319(H) GRANT (WQR-2019-LHC00130) QUANTIFICATION OF LAKE HOPATCONG'S INTERNAL TOTAL PHOSPHORUS LOAD REPORT

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1.0 INTRODUCTION

Lake Hopatcong is the largest lake in New Jersey, with a surface area of 2,686 acres and approximately 39 miles of shoreline. With a maximum depth of 16.7 meters and a mean depth of 5.6 meters, the lake is dimictic and stably stratifies during the growing season each year. Lake Hopatcong is a highly valued resource for the state and has a substantial impact on the local economy. Unfortunately, Lake Hopatcong experienced a large-scale and unprecedented harmful algal bloom (HAB) of cyanobacteria over the course of the 2019 growing season. These HABs resulted in the posting of advisories over large sections of the lake and the closing of all beaches by local/County Departments of Health. These conditions resulted in substantial impacts on the ecological, recreational, and economic resources of the lake and region.

While prevailing weather conditions, including periods of heavy rainfall followed by dry, still-water conditions certainly contributed toward the 2019 HABs, elevated phosphorus concentrations were the main contributing factor. In fact, these blooms were triggered by some of the highest June total phosphorus (TP) concentrations measured over the last 25 years. The elevated June phosphorus concentrations that triggered the bloom were largely the result of external phosphorus loading, in the form of phosphorus-rich stormwater, caused by the heavy rain. However, internal phosphorus loading may have allowed the large-scale HAB to persist in Lake Hopatcong throughout the entire growing season.

Phosphorus is the limiting nutrient, or the nutrient that is naturally lowest in supply relative to the amount required for plant and algal primary productivity, in most freshwater aquatic ecosystems (Schindler, 1977; Schindler, et. Al, 2008). As such, a small increase in phosphorus loading has the potential to stimulate a large amount of primary production. This is the case for Lake Hopatcong, in which a Total Maximum Daily Load (TMDL) for total phosphorus has existed since the New Jersey Department of Environmental Protection (NJDEP) conducted a TMDL analysis for the Upper Musconetcong River Watershed in 2003.

In order to make sound management recommendations for a lake or reservoir, it is important to understand the extent of the pollutant loading to the waterbody. This includes both the external pollutant load, or that derived from the watershed and the internal pollutant load, or that derived from the internal release of nutrients from decomposing organic matter (plants, algae) and from the sediments. While the external pollutant load of Lake Hopatcong has been well-studied and managed over the past 40 years through the implementation of watershed best management practices (BMPs) with various grant funds, the internal phosphorus load has not been studied recently. In the original Clean Lakes Study of Lake Hopatcong, it was estimated that the internal load only accounted for less than 10% of the lake's total annual TP load (Princeton Aqua Sciences, 1983). However, as of 2019, it was estimated that approximately 2,362 lbs. of phosphorus, or 32% of the phosphorus reduction required to meet the TMDL goal, had been removed from the annual Lake Hopatcong phosphorus load through the implementation of watershed BMPs and other restoration measures (Princeton Hydro, 2020). It has been hypothesized that the current internal load constitutes a much larger percentage of the total annual phosphorus load. This hypothesis was based off of the direct decrease in the external load through watershed restoration and the recent trends in growing season weather patterns, which are discussed in this report, that have the potential to cause changes to the physical and chemical properties of the lake that can lead to elevated internal phosphorus loads. Increasing hypolimnetic TP concentrations were also noted.

The internal loading of phosphorus occurs when the hypolimnion becomes anoxic during the summer stratification period in temperate lakes. The cooler bottom layer of water, or hypolimnion, is separated from the warmer upper layer, or epilimnion, due to a difference in water density. Anoxia can occur (DO < 1.0 mg/L) in the hypolimnion during this time when bacteria exhaust the finite source of oxygen when decomposing organic matter. Under oxic conditions, phosphorus in the sediment is weakly bound to ferric iron (Fe^{3+}), forming ferric hydroxyl phosphate complexes that holds a large portion of phosphorus in the sediment if there is enough iron present. These iron complexes are redox sensitive, and when the hypolimnion becomes anoxic the ferric iron gains an electron and becomes soluble ferrous iron (Fe^{2+}). When this occurs, any phosphorous that was bound to iron will be released into the water column in a dissolved form, becoming available for algal assimilation. In addition to the anoxic release of phosphorus from the sediment, phosphorus is also released from oxic sediments, albeit at lesser quantities than anoxic sediment, at varying rates depending on the amount of phosphorus stored in the sediment and the degree of disturbance that occurs at the sediment-water interface. These disturbances can include bioturbation from benthic-dwelling fish or other macro-organisms as well as increased water velocities in shallow areas of the waterbody.

Once the previously bound inorganic phosphorus is released into the overlying water column under anoxic conditions, it becomes potentially available for algal assimilation. As mentioned above, because phosphorus is often the limiting nutrient in lake ecosystems, a small increase in phosphorus can result in a large increase in algal growth. While the internally released phosphorus in the hypolimnion is generally thermally separated from the epilimnion, where much of the phytoplankton and cyanobacteria activity takes place, there are a few ways that this phosphorus can become available for cyanobacteria assimilation, including metalimnetic erosion or partial mixing events, thermocline migration, and through the vertical movement of depth regulating cyanobacteria.

The Lake Hopatcong Commission (LHC) has a 30 year continuous water quality database that was used for the modeling efforts associated with the study. The database was used to calculate the internal phosphorus load on a sampling event basis throughout the monitoring period focused on the growing season, roughly April to October. Lake sampling typically was conducted once monthly from May through September during each monitoring year, although the number of events did vary in some years, and could extend from March into October. By calculating the standing load during each event, it was possible to develop a variety of loading statistics analyzed at various time scales. There were three main tasks completed in preparation of this report, including:

- An updated bathymetric assessment of the main body of Lake Hopatcong
- Supplemental *in-situ* and phosphorus monitoring in Lake Hopatcong during the 2021 season
- Modeling activities to quantify the internal TP load at different time scales and under varying hydrologic conditions

This report aims to quantify and refine the lake's existing internal TP load using a measured data and subsequently determine what ecological implications the internal phosphorus load has on the lake and what management measures for controlling the internal load are most applicable. This project is essentially predicated on the four following questions:

- What are the phosphorus dynamics of Lake Hopatcong as it regards the internal load and has this changed over time?
- How does the internal load change under various hydrologic condition?
- How is the internal load ecologically significant?
- Does the internal load merit management?

2.0 BACKGROUND AND PHOSPHORUS DYNAMICS

This section will first provide a brief overview of lake monitoring and management efforts in Lake Hopatcong over the last 30+ years and the groups and organizations that have been involved in these efforts. Next, an overview of phosphorus dynamics in lake ecosystems, including internal and external loading, will be discussed.

2.1 LAKE HOPATCONG MANAGEMENT BACKGROUND

Lake Hopatcong is likely one of the longest, continuously monitored, lakes in New Jersey. The long-term analysis of the database stemming from those efforts has been extremely valuable in identifying long-term trends and how efforts to meet the TP TMDL have contributed to water quality improvements. The current water quality monitoring program is a modified version of the program that was originally initiated in the Phase I Diagnostic/Feasibility Study of Lake Hopatcong (Princeton Aqua Science, 1983) and continued through the Phase II Implementation Projects. Both the Phase I and some Phase II projects were funded by the US EPA Clean Lakes (314) Program. The modified monitoring program also continued through the development, revision, and approval of the TMDL-based Restoration Plan, as well as through the installation of a series of watershed projects funded through two NJDEP 319 grants and an EPA Targeted Watershed grant. Finally, some additional monitoring was conducted during the 2020 and 2021 seasons as part of the respective NJDEP HABs grant and the 319-grant associated with this project.

The current water quality monitoring program is valuable in terms of continuing to assess the overall health of the lake on a year-to-year basis, identifying long-term trends or changes in water quality, and quantifying and objectively assessing the success and potential impacts of restoration efforts. In addition, the in-lake water quality monitoring program continues to be an important component in the evaluation of the long-term success of the implementation of the phosphorus TMDL-based Restoration Plan, which was approved by NJDEP in April of 2006 (Princeton Hydro, 2006). The monitoring program also provides the data necessary to support the Lake Hopatcong Foundation's and Lake Hopatcong Commission's requests for grant funding to implement both watershed-based and in-lake projects to improve the water quality of Lake Hopatcong.

2.2 MAJOR GROUPS INVOLVED IN LAKE HOPATCONG MANAGEMENT

There are a multitude of organizations involved in the overall management of the largest freshwater lake in New Jersey. As a public waterbody in the state of New Jersey, NJDEP oversees certain management aspects and appropriates funds for lake management and restoration. From the early 1980s to early 2000, the Lake Hopatcong Regional Planning Board oversaw the implementation of various in-lake and watershed-based projects, primarily through State and Federal sources. Because of the large size of Lake Hopatcong and the fact that it spans four municipalities and two counties, there was a need for a representative governing body, with members from each municipality as well as the State, to oversee the preservation, restoration, and enhancement of Lake Hopatcong and its watershed. As a result, the Lake Hopatcong Protection Act was passed in 2001 which created the Lake Hopatcong Commission. The non-profit Lake Hopatcong Foundation was formed in 2012 with a similar goal of improving Lake Hopatcong and its surrounding communities. Finally, as mentioned above, Princeton Hydro and direct predecessors have been involved with the management of Lake Hopatcong since the early 1980s. A brief description of these major groups and their role in the management of Lake Hopatcong is provided below:

NEW JERSEY DEPARTMENT OF ENVIRONMENTAL PROTECTION

Formed in 1970, NJDEP is the state governing agency that is responsible for managing and protecting the state's natural and historic resources and public health. With a major goal of protecting the state's water resources, NJDEP manages and protects this resource by preventing pollution, cleaning up contamination, ensuring ample supply, and investing in strong infrastructure. In relation to Lake Hopatcong, NJDEP oversees overall management efforts, including water quality monitoring, management, or any other permitted activity. NJDEP also oversees and allocates grant funds related to lake monitoring and management in Lake Hopatcong.

LAKE HOPATCONG COMMISSION

The Lake Hopatcong Commission is an independent agency overseen by an 11-member board of Commissioners. Formed in 2001 as part of the Lake Hopatcong Protection Act, the Commission is a state-funded governing body with an annual budget for lake management efforts. As stated on the Lake Hopatcong Commission website, "The Commission is responsible for conducting, managing, and coordinating activities for the preservation, restoration, and enhancement of Lake Hopatcong and its watershed. Members of the Commission work as volunteer representatives of the larger lake community, and are dedicated to permanently preserving the natural, historical, and recreational resources of the watershed for this generation and beyond." Two of the long-standing management efforts that the Commission funds include the annual water quality monitoring program and macrophyte harvesting.

LAKE HOPATCONG FOUNDATION

Founded in 2012, the Lake Hopatcong Foundation (LHF) is a 501(c)3 non-profit dedicated to the improvement of Lake Hopatcong and its surrounding communities. As stated on the Lake Hopatcong Foundation website, "The Lake Hopatcong Foundation dedicates itself to protecting the lake environment and enhancing the lake experience, bringing together public and private resources to encourage a culture of sustainability and stewardship on and around New Jersey's largest lake, for this and future generations."

PRINCETON HYDRO

Princeton Hydro and its predecessor company, Princeton Aqua Science, have been conducting the lake monitoring and management efforts in Lake Hopatcong since the 1980s. In addition to the lake monitoring efforts, Princeton Hydro has also been involved with the permitting, design, installation, and monitoring of various watershed and lake-based projects in the Lake Hopatcong watershed over the last 30+ years.

2.3 PHOSPHORUS DYNAMICS

Although both nitrogen and phosphorus are essential to the growth and development of phytoplankton, filamentous algae, and aquatic macrophytes, in most freshwater aquatic ecosystems phosphorus is the limiting nutrient, or the nutrient that is naturally lowest in supply relative to the demand required for plant and algal primary productivity (Schindler, 1977; Schindler, et. al, 2008). As such, a small increase in phosphorus loading has the potential to stimulate a large amount of primary production. Thus, most of the proactive measures taken to control primary production, accelerated eutrophication, and HABs focuses on controlling and reducing phosphorus.

Undeniably there is a link between watershed development and HABs. Watershed development leads to an increase in runoff volume and the rate at which that runoff enters lakes and the lake's tributaries. The increased volume of runoff increases the mobilization and transport of nutrients and other non-point source (NPS) pollutants. The added volume and rate of runoff also leads to the scour and erosion of streams resulting in increased sediment and nutrient loading to the receiving lake system.

In addition to external sources of nutrient loading, there are internal sources of nutrient loading. This is especially true with respect to phosphorus. Internal phosphorus loading has been shown in many lakes to be an important stimulant and driver of primary production, especially with respect to the development of phytoplankton blooms (Cooke, et al. 2005). While some internal phosphorus loading can be attributed to the natural die-off of plants, phytoplankton, benthic algae, and other organisms, as well as sediment resuspension, for most lakes characterized by a large internal phosphorus load the primary factors responsible for internal phosphorus loading are thermal stratification and associated water column anoxia (the absence of dissolved oxygen). Although this can occur in both shallow and deep lakes, it is most commonly experienced, and has the greatest negative impact, in moderately deep to deep lakes that become thermally stratified during the summer.

2.3.1 EXTERNAL LOADING

External loading refers to the nutrient loading of allochthonous material to a receiving waterbody that originated in the watershed. The major sources of external loading include stormwater runoff, groundwater and septic loading, tributary loading, atmospheric loading, and point-source loading, such as wastewater treatment facilities. As such, the external load to a waterbody is dependent on the land uses and activities within the watershed as well as local hydrology.

The interconnectivity between lakes and their watersheds is a central tenant in NPS pollution control. Watershed size and the land uses, soil types, topography, and geology in concert with variable climatic conditions all influence the quantity of water, its temporal distribution, and the nutrient load into the lake. A direct correlation exists between watershed disturbance and increases in nutrient loading. The conversion of forests to agricultural, residential, commercial, and industrial lands brings about an increase in nutrient loading due to increases in erosion and a multitude of anthropogenic factors.

The North American Lake Management Society (NALMS) has coined the phrase, "A lake is a reflection of its watershed." This phrase conveys the importance of the relationship between watersheds and lakes and helps to orient lake management activities towards preventative measures and correcting the source of lake eutrophication with less reliance on reactive measures. One of the primary areas of focus utilizing this approach is understanding how the watershed transports water and nutrients to the lake and using this information, in concert with in-lake abiotic and biotic data, to develop well-informed management measures.

It was determined in the original Lake Hopatcong Restoration Plan that over 80% of the annual phosphorus load entering Lake Hopatcong originated from surface runoff and septic systems (Princeton Hydro, 2006). This report also estimated that 7% of the annual phosphorus load was the result of tributary loading from Lake Shawnee while 1% was the result of atmospheric loading. However, since 2006, a partial sewering of the Borough of Hopatcong, a mandatory septic pump-out ordinance in the Township of Jefferson, and a multitude of other watershed-based restoration measures has greatly reduced the external phosphorus load to Lake Hopatcong.

2.3.2 INTERNAL LOADING

The physical properties of water are such that water reaches its greatest density at 4°C. As water warms its density decreases. During the summer, due to the sun's heating of a lake's surface, the shallower waters (referred to as the epilimnion) become warmer than the deeper waters (referred to as the hypolimnion). A temperature gradient as low as 1°C/m between two depth intervals can result in a density difference great enough to inhibit the vertical mixing of the water column. When this occurs, the lake becomes thermally stratified. The depth or boundary layer that inhibits vertical mixing is referred to as the thermocline or metalimnion. At times, this boundary can be very sharp, while in other circumstances it may over several meters. Once the thermocline becomes established the vertical mixing of the water column is inhibited resulting in the deeper waters of the lake becoming segregated from the surface waters. The thermocline can develop at any water depth, deep or shallow, and often migrates vertically over the course of a season. The depth at which it forms will be dictated by a number of factors including:

- Time of year, solar intensity, and air temperature

- Water color and clarity
- Lake depth
- Wind and wave action
- Flushing rate (hydraulic retention)
- Lake fetch (the uninterrupted distance that wind blow across a lake's surface)
- Magnitude and strength of the lake's seiche (an internal wave most commonly produced along the thermocline)
- Other factors that affect light penetration and water column stability

The maintenance of dissolved oxygen in lake water is largely a function of the consistent vertical mixing of the water column and the continued and repeated exposure of water to the atmosphere. Although some re-oxygenation may occur as a result of the photosynthetic activity of benthic algae, phytoplankton, and aquatic plants, or the turbulence created at the mouth of a tributary, the majority of re-oxygenation occurs due to the exposure of lake water to the atmosphere. Once stratified, the waters below the thermocline are no longer able to freely mix to the surface, and over time these thermally segregated layers become depleted of oxygen and go anoxic due to community respiration. The rate at which this occurs is variable from lake to lake depending in part on respiration rates, the volume of the segregated layer, the organic composition of the sediments, water temperatures, and other factors. When the density differences between the epilimnion and the hypolimnion are great enough, thermal stratification will persist for long-periods of time leading to the majority of the lake water below the thermocline becoming devoid of oxygen. As a result, a large volume of a lake can become oxygen depleted during periods of stratification. This not only results in a large portion of the lake unable to support organisms dependent on oxygen like fish and zooplankton, but it also alters the chemistry of the lake's sediments.

Under oxic conditions (presence of oxygen) the majority of the phosphorus present in lake sediments is covalently bound to ferric iron (Fe^{3+}), forming ferric phosphate and ferric hydroxyl phosphate complexes. These complexes effectively lock a large portion of potentially biologically available phosphorus in the sediment. However, the ferric hydroxyl phosphate complexes are redox sensitive, and the covalent bond is relatively weak. When water overlying the sediments becomes anoxic the ferric iron gains an electron, becoming soluble ferrous iron (Fe^{2+}). When this occurs, the complexed phosphate is released from the sediment into the water column.

Elevated internal phosphorus loads that go unmanaged can also lead to a feedback cycle of increasing productivity. Essentially, if the internally released phosphorus leads to increased productivity in the epilimnion, usually reflected in cyanobacteria densities and chlorophyll *a* concentrations, this in turn can lead to more acute anoxia in the hypolimnion through an increase in the biological oxygen demand (BOD). More severe anoxia then increases the total release of sediment bound phosphorus, starting the cycle anew. Since the hypolimnion loses oxygen through the respiration associated with bacterial decomposition of organic matter while it is thermally separated from the surface waters, the increased productivity in the epilimnion that results from internally released phosphorus will result in more organic matter to be decomposed in the hypolimnion upon senescence of that load.

2.3.3 IN-LAKE INTERACTIONS AND FATES OF PHOSPHORUS LOADS

The vast bulk of the external phosphorus load enters the lake as surface runoff and tributary discharge, with the exception of groundwater exchanges which can occur throughout the basin. A considerable amount of the current external phosphorus load to Lake Hopatcong is likely in the form of groundwater. While groundwater naturally contains some nutrient loads, especially nitrogen compounds because they are so soluble and thus easily move through groundwater, wastewater leachate from non-sewered areas increases the concentration of groundwater nutrients as does the use of fertilizers and other land use activities associated with developed lands. These types of loads are likely to be preferentially introduced in shallower, near-shore areas within the littoral zone of the lake. As such, the external phosphorus load is generally introduced at or near the surface of the lake where it can be assimilated by algae or plants. Alternatively, the introduced phosphorus can also continue to exist in a dissolved state, precipitate on contact with iron (Fe) and other metals, eventually settle at the bottom of the lake, or be flushed out of the lake. In a phosphorus limited system such as Lake Hopatcong, a large majority of the dissolved inorganic phosphorus load introduced to the lake from the watershed is assumed to be assimilated by plants and algae during the growing season, or flushed out of the system. Since phosphorus has a high affinity to precipitate with cations present in the soil, a portion of the external phosphorus load is in the form of particulate phosphorus, attached to soil particles. Particulate phosphorus, especially larger coarse particulates attached to soil, will settle to the lake bottom after it is introduced to the lake; however, due to internal load mechanisms described above, this particulate phosphorus can potentially be released as dissolved inorganic phosphorus under reducing conditions.

In the hypolimnion, phosphorus arrives from the surficial layers in the form of soluble or particulate phosphorus. However, once the lake becomes stratified, only the heavier particulates or precipitates will settle to the lake bottom due to the difference in densities between the warmer epilimnion and cooler hypolimnion. As such, the large majority of phosphorus measured in the anoxic hypolimnion of a stratified lake is assumed to be the result of internally released phosphorus, although phosphorus in the form of detrital rain does settle in the hypolimnion especially following blooms.

There are also multiple ways in which internally released phosphorus in the hypolimnion can be released into the warmer epilimnion. Pulses of internally released hypolimnetic phosphorus that make their way into the epilimnion can often lead to nuisance algae blooms since the epilimnion is within the photic zone where the majority of the phytoplankton are positioned. The three main ways in which the entrainment of phosphorus across the thermocline occurs include:

- The anoxic boundary extends into the photic zone of the epilimnion where phytoplankton (including cyanobacteria) are actively photosynthesizing
- The lake destratifies or mixes to some extent resulting in the transport and upwelling of the phosphorus rich water into the photic zone of the epilimnion
- Certain genera of cyanobacteria may sink down into the unlit but phosphorus rich waters of the lake, assimilate the available phosphorus, and then because they possess gas vacuoles can buoy back into the photic zone of the epilimnion to photosynthesize and biologically utilize the assimilated phosphorus. This is a metabolic strategy unique to and utilized by a variety of cyanobacteria to effectively capitalize on internally recycled phosphorus and outcompete other phytoplankton (Head et al., 1999)

3.0 DATABASE AND METHODS

This section will provide an overview of the database used for all of the analyses, as well as briefly discuss the sampling and analytical methodologies and describe the importance of each of the measured analytes.

3.1 DATABASE CONSTRUCTION

Princeton Hydro, previously known as Princeton Aqua Science, began conducting monitoring of Lake Hopatcong in the 1980s and has continuously monitored the lake in some capacity every year since at least 1991.

The database for this analysis spans 153 discrete monitoring events from 1991 through 2021. Every year in that timeframe is represented in the analysis with the exception of 1994; the data from this year could not be located. Generally, each year consists of five monthly monitoring events from May through September. Two of the years on record, 1998 and 2002, only had data available for three and four monitoring events, respectively. However, both of these years had data from at least July through September which are at the peak of the growing season and likely the most important months for this study. 2021 had ten total sampling events, including five events from the standard long-term water quality monitoring program as well as five supplemental monitoring events that were covered under this grant. All of the data used from the standard long-term monitoring is from the mid-lake sampling station, known as Station 2.

3.1.1 WATER QUALITY METRICS

All of the base water quality data that was used in this study was collected by Princeton Hydro during the routine monitoring events. These metrics include standard *in-situ* data such as temperature, specific conductance, pH, dissolved oxygen, and Secchi depth. While all of the *in-situ* metrics were compiled and reviewed, the main parameters that were used in the study include temperature, dissolved oxygen, and Secchi depth.

In addition to the *in-situ* metrics, discrete water quality samples were also collected during each monitoring event for laboratory analysis. The discrete metrics that were used in this study include total phosphorus, both surface and deep samples, and surface chlorophyll *a*.

3.1.2 WEATHER DATA

Temperature and precipitation data were compiled and analyzed for this study. Climate records for the database utilized two nearby climate stations, Boonton Reservoir and Aeroflex Airport. The data was downloaded from Northeast Regional Climate Center CLIMOD 2 website (http://climod2.nrcc.cornell.edu/). These stations were the nearest with the most complete records. Boonton records were used from 1991 through 1998, and compared to the 1980 to 2010 climate normals for that station. The Aeroflex records were used from 1999 through the present, and were compared to the 1980 to 2010 climate normals for that station. Please note that 'normal' refers to the average values over that 30-year period and is a standard approach to describing long-term weather trends.

3.2 WATER QUALITY SAMPLING METHODOLOGY

During the standard monitoring events, *in-situ* and discrete data are collected at the mid-lake station, Station 2, once per month from May through September. *In-situ* data is collected with a calibrated multiprobe water quality meter at 1.0-meter increments starting at 0.25 meters below the water's surface and continued to within 0.5 m to 1.0 m of the lake bottom. *In-situ* sampling is generally conducted down to approximately 14.0 m at the sampling station approaching the maximum station depth; the maximum lake depth of 16.7 m is restricted to a very small area outside of Station 2. In addition, water clarity is measured with a Secchi disk.

Discrete water quality samples for the analysis of total phosphorus (TP) and chlorophyll *a* are collected with a Van Dorn sampling device. Total phosphorus samples are collected from approximately 0.5 meters below the lake surface and 0.5 meters above the sediments. Chlorophyll *a* samples are collected from approximately 0.5 meters below the lake surface only. Discrete water samples were appropriately preserved, stored on ice, and transported to a State-certifies laboratory. All laboratory analysis is performed by Environmental Compliance

Monitoring (ECM) of Hillsborough, New Jersey, a state-certified laboratory (#18630). ECM is the preferred contractor because of their excellent detection limits and focus on natural waterbodies rather than wastewater.

3.3 BATHYMETRIC ASSESSMENT METHODOLOGY

A bathymetric assessment is the mapping of water depth in a waterbody. The data from this assessment can be modeled to produce contours (isobaths) of water depth and statistics such as mean depth, maximum depth, and volume of water. The bathymetric study was conducted on 10 May, 11 May, 14 May, 1 June, 11 June, and 21 June. The study area for this project included the main body of Lake Hopatcong and select deep coves that were lacking updated bathymetric data.

For compatibility with future studies, as well as for possible future permitting, the bathymetry data must use elevations and not relative depths (surface water elevation equal to zero). This required the use of an elevation benchmark surveyed with a Leica GS14 survey grade GPS unit. The Water Surface Elevation (WSEL) was measured with a site level and a Philadelphia rod and corrected as necessary. The cited WSEL for the bathymetry was 923.8 NGVD1929.

The study was conducted with a Knudsen Engineering 1612 Echosounder and a Leica GS14 GPS unit. Data was collected along pre-determined transects at 300 foot intervals. The Leica GPS was used in unison with the echosounder to pair sounding data with location.

All data collected was analyzed in-house by Princeton Hydro with two computer software programs. All echosounder data was analyzed with Hypack Max, which allows for the viewing and editing of the raw sounding data. All noise and anomalies within the raw data were edited out. The final sounding data was then imported into ESRI's ArcGIS Geographic Information System (GIS). Within the database, the water depths were converted into NGVD1929 elevations utilizing the WSEL calculated on site during the study.

3.4 KEY WATER QUALITY PARAMETER DESCRIPTIONS

This sub-section will provide detailed information on the key water quality parameters used in this study.

3.4.1 TEMPERATURE AND THERMAL STRATIFICATION

A lake's water temperature is often a primary factor controlling many biological and chemical reactions. Primarily dependent upon solar radiation and secondarily by ambient air temperatures, thermal diffusion is generally aided through wind driven or artificial mixing. Changes in water temperature with depth are primarily dependent upon the degree of light attenuation, water clarity, lake depth, and the topography and vegetative cover surrounding a lake.

The morphology of the lake basin is the primary factor determining temperature distributions throughout the water column. Essentially, shallow basins experience much less spatial variation in temperature distribution throughout the water column than are experienced in deeper basins. An important characteristic of changes in water temperature are the resultant changes in water density. Many deeper lakes within the North American temperate zone experience strong variation in temperature throughout the water column due to seasonality.

Summer thermal stratification results when increasing solar radiation and air temperatures, in conjunction with few days of little wind activity, combine to thermally stratify the water column. Thermal stratification consists of a relatively warm upper water layer, termed the epilimnion, a transition zone, termed the metalimnion or thermocline, and a cold, deep water layer, termed the hypolimnion. The density differences imparted through thermal stratification serves to inhibit wind driven mixing of the water column thereby effectively sealing off the hypolimnetic layer from contact with the atmosphere. This phenomenon has important implications in that bottom waters of thermally stratified systems may become anoxic due to high rates of bacterial decomposition of organic matter and a lack of atmospheric replenishment of dissolved oxygen through diffusion. Resultant conditions of hypolimnetic anoxia include internal sediment release of metals and phosphorus, and reduced fish habitat.

In the late summer and fall, declining air temperatures result in a negative heat income to the lake, and a loss of heat exceeds inputs from solar radiation. Surface waters are thus cooled and induce convection currents which serve to erode the metalimnion of the lake until the water column exhibits a uniform temperature and therefore uniform density. At this point the lake experiences fall turnover. The transition from the final stages of weak summer thermal stratification to fall turnover are often times abrupt, and can occur over a period of a few hours, especially if associated with the high wind velocities of a storm.

Another important impact of temperature is its effect on the solubility of gases. Simply, colder water may hold more dissolved gases, such as oxygen, than warmer water. This phenomenon has important implications in lakes as rates of algal productivity and algal and animal respiration increase during the late summer months when water temperatures are warmest. While algal and plant photosynthesis produces oxygen as a byproduct during the daylight hours, these organisms actively respire during the night thereby consuming dissolved oxygen to metabolize those carbohydrates produced through photosynthesis. As such, increasing algal densities in concert with warm water temperatures may combine to exhaust all available dissolved oxygen.

3.4.2 DISSOLVED OXYGEN AND HYPOLIMNETIC ANOXIA

Dissolved oxygen is crucial to almost all biochemical reactions occurring in freshwater ecosystems. Primary sources of dissolved oxygen are diffusion from the atmosphere and photosynthesis, while sinks are biological respiration and bacterial decomposition of organic matter. The abundance and distribution of dissolved oxygen in a lake system is based on relative rates of producers (photosynthetic organisms) versus consumers (metabolic respiration). Again, as noted above, it is also frequently influenced by the thermal properties of the water column. This affects dissolved oxygen levels not only as a result of stratification, but also in terms of the extent of dissolved oxygen saturation. Simply put, warmer water has less dissolved oxygen retention capacity than does cooler water. As such, the concentration of dissolved oxygen in cooler water is typically greater than warmer water.

As plants (including aquatic macrophytes and single-celled phytoplankton) photosynthesize they take up water and carbon dioxide and through the use of light energy convert those reactants into oxygen and glucose. This serves to increase the net concentration of dissolved oxygen in lakes during the day in the uppermost water layers where there is ample sunlight to support photosynthesis; that area is termed the photic zone. As such, dissolved oxygen concentrations are generally higher in the upper water layers and lower in the deeper water layers due to a lack of photosynthetic activity in conjunction with aquatic animal/bacterial respiration.

As emphasized above, relative concentrations of dissolved oxygen are also due to temperature and density differences throughout the water column. When lakes thermally stratify there is generally a correlated stratification of dissolved oxygen levels. Lower water layers usually contain less dissolved oxygen as they cannot mix with upper water layers whereby dissolved oxygen concentrations would be replenished with atmospheric sources. In highly productive lakes the hypolimnion may become devoid of oxygen due to bacterial decomposition of excessive inputs of organic material. The source of this material may either be from excessive phytoplankton production in the upper water layers that then sink to the bottom when they die (autochthonous) or from excessive watershed derived sediment loading (allochthonous) or more likely a mixture of the two as they are inherently intertwined. Also, as dissolved oxygen concentrations are generally measured during the daytime, when concentrations are highest, there will be far lower concentrations at night when photosynthesis ceases and diffusion is the sole input of oxygen to the lake.

An important consequence of anoxic conditions in the hypolimnion includes both reduced fish habitat and release of metals and phosphorus through internal loading mechanisms. A general guideline for dissolved oxygen concentrations in warm-water lakes is that a concentration of greater than 1.0 mg/L is needed to preclude internal nutrient and metal release while concentrations of 4.0 mg/L and greater should be kept in order to sustain proper warmwater fisheries habitat and even higher concentrations are required for coolwater fishes.

3.4.3 WATER CLARITY

Transparency in lakes is generally determined through the use of a Secchi disk. The Secchi disk is a contrasting white and black disk that is lowered into the lake until no longer visible then retrieved until visible again. The average of those two lengths is termed the Secchi depth. This depth may be influenced by algal density, suspended inorganic particles, organic acid staining of the water, or more commonly a combination of all three. This parameter is often used to calculate the trophic status (productivity) of a lake and as such is a critical tool in lake evaluation. Secchi depths less than 1.0 m are generally associated with reduced water quality due to high concentrations of algae or suspended inorganic sediments and as such is generally associated with impaired quality.

3.4.4 TOTAL PHOSPHORUS

Total phosphorus (TP) represents all species of phosphorus, including organic, inorganic, soluble, and insoluble. Therefore, this measure accounts not only for those dissolved, inorganic species of phosphorus that are readily available for algal assimilation, but also for those species of phosphorus either tightly bound to soil particles or contained as cellular constituents of aquatic organisms which are generally unavailable for algal assimilation.

By monitoring total phosphorus concentrations, the current trophic status of the lake can be determined and future trends in productivity may be predicted. The current concentration threshold recommended by Princeton Hydro for TP concentrations in lakes and ponds to preclude nuisance algal and macrophyte growth is 0.03 mg/L. The NJDEP Surface Water Quality Standard for TP is 0.05 mg/L.

3.4.5 CHLOROPHYLL A

Chlorophyll *a* is the primary photosynthetic component of all algae and as such is often used as a proxy indicator of total algal biomass. Increases in chlorophyll *a* concentration are generally attributable to increases in total algal biomass and are highly correlated with increasing nutrient concentrations. As such, elevated chlorophyll *a* concentrations are an indicator of increased nutrient loading within a waterbody.

Chlorophyll *a* concentrations above 6.0 µg/L are generally associated with eutrophic conditions. Through analysis of many regional waterbodies Princeton Hydro has determined that concentrations above 20.0 µg/L are generally perceived as a water quality issue by those who utilize the lake for recreation. Concentrations above this amount are generally attributed to excessive phosphorus loading and are therefore a visible sign of nutrient impairment.

3.5 PREPARATION OF KEY METRICS

While the water quality parameters and data described above serve as the foundation of our database, those data were then analyzed and used to derive additional metrics required for the internal phosphorus loading analysis. This sub-section outlines those key metrics and how they were derived.

3.5.1 RELARIVE THERMAL RESISTANCE TO MIXING (RTRM)

RTRM is a relative value that quantifies stratification as a function of temperature differential and is a useful tool in determining the relative stability of the water column and the location of the thermocline. RTRM is calculated as:

RTRM = (Density of Upper Layer – Density of Lower Layer) / (Density at 5˚ - Density at 4˚)

The RTRM values for Lake Hopatcong were calculated and graphed utilizing Dr. Bob Kortmann's RTRM spreadsheet.

The RTRM analysis produces numeric values at each depth interval of the study water column, which in this case was the mid-lake station of Lake Hopatcong. However, mixed layers within the system, such as a fully mixed dimictic lake during fall turnover, or the epilimnion during the summer stratification period, result in RTRM values of 0 due to the uniformity in temperature and water density. Higher values indicate a stronger resistance to mixing due to density differences throughout the water column. The depth interval with the highest RTRM value is the planar thermocline, or the point in the water column in which temperature change with depth is the highest. The sum of the RTRM values for each depth interval in a water column represent the overall resistance to mixing of the water column; this generally occurs during the peak of the growing season.

3.5.2 ANOXIC BOUNDARY

The anoxic boundary represents the depth within the water column at which anoxic conditions (DO < 1.0 mg/L) begin. Since stratified lakes do not as a rule exhibit anoxia, it was deemed important to instead focus on measured anoxia rather than thermocline boundary; indeed, stratified oligotrophic (low productivity) lakes may not exhibit a substantial oxygen depression at depth. Conceptually, the lake is viewed as two zones: an anoxic zone, roughly equivalent to the hypolimnion, and the oxic zone, equivalent to the epilimnion.

Anoxic boundary depth was calculated as an interpolation of DO concentrations between the two depth intervals where the water column becomes anoxic (DO < 1.0 mg/L) using the equation:

Anoxic boundary = D1 + (((O1 – 1) * (D2 – D1)) / (O1 – O2))

where D1 = upper depth interval, D2 = lower depth interval, O1 = DO concentration at the upper depth interval, and O2 = the DO concentration at the lower depth interval.

4.0 RESULTS

This section of the report will describe the analysis of the constructed dataset. The analyses are quite wide ranging, describing a number of facets of lake ecology and water quality dynamics in Lake Hopatcong, but the basis of all analyses is the use of measured data. As such, this avoids some of the pitfalls of using models and takes advantage of the extensive and robust long-term monitoring efforts on the part of various parties. The main focus is on internal TP loading, but this must be described in the context of other measured processes to gain a more thorough understanding of the system and how the internal load effects the ecology of the system. This section will primarily focus on the first of the primary research topics covered in the introduction: the internal phosphorus loading dynamics of the lake and changes to the load over time.

4.1 KEY METRICS

The base metric of this report is the anoxic TP load. Internal TP loading is most closely associated with the release of phosphorus bound in the sediments under anoxic conditions. Anoxia is measured widely throughout the growing season and is typically first measured in early June. For this report, anoxia was assumed to develop at DO concentrations of less than 1.0 mg/L and was used as an indicator of internal loading. The anoxic zone therefore includes the portion of the lake that is anoxic. This is typically contiguous with the hypolimnion, that deep layer of water extending from the lake bed to the thermocline or metalimnion when the lake is stratified. However, this is not always the case; at times the lake may be stratified, yet DO concentrations are greater than 1.0 mg/L within the hypolimnion. This most typically occurs early in the growing season while temperatures remain relatively cool and before all available oxygen has been depleted. The anoxic load represents the total quantity of TP present in the anoxic zone. This is calculated by multiplying the volume of the anoxic zone by the measured hypolimnetic TP concentration which was grabbed at a depth 0.5 m to 1.0 m above the lake bed. The anoxic load was calculated for each of the 153 sampling events for which data was available conducted from 1991 through 2021. The anoxic load was very dynamic, reflecting changes not just in TP concentration but in the volume of the anoxic zone.

The anoxic zone volume and area was derived using several datasets. First, the depth to anoxia was calculated for each event using the relative resistance to thermal mixing (RTRM) calculator, which uses the measured DO and temperature profiles as the primary input. Next those depths were translated to volumes and areas using the hypsographic data generated from the bathymetry study conducted by Princeton Hydro in 2021 as part of this grant. The volumes and areas were adjusted for lake stage at the time of sampling. From 2007 onward daily mean lake stage was calculated using the United States Geological Survey (USGS) gage at Landing, New Jersey (USGS 01455400). Daily records were not available prior to this, although the gage has been active since at least 1928. Prior to 2007, monthly averages were used as surrogate values. Overall, the effect of stage adjustment was relatively minor. In addition, there was no available DO data from 2001. Instead, the anoxic boundary was set at 8.87 m, the mean value over the remainder of the dataset.

Next the anoxic load was calculated by multiplying the anoxic zone volume by the measured TP concentration collected near the lake bed. As described above, the source of TP loading in the anoxic zone is varied, but is primarily assumed to be the product of internal phosphorus release.

This was further refined by producing an equivalent anoxic load. It was important to assess the net changes in the loads between events. In instances in which there was no anoxia the anoxic boundary from the successive event was used. This helped establish a baseline or ambient load within that zone prior to the onset of anoxia to better describe daily anoxic loading rates using the net change. Such adjustment was relatively infrequently used and was primarily employed during the first event prior to stratification or anoxia. The loads were generally quite small reflecting not only a very deep thermocline, but low TP concentrations prior to seasonal accumulation.

A complementary load, here termed the oxic load, was calculated for the water column. The oxic load therefore incorporated the volume of the lake above the anoxic zone or equivalent. This allows further analysis of the data.

When summed with the anoxic load, it produces the total load. It also allows for comparisons between the two zones. The oxic zone again may be considered in most instances equivalent to the epilimnion and metalimnion. This is by far the more biologically active area due to increased DO concentrations and availability of sunlight; it is also where lake users observe and interface with the lake.

The load was calculated in a similarly way. The volume was calculated based on the equivalent anoxic zone boundary, but instead of measuring from that boundary down as for the anoxic zone, it calculated the volume of the lake above the boundary. It similarly adjusted the volume based on lake stage at the time of sampling. This volume was then multiplied by the concentration of the surficial TP sample to derive the oxic load.

Most of the other metrics and analyses discussed in this section are derivations of these loads, examined in different ways and timescales, focus on the base inputs and component parts (such as concentrations and volumes), or include other data collected at the same time, including Secchi depth and other measures. The calculations though, are all completed initially at an event level and represent the conditions at the time of sampling. As mentioned above, these loads are very dynamic over time. As such, the data will be examined in different time domains. This includes both intra-annual and inter-annual scales, and in some instances a monthly basis. Below, the results of the analysis will be discussed at the two primary time scales.

4.2 INTRA-ANNUAL ANALYSES

The intra-annual time domain is based on presenting data in year days, which is the numerical day of the year, related to but different than date. This allows sampling data over the 30+ year dataset to be grouped together or binned in neat blocks. This adds significantly more precision to the results allowing examination at a finer scale than months. Generally, a ten-day cluster was used with the exception of the first and last bin representing outlying tails of events conducted early or late in the year. A table relating the year day bins and the equivalent date range is provided below (Table 4-1). The intra-annual results are a way to track and convey the seasonal progression of ecological dynamics covering the examined period. This time domain will be described first as a way to better describe seasonal dynamics prior to examining inter-annual trends in the database.

Table 4-1: Year Day Legend

Year Day	Equivalent
89-140	March through Mid May
141-150	Late May
151-160	Early June
161-170	Mid June
171-180	Late June
181-190	Early July
191-200	Mid July
201-210	Late July
211-220	Early August
221-230	Mid August
231-240	Late August
241-250	Early September
251-260	Mid September
261-270	Late September
271-290	Late September through October

4.2.1 ANOXIC LOAD AND LOADING RATES

The anoxic load is at the heart of this effort. As described above, it integrates anoxic zone volumes and concentrations. Figure 4-1 shows the mean anoxic TP load. In general, the anoxic load is minimal through late June, but rapidly increases before peaking or plateauing in mid-August with a mean of approximately 2,500 kg. That increase is consistent with a major expansion in anoxic volume as the water warms and the thermocline migrates up in the water column, DO becomes depressed at depth, and TP concentrations rise as a consequence of induced internal loading and its accumulation in the anoxic zone. Thereafter it starts to fall as a result of metalimnetic erosion and downward migration of the thermocline. TP concentrations often continue to rise during this time and most of the load reduction is reflected in a shrinking anoxic zone. At times the lake appears to undergo a partial mixing event when stratification breaks down, but then reforms briefly as water temperatures cool. In any case, the September and October loads average well over 1,000 kg per bin and remain significantly elevated relative to the spring events. It should be noted that some of the variability in the figures is related to event frequency in each bin, which varies from 2 to 16 events, although the median is 9. Additionally, the signal reflects the inherent variability of examining over 30 years of data which reflects not only the typical seasonal variability, but longer term systemic changes in lake dynamics.

The anoxic load was also examined on a net basis, calculating the change in load relative to the previous event (Figure 4.2). This helps to contextualize how the load changes over time. On average, there is no significant expansion of the net load until early July. This peaks in mid-July and again reflects the increased anoxic volume, increasing anoxia, and rising TP concentrations. The net load starts to decline thereafter, eventually reaching no net gain by early September. This indicates that this is a crucial period when continuing internal loading is countered by the loss of that load. By mid-September the net load is negative. This is an important concept. Certainly, the lake remains anoxic at this time and therefore continues to exhibit internal loading. However, the load does not accumulate but instead is transformed and redistributed. Again, this is primarily the consequence of a loss of anoxic volume, but negative net loads likely indicate transport to the upper water column through mixing and metalimnetic erosion, a pulse of available phosphorus to the epilimnion, as well as precipitation of phosphorus and sediment bonding under increasing DO concentrations.

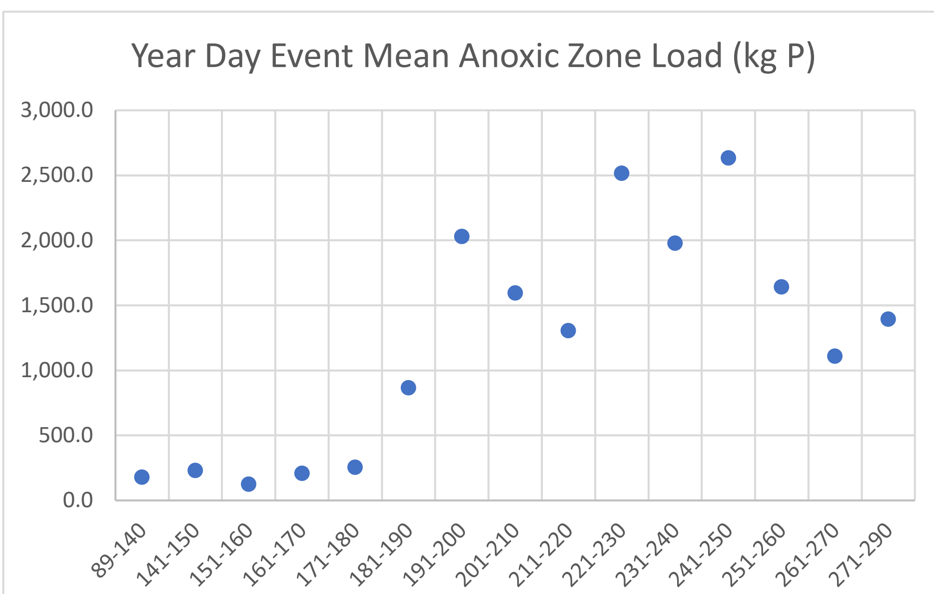

Figure 4-1: Year Day Event Mean Anoxic Zone Load

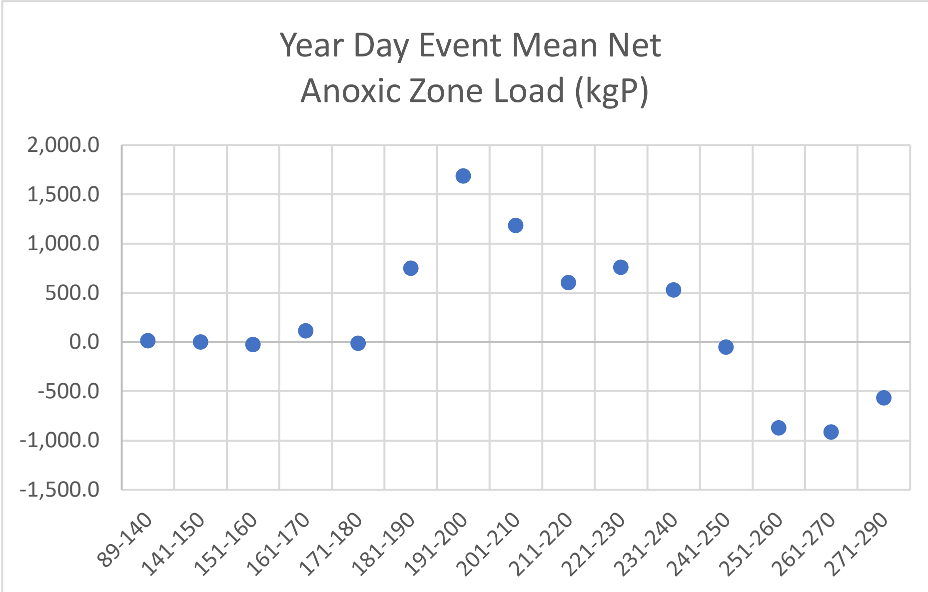

Figure 4-2: Year Day Event Mean Net Anoxic Zone Load

A derivation of the net loads is anoxic zone areal loading rates. These rates are calculated using the event net loads, but are adjusted for both area at the time of the event and the number of days relative to the previous event. The general trend is similar to that described for the net anoxic loads. It should be noted that these values compare very favorably to Nürnberg's published sediment-phosphorus release rates (Nürnberg, 1984). The period of highest loading occurs in mid-July and averages approximately 20.0 mg P m^{-2} d^{-1} (Figure 4-3). From mid-September these loading rates decrease, reflecting a reduced anoxic area in which the loading is occurring, but the negative values must also be viewed, in part, as a pulse to the upper water column and trophic zone of the lake.

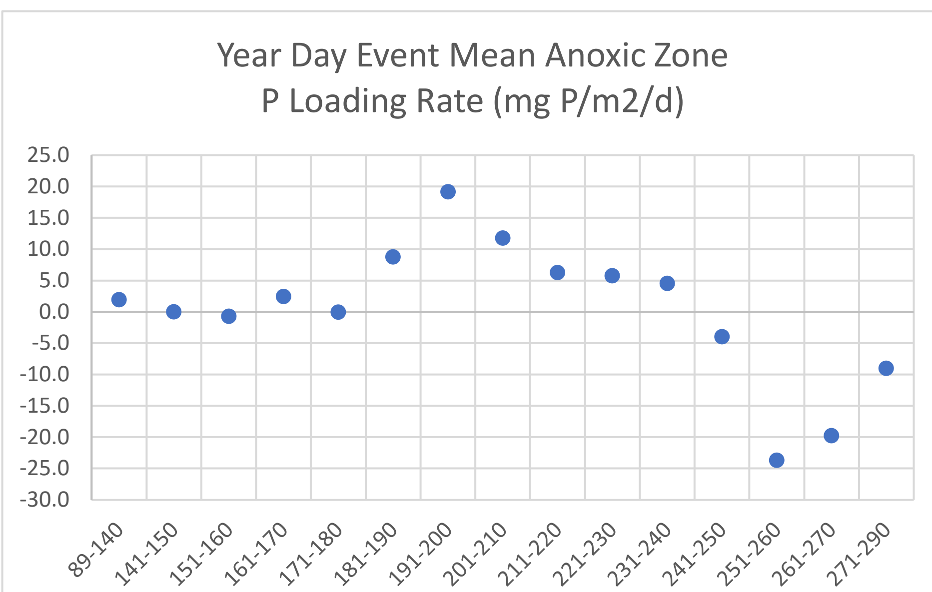

Figure 4-3: Year Day Event Mean Anoxic Zone P Loading Rate

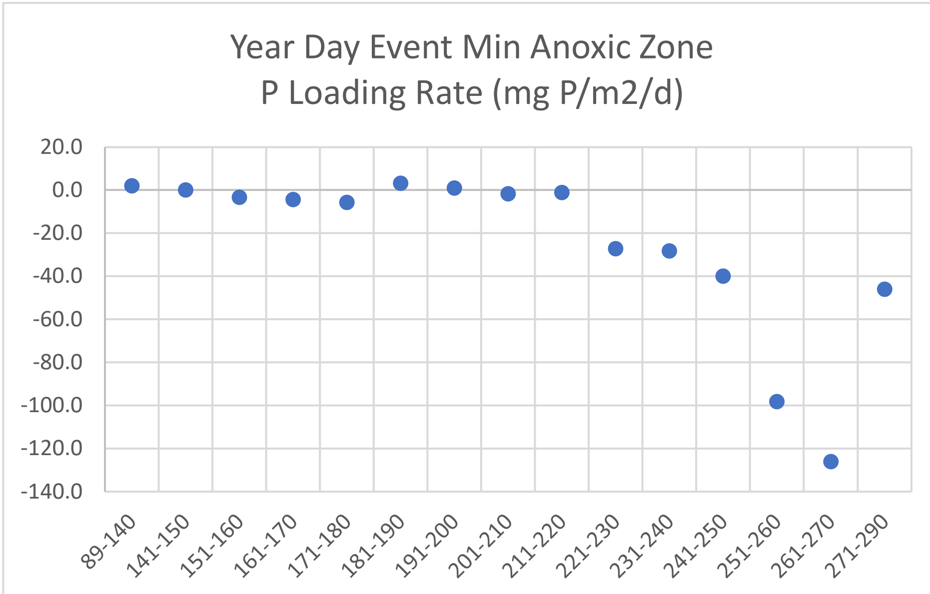

Figure 4-4: Year Day Event Minimum Anoxic Zone P Loading Rate

In addition to the event mean P loading rate, the event minimum P loading rate was explored (Figure 4-4). The plotted data therefore represents the minimum value recorded in each bin from among the constituent events. This shows a different dynamic than the mean loading rate. The negative loading values indicate a net reduction of the anoxic load, and while internal loading is likely still occurring there are other processes that are lowering the load. Here, substantial negative values are observed as early as mid-August. This suggests that extensive nutrient pulses to the trophic zone may be occurring as early as that point, which is often a critical period in terms of cyanobacteria growth and bloom formation. The magnitude of the loading rate is also significant, with four of six events hovering between -25.0 mg P m^{-2} d^{-1} and -50.0 mg P m^{-2} d^{-1}, but two are much higher with one event, recorded on September 23, 1999, reaching -126.1 mg P m^{-2} d^{-1}. This indicates that the rate of reduction can be much more rapid than the slower net positive buildup over time. A review of that date in 1999 shows that the lake had undergone significant mixing with a change in thermocline and anoxic boundary depth of 6.0 m, essentially to the lake bed at the monitoring station, and a 75% decrease in deep TP concentration. Therefore, rapid breakdown of stratification can cause a significant release of phosphorus stored in the anoxic zone to the broader water column at a rate much faster than its accumulation. While that is an outlier event, it is descriptive of the potential for rapid changes to loading dynamics and the distribution of internally generated phosphorus in the lake.

4.2.2 DEEP TOTAL PHOSPHORUS CONCENTRATIONS

A look at the components of the anoxic P load is appropriate, beginning with measured deep TP concentration. Through the end of June, mean deep TP concentrations remain modest, only exceeding a critical TP threshold value of 0.05 mg/L during the mid-June bin (Figure 4-5). Early July marks an inflection point, and concentration begins to rapidly increase as the hypolimnion becomes anoxic consistent with internal phosphorus release from the sediments. It climbs steadily through early September to over 0.25 mg/L, representing the seasonal accumulation of the internal load. It decreases slightly as the lake begins to exhibit signs of vertical mixing in the latter part of September, but remains well above the spring minima. October shows a sharp spike. This seems to reflect the reestablishment of thermal stratification, at least temporarily, and the maintenance of the accumulated load, albeit in a much-reduced anoxic zone. It likely also reflects greater delivery of organic matter from the trophic zone as the growing season algal community senesces. The continued rise in concentrations is a reliable indicator of continuous internal phosphorus release in the hypolimnion during the growing season while the lake is anoxic.

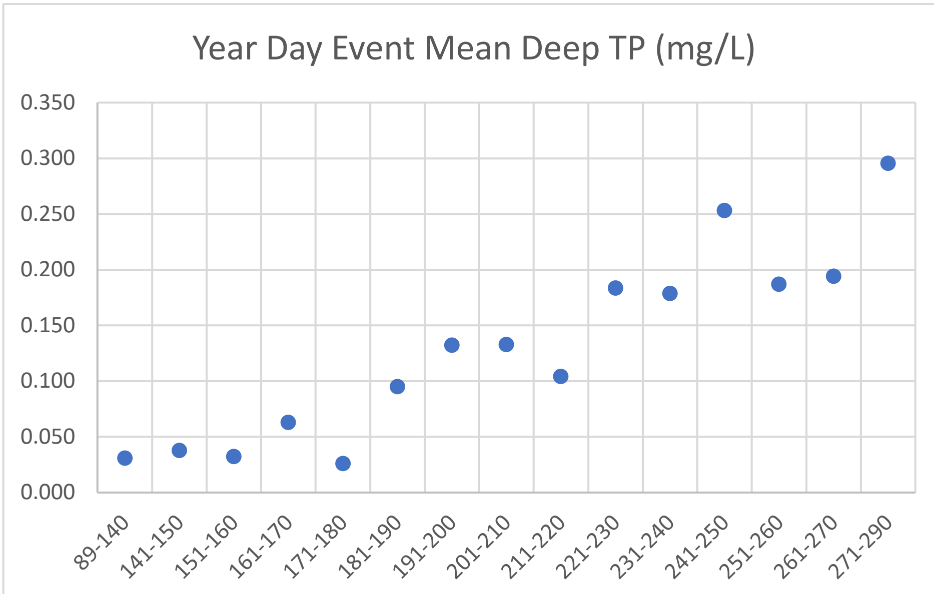

Figure 4-5: Year Day Event Mean Deep TP Concentration

When the maximum values are examined, the pattern is much the same as described for the means, although the magnitude is generally about twice as high. Through late June, concentrations can become elevated, but are generally at 0.10 mg/L and below. A notable exception was on June 12, 2020 when the deep TP concentration spiked to 0.34 mg/L; in part this was precipitated by the sudden and strong onset of anoxia in the lake, but it illustrates that even early growing season concentrations can be quite high. The peak concentration at depth was 0.59 mg/L, sampled on October 6, 2021.

4.2.3 ANOXIC BOUNDARY

The anoxic boundary was also examined and is a critical measure of defining the area and volume of the anoxic zone. Figure 4-6 shows how the boundary changes over the course of the growing season. Please note that this figure is plotted with an inverted depth axis to provide a sectional view of the lake with the surface at 0.0 m and the bottom around 14.0 m. The first onset of anoxia is not noted until early June and is very near the lake bed. It rises steadily through early July averaging 7.2 m through mid-August. It then begins to move downward steadily through the end of September when it exceeds 10.3 m. It is important to note that on average the lake remains anoxic into October, although the anoxic zone boundary is quite deep in the water column. This is another indicator that once anoxia is established in June, portions of the lake remain anoxic well into October driving internal phosphorus releases from the sediments during this time.

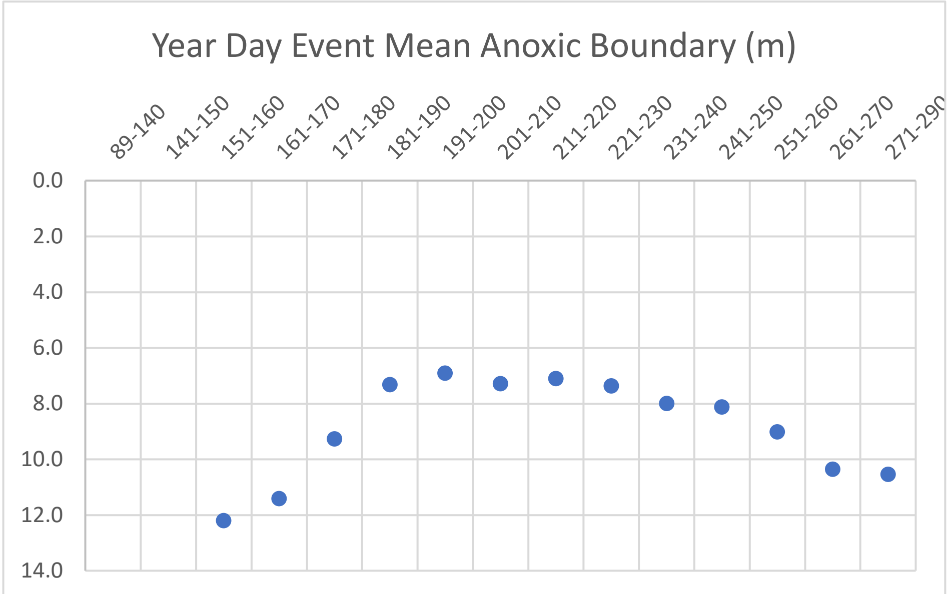

Figure 4-6: Year Day Event Mean Anoxic Boundary

4.2.4 ANOXIC VOLUME

As discussed in section 4.1, the equivalent anoxic volume utilizes an adjustment to establish the background hypolimnetic volume of the lake prior to the onset of anoxia, primarily in the early part of the growing season. When anoxia is present, the established anoxic boundary is used. This is done to compensate for the phosphorus load at depth independent of the internal load, which enables the calculation of both net anoxic loads and derived metrics like internal loading rate. From March through mid-June the equivalent anoxic volume is modest, averaging 6.2 x 10^6 m^3 (Figure 4-7). It quickly begins to rise in late June and reaches a peak in early July at over 16.4 x 10^6 m^3. It then stabilizes through mid-August at an average of 13.3 x 10^6 m^3 and then moves down through October again approaching values observed in the spring at around 4.5 x 10^6 m^3. The equivalent anoxic volume shows general trends similar to the anoxic boundary, but exaggerated. While the anoxic boundary shows relatively minimal movements from early July to mid-August, roughly 6%, the volume varies by nearly 30%. This is because the relationship between depth and volume are not linear, and the area expands at a greater rate than depth moving up in the water column. As such, even relatively small changes in anoxic boundary depth can add significant volume. From late August the volume begins to shrink in a linear fashion through October, although a substantial portion of the lake remains anoxic, on average, in the late growing season.

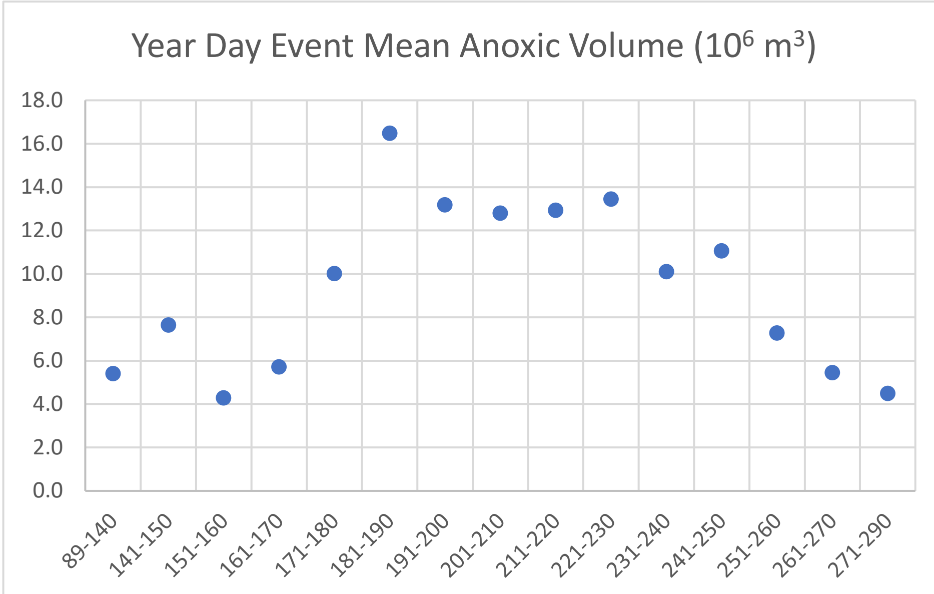

Figure 4-7: Year Day Event Mean Equivalent Anoxic Volume

4.2.5 SURFACE TEMPERATURE

One of the controlling influences on many aspects of lake ecology is temperature. It is particularly important in establishing thermal stratification and anoxia. The onset of thermal stratification establishes the condition in which the hypolimnion is segregated from the epilimnion or the mixed layer. As such, there is no exchange of oxygen with the atmosphere. Because the lake is meso-eutrophic and the hypolimnion relatively warm (often exceeding 10.0°C even during stratification, the available oxygen is quickly consumed through microbial decomposition of organic materials near the lake bed leading to anoxia at depth. At this point internal phosphorus release from the sediments commences.

Surface temperature is an important control on stratification and thermocline dynamics. By early June the lake begins, on average, to exceed 20.0°C (Figure 4-8). During the early spring the lake is well mixed, but as the surface temperature rises the thermocline is established near the lake bed and moves up through the water column. In early July the anoxic boundary and thermocline reach their shallowest depths, yet the lake continues to warm exceeding 25.0°C in mid-July and peaking at over 26.0°C by late July. This continued warming starts to cause the thermocline to begin to migrate down in the column. Surface temperatures slowly start to decrease but continue to hover around 25.0°C through early September. While surface temperatures are falling the lake still remains quite warm continuing to drive the thermocline down in the column, thereby reducing anoxic volume. Surface cooling accelerates through September and October, but average temperatures do not drop below 20.0°C, on average, until October. In the examination of the data 20.0°C seemed to be an important marker for mixing processes in the lake, with notable movements in the thermocline and corresponding changes to anoxic/hypolimnetic volume.

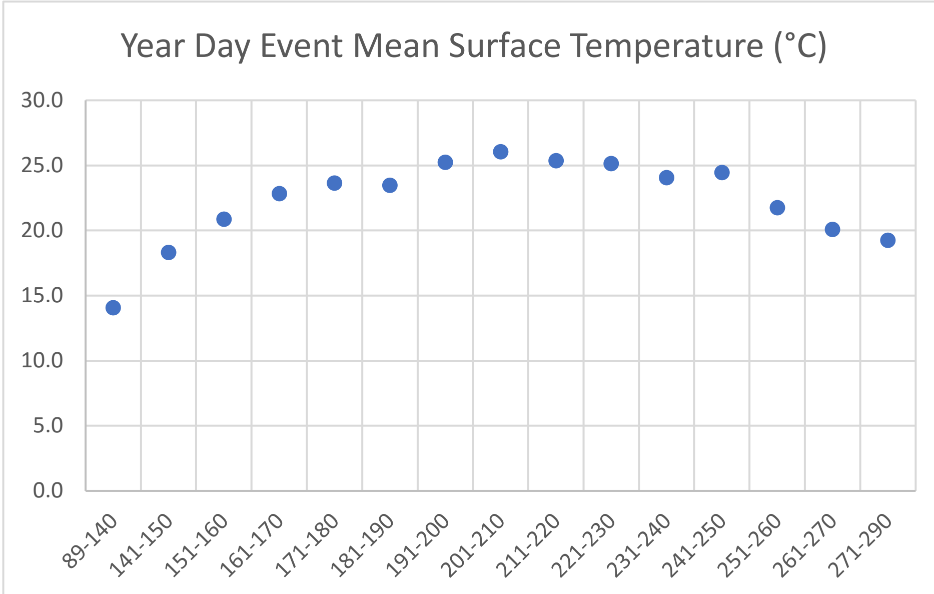

Figure 4-8: Year Day Event Mean Surface Temperature

4.2.6 OXIC LOAD AND LOADING RATES

The sections above chiefly focused on the anoxic load and factors affecting it, the following sections start to examine the oxic loads. It is important to view these in tandem to better understand the ecological impact of the internal load. Like the anoxic load, the oxic load integrates two chief components, the oxic zone volume, the volume of the lake above the anoxic zone, multiplied by the surficial TP concentration. The average oxic zone P load is highest during the earliest portions of the monitoring season averaging approximately 1,270 kg (Figure 4-9). This is driven by higher than average TP concentrations, likely the result of higher external loading from spring precipitation, and high volumes before the onset of anoxia. It is quite variable, but shows an overall decline into early August, reaching the minimum average of just 600 kg. The data is quite variable. Until early July it corresponds to a loss of oxic volume as the thermocline and anoxic boundary are at the shallowest. Thereafter, the oxic volumes start to increase, albeit relatively slowly. TP concentrations, which will be explored in further depth, are also quite variable through this period, but in general are trending lower.

From late July through late August the oxic load is low, but relatively stable. By September though it increases markedly to 975 kg and exceeds 1,085 kg by late September. All facets of lake ecology are experiencing a major seasonal transition from the growing season and late summer into the autumn months, with changes in temperature, stratification, nutrient concentrations, algal biomass, and algal composition among others. The September increases are quite interesting however. Increasing loads during this period are certainly driven by an expansion of the oxic volume as the thermocline migrates deeper in the water column. Yet TP concentrations are also increasing at this point. The cause for increasing TP concentrations is quite interesting, but a significant fraction of the increasing load at this time is believed to represent the loss of the anoxic loads; a pulse of phosphorus into the upper water column caused by metalimnetic erosion and mixing. Much of it too is likely to be rapidly assimilated and expressed in algal cells. It could also represent to some degree the circulation of organic detritus with improved mixing. The load then falls moving into October at an average of 700 kg or about 78% of the total average. Obviously, September is a critical time period for the lake, and significant algal blooms are routinely observed and reported. It is therefore possible that the internal load of phosphorus, which is being distributed into the trophic zone at that time, plays a major role in sustaining high algal biomass at this period.

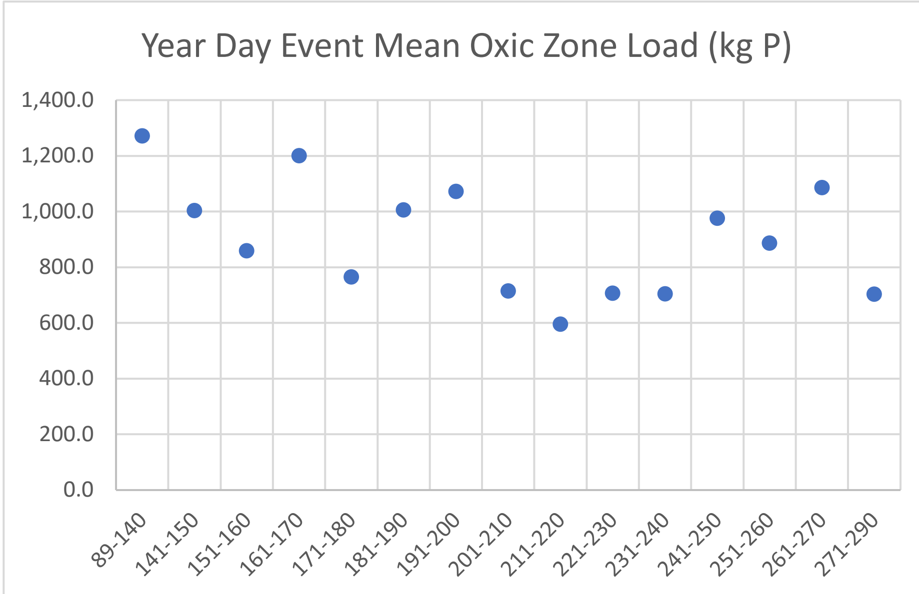

Figure 4-9: Year Day Event Mean Oxic Zone Load

4.2.7 SURFACE TOTAL PHOSPHORUS CONCENTRATIONS

Surface TP concentrations on average vary by a factor of 2 throughout the course of the monitoring season. The surface TP regime may be broken into several discrete units. The first half of the growing season, from March through mid-July, represents one of these units. This is a transitional period from spring to summer growing conditions with all the attendant changes to lake function and ecology. Concentrations are quite variable at this time, but are marked by TP concentrations of 0.025 mg/L on more than 70% of the bins. While these values are generally acceptable, these are means, and concentrations as low as 0.03 mg/L can sustain high algal productivity and primary production. Additionally, throughout this period maximum values exceeded 0.03 mg/L for all bins thus indicating a potential for algal blooms throughout the first half of the growing season.

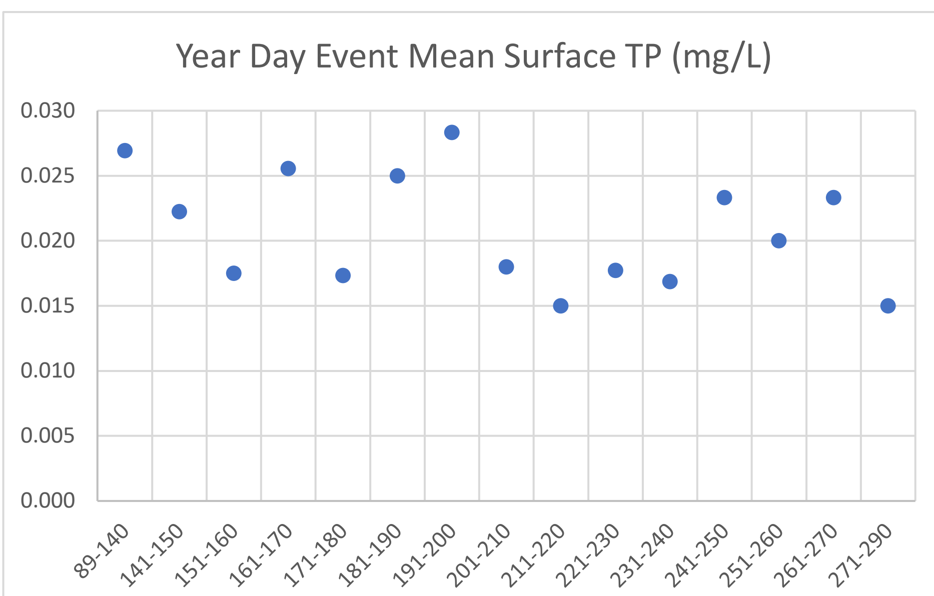

Figure 4-10: Year Day Event Mean Surface TP Concentration

The second grouping occurs over the 40 day period from late July through late August at the height of the growing season. This represents a period of stability and overall low concentrations. Throughout this period TP surface concentrations average just 0.017 mg/L, a favorable value from a management perspective. As with the early growing season though, throughout the dataset all bins also exceeded maximum concentrations of 0.03 mg/L throughout this period. September shows a noticeable increase in TP concentration, ranging on average from 0.020 mg/L to 0.023 mg/L, or a 30% increase relative to the early block. As discussed above, this almost certainly reflects to some extent the expression of the diminishing internal load, which through mixing and other processes, is being transported to the trophic zone.

Finally, moving into October the concentration reaches an annual nadir of 0.015 mg/L. This marks a great reduction in not just the internal load, but the delivery of that load throughout the water column. This is due to enhanced mixing, the dilutionary effect of increasing oxic volume, and likely the precipitation of phosphorus or its binding to the sediments.

4.2.8 ANOXIC AND OXIC LOAD COMPARISON

Finally, it is appropriate to compare the anoxic and oxic P loads. The anoxic load is expressed as a percentage of the total load which is the sum of the anoxic and oxic loads (Figure 4-11). From March through mid-June, the equivalent anoxic load accounts for a small total of the lake's entire phosphorus load, less than 20%. However, with the onset of anoxia, the expansion of the anoxic zone, and the increasing concentration of deep TP due to internal phosphorus release from the sediment it starts to climb by late June, first broaching 22%. The most significant jump occurs in late June, by which point it begins to approach 50%. The rate of increase starts to slacken, but by early September the anoxic load accounts for 72% of the total load. For the most part, when the anoxic load dominates, this phosphorus may largely be sequestered and not transported to the surface strata. However, the thermocline is quite shallow at this time, and it is likely that depth-regulating genera of cyanobacteria may be serving as a vector for phosphorus from depth to the trophic zone.

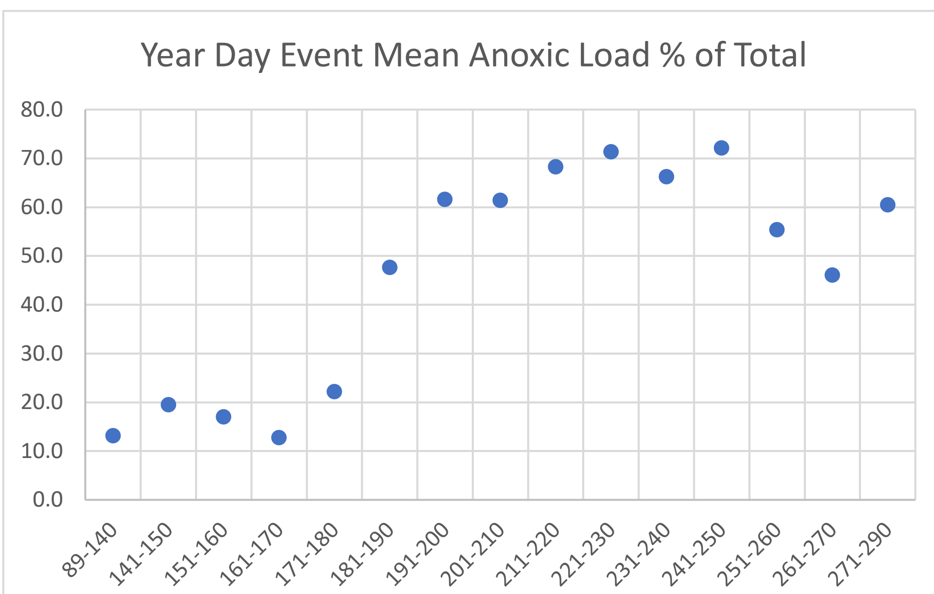

Figure 4-11: Year Day Event Mean Anoxic Load as Percent of Total Load

Mid-September shows the anoxic load fall to less than 60% of the total load, the lowest value since early July. This shift marks not just an increase in the oxic load and a decrease in the anoxic load, but the enhanced availability of this nutrient for algal assimilation, as well as the transformation and transport of phosphorus from the anoxic zone to the oxic zone. Again, this late season shift is believed to be at least partially responsible for sustaining high algal productivity in September in Lake Hopatcong.

4.3 INTER-ANNUAL ANALYSES

The inter-annual analysis shifts time domains to explore how phosphorus load dynamics have changed from 1991 through the present. Here the focus is on annual statistics, rather than the seasonal progression explored through the use of year day bins. Again, the annual data are developed on an event basis.

4.3.1 ANOXIC LOAD AND LOADING RATES

Overall, the anoxic load in Lake Hopatcong has been increasing since 1991 (Figure 4-12). In particular, from 2019 through 2021, the mean event anoxic load has been 2,414 kg of phosphorus, well above any previous year, the next closest being 1999 at 1,591 kg. It is more useful however to consider the data in several discrete blocks. Many of the analyses showed that 2010 was an inflection point of sorts marking a phase shift in lake ecology, although other divisions could be devised. From 1991 through 2010 the anoxic load was very variable, with loads varying by a factor of 5 and year over year changes could be significant. In general, anoxic loads trend down in this time, and from 2003 to 2008 the anoxic load mean did not exceed 1,000 kg. From 2010 onward, loads have risen, particularly in the past three years. From 2010 to 2018 the average annual load was somewhat higher than 1991 to 2009 (997 kg vs 911 kg), but much less variable. Taken as a whole though, 2010 to 2021 averaged 1,433 kg, nearly 57% higher than the earlier period. The potential causes are varied, and will be explored further, but both climate change and general eutrophication are believed to be root causes. In addition, meteorological conditions, well above average rainfall and temperature, which likely reflect climate change, were highly correlated with the conditions observed in 2019.

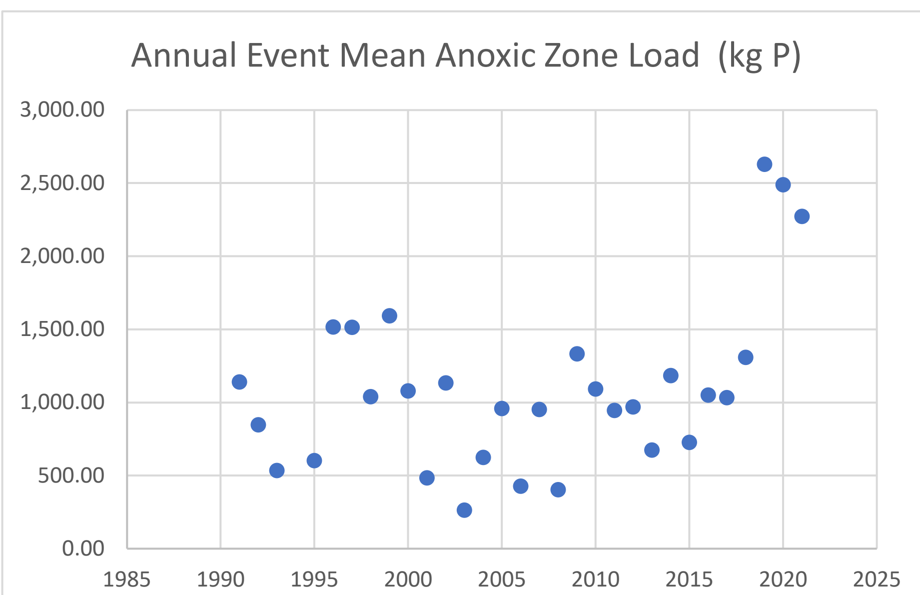

Figure 4-12: Annual Event Mean Anoxic Load

While the use of the annual event mean is useful, the use of the annual event maximum statistic is likely a better representative of the actual internal load of the lake. It more neatly integrates the process of accumulation over time. The use of a mean statistic is hampered by the fact that it includes events when the lake was not stratified and internal loading was insignificant. The annual event maximum is plotted in Figure 4-13. It is quite similar to the event mean figure above, but the magnitude is much greater. In the 18 years of events from 1991 through 2009, the annual maximum anoxic load only exceeded 3,000 kg during four years (22%). In the 12 years since, from 2010 through 2021, this has been broached six times (50%). From 1991 to 2009 the median anoxic load maximum was 2,204 kg; from 2010 through 2021 it is 2,932 kg. The averages showed a similar disparity, 2,347 kg vs 3,293 kg. No matter how the data is parsed, there have been substantial increases in the anoxic load over the last decade.

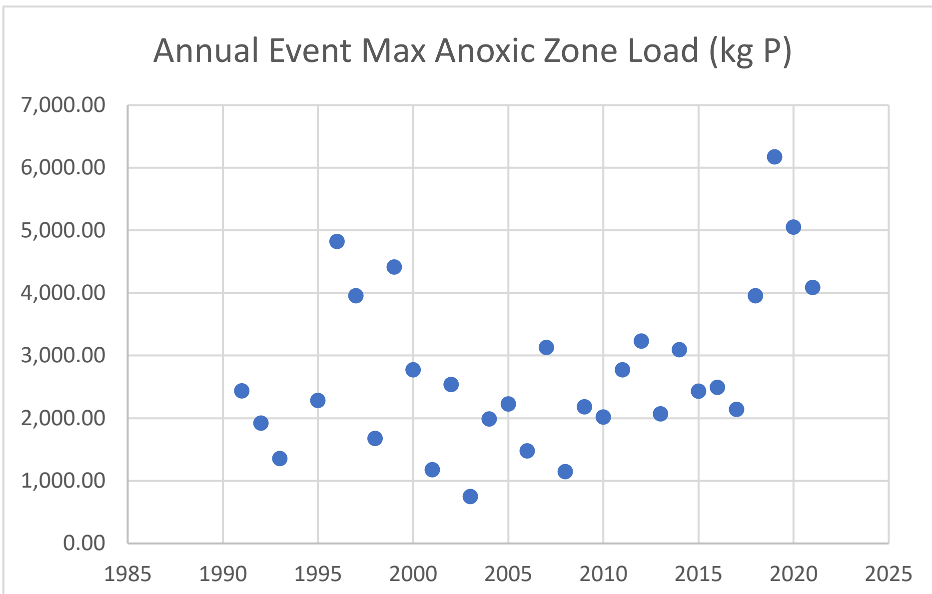

Figure 4-13: Annual Event Maximum Anoxic Load

The intra-annual analysis introduced the concept of using the net anoxic load statistic. It further introduced the concept that net negative events could be considered the release of phosphorus from the anoxic zone into the trophic zone during that critical late growing season period. These releases are often associated with metalimnetic erosion and partial lake mixing events as the thermocline descends through the water column. The same analysis is repeated here. The annual event minimum net anoxic zone load describes the event at which the anoxic zone lone showed the greatest decrease during the year. This decrease is linked to the transport or mixing of this P load with the epilimnion, although other losses including precipitation, settling, and sediment bonding are part of the loss.

Figure 4-14 below shows the minimum net anoxic zone loads, to be interpreted as maximum P pulses to the water column on the whole. The same dynamics are at play here as described above. From 1991 through 2009 the greatest net anoxic load exceeded -1,000 kg during 8 of 18 years (44%). From 2010 through the present this was exceeded all 12 years (100%). The average minimum was -1,137 kg from 1991 to 2009, while it was -1,932 kg from 2010 to 2021. From 2010 until now, the pulsed events are less variable than in the past, but consistently substantially higher. Combined with the base analysis of loads, this shows that not only are total anoxic loads increasing, but that those loads are creating greater phosphorus pulses in the system. From a mechanistic perspective, the generation of greater loads of course must also necessitate greater distribution of the load, but from an ecological perspective it indicates that there is potentially greater availability of that load during the late growing season.

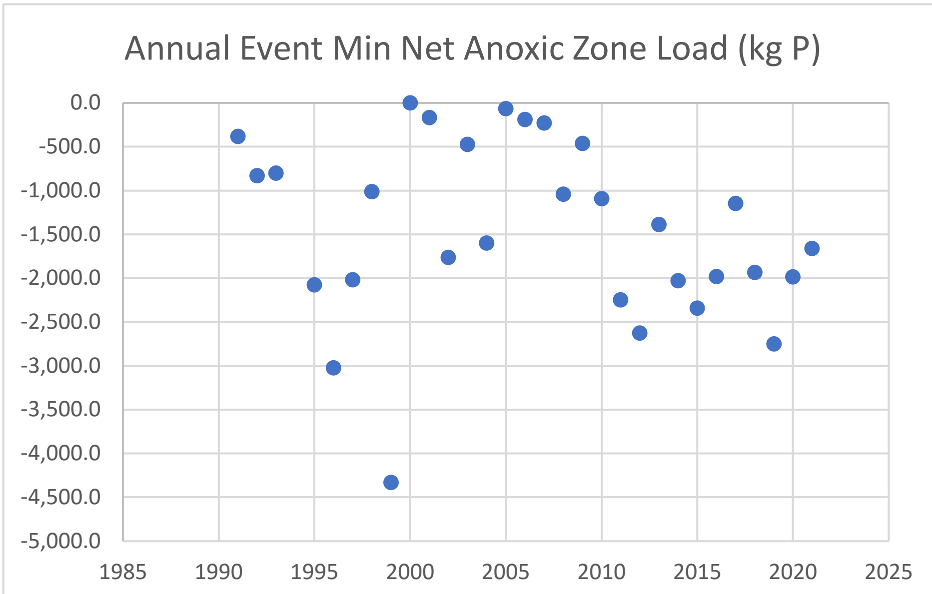

Figure 4-14: Annual Event Minimum Net Anoxic Load

It is also interesting to consider the loading rates. As before, there has been a bit of a phase shift around 2010; prior to that time the loading rates were quite variable (Figure 4-15). Since 2010, there has been a strong increase in areal loading rates, and depending on how the trend is defined, this may stretch back to as early as 2000. This indicates that load is increasing not just a function of increasing anoxic volume, but that on a unit area basis the loading rate is higher. This would correspond well to increases in deep TP concentration. It also suggests that conditions are more suitable for the release of phosphorus from the sediments, which could be related both to the stability and duration of stratification in the lake, or that sediment phosphorus concentrations have increased, perhaps as a function of eutrophication. The minimum loading rates, which are negative and indicate the loss of phosphorus in the anoxic zone, presumably to the upper water column, have also increased in magnitude similar to the net loads shown in Figure 4-14, indicating stronger release of accumulated phosphorus upon mixing.

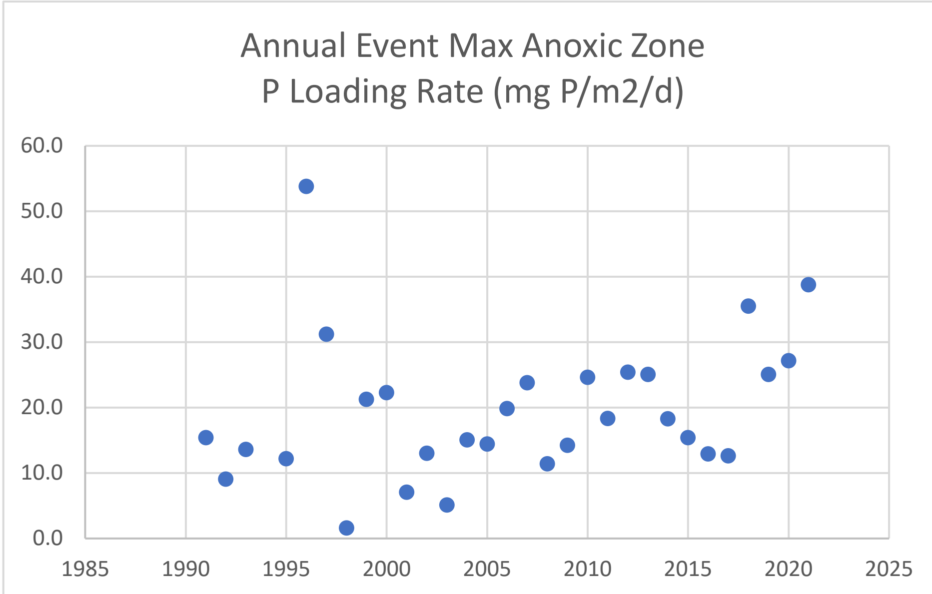

Figure 4-15: Annual Event Maximum Anoxic Zone P Loading Rate

It is also appropriate to review the data in a historical context. In the *Lake Hopatcong Management/Restoration Plan* developed by Princeton Aqua Science in 1983 for the Lake Hopatcong Regional Planning Board, the estimated internal phosphorus load was just 595 kg annually. This was equivalent to 14% of the total phosphorus load to lake which also included tributary inflow, non-point sources, wetfall/dryfall, point sources, and septic tank effluent. It was based on a conservative estimate of loading rates of 6 mg P m^{-2} d^{-1} and a duration of anoxia of just 60 days. This review, based on measured data, rather than models, shows that maximum loading rates are, at times, significantly higher. This must be balanced against the loads developed here. Loading attributable under the original modeling would have reflected the release of soluble reactive phosphorus (SRP), while the measured data includes accumulated SRP as well as particulates and detrital rain. As such, these are not direct comparisons, yet it seems likely that the majority of the measured load, particularly given its seasonal dynamics, directly reflects the SRP product of internal sediment release. Second, the duration of anoxia is also quite limited in the original assumption, although that too was based on field measured concentrations. In this dataset, it seems apparent that anoxia persists for far longer generally starting in early June and continuing well past the cessation of monitoring in October, suggesting that more recent data shows the lake goes anoxic for probably closer to 120 to 150 days, or at least double the earlier estimates.

Only once since 1991, has the calculated maximum anoxic load been less than 1,000 kg. While this calculated load is likely higher than the sole measure of SRP alone, it is clear that the internal load is much greater than originally assumed. More significantly, this load has been demonstrated to have increased significantly over time, particularly during the last few growing seasons and similarly accounts for a much greater percentage of the measured total load in the lake than was evident before.

4.3.2 DEEP TOTAL PHOSPHORUS CONCENTRATIONS

Deep TP concentrations have risen significantly over time (Figure 4-16). The trend has been quite steady, although there is considerable inter-annual variability. Mean hypolimnetic TP concentration did not exceed 0.10 mg/L until 1996. From 1996 through 2021, annual mean deep TP was above 0.10 mg/L 21 times (81%). Since 2019, the average concentration has been 0.208 mg/L, more than double those measured 30 years prior. It is very clear that concentrations are increasing, and greatly impacting the anoxic load.

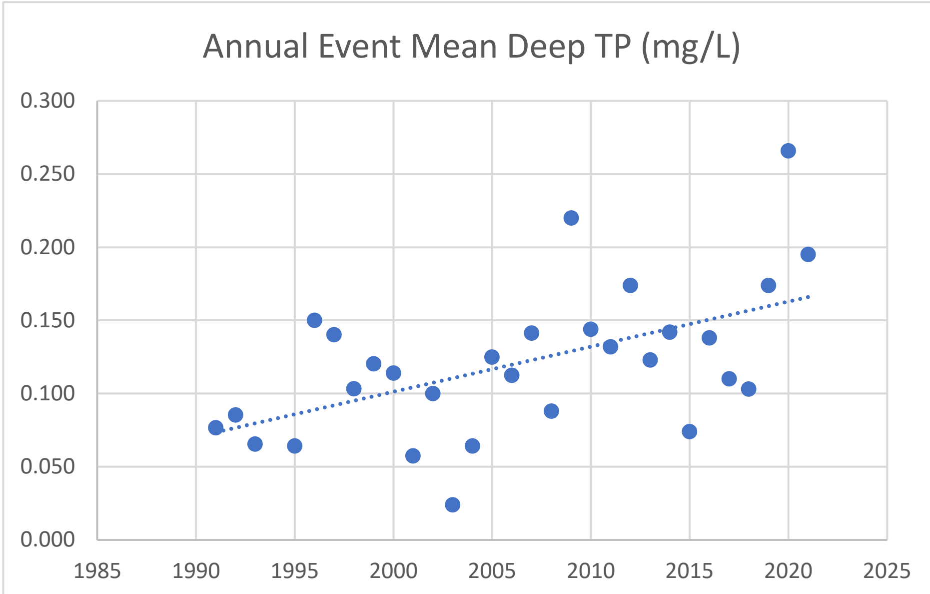

Figure 4-16: Annual Event Mean Deep TP Concentration

The trend is very much the same for the maximum measured deep TP concentration (Figure 4-17). Since 2005, annual maximum deep has exceeded 0.20 mg/L each year. Prior to then, 0.20 mg/L was recorded in just 31% of the monitored years. Four times in the last five seasons the maximum TP has exceeded 0.30 mg/L, reaching an all-time high of 0.59 mg/L in October of 2021. These are extremely high concentrations, and the increase must be in part linked to long periods of anoxia which allows those concentrations to accumulate and increase throughout the period of stratification.

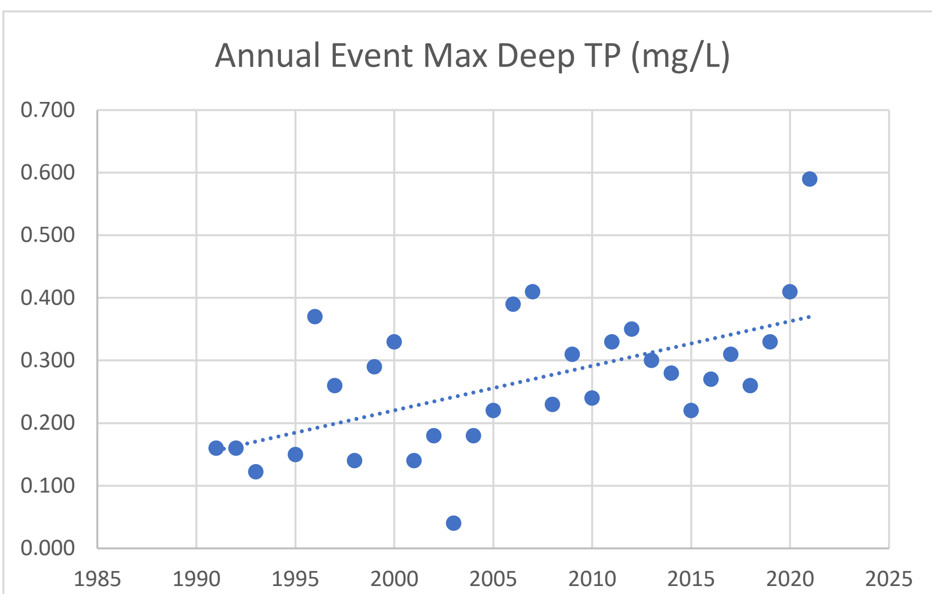

Figure 4-17: Annual Event Maximum Deep TP Concentration

4.3.3 ANOXIC BOUNDARY

The anoxic boundary has in fact shifted over time. From 1991 through 2008 the anoxic boundary was in a state of flux, with a sine wave like oscillation moving down and up in the water column (Figure 4-18). In 2009 and 2010, anoxia (DO < 1.0 mg/L) was not measured, although this is somewhat misleading; concentrations as low as 1.1 mg/L were detected and the sediments at this time were certainly anaerobic. Additionally, elevated TP concentrations during periods of hypoxia (DO < 2.0 mg/L) also confirm internal phosphorus release during this time. The boundary reached its deepest average in 2011 at nearly 11.5 m, courtesy of a single positive detection of anoxia. Since then, however, the annual event mean anoxic boundary has climbed steadily upward in the water column. It should be noted that 2019 had the shallowest average anoxic boundary during the period of record at just 7.1 m recorded over the course of 4 events.

It is also appropriate to examine the annual minimum anoxic boundary at each year (Figure 4-19). Prior to 2010, the minimum anoxic boundary was less than 6.0 m twice (13%). Since 2010, it has been less than 6.0 m five times (45%). This represents a definite upward migration in the water column. It should also be noted that based on hypsographic data lake area starts to increase greatly at around the 6.0 m isobath indicating that small movements in the anoxic boundary near this depth may expose significantly more lake bed to anoxia and thus induce increased loading.

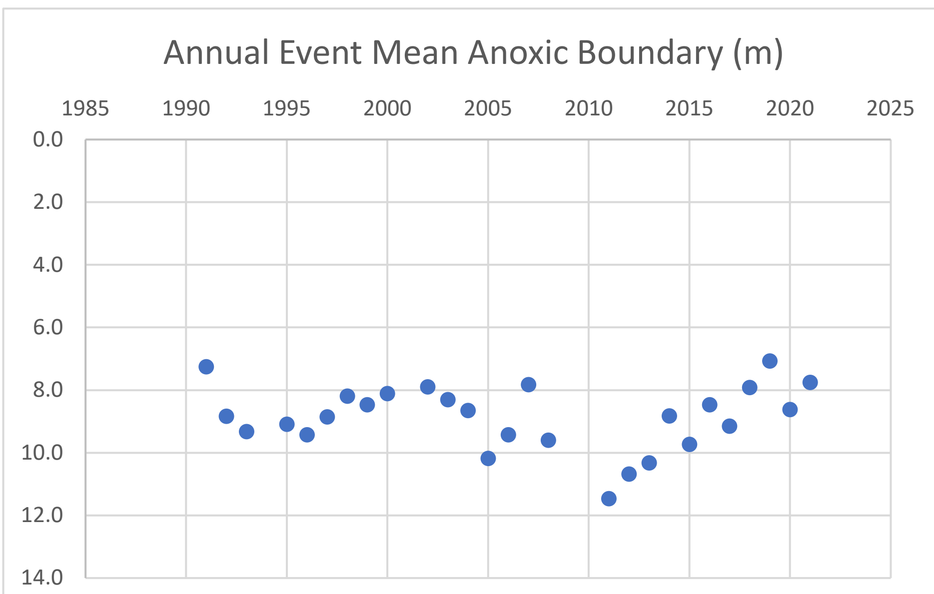

Figure 4-18: Annual Event Mean Anoxic Boundary

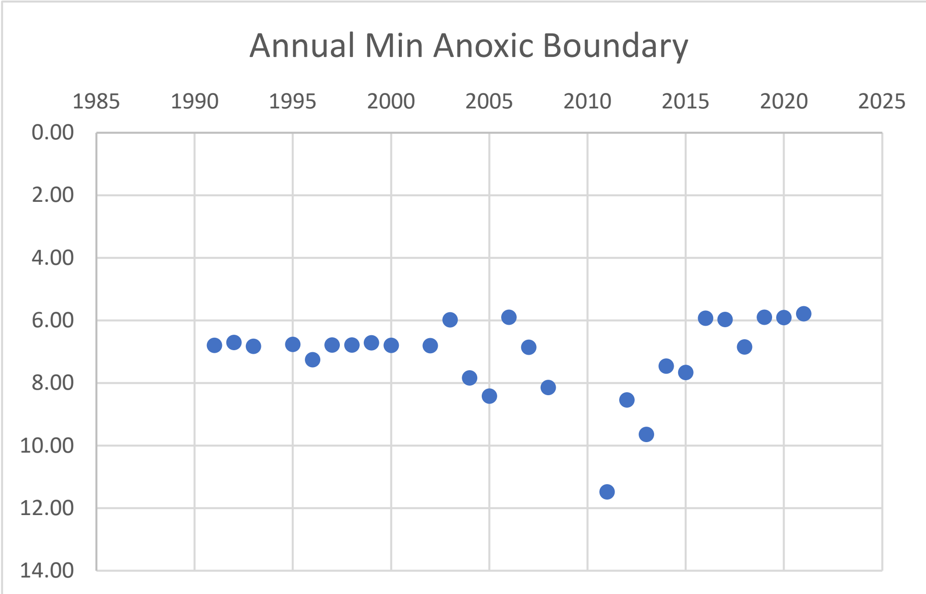

Figure 4-19: Annual Event Minimum Anoxic Boundary

4.3.4 ANOXIC VOLUME

The equivalent anoxic volume was also examined over the course of monitoring. This differs somewhat from the anoxic boundary patterns due to corrections described above including for periods when the lake was not anoxic, typically early in the year, or when it was hypoxic but not anoxic. It is also a more consistent record, because this statistic was calculated for each sampling event.

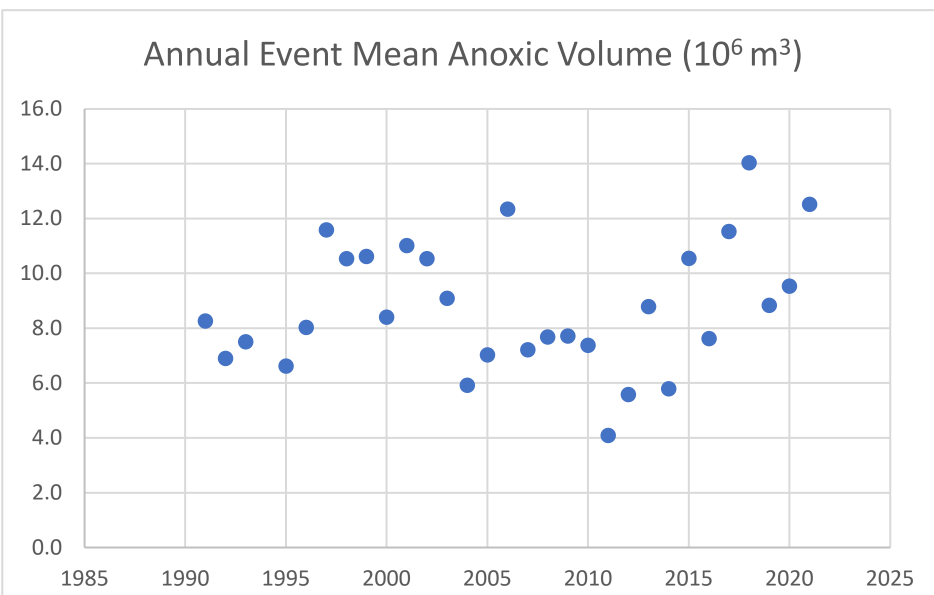

Figure 4-20: Annual Event Mean Anoxic Volume

Since 1991, the growing season average anoxic zone volume has increased (Figure 4-20). This increase is not linear however. There was a relatively steady increase from the onset to around 2005. For a period of 6 or 7 years around this time it decreased substantially, bottoming out at just 4.1 x 10^6 m^3 in 2011. Since then, it has increased at a very rapid pace, although successive years seem to evince an alternating step-wise pattern. The two highest years in the dataset have occurred within the last four monitoring seasons, including 2021. The pattern is largely the same for the maximum anoxic volume (Figure 4-21). Since 2015 the maximum volume has exceeded 17.5 x 10^6 m^3 all years but one (86%), while prior to that time that volume was exceeded just twice (9%). This again suggests, that at least for some crucial period in the growing season, the anoxic volume has considerably expanded since 1991.

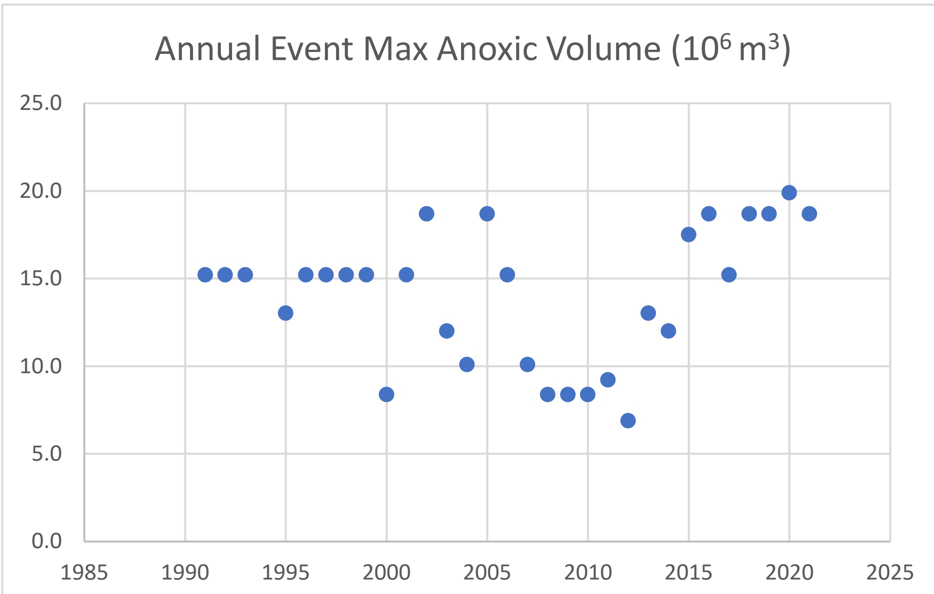

Figure 4-21: Annual Event Maximum Anoxic Volume

4.3.5 OXIC LOAD AND LOADING RATE

The oxic load data is extremely interesting. The anoxic load has shown a substantial increase by nearly all examined parameters and driven largely by increasing anoxic zone volume and increasing phosphorus concentrations. For the oxic zone load the reverse is true, and it has shown strong decreases in average value since 1991 (Figure 4-22). Again, using 2010 as a handy point of demarcation, since that time the average oxic load has exceeded 800 kg P during 33% of the years, while prior to that time that mark was bested during 66% of the sampling years. For the reasons explained for the anoxic zone metrics, it is useful to examine the annual maximum oxic loads as well (Figure 4-23). Since 2010 the mean value of the annual maxima has been 1,212 kg, while the period from 1991 to 2009 averaged 1,643 kg. Together, these values show an undeniable decline in trophic zone nutrient loads. This points to very high efficacy in managing the external phosphorus loading at the lake through non-point source management, stormwater management, sewering, and other watershed management practices. They have undoubtedly proven effective.

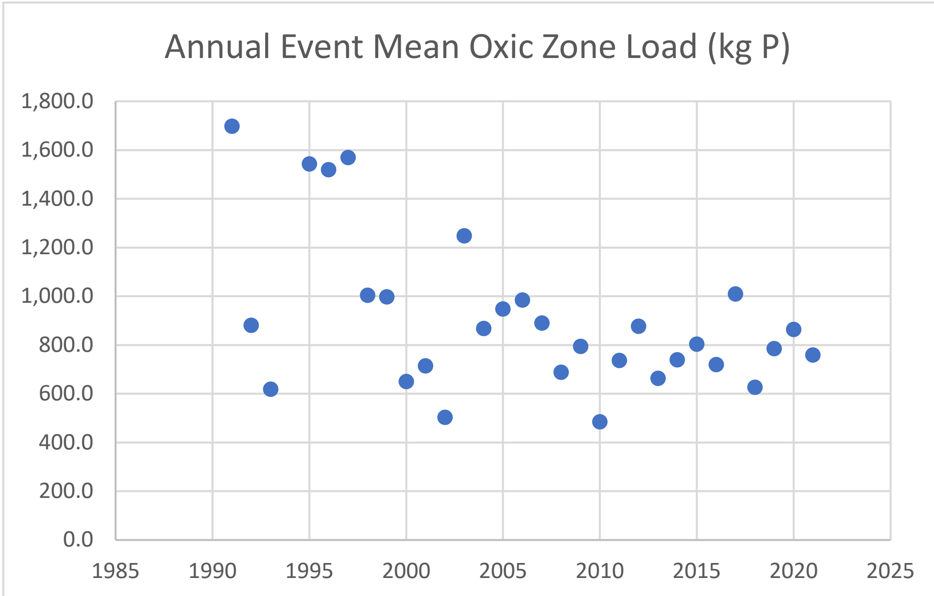

Figure 4-22: Annual Event Mean Oxic Zone P Load

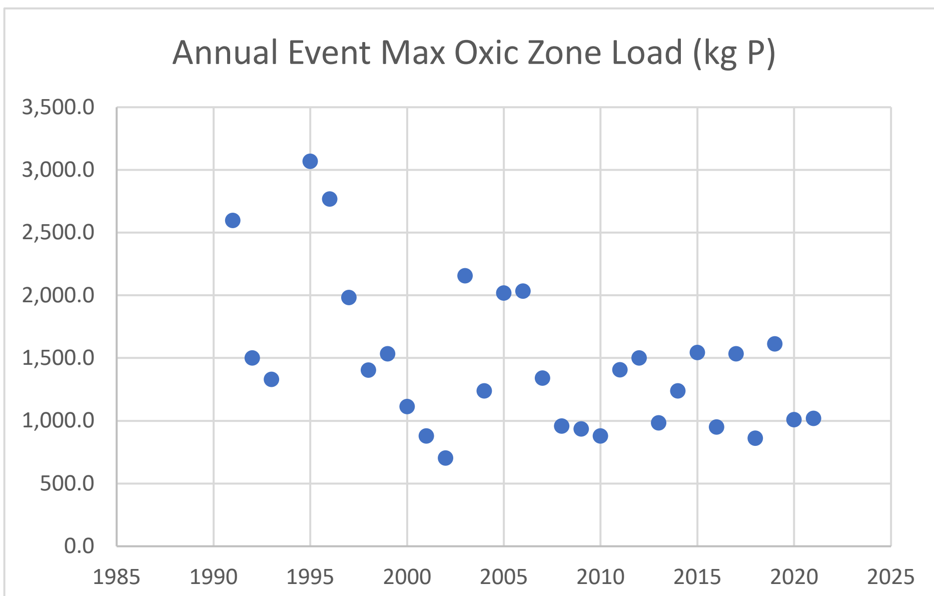

Figure 4-23: Annual Event Max Oxic Zone P Load

One area in which there may be some indication that the oxic zone load is being impacted is an analysis of the net oxic loads, which examines the change in load from event to event. In looking at the annual maxima, the greatest change between successive sampling events, these have shown a decline very similar to the other discussed oxic load metrics, namely that they have decreased substantially (Figure 4-24). Yet, 2010 seems to be another inflection point, and possibly as early as 2005, where the maximum net load, essentially a pulse of phosphorus to the system, has been increasing. 2019, a particularly severe year for the formation of HABs at Lake Hopatcong, saw a maximum net increase of 1,180 kg; this was the highest value since 2003. It represents a significant addition of phosphorus to the photic zone. This recent trend may link to higher loads being generated in the anoxic zone, and with the potential for release of that load into the oxic zone through metalimnetic erosion.

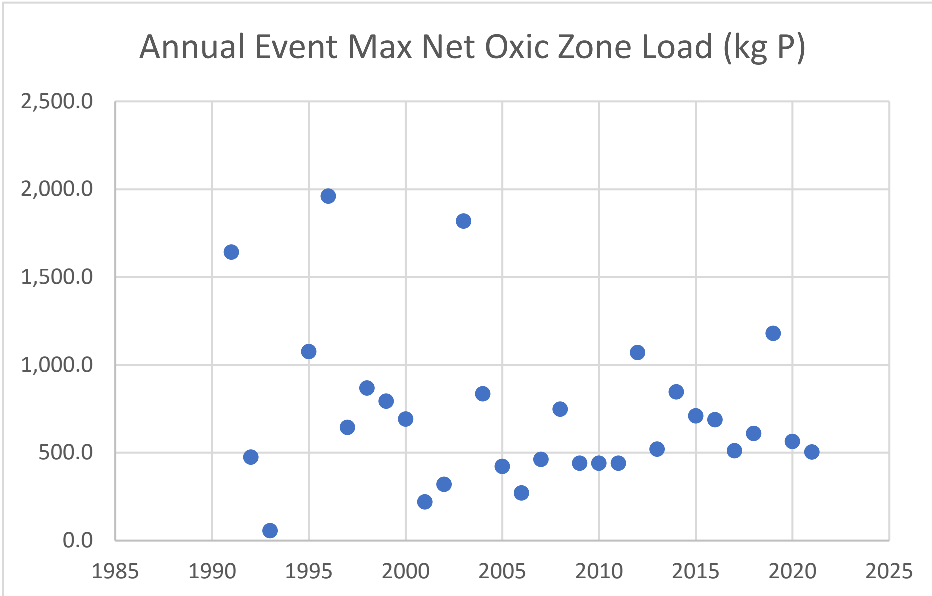

Figure 4-24: Annual Event Max Net Oxic Zone P Load

4.3.6 SURFACE TOTAL PHOSPHORUS CONCENTRATIONS

Surface TP concentrations were also examined. Again, the annual mean surface TP concentrations have fallen significantly since 1991 (Figure 4-25). Prior to 2010 the event mean surface TP concentration exceeded 0.02 mg/L during 61% of the monitoring years, but since then this has been exceeded just 25% of the time. The trend seems to indicate however that those values are showing an increasing trend in that time frame, and those three exceedances all occurred within the last five years. The analysis of the maximum surface concentrations is much the same, although the maximum concentrations are clearly higher than the mean values (Figure 4-26). While the mean values seemed to indicate a recent upward trend, there is no clear indication of that for the maximum values. It should be noted again that 2019 was an outlier, with a maximum concentration of 0.04 mg/L measured at this surface. While that value is higher than desirable, and certainly capable of supporting high-intensity algal blooms, it is a modest concentration and below the state water quality standard of 0.05 mg/L. Overall, the decline in surface phosphorus concentrations mirrors the oxic load data. It also correlates to watershed nutrient control strategies and a reduction in TP loads to the lake. There are signals however that internal loading is contributing to the observed oxic load, particularly late in the growing season.

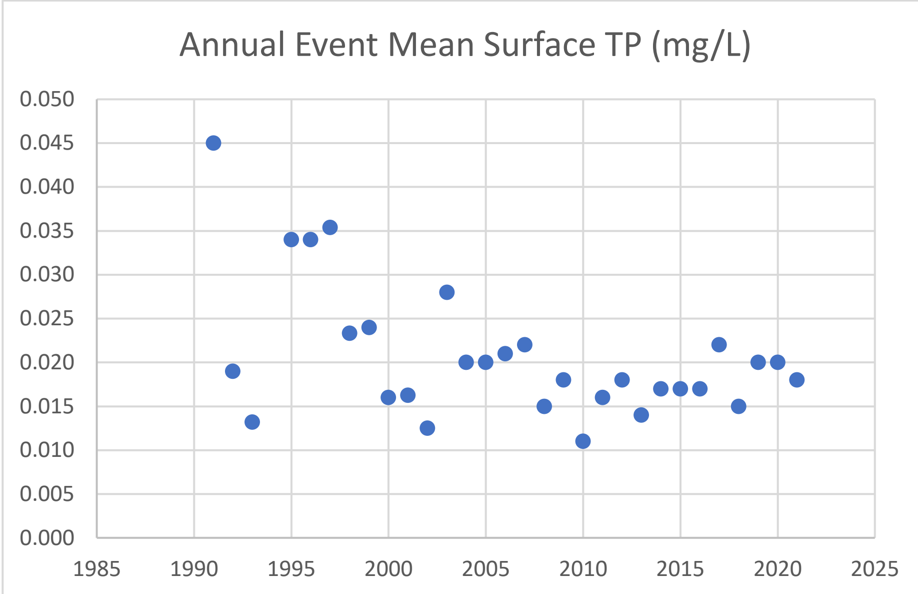

Figure 4-25: Annual Event Mean Surface TP Concentration

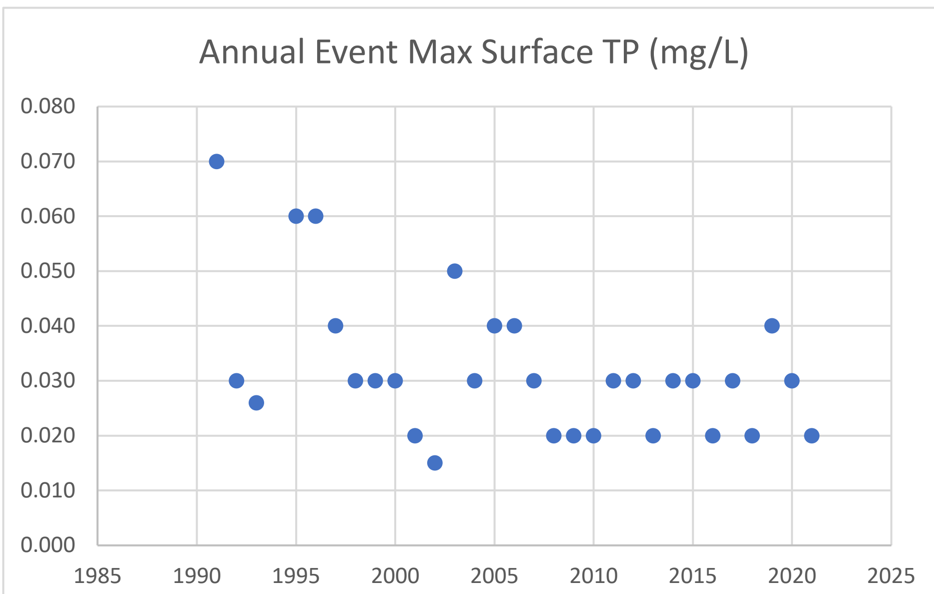

Figure 4-26: Annual Event Max Surface TP Concentration

4.3.7 ANOXIC AND OXIC LOAD COMPARISON

Last, the anoxic load as a percentage of total load was reviewed (Figure 4-27). This too has shown an upward trajectory over the course of monitoring, although there is a high degree of variability in the data. Again, there is an indication that since 2010 there has been an acceleration of the anoxic load relative to the total. As maximum is a better representation of load dynamics at critical points in the growing season, this data is shown in Figure 4-28. Again, the anoxic load has been increasing as a percentage of total load through time. All of the instances in which the anoxic load has broached 90% of the total load have occurred since 2015, including the last three years of monitoring. This indicates not only the dynamics of shifting loads with the anoxic load increasing and the oxic load decreasing, but also suggests that the internal load may be much more biologically important than it was in the past.

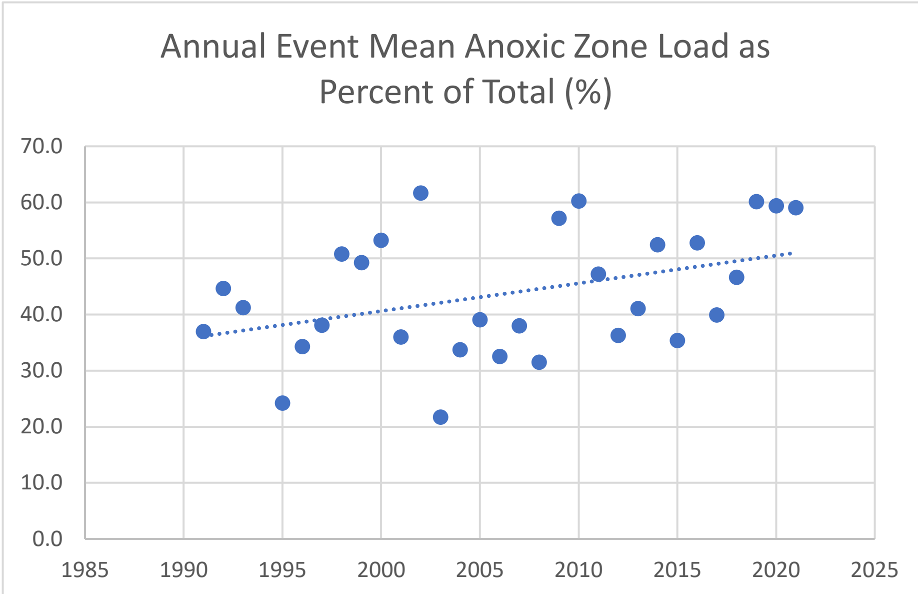

Figure 4-27: Annual Event Mean Anoxic Load as Percent of Total Load

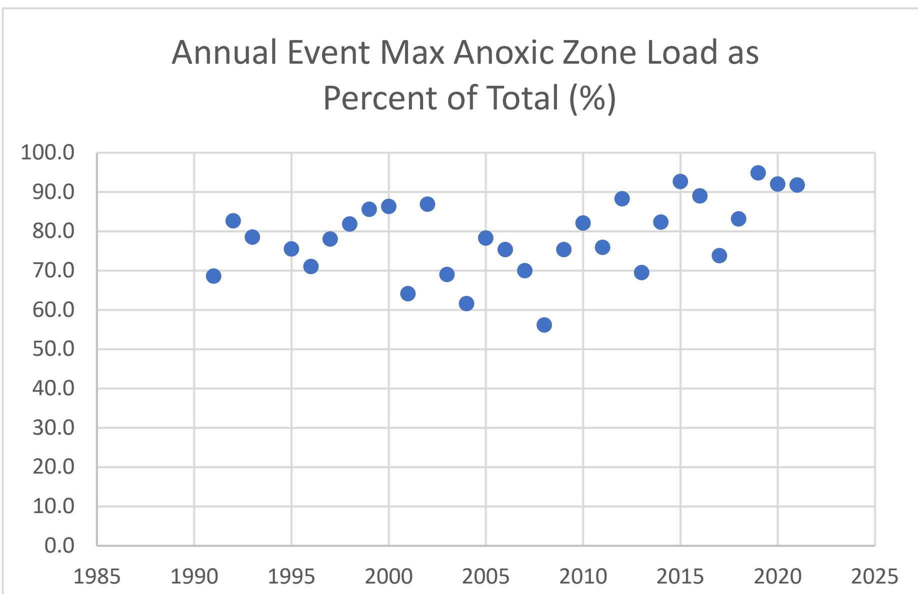

Figure 4-28: Annual Event Maximum Anoxic Load as Percent of Total Load

5.0 FACTORS AFFECTING THE INTERNAL LOAD

The results section above explored the seasonal and inter-annual dynamics of internal phosphorus loading. In summary, the anoxic load of Lake Hopatcong has shown a major increase in the past 30 years, with both primary components of the load, the anoxic volume and hypolimnetic TP concentrations, also increasing. The anoxic load expressed as a percentage of total load has grown. There are also indications that the load is contributing, at times, particularly in September, to increased phosphorus availability at the surface which may be driving ecological changes in the system. While the internal phosphorus load is the main focus of this project, other aspects of loading were investigated. The oxic load, which can be considered the upper water column including the epilimnion and trophic zone, has decreased substantially during the same time with a marked decrease in epilimnetic TP concentrations. This is almost certainly a function of the successful implementation of watershed phosphorus control strategies. As alluded to above, though, nutrient pulses to the upper water column in the late growing season seem to be growing, making more phosphorus available which in turns seems to be sustaining higher algal biomass.

The primary focus of this section will be on exploring some of the potential factors that could be contributing to these changes. There are a number of possibilities, but the primary focus here will be on climate and weather. Changes to external loading should certainly be considered a factor. In fact, it is already recognized here as a primary reason why surficial phosphorus concentrations and loads have decreased. That topic will not be explored in further detail here. Indeed, there is a vast corpus of work and documentation that details those efforts, as it has been the primary focus of ecological lake management practices over the last 40 years. Additionally, lower external loading may be expected to, eventually, reduce the internal loading; that is not occurring here. Lastly, the concept of eutrophication from a limnological perspective will be explored.

5.1 SURFACE TEMPERATURE

Surface temperature is an important consideration. It is a primary control on stratification patterns. It is also a good indicator of total system energy, and a controlling factor on biological and chemical reactions. Here, it is considered a regulator of system productivity and the production and consumption of oxygen. Lake water temperatures, particularly at the surface, are driven primarily by solar irradiance and the absorption of solar energy, as well as atmospheric temperatures. Flow, hydraulic retention, and color can also be important influences on temperatures.

Surface temperature was already discussed in section 4.2.5 and how it varies in the intra-annual time domain, but this section will focus on the inter-annual changes. Since 1991, average surface temperatures have trended up (Figure 5-1). The magnitude is subtle however. Using a linear regression, the growing season mean surface temperature has risen from 21.51°C in 1991 to 22.37°C. Overall, this is a relatively modest difference on the temperature scale, yet it reflects a tremendous amount of thermal energy in this large waterbody. Practically speaking, it also suggests earlier warming and later cooling. This in turn can be assumed to extend the period of stratification and anoxia. It is likely linked to a shallower anoxic boundary. How it effects the distribution of the internal phosphorus load is less clear. Higher temperatures may suggest accelerated metalimnetic erosion, in which the thermocline is driven deeper into the water column in the late growing season, however this could be offset by a time lag and later cooling. Additionally, warmer water temperatures would be linked to increased productivity, especially for the cyanobacteria, and an extension of the growing season. This increase in productivity could drive accelerating internal loading in successive growing seasons. Higher temperatures would also be expected to increase the rate of oxygen consumption in the hypolimnion. Whatever the impacts are, it is clear that surface water temperature has increased in the lake.

While the surface of the lake has warmed less than 1.0°C on average in the last 30 years, the maximum annual temperatures have shown a much larger jump. Again, using a regression analysis, the average maximum temperature in 1991 was approximately 24.68°C, but by 2021 had jumped by over 2.1°C to 26.85°C. While maximum temperatures represent a brief moment in time, this usually occurs at the peak of seasonal productivity and therefore may have an outsized influence on driving productivity and the thermal regime of the lake.

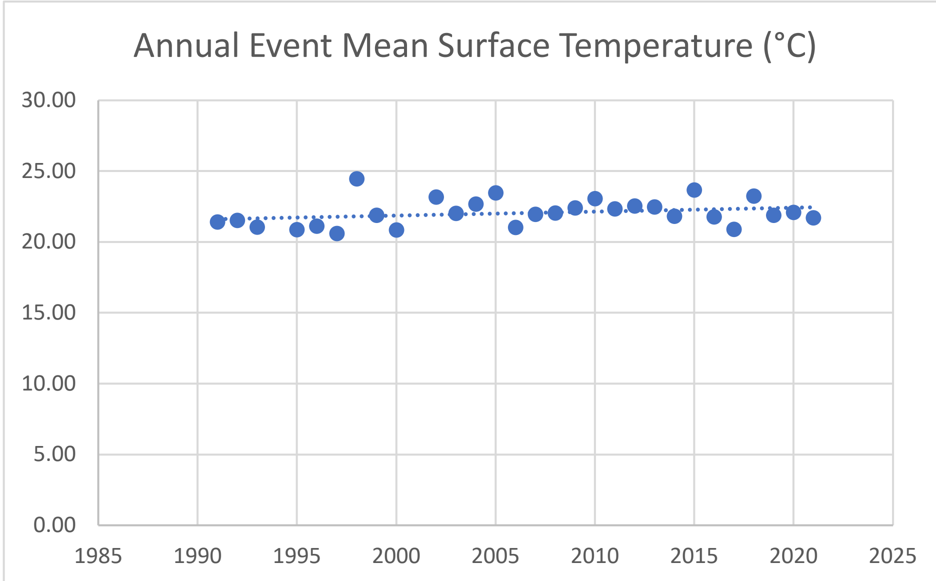

Figure 5-1: Annual Event Mean Surface Temperature

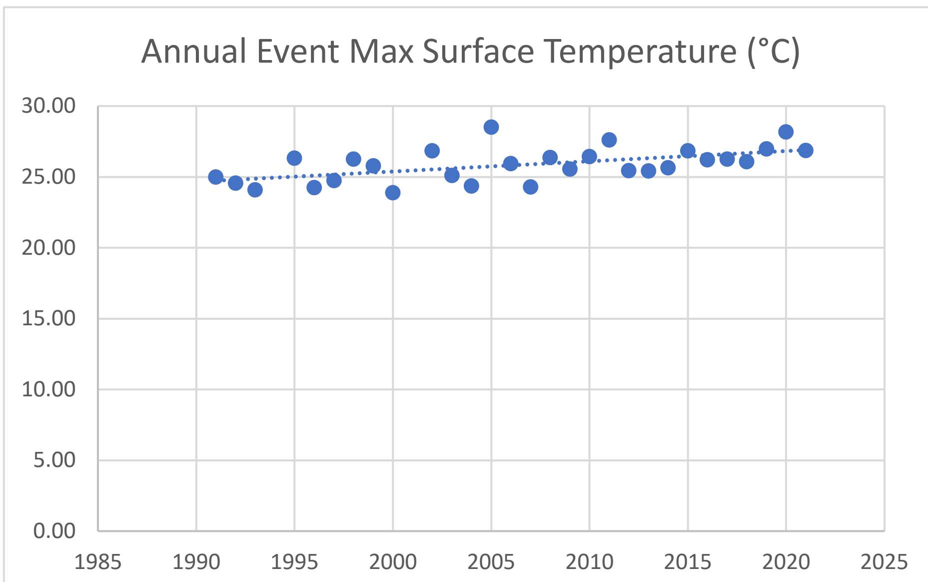

Figure 5-2: Annual Event Max Surface Temperature

5.2 CLIMATE DEPARTURE

Two aspects of climate were tracked for this dataset: precipitation and temperature. The effects of temperature have been discussed above. Precipitation is also a very important driver in lake ecology. Precipitation, via runoff, tributary flow, and groundwater flux, is the primary vector of phosphorus loading to lakes in the absence of point sources, including both particulate and dissolved forms. However, it impacts a number of other aspects of lake ecology like flow, flushing rate, and retention period. Heavy rains can disrupt algal blooms. Storm events are associated with other weather phenomena, and heavy winds may induce mixing or alter the structure of the thermocline while cloudy weather can reduce photosynthetic rates and growth. Depending on the time of year they may affect temperatures; summer storms are often associated with cooling, while spring and fall events are typically warm, particularly during hurricane season when the remnants of tropical depressions often pass through the area.

Here climatic changes have been analyzed using departures. The departures represent deflection from the 30-year climate norms. This helps place the data in a frame of reference, i.e., whether or not it is above average or below average and the magnitude of the difference. Climate records for the database utilized two nearby climate stations, Boonton Reservoir and Aeroflex Airport. These stations were the nearest with the most complete records. Boonton records were used from 1991 through 1998, and compared to the 1980 to 2010 climate normals for that station. The Aeroflex records were used from 1999 through the present, and were compared to the 1980 to 2010 climate normals for that station. The use of departures also helps to normalize the differences between the two weather stations.

The departures were examined on a monthly scale. Days with more than 5 days of missing data were excluded, and any years with missing months were similarly excluded from the analysis. For both precipitation and temperature, departures were examined on a monthly basis from April to September as an analog for the growing season and the monitoring season. For precipitation the monthly difference between the record and the climate norms were simply summed. For temperature, the average climate norms from the period were subtracted from the average of recorded temperatures.

Figure 5-3 shows the precipitation departures from the norms. It is clear that over time precipitation has been increasing. The distribution however is interesting. Prior to 2010, 50% of the years exceeded the climate norms, essentially a normal distribution. From 2010 onward, only 42% of the years exceeded the climate norms. This indicated that above average rainfall totals were less frequent in more recent data. Yet comparing the averages of those two time periods was quite different. From 1991 to 2009 the average departure was -1.45 inches, indicating that while the distribution of wet and dry years was equitable, the deflection was greater in dry years than wet years. From 2010 to the present that average was 3.09 inches. During this period, drought years were milder, but wet years were exceedingly wet. Prior to 2010 only one year showed a growing seasonal rainfall total more than 5.0 inches above normal, whereas this has happened five times since 2010 and three of those events have occurred since 2018. While the chance of a wet year may be down slightly, wet years tend to be very wet and droughts of lower intensity.

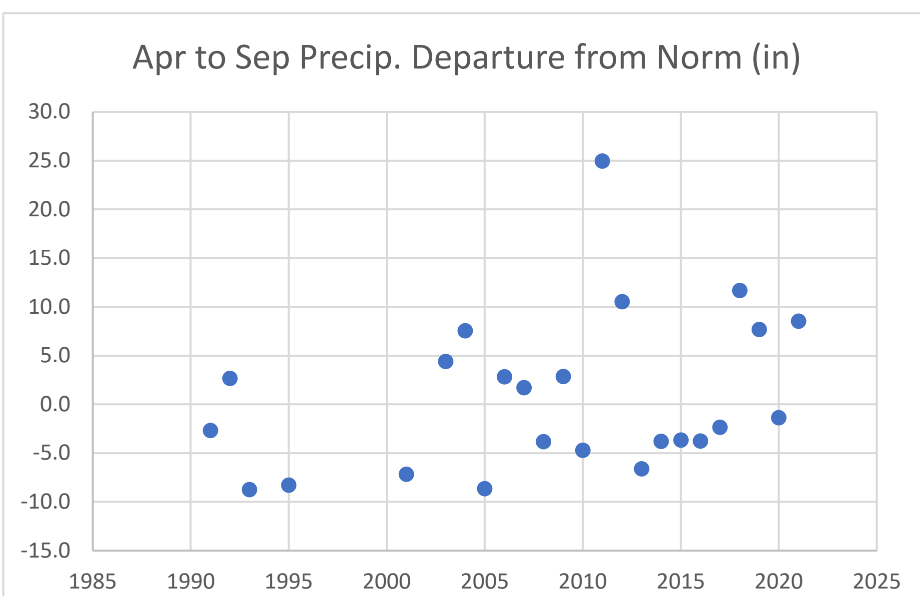

Figure 5-3: April to September Precipitation Departure from 30-year Climate Norms

The interpretation of the temperature data is more straightforward and temperatures have clearly been increasing relative to the climate normals. From 1991 through 2009 about 46% of the years were warmer than climate norms, with an average departure of -0.01°F; again, this describes an almost perfect distribution. From 2010 through 2021 92% of the years have been above the climate norms and the average departure during this time was 1.51°F. Clearly, the summer months are significantly warmer in the last decade relative to the two previous decades.

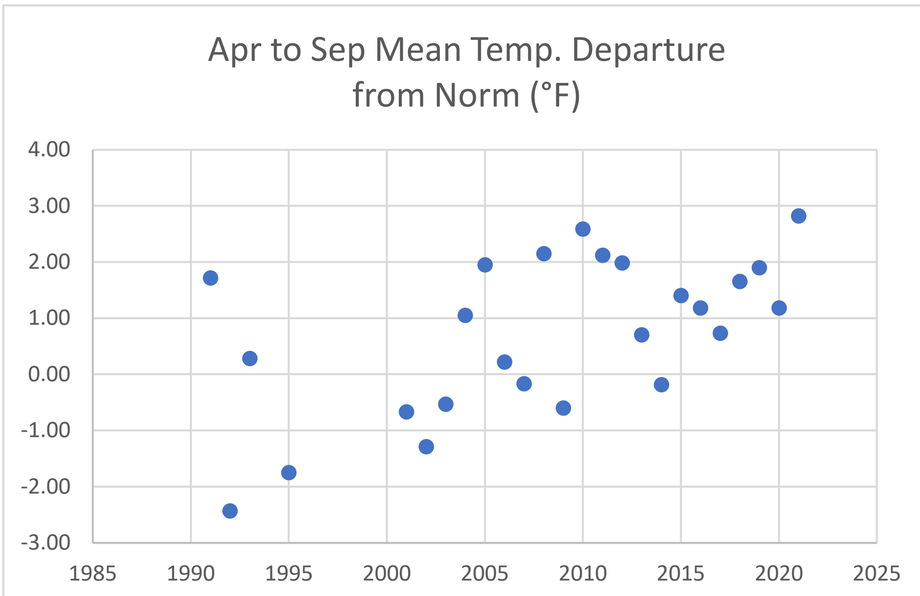

Figure 5-3: April to September Temperature Departure from 30-year Climate Norms

In addition to these two factors a departure index was calculated. The departure index is simply the sum of the precipitation departure and the temperature departure. No adjustments were made because over the 30 year dataset the average departures of both precipitation and temperature varied by about 14% relative to each other. This is a unitless metric, but helps describe the paired weather system during each of the monitoring years. Warm wet years will have high magnitude positive values, while cool dry years will have large negative values. Average years should have values near 0. The utility of the metrics fails somewhat in the case of warm dry years and cool wet years, but is still adds another way to gauge how climate has changed.

The data is plotted in Figure 5-4 below. The preponderance of warm wet growing seasons has grown dramatically since 2010. During this time, there were five years in which the index value exceeded 5.0, and were very wet and very warm. This includes three of the last four growing seasons. Prior to 2010, a departure index score over 5.0 was only seen once before, in 2004. Again, it is interesting to note the distribution. Even in the period from 2010 to 2021, the index value was negative for 58% of the seasons. In all but one of those seasons temperatures were higher than average, and those negative values reflect dry conditions. The post-2010 period can be characterized as very warm, but with variable rainfall, however wet seasons tend to be extremely wet. In reality, this has primarily been driven by late-growing season storm events, the remnants of hurricanes and other tropical systems passing through the area. This type of variability helps to explain the differences in observed growing season ecology at the lake.

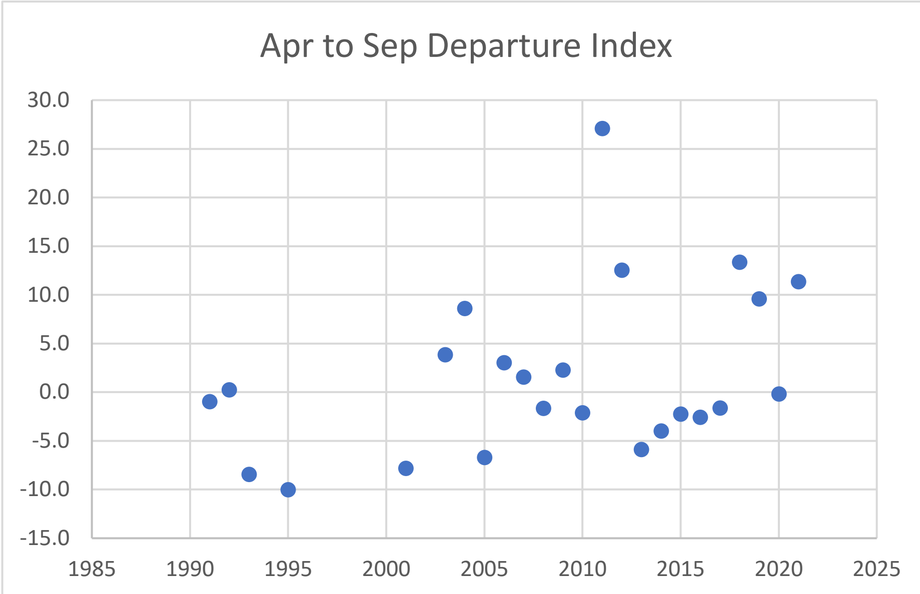

Figure 5-4: April to September Departure Index

5.3 CLIMATE DEPARTURE AND LOADING ANALYSIS

The climatic data was then used to analyze the various load metrics, for both anoxic and oxic loads. This was initially tried at various scales, including an April to June window, July to September, as well as the growing season April to September as reported above. The analytical strength was highest when using the growing season data, particularly for the anoxic load. April to June data generally yielded the weakest results, with one exception to be discussed later. June to September yielded clearer results and higher R^2 values. This suggests that the weather, the manifestation of climate on an immediate scale, experienced late in the growing season is more important to driving overall loads than April to June. However, since the full season data yielded the best results, it is also clear that the early growing season establishes the antecedent conditions that can mitigate or contribute to late growing season patterns.

5.3.1 MAXIMUM ANOXIC LOADS

The annual maximum anoxic load was compared to the departures for both precipitation and temperature as well as the departure index. For precipitation, there was a positive trend indicating that annual maximum anoxic load, the preferred statistic for the internal phosphorus load, increased with increasing departure, such that wetter years had higher internal loads (Figure 5-5). The correlation remains fairly low with an R^2 value of just 0.135. Whether this implies causation or simple correlation is unknown. It is possible that increasing precipitation contributes a greater load through some mechanism that is reflected in the hypolimnion. This would probably suggest settleable materials, like coarse particulates rather than a soluble material. It may be more appropriate however to consider the lower end of the spectrum. Very dry years had small internal maximum anoxic loads, and these were clustered much more tightly. This may suggest that more stable hydrology, a consequence of lower rainfall, helps to minimize internal loading. Conversely, higher rainfall imparts a much greater degree of variability, but a greater chance of higher internal loading.

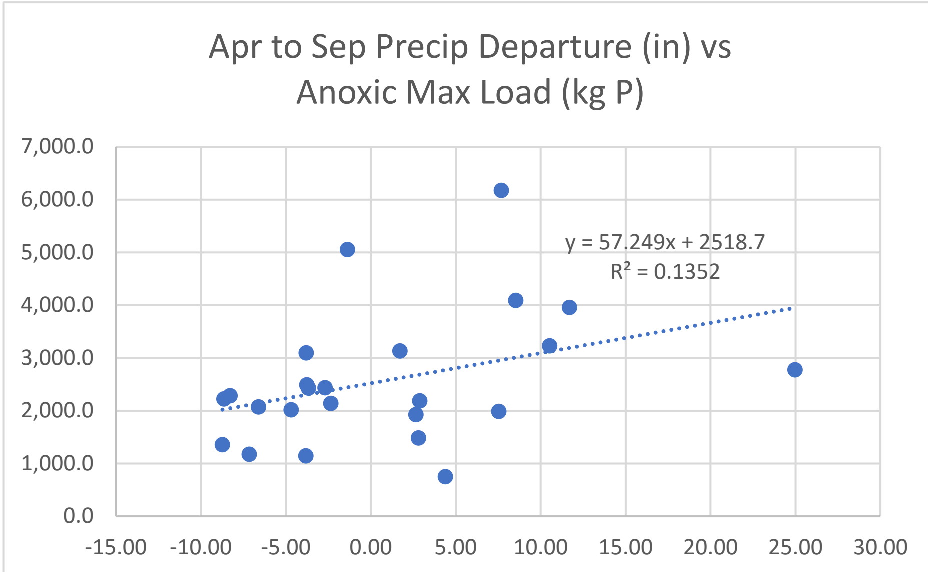

Figure 5-5: April to September Precipitation Departure vs Anoxic Maximum Load

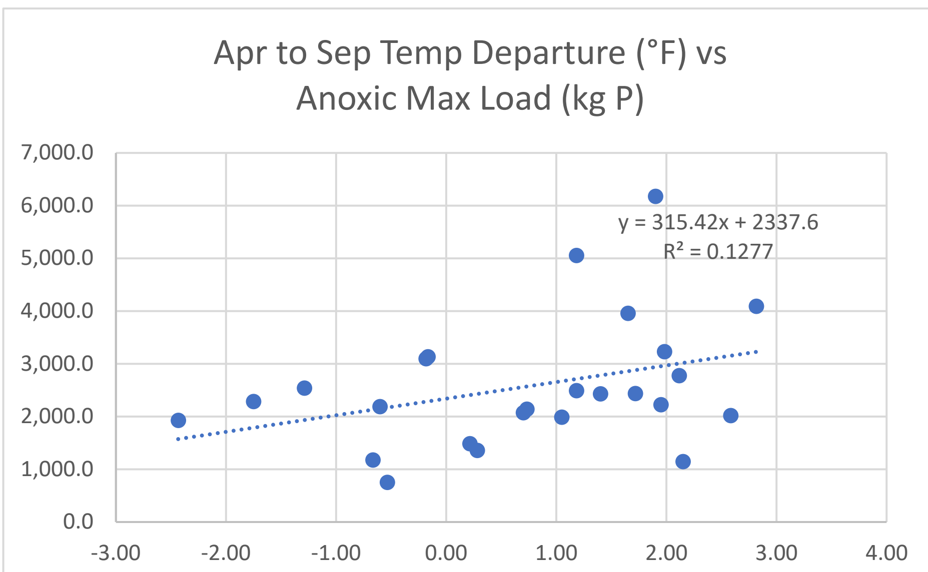

Figure 5-6: April to September Temperature Departure vs Anoxic Maximum Load

The temperature departure data is plotted above in Figure 5-6. The correlation (R^2 = 0.127) was actually somewhat weaker than it was for precipitation, yet there was a more even distribution of departures. In general, the anoxic load generally increased with greater temperature. Again, it is important to point out that whether this is causation or correlation is unknown. It may simply reflect that recent years, which were warmer, had much greater anoxic loads than previously. Some of the potential mechanisms have been discussed, but certainly it seems realistic that increased temperature extends the time of anoxia and the anoxic zone volume to increase the duration of sediment release and its accumulation in the hypolimnion.

Using the departure index yielded the best results, although R^2 is still relatively weak at 0.165. In general, the maximum anoxic load increased during warmer and wetter years (Figure 5-7); note the years are labeled in this figure. In particular, 2012, 2018, 2019, and 2021 all had extreme departure index values and all had very high loads. It is also interesting to note 2011; this is an extreme outlier the result of two consecutive months with over 15.0 inches of rain. Such extreme weather events are so disruptive to the normal hydrology and function of the system that their impact cannot be fully realized. While high departures, wet and warm conditions, are correlated with higher anoxic loading rates, the relationship is not always linear and there are a variety of factors that can affect this function.

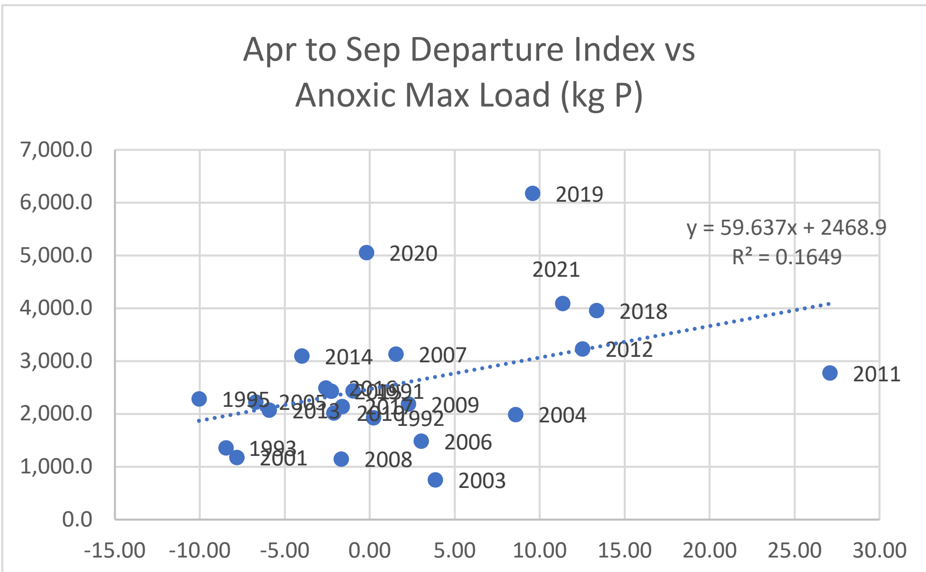

Figure 5-7: April to September Departure Index vs Anoxic Maximum Load

Above it was mentioned that the data was analyzed at different time scale. One of these produced an interesting result. The April to June departure index for maximum anoxic load correctly indicated that 2019 was a significant outlier in terms of both departure magnitude and loading, plotting to the right and above all the remaining dates. Generally, this figure showed a weak positive trend and poor correlation. This suggests that while the April to June time period under a typical range of departure index values is not especially predictive for anoxic load, that there is a potential that extremely warm and wet springs may be associated with extremely high loading anoxic loading. In the future, these conditions should be monitored closely.

5.3.2 MAXIMUM OXIC LOADS

The annual maximum oxic loads were analyzed in the same manner. Curiously, the R^2 values were much weaker for all of the oxic loading analyses than for the anoxic loads. The reverse was expected: the oxic zone, at the top of the water column, is seemingly much more intimately and immediately affected by changes in temperatures and hydrology. This, however, was not borne out it in the analysis.

Increasing precipitation departure was associated with a small and very weak decreasing trend (Figure 5-8). The variability was much higher during rainfall deficits. This may suggest, despite the potential for increased external loading during periods of higher rainfall, that increasing rainfall increases flushing in the system thus limiting the expression of the oxic load in the upper water column. It may also suggest a dilutionary effect of direct precipitation on the lake. The correlation value is likely impacted by the extreme outlier of nearly 25.0 inches (2011), however visual analysis does indicate that increasing rainfall generally reduces maximum oxic load at the lake.

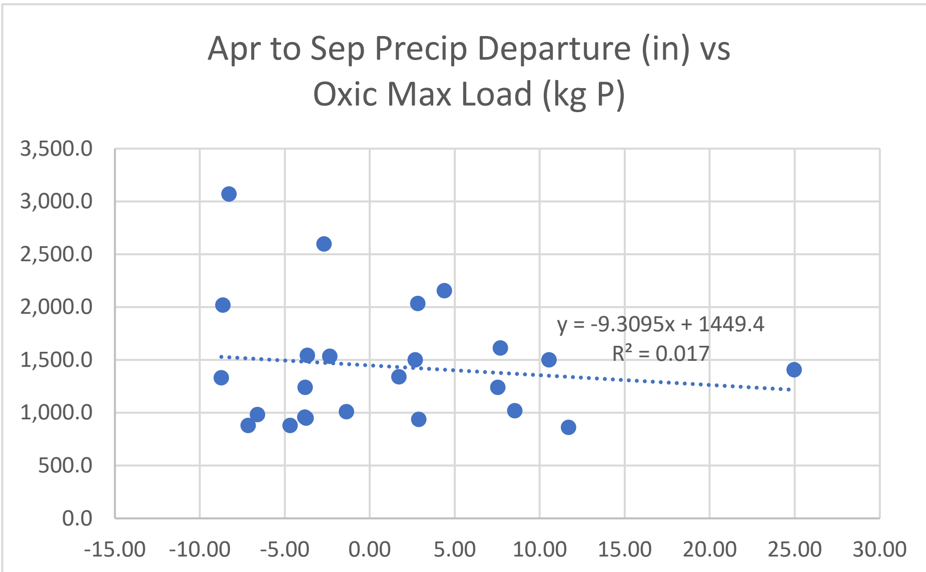

Figure 5-8: April to September Precipitation Departure vs Oxic Maximum Load

The analysis was again repeated for temperature. There is a very small decreasing trend in maximum oxic load with April to September average temperature departure (Figure 5-9). As before though, the correlation is exceedingly weak. There is no obvious mechanism by which higher temperatures should contribute to decreased expression of phosphorus in the epilimnion. This may simply reflect the fact that there has been decreasing external loading to the lake as temperatures increase, suggesting no real causation in this relationship.

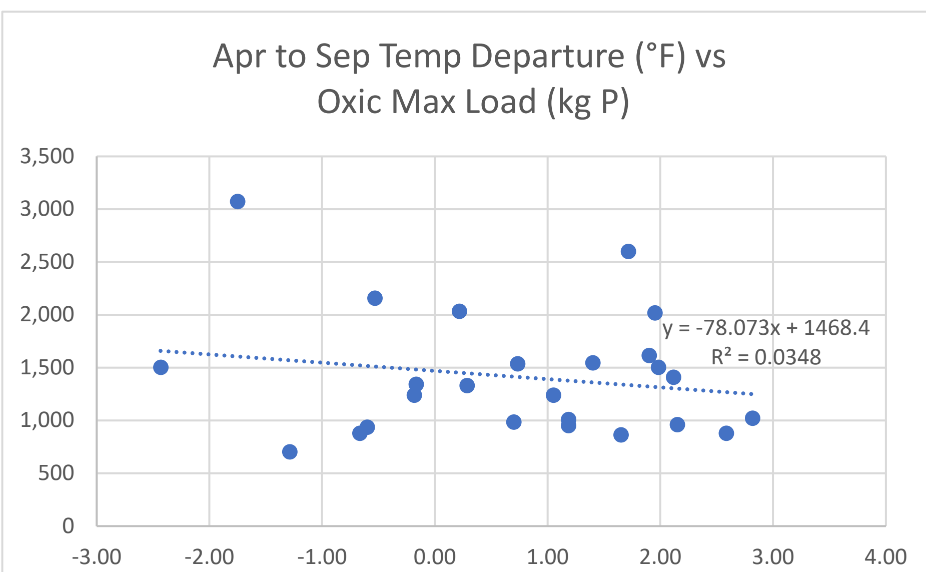

Figure 5-9: April to September Temperature Departure vs Oxic Maximum Load

The oxic load was compared to the departure index (Figure 5-10). Here too the R^2 value is very low, but there is a trend that indicates warm and wet years are associated with lower oxic loads than other meteorological conditions. In particular, there is a well-defined cluster of warm wet years (2004, 2012, 2018, 2019, and 2021) when oxic loads were low to moderately low. The most likely cause again seems to be the potential for enhanced flushing during such conditions, although lower external loading over time coupled with increasing rainfall departure must also be considered a major factor in this finding.

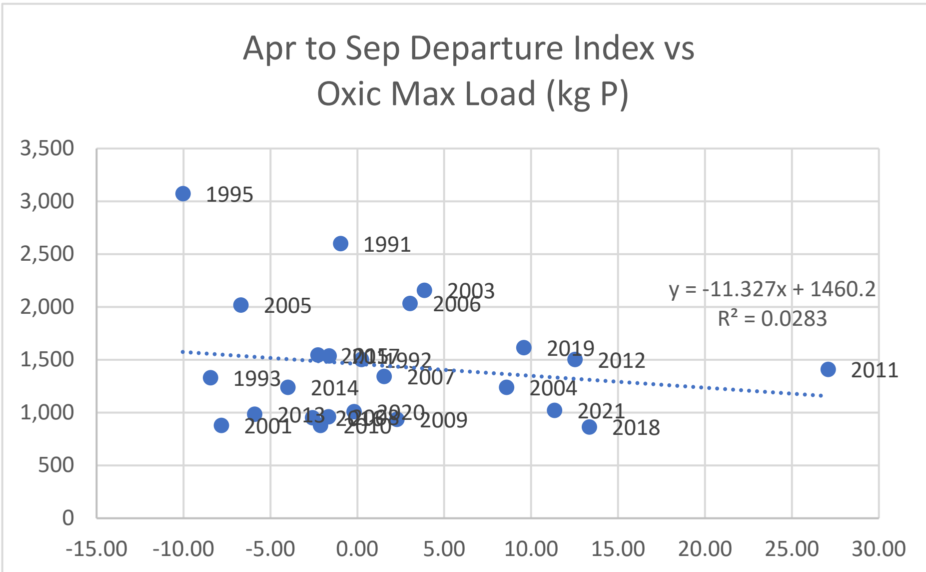

Figure 5-10: April to September Departure Index vs Oxic Maximum Load

5.4 EUTROPHICATION

As discussed above, climate has changed over time. In particular there has been a strong rise in growing season average temperature relative to the climate normals. The precipitation regime has also changed and while the frequency of wet years relative to dry years has been fairly stable, the wet years tend to be much wetter on average while the dry years are not as dry. The frequency of warm wet years has also increased substantially. Both increasing temperature and increasing precipitation during the growing season has shown substantial correlation with increasing anoxic loads. The same analyses performed for the oxic load actually show slight decreases with rising temperature and precipitation and poor correlation, suggesting that climate factors are less important for this load than the anoxic load, and generally may have minimal effect. Decreases in the oxic load with increasing rainfall are theorized to be related to increased flushing, while increasing anoxic load is probably associated with an expansion of the anoxic zone with warmer growing season temperatures.

The base effects of both loads must also be considered over the course of this project with anoxic loads strongly increasing and oxic loads decreasing. Decreasing oxic loading is a reflection of decreasing external loading to the lake, the result of nutrient management efforts, including via regulation like stormwater management rules, efforts to implement various best management practices (BMPs), and the construction of various sanitary sewers throughout the watershed. It may be expected that these reductions in external loading would be reflected in the internal load, yet they are not. This suggests that there are other factors at work.

One of these factors is the concept of eutrophication. Eutrophication is the increase in trophic state over time, usually associated with an increase in nutrient concentrations and/or loads and manifested in increased primary productivity of both algae and plants. The popular usage of eutrophication refers to cultural eutrophication which describes the process in which increasing nutrient concentrations and primary productivity, along with associated impacts, are the result of cultural processes. This includes a suite of factors which increase nutrient and solids loading to a lake system including watershed development, increasing stormwater volumes as the result of higher impervious coverage, the discharge of nutrient-enriched wastewater, agriculture and the use of fertilizers, and other related factors. This process also occurs naturally, even in undeveloped watersheds, and while the cause for increased loading varies one of the biggest differences is the rate of change. This also addresses the concept of lake aging or the accumulation of nutrients and solids over time with changes to observed nutrient loads, lake ecology and productivity, and even the morphometry/bathymetry of a lake. Undoubtedly, Lake Hopatcong has been subject to both natural and cultural eutrophication.

Originally, the lake consisted of two separate natural glacial lakes. These would have been subject to natural lake aging and the accretion of solids and nutrients over time, but likely at a very slow pace and the water quality likely would have been considered quite high. As early as the 1750s some dam construction was undertaken to support colonial industries. In the 1830s major dam construction was undertaken to link the basins to support regional commercial transportation through the Morris Canal. This resulted in the creation of a single large lake, deepened by approximately 12 feet, and similar in size and shape to the lake in its current form. Over the course of the nearly two centuries since, the watershed, and particularly the near shoreline areas, have seen tremendous development. As such, the lake has been accumulating solids and phosphorus loads with anthropogenic origins since that time. The rate of accumulation varies based on factors like hydrology and lake shape, such that only a portion of the phosphorus load is retained on annual basis, but over 200 years this is likely a very large load.

By the early 1980s, the management plan for the lake recognized the primary loader of phosphorus was septic effluent. This has been addressed in part by the construction of various sewer systems throughout the watershed, but this varies greatly by municipality, and many residents within the watershed continue to be served by individual on-lot septic systems. By directly managing that load, and other loads associated with cultural eutrophication, the external load to the lake has decreased. Yet even at a reduced rate, a portion of the phosphorus load continues to be retained each year adding to the existing loads accumulated over the course of centuries. It is not unexpected that the internal load therefore may continue to expand while other loads shrink and this may simply reflect the integrated eutrophication of the lake over time.

6.0 LAKE ECOLOGY AND THE INTERNAL LOAD

This section will examine some different aspects of lake ecology and how they have changed over time. This will focus on several metrics that are intimately linked to nutrient concentrations and nutrient loads including Secchi depth and chlorophyll. It also includes a trophic state analysis. Lastly, some of the impacts of changing anoxic zone volume and structure and lake water temperature will be reviewed as it relates to trout habitat.

6.1 SECCHI DEPTH

Secchi depth is a measure of lake clarity or surficial turbidity. Secchi clarity is generally governed by the density of suspended solids in the water column. In lake systems this is generally assumed to be algal particles, although other particles may be present, particularly at tributary mouths and along the shoreline as a result of discharge and runoff, resuspension of bed materials in the shallows, and dissolved color. In the field its measurement can also be affected by light conditions, sky cover, and surface disturbance, but it is usually held as a surrogate for algal density. Secchi depths of less than 1.0 m are usually perceived as turbid to most people and depths below 2.0 m are usually an indication of eutrophy.

Secchi depth was analyzed on both intra-annual and inter-annual domains since 1991. Secchi depth averages approximately 1.8 m during the earliest bin period through mid-May (Figure 6.1). Generally, this is a time of limited algal biomass due to cool water temperatures, although density can be affected by spring growth. On average, Secchi increases through early June. This is often noted as a "clear-water phase" and marks the change between spring and summer algal communities. There is often a noted increase in zooplankton density at or before this time which clear the water column through grazing. This is relatively short-lived and by mid-June Secchi starts to decrease as algal densities rise and cyanobacteria become more prominent. Secchi clarity then begins to slowly decrease throughout most of the remaining growing season. It first falls below 2.0 m again in mid-August and reaches a seasonal nadir in mid-September. This tracks with chlorophyll and algal density metrics. In late September and October Secchi shows some improvement.

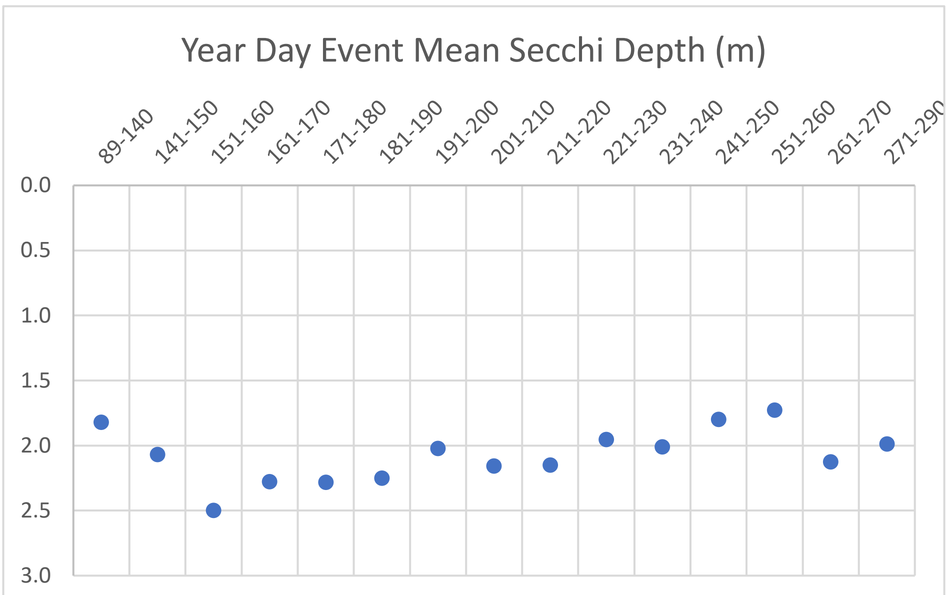

Figure 6-1: Year Day Event Mean Secchi Depth

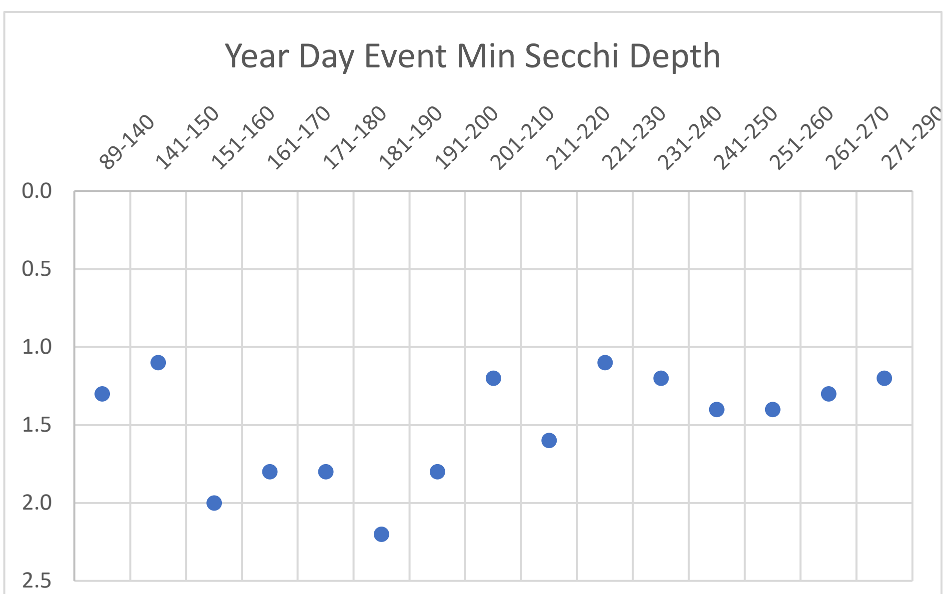

Figure 6-2: Year Day Event Minimum Secchi Depth

It is also interesting to view the Secchi depth minima throughout the year. After some initial phytoplankton growth in the spring, often chlorophytes (green algae) and diatoms, the clear water phase is very evident during the early summer (Figure 6-2). By late July though Secchi clarity can drop sharply and remains low into October. From late July to October the minimum Secchi depth averages 1.3 m and has dropped as low as 1.1 m. This corresponds to that time of the year when cyanobacteria blooms are most frequent. It is interesting to note that while Secchi on average is lowest in September, the event minima are recorded in mid- to late August. This suggests that the period just before and just after the thermocline begins to migrate down in the water column is extremely important in regulating algal growth and clarity. Average annual Secchi depth has been relatively steady over the course of monitoring, although since 2010 there has been a trend in which Secchi depth has been decreasing (Figure 6-3). Two of the three lowest years on average have been since 2019.

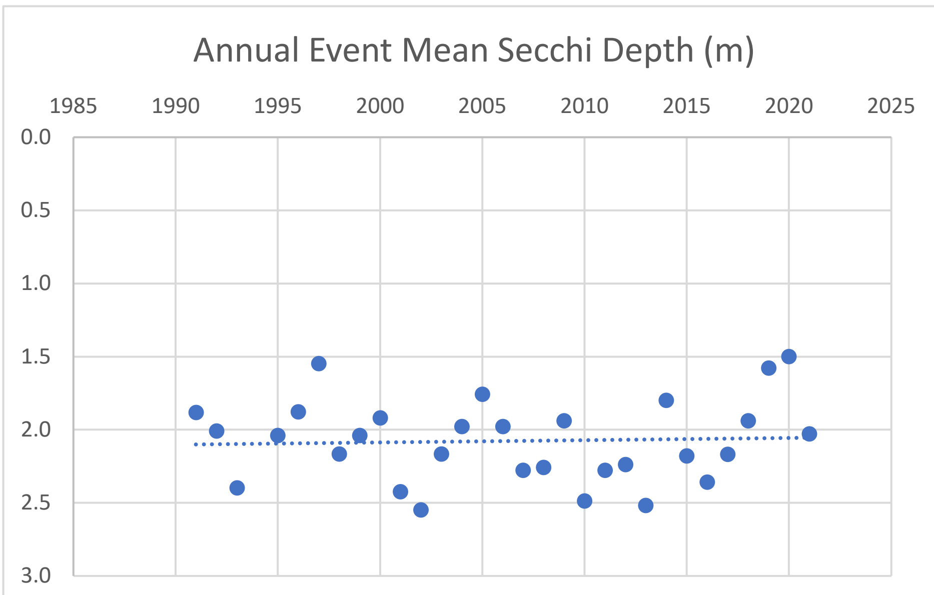

Figure 6-3: Annual Event Mean Secchi Depth

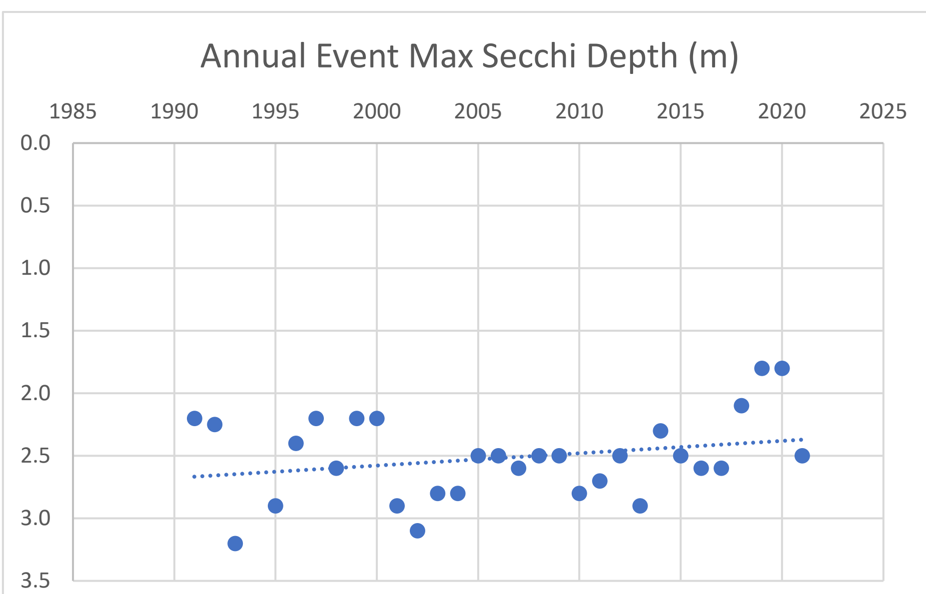

Figure 6-4: Annual Event Maximum Secchi Depth

Annual maximum Secchi depth was also plotted (Figure 6-4). This has decreased over time. The two poorest years on record were 2019 and 2020 in which the annual maximum Secchi depth was just 1.8 m. This is lower than the minimum Secchi depth of some years, including as recently as 2016. Since the annual maxima tend to be recorded around early June, this reduction in maximum value can be interpreted as indicating higher algal density earlier in the growing season than occurred previously.

The annual event minima were also explored. Interestingly, these have actually increased over time (Figure 6-5). The frequency of annual minima below 1.5 m seems unchanged and the lake continues to exhibit impacted clarity. The cause of this is not entirely clear. While epilimnetic phosphorus concentrations have fallen, chlorophyll, a measure of algal density, has increased on average. This would suggest that there factors other than algal density that influence Secchi depth at the lake. Decreasing loading of solids, achieved through watershed management measures, may be responsible for increasing minimum Secchi depths.

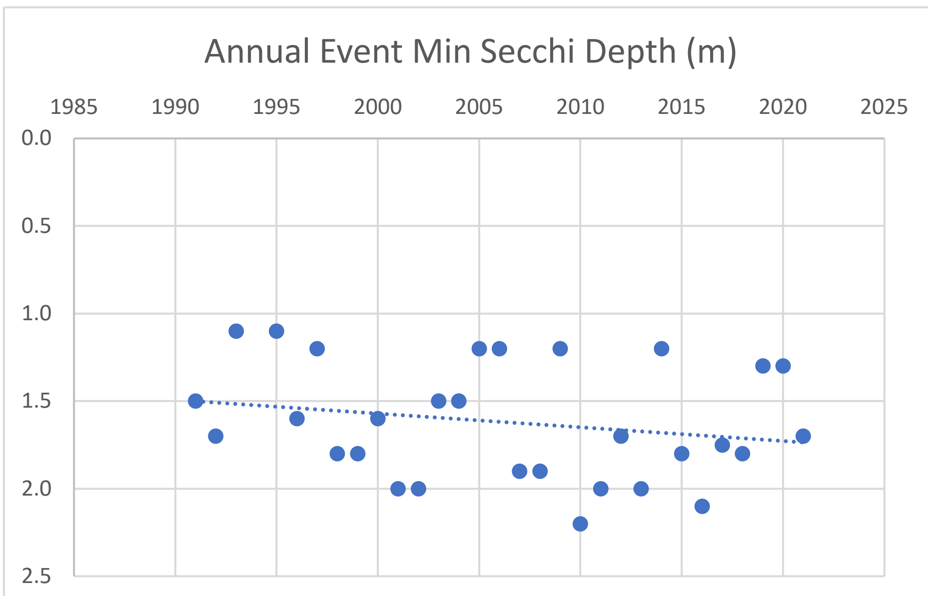

Figure 6-5: Annual Event Minimum Secchi Depth

6.2 CHLOROPHYLL

Chlorophyll is a photosynthetic pigment common to all plants and algae. In lake ecology it is used as a surrogate for algal biomass. Concentrations in excess of 10.0 µg/L are often associated with eutrophic conditions, while those in excess of 30.0 µg/L are typically only encountered during high density algal blooms.

The early growing season is marked by slightly elevated chlorophyll concentration, but these tend to fall though the clear-water phase, eventually bottoming out in early July at 5.75 µg/L (Figure 6-6). After this, it climbs steadily, with a maximum average value of 18.6 µg/L in early September. It drops thereafter, but stays above 10.0 µg/L through the remainder of September and into October. These are reflective of elevated, although not extreme values. They are however, averages, and the maximum values were similarly analyzed as shown in Figure 6-7. This paints a different picture of the lake, and shows that high density algal growth, here defined at concentrations of chlorophyll in excess of 20.0 µg/L, can be encountered throughout most of the growing season, including during the first bin from March into May. The highest events though are observed in mid- and late August, peaking at over 30.0 µg/L and consistent with high density bloom events. Into early September concentrations can be as high as 25.0 µg/L, and above 15.0 µg/L throughout the remainder of the year. Again, this points to the late growing season as the critical time. This suggests that these high density growth events are triggered right before the thermocline starts to shift in the water column. Two mechanisms may be responsible for the growth. First, while conditions remain relatively stable, the thermocline is near its shallowest and the water is still close to peak temperature, cyanobacteria may be regulating water column position to access high phosphorus concentrations just below the trophic zone and returning back to the surface. Additional growth is observed slightly later as metalimnetic erosion starts to mix that high phosphorus concentration water from the anoxic zone into the epilimnion, which sustains high algal density well into October.

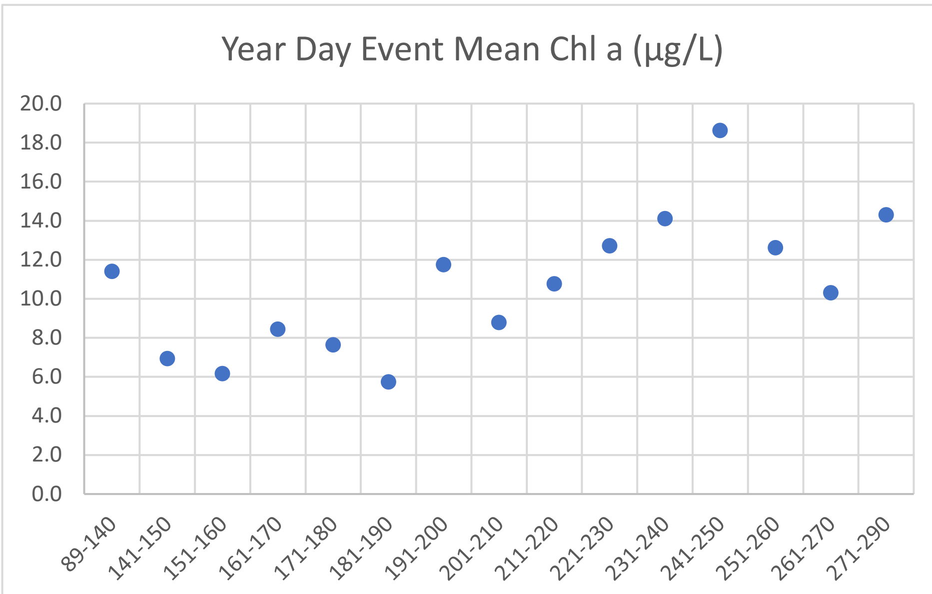

Figure 6-6: Year Day Event Mean Chlorophyll a Concentration

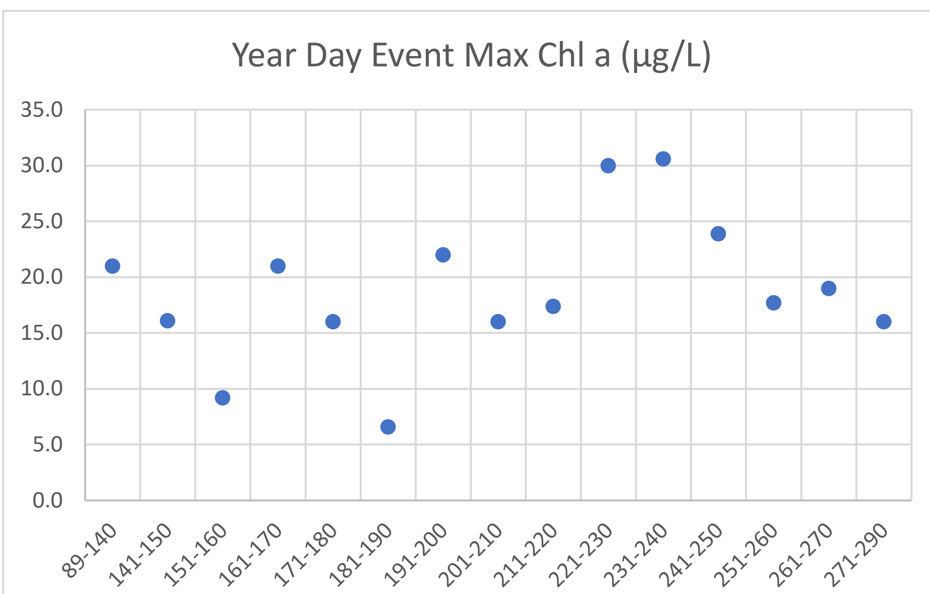

Figure 6-7: Year Day Event Max Chlorophyll a Concentration

The inter-annual variability of chlorophyll was explored. Chlorophyll, and therefore algal density, has risen steadily throughout the years (Figure 6-8). The first year with an average concentration over 10.0 µg/L was 1997, although this would become increasingly frequent. Prior to 2010, the average chlorophyll concentration was 9.74 µg/L, but since then it has increased 18.5% to 11.55 µg/L. During this same time surface TP concentrations fell by 26%. In general, chlorophyll would be expected to decrease as a direct cause of decreased nutrient concentrations. This suggests that the utilization of phosphorus, or other factors, has changed relative to algal production. This could represent a shift in community composition to organisms better suited to utilize phosphorus resources, a shift in the loading patterns, or a shift in the timing and availability of phosphorus. Likely, all three have happened and while surface concentrations are down, anoxic concentrations and loads are much higher, show stronger pulsing events in the late growing season, increasing dominance of cyanobacteria in the plankton, and a shallower and more accessible anoxic zone, all of which could explain this seeming contradiction. Annual event maxima have also increased over time (Figure 6-9).

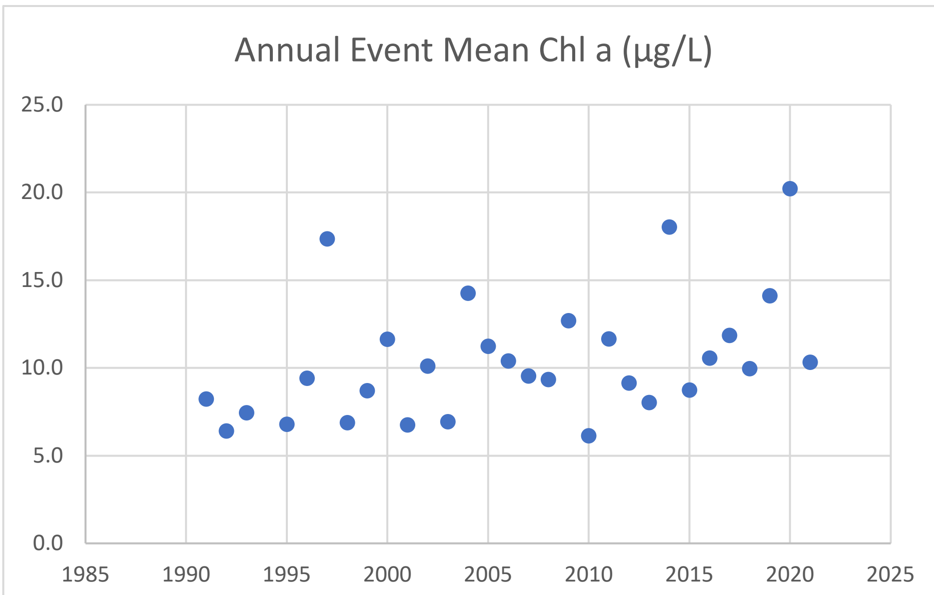

Figure 6-8: Annual Event Mean Chlorophyll a Concentration

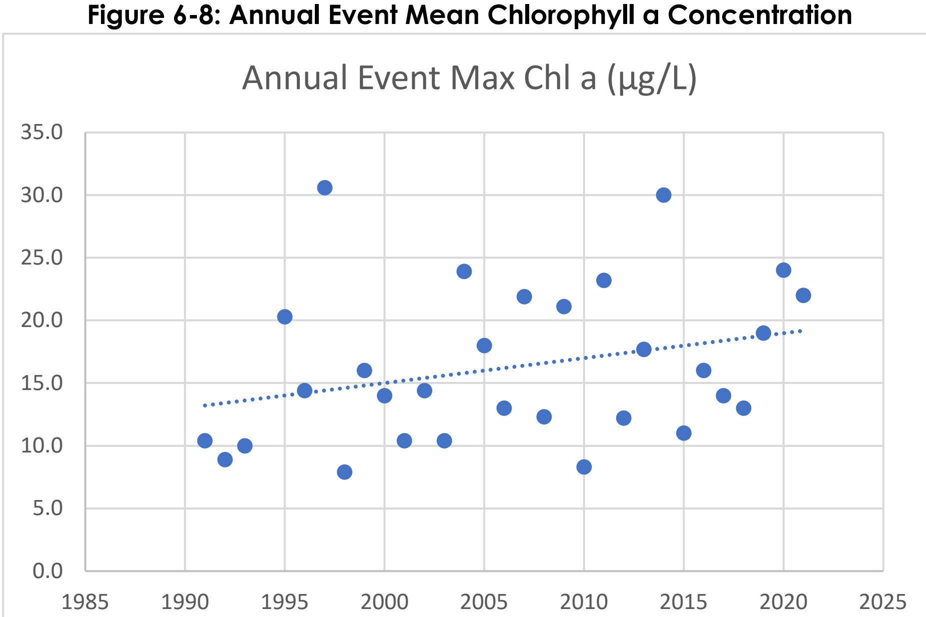

Figure 6-9: Annual Event Max Chlorophyll a Concentration

6.3 TROPHIC STATE AND RESIDUALS ANALYSIS

The trophic state concept is used to assess the relative productivity of system. Generally, lakes are categorized in one of three states:

- Oligotrophic – low productivity, usually defined by clear water and low biological carrying capacity
- Mesotrophic – moderate productivity with increasing coloration and biomass loads
- Eutrophic – high productivity with high algal or plant density, abundant fish stocks, and eutrophic impacts to water quality

The trophic state concept is often misunderstood as a categorical designation, rather than a continuum from a nutrient- poor state to a nutrient-rich state, eutrophy. Lakes can be classified on the basis of several water quality parameters, including total phosphorus, chlorophyll, and Secchi depth. The Carlson Trophic State Index (TSI) utilizes all three of these parameters. The basic assumptions of the model are that suspended particulate matter is the sole control of water clarity, that algae are the sole source of this particulate matter, and that algal density is governed by phosphorus concentration. If all assumptions are met, each parameter should yield the same value. Based on a scale, Carlson's TSI provides a method to easily classify lake productivity. The interpretation of the scale is provided below (Figure 6-10).

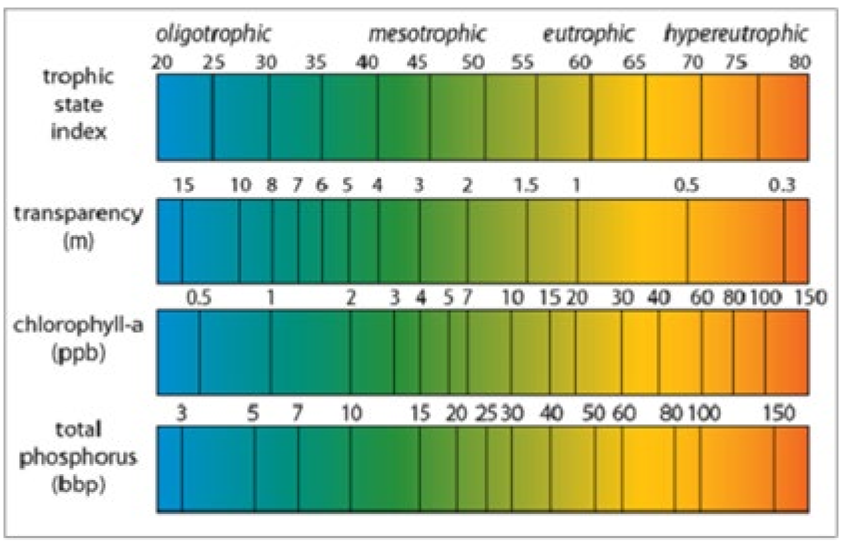

Figure 6-10: Carlson's TSI Scale

The TSI values can vary among the three input values and calculated indices. This occurs if all model assumptions are not met. A residuals analysis can therefore be performed to determine if any systemic differences between trophic variables may provide additional insight into lake productivity and trophic state. The residuals analysis examines the differences between the indices. Figure 6-11 below provides some of those interpretations. At its most basic however, the Y-axis represents phosphorus limitation. Phosphorus is considered the limiting nutrient in most cases throughout mid-Atlantic lakes and describes a state when biological demand is higher than phosphorus nutrient availability. Here it is assumed to occur when the chlorophyll index (TSI_{Chl}), indicating algal biomass, is higher than the phosphorus index (TSI_{TP}). Stated differently, algal biomass is higher than predicted based on TP concentration. The X-axis describes differences in particle sizes. This is calculated by subtracting the Secchi depth index (TSI_{SD}) from the TSI_{Chl}. Larger particles are said to predominate at positive values when the TSI_{SD} is below the value predicted by chlorophyll concentration. The TSI_{SD} value is inverse, and higher Secchi, meaning better clarity, is assigned smaller values. This essentially means that clarity is better than biomass would indicate. This happens if the cells are very large or if there are large colonies; cyanobacteria often exhibit large-colony growth forms.

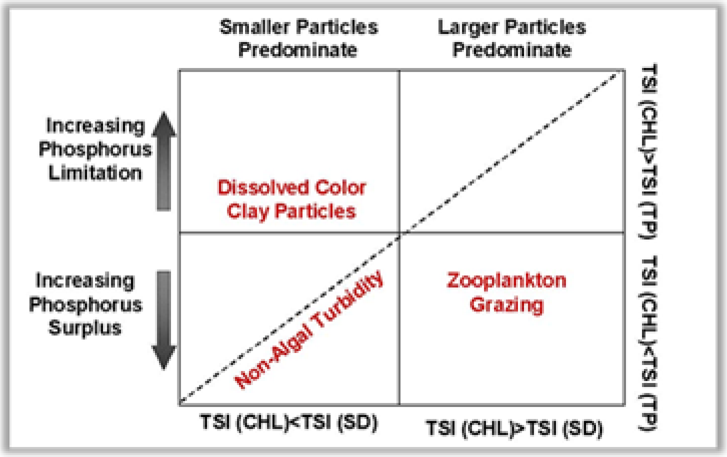

Figure 6-11: Carlson's TSI Residuals Interpretation

The three indices were plotted against the year day bins (Figure 6-12). In general, TSI scores between 50 and 55 usually are considered the boundary between mesotrophic and eutrophic conditions. In early spring, productivity is already moderately high, but declines somewhat for all three indices through early June and the clear-water phase. It remains relatively stable through early July and all three indices converge at a high mesotrophic value. By mid-July things start to shift. There is more divergence between the three values and TSI_{Chl} approaches a eutrophic value and then climbs through early September when it nears 60.0, a squarely eutrophic value. It decreases somewhat at later times, but remains relatively high and reflects higher algal biomass in the late growing season. TSI_{SD} is relatively stable overall. TSI_{TP} continues to decline through late August, but surges in September. While it remains towards mesotrophy, this increase at this time likely represents pulses from the internal load distributed into the upper water column and sustaining high algal biomass at this point.

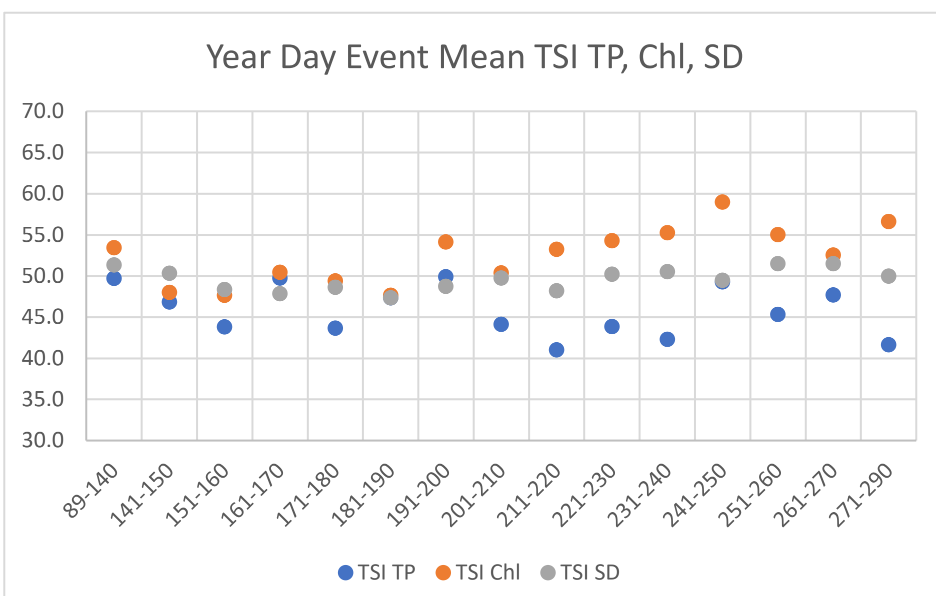

Figure 6-12: Year Day Event Mean TSI_{TP}, TSI_{Chl}, and TSI_{SD}

The residuals analysis was performed for this dataset. In the early part of the year there is little indication of phosphorus limitation (Y-axis value) or particle size distribution (Figure 6-13). Moving into mid-July phosphorus limitation starts to increase, and from early August through late September it is quite high. Phosphorus limitation is quite interesting during this period. It indicates higher algal biomass than would be anticipated given the measured phosphorus concentrations. This is likely a result of both the high internal load at this point and the composition of the phytoplankton, which is typically dominated by cyanobacteria. Cyanobacteria have a number of competitive advantages over most other phytoplankton. Capabilities vary among genera, but they are able to utilize both organic and inorganic forms of phosphorus, and dissolved and particulate species, while most other phytoplankton are constrained primarily to dissolved inorganic phosphorus species. Thus, even at lower concentrations, more types of phosphorus are available to cyanobacteria. The production of cyanotoxins is in part an adaptation to counter phosphorus limitation and the production and release of cyanotoxins can cause other algae to release phosphorus which is then assimilated by the cyanobacteria. Lastly, as mentioned at several junctures, some genera of cyanobacteria regulate their position in the water column through gas vacuoles. This allows them sink below the photic zone to the thermocline to uptake phosphorus found at a higher concentration due to internal loading. In Lake Hopatcong, this boundary is often quite shallow, and has been measured at less than 6.0 m from mid-July to mid-August which suggests that minimal movement is required to transition between the photic zone to the thermocline and back.

The X-axis values tend to be relatively minimal for much of the growing season. They do exhibit consistent positive values from late July through the end of the growing season, indicating that the particle sizes are quite large at this period. This tracks well with the predominance of colonial cyanobacteria at this time. It could also be an indicator that larger non-algal particles are present. This is certainly a possibility as organic detritus begins to mix through the water column later in the summer and around turnover.

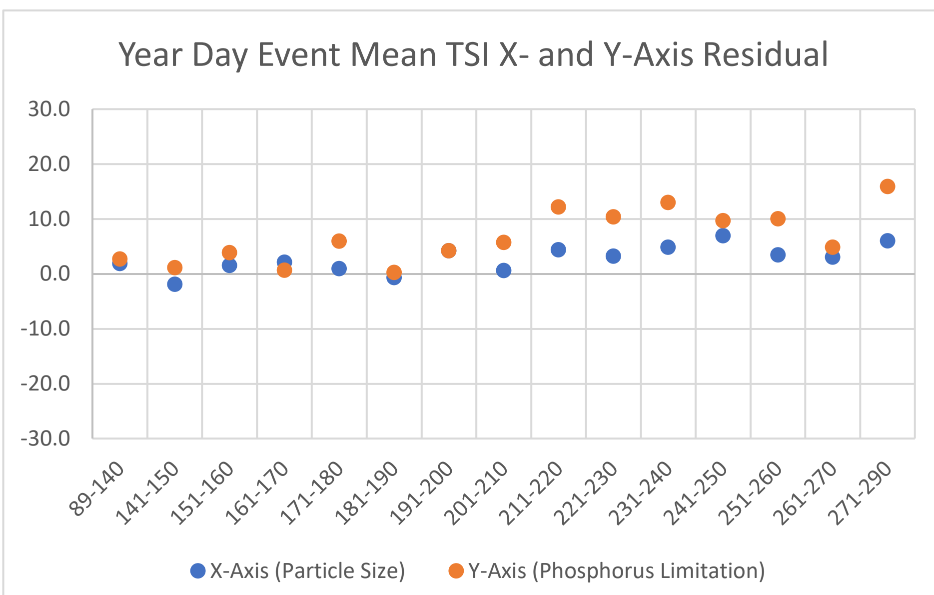

Figure 6-13: Year Day Event Mean TSI X-Axis and Y-Axis Residual

The TSI was used to evaluate trophic conditions over the course of the study. Two strong patterns emerged in the analysis of the data: TSI_{TP} has shown a marked decline over time, while TSI_{Chl} has increased (Figure 6-14). In practical terms, this shows that epilimnetic phosphorus concentrations have decreased, while chlorophyll and algal biomass has increased. As mentioned before, the decrease in phosphorus concentrations is largely related to efforts to control external loading, which has been successful. The increase in plankton biomass is likely related to increasing anoxic load, expanded anoxic zone volume, and increased duration of anoxia. It also is related to more efficient utilization of phosphorus resources, or the use of alternative phosphorus sources, for example through buoyancy regulation. This would tend to suggest that despite the success in managing the external load, reduced loading and measured nutrient concentrations at the surface are insufficient to limit increasing phytoplankton growth.

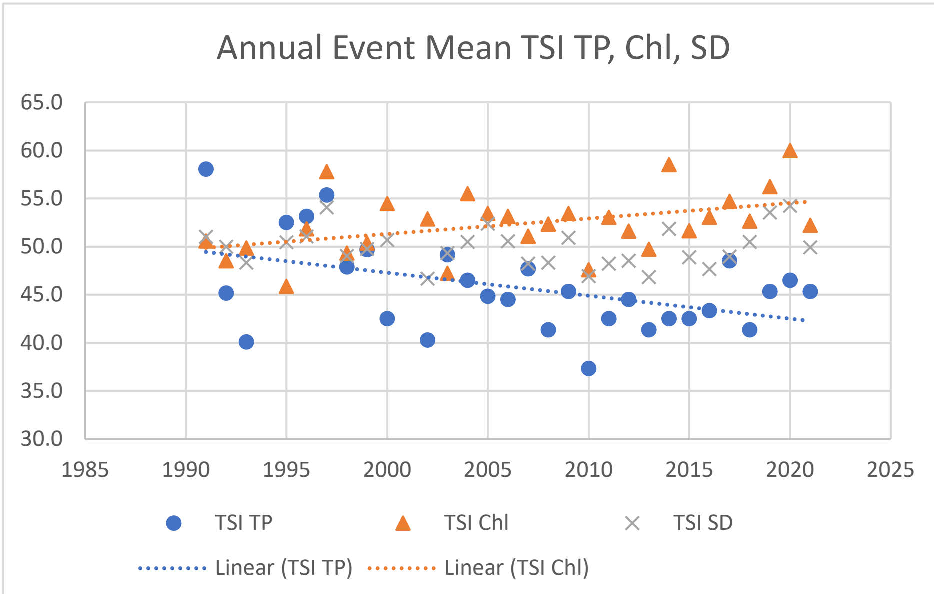

Figure 6-14: Annual Event Mean TSI_{TP}, TSI_{Chl}, and TSI_{SD}

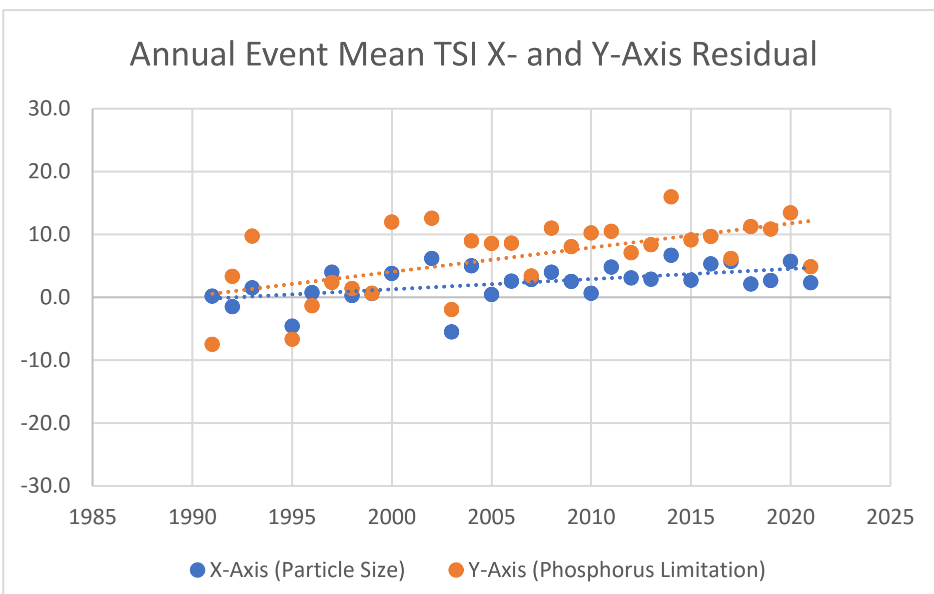

Figure 6-15: Annual Event Mean TSI X-Axis and Y-Axis Residual

The residuals analysis shows that phosphorus limitation, indicated by the Y-axis series, has been increasing over time, and recently has been quite high (Figure 6-15). This reflects both decreasing P concentrations and increasing biomass. Generally, it is a positive step in lake management to induce phosphorus limitation, yet additional efforts are clearly needed in the face of increasing algal biomass. A weaker trend is the increase in particle size. This is quite modest, but likely indicates the shift towards colonial cyanobacteria over the course of this dataset. Increasing dominance of cyanobacteria is related to the phosphorus availability dynamics in the system, as well as climatic changes.

6.4 TROUT HABITAT

Available trout habitat was also explored here. The link between trout habitat and internal loading is not direct, but many of the same factors influence both, chiefly water temperature and anoxia. In addition, the relationship between total productivity and anoxia should not be ignored. Indeed, internal loading is often considered as a type of positive feedback loop in which the release of phosphorus from the sediments increases primary productivity, which in turn can cause more intense anoxia as that algal load senesces and is decomposed.

Trout habitat is rather loosely defined. Optimal habitat occurs at temperatures less than 24°C at DO concentrations in excess of 5.0 mg/L, while carryover habitat is defined at temperatures between 24.0°C and 26.0°C at more than 5.0 mg/L. These classifications are closer to higher end bounds, and both lower temperatures and higher oxygen concentrations would be preferentially used. Higher temperatures and lower DO concentrations are not immediately lethal, but the effect increases with departure from the bound and is probably more critical as it relates to DO. Duration of exposure to those conditions is important, as well as the size/age-class of the fish; generally, larger and older fish have a higher tolerance for extreme conditions (Elliot, 1994).

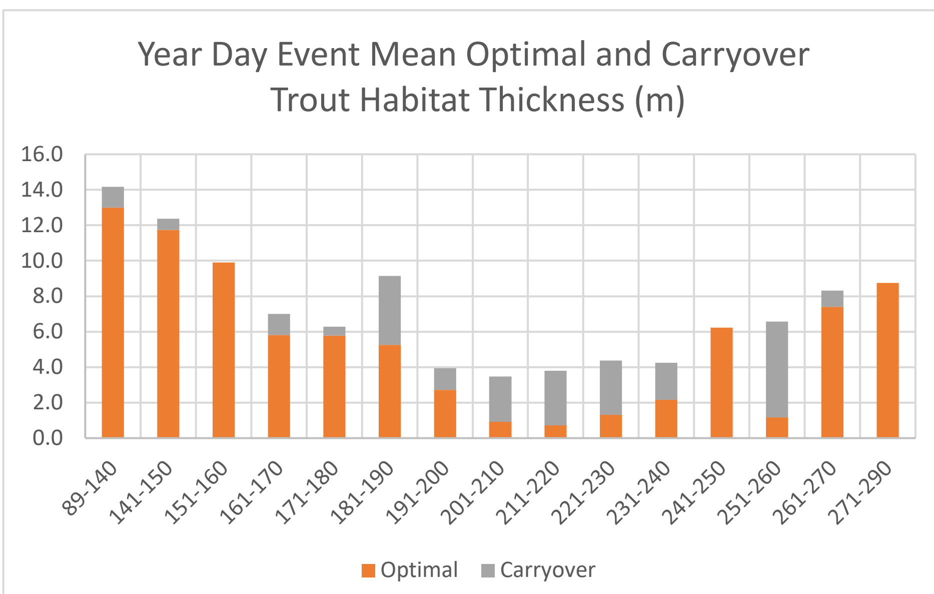

Figure 6-16: Year Day Event Mean Optimal and Carryover Trout Habitat Thickness

Trout habitat thickness varies widely through the course of a year (6-16). The upper bound of trout habitat, at or near the surface is chiefly governed by water temperature, while the lower bound, at depth, tends to be regulated by DO concentration. Habitat thickness is greatest during the early spring when temperatures are cool and DO concentrations high. It declines fairly steadily through early July as a consequence of both increasing temperature and expansion of the anoxic zone. Mid-July through late August represents the critical period when trout habitat hovers around 4.0 m on average, but even then, this is dominated by carryover habitat that is marginal to support trout. As water temperatures start to cool and the thermocline migrates down in the water column in early September trout habitat thickness begins a steady expansion through October.

Perhaps the more important analysis is the minimum thickness. In general, the spring and late growing seasons offer adequate trout habitat throughout the period of examination. More importantly though, over the course of 30 years, no trout habitat was identified at the limnetic deep water sampling station at times ranging from as early as mid-July through late August. This graph does not address duration of events, only that there was some event during each of those bins in which trout habitat was not identified mid-lake, but does not examine habitat in other portions of the lake. Similarly, it does not mean that lethal effects were immediately achieved, and does not address other refuge habitat in the lake like seeps and springs or tributaries. It does indicate though that the conditions can be very poor throughout the height of summer.

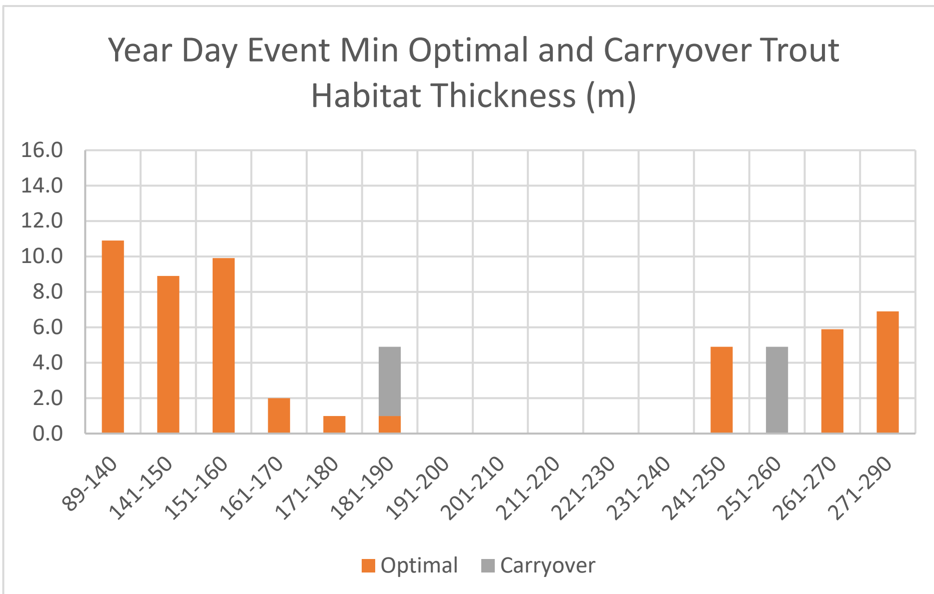

Figure 6-17: Year Day Event Minimum Optimal and Carryover Trout Habitat Thickness

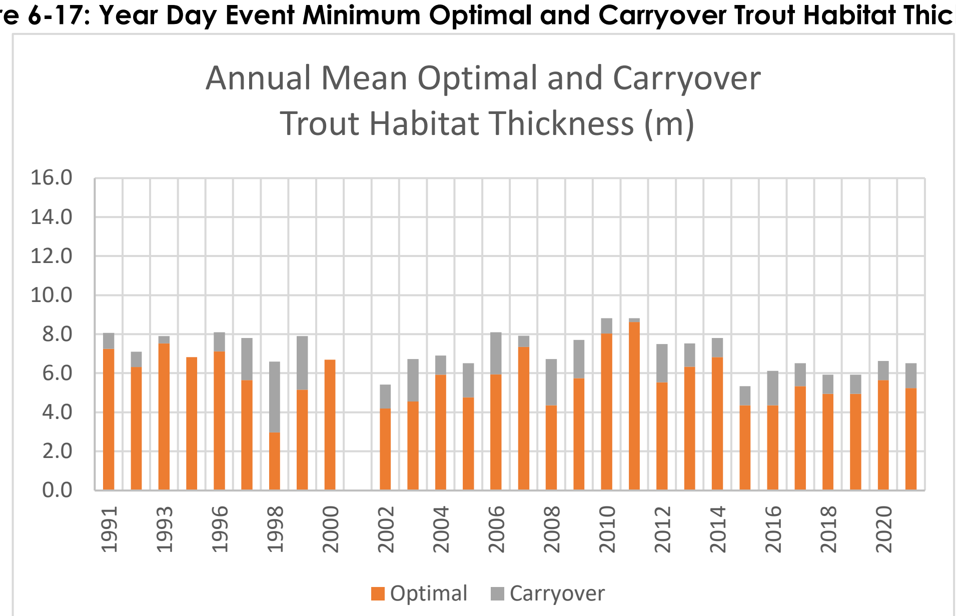

Figure 6-18: Annual Event Mean Optimal and Carryover Trout Habitat Thickness

Trout habitat was similarly analyzed since 1991 (Figure 6-18). For most of the period of record (note: 2001 data unavailable), available trout habitat has been fairly steady. The early 2000s marked a low point, but there was subsequent recovery until about 2010. Since then, it has shown a marked decrease of more than 1.0 m. This was due to a combination of increasing surface water temperature and a shallower anoxic boundary. As with the intra-annual analysis, the minimum trout habitat thickness is of greater consequence. For most of the record, viable trout habitat was recorded during each event during the annual monitoring period and did not fall below 4.0 m. The no habitat condition was recorded just twice prior to 2010 at the mid-lake station, in 1995 and 2002. In each instance this was a single event during the summer, suggesting that its maximum duration was on the order of several weeks. From 2010 through the present no trout habitat condition was identified during an event on five occasions spanning 2010, 2016, 2018, 2019, and 2020. In four other years, 2011, 2015, 2017, and 2021, it reached a minimum value of just 1.0 m. While overall trout habitat has decreased slightly, the frequency of critical events when trout habitat thickness is 1.0 m or less has greatly accelerated since 2010 and has occurred during 83% of those years. Again, a direct link to internal phosphorus loading is tangential, but there are certainly correlations to the conditions that have driven increased loading and impacts to available trout habitat.

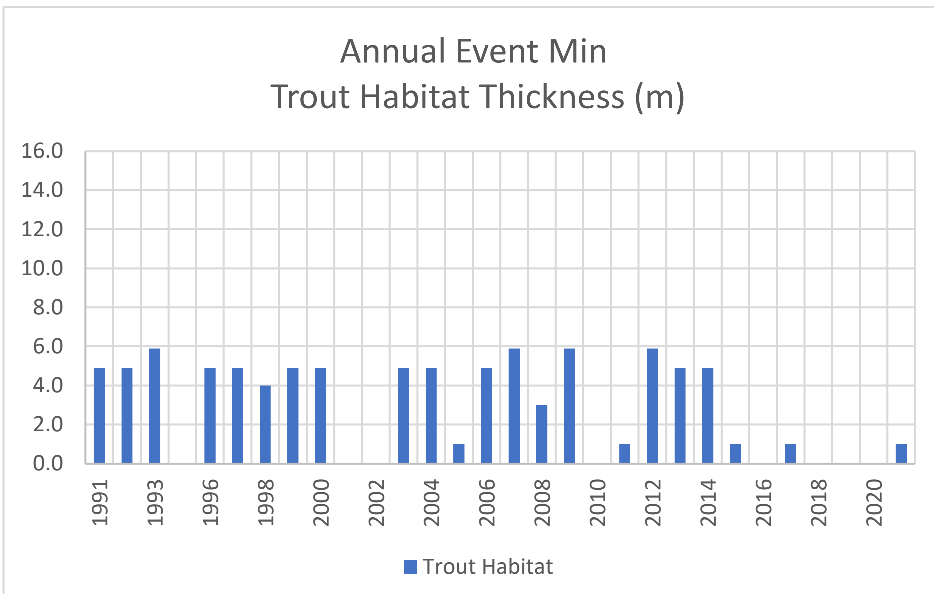

Figure 6-19: Annual Event Minimum Trout Habitat Thickness

7.0 MERITS OF MANAGING THE INTERNAL PHOSPHORUS LOAD

One of the most important questions for this study, as outlined in the introduction of this document is: does the internal load of Lake Hopatcong merit active management? In short, the answer to this appears to be yes, and those reasons have been extensively discussed above in sections 4.0, 5.0, and 6.0. This section will highlight some of the most important reasons discussing how the internal load or anoxic load has changed and how it more broadly has impacted the ecology of the lake.

7.1 CHANGES TO THE INTERNAL LOAD

While this report has covered a wide range of analyses focusing on many different aspects of lake ecology, the primary focus was on the anoxic load or internal load. No matter how it has been parsed, the anoxic load has increased significantly since the beginning of the dataset. The trend has not always been linear. During the first decade, the 1990s, the data was very variable, but by 2000 the loads had started to decline. Since 2010 it has grown rapidly. The three highest average anoxic phosphorus loads have been observed in the last three years. The best metric to express the annual internal load is the event maximum anoxic load. The pattern is much the same. The average of the maximum anoxic load over the last three years has been 5,100 kg; from 1991 to 1993 it was 1,900 kg. Since 2010, 50% of the years have exceeded an internal load of 3,000 kg and all were above 2,000 kg. Previous to that, just 22% of the years exceeded 3,000 kg and 44% were less than 2,000 kg.

The causes of this observed increase are varied. At a base level, the anoxic volume of the lake, including at both event mean and event maximum scales, has increased. The average increase likely suggests an increase in duration, while the maximum indicates that the anoxic zone has both a larger footprint and that the anoxic boundary is shallower in the water column. The other component of the load, TP concentration, has also increased using both mean and maximum metrics. The rise in concentration is likely a factor of longer duration of anoxia allowing for more accumulation, but it may also reflect increased productivity and continuing accretion and retention of phosphorus in the sediments in both recent and geologic time scales. Higher productivity, reflected in algal growth, is likely driven by increasing internal loading, but it in turn also increases the annual detrital loads to the lake bed increasing the consumption of available oxygen in the hypolimnion.

While all those components discussed above interact as a complex system and all affect the internal load, it is likely that systemic factors are the primary driver. A record of climatic changes was developed for both temperature and precipitation using growing season departures from the 30-year climate norms. Temperatures have climbed steadily over the course of this project. Since 2010, all but two years have shown an average growing season temperature departure of over 1.0°F, and four have exceeded 2.0°F. Growing season precipitation patterns have changed. While the relative frequency of wet versus dry growing season (based on departure from growing season norms) is little changed, since 2010 the wet years tend to be much wetter, often with extreme storm systems, while droughts are less severe.

These effects are observed in the data. Surface water temperatures have increased on average over time, and the annual maximum temperature on average has increased by nearly 2.1°C since 1991, a tremendous quantity when considered in terms of thermal energy in this large lake and an important control on thermal stratification and thermocline formation. The departures, including temperature, precipitation, and a combined departure index, all were positively correlated with increased internal loading. Extremely warm and wet springs were also correlated with extreme anoxic loading. While the seasonal climate will continue to vary from year to year, there is no indication that the increasingly warm and to a lesser extent wet conditions will not continue to progress in the short-term. If so, the data indicates that the internal load may continue to rise.

7.2 CHANGES TO LAKE ECOLOGY

There have been a number of attendant changes to the ecology of Lake Hopatcong correlated to the changes described above. Probably of greatest interest is algal density measured via chlorophyll. Average chlorophyll concentrations have increased sharply since 1991. There is both a long-term increase evident throughout the entire dataset, as well as an acceleration since 2010. This is primarily manifested in cyanobacteria, and there have been significant HAB events within the last few years, particularly in 2019.

There are several ways in which cyanobacteria or more generally primary productivity has increased as a result of increased internal loading. In general, the internal load or the anoxic load is generally sequestered in the hypolimnion, and theoretically unavailable to algae in the trophic zone in the upper water column. This only holds true while the lake is stably thermally stratified. As the thermocline starts to migrate this can start to liberate some of the from the water from the anoxic zone and mix it into the upper water column and trophic zone where most algal productivity and biomass is concentrated. This can occur through metalimnetic erosion as the thermocline gradually moves deeper in the water column in August and September or more suddenly in the case of fall turnover and the complete disruption of thermal stratification somewhat later in the year. In either case, this represents a pulse of nutrients that can sustain high levels of algal biomass. These pulses can be detected in several ways. First, a reduction in anoxic load indicates a pulse of nutrient. On average, this first starts to occur in early September, however significant anoxic load reductions have been observed as early as mid-August. Second, it is reflected in surficial TP concentration: from late July to late August surficial TP concentrations average 0.017 mg/L, but increase to 0.022 mg/L in September, an increase of nearly 30%. Over time these late season pulses have grown much stronger. From late August to early September mean chlorophyll concentration jumps from 14.1 µg/L to 18.6 µg/L. The maximum values tend to be somewhat earlier from mid- to late August at 30.0 µg/L. Another spike is observed in October when mixing is more complete. Together these data show how the anoxic load eventually is expressed at the surface through late season mixing processes and the breakdown of stratification.

Another way in which the anoxic load may be expressed at the surface is through cyanobacteria movements in the water column. This has been discussed several times above, but numerous cyanobacteria genera, including the familiar and problematic *Anabaena* and *Aphanizomenon*, can move through the water column by regulating cellular buoyancy through gas vacuoles. At times they will descend through the water column to the thermocline, assimilate the enriched phosphorus found at that depth, and buoy back towards the surface to photosynthesize. Over time, the anoxic boundary has moved toward the surface of the lake. On average, it was deepest around 2010, at roughly 11.5 m, but has moved up to less than 8.0 m in recent years, while the annual minimum is shallower than 6.0 m. This would indicate that accessing higher phosphorus concentrations at depth is less costly than it was previously, and the reward richer with higher anoxic zone TP concentrations. The anoxic zone boundary generally remains quite shallow through mid-August before descending and thus likely decreasing the rate of transport of phosphorus between the thermocline and epilimnion. It is interesting to note that peak chlorophyll values tend to occur right at this time and before more thorough metalimnetic erosion indicating that the start of bloom events may originate with this theory of algal phosphorus vectoring prior to mixing.

Chlorophyll has certainly increased over time, and there appears to be two viable pathways in which the internal loading is sustaining that rise, starting in mid-August: cyanobacteria position regulation and metalimnetic erosion.

7.3 INTERNAL LOAD, OXIC LOAD, AND TOTAL LOAD METRICS

The expansion of the anoxic load as well as correlated increases in productivity and algal biomass have been explored. It is important to contextualize the increase in the anoxic load to other components of the lake's load including, the oxic load and the total load.

The oxic load is the load of the upper water column including the epilimnion and portions of the metalimnion, essentially areas of the lake outside of the anoxic zone. This has been discussed in detail in sections 4.2.6 and 4.3.5 amongst others, but bears repeating. The oxic zone load has fallen considerably since 1991, primarily as a function of decreased TP concentration. The reduction in TP concentration is certainly not a function of reduced algal biomass, in fact this has grown. Instead, the combined efforts to manage the watershed or external load to the lake have been successful, and includes various efforts such as stormwater management and sewering portions of the watershed. This is a strong indicator of the efficacy of these efforts, and the management of this load should continue apace. Despite this, those concentrations reductions have not translated to decreased algal productivity.

Summing the oxic load and anoxic load produces the total load. Again, this represents the standing load in the lake as it was calculated for each event and not the integrated annual load. Indeed, the lake only retains a portion of its phosphorus at any given time or interval, usually expressed as a phosphorus retention coefficient. Significant portions of the load are continually lost through discharge at the dam to the Musconetcong River system. Since 1991, the anoxic load has increased as a percentage of the total load, reflecting not only the increase in the internal load, but the decreasing oxic load. Since 2019 the equivalent anoxic load has accounted for on average well over 55% of the total load; in the previous 27 years of monitoring data this value was broached just three times (11%). However, these values include the late and early growing when those anoxic loads are at near minimum. Comparisons of the annual maximum values may be more useful and generally target the late summer period. Since 2010, the maximum anoxic load has averaged 84.6% of the total load, while previously it was 74.7%. It is not a recent phenomenon therefore that the anoxic load often accounted for a bulk of the load late in the growing season, but it has increased as a percentage of total load, increased in absolute terms, and appears to be more available through changes in anoxic zone volume and mixing processes. At the same time, algal productivity has increased.

7.4 MANAGING THE INTERNAL LOAD

Together, the sections above lay out how the anoxic load of Lake Hopatcong is related to increasing productivity in the lake. The most compelling data however is the disconnect between reduced TP concentrations and oxic zone load and the increasing frequency, severity, and duration of algal blooms. The mechanisms for how the anoxic load have contributed to that increased productivity has been explored. The more salient point though is that a continued focus on just the external load and managing the trophic zone is likely insufficient to curb increasing productivity at the lake. This would be especially true if the increasing anoxic load is the result of larger systemic processes such as increasing temperature and lake aging and eutrophication. Unless temperature increases abate or reverse, the load is not likely to decrease naturally, nor are the phosphorus loads in sediments accumulated over the years going to decrease. Ultimately this will require a shift in the management paradigm with some focus on the internal load. As stated above, and stressed here again, it should not be made at the expense of the continuing to manage the watershed load. Indeed, reverses to those load reductions will only compound the problem by increasing phosphorus availability at the surface and adding even more load to the bed that could be liberated.

8.0 POTENTIAL INTERNAL LOAD MANAGEMENT OPTIONS

Section 7.0 discusses some of the reasons for managing internal loads in Lake Hopatcong. As mentioned, in some senses it is difficult to separate management of the anoxic load, internal load, and external load. Throughout this document the focus has been on the anoxic load. The anoxic load accounts for total loading in the anoxic zone, and there is a distinction between the anoxic load and the internal phosphorus load. The internal load refers to the release of SRP from the sediments under anoxic conditions. It is an important part of the anoxic load, likely the vast majority, especially as the anoxic zone load grows rapidly during the summer. Yet an exact determination of what fraction of anoxic load consists of the internal load cannot be made without additional laboratory analytical data which was not collected throughout most of the study. In addition to the internal phosphorus leachate, the anoxic load also consists of detrital rain, mostly senesced algal cells that contain phosphorus, settleable particles from watershed erosion, phosphorus precipitates formed under oxygenated conditions, as well as dissolved phosphorus loads from the watershed. However, in using the equivalent anoxic load metric a good estimate of the fraction of the anoxic load that is not the internal load can be made by looking at the average in the early part of the growing season prior to the development of a strong anoxic zone. Through late June, this averages just 214.0 kg of phosphorus. When comparing it to the highest event mean in early September it would account for just 8.1% of that load. If compared to a recent annual maximum value it is just 3.5% of that load. In either case, therefore, the internal load likely represents well over 90% of the anoxic load, and probably closer to 95% once the internal load is well developed in the late summer.

While the focus will be on managing the internal load, it is still important to maintain external load management to reduce to the extent practicable that smaller fraction of the anoxic load that is not released from the sediments. There are of course obvious benefits to the management of the trophic zone where most of the external load first enters the lake and where primary productivity occurs.

There are two primary methods available to lake managers to manage internal loads: aeration/destratification and nutrient inactivants. In general, aeration and destratification systems rely on the use of technologies that increase available DO concentrations in an effort to maintain sufficient oxidation potential to limit the release of phosphorus from the lake sediments. Nutrient inactivants on the other hand are chemical additives that directly bond to phosphorus and generally are unaffected by anoxia and thus maintain the chemical bonds to limit the release of phosphorus. Like aeration, there are a number of products and varying ways in which they can be deployed. The following sections of the document will explore each of these technologies, and discuss the benefits and drawbacks, installation, operation, and maintenance, and potential applicability to Lake Hopatcong.

8.1 AERATION AND DESTRATIFICATION

Aeration and destratification systems utilize a wide variety of technologies. Several of the major classifications will be described and discussed, as well as system benefits and disadvantages.

8.1.1 DESTRATIFICATION

Destratification (complete water column mixing) aeration systems use compressed air to vertically circulate the entire water column thereby preventing thermal stratification from occurring or from persisting. This results in a water column characterized by relatively uniform surface to bottom water temperatures and densities. As a result, the entire water column can easily circulate from surface to bottom. Lake water reoxygenation occurs due to the constant and consistent vertical mixing of the water column and the exposure of the water to the atmosphere. Although compressed air facilitates water column mixing, it is the exposure of the water to the atmosphere rather any direct oxygen transfer associated with the compressed air that is responsible for the vast majority of reoxygenation; however, advances in diffuser technology and the production of increasingly smaller air bubbles may directly transfer some limited amount of oxygen to the column. The primary goal therefore is to limit the formation of a stable hypolimnion and circulate oxygenated water to the bed in order to prevent anoxia and the leaching of phosphorus from the sediments. A secondary benefit of these types of systems is that they disrupt the formation of stable mid-column habitat which is often crucial to the growth of various nuisance algae including the common cyanobacteria *Oscillatoria.*

Destratification systems create a vertical convection current that results in the bottom waters being circulated to the surface of the lake, replicating the natural mixing of a lake during periods of turnover or when the water column is of uniform water temperature and density. This is accomplished by the strategic placement of air diffusers throughout the lake, but especially within the lake's deeper reaches. The air compressors and the negatively buoyant air lines account for majority of the cost associated with destratification aeration systems. The diffusers are relatively inexpensive (approximately $600 per diffuser unit, although this can vary widely depending on design, especially when incorporating multi-head diffusers). The compressor or more likely multiple compressors must be housed in suitably-sized compressor buildings. To mitigate the noise and heat resulting from the operation of the compressors, the compressor buildings must be both heat and sound insulated and vented. Destratification systems are operated continuously typically starting in early- to late-spring (before thermal stratification occurs) and throughout the entire summer until the early fall and the natural breakdown of stratification due to lake cooling. Operational costs are a function of the size of the required compressor. The compressors for a destratification system servicing a relatively large lake typically require 3-Phase, 220-volt or 440-volt power sources. The annual maintenance for these systems primarily involves inspection and servicing of the compressor. Most modern lines are very robust, often offered with lifetime warranties, as are the diffusers, which utilize self-scrubbing technologies and it is not unreasonable to expect at least 20 years of service from those components, although periodic inspection, probably most efficiently conducted by a dive crew, is recommended.

While these systems are of great utility in many settings, there are some drawbacks to their use in Lake Hopatcong for management, operational, and practical reasons. In its current state, Lake Hopatcong is a dimictic lake. This means that it undergoes two mixing cycles per year, once in the spring as water temperatures are increasing and again in the fall as temperatures are cooling. Otherwise, the lake is stratified throughout the winter and again throughout the summer. In winter, depending on temperature, the lake likely exhibits inverse stratification in which cooler waters overlie warmer water; the maximum density of freshwater is achieved at 4°C (39°F) and will therefore be found at depth when the lake is frozen. Typical stratification occurs during the summer months in which the epilimnion is warm and the hypolimnion cooler. Destratification therefore would eliminate this period of summer stratification, a great change to the current regime. In its current state, the internal load is sequestered through most of the year, although cyanobacteria migration in the water column counters this benefit, as does the late season release of phosphorus (on average starting in early September) to the trophic zone which is linked to algal bloom formation. More importantly though it would significantly impact coolwater fishery habitat structure. While increased DO concentrations, especially at depth would be of benefit, the impact to the thermal structure could be detrimental. Any temperature refuge available, pockets of oxygenated water, would be eliminated and the water column would be of nearly uniform temperature, expected to align closely with the current surface temperature regime of the lake. Certainly, in its current form trout habitat shrinks greatly during the summer months, but is generally sustained. This could change and disrupt the current trout fishery, an important recreational resource of the lake and valuable to many lake users.

From a design consideration perspective, the scale of the system as well as the morphometry of the lake could provide challenging. Lake Hopatcong is highly dendritic, and can generally be described as long and narrow with numerous branching coves and sub-basins. The target area for such a system should include, at a minimum, the entirety of the lake subject to thermal stratification and anoxia. The relevant anoxic zone target area is provided in Table 8-1.

Table 8-1: Anoxic Zone Target Area (Mean from 2019-2021)

Metric	SI Value	SI Units	Imp. Value	Imp. Units
Maximum Anoxic Area	3.95E+06	m2	976.0	acres
Minimum Anoxic Boundary	5.87	m	19.3	feet
Mean Anoxic Zone Depth	10.71	m	35.1	feet
Maximum Depth	16.7	m	54.8	feet
Maximum Anoxic Load	5,104.70	kg P	11,253.90	pounds P

The greatest expense in the installation of destratification systems is typically the air line cost. Here the anoxic zone extends approximately 4.5 miles from one extreme to the other following an arcing path along the centerline of the lake. While a single system might be proposed, a number of sub-systems is more likely. This would, to some degree, shorten the required line runs, especially if larger supply lines are used. It would also eliminate some of the system inefficiencies inherent in very long runs, including friction losses. The design could vary widely; it may consist of larger diameter trunk lines that are fitted with manifolds to feed each individual diffuser, generally only possible in shorter line runs, or may consist of individual air lines to each diffuser. If a single compressor station was used, and centrally sited, the longest line runs would be in excess of 12,000 feet at a minimum. The number of diffusers could vary greatly as well and could range from 100 up to possibly as many as 500 units. Spacing is determined of course by the design of each diffuser, but the general rule is that required spacing is inversely proportional to depth. The air lift or circulation potential increases with depth because the bubbles rise in a cone covering a wider area. Density would therefore increase in shallower portions of the target zone. Between the number of diffusers and the length of the line runs, the total line runs would likely quickly range into multiple 100,000s of feet.

Another factor to be considered is boat traffic and recreational use of the lake. While the in-lake components are subsurface and the lines rest on the lake bed, they are not uncommonly snagged by anchors. Depending on how this is treated, this could result in the detachment of the line from the diffuser or the line hauled to the surface. Proper use of negatively buoyant line and anchoring/mooring should reduce the risk of the line remaining near the surface, but this does happen and becomes a navigation hazard; additionally, damage to the line can occur due to propellors.

Siting concerns must also be considered with the establishment of one or multiple compressor houses that will become fixed infrastructure. Additionally, systems of this size will require at least three-phase 440-V electric service to power the compressors given the volume of air necessary. Utility fees alone for annual operation are likely to run to well over $100,000. Annual service requirements for the most part would be expected to be relatively low. This would include items such as filter changes, basic inspection, and potentially oil changes. If multiple smaller systems or fewer but larger compressors are used the total annual maintenance is expected to be about the same at perhaps $10,000 to $15,000.

Ownership of the system, ownership of the land, and responsibility for utility costs, operation, and maintenance would all need to be decided well in advance of placement and could be complicated given the number of lake stakeholders and the number of municipalities abutting the lake.

It is very difficult to estimate capital costs at this point, given that basics like air line length, number of diffusers, and compressor specifications or even the number of systems has not been determined. However, some preliminary figures are provided based on recent price estimates developed at other similarly sized lakes in New York and New Jersey and scaled appropriately. The cost estimates include just the labor and machinery costs, but exclude design, permitting, electrical service, and compressor house construction. At a minimum, costs are likely to start at a minimum of $900,000, with a median of roughly $1,200,000 to $1,650,000, up to a $2,500,000 and possibly more.

8.1.2 HYPOLIMNETIC AERATION SYSTEMS

An alternative to the use of destratification is the use of hypolimnetic aeration systems. Conventional full-lift hypolimnetic aeration systems, as well as depth specific/Layer Air™ systems, make use of a "tube within a tube" design. Like destratification systems, they use compressed air to lift deep oxygen-poor water higher in the water column, sometimes to the surface, but with hypolimnetic systems the deep anoxic water is returned to the bottom of the lake following re-oxygenation; this return is driven by temperature-dependent density differences between the waters. Even though a major transfer of DO results between the warm and cold water, because of the limited duration of time needed to mix the cold, oxygen-poor bottom water with the warm, oxygen-rich surface water, only a nominal increase in water temperature is experienced. Thus, the lake remains thermally stratified. Hypolimnetic systems may be operated to maintain either high (>4.0 mg/L) or minimal (1.0 mg/L to 2.0 mg/L) DO concentrations at the lake bottom or targeted aeration zone. If the goal is to use the aeration system to control internal phosphorus loading, the targeted DO concentration may only utilize the lower bound as sufficient to prevent internal phosphorus leaching. Conversely, if the goal is to create or maintain coldwater fish habitat as well as control internal phosphorus loading, the targeted DO concentration may be significantly higher. The hypolimnetic unit may be equipped with a hatch, air tube, or vent that allows any hydrogen sulfide present in the anoxic bottom waters to be released into the atmosphere.

An adaptation of or alternative to standard hypolimnetic aeration is depth specific aeration. Depth specific or Layer Air™ systems also use a "tube within a tube" design approach similar to conventional hypolimnetic systems (Moore, et al., 2015). The difference is that water is drawn into the unit at a specified depth, usually the upper stratum of the hypolimnion or from the metalimnion. Once again compressed air is used to lift oxygen-poor water higher in the water column. The anoxic or hypoxic water is then mixed with the more highly oxygenated surface water and then returned to the stratum from which it was drawn. Typically, multiple mixing units are positioned within one or more strata. The designated stratum often provides the water temperature needed to provide critical holdover summer habitat for coldwater fish. Although depth specific/Layer Air™ systems are designed and operated to preserve thermal stratification, they usually are not designed to eliminate bottom water anoxia. Thus, the epilimnetic and metalimnetic strata will be well-oxygenated, but the hypolimnetic strata may remain anoxic. Because deep water anoxia is not abated, hypolimnetic internal phosphorus loading may still occur. But by creating a thermally separated, oxygen-rich mid-water depth zone, it is possible to maintain separation of the phosphorus-rich hypolimnetic water from the photic zone of the epilimnion where photosynthesis occurs. Any phosphorus liberated from the anoxic lake bottom thus remains contained at the bottom of the lake and does not become available for biological uptake.

For depth specific/Layer Air™ and hypolimnetic systems operated in a manner that does not completely prevent anoxia and associated internal phosphorus loading, it will be necessary to manipulate the operation of the system immediately in advance of the lake's natural turnover to prevent the late season upwelling of phosphorus rich water to avoid an autumnal algae or cyanobacteria bloom. In such cases, the lake's thermal and DO profile will need to be closely monitored. As the water column begins to naturally cool and surface to bottom water temperatures become increasingly uniform, mixing and reoxygenation of the deeper portions of the lake is intensified resulting in enough deepwater DO to create the oxic conditions necessary for the liberated inorganic phosphorus to re-bind with available dissolved iron. Once bound again to iron, the sedimentary recycled phosphorus can no longer be bioassimilated by algae or cyanobacteria. These operational considerations as well as the generally shallow anoxic boundary would both suggest that these systems are not well suited to use at the lake.

The power needs, operation, and maintenance of conventional hypolimnetic and depth specific/Layer Air™ systems are very similar. The compressors tend to be relatively large (at least 20 HP) and usually require a 3-Phase, 220-volt or 440-volt power source. The compressors for either type of system must be housed in a large footprint (at least 30' x 30'), properly insulated, and vented compressor building. To decrease the length and cost of the air line runs, the building should be located close to the shore and as close to deepest area of the lake as possible. The majority of the material costs associated with hypolimnetic and depth specific/Layer Air™ systems are related to the length, number, composition, and size of the air lines as well as the size and number of compressors. Additionally, the system's cost will be a function of the type and number of mixing units, all supporting system elements (filters, expansion tank, flow meters, etc.), and the construction of the compressor building. The compressors and supporting system elements must be serviced at least annually by manufacturer certified service providers and professional divers may be needed to periodically inspect and service the mixing units and the air lines. Utility costs are a function of the number and size of the compressors but can be expected to be significant ($30,000-$50,000 annually) given that hypolimnetic and depth specific/Layer Air™ systems must be operated continuously from once the lake is thermally stratified in early- to late-spring until early fall after the lake turns over.

Certainly, hypolimnetic aeration systems offer a significant advantage over destratification systems, the preservation of thermal stratification such that impacts to coldwater fishery habitat is minimized. Depending on its operational goal, it may even be used to expand that habitat by significantly increasing mid- and deep water column DO concentrations. This would be advantageous for the maintenance of trout habitat in Lake Hopatcong although likely to difficult to achieve in practice given the thermal structure and DO regime of the lake.

But much like standard submerged destratification systems, there are a number of drawbacks, and most of these are held in common. One significant difference is the size of the in-lake apparatus. Full lift systems may be very large, extending from the lake bed to at least the thermocline and potentially to the surface and are usually at least several feet in diameter and sometimes significantly larger. The units often use a large buoyancy chamber at the top of the water column which helps to maintain the unit in a vertical attitude and makes the unit telescopic so that it can be collapsed in place. The life-span is not particularly long. In addition, those that have the chamber require winterization, and they are typically flooded and moved subsurface to avoid ice damage. This of course introduces another navigation hazard to the lake. The depth specific units do not reach the surface, yet they are physically large structures placed in the lake. Additionally, due to their size they require large cranes to launch and divers to help with the installation.

The number of units needed to adequately aerate the lake to prevent internal loading is not known at this time, but there are likely to be a minimum of ten such units, and possibly more given the total volume to be aerated or the design of the unit. Unlike destratification which is more confined by simple depth, here the volume of water that can be adequately aerated is more important to achieve the desired DO management concentration. Certainly, the length of air line runs will be much less in total if these systems are used, yet the total length of runs will continue to be a problem and will certainly impact operational efficiency of the units.

Compressor maintenance and utility costs will likely be similar to that described for destratification, but the maintenance and life span of the unit itself must be considered. This requires larger equipment and is logistically more difficult and would include in-the-water operations. The operation of the unit itself can be more complicated, especially if the telescopic devices are used which must be floated to the surface in the spring and sunk in the fall. Monitoring DO concentrations on a consistent basis is critical and the system must be adjusted accordingly to maintain the desired DO concentration and preserve stratification. This is much more complicated than destratification, yet even for destratification systems it may be important to adjust the system as necessary to maintain the proper vertical mixing and DO while limiting resuspension of sediments and entrainment of phosphorus and particulates. Achieving this balance would be even more difficult for the depth specific aeration, especially if used to improve fisheries habitat. The unit must be operated to prevent destratification, while maintaining the proper DO concentration, and then during periods of metalimnetic erosion and prior to fall turnover must be adjusted to precipitate phosphorus prior to mixing.

Again, questions of responsibility for operation and maintenance must be considered, as well siting and placement of the compressor houses and access to the required electrical service.

Again, providing any realistic assessment of system difficult is exceedingly difficult, yet for planning purposes such systems are likely to start at approximately $2,500,000. Overall, while the preservation of habitat is a distinct advantage, any such system will be exceedingly expensive and likely difficult to operate and maintain.

8.1.3 DIRECT INJECTION PURE OXYGEN SYSTEMS

Pure oxygen systems (also referred to as direct oxygen or DOX systems) are typically used in large, deep lakes and run-of-river reservoir systems. As with hypolimnetic systems, stratification is maintained but anoxia is overcome. Thus, DOX systems are well suited for managing internal phosphorus loading in deep lakes where stratification must be maintained to support a coldwater fishery. Unlike hypolimnetic systems, DOX systems do not rely on an air lift approach to circulate and oxygenate the hypolimnion or rely on atmospheric oxygen to reoxygenate the lake. Rather, oxygen is directly introduced to the lake water.

There are basically two types of direct injection oxygen aeration systems: line aeration systems and Speece cone systems. Although the design and operation of line aeration and Speece cone systems are very different, the underlying approach is similar involving the direct mixing of oxygen gas with lake water. Speece cones utilize a water pump to transfer the oxygen-poor hypolimnetic water into a large, cone-shaped chamber where the water is then mixed with oxygen gas before being released back into the hypolimnion. This form of aeration may also be referred to as slip stream aeration when the Speece cone is located on shore rather than placed directly in the lake. Speece cones are typically very large and their operation requires the use of a large water pump. The purpose of the pump is to extract water from the hypolimnion, pump it into the Speece cone, mix the water with the injected oxygen, and then pump the re-oxygenated water back into the lake's hypolimnion. Oxygen is supplied either by an onsite oxygen generator or oxygen stored in onsite tanks. The oxygen-poor water and oxygen gas are mixed at the top of the conical shaped structure. The shape of the Speece cone maximizes the contact and mixing of the oxygen-poor water with the introduced oxygen gas. The pumping, reoxygenation, and mixing process does not increase the temperature of the hypolimnetic water or result in enough turbulence to cause the lake to thermally destratify.

Line diffuser DOX systems use fine-pore oxygen diffusion lines to introduce O_2 directly into the water. This approach does not rely on any pumps to draw in hypolimnetic water. Rather oxygen gas is released under low pressure into the lake via porous air lines anchored along the bottom but suspended above the sediments. As with Speece cone systems, oxygen may be supplied from large land-based oxygen gas storage tanks or by means of a land-based onsite oxygen generator. At a minimum, the amount and rate of oxygen delivered into the hypolimnion must be enough to exceed the lake's computed biological and sediment oxygen demand. This will vary seasonally with peak demands occurring in the middle of the summer. When the goal is to sustain coldwater fish habitat, more oxygen may be supplied and higher hypolimnetic DO concentrations maintained. If the goal is to control internal phosphorus loading, the targeted DO concentration may be as low as 1.0 mg/L to 2.0 mg/L DO. In either case, the rate at which the oxygen gas bubbles are released from the porous air lines is not enough create the turbulence needed to thermally destratify the lake. Thus, anoxia is prevented and internal phosphorus loading is controlled, but the lake remains in a thermally stratified state.

Speece cone and line diffuser aeration systems are operated continuously from late spring/early summer through late summer until the fall turnover. The system is started after thermal stratification and hypolimnetic anoxia occurs and is shut down after the fall turnover, when the lake is thermally destratified and there is no longer the potential for deep water anoxia. Of the two pure oxygen aeration techniques, the DOX line diffuser systems are less expensive and easier to construct, install, and operate than the Speece cone systems.

The power requirements for DOX line diffuser systems can be met using a standard 110 or 220-volt, single phase power source. The amount of land and the size of the building needed for DOX systems will vary depending on whether the system uses oxygen storage tanks or an oxygen generator. For an oxygen tank system, a secure, fence-enclosed concrete pad needs to be constructed along with a structure large enough to house all the supporting metering and gauging equipment. In total, this may require as much as 0.25 acres of land.

Conversely, if the system's oxygen supply is met using an oxygen generator, the building needed to house the system will be much smaller (30' x 30'), similar to that associated with a hypolimnetic or destratification system. In either case the pad/building should be located near the shoreline and as close to the deepwater area of the lake as possible. This will help decrease costs associated with the oxygen supply air lines. Inspection and maintenance of direct oxygen injection systems focus on the routine inspection (at least bi-weekly) of the oxygen reserves in the storage tanks or the operation of the oxygen generator. Annually, divers will need to inspect all in-lake components of the system, in particular the oxygen supply lines. Some systems though may be constructed with built in buoyancy lines so that the system can be floated to the surface at need, although this would double the length of line runs. Operational costs are highly variable and dependent on the lake's biological oxygen demands and hypolimnetic DO goals. For systems supplied by land-based oxygen storage tanks, the cost of tanker truck oxygen deliveries must be taken into consideration.

Speece cones or slip stream systems seem poorly suited for use at Lake Hopatcong due to space, power consumption, and oxygen demand requirements. The line aeration DOX systems seem to provide a better benefit with low power consumption, preservation of stratification, and little impact to coldwater fishery habitat and could be effective in limiting internal loading in the lake.

The main issues repeat themselves from the other aeration methods. One of the biggest concerns would be siting the apparatus. It is estimated that at a minimum 0.25 acres would be required to construct a concrete pad, position the transfer equipment, and maintain the large oxygen tanks or onsite generating equipment needed to operate the system. Oxygen storage safety and public perception must also be weighed, especially in residential areas. Additionally, such a space would have to be sufficiently large to allow regular and repeated delivery of liquid oxygen. There is also the in-lake apparatus to consider, again contending with issues of length of porous hose runs, although friction and overheating would be of a lesser concern because the systems are operated at such low pressures; also, these lines are usually suspended off the bed greatly increasing the risk of the lines being entangled with anchors.

The chief design concern with these systems is oxygen demand. The systems must be designed to supply and distribute sufficient oxygen to counteract the hypolimnetic oxygen demand which would include both sediment oxygen demand and oxygen demand related to decomposition of suspended organic materials in the water column; both are likely to be quite high based on the rapid depletion of oxygen upon stratification and the presumably high detrital loads of senesced algal cells. This demand will vary in both rate and scale as the growing season progresses and the anoxic zone volume increases. This determination can be made in several ways including the analysis of *in-situ* DO data collected with a water quality probe or the incubation of sediment samples and hypoxic water in a controlled laboratory environment. Another advantage to the use of line aeration is that latent oxygen demand can be suppressed after some time, although the effect is slow to be realized; a recent paper reported than oxygen demand had decreased by 75% over the course of several decades in ta studied system (Horn, et al., 2019).

Again, it is very difficult to estimate the cost of one of these systems and costs are usually proportional to design oxygen supply needs, but capital costs of $1,000,000 to $2,000,000 are likely based on similar recent cost estimates. Operational costs are likely to be quite high. While the actual energy costs are anticipated to be quite low, the bulk of the costs will be for the delivery and supply of oxygen necessary to operate the system, as well as labor costs associated with regular system inspection and operation, which is a more technical nature than some of the other systems and will require carefully monitoring *in-situ* hypolimnetic DO concentrations and adjusting the system accordingly as demands change over the growing season. Annual operational costs are anticipated to start at around $250,000.

8.1.4 COMPLEMENTARY MANAGEMENT PRACTICES

While the various aeration systems can be important tools to manage internal loading, they are but one of the tools to manage lake water quality. A holistic approach includes employing varying management schemes concurrently to most efficaciously manage the load. As such, source control of watershed loads should remain a major focus of management activities. In addition, the use of a nutrient inactivant is often recommended prior to the startup of an aeration system. In essence, this helps to jump start the process of managing the internal load by limiting near term loading rates, addressing any extant load, and generally increasing efficacy as well as limiting impacts on startup, especially for destratification systems, including inadvertently mixing phosphorus or suspended matter through the water column.

8.1.5 PERMITTING PROCESS

While NJDEP does not specifically have a permitting process for the installation of an aeration system, there are components of the associated infrastructure that may require State approval. For example, if an aeration system were designed for Lake Hopatcong, there would likely be multiple compressor locations given the size of the lake and the cost of the weighted line. A system based on one large compressor house for the entire lake would likely be substantially higher in cost, again due to the cost of the weighted line, rather than having multiple, smaller compressor house locations. However, this means that such a design would need to take into account all shoreline areas being considered for compressor and line infrastructure.

In addition to addressing local property owner concerns, any potential freshwater wetlands would need to be identified through a wetlands/transition area letter of interpretation (LOI). Besides wetlands, flood hazard and riparian zones need to be identified at the shoreline compressor/infrastructure sites. Thus, depending on site specific conditions, it is certainly possible that a Freshwater Permit and/or a Flood Hazard Permit may be required. Prior to any final design plans being developed, the shoreline compressor locations need to be clearly established and it needs to be determined if any NJDEP permits are required. Other factors that need to be considered are if there are any local electrical infrastructure issues that need to be addressed and if there are any local municipal or County permit approval requirements.

Finally, regardless of the level of permitting that is required for such a proposed project, it should be noted that for the State's largest public waterbody, the installation of any whole-lake aeration system would require some level of review and approval by NJDEP.

8.1.6 GENERAL ASSESSMENT

Overall, the potential use of aeration technologies at Lake Hopatcong seems limited for varying reasons. While the systems could meet the base management goal of reducing the internal load, there are other practical and cost considerations. From a management perspective, the use of destratification systems, the most commonly used aeration technology in the region, is a non-starter if the preservation of trout habitat is important.

The size, shape, and depth of the lake as well as the anoxic zone (nearly 1,000 acres) is a major concern and a major factor in the price point. For almost all of these systems air line or other line run lengths are likely to be very high, even with the use of multiple sub-systems in tandem. Typically, it is the line costs and their installation that account for most of the capital investment. From an operational perspective, long line runs limit efficiency and may require larger equipment to overcome those losses.

All the systems require extensive and complex in-lake infrastructure. In common, all will have various line runs, while most of these would lie directly on the lake bed; in the case of line diffusion technology the lines will be suspended off the bottom. Hypolimnetic aeration and depth specific aeration would include large tower like structures, and in some cases may extend to the surface. Given the amount of boat traffic, these systems could potentially be navigation hazards and all would be at risk from anchors.

Siting of the on-shore apparatus will be important, especially if multiple sub-systems are used, a significant issue at a waterbody that has extensive shoreline development. Access to specialized high-voltage electrical service is a concern. Permitting is another consideration and could add considerable time and cost to the project.

All the systems are expensive, but these could be viewed as one time or generational costs. However, all have extensive operational and maintenance costs. Utility fees would be high for destratification and hypolimnetic aeration. Line aeration would require continual resupply of oxygen and onsite storage or the construction of oxygen generators. All will require, to varying levels, careful management and technical operation.

The last concern speaks to responsibility. Who will own the systems and on-shore lots? How will responsibility for operation be decided? Where are the systems sited? How will annual maintenance and utility costs be provided? Overall, if such a system is selected, it is a significant and long term commitment to the management of the lake. Additionally, to continue to manage the internal load they would need to be operated for the foreseeable future. While long term operation may reduce the rate of loading over time, the base conditions that drive stratification and anoxia, namely water temperatures, lake morphometry, and eutrophic conditions, are unlikely to change and when not actively managed internal loading and phosphorus leaching will continue to manifest in the lake and impact its ecology.

8.2 NUTRIENT INACTIVATION

Nutrient inactivation is a common in-lake management tool utilized to control phosphorus availability and internal phosphorus loading. Over the past forty years, nutrient inactivation has been successfully implemented in the management and restoration of numerous lakes and reservoirs located in New Jersey, Pennsylvania, New England, and the mid-West (Cooke, et al., 2005). Some of these products have been used for over four decades in the restoration of eutrophic, phosphorus-rich lakes (Cooke, et al., 1977; Cooke, et al., 2005). Numerous studies and supporting data, going back to the 1980s, demonstrate the effectiveness of nutrient inactivant treatments, in many cases accomplished following a single application of a nutrient inactivant (Huser, et al., 2016).

Nutrient inactivant products are used in two, often overlapping, ways:

- To decrease water column dissolved phosphorus concentrations through binding and precipitation
- To bind phosphorus within sediment interstitial pore water or as that phosphorus is released from the sediments during periods of deep-water anoxia

The primary objective of any nutrient inactivant project is to decrease the concentration of biologically available phosphorus within a waterbody. There are a number of nutrient inactivant products that can be used to manage phosphorus availability. Examples of the most commonly used products are:

- Aluminum sulfate (alum)
- Sodium aluminate
- Buffered alum (alum + sodium aluminate)
- Polyaluminum chloride (PACl)
- PhosLock (lanthanum)

- Various ferric (iron) products

Of the above, the most commonly used nutrient inactivant is aluminum sulfate (alum). Alum has been used for over 70 years in drinking water treatment plants as a coagulant to strip water of particulate material. Dating back to at least the 1980s, alum has also been widely used in managing and restoring lakes. A primary advantage of alum is that it is produced in bulk and is therefore relatively inexpensive and easy to obtain.

Upon contact with water, alum forms a fluffy, amorphous aluminum hydroxide precipitate. This precipitate is the result of the liberation of aluminum ions, which are immediately hydrated, and through a progressive series of hydrolysis form aluminum hydroxide. The resulting colloidal, amorphous floc has high coagulation properties that bind, concentrate, and settle particulate phosphorus and other suspended materials from the water column. The aluminum hydroxide also effectively binds dissolved forms of phosphorus. Because the floc is heavier than water, it settles out of the water column over a 12 to 24 hour period and becomes integrated into the lake's soft sediments. The remaining active aluminum hydroxide contained in the floc binds interstitial pore water phosphorus, including that released under periods of anoxia. Any bound phosphorus becomes inactivated, meaning that it will not be leached nor available for biological uptake by benthic algae and phytoplankton.

The phosphorus bound by the alum is insoluble in water. The major benefit of using alum and the other aluminum and lanthanum-based nutrient inactivants is that the bound phosphate is not redox sensitive and the bound phosphorus will remain biologically unavailable even if the overlying waters become anoxic. This cannot be accomplished using the iron based inactivant products. Welch and Cooke (1999) showed that following a single surface application of alum, the internal loading rate in 7 out of 7 dimictic, eutrophic lakes was reduced on average by 80%, while water column bioavailable concentrations of phosphorus remained low for an average of 13 years.

The primary disadvantage of alum is that it will cause the pH of a treated lake to decrease and become more acidic. The extent to which this happens is largely a function of the lake's buffering capacity, defined by its natural alkalinity. If the pH following an alum treatment drops below 6.0 to 5.5, the aluminum in the applied alum can enter a dissolved state which may be toxic to fish and other aquatic life. However, the literature also shows that alum can be safely utilized with no negative consequences to aquatic biota if changes in pH are effectively managed (Welch and Cooke, 1999; Steinman et al. 2004). Essentially this means applying alum only in lakes having adequate buffering capacity as well as applying alum carefully in an amount that does not cause the pH to drop excessively. It should be noted that decreased pH as a direct result of alum application is quite short-lived and the upper water column often recovers by the time the precipitate has settled out of the water column. In the long term, if the treatment is successful and hypolimnetic phosphorus concentrations are significantly curtailed such that algal productivity is decreased, pH is likely to decrease as a result of decreased productivity although this effect would be very minor.

Another disadvantage to the use of alum is that its efficacy is finite. Once bound to phosphorus it does not continue to bind additional phosphorus. In addition, particularly in lakes where there is significant sedimentation, the floc blanket may eventually become buried and thus prove less efficacious in capturing phosphorus at the sediment water interface. However, this can be easily countered by adjusting the dose rate; typically, applications are designed to capture five years of internal load. As such, repeated treatments are often necessary, but this is generally on the scale of 5 to 10 years, and should decrease as more and more phosphorus is bound over time.

While there are some disadvantages to the use of the nutrient inactivants, these are generally easily managed. Total application rates can be adjusted to provide longer efficacy. Toxicity issue related to aluminum moving into the soluble phase and the related issue of decreasing pH during treatment are also managed. First, product selection and determining dose rates is key to avoiding these issues. One of the key tools is the use of bench testing. The bench test involves the collection of large-volume, depth integrated samples from the subject lake. At a laboratory, iterative doses of various nutrient inactivant products are made at successive increasing dose rates. At each step the water is monitored for a variety of parameters including total and dissolved phosphorus, pH, and dissolved aluminum at a minimum in addition to other parameters like alkalinity. The goal is to mimic the application process in the lake and determine the product performance with special attention provided to pH changes, dissolution of aluminum, and phosphorus capture. Based on the bench test different dose rates are calculated including the effective dose to remove the desired load as well as the safe dose, which is the dose that can be applied without inducing pH to drop below a safety threshold, usually somewhere 6.2, or produce dissolved aluminum above a low threshold. The selection of the product is also a factor. In lakes that are poorly buffered that are subject to rapid changes in pH a buffered product could be used or another alternative like PACl. Issues can also arise in extremely basic (high pH) lakes, and in such a system an acidified product might be considered.

Another way to limit toxicity and acidification is to monitor water quality during the application. While a proper dosing design should largely eliminate these impacts, monitoring pH in the water column is the most reliable way to address this issue. If pH should reach the safety threshold, which is generally an order of magnitude above the pH point at which there will be significant dissolution of aluminum, a treatment plan will identify the appropriate measures in which the treatment can be paused until the pH recovers, moved to a different section of the target treatment area, or the total dose or rate of application reduced.

Despite these issues, there are many advantages to the use of nutrient inactivants. The first is that these are highly efficacious. If designed correctly, the bulk of the internal load would be almost immediately controlled. This may not yield obvious immediate results in the trophic zone, but during that critical period in the late growing season it should limit productivity. The application of nutrient inactivants, when used as a single dose intended to blanket the lake bed within the anoxic zone, is a discrete event; here the application may be on the order of 10 to 20 days given the size and volume to be treated and depending on the contractor and the size of the treatment vessels. As such, there is no continuing operation or maintenance. There is no permanent infrastructure. At worst, the lake may require future treatments, usually at around 5 year intervals. When applied, the product is usually visible as a thin cloudy slurry during the application, but the product settles quickly and most visible traces diminish in under a day. Typically, because of its coagulant properties, suspended solids are stripped from the water column and there is often a lingering benefit of greatly improved clarity that may last several days to several weeks, although it is usually beneficial to apply when suspended solids loads are low to improve phosphorus retention.

Like the aeration products, there is no specific permitting process for the use of nutrient inactivants. These products have been deemed to not be pesticidal and therefore are not regulated in any real sense, although NJDEP should certainly be consulted in the process. The potential need for securing additional permits is unsure, but flood hazard or wetlands are both possibilities, although wetland areas will be avoided entirely with the anoxic zone target area essentially following the 20 foot isobath and no infrastructure will be erected.

While pricing for aeration products is difficult without some preliminary design and identified vendor, this project developed much of the data required to assess costs. The target application area has been largely identified and quantified (see Table 8-1), as well as the need to target a maximum internal load of approximately 5,100 kg of phosphorus annually. The major missing component at this time is the bench test, which will be used to select the final product and develop the application rates. However, if an assumption is made that alum will be the selected inactivant and that the application will target five years of internal load control, a reasonably priced estimate can be developed. Such an application would cost around $360,000. The largest line item is the product cost followed by application labor. This would also include monitoring both during and pre- and post-treatment, bench test analysis, development of the application program, consultation with NJDEP, and related matters.

Overall, Lake Hopatcong is an ideal candidate for nutrient inactivation. It is a large, deep lake. It is dimictic, with strong hypolimnetic anoxia, and a large internal load. A single application could provide efficacious control for several years. While the application period may cause some disruption to the recreational use of the lake this would be limited to that short time during the treatment; once completed there will be no in-lake structures. More importantly, there is no maintenance burden or ongoing significant utility costs. It is also very price competitive. The capital expense of a single 5-year nutrient inactivant treatment is about one-third of the lower end of price estimates for the various aeration systems.

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