needed (August-September) and oxygen demand is at the low end for lakes where oxygenation is conducted. Consequently, operational costs are anticipated to be on the order of $100 per acre per year, or about $115,000, with a range of $50,000 in a “good” year to $230,000 in a “bad” year for oxygen demand ($160,000 with a range of $68,000 to $310,000 with 35% for contingencies).

Table 3-2
Comparison of Unit Costs from Case Studies

Oxygenation System Type	Capital Cost/Target Acres ($/acre)	Annual O&M Cost/Target Acres ($/acre/year)
Diffused Air		
Number of Case Studies	7	6
Average	$5,469	$336
Maximum	$8,000	$714
Minimum	$1,623	$44
Diffused Oxygen		
Number of Case Studies	10	10
Average	$2,141	$376
Maximum	$5,710	$1,003
Minimum	$370	$80

3.2.3 Phosphorus Precipitation and Inactivation

As an alternative for reducing algal bloom potential, phosphorus inactivation through an in-lake treatment is highly applicable to Lake Auburn as a long-term control measure. Regulatory acceptability of phosphorus inactivation appears positive in Maine, as this technique has been applied in other Maine lakes in the past. Previous applications of aluminum in Maine have taken considerable planning, an effort not yet conducted for Lake Auburn, but it would appear that a properly justified and planned phosphorus inactivation project could be approved for Lake Auburn. If chosen, additional studies would be required as part of designing the alum application, including additional sediment testing and assays to determine a safe and effective dose.

Phosphorus can be inactivated in either the water column or within surficial sediments. Phosphorus precipitation by chemical complexing removes phosphorus from the water column and binds phosphorus in surficial sediments. This technique can control algal abundance until the phosphorus supply is replenished. Phosphorus inactivation that focuses on phosphorus precipitation from the water column is not very efficient at lower concentrations of phosphorus (<100 µg/l where aluminum is the binder, possibly lower where lanthanum is used), and is therefore not advantageous for many epilimnetic lake situations. It has greater applicability to stormwater management situations, usually through an injection system triggered by rain or increasing flows, and is very effective on hypolimnetic water where phosphorus has accumulated to levels over 100 μg/l.

Phosphorus inactivation of surficial sediments aims to achieve long-term control of phosphorus release from lake sediments by adding enough phosphorus binder to the upper 4-10 cm of sediment to minimize releases, which are usually a function of dissociation from iron-based compounds under low oxygen conditions. This technique is most effective after nutrient loading from the watershed is sufficiently reduced, as it acts only on existing phosphorus reserves, not new ones added post-treatment. In-lake treatments are used when phosphorus budget studies of the lake indicate that the primary source of the phosphorus is internal (i.e., recycled from lake sediments).

Aluminum has been widely used for phosphorus inactivation, mostly as aluminum sulfate (alum) and often in combination with sodium aluminate (aluminate), as it binds phosphorus well under a wide range of conditions, including anoxia. Several other aluminum compounds have been less frequently applied and tend to be much more expensive. Lanthanum has more recently become commercially available for phosphorus inactivation, and may be preferable to aluminum where phosphorus stripping from the water column is the primary intent or where many sensitive organisms are present and toxicity is a large concern. Lanthanum is combined with bentonite clay in the commercially available product (PhosLock), which further acts to seal the bottom upon settling. The theory is sound, yet only a limited track record is currently available for lanthanum, with no treatments in New England as of yet. Results with lanthanum in other regions have generally been positive, but the cost is higher than for aluminum.

Benefits of Phosphorus Precipitation and Inactivation

Commonly applied aluminum doses for sediment phosphorus inactivation range from about 10 to 50 g/m^2, with treatments up to 100 g/m^2 known from New England. Aluminum compounds are added to the water and colloidal aggregates of aluminum hydroxide are formed. These aggregates rapidly grow into a visible, brownish to greenish white floc, a precipitate that settles to the sediments over the following hours, carrying sorbed phosphorus and bits of organic and inorganic particulate matter in the floc. After the floc settles to the sediment surface, the water will usually be very clear. If enough aluminum is added, an initial layer of 1 to 2 inches of aluminum hydroxide floc will cover the sediments, mix with the upper few centimeters over a period of weeks, and significantly retard the release of phosphorus into the water column as an internal load.

Nutrient inactivation has received increasing attention over the last decade as long lasting results have been demonstrated in multiple projects, including several in Maine. Where aluminum has not reduced algal densities, either the dose was inadequate or watershed sources were more important than internal loads. Furthermore, Lake Auburn is not well buffered so precautions would be needed to guard against pH shifts that would adversely affect water quality. Consequently, it is necessary to have a reliable assessment of the relative magnitude of loads and to know the proper aluminum dose. Planning for a phosphorus inactivation treatment requires substantial lead time.

Numerous examples exist of successful aluminum treatment in lakes throughout New England. Annabessacook Lake in Maine suffered algal blooms for 40 years prior to the 1978 treatment with aluminum sulfate and sodium aluminate. A 65% decrease in internal phosphorus loading was achieved, blue-green algae blooms were eliminated, and conditions remained much improved for about 30 years; conditions are still improved over pre-treatment status, but a gradual decline has been noted. Similar results have been obtained in Cochnewagon Lake in Maine. Kezar Lake in New Hampshire was treated with aluminum sulfate and sodium aluminate in 1984 after a wastewater treatment facility discharge was diverted from the lake. Both algal blooms and oxygen demand were depressed for several years, but began to rise more quickly than expected. Additional controls on external loads reversed this trend and conditions have remained markedly improved. No adverse impacts on fish or benthic fauna have been observed despite careful monitoring.

Aluminum sulfate and sodium aluminate were employed with great success at Lake Morey, Vermont. A pretreatment average spring total phosphorus concentration of 37 µg/l was reduced to 9 µg/l after treatment in late spring of 1987. Although upper water phosphorus levels have varied since then, the pretreatment levels have not yet been approached. Oxygen levels increased below the epilimnion, with as much as 10 vertical feet of suitable trout habitat reclaimed. Some adverse effects of the treatment on benthic invertebrates and yellow perch were observed immediately after treatment (e.g., smothering of some invertebrates by the floc layer and poor growth by yellow perch for a season), but these proved to be transient phenomena and conditions have been acceptable and stable for over two decades.

Although some short-term effects have been noted, there do not seem to be any significant negative long-term impacts of phosphorus inactivation. Bioaccumulation of aluminum has not been reported. Reducing algal production might be expected to reduce fish production and increased transparency may allow macrophytes to increase and extend their depth distribution into deeper waters as sunlight penetration increases. However, no dissatisfaction with treatment results has been expressed in the studied cases, but it should be noted that use of aluminum may not appreciably reduce phosphorus levels in the water column. Lake Auburn phosphorus levels are <20 µg/l even during algal blooms, and aluminum would be very inefficient at reducing those levels further. A very high dose of aluminum may be needed (>10 mg/l), which would increase toxicity risk. Lanthanum may be more appropriate in this case, but no testing has been done and there are no similar examples for comparison. Treatment in spring, when phosphorus levels in the water column are often lowest and release from sediments is at a minimum, is the preferred approach for maximizing immediate benefits.

Potential Drawbacks of Phosphorus Precipitation and Inactivation

Lake Auburn contains important populations of salmonids (lake trout or Togue and salmon) and may harbor a wide array of benthic invertebrates, information on which is very limited. Additional studies would be needed to prepare for a possible treatment because the primary drawback to aluminum treatments is that reactive aluminum can be toxic to fish and invertebrates. The safe level of dissolved aluminum is considered to be about 50 µg/l. The amount of reactive aluminum is strongly influenced by pH and is very low between pH values of 6 and 8. A successful aluminum treatment must deliver enough aluminum to the surficial sediments to inactivate most of the iron-bound phosphorus present while keeping the reactive aluminum level in the water column at a low enough level to avoid toxicity, or keeping sensitive organisms out of the treatment area. Sediment testing of available phosphorus and lab assays for both the amount of aluminum needed and the effect of that aluminum dose on fish and invertebrates helps with treatment planning. Where the dose exceeds the toxicity threshold, the dose can be sequentially delivered in smaller amounts and a treatment pattern that minimizes exposure of sensitive organisms can be developed, but there may be some risk of toxicity.

Once reacted, the resultant aluminum compounds are non-toxic and rather stable. Short-term effects are therefore more likely than long-term impacts, and involve aluminum toxicity at low or high pH. In some cases dissolved aluminum concentrations have exceeded the safe level, but in most cases detectable fish and invertebrate kills have been avoided. In low alkalinity Kezar Lake, New Hampshire, dissolved aluminum concentrations were as high as 400 µg/l after application of alum and sodium aluminate, but no fish kills were observed. In Lake Morey, Vermont, dissolved aluminum reached concentrations as high as 200 µg/l in the epilimnion where the pH was near 8.0 SU. Despite the high aluminum concentrations, no direct fish mortality was observed. Careful study of remaining mussels in Mystic Lake during treatment revealed no impacts, not even behavioral changes. Mussels can close their shells and survive for days to a week, by which time toxicity risk is negligible, but in Mystic Lake there was no adverse reaction to aluminum floc and no later mortality. Losses of benthic invertebrates were reported in Lake Morey, but mainly from smothering under the aluminum floc. The eventual incorporation of the floc into the surficial sediments leads to transient impacts on benthic invertebrates.

Fish kills early in the use of aluminum in lakes resulted from lack of buffering. In these cases, the pH dropped to well below 6.0 SU and aluminum toxicity ensued. A fish kill was reported following aluminum sulfate and sodium aluminate addition to low alkalinity Hamblin Lake in Barnstable, Massachusetts in 1995 as a consequence of overbuffering and high pH (values as high as 9.3 SU), leading to aluminum toxicity and possibly pH shock. A kill similar to that at Hamblin Pond occurred at Lake Pocotopaug in Connecticut in 2000, during the early stages of a treatment with a similarly overbuffered mix of alum and aluminate. Fish bioassays documented that the impact was from elevated aluminum and high pH. Altering the treatment protocols with regard to alum:aluminate ratio and maximum aluminum dose to any location on any day resulted in no fish mortality in the lake during completion of the treatment in 2001. Fish kills have become a rare occurrence, however, as dose adjustments and buffering of treatments in low alkalinity lakes have become standard. It is possible to perform treatments on low alkalinity lakes without inducing aluminum toxicity, but there is still a risk.

The precipitation of the floc may also carry many other organisms, such as algae and small zooplankton, to the bottom. Changes in the algal community are expected. However, no studies indicate any major shift in zooplankton immediately following treatment. Data for zooplankton in several Maine lakes treated between 1978 and 1986 and monitored before treatment and just after treatment suggest no adverse impacts on zooplankton community composition, density or mean size. Impacts may well have occurred in the treatment zone, but refuges, resting stages, rapid reproduction and re-distribution act to minimize zooplankton impacts.

Less is known about lanthanum treatments, but the toxicity profile for this compound is more favorable than for aluminum. The primary issue appears to be smothering of benthic organisms by the bentonite clay that is an integral part of the commercially available formulation. Lanthanum seems to better absorb phosphorus from the water column than aluminum, but may not hold the phosphorus in the sediment as well as aluminum from very limited study.

Cost Effectiveness

The cost of phosphorus inactivation is dependent on the inactivator chosen, the needed dose, and distance from suppliers, plus the environmental constraints placed on the application, including dose limitations, application timing, monitoring and contingencies. Application costs have ranged from $1,000 to $10,000 per acre, with the more costly applications linked to precautions relating to sensitive species. For Lake Auburn, a cost of between $2,500 and $4,000 per acre is possible, based on recent treatments in other New England lakes. The exact area to be treated is not known, but could be all the lake area potentially subject to anoxia, which would be about 1,150 acres, just over half the lake area. The area between 10 and 20 meters of depth has the highest available sediment phosphorus levels and has only recently been subjected to anoxia, an area of about 500 acres. Another strategy would be to treat only a portion of the lake in each of successive years. This range of potential treatment area suggests a wide cost range of $1.3 million to $4.6 million ($1.8 to $6.2 million with 35% for contingencies), owing to both uncertain dose and uncertain treatment area. The range would be greatly narrowed by a proper planning effort, but based on a probable average dose of 50 g/m^2, with an average cost of $60/gram Al/acre, and assuming a treatment area of 1,000 acres, the cost would be $3 million ($4.1 million with 35% for contingencies).

3.2.4 Summary of Alternatives

This analysis shows that hypolimnetic oxygenation and phosphorus inactivation are feasible and would likely have a significant impact on algal populations and turbidity. Table 3-3 provides a summary of the benefits and drawbacks of each potential control.

3.3 Long-term Management Recommendations

Based on the data presented in the Phase 1 report and in Section 2 of this report and the alternatives prepared for the algicide application (CDM Smith and WRS, 2013), it is recommended that a long-term management plan be implemented consisting of three elements: continued monitoring of watershed and lake water quality, watershed management activities, and in-lake management activities.

3.3.1 Monitoring Program and On-going Data Assessment

In response to the changes in water quality in Lake Auburn in 2011 and 2012, AWD/LWD expanded the 2013 monitoring program and collected over 6,000 data points. The 2013 monitoring program[2] should be continued with a few changes. A brief description of the program, focusing on the main elements to be continued and recommended changes, is provided below.

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[2] The monitoring program described here focuses on physical lake parameters, nutrients and algae and should supplement the on-going monitoring efforts to sample and analyze for microbiological parameters and other requirements of the Safe Drinking Water Act.

- The location of all sites should be verified with respect to both their spatial and depth locations.

- Weekly *in situ* parameter collection (temperature and dissolved oxygen profiles, pH, conductivity and Secchi depth) should be continued at all five open water locations, with special attention to taking the first profile in the spring before stratification sets in to provide information to support future lake management decisions.

- The Bates College meter array at the deep hole site should be continued. Future data will allow for fine-resolution understanding of thermal structure and vertical mixing dynamics in the lake. AWD/LWD should also continue working with the University of Maine Farmington to collect additional temperature data.

- In 2013, only some of the discrete samples for nutrient parameters were collected as described in the sampling plan in order to meet budget constraints. For 2014 and future years, we recommend scaling back the 2013 program as planned to reduce the total number of samples collected. Station 8 should be sampled at least monthly at the surface (1 meter below water surface) and bottom (1 meter above the bottom) when no stratification is observed. Once stratification is observed Station 8 should be sampled at least monthly at the surface, just above the thermocline, in the thermocline, at the top of the hypolimnion, and at approximate 4 m intervals to the bottom sample, which is taken 1 meter off the bottom. At the remaining four stations (12, 30, 31, and 32), surface (1 meter below water surface) and bottom (1 m above the bottom) samples should be collected monthly. The bottom samples are particularly important if a future year shows the return of discontinuous low oxygen waters at the thermocline as they could provide information about regeneration of phosphorus from these relatively shallow sediments.

  In addition, after anoxic conditions are found at each of the other four open water stations, a profile of discrete samples for nutrient analysis should be taken at least monthly to supplement the top and bottom samples. Samples should be collected at the surface, just above the thermocline, in the thermocline, at the top of the hypolimnion and at approximate 4 m intervals to the bottom sample, which is taken 1 meter off the bottom. These samples will allow for an improved estimate of the mass of phosphorus in the lake once the processes start that regenerate phosphorus from the sediments and examine nutrient limitation.

  Also, the analytical procedures for nutrient parameters should be reviewed prior to the start of sampling in 2014. In 2013, the nitrogen analytical parameters reported were TKN and nitrite + nitrate. In 2013, the laboratory only reported values in whole tenths of TKN (e.g., 0.2, 0.3 mg/l).The reporting limit should be examined prior to the start of the 2014 sampling program.

- Discrete water samples in the tributary and outlet should continue in future years, including measurement of flows at the Basin Dam and Townsend Brook.

Table 3-3
Summary of In-lake Management Alternatives

Alternative	Benefits	Drawbacks	Cost Effectiveness[1]	Reduction in Turbidity?
Dredging	♦ Net positive impact for both deep and shallow alternatives. ♦ Would remove nutrient reserves from sediment, reducing anoxic phosphorus release rate. ♦ May increase and improve spawning habitat for many fish species. ♦ May remove pollutant reserves from sediment, improving overall aquatic habitat quality	♦ Impacts to biota, especially invertebrates, is likely.	♦ Deep dredging (9 to 15 m) for sediment phosphorus removal would likely cost about $86 million. .	♦ Yes, but cost prohibitive.
Phosphorus Inactivation	♦ Possibility of long-term control of phosphorus released from sediments. ♦ Some phosphorus may be removed from the water column, but Lake Auburn phosphorus concentrations are too low for phosphorus inactivation to have a significant impact on immediate water column phosphorus.	♦ Can be toxic to fish and invertebrates. ♦ Most applications today have no adverse effects; fish kills early in the use of phosphorus inactivation resulted from improper buffering. ♦ Small organisms may be carried to the bottom, but data show that no significant impacts to populations are expected.	♦ Vary based on the chemical chosen, the needed dose, and environmental constraints due to sensitive species. ♦ Cost for Lake Auburn likely between $2,500 and $4,000 per acre. ♦ Total cost range likely between $1.8 million and $6.2 million.	♦ Yes. More study needed to assess necessary dose, application area, and expected duration of benefits.

Alternative	Benefits	Drawbacks	Cost Effectiveness[1]	Reduction in Turbidity?
Hypolimnetic Oxygenation	♦ Can make a marked improvement in lake conditions, but effects are not uniform or consistent within and among aquatic systems. ♦ System must be kept running year after year. ♦ Increased oxygen would improve habitat for fish and other aquatic organisms	♦ System failure after establishing an oxic zone may cause a fish kill if anoxia occurs. ♦ Oxygen or nitrogen supersaturation could cause gas bubble disease, but formation of the harmful size bubbles from oxygenation is not expected.	♦ Oxygen demand appears to be at the low end of the range where oxygenation is applied. ♦ Expected capital cost is \$1,600 - \$2,600 per acre. ♦ Approximately half the lake could require oxygenation, representing a total capital cost of \$2.6 million to \$4.2 million. ♦ Operational costs anticipated to be on the order of \$100 per acre per year or about \$155,000, with a range of \$68,000 in a "good" year to \$310,000 in a "bad" year	♦ Yes, but more study is needed to determine amount of oxygen needed, area to be oxygenated

[1] *The capital and O&M costs presented in this table were marked up to include 35% for contingencies given the planning level basis of the cost estimates.*

- Target future wet weather sampling at the North Auburn Dam and Townsend on larger storm events. While samples collected in 2013 showed slightly elevated total phosphorus during wet versus dry weather. The analytical procedure for TSS should be reviewed to target a detection limit as least as low as 0.5 mg/l.

- Composite samples should be collected as they were in 2013 for chlorophyll and algal identification and enumeration.

- *Gloeotrichia* monitoring should continue at the shoreline stations to continue to collect data consistent with the long-term record.

- Continue the intensive sampling program at Little Wilson and Mud Ponds and add samples from the Basin

- Monitoring sediment accumulation in the Basin and upstream ponds at the beginning and end of each sampling season. This can be done by permanently mounting a staff gage on the North Auburn Dam and then measuring the distance to the sediment relative to this fixed point using a rod with a flat disc mounted on the end.

- Maintain a log of boards used and use of sluice gate to allow calculation of flows through the dam.

Data from the monitoring program should be evaluated on both an on-going and annual basis. On-going evaluation allows for identification of trends that need to be addressed immediately. These could include increases in turbidity in the lake, atypical results from watershed monitoring, or occurrence of nuisance algae.

In addition, an annual data summary should be prepared at the end of each year. The summary should update trend plots on precipitation, annual isopleths of temperature and dissolved oxygen, Secchi depths, turbidity, time histories for phosphorus in the surface waters (0 to 5 m) and bottom waters (30 to 35 m), time histories of chlorophyll, algal cell count and algal biomass. The following calculations should be made from the available data: anoxic factor, number of days maximum turbidity exceeds 1 NTU, N:P ratios, time history of total mass of phosphorus in the lake by thermal layer (epilimnion, metalimnion, hypolimnion), estimated watershed load of phosphorus, and assessment of the timing, abundance and dominance of algal groups.

3.3.2 Watershed Management

AWD/LWD and the LAWPC should continue and strengthen their on-going watershed management program that serves as the first barrier in their multiple barrier approach to providing safe drinking water. Estimates of the external load of phosphorus to Lake Auburn show that it has been controlled to a substantial extent through existing watershed programs to control non-point sources of phosphorus. However, recent investigation (CEI, 2013a) has suggested some areas of concern within the watershed, particularly the occurrence of many sediment deltas of recent origin around the lake's perimeter and in the watershed. These deltas are hypothesized to have resulted from more intense storms that have occurred since 2005 and are of concern because sediment adds phosphorus load to the lake and most likely oxygen demanding substances as well. Ultimately, the reserves of phosphorus in Lake Auburn's sediments have their origin in the watershed and minimizing watershed loading is in the best interest of the lake and its users.

Since the Phase 1 report, LAWPC has made good progress in further strengthening the watershed management, through implementation of both structural and non-structural BMPs. Recent actions taken by AWD/LWD are described in Section 1.

One of the most important controls for sediment load that exists in the watershed today are the impoundment behind the North Auburn Dam (the Basin) and other upstream ponds that exist as part of a linked stream pond system. These ponds serve as natural sedimentation basins and monitoring sediment accumulation and maintaining their capacity to store sediment (e.g., through dredging if needed) should be a high priority.

Tributary sampling conducted in 2013 shows that phosphorus concentrations entering the lake from Townsend Brook are about double those entering from the basin (though the load in Townsend Brook is smaller than the Basin due to the relatively smaller flow in Townsend Brook). The reason for this is not known, and could include reduction in load from the linked pond-stream system that occurs in the Basin watershed or a source(s) of higher phosphorus load (e.g., agricultural activities) that exist in the Townsend Brook watershed. This lead to LAWPC commissioning Summit Environmental to conduct an investigation of phosphorus sources in the lower Townsend Brook watershed. The study (Summit Environmental, 2014) found that phosphorus concentrations along the brook were relatively uniform (note they were slightly lower than those measured in the AWD/LWD sampling program); phosphorus concentrations were higher in some of the tributaries to the brook but these increases were not seen downstream of the tributaries entering the brook due to dilution with water in the brook. The study recommended further evaluation including:

- additional water quality monitoring during the growing season, and
- repairs to the former fish hatchery pond outfall and the Gammon's Nursery irrigation pond outfall and implementing, if possible, erosion control measures between these outfalls and Townsend Brook to reduce phosphorus load, particularly during runoff events.

CEI (2013b) prepared a Phase 2 report that included recommendations for structural and non-structural BMPs for the watershed. The structural BMPs focused on controls of sediment from problem areas noted in the lake and watershed or control of potential spills that could occur from the roads that surround the lake. We recommend that AWD/LWD continue to take advantage of opportunities to implement these BMPs as they arise (e.g., through use of dedicated watershed funds and available grant funding), focusing initially on controlling erosion at locations that discharge directly to the lake.

The CEI Phase 2 report also included recommendations on non-structural BMPs. The recommendations from that report are summarized below. AWD/LWD has already started to implement many of these and we recommend full implementation of these recommendations.

- Amending regulations to promote low impact development, amend the zoning ordinance to strengthen subsurface watershed disposal requirement, update the phosphorus control ordinance, amend the ordinance to prohibit certain land uses within the watershed, and adding a steep slope ordinance;
- developing a public education program, including an enhanced website and educational materials, and education programs for schools, farmers, residents on the value of buffers and gravel road maintenance
- continuing watershed monitoring activities, including tributary water quality and forestry activity;
- setting aside funds for maintaining watershed lands;
- continuing with key parcel land acquisition and conservation efforts of environmentally sensitive lands;
- developing trail networks on LAWPC properties;
- continuing to control invasive species; and
- continuing with the gull management program.

In addition to these recommendations, we recommend that AWD/LWD continue with identification and control of controllable sources of phosphorus in the watershed including working with the golf course to reduce use of fertilizers, working with farmers in the surrounding area to encourage best practices, and modeling best management practices concerning the timber harvesting plan that was developed in 2013 and encouraging their use throughout the watershed.

3.3.3 In-lake Management

The data and analysis presented in this report indicate that implementation of an in-lake management system could be needed within the next several years to drive a significant improvement in water quality in Lake Auburn. It is not needed immediately, however, due to the contingency plan AWD/LWD has to apply algicide if algal blooms similar to 2011/2012 recur. While the lake's water quality could continue to improve, as was seen in many indicators in 2013, changing climatic factors (e.g., more frequent intense rain events and the increase in stratification stability as presented in Section 2.6) will pose hurdles to continued water quality improvement. Nonetheless, because recurrence of degraded water quality similar to 2011 or 2012 is heavily dependent climate and other factors outside of our control, it is sensible to begin planning now to implement an in-lake management system.

The analysis in Section 3.2 indicated that AWD/LWD consider implementation of either an alum treatment or a pure oxygen system. Table 3-4 summarizes the relative benefits and drawbacks of both in-lake management activities.

The principal advantages of an alum treatment are that it is a simple, quick, and effective treatment that requires no additional ongoing operation and maintenance cost after the initial capital investment. An alum treatment is not permanent, however, and the internal load will gradually increase as more phosphorus is added to the lake from external sources. Improving control of phosphorus entering the lake through watershed management will cause the treatment to last longer, delaying the capital cost before another treatment is required. There is also some risk to the aquatic environment if not carefully implemented.

Oxygenation is another treatment method that can be used to reduce the internal load rate. The principal advantages of oxygenation are that there are not any issues with regulatory or citizen acceptance as the only substance being added to the lake is pure oxygen. In addition, an oxygenation system will improve habitat suitability without any significant ecological impacts. The downside to an oxygenation system is that there is an ongoing operation and maintenance cost associated with running and maintaining the equipment and purchasing oxygen to supply the system. An oxygenation system would only need to be run on an as needed basis.

From the perspective of effectiveness of phosphorus control, there is no technical reason to prefer one of these treatments over the other. The analysis in Section 3.2 also indicates that planning level costs for these two systems are similar, with the higher capital cost of alum being balancing with the operation and maintenance cost of the oxygenation system. We recommend AWD/LWD conduct the necessary preparations to be able to implement either system should an in-lake system be required. Ultimately the selection of a system will require consideration of the tradeoffs related to operating an aeration system on an on-going basis, stakeholder acceptance, regulatory approval, potential for benefits to the lake's aquatic life, and flexibility provided by the management action.

3.3.4 Summary of Recommendations

The proposed plan of action is as follows.

1. Continue to monitor the lake water quality with the modified monitoring program described above, and prepare an annual summary of the monitoring data.

2. Continue to maintain and strengthen the watershed management program to control external loads of phosphorus through both structural and non-structural BMPs as both a best practice as part of a multiple barrier system for providing safe drinking water, and also actions that could potentially delay the need for and/or improve the success of any large scale in-lake management system.

Maintain the ability to apply algicide (by renewing the permit as necessary) as a short-term measure to control a significant bloom before it develops. The timing of treatment would be triggered by an increase in turbidity above 1.5 NTU as a rolling two-day average with collaboration that algae are responsible for the elevated turbidity. The greatest concern is with the cyanobacteria *Anabaena* and *Microcystis*, which are expected to reach maximum abundance in September or October. Copper sulfate should be applied in accordance with the Algicide Application Plan (CDM Smith and WRS, 2013) and the Maine Department of Environmental Protection Waste Discharge License (Maine DEP, 2013).

3. Plan for the implementation of either a diffused oxygen distributor system to oxygenate a portion or all of the hypolimnion in Lake Auburn or an alum treatment to bind phosphorus in the sediment in the lake, including:

    a. initiating outreach to the regulatory agencies and stakeholders to obtain their buy-in for either in-lake management approach and decision criteria for implementation

    b. establishing the ability to fund the capital costs to implement either management approach and to continue to fund operation and maintenance of an oxygenation system,

    c. identifying the location for land facilities for an oxygenation system (secured fenced area for liquid oxygen storage tank on a concrete pad and vaporizer),

    d. collecting additional sediment samples to help refine the area for an alum treatment. Although existing data are sufficient to design a responsive alum treatment for inactivation of phosphorus in surficial sediments, greater accuracy and possible cost savings could be realized with additional testing. It is recommended that 15 to 20 samples be collected in water 30 to 70 feet deep, each sample representing the upper 2 to 4 inches of sediment, and that those samples be tested for iron-bound phosphorus and response to aluminum assays at simulated doses of 25, 50 and 75 g/m^2.

    e. identifying permitting requirements for both management approaches, and

    f. preparing a draft procurement document for both management approaches with performance specifications, which will need to be reviewed and validated with most recent information after the decision is made to implement one of the approaches.

4. If an algicide treatment is required, immediately review that current year's water quality data and climatic events to determine a likely cause(s), and unless the cause(s) is highly unusual (e.g., 2011's breakdown of stratification due to passage of Hurricane Irene) initiate design and permitting to implement either a diffused oxygen distributor system or alum application by July of the following year. The design of the oxygenation system would use all the temperature and dissolved oxygen profiles, including those collected in the year when the algicide was applied. The area to be treated with alum would be informed by current and additional data on sediment phosphorus levels.

Table 3-4
Comparison of Hypolimnetic Oxygenation with Phosphorus Inactivation

Factor	Hypolimnetic Oxygenation	Phosphorus Inactivation
Capital cost[1]	$2.6 to 4.2 million (best est. $3.4 million)	$1.8 to 6.2 million (best est. $4.1 million)
Annual operational cost[1]	$68,000 to $310,000 (best est. $155,000)	None
Mode of algal control	Suppresses phosphorus release from sediment; also reduces iron release, which may aid in cyanobacteria control	Suppresses phosphorus release from sediment, removes some phosphorus from water column during treatment
Additional benefits	Enhances deep-water habitat through increased oxygen levels; may reclaim entire hypolimnion or any targeted portion thereof	May reduce oxygen demand from settling algae; tends to reduce metalimnetic demand and may extend oxygen about 10 feet deeper into hypolimnion than might otherwise occur
Longevity of benefits	Indefinite while system is running; benefits lost quickly (days to weeks) when system is not running. System would likely run about 2 months each year.	Typically at least 10 years for deeper lakes, usually close to 20 years, not expected beyond 30 years. Control of watershed load needed to extend longevity.
Longevity of equipment	Most systems have maintenance schedule to replace in-lake portion of system about every 10 years	No equipment to replace, but may require repeat application any time after 10 years
Toxic impacts	None	Risk during actual treatment depends on pH and dose.
Sediment Impacts	Will encourage decomposition of organic sediment and lowering of oxygen demand; may have decreasing (but not eliminated) need for oxygen addition with time.	Adds minor amount of material to sediment. Some smothering of some benthic organisms expected; recovery expected within 2 years, possibly with better quality assemblage

[1] *The capital and O&M costs presented in this table were marked up to include 35% for contingencies given the planning level basis of the cost estimates.*

Section 4
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Appendix A
Initial Screening of Long-term Management Alternatives

Table A-1
Initial Screening of Long-term Management Alternatives
Reprinted from the Diagnostic Study of Lake Auburn and its Watershed: Phase I (CDM Smith, 2013)

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICABILITY TO LAKE AUBURN
PREVENTION				
1) Watershed Management	♦ Includes wide range of watershed and lake edge activities intended to eliminate nutrient sources or reduce delivery to lake ♦ Essential component of algal control strategy where internal recycling is not the dominant nutrient source, and desired even where internal recycling is important	♦ Acts against the original source of algal nutrition ♦ Creates sustainable limitation on algal growth ♦ May control delivery of other unwanted pollutants to lake ♦ Facilitates ecosystem management approach which considers more than just algal control	♦ May involve considerable lag time before improvement observed ♦ May not be sufficient to achieve goals without some form of in-lake management ♦ Reduction of overall system fertility may impact fisheries ♦ May cause shift in nutrient ratios to favor less desirable algae	♦ Applicable, but not as a short-term measure to control algae and thus limit potential high turbidity to avoid violating the terms of the filtration waiver. Watershed management is a required part of comprehensive plan to reduce nutrient concentrations in the lake.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICABILITY TO LAKE AUBURN
IN-LAKE MECHANICAL/PHYSICAL CONTROLS				
2) Circulation and destratification	♦ Use of water or air to keep water in motion ♦ Intended to prevent or break stratification ♦ Generally driven by mechanical or pneumatic force	♦ Reduces surface build-up of algal scums ♦ May disrupt growth of blue-green algae ♦ Counteraction of anoxia improves habitat for fish/inverte-brates ♦ Can eliminate localized problems without obvious impact on whole lake	♦ May spread localized impacts ♦ May lower oxygen levels in shallow water	♦ Not applicable. Although circulation would reduce or eliminate internal loading of phosphorus from sediments, it would disturb the cold water fishery, reducing habitat for lake trout.
3) Drawdown	♦ Lowering of water over autumn period allows oxidation, desiccation and compaction of sediments ♦ Duration of exposure and degree of dewatering of exposed areas are important ♦ Algae are affected mainly by reduction in available nutrients.	♦ May reduce available nutrients or nutrient ratios, affecting algal biomass and composition ♦ Opportunity for shoreline clean-up/structure repair ♦ Flood control utility ♦ May provide rooted plant control as well	♦ Possible impacts on non-target resources ♦ Possible impairment of water supply ♦ Alteration of downstream flows and winter water level ♦ May result in greater nutrient availability if flushing inadequate	♦ Not applicable. The water surface elevation in Lake Auburn cannot be lowered enough to achieve these benefits. Furthermore, before managed releases could occur measures to retain salmon in the lake would need to be implemented.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICABILITY TO LAKE AUBURN
4) Dredging	♦ Sediment is physically removed using suction or cutterhead dredges ♦ Dredges create a slurry that is hydraulically pumped to containment area and dewatered. Sediment is retained; water is discharged. ♦ Nutrient reserves are removed and algal growth can be limited by nutrient availability	♦ Can control algae if internal recycling is main nutrient source ♦ Can reduce sediment oxygen demand ♦ Can improve spawning habitat for many fish species ♦ Allows complete renovation of aquatic ecosystem ♦ Possible mechanism to control *Gloeotrichia* cysts	♦ Often leaves some sediment behind ♦ Temporarily removes benthic invertebrates ♦ Interference with recreation or other uses during dredging ♦ Can result in short term elevated turbidity levels	♦ Applicable but costly compared other approaches. Control of sediment phosphorus would require dredging in 9 to 15 m of water. Dredging to control *Gloeotrichia* would require dredging of shallow sediments down to either 4.6 or 13 m. Further investigation on *Gloeotrichia* recruitment areas would be needed before dredging would be recommended. In both cases, dredging would be truly restorative.
5) Selective withdrawal from the water intake	♦ Discharge of bottom water which may contain (or be susceptible to) low oxygen and higher nutrient levels ♦ May be pumped or utilize passive head differential	♦ Removes targeted water from lake efficiently ♦ May prevent anoxia and phosphorus build up in bottom water ♦ May remove initial phase of algal blooms which start in deep water	♦ May promote mixing of remaining poor quality bottom water with surface waters ♦ May cause unintended drawdown if inflows do not match withdrawal	♦ Potentially applicable but not in the short term; raw water intake was recently extended 900 ft. and cannot be modified for a hypolimnetic withdrawal. This alternative would require construction of a new longer intake.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICABILITY TO LAKE AUBURN
6) Hypolimnetic aeration or oxygenation	♦ Addition of air or oxygen provides oxic conditions ♦ Maintains stratification ♦ Can also withdraw water, oxygenate, then replace	♦ Oxic conditions reduce P availability ♦ Oxygen improves habitat ♦ Oxygen reduces build-up of reduced compounds	♦ May disrupt thermal layers important to fish community ♦ Theoretically promotes supersaturation with gases harmful to fish	♦ Applicable. Would prevent anoxic conditions from occurring, reducing phosphorus release rate from sediments and protecting the cold water fishery.
7) Phosphorus inactivation	♦ Typically salts of aluminum, iron or calcium are added to the lake, as liquid or powder ♦ Phosphorus in the treated water column is complexed and settled to the bottom of the lake ♦ Phosphorus in upper sediment layer is complexed, reducing release from sediment ♦ Permanence of binding varies by binder in relation to redox potential and pH	♦ Can provide a decrease in phosphorus concentration in water column ♦ Can minimize release of phosphorus from sediment ♦ May remove other nutrients and contaminants as well as phosphorus ♦ Flexible with regard to depth of application and speed of improvement	♦ Possible toxicity especially by aluminum ♦ Possible release of phosphorus under anoxia or extreme pH ♦ May cause fluctuations in water chemistry, especially pH, during treatment ♦ Possible resuspension of floc in shallow areas ♦ Adds to bottom sediment	♦ Applicable; internal load is a major source of phosphorus and inactivation with aluminum is possible. Requires more local data to implement, so cannot be used as a short term mitigation technique.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICABILITY TO LAKE AUBURN
8) Dilution and flushing	♦ Addition of water can dilute nutrients and flush system to minimize algal buildup ♦ Can occur continuously or periodically	♦ Dilution reduces nutrient concentrations without altering load ♦ Flushing minimizes detention so the response to pollutants may be reduced	♦ Diverts water from other uses ♦ Flushing may wash desirable zooplankton from lake ♦ If water used is of poorer quality nutrient loads can increase	♦ Not applicable. Although groundwater resources could be used to enhance dilution and flushing, they have not been quantified and if taken from within the basin would ultimately reduce baseflow in tributary streams.
ALGICIDE				
9) Algicides	♦ Liquid or pelletized algicides applied to target area ♦ Algae killed by direct toxicity or metabolic interference ♦ Typically requires application at least once/yr, often more frequently	♦ Rapid elimination of algae from water column , normally with increased water clarity ♦ May result in net movement of nutrients to bottom of lake	♦ Possible toxicity to non-target species ♦ Restrictions on water use for varying time after treatment ♦ Increased oxygen demand and possible toxicity ♦ Possible recycling of nutrients	♦ Applicable. Can be applied as a temporary stopgap measure until more permanent measures can be implemented. Most effective when applied prior to the exponential growth phase.

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES	APPLICABILITY TO LAKE AUBURN
9a) Copper sulfate	♦ Cellular toxicant, disruption of membrane transport ♦ Applied as wide variety of liquid or granular formulations	♦ Effective and rapid control of many algae species ♦ Approved for use in most water supplies	♦ Possible toxicity to aquatic fauna ♦ Accumulation of copper in system ♦ Resistance by certain green and blue-green nuisance species ♦ Lysing of cells releases nutrients and toxins	♦ Applicable; requires a permit from Maine DEP. This is the recommended form of algicide for application to Lake Auburn.
9b) Peroxides	♦ Disrupts most cellular functions, tends to attack membranes ♦ Applied as a liquid or solid. ♦ Typically requires application at least once/yr, often more frequently	♦ Rapid action ♦ Oxidizes cell contents, may limit oxygen demand and toxicity	♦ Much more expensive than copper ♦ Limited track record ♦ Possible recycling of nutrients	♦ Applicable. Less disruptive than copper, but more expensive. Tends to work best on cyanobacteria, but unlikely to prevent all blooms in fertile system. This form of algicide is not recommended for Lake Auburn.
CULTURAL CONTROLS				
10) Enhanced grazing	♦ Manipulation of biological components of system to achieve grazing control over algae ♦ Typically involves alteration of fish community to promote growth of grazing zooplankton	♦ May increase water clarity by changes in algal biomass or cell size without reduction of nutrient levels ♦ Can convert unwanted algae into fish ♦ Harnesses natural processes	♦ May involve introduction of exotic species ♦ Effects may not be controllable or lasting ♦ May foster shifts in algal composition to even less desirable forms	♦ Potentially applicable, but the addition of fish may have detrimental trophic effects on the existing fish population. Minimal information available on current grazing capacity.

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