Climate Change Vulnerability of Eutrophication and Algal Blooms in New York State

Final Report | Report Number 22-32 | March 2022

NEW YORK STATE | NYSERDA

NYSERDA’s Promise to New Yorkers:

NYSERDA provides resources, expertise, and objective information so New Yorkers can make confident, informed energy decisions.

Our Vision:

New York is a global climate leader building a healthier future with thriving communities; homes and businesses powered by clean energy; and economic opportunities accessible to all New Yorkers.

Our Mission:

Advance clean energy innovation and investments to combat climate change, improving the health, resiliency, and prosperity of New Yorkers and delivering benefits equitably to all.

Climate Change Vulnerability of Eutrophication and Algal Blooms in New York State

Final Report

Prepared for

The New York State Energy Research and Development Authority

Albany, NY

Amanda Stevens
Project Manager

Prepared By:

Hazen and Sawyer

New York, NY

Hazen and Sawyer
Ben Wright, PE
Principal Investigator

Elizabeth Crafton-Nelson, Ph.D.
Director of Water Quality Protection and Restoration

W. Josh Weiss, PhD, PE, D.WRE
Director of Innovations – Water Resources

and

Upstate Freshwater Institute

Syracuse, NY

Susan M. O'Donnell
Dave M. O'Donnell
Senior Engineers

David A. Matthews, PhD
Director

NYSERDA Report 22-32 NYSERDA Contract 60447 March 2022

Notice

Abstract

A growing concern in New York State and across the United States is the occurrence of cyanobacterial harmful algal blooms (CHABs), which are caused by excessive growth of cyanobacteria. Certain cyanobacterial taxa are capable of producing hepatotoxins, neurotoxins, and endotoxins, which can cause widespread economic loss associated with decreased recreational spending and increased water treatment costs. In general, the primary drivers of cyanobacterial blooms are elevated water temperatures and nutrient levels. Both of these factors are either directly or indirectly affected by climate change through warming temperatures and increased potential for extreme storms. Through the use of mechanistic water quality models, this project explored the possible impact future climate conditions may have on hydrodynamics and the likelihood of CHABs in three Central New York Finger Lakes (Cayuga Lake, Owasco Lake, and Skaneateles Lake). The project results suggest that future climate change will produce conditions in New York State lakes that favor the growth of cyanobacteria and increase the likelihood of CHABs. While a continued focus on nutrient load reduction is critical to prevent worsening conditions, nutrient management cannot be expected to completely stop all cyanobacterial blooms given future temperature trends and that oligo-mesotrophic lakes are at risk of CHABs. Given the complexity of the problem with cyanobacteria blooms, it will take a combination of short- and long-term measures to effectively manage CHABs across the lakes in the State.

Keywords

Climate change, harmful algal blooms, HABs

Table of Contents

List of Figures

List of Tables

Summary

A growing concern in New York State and across the United States is the occurrence of cyanobacterial harmful algal blooms (CHABs), which are caused by excessive growth of cyanobacteria. Certain cyanobacterial taxa are capable of producing hepatotoxins, neurotoxins and endotoxins, which can cause widespread economic loss associated with decreased recreational spending and increased water treatment costs. In general, the primary drivers of cyanobacterial blooms are elevated water temperatures and nutrient levels. Both of these factors are either directly or indirectly affected by climate change through warming temperatures and increased potential for extreme storms. Cyanobacteria populations are also influenced by trophic interactions with other organisms and distributed by wind and wave action in a lake.

This project explored, through the use of mechanistic water quality models, the possible impact future climate conditions may have on hydrodynamics and the likelihood of CHABs in three Central New York Finger Lakes (Cayuga Lake, Owasco Lake, and Skaneateles Lake). These lakes are relatively unproductive (oligotrophic-mesotrophic), and serve as public water supplies, venues for a wide range of recreational pursuits, and vital economic engines for the region. The general approach adopted for this project was to use local meteorological data (e.g., air temperature and precipitation), downscaled from multiple Global Climate Models (GCMs) to drive the two-dimensional in-lake simulation model CE-QUAL-W2 for each lake. Modeling of Cayuga Lake and Skaneateles Lake was limited to hydrodynamic processes (i.e., water motion, temperature), while simulations for Owasco Lake included nutrient cycling and growth of three phytoplankton groups (diatoms, green algae, cyanobacteria).
In addition to the CE-QUAL-W2 models, a watershed model (GWLF-E) was used to estimate daily flows and nutrients loads for Owasco Lake tributaries. These inputs were required to establish hydrologic and material loads that support water quality modeling of Owasco Lake.

The climate projections were developed using monthly change factors based on the GCMs for available greenhouse gas scenarios (RCP[1] 4.5 and 8.5) and future time slices (2020,[2] 2050, and 2080) from the New York State Energy Research and Development Authority (NYSERDA) ClimAID data set. The change factors were applied to the historical data to create the representative future daily temperature and precipitation projections for the specific locations evaluated in the modeling. The monthly change

factors were applied to location-specific data for each lake to develop projections used for the modeling. The monthly change factors are additive for temperature and multiplicative for precipitation. A subset (n=34) of the 210 climate scenarios were selected to get broad coverage of potential future changes to precipitation (-2% to 20%) and temperatures (+1℃ to +7℃).

Because invasive dreissenid mussels (zebra and quagga mussels) have established large populations in all three study lakes, we hypothesized that selective filtration and nutrient recycling by these mussels may contribute to recent increases in the frequency and severity of CHABs. Dreissenid mussels have been found to be an important determinant of CHABs in New York State lakes, particularly in lakes with lower total phosphorous (TP) and chlorophyll-*a* (Chl-*a*) concentrations. Simulations were run with and without a calibrated dreissenid mussel sub-model for Owasco Lake to identify changes in cyanobacteria populations under base climate conditions. To advance our understanding of the role of wind-driven transport in determining the spatial distribution of CHABs, the hydrothermal sub-model was run for each of the three study lakes with a buoyant conservative tracer used to mimic the transport of cyanobacteria due solely to water motion.

A variety of metrics were evaluated to compare the output from the 34 modeled climate scenarios, which included heat income, stratification (duration, depth, start and end dates), Schmidt stability, temperatures (upper water maximum temperature, average temperature, number of days >25°C), total phosphorus, chlorophyll-*a* (Chl-*a*), and phytoplankton community composition (diatoms, green algae, cyanobacteria). The lakes exhibited strong shifts in metrics related to air temperature changes (Figure E-1 and Figure E-2), indicating that temperature effects of climate change will translate to warmer lake waters more favorable for growth of cyanobacteria.

In contrast, the results from the eutrophication modeling, which included both temperature and precipitation effects, were less distinct (Figure E-3). Evaluation of the modeled results indicated that increasing temperatures would shift the population distribution away from diatoms, which are heat-sensitive, and toward green algae and cyanobacteria. Precipitation changes would increase phosphorus loading of the lake (Figure E-4 and Figure E-5), but the simulated change was relatively small for most scenarios. Scenarios with more substantial shifts in phosphorus exhibited increases in overall algal biomass, and higher temperatures enabled cyanobacteria to flourish. Further, review of data on a daily level indicated that for scenarios where there is not a major shift in population, cyanobacteria concentrations can exceed those in the base climate modeling for a significant number of days on average (Figure E-6). In addition, the modeling identified that dreissenid mussels caused a reduction in phytoplankton biomass but an increase in cyanobacteria concentrations due to selective filtration. Hydrodynamic modeling established that advection of surface layers by wind, predominantly from the south, could explain the higher frequency and severity of CHABs in the northern ends of the study lakes.

These results suggest that future climate change will produce conditions in New York State lakes that favor the growth of cyanobacteria and increase the likelihood of CHABs. While a continued focus on nutrient load reduction is critical to prevent worsening conditions, nutrient management cannot be expected to completely stop all cyanobacterial blooms given future temperature trends and that oligo-mesotrophic lakes are at risk of CHABs. Lake monitoring to identify presence of blooms is necessary to support beach closures and public announcements to prevent public exposure to potential toxins. Additionally, while many typical water treatment processes are effective against cyanobacteria and cyanotoxins, plant operators may need to adjust processes to maximize removal efficiencies during a bloom. In-lake measures, such as mixing or hypolimnetic aeration, may help limit cyanobacterial growth in smaller waterbodies with anoxic hypolimnia, but are impractical in large lakes. Application of algaecides, particularly hydrogen peroxide-based chemicals, can be a useful reactive measure to selectively target cyanobacteria as a bloom is forming to prevent its growth and shift the population distribution back to green algae. Given the complexity of the problem with cyanobacteria blooms, it will take a combination of short- and long-term measures to effectively manage CHABs across the lakes in the State.

Figure S-1. Annual Average Temperature Change Factor for Each Modeled Scenario

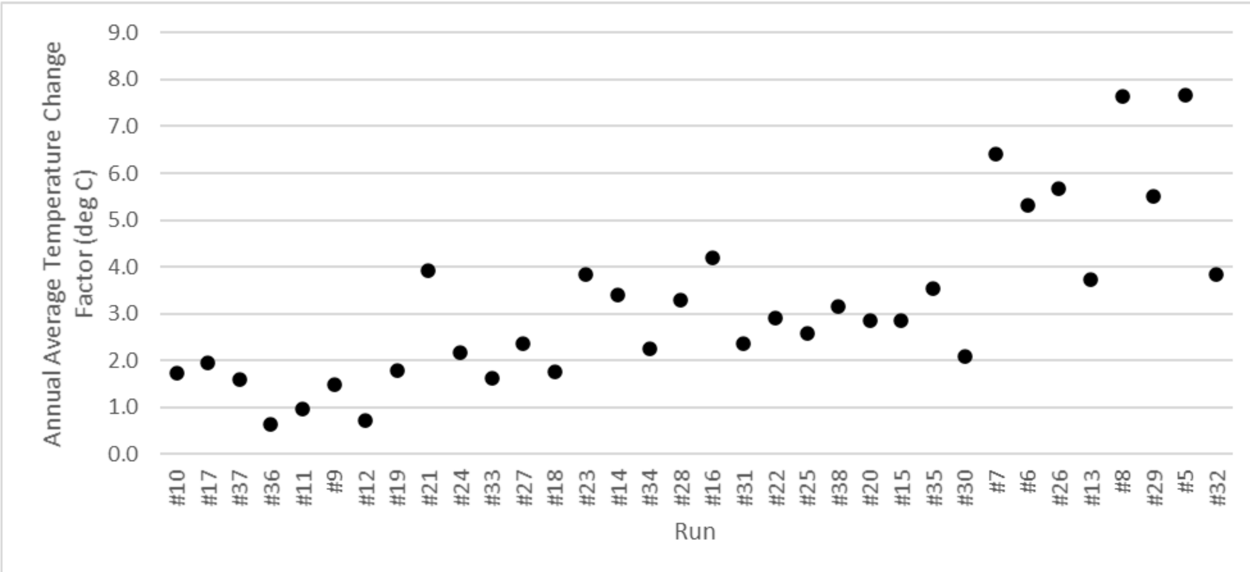


Figure S-2. Owasco Lake: Days Greater than 25°C (April–October)

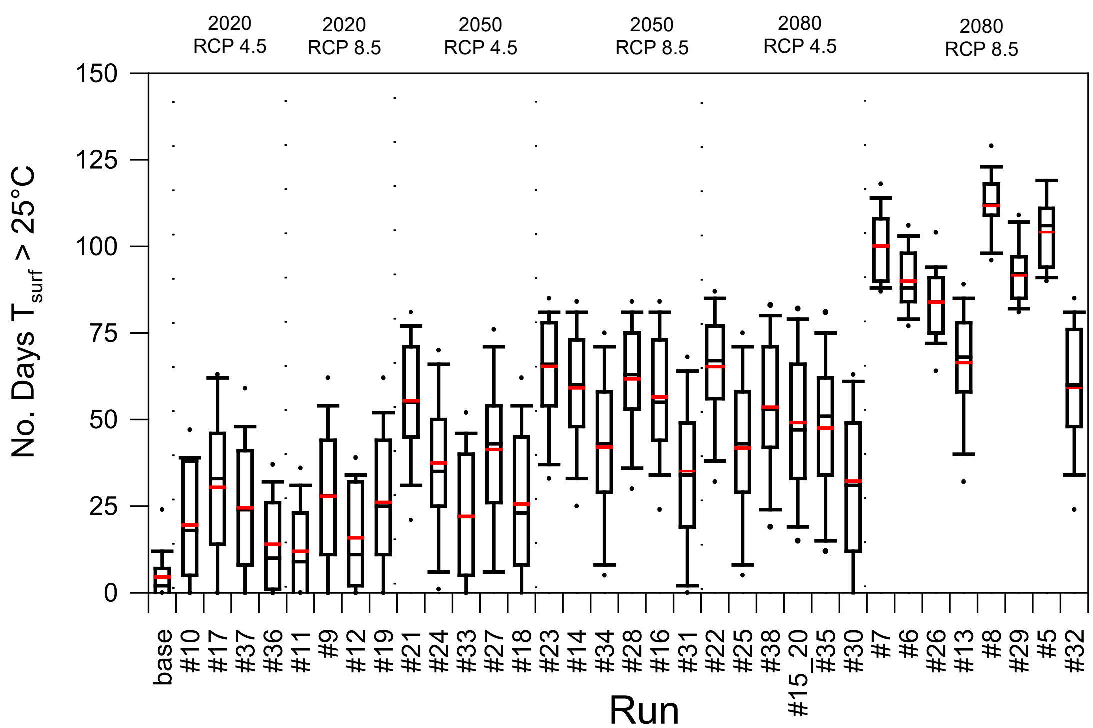


Figure S-3. Owasco Lake Bar Chart of Algae Types: 19 Yearly Averages for Modeled Scenarios 5–35 (July–Sept)

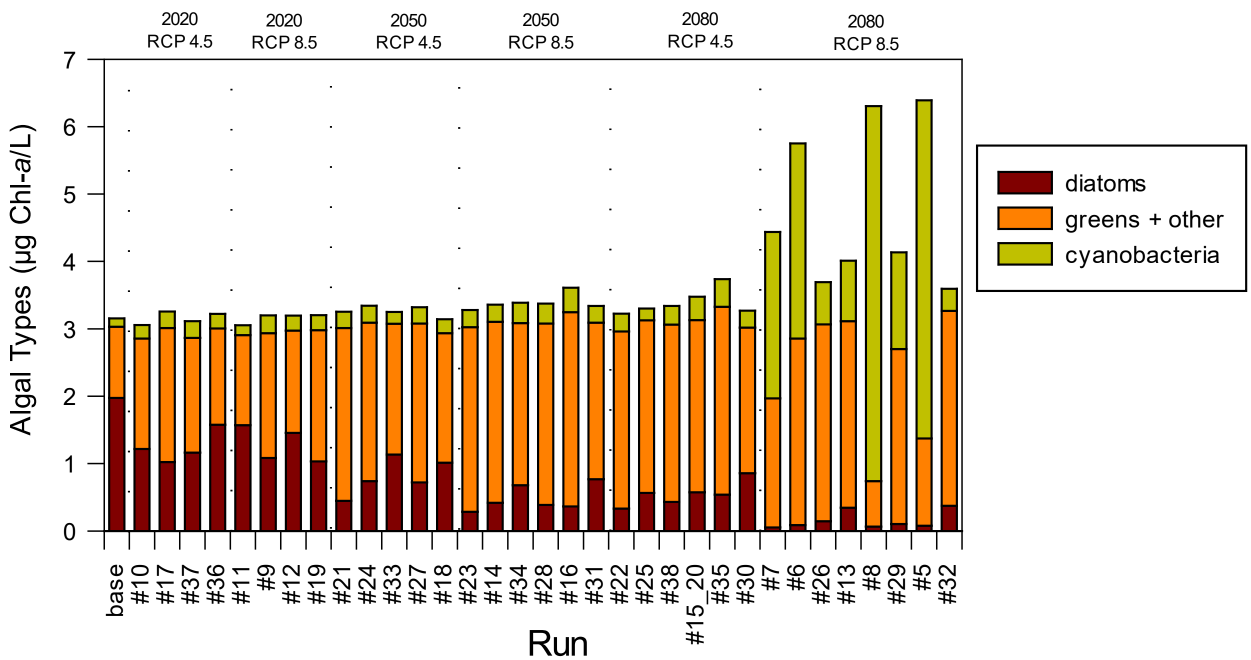


Figure S-4. Annual Average Precipitation Change for Each Modeled Scenario

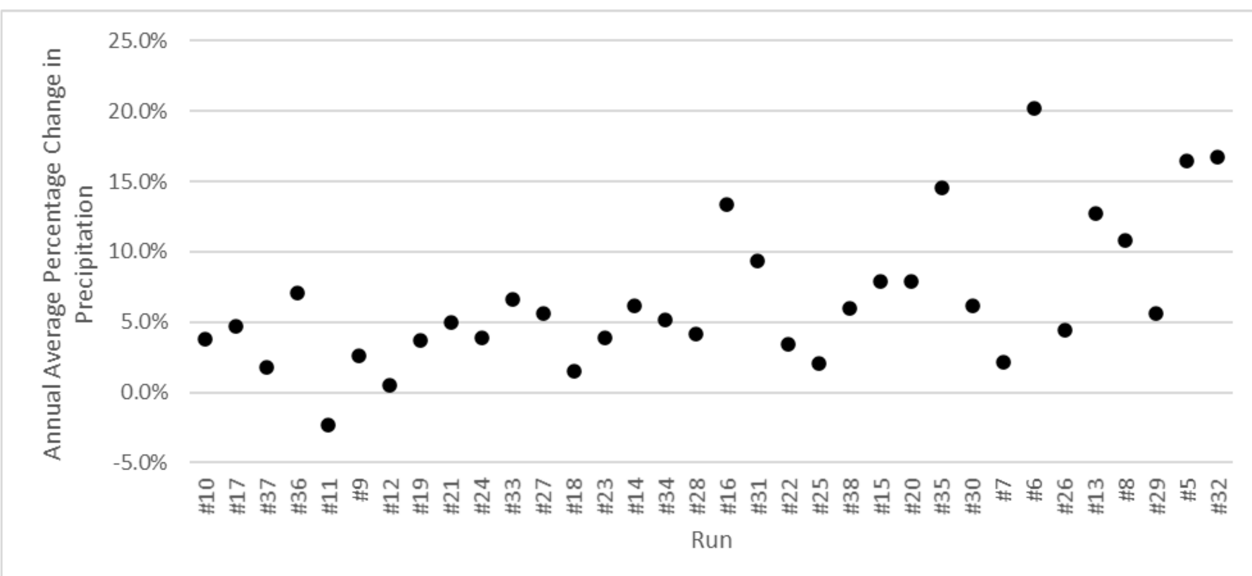


Figure S-5. Average Total Phosphorous at the Surface: Owasco Lake (April–Oct)

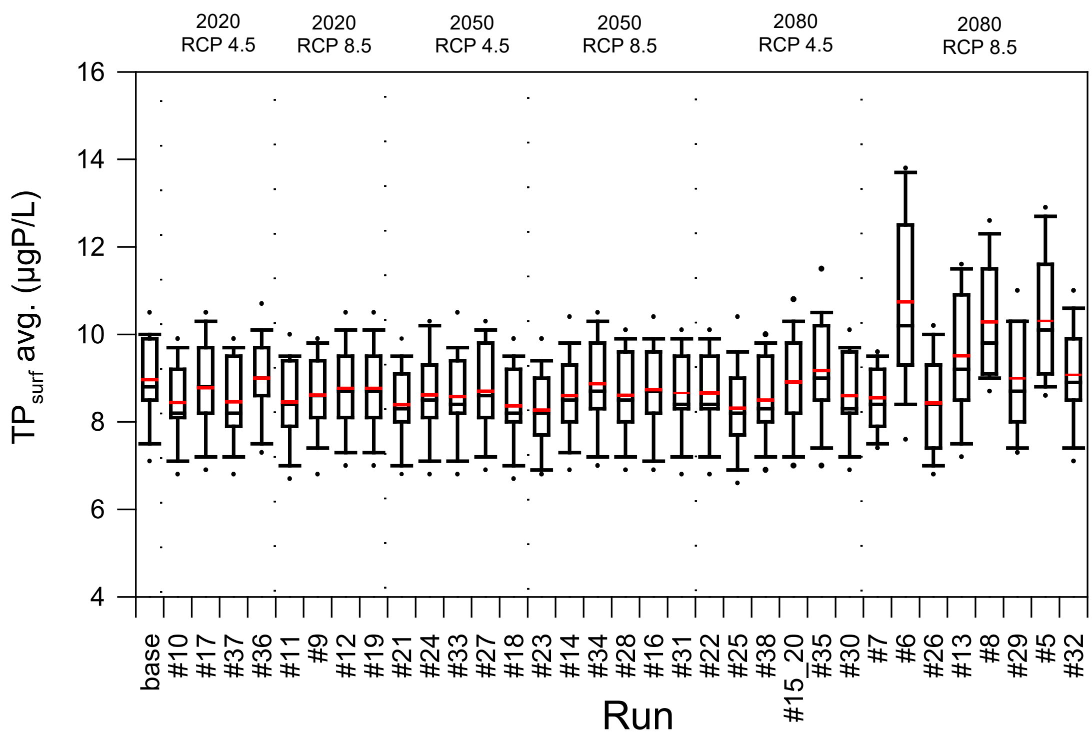
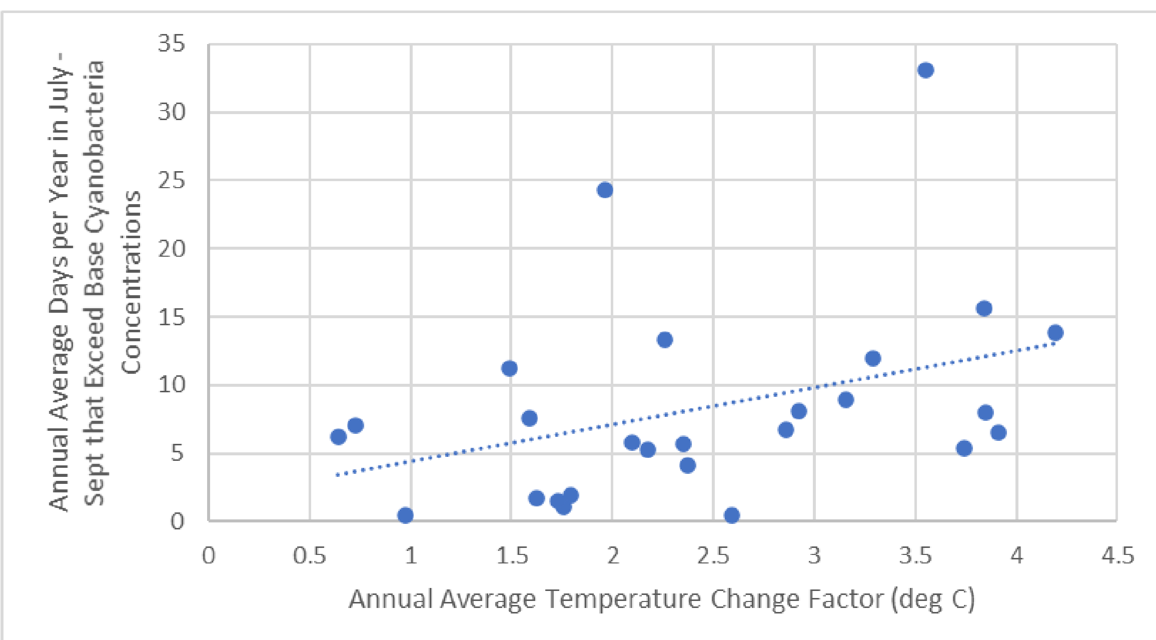


Figure S-6. Annual Average Days per Year with Cyanobacteria Concentrations above the Base Year for Each Climate Scenario



1 Introduction

New York State has an abundance of freshwater resources in the form of rivers, streams, and lakes. These resources are used for drinking water, recreation, fishing, tourism, agriculture, and manufacturing. Water quality is an important factor for the beneficial use of the State's water resources. A growing concern in the State and across the U.S. is the occurrence of cyanobacterial harmful algal blooms (CHABs), which are caused by excessive growth of cyanobacteria.[3] Certain cyanobacterial taxa are capable of producing hepatotoxins, neurotoxins and endotoxins, which can cause widespread economic loss associated with decreased recreational spending and increased water treatment costs (Dodds et al., 2009; Sanseverino et al., 2016). In general, the primary drivers of cyanobacterial blooms are elevated water temperatures and nutrient levels (Newcombe, 2012). Both of these factors are either directly or indirectly affected by climate change through warming temperatures and increased potential for extreme storms. CHABs may also be promoted by dreissenid (quagga, zebra) mussels (Knoll et al., 2008; Sarnelle et al., 2012) and wind-blown accumulation in shoreline areas (Chorus & Bartram, 1999). This project was designed to explore how these drivers may potentially affect the occurrence of cyanobacterial blooms in three large, oligo-mesotrophic Finger Lakes in New York State. The focus of this research was to use modeling tools and climate change projections to simulate how cyanobacterial blooms may change as the result of future climate conditions and to suggest adaptation options to improve resilience to blooms.

Harmful algal blooms are a pivotal issue for many recreational and source water systems. Since the highly publicized shutdown of the Toledo, OH water supply due to a cyanobacteria bloom in 2014 and the subsequent publication of federal health advisories, many locales have experienced and responded to cyanobacterial blooms with varying levels of success. A high-level review of available data provides little insight into the drivers of such events and in many cases is contradictory. While CHABs are not a new phenomenon, they have increased in severity and frequency over the past century and are projected to increase further as the impacts of climate change worsen (Carey et al., 2012; Paerl & Paul, 2012).

CHABs result in numerous impacts to the beneficial uses of lakes. Because of the production of toxins, lakes experiencing a bloom may be restricted for recreation, especially water contact recreation.[4] However, boating and fishing are also not recommended around blooms (NYS Department of Health, 2016). Toxins, tastes, and odor compounds, and biomass associated with cyanobacteria may cause problems for drinking water treatment. These problems may require treatment plant operational changes to maintain the safety and aesthetics of drinking water.

This project explored, through the use of water quality models, the possible impact future climate conditions may have on hydrodynamics and the likelihood of CHABs in three Central New York Finger Lakes (Cayuga Lake, Owasco Lake, and Skaneateles Lake). These lakes serve as public water supplies, venues for a wide range of recreational pursuits, and vital economic engines for the region. These ecosystem services are currently impacted by CHABs and understanding the potential of climate change on CHABs frequency and severity is important for management. The modeling approach was guided by the principle of parsimony. Accordingly, the watershed and in-lake models employed only the level of complexity necessary to simulate the key processes understood to impact lake hydrodynamics and the occurrence of CHABs.

2 Lake Ecology, Cyanobacteria, and Potential Impacts

Lake[5] ecology is driven by watershed characteristics, morphometry (shape, depth, size) and climate (hydrology and temperatures). These factors influence the amount of water flowing through the lake, its quality and internal dynamics, such as mixing, particle settling, etc., which in turn influence the habitats and biological populations. Large lakes are rarely homogenous with often times great variability between the near-shore, littoral zone, and the deep, open water sections. Winds, tributary inflows, storms, and temperature stratification drive internal mixing and distribution of nutrients and plankton through the water column.

The classical seasonal pattern for biological succession of a lake begins after spring turnover with the growth of diatoms, which prefer cooler water and have a competitive advantage in low-light conditions (Wetzel, 2001). As water temperatures and light availability increases through the spring and summer, the phytoplankton population shifts toward chlorophytes (green algae) and cyanophytes (cyanobacteria). As primary producers, these three populations (diatoms, green algae, and cyanobacteria) may be grazed upon by microzooplankton, small fish, crustaceans, and mollusks. The feeding preferences of these primary consumers can affect population dominance of the primary producers through the year. Diatoms and green algae are the preferred taxa for a lake ecosystem because they are nutrient-rich and an ideal primary producer in terms of food sources. Strong production of these species promotes flow of both nutrients and energy up the food chain.

Alternately, cyanobacteria can outcompete green algae under warm, quiescent conditions, and once cyanobacteria dominance is established, it becomes difficult to suppress their growth and shift dominance back to green algae. Cyanobacteria dominance is problematic because they have the potential to produce a myriad of toxins, other adverse irritants, and taste and odor compounds. Further, cyanobacteria adversely impact the food web because they are a poor-quality food resource that may be avoided by grazing species higher in the food chain. This furthers the ecosystem imbalance since nutrients and energy stay in the lowest portion of the food chain with minimal upward flow. Additionally, grazing provides limited control on the population density of cyanobacteria. This overgrowth of primary producers with minimal grazing results in degraded water quality.

2.1 Factors Affecting Cyanobacteria Dominance

Cyanobacteria are endemic to most freshwater lakes. When populations are small, they do not result in negative consequences. However, under some conditions, cyanobacteria can multiply quickly and become concentrated to a level that is detrimental to lake water quality. Referred to as cyanobacterial harmful algal blooms,[6] these events have become more frequent and are expected to increase under projected future climate change.[7]

Cyanobacteria dominance in phytoplankton assemblages is commonly observed in temperate freshwater systems and has been extensively documented[8]. While not every factor that influences the cyanobacteria life cycle is known, and cyanobacteria have been documented under conditions contrary to their preferred environment (e.g., cold water conditions, rivers, low nutrient lakes, etc.), high-water temperature, adequate to above average nutrients (notably phosphorous) and lake stratification are key risk factors for CHAB formation (Figure 1).

Figure 1. Factors Affecting Cyanobacterial Harmful Algal Blooms Development

Adapted from Table 3 of Hazen Harmful Algal Bloom White Paper, Summer 2015.

Cyanobacteria Potential	Water Temperature (°C)	Total Phosphorus µg/L	Occurrence of Thermal Stratification
Very Low	< 15	< 10	Rare or Never
Low	15 - 20	< 10	Infrequent
Moderate	20 - 25	10 - 25	Occasional
High	> 25	25 - 100	Frequent and persistent
Very High	> 25	> 200	Frequent and persistent / strong

The optimal growth temperature for cyanobacteria is approximately 25°C, which is higher than some of their algal counterparts. Higher temperatures also influence lake stratification,[9] which also promotes cyanobacteria growth and dominance.[10] Stratification typically creates a vertical nutrient gradient with limited nutrients available in the epilimnion (upper layer). Cyanobacteria can regulate their buoyancy and are thereby able to acquire nutrients from lower in the water column during the diurnal cycle without relying on advection[11]. Cyanobacteria can, therefore, take advantage of nutrients in the lower depths, while also benefitting from high light levels in the photic zone. The longer stratification persists, the more important nutrient-related competition between organisms becomes. Thus, the ability of cyanobacteria to regulate buoyancy, moving up and down the water column, provides a competitive advantage over other primary producers.

While these conditions are the most conducive for bloom formation, there are a myriad of factors that influence whether a bloom occurs in any given year. Research is ongoing to continue to identify other specific factors to support management efforts.

2.2 Cyanobacteria and Invasive *Dreissenid* Mussels

Zebra mussels (Dreissena polymorpha) and quagga mussels (Dreissena bugensis), collectively referred to as dreissenid mussels, are invasive aquatic organisms that were introduced to the U.S. in the late 1980s and proceeded to spread throughout the northeast and Midwest. In New York State, the introduction of zebra mussels began in the 1990s and was followed by a secondary introduction by quagga mussels in the 2000s (Hetherington et al., 2019; Watkins et al., 2012). Currently in the State there are established populations of zebra mussels in 38 counties and established populations of quagga mussels in 16 counties (USGS, 2020a; USGS, 2020b). As dreissenids became established, there were observed shifts in lake ecology that impacted primary producers (Fishman et al., 2009). Dreissenids are thought to promote blooms of cyanobacteria through a variety of mechanisms, including: (1) removal of particles resulting in clearer water, (2) increased recycling of phosphorus, and (3) selective rejection of certain strains of cyanobacteria (Fishman et al., 2009; Vanderploeg et al., 2001). Dreissenid mussels have been found to be an important determinant of CHABs in New York State lakes, particularly in lakes with lower total phosphorous (TP) and chlorophyll-a (Chl-a) concentrations (Matthews et al., in prep).

3 Climate Change Projections

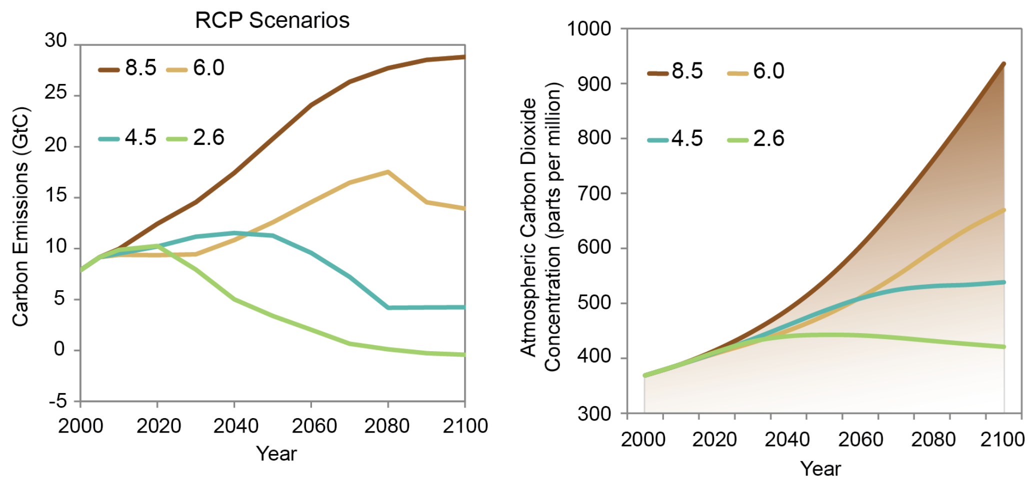
The principal force behind all of Earth's climate is the sun's radiation and the energy balance between how much energy is absorbed by the Earth versus reflected back into space. Increasing concentrations of heat-trapping gases in the atmosphere, referred to as greenhouse gases (GHG), has resulted in radiative forcing that is rapidly increasing the Earth's energy balance and leading to climate change (North, Pyle, & Zhang, 2014). Radiative forcing, measured as watts per square meter (W/m^2), has been increasing proportionally with GHG concentrations. As more energy is absorbed by the Earth, global temperatures increase, which drives other changes to climate. Currently, the Earth's energy balance is at a radiative forcing of approximately 3.0 $W/m.^2$

To understand and predict how changes in radiative forcing will affect the climate, General Circulation Models (GCMs) are used to simulate the Earth's atmosphere and oceans, and the resulting weather patterns. Many research institutions around the globe have developed GCMs for analyzing global climate. These organizations frequently collaborate and compare data and modeling tools. In order to facilitate collaboration and comparison of models and results, the climate modeling community developed a set of future scenarios to serve as a common basis for climate change analyses. Referred to as Representative Concentration Pathways (RCP), these scenarios span the range of radiative forcing values through the year 2100, discussed in the peer-reviewed literature (e.g., 2.6 to 8.5 W/m^2) and are correlated with global GHG concentrations (Figure 2) [RCP Database, 2009; Van Vuuren et al., 2011].

- RCP 2.6 (i.e., radiative forcing of 2.6 W/m^2) corresponds to stabilization of emissions in the beginning of the century as proposed by the 2015 Paris Agreement. Note that this scenario requires a reduction below current levels of observed radiative forcing.
- RCP 4.5 emissions peak from 2040 to 2050 then stabilize and decline resulting in stabilization of radiative forcing before the end of the century.
- RCP 6.0 emission peak and decline around 2080 resulting in stabilization of radiative forcing after 2100.
- RCP 8.5 corresponds to growing emissions through the end of the century and a peak radiative forcing of 8.5 W/m^2 after 2100.

Figure 2. Trends in Global GHG Emissions and Concentrations for Four RCPs

Source: Melillo, Richmond, and Yohe, 2014

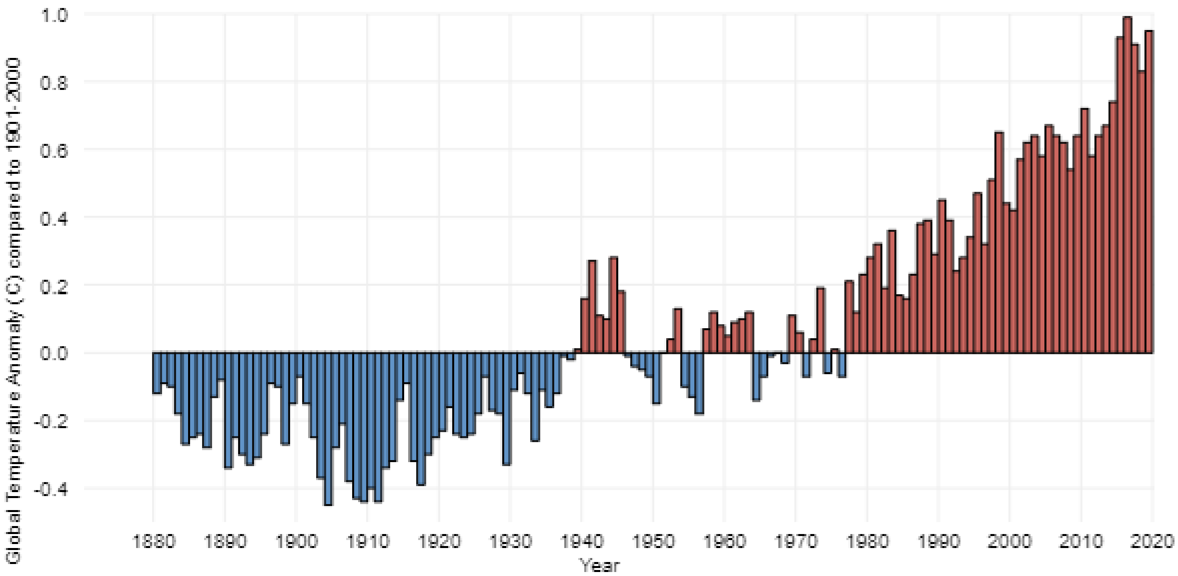


There is no assumption of likelihood in these scenarios because they are based on global decisions regarding GHG emissions.

The New York State Energy Research and Development Authority (NYSERDA), as part of its ClimAID Climate Risk Information initiative, commissioned the development of seven regional downscaled climate change projections for the State based on GCM simulations from the Coupled Model Intercomparison Project Phase 5 (CMIP5) developed for the United Nations Intergovernmental Panel on Climate Change (IPCC) Fifth Assessment Report (Horton, Bader, Rosenzweig, DeGaetano, and Solecki, 2014). The projections include daily temperature and precipitation values for 35 individual GCMs for two representative concentration pathways (RCP 4.5 and RCP 8.5) and three time slices (2020s, 2050s, and 2080s). The regional data also include the historical basis for comparison.

There has been a consistent increasing trend in observed global temperatures for the last few decades (Figure 3), which is projected to continue based on current trends in GHG emissions and GCM modeling. These temperature changes have the potential to directly influence the factors that contribute to more frequent CHABs.

Figure 3. Average Annual Temperature Difference Compared to the 20th Century Average (NOAA, 2020)



Increasing ambient temperatures and more frequent heat waves have the potential to result in warmer water temperatures. Longer periods of warmer weather also have the potential to cause higher relative thermal resistance with a resulting longer periods of lake stratification. These factors are expected to lead to increasing cyanobacteria growth rates and promote cyanobacteria dominance.

Rising temperatures will also influence the hydrologic cycle, which drives local rainfall and corresponding nutrient loadings to lakes. In general, higher temperatures are expected to intensify the earth's water cycle and cause more frequent storm events due to both the increased rates of evaporation and because higher air temperatures can hold more water vapor in the atmosphere (Kunkel, et al., 2013).

4 Approach and Model Development

The overall approach of this research is to quantify potential changes to lake conditions that influence the occurrence of CHABs by using models to compare historical and projected climate drivers (e.g., changing temperature and rainfall).

4.1 Study Systems

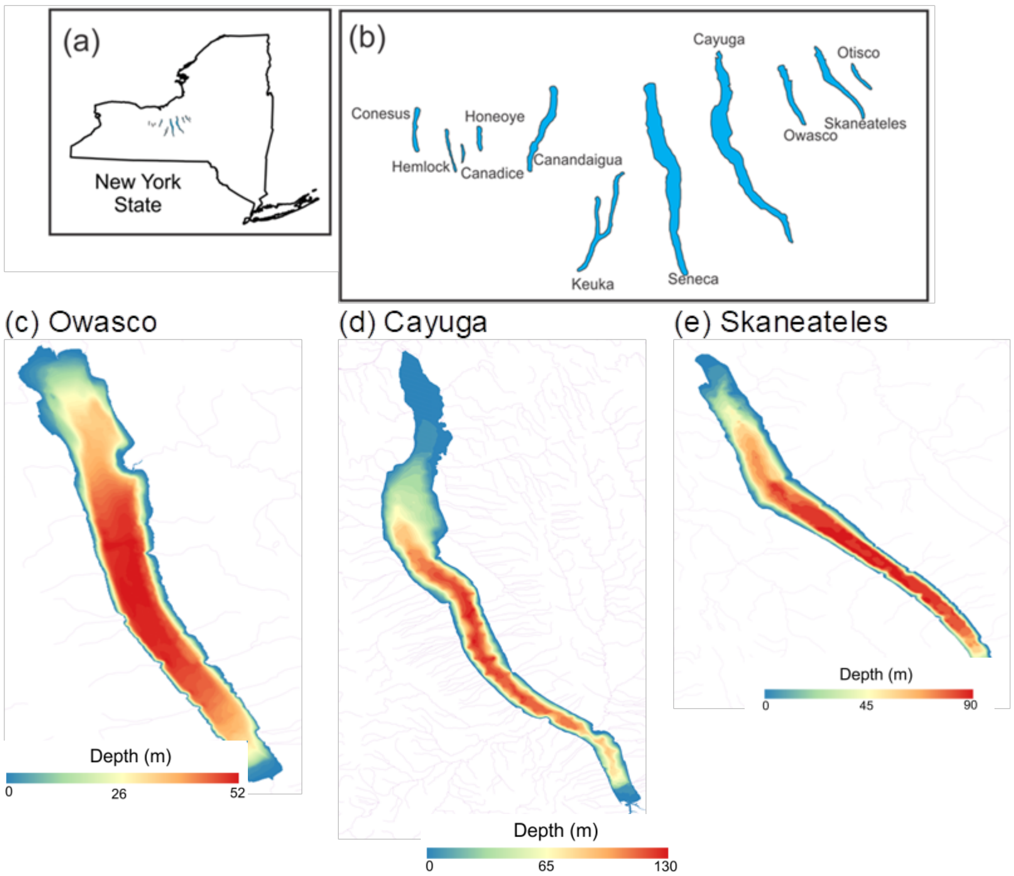
The Finger Lakes of central New York State consist of 11, elongated, north-south oriented lakes (Figure 4a, b). These lakes originated as pre-glacial stream valleys, which were subsequently enlarged and deepened by a combination of ice and sub-glacial meltwater erosion during the Pleistocene (Mullins & Hinchey, 1989 ; Mullins et al., 1996). The modern Finger Lakes were last structured during late Wisconsinan by a surge of the Laurentide ice sheet (Lajewski et al., 2003). European settlement of these watersheds occurred in the late 1700s and early 1800s. The Finger Lakes were the focus of some of the earliest limnological investigations (Birge & Juday, 1914; Birge & Juday, 1921) in the United States. The Finger Lakes serve as vital drinking water sources for the region, provide multiuse recreational opportunities, and are the focus of the substantial Finger Lakes tourism industry. Reports of CHABs have increased across the Finger Lakes in recent years, threatening drinking water supplies and recreational uses. CHABs were documented in all 11 Finger Lakes during 2017. Hydrodynamic and water quality modeling of three selected Finger Lakes (Owasco, Cayuga, Skaneateles) provides an opportunity to better understand the potential impacts of climate change, invasive dreissenid mussels, and hydrodynamics on CHABs.

There are a variety of potential causes of CHABs, including excessive loading of the nutrients phosphorus and nitrogen that supports excessive growth of cyanobacteria (Beaulieu et al., 2013; Beaver et al., 2018; Dolman et al., 2012), low ratios of nitrogen to phosphorus (Graham et al., 2004; Orihel et al., 2012), higher water temperatures (Beaulieu et al., 2013; Beaver et al., 2014), and effects of invasive dreissenid mussels (Knoll et al., 2008; Sarnelle et al., 2012). The specific causes of CHABs in Owasco Lake, Cayuga Lake, and Skaneateles Lake remain unknown at this time. It is noteworthy that these lakes are not nutrient-rich or shallow, conditions typically associated with CHABs. Reports of CHABs in lakes with low to moderate nutrient concentrations are increasing (Carey et al., 2008; Sarnelle et al., 2012; Sorichetti et al., 2014; Winter et al., 2011), underscoring the myriad physical, chemical, and biological factors that contribute to cyanobacterial dominance (Heisler et al., 2008; Paerl & Otten, 2013). Owasco, Cayuga, and Skaneateles Lakes were included in the 12 priority lakes for which HAB Action Plans were developed by New York State Department of Environmental Conservation (NYSDEC) to identify the factors contributing to HABs and guide strategies to reduce blooms (NYSDEC et al., 2018a, b, c).

Owasco Lake, a mesotrophic lake located in Cayuga County south of the city of Auburn, is the sixth largest and third easternmost of the Finger Lakes (Figure 4c). The lake is 18 kilometers (km) long and relatively narrow (maximum width of 2 km) with a maximum depth of 54 m, an average depth of 29 m, a volume of 780x106 m3, and a surface area of 26.7 km2. The lake is a source of drinking water for the City of Auburn and Town of Owasco, and is a valued recreational resource used for fishing, boating, and swimming. Approximately 50% of the watershed is classified as pasture, hay, and cultivated crops. Forested and developed lands are estimated to account for 29% and 5% of the watershed, respectively. In 2016, Owasco Lake was included in the New York State Section 303(d) List of Impaired Waters due to frequent CHABs. The lake is not listed for CHABs in the latest 2020–2022 Draft 303(d) List.[12] From 2013 through 2017 there were 84 confirmed CHABs in Owasco Lake, including 55 confirmed CHABs with high cyanotoxin levels (NYSDEC et al., 2018a). CHABs pose a threat to drinking water supplies, aesthetics, and contact recreation, as highlighted by the loss of 61 beach days between 2014 and 2017 (NYSDEC et al., 2018a). CHABs continued to be documented in Owasco Lake during the summer and fall of 2018, 2019, and 2020.

Figure 4. Maps Depicting Locations and Depths of the Finger Lakes

(a) The location of the Finger Lakes within New York State, (b) the relative positions of the 11 Finger Lakes, (c) bathymetric map of Owasco Lake, (d) bathymetric map of Cayuga Lake, and (e) bathymetric map of Skaneateles Lake. Bathymetric maps are not to scale.



Cayuga Lake, a mesotrophic lake located in Cayuga, Seneca, and Tompkins County north of the city of Ithaca, is the second largest and fourth easternmost of the Finger Lakes (Figure 4d). The lake is 61.5 km long and relatively narrow (mean width of 2.8 km) with a maximum depth of 133 m, an average depth of 55 m, a volume of 950 x 10^7 $m,^3$ and a surface area of 173 km^2 (NYSDEC et al., 2018b). Cayuga Lake serves as the primary drinking water source and/or backup source for nearly 100,000 watershed residents (NYSDEC et al., 2018b), including the Village of Seneca Falls, the Towns of Dryden, Ithaca, and Lansing and the Villages of Cayuga Heights and Lansing, and Wells College. Cayuga Lake is also a valued recreational resource used for fishing, boating, and swimming. Approximately 50 percent of the watershed is classified as pasture, hay, and cultivated crops. Natural areas and developed lands are estimated to account for 35 percent and 6 percent of the watershed, respectively. From 2002 to 2020, Cayuga Lake was included in the State Section 303(d) List of Impaired Waters as impaired by phosphorus and silt/sediment. This listing was removed in the latest 2020–2022

Draft 303(d) List. A Total Maximum Daily Load (TMDL) for phosphorus in Cayuga Lake is currently under development. There were 12 confirmed CHABs occurrences in the lake from 2013 through 2017, including three confirmed CHABs with high toxins (NYSDEC et al., 2018b). Large portions of the lake shoreline were in bloom during late summer in 2017, resulting in closures of six beaches for a total of 62 beach days. CHABs continued to be documented in Cayuga Lake during the summer and fall of 2018, 2019, and 2020.

Skaneateles Lake, an oligotrophic lake located in Onondaga, Cayuga, and Cortland Counties south of the Village of Skaneateles, is the fifth largest and second easternmost of the Finger Lakes (Figure 4e). The lake is 26 km long and relatively narrow (maximum width of 1.5 km) with a maximum depth of 96 m, an average depth of 44 m, a volume of 156 x 10^7 m,^{3} and a surface area of 35.6 km.^{2} Skaneateles Lake provides drinking water to the City of Syracuse[13] and portions of the Towns of DeWitt, Onondaga, Geddes, Camillus, Salina, and Skaneateles, and to the Villages of Skaneateles, Jordan, and Elbridge (NYSDEC et al., 2018c). The lake is a valued recreational resource used for fishing, boating, and swimming. Approximately 36 percent of the watershed is classified as pasture, hay, and cultivated crops. Natural and developed lands are estimated to account for 40 percent and 5 percent of the watershed, respectively. A widespread/lakewide CHAB was documented in Skaneateles Lake during September and early October of 2017 (NYSDEC et al., 2018c). Less widespread CHABs, mostly confined to shoreline areas, were documented in 2018, 2019, and 2020.

4.2 Approach and Model Development

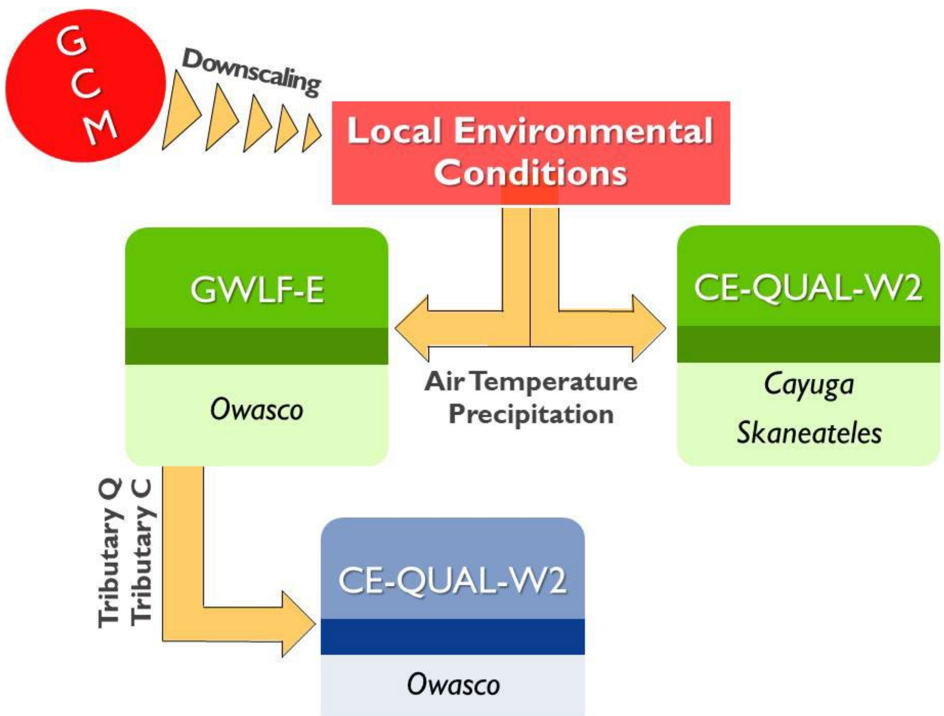
4.2.1 Analytical Approach

The primary objective of this project was to advance our understanding of CHABs through investigation of the potential effects of future climate change, invasive mussels, and hydrodynamics on the frequency and severity of CHABs using existing mathematical models developed for the study lakes. Climate forecasts were obtained from 14 global circulation models (GCMs) using two representative carbon pathways (RCP 4.5, 8.5) and two time slices (2050, 2080). The combination of an RCP with a downscaled GCM for a particular time slice is considered a "future climate scenario," which can be used to explore possible impacts on a selected region. Four of the GCMs (ACCESS1.0, FGOALS-g2, IPSL-CM5A-LR, MRI-CGM3) were used to generate climate forecasts for each of the four RCP-time slice combinations. Additionally, contemporary meteorological data were used the generate baseline model simulations that represent current (e.g., 2000–2018) conditions.

The general approach adopted for this project was to use local meteorological data (e.g., air temperature, precipitation) downscaled from multiple GCMs to drive the two-dimensional in-lake simulation model CE-QUAL-W2 [Figure 5]). For each climate change scenario, a watershed model, GWLF-E (www.modelmywatershed.org), was used to estimate daily flows and concentrations for Owasco Lake tributaries. These inputs were required to establish hydrologic and material loads that support water quality modeling of Owasco Lake. Detailed loads were not required because only hydrodynamic processes (i.e., water motion, temperature) were considered for Cayuga Lake and Skaneateles Lake.

Figure 5. Schematic Representation of the Modeling Process for Owasco, Cayuga, and Skaneateles Lakes

This approach supports hydrodynamic modeling for all three lakes and water quality modeling for Owasco Lake.



4.2.2 Climate Projection Selection

The climate projections were developed using monthly change factors based on the GCMs for each RCP and time slice. For this study, the change factors were then applied to the historical data to create the representative future daily temperature and precipitation projections for the specific locations evaluated. All of the lakes included in this study were in the ClimAID region 1. The monthly change factors were applied to location-specific data for each lake to develop projections used for the modeling. The monthly change factors are additive for temperature and multiplicative for precipitation per the example formulas below.

$$Adjusted\ Jan\ 1st\ PRCP = (Historical\ Jan\ 1st\ PRCP) * (January\ PRCP\ Change\ Factor)$$

$$Adjusted\ Jan\ 1st\ TEMP = (Historical\ Jan\ 1st\ TEMP) + (January\ TEMP\ Change\ Factor)$$

It was determined that it would not be possible to simulate all 210 ClimAID scenarios. Therefore, three subsets of the available climate projections were developed to assess a range of potential future conditions. The basis for the three subsets were:

Seasonal Percentiles The temperature and precipitation change factors were reviewed for the 210 scenarios to select 16 scenarios based on percentile (approximately 10th, 25th, 75th, 90th) aggregated by season (Dec–Feb, Mar–May, Jun–Aug, Sep–Nov). The objective of this subset was to get a sampling of the range of potential change by season and change factor. Figure 6 presents one seasonal distribution of change factors with dividing lines for the percentiles.

Model Series Because the selection of scenarios based on seasonal percentiles was neutral with respect to time slices and RCP, it was determined a number of series of simulations would be helpful to evaluate differences in time slice and RCP. Scenarios were selected for 2050 and 2080 time slices for both RCPs for five GCM models for comparison.

Intra-annual Variability Scenarios were also evaluated for intra-annual variability of the change factors. For example, scenarios with consistently high-change factors for each month would have low intra-annual variability, while change factors that vary from high to low have high intra-annual variability. This factor was selected to evaluate the effect of consistency of annual change on modelling results.

Table 1 through Table 3 below presents the model details for each selected scenario. While only the seasonal or annual change factor is presented, the 12 monthly change factors were used to adjust historic temperature and precipitation values for the modeling.

Figure 6. Change Factors for Temperature and Precipitation Plotted Together Showing the Ranges of the Percentiles

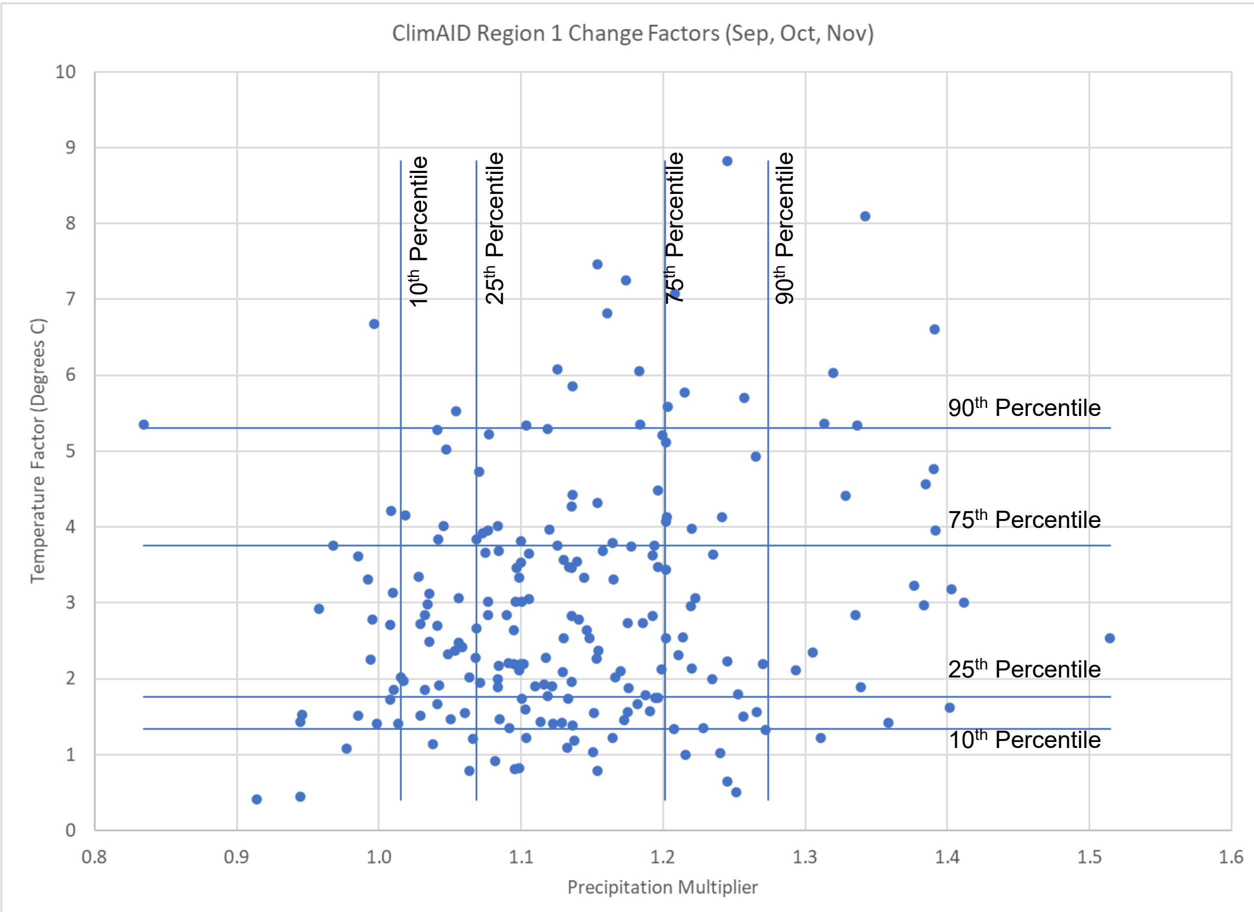


Table 1. Climate Model Details for Selected Scenarios Based on Percentile Change

Minimum Combined Temperature and Precipitation by Season					
Model Scenario	**Model Name**	**Season**	**Percentile**	**Precipitation Factor**	**Temp Factor (℃)**
GCM 8.5, 2020	IPSL_CM5A_LR	Mar-Apr-May	10	0.89	0.63
GCM 4.5, 2020	GISS_E2_H	Jun-Jul-Aug	14	0.84	1.15
GCM 8.5, 2020	inmcm4	Sep-Oct-Nov	10	0.76	0.41
GCM 8.5, 2020	MRI_CGCM3	Dec-Jan-Feb	4	0.87	0.30
25th Percentile Combined Temperature and Precipitation by Season					
Model Scenario	**Model Name**	**Season**	**Percentile**	**Precipitation Factor**	**Temp Factor (℃)**
GCM 8.5, 2080	GFDL_ESM2M	Mar-Apr-May	75 ± 3	1.29	3.66
GCM 8.5, 2050	FGOALS_g2	Jun-Jul-Aug	75 ± 4	1.17	3.54
GCM 4.5, 2080	IPSL_CM5A_MR	Sep-Oct-Nov	75 ± 4	1.19	3.76
GCM 8.5, 2050	MIROC5	Dec-Jan-Feb	75 ± 3	1.33	4.37
75th Percentile Combined Temperature and Precipitation by Season					
Model Scenario	**Model Name**	**Season**	**Percentile**	**Precipitation Factor**	**Temp Factor (℃)**
GCM 4.5, 2020	HadGEM2_CC	Mar-Apr-May	25 ± 4.5	1.14	1.41
GCM 4.5, 2050	MRI_CGCM3	Jun-Jul-Aug	25 ± 3	1.02	1.73
GCM 8.5, 2020	NorESM1_ME	Sep-Oct-Nov	25 ± 6	1.08	1.89
GCM 4.5, 2080	IPSL_CM5A_MR	Dec-Jan-Feb	25 ± 3	1.15	2.02
Maximum Combined Temperature and Precipitation by Season					
Model Scenario	**Model Name**	**Season**	**Percentile**	**Precipitation Factor**	**Temp Factor (℃)**
GCM 8.5, 2080	MIROC_ESM_CHEM	Mar-Apr-May	99	1.62	7.95
GCM 8.5, 2080	CSIRO_Mk3_6_0	Jun-Jul-Aug	94	1.43	6.08
GCM 8.5, 2080	ACCESS1_0	Sep-Oct-Nov	96	1.39	6.61
GCM 8.5, 2080	HadGEM2_ES	Dec-Jan-Feb	97.5	1.61	7.58

Table 2. Climate Model Details for Selected Scenarios Based on Completed Series

Model Scenario	Model Name	Annual Precipitation Factor	Annual Temperature Factor (℃)
RCP45, 2050	ACCESS1_0	1.050	3.91
RCP45, 2080	ACCESS1_0	1.034	2.92
RCP85, 2050	ACCESS1_0	1.039	3.84
RCP45, 2050	FGOALS_g2	1.039	2.18
RCP45, 2080	FGOALS_g2	1.021	2.59
RCP85, 2080	FGOALS_g2	1.044	5.68
RCP45, 2050	IPSL_CM5A_LR	1.056	2.37
RCP85, 2050	IPSL_CM5A_LR	1.041	3.29
RCP85, 2080	IPSL_CM5A_LR	1.056	5.51
RCP45, 2080	MRI_CGCM3	1.061	2.10
RCP85, 2050	MRI_CGCM3	1.093	2.35
RCP85, 2080	MRI_CGCM3	1.167	3.84
RCP45, 2020	MRI_CGCM3	1.071	0.64
RCP45, 2020	IPSL_CM5A_LR	1.018	1.59
RCP45, 2080	IPSL_CM5A_LR	1.060	3.15

Table 3. Climate Model Details for Selected Scenarios Based on Intra-Annual Variability

Model Scenario	Model Name	Annual Precipitation Factor	Annual Temperature Factor (℃)
RCP45, 2050	GFDL_ESM2M	1.066	1.63
RCP85, 2050	GFDL_ESM2M	1.051	2.26
RCP45, 2080	MIROC5	1.145	3.55

4.2.3 Mathematical Models of Lakes Owasco, Cayuga, and Skaneateles

This project takes advantage of previously tested mathematical simulation models of Owasco, Cayuga, and Skaneateles Lakes to investigate potential water quality changes associated with future climate change. Models previously developed to support a TMDL analysis for Cayuga Lake—Upstate Freshwater Institute (UFI), 2017, a Nine Element (9E) Plan for Owasco Lake (UFI, 2020), and spill response analysis for Skaneateles Lake (Driscoll et al., 2006)—were applied to simulate lake conditions for a wide range of climate change scenarios. Additional details regarding calibration and testing of CE-QUAL-W2 for the study lakes, including performance criteria, are available in the references cited above. For the current project, these models were applied to advance our understanding of CHABs and to investigate the potential effects of climate change and invasive mussels on the frequency and severity of CHABs in Cayuga Lake, Owasco Lake, and Skaneateles Lake.

The widely used two-dimensional, laterally averaged hydrothermal/water quality model CE-QUAL-W2 was set up, calibrated, and tested for the three study lakes. CE-QUAL-W2 is currently maintained by Portland State University (Cole & Wells, 2018; http://www.ce.pdx.edu/w2/). QE-QUAL-W2 is a public access model that has been successfully applied to hundreds of lakes, rivers, and reservoirs (Zhang et al., 2015; Zhu et al., 2017) and used in support of TMDL analysis and 9E Plans (Interstate Commission on the Potomac River Basin, 2012; JM Water Quality, LLC, 2008; LimnoTech, 2016; von Stackelberg et al., 2016). CE-QUAL-W2's hydrothermal sub-model, which simulates water motion, temperature and stratification, was applied to all three lakes. The model is capable of representing various transport processes, including advective transport of buoyant cyanobacteria caused by wind. The two-dimensional framework is particularly well-suited for simulating longitudinal patterns in long, narrow lakes such as Owasco, Cayuga, and Skaneateles (Figure 7). The water quality sub-model was applied only to Owasco Lake. Multiple forms of phosphorus, nitrogen and organic matter are simulated in the water quality sub-model, including particulate and dissolved fractions, which are partitioned according to labile and refractory components. Phytoplankton biomass is simulated for three algal groups: diatoms, green algae, and cyanobacteria. In CE-QUAL-W2, the growth rate of each functional group was adjusted using a temperature-dependent growth rate multiplier (fractional value of 0 to 1.0). The shape of the multiplier as a function of temperature was determined by four calibration coefficients for each phytoplankton group and guided by literature values.[14] The temperature multiplier used for cyanobacteria was 0.1 at 18°C and increased to 1.0 (the maximum growth rate) at 26°C.

For this project, CE-QUAL-W2 was modified by the addition of a sub-model to address the impacts of invasive dreissenid mussels (quagga and zebra mussels) on nutrient cycling, algal biomass, and composition of the phytoplankton community. Although mussel growth and mortality were not modeled, impacts associated with filtering of particulate constituents (e.g., particulate organic matter, algae) and excretion of dissolved constituents (e.g., soluble reactive phosphorus, ammonium) was accounted for in the model (Figure 8). A similar approach was adopted previously for Cayuga Lake (UFI, 2017) and also by Boegman et al. (2008), Descy et al. (2003), and Zhang et al. (2008).

Figure 7. Lake Model Segments with the Primary Segment Used in Modeling Analysis

Shaded in blue for (a) Owasco Lake, (b) Cayuga Lake, and (c) Skaneateles Lake. Note that lakes are not shown to scale.

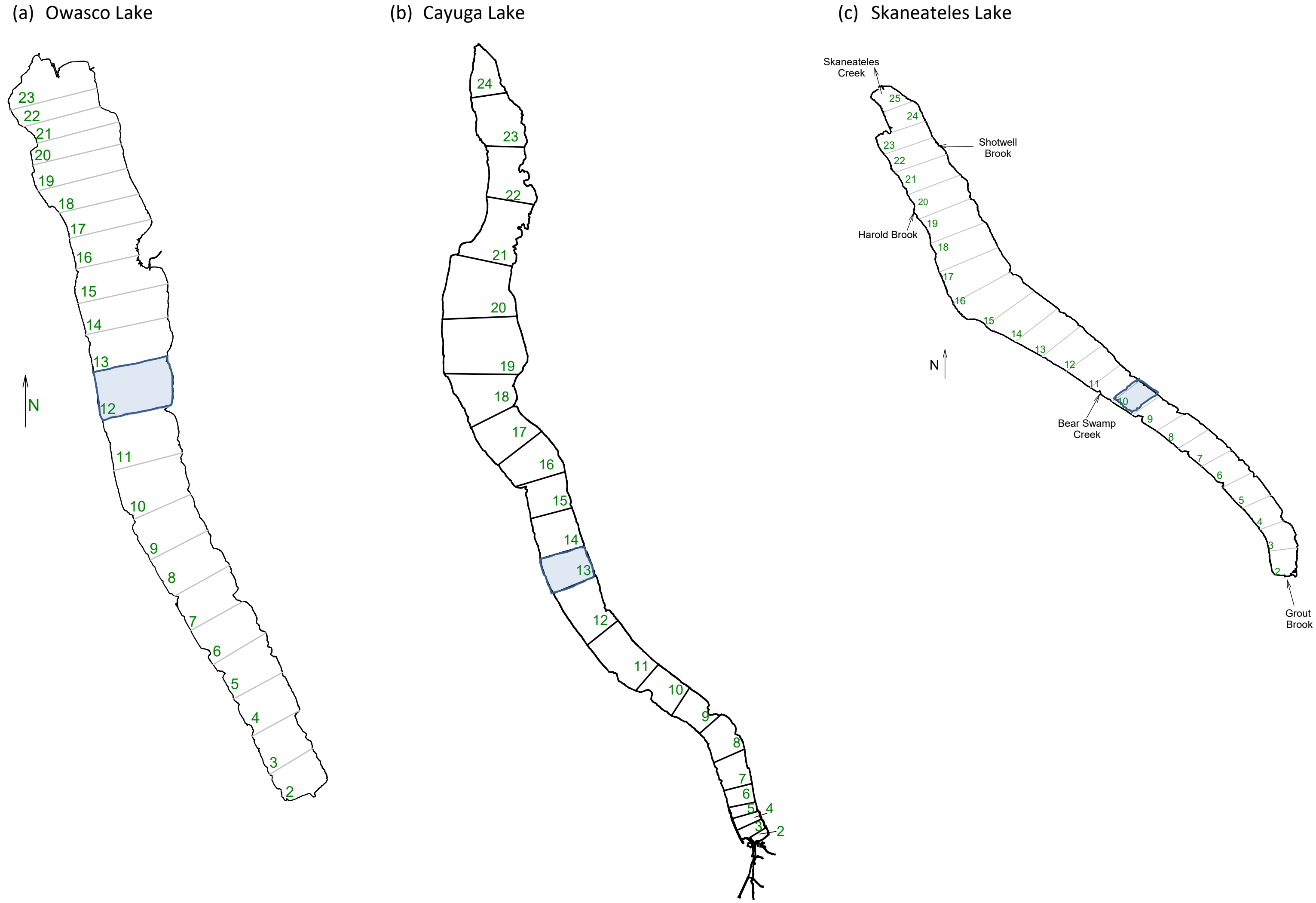
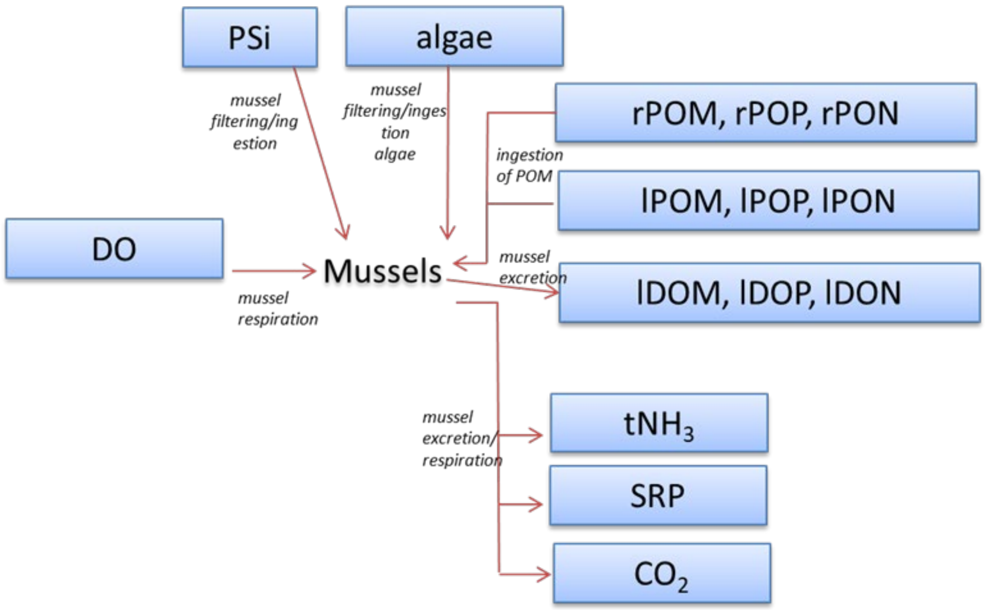


Figure 8. Conceptual Diagram Summarizing the Dreissenid Mussel Sub-Model Added to CE-QUAL-W2



4.2.4 Application of the Watershed Model: GWLF-E

The GWLF-E model is a combined distributed/lumped parameter watershed model, allowing for multiple land use/land cover scenarios. GWLF-E is integrated with Model My Watershed, an open-source online application maintained by the Stroud Water Research Center (www.modelmywatershed.org) that allows users to define an area of interest and model water quality and hydrology. GWLF-E has been successfully applied in a variety of climate change studies (Markensten et al., 2010; Samal et al., 2012, 2013; Tu, 2009). The online tool uses available geospatial information system (GIS) layers to derive inputs for the area of interest and local meteorological data for 1961 through 1990 to model water quality. A model input file for the Owasco Lake watershed was exported from the Model My Watershed online application and updated using temperature and precipitation data for the 2000–2018 period. Other model drivers were updated using the best available local information or using recommended values from the GWLF user manuals (Haith, 1992; Evans, 2002). After all adjustments were made, the modeled flows compared well to the flow budget developed for Owasco Lake (Nash Sutcliffe Efficiency = 0.71). The model was calibrated to measured flows and concentrations of total phosphorus, total nitrogen (TN), and total suspended solids (see UFI, 2020 for additional details).

To update the material loads that drive the water quality model, GWLF-E driver files were updated with temperature and precipitation data for each individual future climate scenario. All other calibrated model driver values were kept constant. The resulting loads were then compared to “baseline” loads based on meteorological data for the 2000–2018 period. Changes to flows and nutrient concentrations under different future climate scenarios were adjusted on a monthly basis according to the ratio:

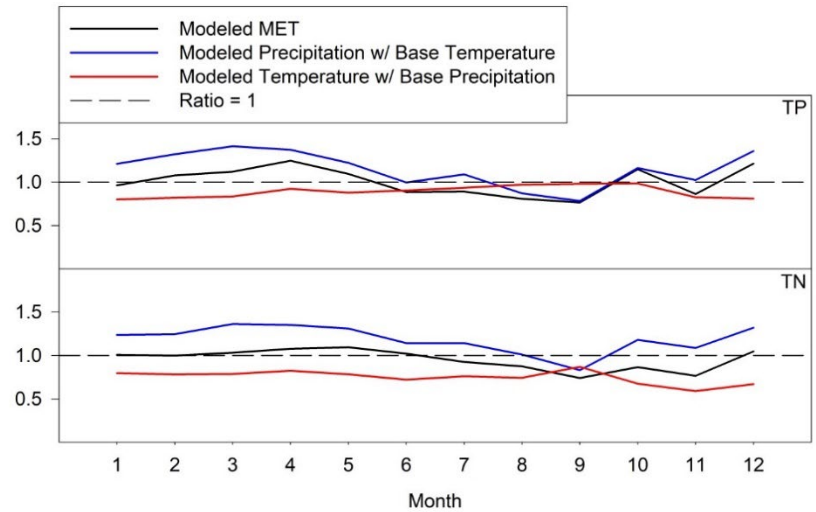
$$\frac{\boldsymbol{Future\ Climate\ Scenario\ value}}{\boldsymbol{Baseline\ Scenario\ value}}$$

This ratio was used as a multiplier to adjust nutrient loading to Owasco Lake under each future climate scenario.

To better understand the response of GWLF-E to future climate scenarios, we examined the net impact of adjusting temperature and precipitation independently. GWLF-E was run with 20 years of modeled future precipitation data (RCP 8.5, 2080) while holding temperature at baseline conditions (Figure 9, blue/top lines) and also with 20 years of modeled future temperature data while holding precipitation at baseline conditions (Figure 9, red/bottom lines). Despite some notable seasonal variations, the overall impact of modeled future increases in precipitation was an increase in watershed loading of TP and TN, while modeled future temperature increases resulted in decreased loads during most of the year (Figure 9). Modeled future values for both precipitation and temperature resulted in generally higher TP loads during the spring and winter months and lower loads during summer and early fall (Figure 9 top, black/middle line). A similar seasonal pattern was indicated for TN, with higher loads during spring and lower loads during summer and early fall (Figure 9 bottom, black/middle line). These findings with respect to increasing precipitation and nutrient loadings are consistent with other studies on the topic (Fischbach, Lempert, Molina-Perez, Tariq, Finucane, & Hoss, 2015).

Figure 9. Monthly Median Ratios of Phosphorous and Nitrogen

Total phosphorous (TP) and total nitrogen (TN) Watershed Loads from GWLF-E using Modeled Precipitation with Baseline Temperature Data (Blue/Top Line), Modeled Temperature with Baseline Precipitation Data (Red/Bottom Line), and Modeled Values for both Precipitation and Temperature (Black/Middle Line). The dashed line represents a ratio of 1.0.



4.2.5 Approach for Developing Representative Model Forecasts

Using modeling tools to forecast future conditions and how management decisions will affect the ecosystem modeled is a common use of water quality models (Chapra, 1997). These types of forecasts, which correspond to future environmental forcing conditions, are called *a priori* simulations because they correspond to future conditions that have not occurred during the model testing period. Since these specified forcing conditions can greatly influence the model predictions, it is important for these forcing conditions to be representative (Bierman & Dolan, 1986; Gelda et al., 2001). UFI has adopted a strategy for forecasting where long-term historic meteorological and flow data are used as drivers.[15] The model is run for the historic period of record to simulate "baseline conditions." The goal is to simulate this period to demonstrate the range of in-lake conditions that occur due to natural year-to-year variability in meteorology and flow. The baseline run is then compared to runs for various future scenarios, which also incorporate the observed natural variability in forcing conditions as multipliers applied to downscaled temperature and precipitation data. This approach provides a more realistic (i.e., probabilistic) picture of ecosystem response to naturally occurring variability in environmental drivers, such a meteorology and runoff.

For Owasco, Cayuga, and Skaneateles, the baseline periods were 2000–2018, 1998–2013, and 2000–2018, respectively. Accordingly, environmental drivers were developed for each of these periods.

Due to the influence of the water residence time (approximately three years for Owasco Lake, 10 years for Cayuga Lake, and 18 years in Skaneateles Lake), the effects of climate change on lake conditions will require several years to be fully manifested. Therefore, two sets of model runs were conducted for each lake to properly simulate the in-lake water quality impacts resulting from forecasted changes in temperature and precipitation. The first run (spin-up) was initialized with representative initial conditions and run for a minimum simulation period corresponding to approximately 90 percent of steady-state, assuming characteristics of a mixed reactor. The last year of the spin-up simulation provided the initial conditions for the desired model simulation.

4.3 Analytical Approach for Modeling Other Factors that Influence CHABs

As noted previously, cyanobacteria blooms occur through complex trophic interactions and are influenced by hydrodynamic effects within a lake. This section describes additional analyses conducted to model the level of influence from invasive dreissenid mussels on CHAB formation and the effect of wind on spatial distribution within the study lakes.

4.3.1 Modeling the Effects of Dreissenid Mussels on CHABs

As stated previously, invasive dreissenid mussels have resulted in ecological changes in New York State lakes. The quagga mussel invasion, for example, has resulted in increased coverage of soft substrates and more total mussel biomass in invaded lakes (e.g., Hetherington et al., 2019). Dreissenid mussels have been observed in all three study lakes, including quagga mussels inhabiting even the deepest waters. Of particular significance to the three study lakes is the finding that dreissenid mussels promote CHABs more in lakes with low-to-moderate phosphorus concentrations than in those with high phosphorus concentrations (Knoll et al., 2008; Sarnelle et al., 2012). We hypothesize that a primary invasion of zebra mussels and the secondary invasion by quagga mussels was an important factor in promoting CHABs in a number of oligotrophic and mesotrophic State lakes. Dreissenid mussels have been found to be an important determinant of CHABs, particularly in lakes with lower TP and Chl-*a* concentrations (Matthews et al., in prep). Significant association between CHABs and the interaction between dreissenids and lake fetch[16] suggests that lake-wide increases in cyanobacterial abundance are promoted by the presence of dreissenids and magnified along shorelines in lakes with long fetch lengths (Matthews et al., in prep). This mechanism may be particularly important in long and narrow lakes such as Owasco, Cayuga, and Skaneateles.

To further investigate the effects of dreissenid mussels on CHABs we performed two 19-year simulations with the water quality model for Owasco Lake, one simulation with the dreissenid mussel sub-model active and another with the sub-model disengaged. Contemporary meteorological conditions were used for both simulations. The model simulations were intended to represent water quality conditions in Owasco Lake with dreissenid mussels at current densities (UFI, 2020) and in their absence.

4.3.2 Modeling the Impact of Wind-Driven Transport on the Spatial Distribution of CHABs

Documented CHABs have been reported to occur predominately along shorelines in northern portions of Owasco, Cayuga, and Skaneateles Lakes (NYSDEC et al., 2018a, b, c). The predominance of CHABs in shoreline areas is likely driven by both in situ cyanobacterial growth and hydrodynamic influences. In certain instances, favorable environmental conditions (e.g., temperature, light, nutrients, turbulence) along the shoreline may allow cyanobacteria to proliferate. Alternatively, the shoreline may simply function as a natural barrier to the transport of buoyant cyanobacteria resulting in a concentrated accumulation or surface scum. Relatively low lake-wide concentrations of cyanobacteria can be magnified a thousand-fold to million-fold when concentrated as scums along the shoreline (Chorus & Bartram, 1999). Lakes with long fetches, such as the State's Finger Lakes, may be particularly susceptible to accumulations of cyanobacteria in downwind shoreline areas. Hydrodynamic processes, including wind-driven accumulation and transport, have been demonstrated to influence the spatial distribution of CHABs in large lakes (Carmichael & Boyer 2016; Francy et al., 2016; Li et al., 2014) and coastal bays (Raine et al., 2010).

To advance our understanding of the role of wind-driven transport in determining the spatial distribution of CHABs, the hydrothermal sub-model was run for each of the three study lakes with a buoyant conservative tracer used to mimic the transport of cyanobacteria due solely to water motion. The model simulations were conducted for the August 15 to August 30 interval of 19 years (2000–2018). At noon on August 15 of each simulation year, the top computational layer in all model segments was initialized to an arbitrary value of 1000 µg/L of cyanobacterial biomass (i.e., buoyant conservative tracer). Animated contour plots were created to evaluate and select representative data to present in plan view plots. Using noon data, the mass of cyanobacteria above 10 meters depth in each segment was divided by the volume of the surface layer to determine a concentration for each segment.

On-site wind measurements from Owasco Lake established that winds are predominately from the south[17] during May–October (Figure 10a). This wind direction is consistent with accumulations of cyanobacteria in northern areas of the lake. It's noteworthy that southerly winds are much less common at the Syracuse Airport (Figure 10b) located just 43 km (27 mi) to the northeast of Owasco Lake. This is likely a consequence of the north-south orientation of the glacial valleys, which transform westerly and easterly winds to southerly winds over the Finger Lakes. Winds during the August 15, 2015 to August 20, 2015 interval selected for presentation were also predominately from the south (Figure 11). Winds from other directions do occur (Figure 10) and undoubtedly contribute to CHABs in other areas of the lakes. Simulations with a three-dimensional model would be required to identify the impact of wind on the east-west distribution of cyanobacteria.

Figure 10. Wind Rose Plots Showing Wind Speed, Direction, and Frequency for May to October of 2014–2018

(a)Owasco Lake, and (b) Syracuse Hancock International Airport. The Owasco Lake data was collected from on-lake buoy operated by the Finger Lakes Institute. The Syracuse Airport data was available from the NOAA weather station, KSYR.

(a) (b)

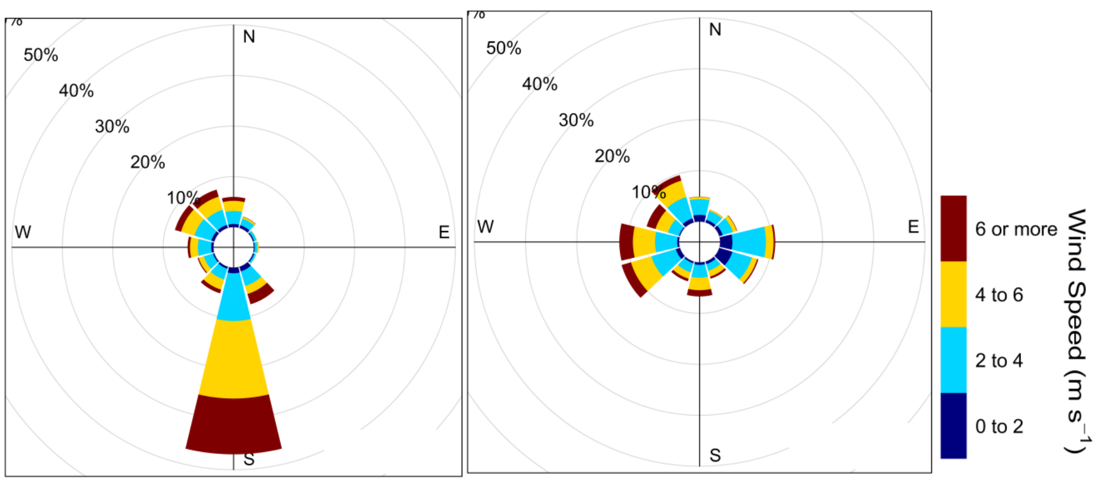
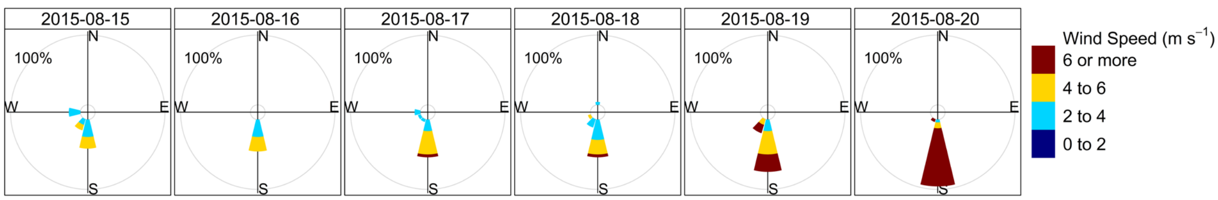


Figure 11. Daily Wind Rose Plots Showing Wind Speed, Direction, and Frequency for the August 15, 2015 to August 20, 2015 Interval

Color segments show the percentage of time that wind speeds were within each 2 meters per second range on a given date. Calm winds (0–2 m/s) are not shown.



5 Modeling Results

A variety of metrics were evaluated to compare the output from the 34 modeled scenarios. These included:

- Heat income
- Stratification (duration, depth, start and end dates)
- Schmidt stability
- Temperatures (upper water maximum temperature, average temperature, number of days >25°C)
- Total phosphorus
- Chlorophyll-*a*
- Phytoplankton community (diatoms, green algae, cyanobacteria)

Figure 12 presents the average annual temperature change factors for each of the scenarios ordered based on time slice and RCP. There is a generally increasing trend with time slice and RCP, but there can be substantial variability between models. Most of the change factors range from approximately +1℃ to +4℃, except for the RCP 8.5, 2080 scenarios that range from approximately +4℃ to +8℃. Note that none of the scenarios exhibited a reduction in temperatures, which is consistent between recent earth surface temperatures and current GCM simulations.

Figure 12. Annual Average Temperature Change Factor for Each Scenario

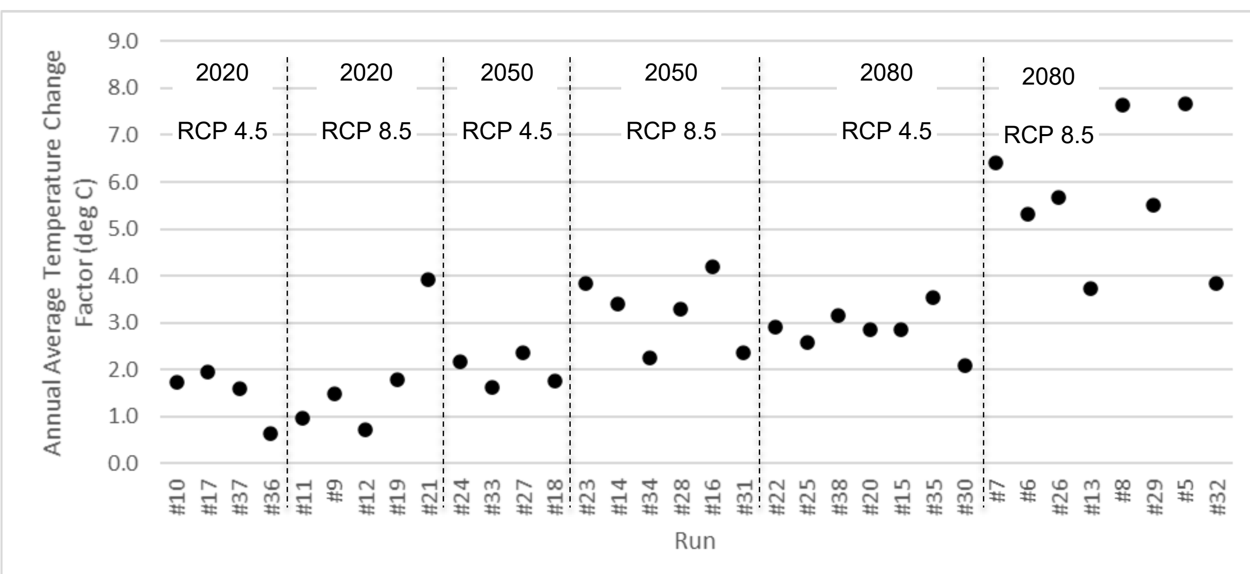
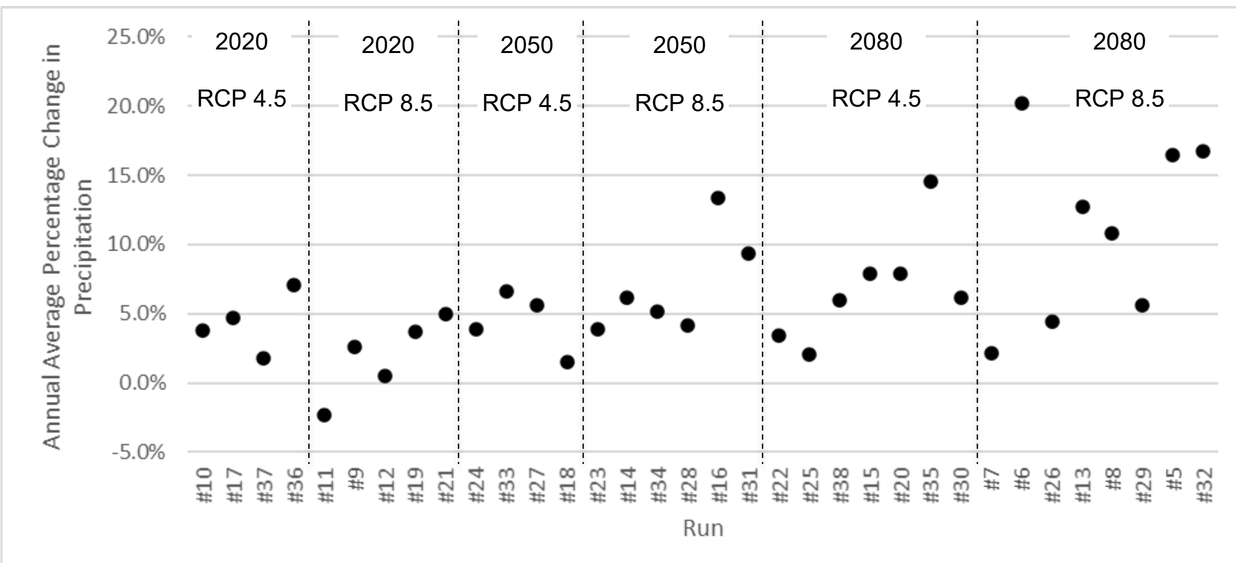


Figure 13 presents the average annual percent change for precipitation for each modeled climate scenario. There is a generally increasing trend with time slice and RCP, but there can be substantial variability between models. Most of the change factors range from approximately zero to +10% with some scenarios exhibiting precipitation change up to approximately +20%, typically in the 2080 RCP 8.5 scenarios.

The sections below present representative plots of the results for specific study lakes. In general, the trends for each lake were consistent; however, the values varied due to variations in physical lake characteristics.

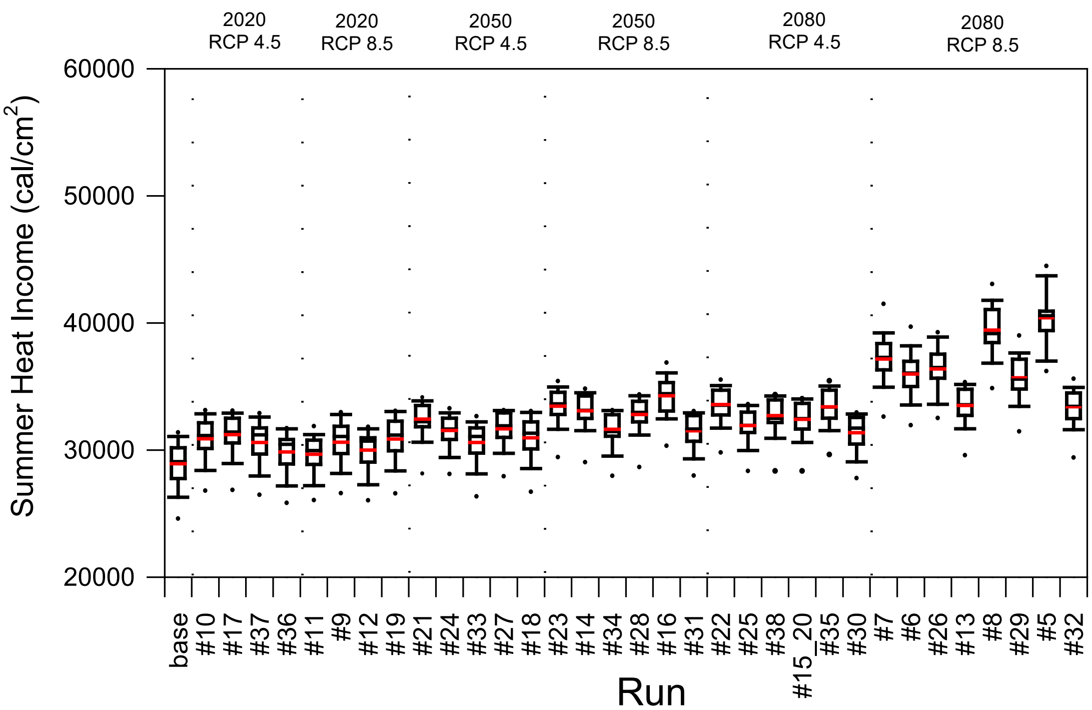
Figure 13. Annual Average Precipitation Change for Each Scenario



5.1 Heat Income

Summer heat income is the amount of heat necessary to raise the lake temperature from isothermal at 4°C to its maximum summer heat content (Wetzel, 2001). It is a function of direct heating of the surface layers, distribution of this heat by wind and the bathymetry of the lake. Differences between the runs in water temperature will be driven by the differences in air temperature, which is about the same between the lakes, but will differ between climate scenarios. Figure 14 presents the summer heat income for Owasco Lake. The box and whisker plots[18] are comprised of each individual year in the simulations for each climate scenario. There is a strong correlation between the annual temperature change factor and the summer heat income. In general, each lake exhibited similar patterns. The Julian day[19] for the summer maximum heat content was also evaluated. The range was approximately from 220 to 260 (early August to mid-September) for each of the climate scenarios, and there was no trend with temperature change factors.

Figure 14. Owasco Lake Summer Heat Income



The simulated summer heat income increased modestly from the base case to 2080-RCP 4.5 and abruptly for 2080-RCP 8.5 (Figure 14), consistent with temperature change factors (Figure 12). Although heat income and heat content have little direct bearing on cyanobacteria growth, they provide context to the impact that seasonal variations and climate change air temperatures may have on lake thermodynamics. The following sections will evaluate how relevant temperature factors are affected by higher temperatures.

5.2 Stratification

In the modeling, we defined the lake as stratified if the temperature difference between the surface and bottom was greater than 4°C. Figure 15 and Figure 16 present box and whisker plots for duration of summer stratification under base and climate scenarios for Owasco and Cayuga Lakes. While there are small differences between the results based on individual lake bathymetry, the changes are generally consistent and match each lake's trend in changing heat content. Further, the results demonstrate that relatively minor changes in air temperature may result in dramatic change in stratification duration. Annual temperature changes as low as +2°C (scenarios #10, #17, #37) increases the median (red line on plots) to the maximum of the base scenario. Larger changes in temperatures result in even further duration increases.

Figure 15. Owasco Lake Duration of Summer Stratification

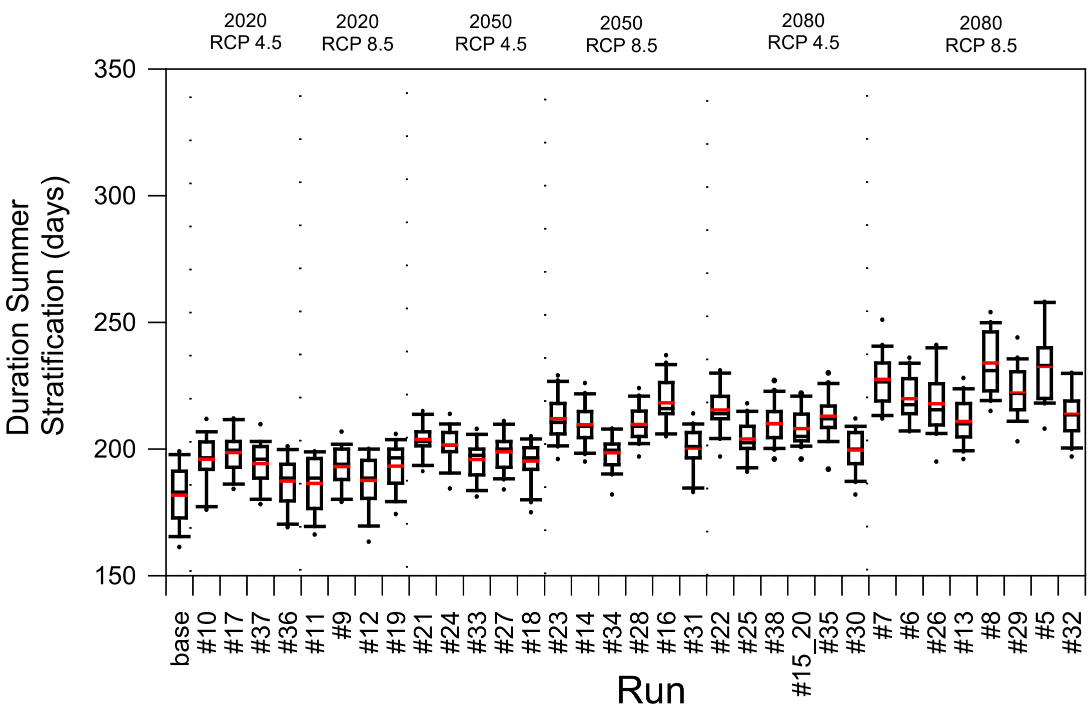
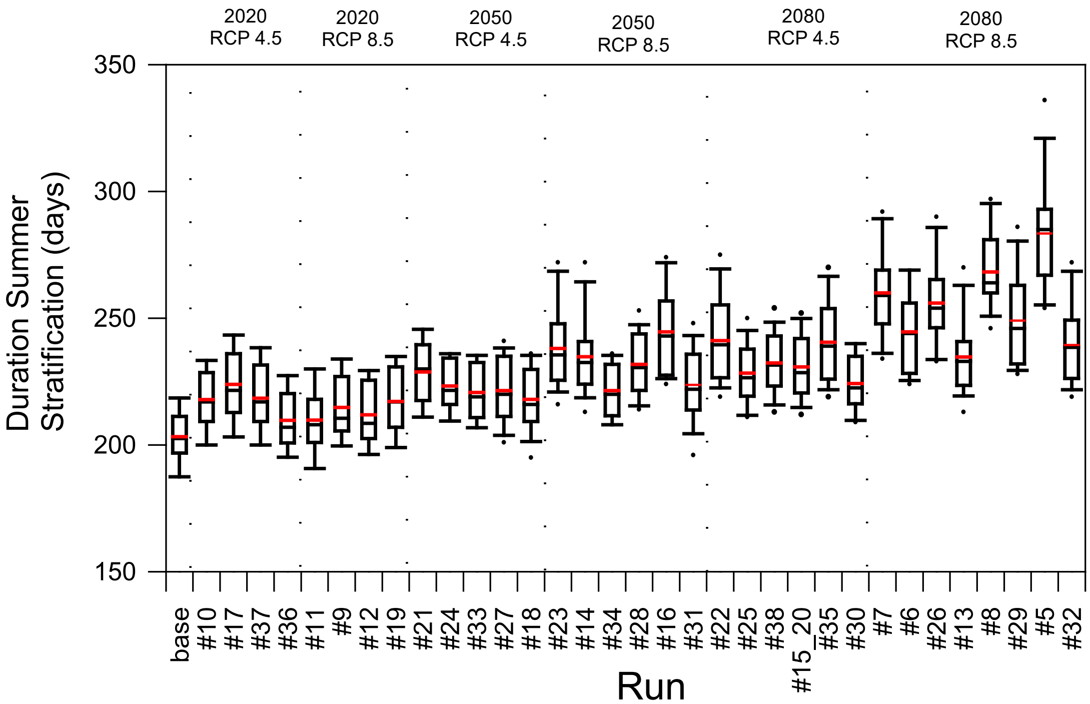


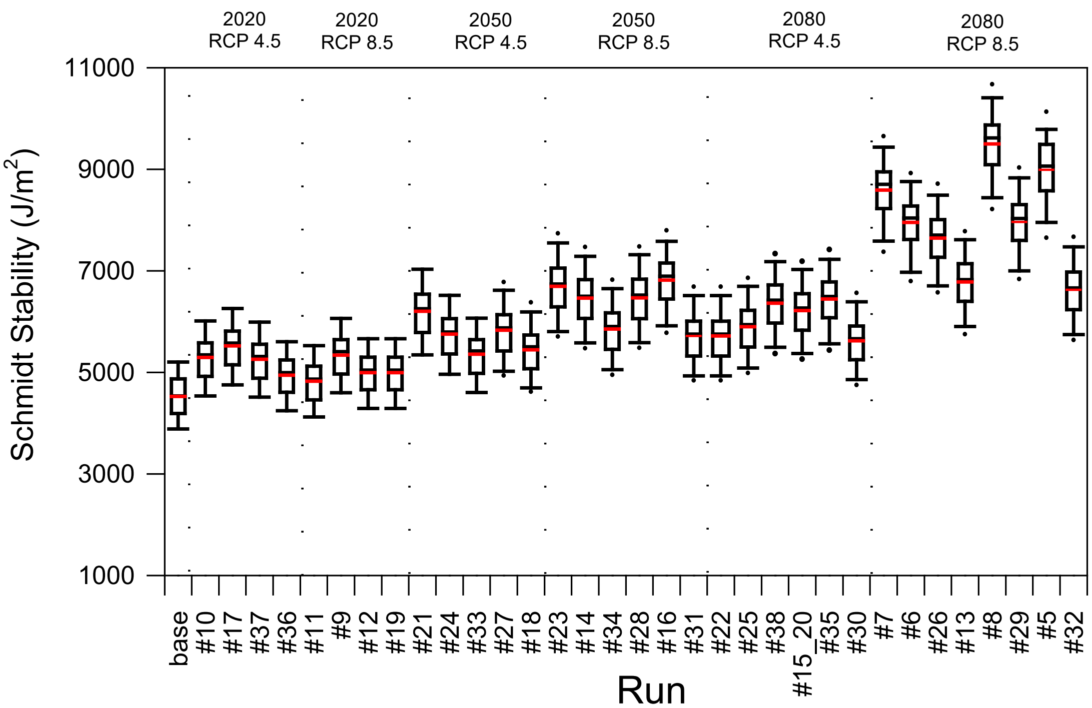
Figure 16. Cayuga Lake Duration of Summer Stratification



Duration of stratification results in both the hydrothermal model (temperature shifts only) and the water quality models (both temperature and precipitation shifts) were compared for Owasco Lake and were essentially the same, indicating that most of the effect on stratification is from air temperature. The start and end dates for stratification were compared as well. There was no consistent shift in the start and end dates or the variability between runs as temperature shifts increased. Overall, both the start and end dates experienced some shift as stratification duration became longer.

The Schmidt stability index was calculated daily from predicted lake temperature and averaged over the summer stratification period. The Schmidt stability index is a measure of the amount of mechanical energy needed to mix an entire body of water to uniform temperature (i.e., lake turnover). Increasing values indicate greater resistance to mixing (i.e., stronger stratification). The Schmidt stability index for the lakes in this study exhibited increasingly stronger stratification with increasing atmospheric temperatures (Figure 17). Therefore, not only is the stratification getting longer, but the water column is more strongly stratified which improves the likelihood of increased cyanobacteria growth (Becker, et al., 2010; Taranu, et al., 2015).

Figure 17. Schmidt Stability Index for Lake Cayuga (April–Oct)



5.3 Lake Temperature

Temperature metrics included (1) upper water maximum, (2) average temperature, and (3) days over 25°C for each lake. As with the heat content and stratification, there was a strong correlation between the temperature metrics and the annual temperature change factors from the climate scenarios (Figure 18). However, under the base scenario, days over 25°C did not occur in all years and reached a maximum of approximately 14 days. For the climate scenarios, even the lowest annual average temperature change factor of approximately 0.6°C to 0.7°C (runs #36 and #12, see Figure 12) resulted in an increase (over the baseline) of 9 and 11 additional days over 25°C. This metric is one of the few defined thresholds for cyanobacterial growth, as noted in many studies (Chang et al., 2020; Wehr et al., 2015; Carey et al., 2012). Table 4 presents the mean annual change in number of days above 25°C for each of the lakes modeled. The three lakes all had increasing days of surface water above 25°C as the time slice and RCP increased with largest change observed at RCP8.5 and the 2080 time slice. Between lakes, Owasco Lake had the greatest increase in days >25°C while Cayuga Lake had the smallest increase. This result suggests that lakes with smaller volumes are more susceptible to water temperature increases from climate change.

Maximum summer temperatures in the upper waters followed similar patterns as the days >25℃, but average summer temperatures for the upper waters showed smaller change (Figure 19). Because cyanobacteria are always present in the waterbodies, they can take advantage of short periods of favorable conditions, even if average conditions are less than ideal.

Figure 18. Owasco Lake Number of Days Greater than 25°C (April–Oct)

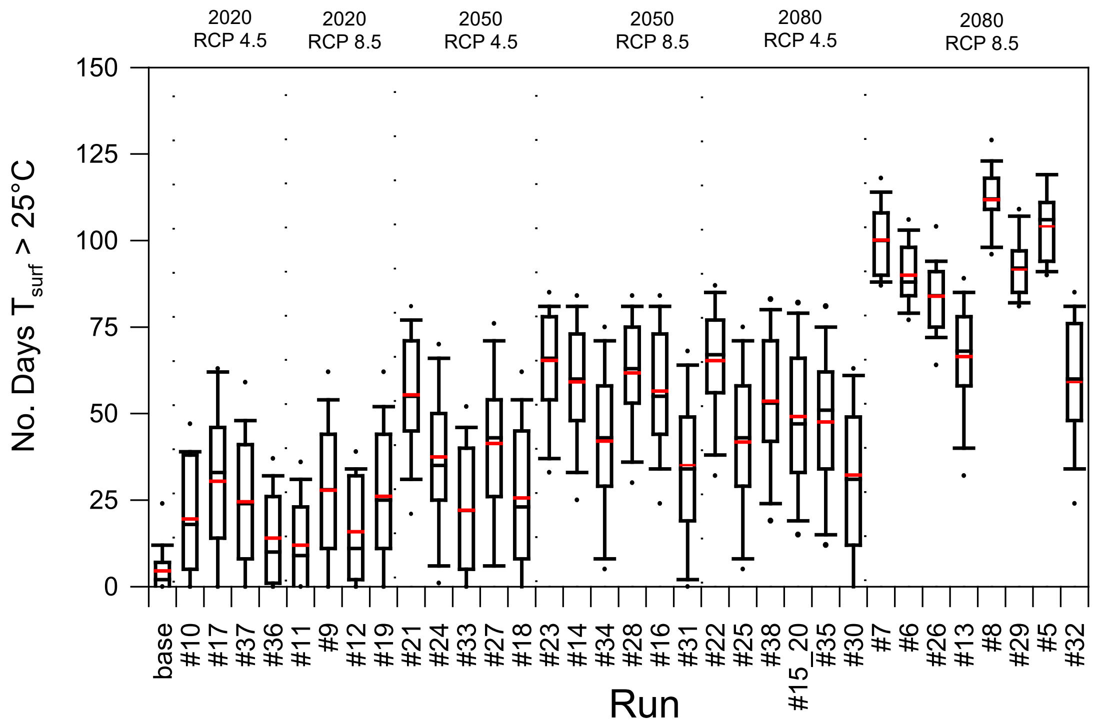
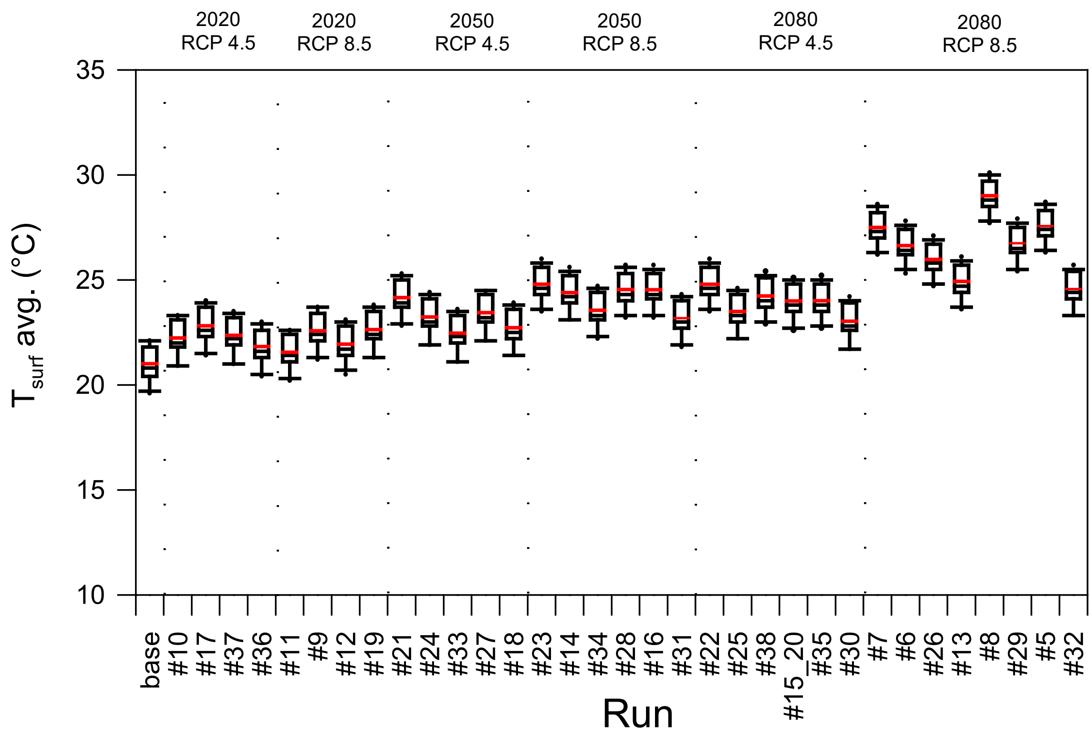


Table 4. Change from Baseline in Mean Number of Days per Season (April–Oct) that Surface Water Temperatures exceed 25°C for Modeled Scenarios (5-35)

Time Slice	RCP	Run No.	Owasco	Cayuga	Skaneateles
2020	4.5	#10	15	5	9
		#17	26	15	20
		#37	20	8	13
		#36	9	3	5
		avg.	**18**	**8**	**12**
	8.5	#11	7	1	3
		#9	23	12	17
		#12	11	3	6
		#19	22	10	15
		avg.	**16**	**7**	**10**
2050	4.5	#21	51	37	44
		#24	33	22	26
		#33	18	7	11
		#27	37	26	30
		#18	21	9	14
		avg.	**32**	**20**	**25**
	8.5	#23	61	46	57
		#14	55	40	49
		#34	37	26	30
		#28	57	43	53
		#16	52	38	48
		#31	31	18	23
		avg.	**49**	**35**	**43**
2080	4.5	#22	60	46	58
		#25	37	26	30
		#38	49	36	43
		#15_20	45	33	39
		#35	43	32	39
		#30	28	16	21
		avg.	**44**	**31**	**38**
	8.5	#7	95	90	96
		#6	85	78	86
		#26	79	70	77
		#13	62	47	59
		#8	107	103	109
		#29	87	80	87
		#5	99	94	100
		#32	55	40	50
		avg.	**84**	**75**	**83**

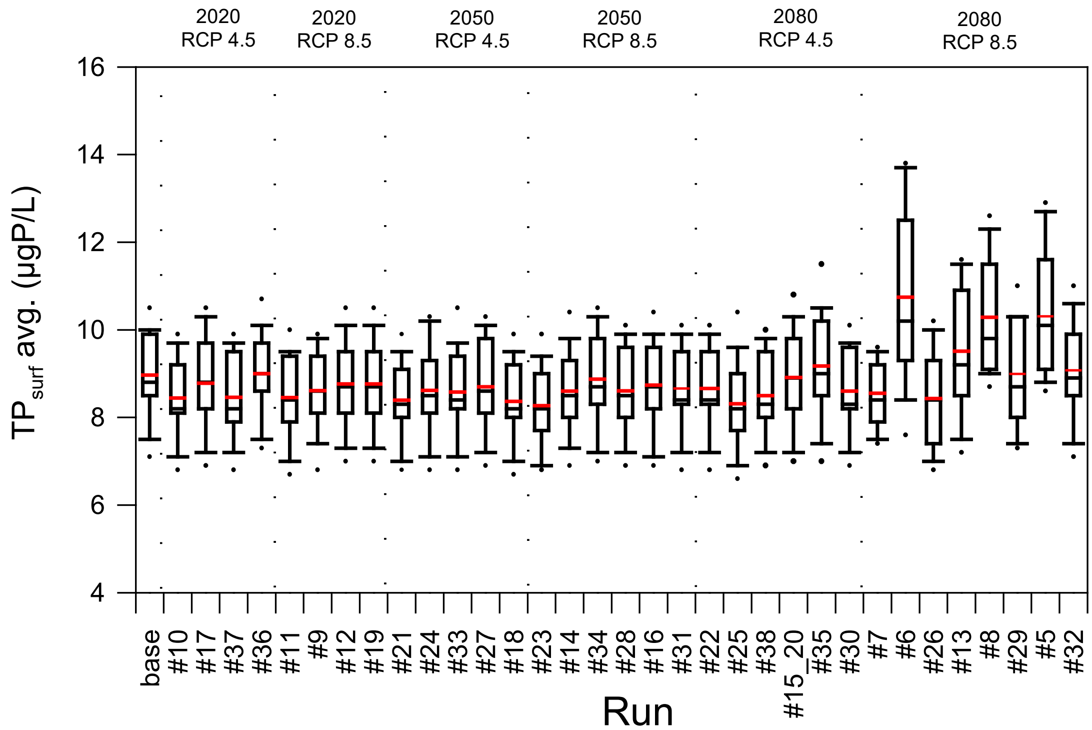
Figure 19. Owasco Lake Average Upper Water Temperatures for Summer (June–Sept)



5.4 Phosphorous and Chlorophyll- *a*

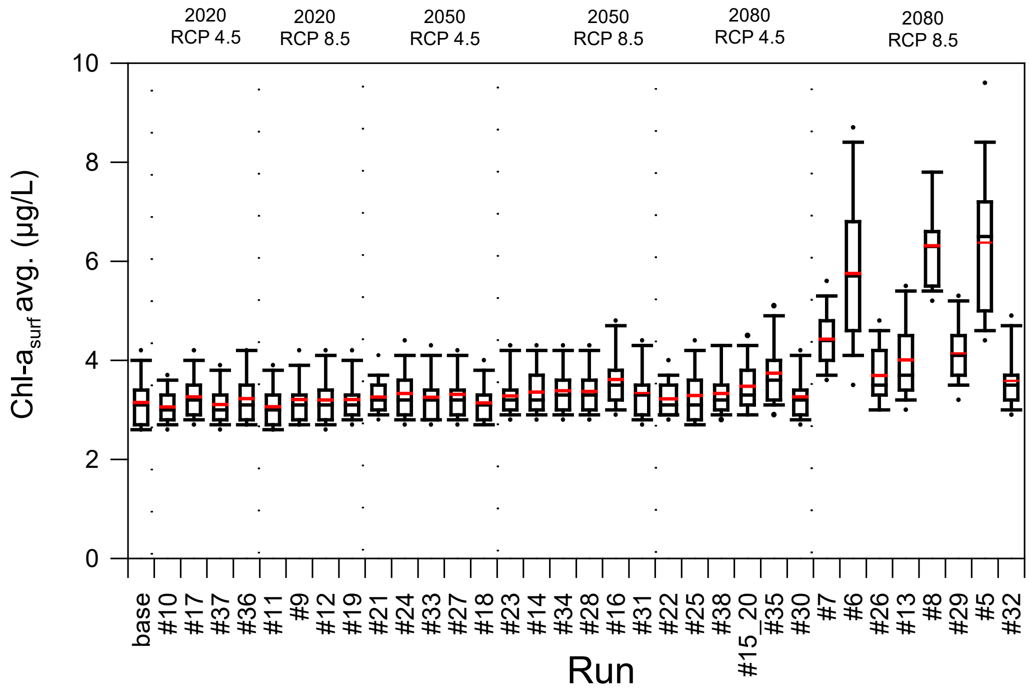
Unlike the temperature effects, Owasco Lake TP showed little change related to temperature or precipitation impacts from climate change scenarios (Figure 20). The average TP is between 8 and 11 µgP/L with a range between 7 and 14 µgP/L.[20] The 2080 time slice for an RCP of 8.5 had greater year to year and model to model variability than the lower RCPs and earlier time slices.

Figure 20. Average Total Phosphorous at the Surface, Owasco Lake (April–Oct)



Chlorophyll-*a* (Chl-*a*) exhibited a similar pattern in Owasco Lake as TP (Figure 21). On average the Chl-*a* is between 3 and 3.5 µg/L with a range between 2.8 and 4.5 µg/L. There is little variability between years for a given model run and between model runs until the 2080 time slice for an RCP of 8.5. The variability between years and model runs increased, the average is between 3 and 6 µg/L with the range between 2 and 8 µg/L.

Figure 21. Average Chl-*a* at the Surface, Owasco Lake (July–Sept)



5.5 Algae and Cyanobacteria Biomass

Modeled summer phytoplankton biomass in Owasco Lake is presented for the three phytoplankton groups in Figure 22. These results track closely with Chl-*a* (Figure 21) for each scenario. These data highlight a number of changes in community composition forecast to occur under future climate scenarios. The proportion of diatoms is decreasing with increasing temperatures, which is shown in Figure 23. Overall biomass across most of the scenarios increases slightly as green algae replace diatoms.[21] The seasonal average concentration of cyanobacteria increases only slightly until the most extreme scenarios where the temperature change factors are above 5°C and precipitation change factors of approximately +5% to +20% (scenarios #5, #6, #8, #26, #29). Scenario #7 is somewhat anomalous because, while it has a high temperature change factor of approximately +6°C, it has a relatively low change in precipitation at +2%. There is a complex interplay between changing precipitation and temperature patterns exhibited in these runs. Increasing precipitation influences the availability of phosphorus that increases the overall productivity of the system and cyanobacteria are better able to take advantage of the additional nutrients when coupled with increased water temperatures (Reinl et al., 2021). It is recognized that these extreme changes are not a near term risk. However, these changes to seasonal average conditions do not capture the potential for short term blooms that could cause negative impacts to lakes from smaller changes to temperatures.

Figure 22. Summer Average (July-September) Phytoplankton Biomass in Owasco Lake According to Functional Group

Each bar represents the average of 19 years of model output for a given climate scenario.

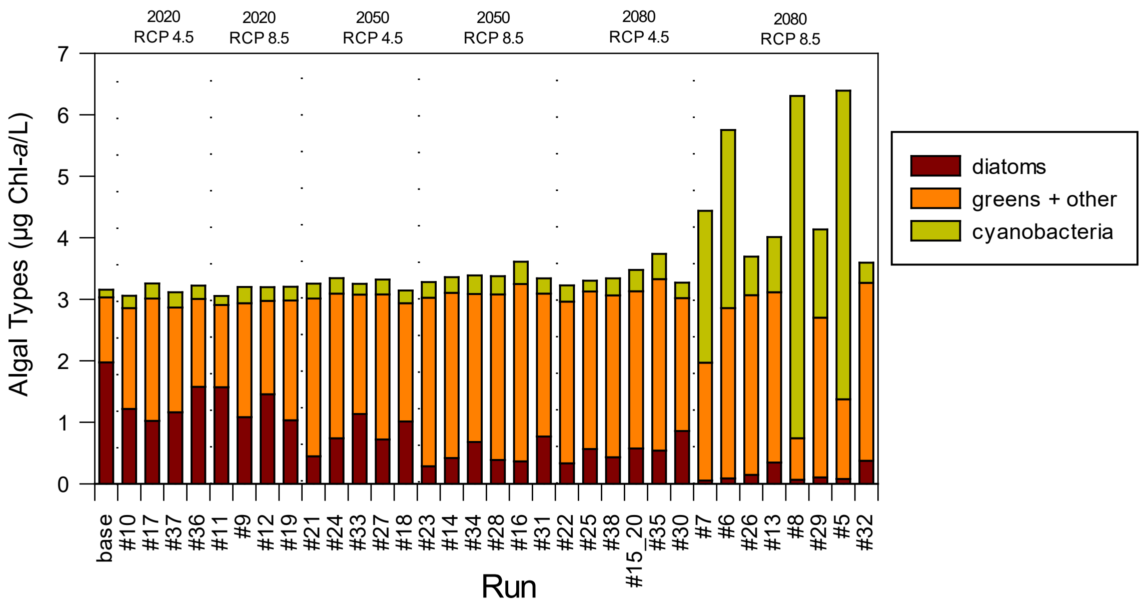
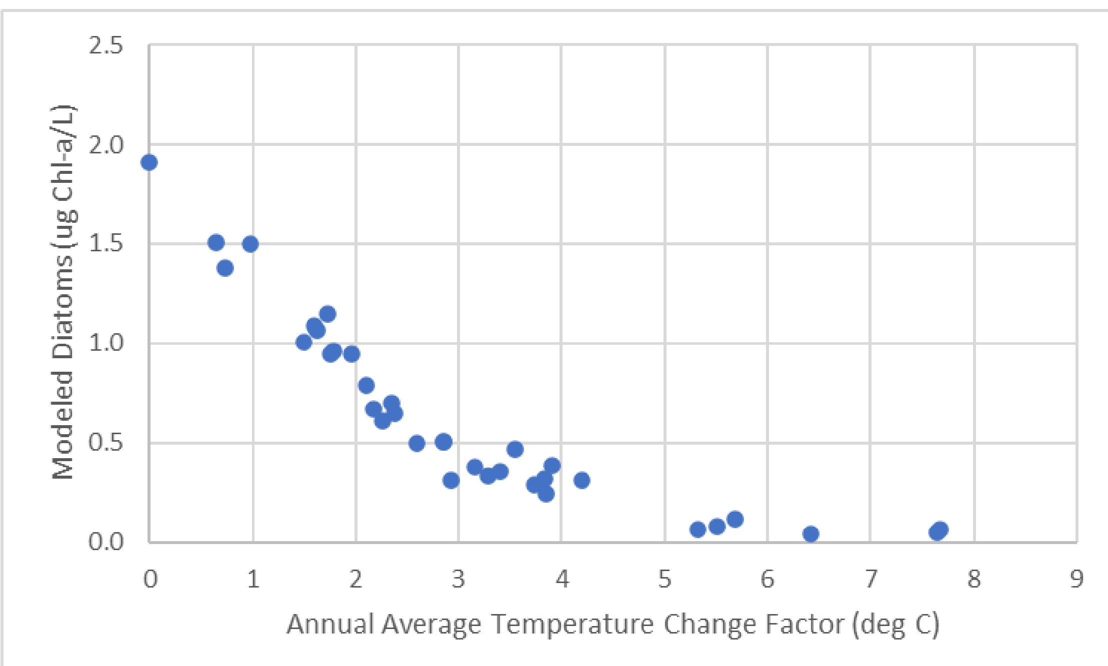
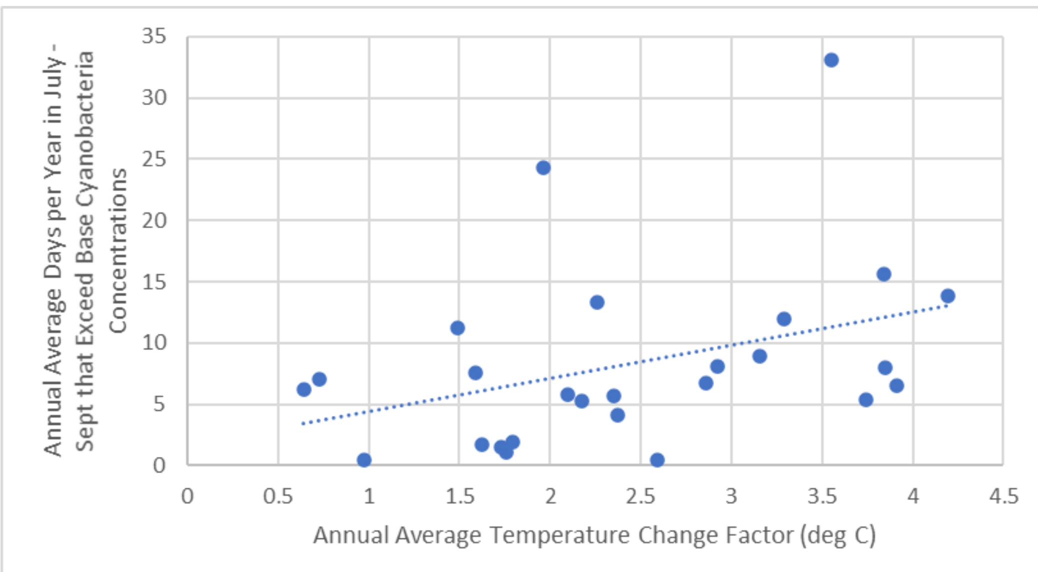


Figure 23. Relationship between Average July–Sept. Diatom Concentration and Temperature Change Factor for each Owasco Lake Scenario



There is no universally accepted quantitative definition of what constitutes a cyanobacterial bloom. In New York State since 2018, NYSDEC defined a bloom as blue green chlorophyll level of 25 µg/L blue green Chl-*a* and qualitative assessment of cyanobacteria under a microscope to verify "dominance." In 2020, New York State DEC broadened the Confirmed Bloom status definition to be applied to visual reports (previously designated a Suspicious Bloom) (NYSDEC, undated). Depending on the lake, blooms can extend lakewide or be limited to coves and shoreline areas. For this analysis, the Owasco Lake statistics for the baseline modeling scenario were used as a threshold for comparison for each climate scenario. The number of days for each scenario that exceeded the maximum cyanobacterial concentration of the base year was tabulated and averaged over the 19-year simulation. Figure 24 presents those results for Owasco Lake scenarios with temperature change factors below +5°C.

Figure 24. Annual Average Days per Year with Cyanobacteria Concentrations above the Base Year for Temperature Change Factors below +5℃ for Owasco Lake



While there is a wide range of variability, many scenarios exhibit substantially more days over the base year as temperatures increase. In the near term, there is not expected to be a major shift in the composition of nutrients or phytoplankton in lakes in the State, but these data indicate that higher temperatures have the potential to contribute to algal blooms at an increasing rate that is disruptive to beneficial uses and requires management.

5.6 Effects of Dreissenid Mussels

As discussed in section 4.3.1, the baseline model for Owasco Lake was run with and without dreissenid mussels to assess their impacts on CHABs. The modeled results suggest that dreissenid mussels have only a modest effect on total phytoplankton biomass, as indicated by the 9% average decrease in Chl-*a* during July–September with the mussel sub-model active (Table 5). The simulated decrease in Chl-*a* is consistent with increased grazing pressure on the phytoplankton community by dreissenid mussels.

Table 5. Impact of Dreissenid Mussels on Chl-*a* and the Relative Abundance of Diatoms, Green Algae, and Cyanobacteria in the Upper Waters (0-10 m) of Owasco Lake during July–September

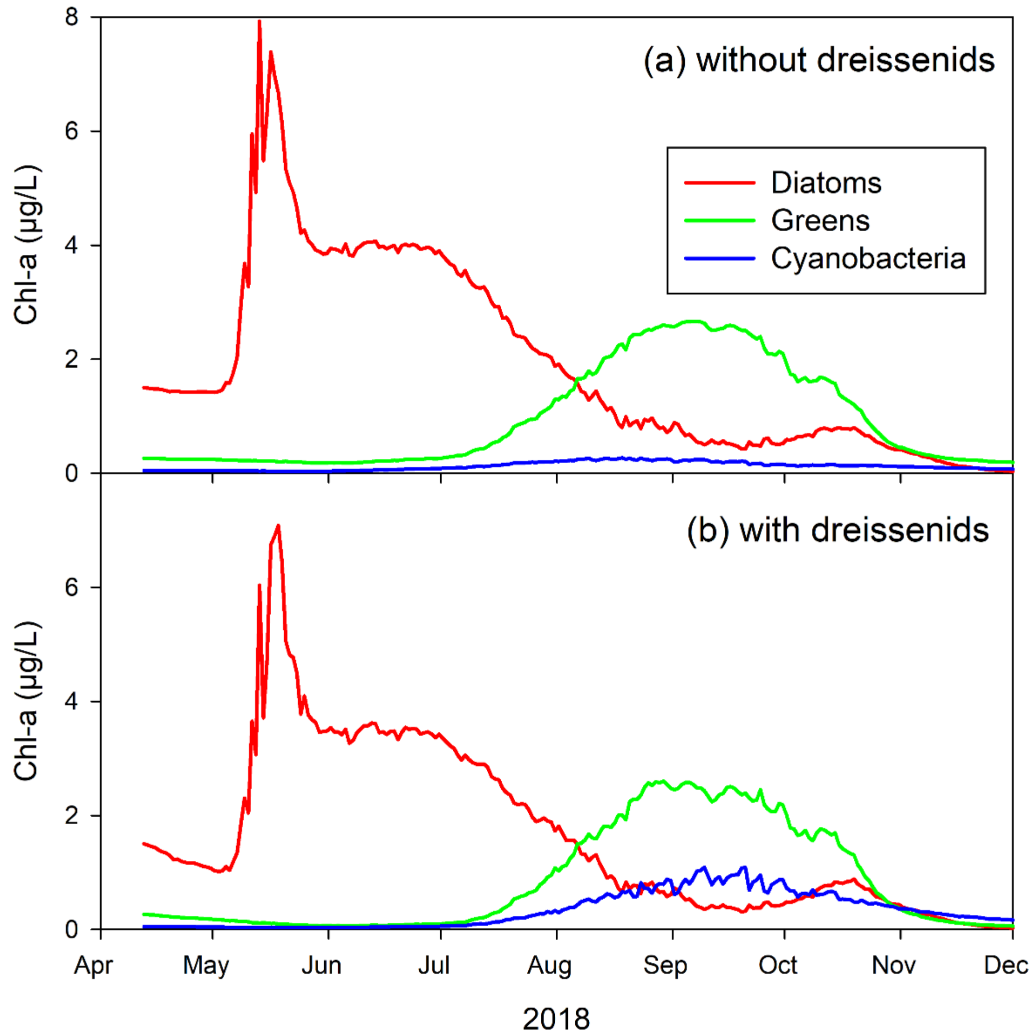
All values represent the change attributable to dreissenid mussels (positive values indicate increases with dreissenid mussels activated in model).

Simulation Year	Chl-*a* (µg/L)		Diatoms		Greens		Cyanobacteria	
	change	percent change	change	percent change	change	percent change	change	percent change
2014	-0.5	-12	-0.7	-27	0.1	8	0.1	53
2015	-0.9	-19	-0.7	-24	-0.3	-17	0.1	65
2016	0.2	8	-0.1	-11	0.1	9	0.3	130
2017	-1.0	-20	-0.6	-19	-0.4	-28	0.1	47
2018	-0.0	0	-0.2	-13	-0.2	-9	0.3	170
Average	-0.4	-9	-0.5	-19	-0.1	-8	0.2	93

Modeled changes in phytoplankton community composition attributable to dreissenids were large compared to changes in Chl-*a*. Averaged over 2014–2018, the presence of dreissenid mussels caused decreases of 13% and 9% in diatoms and green algae, respectively (Table 5). In contrast, the 2014–2018 average cyanobacterial biomass nearly doubled during summer as a result of selective filtration by mussels. The simulated increase in cyanobacteria ranged widely from 47% in 2017 to 170% in 2018. Mussels had little impact on the seasonal succession of the phytoplankton, with dominance by diatoms in spring and early summer and by green algae in late summer and fall (Figure 25). Although cyanobacteria increased during late summer in both the mussel and no-mussel scenarios, the increase was much larger due to the effects of dreissenid mussels (Figure 25). Mussel feeding preference was the primary cause for the modeled decrease in diatoms and green algae and the corresponding increase in cyanobacteria. In the calibrated model, the coefficient representing mussel feeding preference was assigned a value of 1.0 for diatoms and green algae and 0.2 for cyanobacteria. These values are qualitatively consistent with selective rejection of cyanobacteria (i.e., Microcystis) by dreissenid mussels due to toxicity or large colony size (Vanderploeg et al., 2013). Even the peak levels of cyanobacterial biomass associated with dreissenids were quite low (~ 1.0 μg/L as Chl-*a*; Figure 25b) and below concentrations typically associated with blooms. However, seemingly inconsequential cyanobacterial densities in the main lake (Figure 25a) can cause blooms when they are concentrated in the upper few centimeters of the water column and accumulate along the shoreline.

Figure 25. Modeled Seasonal Patterns for Diatoms, Green Algae, and Cyanobacteria Biomass in the Upper Waters of Owasco Lake During 2018

(a) Without the Effects of Dreissenid Mussels and (b) With the Effects of Dreissenid Mussels.



The finding that dreissenid mussels promote CHABs in New York State lakes, particularly in lower productivity systems, is supported by a number of studies in the Great Lakes region. Alteration of the phytoplankton community by dreissenid mussels has been documented in lakes in New York State (Idrisi et al., 2001; Matthews et al., 2001) and throughout the Great Lakes region (Barbiero et al., 2006; Makarewicz et al., 1999; Nicholls et al., 2002). Of particular note were reports of CHABs in nearshore regions of the Great Lakes following invasion by zebra mussels (Nicholls et al., 2002; Vanderploeg et al., 2001). Increased relative abundance of cyanobacteria (i.e., Microcystis) in inland lakes with low-to-moderate nutrient concentrations (TP=10-20 μg/L) has also been attributed to the effects of dreissenids (Knoll et al., 2008; Sarnelle et al., 2010). Results from New York State lakes further support the conclusion that dreissenids are an important cause of CHABs in lakes with low-to-moderate nutrient levels (Knoll et al., 2008; Sarnelle et al., 2005, 2010). Promotion of CHABs by dreissenid mussels confounds the well-established relationship between nutrient enrichment and cyanobacterial abundance in north-temperate lakes and suggests that reducing nutrient inputs may not be sufficient to control CHABs in invaded lakes (Raikow et al., 2004). However, experimental evidence suggests that these effects are reversible if densities of invasive dreissenid mussels can be controlled (Sarnelle et al., 2005).

5.7 Effects of Wind on the Spatial Distribution of Cyanobacteria

As described in section 4.3.2, we explored the impact of wind-driven transport on the spatial distribution of CHABs by running the baseline model for the three study lakes. Model simulations for the August 15, 2015 to August 20, 2015 interval indicated that the transport of surface waters by southerly winds is sufficient to explain accumulations of cyanobacteria in northern regions of Owasco, Skaneateles, and Cayuga Lakes (Figure 26, Figure 27). The initial uniform distribution of cyanobacteria in Owasco and Skaneateles Lakes on August 15 (Figure 26a) was transformed to a distinct north-south gradient by August 17 (Figure 26c) as the speed of southerly winds increased (Figure 11). Both the southerly winds and the gradient in cyanobacteria continued to strengthen over the following three days (Figure 11, Figure 26d-f). The absence of an increase in cyanobacteria in the northernmost segment of Skaneateles Lake was unexpected and deserves further investigation. The model simulation for Cayuga Lake also resulted in a progressively stronger north-south gradient as winds from the south increased over the six-day period (Figure 27a-f). However, higher concentrations of cyanobacteria did not extend north of a constriction and bend in the lake. Shallow water and a causeway located at the northern end of Cayuga Lake may limit the transport of surface water to that area. Additional hydrodynamic investigations are required to better understand the absence of simulated increases in cyanobacteria in the northernmost regions of Skaneateles and Cayuga Lakes.

These southerly winds have also been found to transport warmer surface water northward and cause upwelling of cooler water in the southern portion of Owasco Lake, resulting in a small but persistent temperature gradient (UFI, 2020). As with many biological processes, phytoplankton growth rates are a function of water temperature and certain taxa are favored within particular temperature ranges. Diatoms, for example, are generally favored at lower temperatures while cyanobacteria have a competitive advantage at higher temperatures. The combined effects of wind-blown accumulation and higher surface water temperatures are expected to favor cyanobacteria and the occurrence of CHABs in the northern regions of the study lakes and in the other Finger Lakes. The simulated distributions of cyanobacteria presented here have furthered our understanding of the documented preponderance of CHABs along shorelines in northern regions of Owasco, Cayuga, and Skaneateles Lakes (NYSDEC et al., 2018a, b, c).

Figure 26. Simulated Distribution of Cyanobacteria in Owasco and Skaneateles Lakes Resulting from Wind Induced Surface Advection

A conservative, buoyant constituent ("cyanobacteria") was introduced into the surface layer of all computational segments on August 15, 2015. Time progression of cyanobacteria transport resulting from a predominantly southerly wind over six days: (a) Aug 15, (b) Aug 16, (c) Aug 17, (d) Aug 18, (e) Aug 19, and (f) Aug 20.

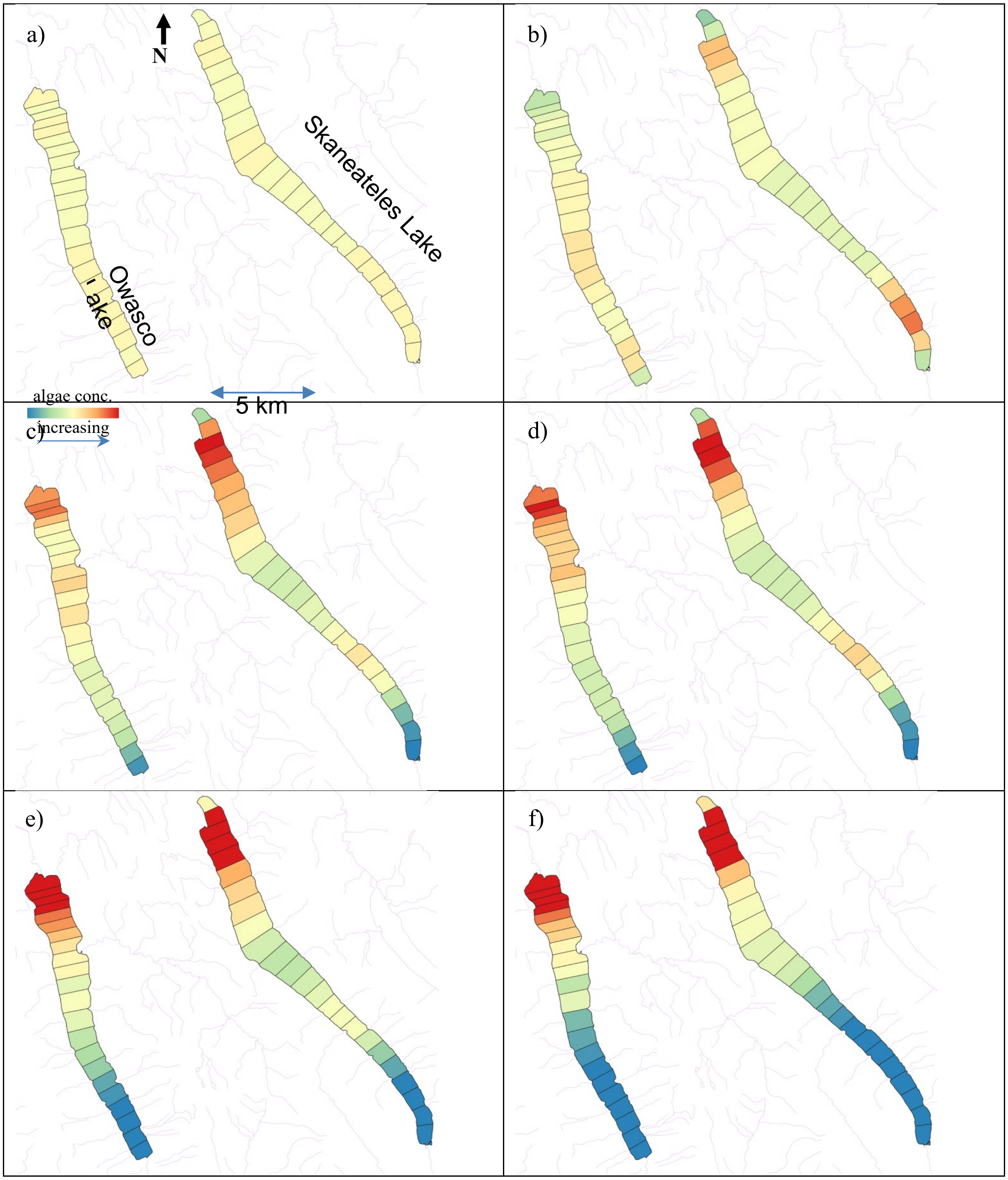
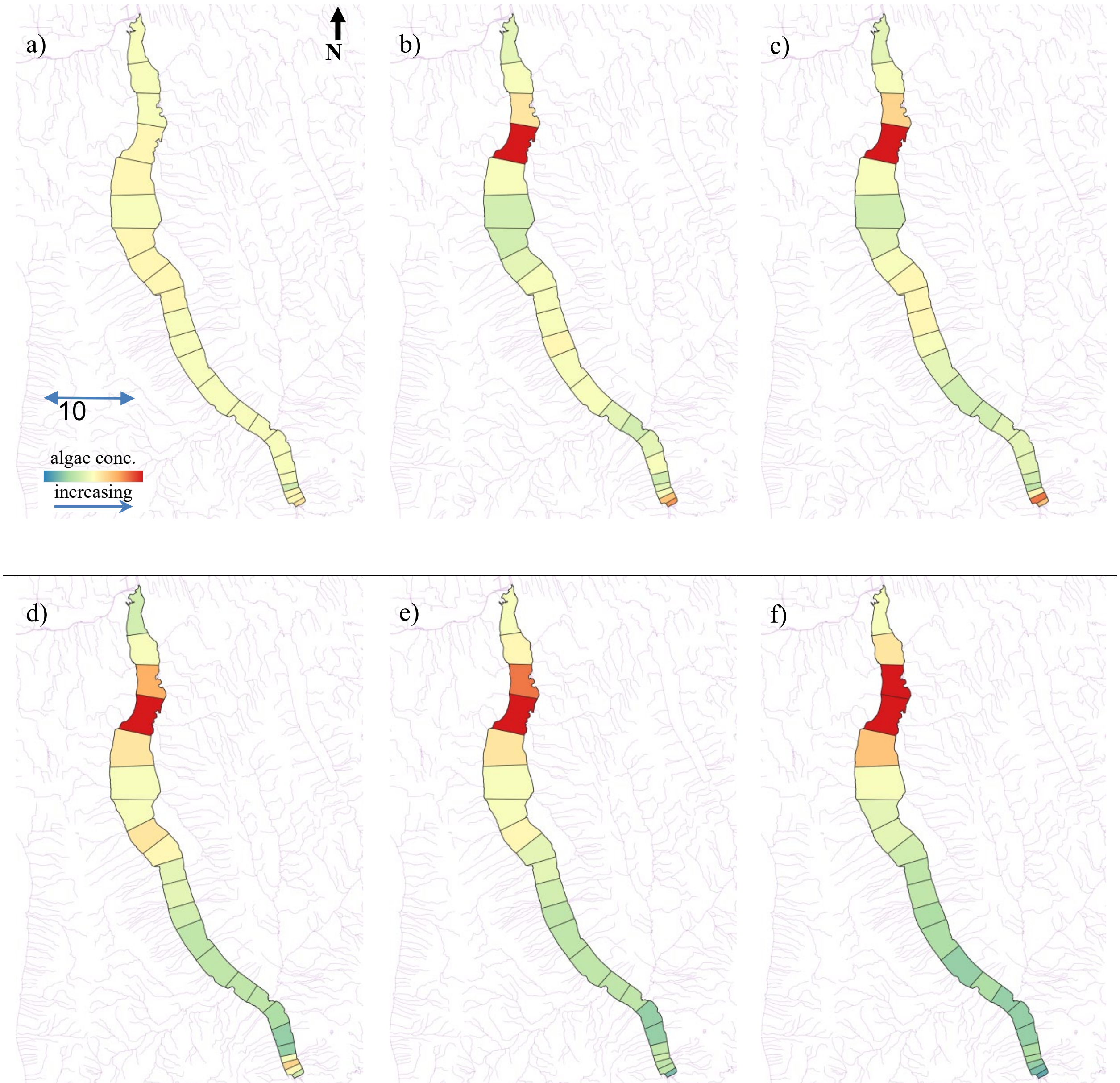


Figure 27. Simulated Distribution of Cyanobacteria in Cayuga Lake Resulting from Wind Induced Surface Advection

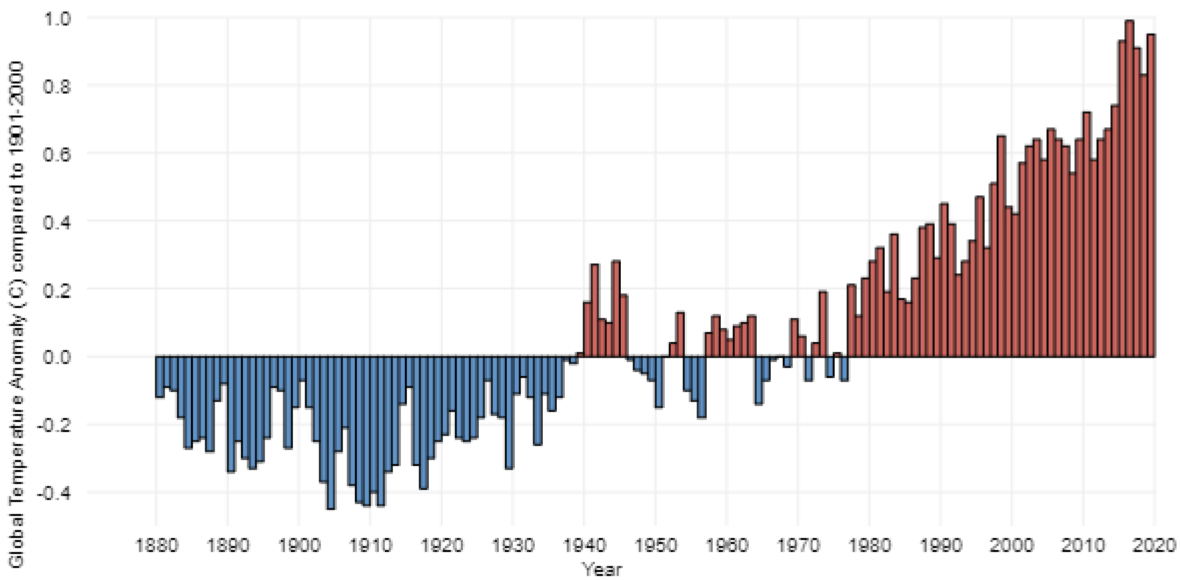
A conservative, buoyant constituent ("cyanobacteria") was introduced into the surface layer of all computational segments on August 15, 2015. Time progression of cyanobacteria transport resulting from a predominantly southerly wind over six days: (a) Aug 15, (b) Aug 16, (c) Aug 17, (d) Aug 18, (e) Aug 19, and (f) Aug 20.



6 Climate Change Mitigation for Cyanobacteria Blooms

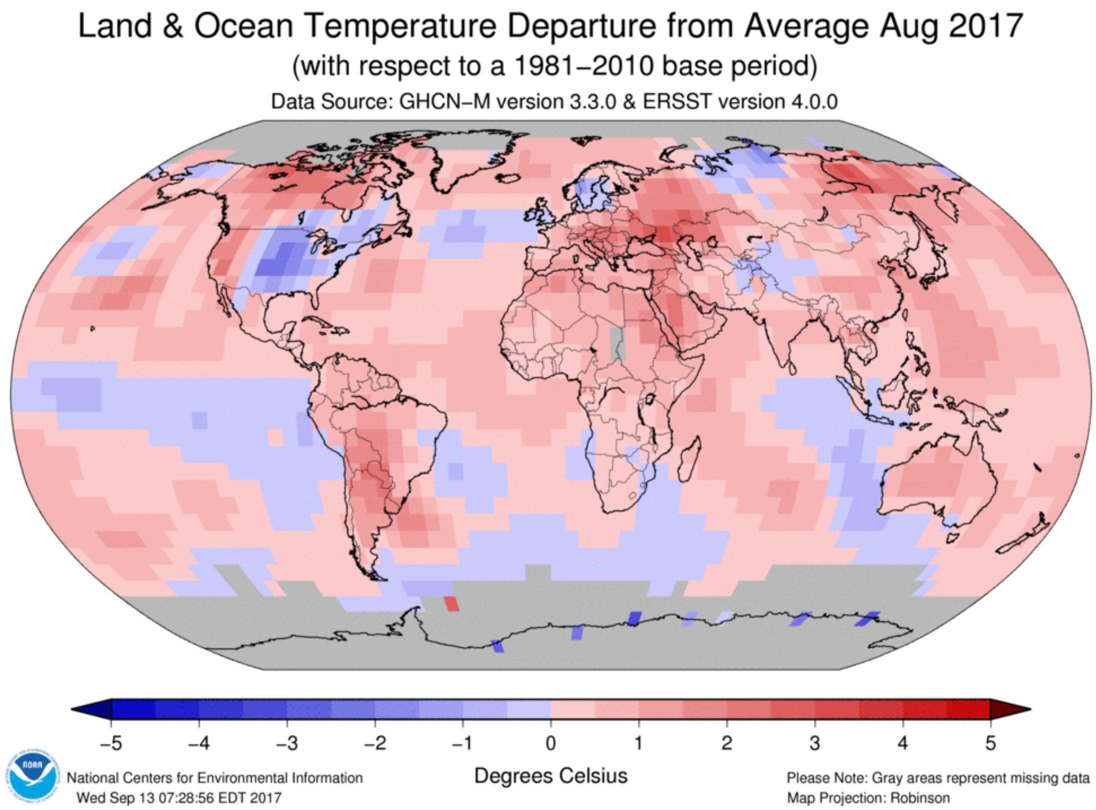
The results of the modeling show that rising temperatures have the potential to increase the prevalence of CHABs in New York State's surface water bodies. At the global scale, temperatures have been consistently rising over the last few decades (Figure 28). However, global average temperatures do not always equate to higher local temperatures across all seasons or years. Figure 28 shows the temperature anomaly for August 2017, which was globally one of the hottest on record, but the temperatures in the northeast U.S. were lower than average.

Figure 28. Average Annual Temperature Difference compared to the 20th Century Average (NOAA, 2020)



Given that rising temperatures are driven by global greenhouse gas emissions, it is not possible to address the root cause of the temperature changes at the local watershed scale. Increased cyanobacteria blooms are one of the many potential negative consequences of greenhouse gas emissions and should serve as a motivating factor for emission reductions. However, even with dramatic reductions in greenhouse gas emissions in the near term, it would take decades for those changes to be reflected in global average temperatures (Martinich et al., 2018). Therefore, continued increasing temperatures need to be accepted as the new normal based on current trends, and lake managers have to be able to implement management options to maintain lake environments to adapt to changing conditions.

Figure 29. August Global Temperature Anomaly (NOAA, 2017)



7 Cyanobacteria Management and Control Options

There are a variety of control options available for nuisance aquatic species, which fall into the general management categories of physical, biological, and chemical. Physical control techniques interfere with an organism's life cycle through physical removal or by physically disrupting its habitat. Biological control is based on the concept that a new introduced species that regulates the life cycle of the target species can help provide balance to an ecosystem and prevent negative impacts. Chemical control uses pesticides to kill the target organisms or prevent reproduction.

The lake management objective for CHABs is to reduce overall phytoplankton growth and to shift the phytoplankton composition from cyanobacteria to green algae and diatoms. Options to manage CHABs under projected future climate change conditions consist of a mix of short- and long-term actions that are intended to help maintain ecological function and reduce the prevalence of blooms. These methods include reducing nutrient loadings, inhibiting internal nutrient cycling, and implementing chemical control measures to directly address blooms (NYSDEC, undated).[22] Biological methods to influence trophic interactions within a lake by selectively introducing or removing fish species in order to encourage zooplankton that graze on cyanobacteria have been tried in some lakes but is not widespread (Rollwagen-Bollens et al., 2018). Invasive species management also plays a key role to prevent or manage species, such as zebra and quagga mussels, that contribute to ecosystem imbalances.

7.1 External Nutrient Load Reductions

While elevated nutrients were not an issue for the meso- and oligotrophic lakes included in this analysis, there are a number of eutrophic lakes in New York State that could benefit from external load reductions. Nutrients may originate from point and nonpoint sources in a watershed, and proper management requires an assessment of all potential sources. Watershed-level nutrient management requires coordination of many stakeholders and may necessitate costly solutions to address agricultural runoff, wastewater discharges, storm drain discharges and other pollutant sources. Regardless, nutrient reduction is a long-term strategy that has been successfully implemented to addresses a root cause of algal blooms. The Occoquan Agreement in Virginia, for example, was a multi-jurisdictional cooperative effort to replace a number of poorly performing publicly owned treatment works with a single advanced wastewater treatment plant among other provisions. The nutrient control provided by this agreement resulted in significant benefit to algae dynamics in the Occoquan Reservoir system since the 1970s.

Watershed scale nutrient load reduction is most typically implemented through regulatory action and the implementation of TMDLs. One of the largest TMDLs is the Chesapeake Bay TMDL for nutrients and sediments that covers approximately 64,000 square miles, including portions of southern New York State. Alternately, as with the Occoquan Agreement, cooperative agreements between local jurisdictions and other watershed stakeholders can be developed to voluntarily enact watershed best management practices without regulatory requirements. The NYC Watershed Memorandum of Agreement signed in 1997 between NYC and the local communities within the approximately 2,000 square mile watershed is a prime example of such an endeavor.

Whether through regulations or voluntary agreements, watershed scale nutrient management is a long-term effort that may require years to develop and implement. Further, many watersheds have diverse stakeholders that may have varying degrees of engagement on the issue, which requires broad and sustained outreach to build cooperation and consensus on management efforts. However, for many lakes, regular nutrient inputs from the watershed are a primary cause of the CHABs and should be incorporated with other methods of management, which will benefit the long-term health of the watershed and waterbody.

7.2 Internal Lake Nutrient Cycling Control

Internal nutrient cycling occurs through the uptake and release of nutrients through biological (e.g., growth and decay) and geochemical processes. A prominent process that results in the release of nutrients into the water column is due to the hypolimnion becoming anoxic during summer stratification. The anoxic conditions result in the release of phosphorus and/or nitrogen compounds from the sediment, which can be a major contributor to cyanobacterial growth. While this is not an issue in the lakes modeled in this study, it is common in many lakes in New York State and throughout the U.S.

Nutrient sequestration or inactivation using compounds, such as alum, can remove phosphorus from the water column and/or minimize internal phosphorus flux from legacy phosphorus accumulation in the sediment. Aluminum salts, like those used in coagulation, bind with phosphorus to form aluminum phosphorus compounds, immobilizing dissolved phosphorus. Further, the layer of alum along the bottom of the lake can provide a protective layer that prevents the release of phosphorus to the water column (Lake, Coolidge, Norton, & Amirbahman, 2007). Alum treatment can provide nutrient benefits for

10 to 20 years but requires sufficient benthic sampling and appropriate application design to be successful. In the State, the use of alum to sequester phosphorus is currently prohibited. The State is developing an approach for legally using nutrient inactivation in the State. Alum use is expected to remain prohibited until that process is complete. (NYSDEC, 2019; NYSDEC, 2020).[23]

Hypolimnetic aeration can also prevent phosphorus release by oxygenation of water directly above the sediments. The goal of hypolimnetic aeration is to meet the oxygen demand of the hypolimnion using ambient air or pure oxygen, while preserving lake stratification (Beutel and Horne, 1999). While the systems can be costly and require regular operation and maintenance, a hypolimnetic aeration system can achieve other water quality goals to support lake ecosystem and drinking water quality. The system would not necessarily need to be deployed throughout the entire lake but could be targeted in areas of high-sediment nutrient concentrations. The systems would need to be run through the summer months with oxygen levels monitored to ensure appropriate levels are maintained.

Neither of these methods for managing internal nutrient cycling permanently addresses the nutrient imbalance in a lake. The lake would likely return to its prior state without repeat applications of alum to sequester nutrients or continual operation of a hypolimnetic oxygenation system. Older literature cites the use of dredging to physically remove nutrient-laden sediments from lake bottoms, but dredging has numerous obstacles (e.g., costs, permitting, dredged spoil disposal, etc.) that limit its use in this application.

7.3 Chemical Control of Cyanobacteria

Algaecides are a commonly used chemical-control mechanism for algae and cyanobacteria. Such products are effective, low-cost, and relatively simple to apply. Regulations and approved pesticides vary by state. In the state of New York, copper—or hydrogen peroxide-based algaecides are the only approved pesticides for algae control (NYCDES, undated). There are numerous copper-based products on the market, many of which are a form of copper sulfate. Copper-based products are non-specific and broadly toxic to many species including plants and fish. Copper algaecides have been used for many years to control algal and cyanobacterial blooms in lakes. Copper algaecides do not address the cause of the imbalance that may be leading to a CHAB event and repeat applications may be required through a season to control the population until fall turnover. Additionally, copper will accumulate in the environment.

As an alternative to copper-based algaecides, hydrogen peroxide-based products are becoming more commonly used throughout the US and offer many benefits over their copper-based counterparts. In addition to leaving no long-term residuals, one of the primary benefits is the ability to selectively treat cyanobacteria. This is possible because cyanobacteria are prokaryotic and more sensitive to oxidative stress than eukaryotic algae. A side reaction of photosynthesis, the Mehler reaction, occurs in eukaryotic algae, but not in prokaryotic cyanobacteria. This reaction produces hydrogen peroxide in the cell. To avoid internal cellular damage, eukaryotic algae produce an enzyme which protects the cell contents. This enzyme is absent in cyanobacteria, making them more vulnerable to peroxide treatment.[24]

In addition to the benefit of selective treatment, hydrogen peroxide treatments have been shown to disrupt the circadian rhythm in cyanobacteria, which controls reproduction and the synthesis of 2-methylisoborneol (MIB), geosmin, and cyanotoxins (Sandrini et al., 2020; Schuurmans et al., 2018). A key aspect of this disruption is the suppression of key genes associated with microcystin production. The prolonged suppression of cyanobacteria observed following a hydrogen peroxide treatment is often attributed to this disruption as well as a shift in dominance to green algae.

The ability to selectively treat cyanobacteria with hydrogen peroxide can be used to shift the dominance away from cyanobacteria to a more beneficial species. This approach differs from copper treatments, where toxicity is non-specific and all groups of phytoplankton (green algae and cyanobacteria), as well as high-level organisms that consume cyanobacteria are impacted. The use of copper can create an opportunity for dominance that cyanobacteria will capitalize on, as their prokaryotic nature allows them to rebound quickly.

Algaecides are regulated pesticides, and their application is governed by the label instructions. The maximum application is typically no more than half a waterbody to allow fish to move away from treated areas. For large lakes, application across a large surface area is impractical. Targeted treatment in key areas based on field surveys is recommended, which reduces the cost as well as the potential ecological impacts from algaecides.

7.4 Other Methods of Cyanobacteria Control

In addition to the methods described, there are other potential methods for managing CHABs that have been implemented or are being researched (USEPA, 2020). These methods may be appropriate for specific lakes based on size, morphology and uses.

Mixing aeration, mechanical mixing, and hydrologic manipulation are various methods used to physically disrupt cyanobacteria's ability to regulate buoyancy through maintaining turbulence. Mixing has been implemented successfully in several lakes to improve water quality. The important factors to consider are (1) the mixing rate should be sufficiently high to entrain the cyanobacteria in the turbulent flow, (2) mixed layers should include a large portion of the photic zone, and (3) a large enough surface of the lake should be covered to limit areas for cyanobacteria to flourish (Visser, Ibelings, Bormans, & Huisman, 2016). Mixing systems are not widespread given the cost of installing and maintaining the systems, and the systems may interfere with recreation and other uses of a lake.

An ultrasound device can be used to control CHABs by emitting ultrasonic waves of a particular frequency such that the cellular structure of cyanobacteria is destroyed by rupturing internal gas vesicles used for buoyancy control (Wu, Joyce, & Mason, 2011). Ultrasound may not affect all cyanobacteria species similarly and also may impact green algae species (USEPA, 2020). Further, sufficient lake coverage is needed, similar to mixing, and as such it has some of the same drawbacks as mixing systems.

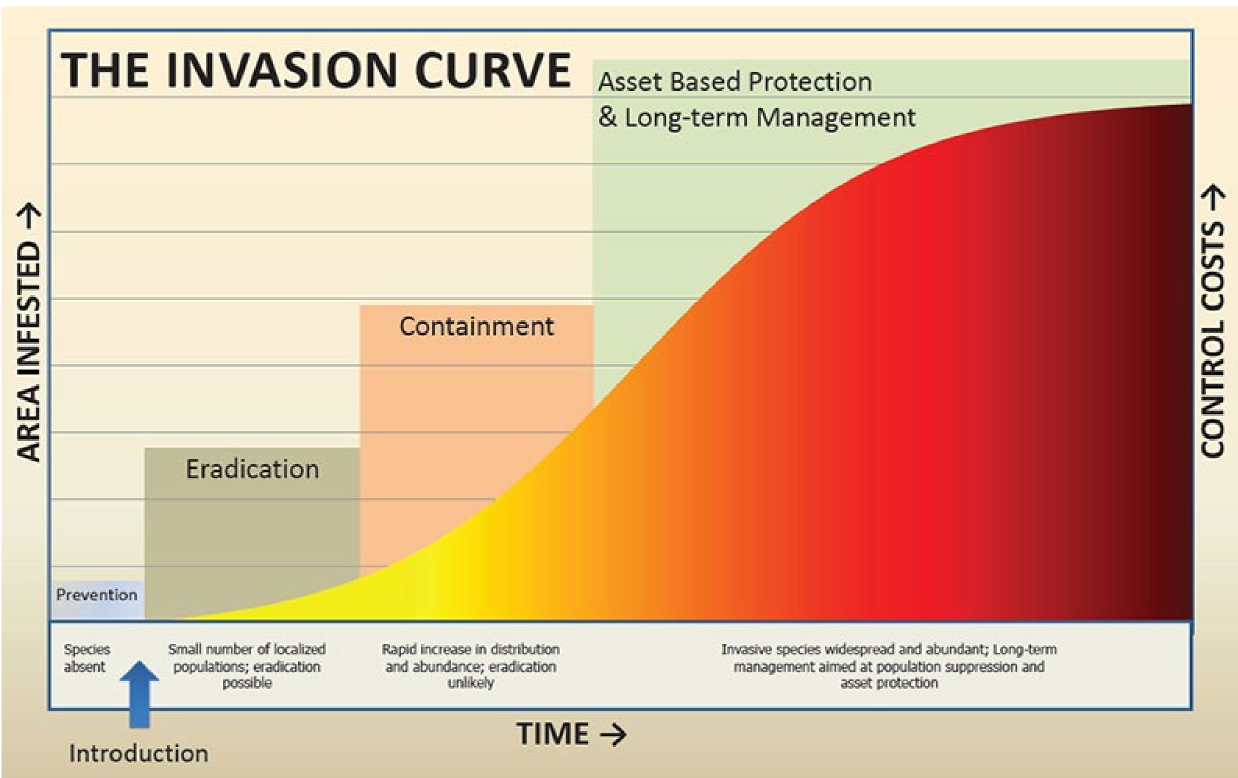
Surface skimming is appropriate when a bloom results in a surface scum, usually at the later stages. This method is typically useful for small areas around recreational areas; it is not useful for large areas or low-density blooms (USEPA, 2020). A benefit of skimming is that the associated nutrients and toxins are removed from the lake. A downside of this method is the fate/disposal of the concentrated bloom material and potential toxins.

Littoral Restoration: An issue with most of the methods used to manage CHABs is that the methods do not address the underlying imbalance in the system. Therefore, reactive response may temporarily improve conditions by reducing the bloom, but the fundamental conditions that promote bloom formation remain. Littoral restoration is an area of growing research to address the imbalance of a lake system through restoration of the most productive portions of the lake to encourage a diverse, stable population of primary producers (Crafton, Rosenfeldt, and Baughman, 2019). The goal is to encourage competition for available nutrients to prevent any one species from becoming dominant.

7.5 Invasive Species Management

Invasive species can contribute to the disruption of a lake ecosystem that may contribute to cyanobacteria dominance as has been demonstrated by the zebra and quagga mussels in the lakes included in this analysis. Currently, there is no available method to effectively eradicate invasive mussels once a population becomes established (Finger Lakes PRISM, 2020; State of Michigan, 2015). Even for species that can be effectively controlled or eradicated, the cost is high (Figure 30). Therefore, the most effective approach is to focus on preventing the spread of invasive species to new lakes to avoid the negative consequences.

Figure 30. Invasive Species Curve (USACE, 2014)



8 Managing Impacts of Cyanobacterial Harmful Algal Blooms

There are no simple solutions to reducing the prevalence of CHABs, nor for reversing the climate change temperature trends that may make blooms more frequent or severe. However, even if current climate trends cannot be easily reversed, our analysis suggests that CHABs could be limited by avoiding the worst-case climate scenarios. It is, therefore, necessary to continue to work to manage the impacts of blooms to protect public health as long-term solutions are implemented. Many of the negative consequences to public health from CHABs are currently being managed through monitoring and targeted responses. The two primary modes for exposure to water contaminated with cyanobacteria are through recreation and drinking water. Recreational exposure can continue to be addressed through visual observations and water quality monitoring, coupled with outreach and education to help people recognize when water quality conditions may be unsafe. There will continue to be negative consequences from beach closures, lost revenues, and inconvenience to the public, but the goal is to prevent morbidity and mortality from cyanotoxins.

While there have been high-profile instances of CHABs causing drinking water supplies to be shut down (e.g., Toledo, OH in 2014), most typical drinking water treatment processes can manage CHABs to protect public health (NYS Department of Health, 2018; USEPA, 2018). Filtration, which may also include coagulation, flocculation, and sedimentation, are highly effective for removal of cyanobacteria and intracellular[25] cyanotoxins but provide minimal removal of extracellular[26] cyanotoxins. Oxidants (e.g., chlorine, chloramine, chlorine dioxide, etc.) for inactivating pathogens can also be effective for oxidizing cyanotoxins (see Table 6. Effectiveness of Typical Oxidants for Disinfection of Algal Cells and Oxidation of Extracellular Toxins.). Oxidants could also lyse cells, potentially releasing toxins (extracellular). Oxidant usage should be minimized in advance of filtration processes in order to maximize the removal of toxins. In addition to oxidants, activated carbon, in powder or granulated form, can be utilized for management of extracellular cyanotoxins through adsorption.[27] The presence of a CHABs in a drinking water supply may require utilities to adjust treatment processes and perhaps increase chemical usage to effectively manage potential toxins.

Table 6. Effectiveness of Typical Oxidants for Disinfection of Algal Cells and Oxidation of Extracellular Toxins

Adapted from Table 7 of Hazen Harmful Algal Bloom White Paper, Summer 2015.

	Oxidants						
	Free Chlorine	Mono-chloramine	Chlorine Dioxide	Per-manganate	Ozone	Advanced Oxidation*	Ultra-violet Light
Cyanobacteria Disinfection (may lead to toxin release)	Effective	Moderate	Moderate	Moderate	Effective	Moderate	No
Microcystin	Moderate (pH dependent)	Slow/no oxidation	Slow/no oxidation	Effective	Effective	Effective	No
Cylindrospermopsin	Effective	No	No	No	Effective	Effective	No
Anatoxin A	No, Slow	No	No	Moderate	Effective	Effective	N/A
Saxitoxin	Effective	N/A	N/A	No	No	N/A	N/A

* Advanced Oxidation is a process the uses ozone, hydrogen peroxide and/or UV light to create hydroxyl radicals for disinfection.

It is recommended that utilities plan ahead to address intermittent CHABs by conducting an evaluation of existing treatment processes to identify effectiveness and steps for optimization. This could include jar tests, bench-scale tests, or desktop evaluations, such as the HazenAdams CyanoTOX® model.[28] New York State does not have a recommended approach for optimizing for cyanotoxin removal and considers each system on a case-by-case basis (Yeager, & Carpenter, 2019). Public water supply systems should discuss CHABs response plans with the NYS Department of Health to ensure regulatory compliance.

9 Conclusions

These results suggest that future climate change will produce conditions in New York State lakes that favor the growth of cyanobacteria and increase the likelihood of CHABs. While a continued focus on nutrient-load reduction is critical to prevent worsening conditions, nutrient management cannot be expected to completely stop all cyanobacteria blooms given future temperature trends and that oligo-and mesotrophic lakes are at risk of CHABs. Lake monitoring to identify presence of blooms is necessary to support beach closures and public announcements to prevent exposure to potential toxins. Additionally, while many typical water treatment processes are effective against cyanobacteria and cyanotoxins, plant operators may need to adjust processes to maximize removal efficiencies during a bloom. In-lake measures, such as mixing or hypolimnetic aeration, may help limit cyanobacteria growth in smaller waterbodies, but are impractical in large lakes. Application of algaecides, particularly hydrogen peroxide-based chemicals, can be a useful reactive measure to selectively target cyanobacteria in localized areas as a bloom is forming to prevent its growth in order to shift the population distribution back to green algae. Given the complexity of the problem with cyanobacteria blooms, it will take a combination of short- and long-term solutions to effectively manage across the lakes in the State.

10 References

Barbiero, R. P., Rockwell, D. C., Warren, G. J., & Tuchman, M. L. (2006). Changes in spring phytoplankton communities and nutrient dynamics in the eastern basin of Lake Erie since the invasion of Dreissena spp. *Can J Fish Aquat Sci*, 63, 1549-1563.

Beaulieu, M., Pick, F., & Gregory-Eaves, I. (2013). Nutrients and water temperature are significant predictors of cyanobacterial biomass in a 1147 lakes data set. *Limnol Oceanogr*, 58(5), 1736-1746.

Beaver, J. R., Manis, E. E., Loftin, K. A., Graham, J. F., Pollard, A. I., & Mitchell, R. M. (2014). Land use patterns, ecoregion, and microcystin relationships in U.S. lakes and reservoirs: A preliminary evaluation. *Harmful Algae*, 36, 57-62.

Beaver, J. R., Tausz, C. E., Scotese, K. C., Pollard, A. I., & Mitchell, R. M. (2018). Environmental factors influencing the quantitative distribution of microcystin and common potentially toxigenic cyanobacteria in U.S. lakes and reservoirs. *Harmful Algae*, 78, 118-128.

Beutel, M. W., & Horne, A. J. (1999). A review of the effects of hypolimnetic oxygenation on lake and reservoir water quality. *Lake and Reservoir Management*, 15(4), 285-297.

Bierman, V. J., & Dolan, D. M. (1986). Modeling of phytoplankton in Saginaw Bay: II Post-audit phase. *J Environ Eng*, 112, 415-429.

Birge, E. A., & Juday, C. (1914). *A limnological study of the Finger Lakes of New York.* Bull US Bureau of Fisheries, 32, 525-609.

Birge, E. A., & Juday, C. (1921). *Further limnological observations of the Finger Lakes of New York*. Bull US Bureau of Fisheries, 37, 961-975.

Boegman, L., Loewen, M. R., Culver, D. A., Hamblin, P. F., & Charlton, M. N. (2008). Spatial-dynamic modeling of algal biomass in Lake Erie: Relative impacts of dreissenid mussels and nutrient loads. J *Environ Eng*, 134(6), 456-468.

Burson, A., Matthijs, H. C., de Bruijne, W., Talens, R., Hoogenboom, R., Gerssen, A., ... & Huisman, J. (2014). Termination of a toxic Alexandrium bloom with hydrogen peroxide. *Harmful Algae*, 31, 125-135.

Carey, C. C., Weathers, K. C., & Cottingham, K. L. (2008). Gloeotrichia echinulata blooms in an oligotrophic lake: helpful insights from eutrophic lakes. *J Plankton Res*, 30(8), 893-904.

Carmichael, W. W., & Boyer, G. L. (2016). Health impacts from cyanobacteria harmful algae blooms: Implications for the North American Great Lakes. *Harmful Algae*, 54, 194-212.

Chapra, S. C. (1997). *Surface water-quality modeling*. New York, NY: McGraw-Hill.

Chorus, I., & Bartram, J. (1999). *Toxic cyanobacteria in water: A guide to their public health consequences, monitoring and management*. London, UK: E & FN Spon.

Cole, T. M., & Wells, S. A. (2018). *CE-QUAL-W2: A Two-dimensional, Laterally Averaged, Hydrodynamic and Water Quality Model, Version 4.1*. Submitted to US Army Engineering and Research Development Center, Vicksburg, MS.

Crafton, E., Rosenfeldt, E., & Baughman, D. (2019). Lake Management Strategies for Source Water Quality Management. *Proceedings of the 2019 Georgia Water Resources Conference*, April 16-18, 2019, Athens, Georgia.

Descy, J. P., Everbecq, E., Gosselain, V., Viroux, L., & Smitz, J. S. (2003). Modelling the impact of benthic filter-feeders on the composition and biomass of river plankton. *Freshwater Biol*, 48, 404-417.

Dodds, W. K, Bouska. W., Eitzmann, J. L., Pilger, T. J., Pitts, K. L., Riley, A. J., Schloesser, J. T., & Thornbrugh, D. J. (2009). Eutrophication of US Freshwaters: Analysis of potential economic damages. *Environ Sci Tech*, 43(1), 12-19.

Dolman, A. M., Rücker, J., Pick, F. R., Fastner, J., Rohrlack, T., Mischke, U., & Wiedner. C. (2012). Cyanobacteria and cyanotoxins: The influence of nitrogen versus phosphorus. *PLOS ONE*, 7(6), e38757. Doi:10.1371/journal.pone.0038757.

Donald, D. B., Bogard, M. J., Finlay, K., & Leavitt, P. R. (2011). Comparative effects of urea, ammonium, and nitrate on phytoplankton abundance, community composition, and toxicity in hypereutrophic freshwaters. *Limnology and Oceanography*, 56(6), 2161-2175.

Driscoll, C. T., Cleckner, L. B., Effler, S. W., Munson, R., Yavitt, J., & Roy, K. (2006). *Strategically targeted academic research in environmental quality systems for aquatic ecosystems*. Annual Report to the United States Environmental Protection Agency submitted by Syracuse University.

Evans, B. M., Lehning, D. W., Corradini, K. J., Petersen, G. W., Nizeyimana, E., Hamlett, J. M., Robillard, P. D., & Day, R. L. (2002). A comprehensive GIS-based modelling approach for predicting nutrient loads in watersheds. *J. Spatial Hydrology*, 2(2).

Finger Lakes PRISM. (2020). *Zebra Mussel*. Accessed at http://fingerlakesinvasives.org/invasive_species/zebra-mussel/ on 1/12/2021

Fischbach, J. R., Lempert, R. J., Molina-Perez, E., Tariq, A. A., Finucane, M. L., & Hoss, F. (2015). *Managing water quality in the face of uncertainty: A robust decision-making demonstration for EPA's national water program*. Rand Corporation.

Fishman, D. B, Adlerstein, S. A., Vanderploeg, H. A., Fahnenstiel, G. L., & Scavia, D. (2009). Causes of phytoplankton changes in Saginaw Bay, Lake Huron, during the zebra mussel invasion. *J Great Lakes* Res, 35, 482-495.

Francy, D. S., Brady, A. M. G., Ecker, C. D., Graham, J. L., Stelzer, E. A., Struffolino, P., Dwyer, D. F., & Loftin, K. A. (2016). *Harmful Algae*, 58, 23-34.

Gal, G., Hipsey, M.R., Parparov, A., Wagner, U., Makler, V., Zohary, T. (2009). Implementation of ecological modeling as an effective management and investigation tool: Lake Kinneret as a case study. *Ecological Modeling*. 220 (13-14) P 1697-1718.

Gelda, R. K., Effler, S. W., & O'Donnell, S. M. (2001). Probabilistic model of ammonia and toxicity status for urban lake. *J Water Res Pl-ASCE*, 127(5), 337-347.

Gelda, R. K., & Effler, S. W. (2007). Testing and application of a two-dimensional hydrothermal model for a water supply reservoir: Implications of sedimentation. *J Environ Eng Sci*, 6(1), 73-84.

Gelda, R. K., & Effler, S. W. (2008). Probabilistic model for temperature and turbidity in a reservoir withdrawal. *Lake Res Manage*, 24(3), 219-230.

Graham, J. L., Jones, J. R., Jones, S. B., Downing, J. A., & Clevenger, T. E. (2004). Environmental factors influencing microcystin distribution and concentration in the Midwestern United States. *Water Res*, 38, 4395-4404.

Haith, D. A., Mandel, R., & Wu, R. S. (1992). GWLF: Generalized watershed loading functions (User's Manual, version 2.0). Cornell University.

Heisler, J., Gilbert, P. M., Burkholde, J.M., Anderson, D. M., Cochlan, W., Dennison, W. C., Dortch, Q., Gobler, C. J., Heil, C. A., Humphries, E., Lewitus, A., Magnien, R., Marshall, H. G., Sellner, K., Stockwell, D. A., Stoecker, D. K., & Suddleson, M. (2008). Eutrophication and harmful algal blooms: A scientific consensus. *Harmful Algae*, 8, 3-13.

Hetherington, A. L., Rudstam, L. G., Schneider, R.L., Holeck, K. T., Hotaling, C. W., Cooper, J. E., & Jackson, J. R. (2019). Invader invaded: Population dynamics of zebra mussels (Dreissena polymorpha) and quagga mussels (Dreissena rostiformis bugensis) in polymictic Oneida Lake, NY, USA (1992-2013). *Biol Invasions*, 21(5), 1529-1544.

Hipsey, M. R., G. Gal, J.P. Antenucci, T. Zohary, V. Makler, and J. Imberger. (2006). Lake Kinnerret Water Quality Modeling System. *Proceedings of the 7th Annual Conference on HydroScience and Engineering*, Philadelphia, USA. Drexel University, College of Engineering.

Horton, R., D. Bader, C. Rosenzweig, A. DeGaetano, and W. Solecki. 2014. *Climate Change in New York State: Updating the 2011 ClimAID Climate Risk Information*. New York State Energy Research and Development Authority (NYSERDA), Albany, New York.

Huber, V., Wagner, C., Gerten, D., & Adrian, R. (2012). To bloom or not to bloom: contrasting responses of cyanobacteria to recent heat waves explained by critical thresholds of abiotic drivers. *Oecologia*, 169(1), 245-256.

Idrisi, N., Mills, E. L., Rudstam, L. G., & Stewart, D. J. (2001). Impact of zebra mussels (Dreissena polymorpha) on the pelagic lower trophic levels of Oneida Lake, New York. *Can J Fish Aquat Sci*, 58, 1430-1441.

Interstate Commission on the Potomac River Basin. (2012). *Modeling framework for simulating hydrodynamics and water quality in the Liberty Reservoir, Baltimore and Carroll Counties, Maryland.* Submitted to Maryland Department of the Environment, Baltimore, MD. Submitted by Interstate Commission on the Potomac River Basin, Rockville, MD.

JM Water Quality LLC. (2008). East Canyon Reservoir 2008 CE-QUAL-W2 Water quality model, Utah DEQ Phosphorus TMDL. Submitted to SWCA. Submitted by JM Water Quality, LLC.

Kaushal, S. S., Likens, G. E., Pace, M. L., Utz, R. M., Haq, S., Gorman, J., & Grese, M. (2018). Freshwater salinization syndrome on a continental scale. *Proceedings of the National Academy of Sciences*, 115(4), E574-E583.

Knoll, L. B., Sarnelle, O., Hamilton, S. K., Kissman, C. E. H., Wilson, A. E., Rose, J. B., & Morgan, M. R. (2008). Invasive zebra mussels (Dreissena polymorpha) increase cyanobacterial toxin concentrations in low-nutrient lakes. *Can J Fish Aquat* Sci, 65, 448-455.

Kunkel, K. E., Karl, T. R., Easterling, D. R., Redmond, K., Young, J., Yin, X., & Hennon, P. (2013). Probable maximum precipitation and climate change. *Geophysical Research Letters*, 40(7), 1402-1408.

Lajewski, C. K., Mullins, H. T., Patterson, W. P., & Callinan, C. W. (2003). Historic calcite record from the Finger Lakes, New York: Impact of acid rain on a buffered terrane. *Geo Soc of Am Bull*. 115(3), 373-384.

Lake, B. A., Coolidge, K. M., Norton, S. A., & Amirbahman, A. (2007). Factors contributing to the internal loading of phosphorus from anoxic sediments in six Maine, USA, lakes. *Science of the total Environment*, 373(2-3), 534-541.

Lee, T. A., Rollwagen-Bollens, G., Bollens, S. M., & Faber-Hammond, J. J. (2015). Environmental influence on cyanobacteria abundance and microcystin toxin production in a shallow temperate lake. *Ecotoxicology and environmental safety*, 114, 318-325.

Li, W., Qin, B., & Zhu, G. (2014). Forecasting short-term cyanobacterial blooms in Lake Taihu, China, using a coupled hydrodynamic-algal biomass model. *Ecohydrol*. 7, 794-802.

LimnoTech. (2016). CE-QUAL-W2 Lake response modeling of Petenwell and Castle Rock Reservoirs for the Wisconsin River TMDL. Submitted to US EPA Region 5.

Makarewicz, J. C., Lewis, T. W., Bosch, I., Noll, M. R., Herendeen, N., Simon, R. D., Zollweg, J., & Vodacek, A. (2009). The impact of agricultural best management practices on downstream systems: Soil loss and nutrient chemistry and flux to Conesus Lake, New York, USA. *J Great Lakes Res*, 35, 23-36.

Markensten, H., Moore, K., & Persson, I. (2010). Simulated lake phytoplankton composition shifts toward cyanobacteria dominance in a future warmer climate. *Eco Appl*, 20(3), 752-767.

Martinich, J., B.J. DeAngelo, D. Diaz, B. Ekwurzel, G. Franco, C. Frisch, J. McFarland, and B. O'Neill. (2018). Reducing Risks Through Emissions Mitigation. In *Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II* [Reidmiller, D.R., C.W. Avery, D.R. Easterling, K.E. Kunkel, K.L.M. Lewis, T.K. Maycock, and B.C. Stewart (eds.)]. U.S. Global Change Research Program, Washington, DC, USA, pp. 1346–1386.

Matthews, D. A. In prep. Factors associated with cyanobacterial harmful algal blooms in New York State lakes: findings from a statewide citizen science monitoring program.

Matthews, D. A., Effler, S. W., Matthews (Brooks), C. M., Siegfried, C. A., & Spada, M. E. (2001). Responses of Onondaga Lake, New York, to early stages of rehabilitation: Unanticipated ecosystem feedbacks. *Water Environ Res*, 73, 691–703.

Matthijs, H. C., Visser, P. M., Reeze, B., Meeuse, J., Slot, P. C., Wijn, G., ... & Huisman, J. (2012). Selective suppression of harmful cyanobacteria in an entire lake with hydrogen peroxide. *Water Research*, 46(5), 1460-1472.

Mullins, H. T., & Hinchey, E. J. (1989). Erosion and infill of New York Finger Lakes: Implications for Laurentide ice sheet deglaciation. *Geology*, 17, 622-625.

Mullins, H. T., Hinchey, E. J., Wellner, R. W., Stephens, D. B., Anderson, W. T., Dwyer, T. R., & Hine, A. C. (1996). Seismic stratigraphy of the Finger Lakes: A continental record of Heinrich event H-1 and Laurentide ice sheet instability. GSA, Special Paper 311, pp. 1-36.

Neilan, B. A., Pearson, L. A., Muenchhoff, J., Moffitt, M. C., & Dittmann, E. (2013). Environmental conditions that influence toxin biosynthesis in cyanobacteria. *Environmental microbiology*, 15(5), 1239-1253.

Newcombe, G., 2012. *International Guidance Manual for the Management of Toxic Cyanobacteria*. IWA Publishing.

New York State Department of Environmental Conservation (NYSDEC). (undated). Harmful Algal Blooms (HABs) Program Guide Version 3. pp. 38. Available at https://www.dec.ny.gov/docs/water_pdf/habsprogramguide.pdf

New York State Department of Environmental Conservation (NYSDEC), New York State Department of Health (NYSDOH), New York State Department of Agriculture and Markets (NYSDAM). (2018a). Harmful algal bloom action plan: Owasco Lake. pp. 119.

New York State Department of Environmental Conservation (NYSDEC), New York State Department of Health (NYSDOH), New York State Department of Agriculture and Markets (NYSDAM). (2018b). Harmful algal bloom action plan: Cayuga Lake. pp. 124.

New York State Department of Environmental Conservation (NYSDEC), New York State Department of Health (NYSDOH), New York State Department of Agriculture and Markets (NYSDAM). (2018c). Harmful algal bloom action plan: Skaneateles Lake. pp. 95.

New York State Department of Health. (2016). Blue-green Algae and Health. Accessed at: https://www.health.ny.gov/publications/6563.pdf

New York State Department of Health. (2018). How Public Water Supplies Optimize Treatment During Harmful Blue-green Algae Blooms. Accessed at: https://www.health.ny.gov/publications/6632.pdf

Nicholls, K. H., Heintsch, L., & Carney, E. (2002). Univariate step trend and multivariate assessments of the apparent effects of P loading reductions and zebra mussels on the phytoplankton of the Bay of Quinte, Lake Ontario. *J Great Lakes Res*, 28, 15–31.

NOAA. (2017). Global Climate Report - September 2017. Accessed at https://www.ncdc.noaa.gov/sotc/global/201708 on 11/8/20

NOAA. (2020). Understadning Global Temperature Change. Accessed at https://www.climate.gov/news-features/understanding-climate/climate-change-global-temperature on 11/11/20

NYSDEC. (2019). Harmful Algal Bloom Action Plan Conesus Lake. Accessed at https://www.dec.ny.gov/docs/water_pdf/conesushabplan.pdf on 12/15/20

NYSDEC. (2020). Harmful Algal Bloom Action Plan Lake Agawam. Accessed at https://www.dec.ny.gov/docs/water_pdf/habapagawam.pdf on 12/15/20

Orihel, D. M., Bird, D. F., Brylinsky, M., Chen, H., Donald, D. B., Huang, D. Y., Giani, A., Kinniburgh, D., Kling, H., & Kotack, B. G. et al. (2012). High microcystin concentrations occur only at low nitrogen-to-phosphorus ratios in nutrient-rich Canadian lakes. *Can J Fish Aquat Sci*, 69, 1457-1462.

Owens, E. M., Effler, S. W., Doerr, S. M., Gelda, R. K., Schneiderman, E. M., Lounsbury, D. G., & Stepczuk, C. L. (1998). A strategy for reservoir model forecasting based on historic meteorological conditions. *Lake Res Manage*, 14(2-3), 322-331.

Paerl, H. W. & Otten, T. G. (2013). Harmful cyanobacterial blooms: Causes, consequences, and controls. *Microbial Ecol*, 65(4), 995-1010.

Raikow, D. F., Sarnelle. O., Wilson, A. E., & Hamilton, S. K. (2004). Dominance of the noxious cyanobacterium Microcystis aeruginosa in low-nutrient lakes is associated with exotic zebra mussels. *Limnol Oceanogr*, 49(2), 482-487.

Raine, R., McDermott, G., Silke, J., Lyons, K., Nolan, G., Cusack, C. (2010). A simple short range model for the prediction of harmful algal events in the bays of southwestern Ireland. *J Marine Syst*, 83, 150-157.

Reinl, K. L., Brookes, J. D., Carey, C. C., Harris, T. D., Ibelings, B. W., Morales-Williams, A. M., De Senepont Domis, L. N., Atkins, K. S., Isles, P. D. F., Mesman, J. P., Morth, R. L., Rudstam, L. G., Stekzer, J. A. A., Venkiteswaran, J. J., Yokota, K, & Zhan, Q. (2021). Cyanobacterial blooms in oligotrophic lakes: shifting the high-nutrient paradigm. *Fresh Biol*, 66(9), 1846-1859.

Rollwagen-Bollens, G., Lee, T., Rose, V., & Bollens, S. M. (2018). Beyond eutrophication: Vancouver Lake, WA, USA as a model system for assessing multiple, interacting biotic and abiotic drivers of harmful cyanobacterial blooms. *Water*, 10(6), 757.

Samal, N. R., Pierson, D. C., Schneiderman, E., Huang, Y., Read, J. S., Anandhi, A., & Owens, E. M. (2012). Impact of climate change on Cannonsville Reservoir thermal structure in the New York City water supply. *Water Qual Res J Can*, 47(3-4), 389-405.

Sandrini, G., Piel, T., Xu, T., White, E., Qin, H., Slot, P. C., ... & Visser, P. M. (2020). Sensitivity to hydrogen peroxide of the bloom-forming cyanobacterium Microcystis PCC 7806 depends on nutrient availability. *Harmful Algae*, 99, 101916.

Sanseverino, I., Conduto, D., Pozzoli, L., Dobricic, S. & Lettieri, T. (2016). *Algal bloom and its economic impact*; EUR 27905 EN; doi:10.2788/660478.

Sarnelle, O., Morrison, J., Kaul, R., Horst, G., Wandell, H., & Bednarz, R. (2010). Citizen monitoring: Testing hypotheses about the interactive influences of eutrophication and mussel invasion on a cyanobacterial toxin in lakes. *Water Res*, 44, 141-150.

Sarnelle, O., Wilson, A. E., Hamilton, S. K., Knoll, L. B, & Raikow, D. F. (2005). Complex interactions between the zebra mussel, Dreissena polymorpha, and the harmful phytoplankter, Microcystis aeruginosa. *Limnol Oceanogr*, 50(3), 896-904.

Sarnelle, O., White, J. D., Horst, G. P., & Hamilton, S. K. (2012). Phosphorus addition reverses the positive effect of zebra mussels (Dreissena polymorpha) on the toxic cyanobacterium, Microcystis aeruginosa. *Water Res* 46, 3471-3478.

Schuurmans, J. M., Brinkmann, B. W., Makower, A. K., Dittmann, E., Huisman, J., & Matthijs, H. C. (2018). Microcystin interferes with defense against high oxidative stress in harmful cyanobacteria. *Harmful Algae*, 78, 47-55.

Smith, A. J., Duffy, B. T., Onion, A., Heitzman, D. L., Lojpersberger, J. L., Mosher, E. A., & Novak, M. A. (2018). Long-term trends in biological indicators and water quality in rivers and streams of New York State (1972‐2012). *River Research and Applications*, 34(5), 442-450.

Sorichetti, R. J., Creed, I. F., & Trick, C. G. (2014). Evidence for iron-regulated cyanobacterial predominance in oligotrophic lakes. *Freshwater Biol*, 59, 679-691.

State of Michigan. (2015). Status and Strategy for Zebra and Quagga Mussel Management. Accessed at: https://www.michigan.gov/documents/invasives/egle-ais-dreissenids_708905_7.pdf

Tu, J. (2009). Combined impact of climate and land use changes on streamflow and water quality in eastern Massachusetts, USA. *J Hydrol*, 379(3-4), 268-283.

USACE. (2014). *Slowing the spread of new invasives*. Accessed at http://www.usace.army.mil/Media/News-Archive/Story-Article-View/Article/486891/slowing-the-spread-of-new-invasives/ on 12/15/2020

U.S. Army Corps of Engineers (USACE). (2015). Water Quality Modeling Report (report number CENWO-ED-HA/WQMR/Zorinsky2015), Application and Calibration of the CE-QUAL-W2 Version 3.71 Hydrodynamic and Water Quality Model for Zorinsky Reservoir. Water Control and Water Quality Section Hydrologic Engineering Branch Engineering Division Omaha District U.S. Army Corps of Engineers. Omaha District, Omaha Nebraska. Pp. 171.

USEPA. (2020). *Control Measures for Cyanobacterial HABs in Surface Water.* Accessed at https://www.epa.gov/cyanohabs/control-measures-cyanobacterial-habs-surface-water on 1/10/2021

USEPA. (2018). *Water Treatment Optimization for Cyanotoxins Version 1.0.* Office of Water (MS-140) EPA 810-B-16-007. Washington, DC.

Upstate Freshwater Institute (UFI), Department of Biological and Environmental Science Cornell University (DBESCU). (2017). Phase II Final Report: A Phosphorus/Eutrophication Water Quality Model for Cayuga Lake, New York. Submitted by Upstate Freshwater Institute, Syracuse, NY. Ithaca, NY pp. 1-227.

Upstate Freshwater Institute (UFI). (2020). A Water Quality Model for Owasco Lake, New York. Prepared for New York State Department of Environmental Conservation, Lake Monitoring and Assessment Section 625 Broadway, Albany, NY. Submitted September 2020.

USGS. (2020a). Zebra Mussel. Accessed at https://nas.er.usgs.gov/queries/CollectionInfo.aspx?SpeciesID=5&State=NY on 11/20/20

USGS. (2020b). Quagga Mussel. Accessed at https://www.invasivespeciesinfo.gov/aquatic/invertebrates/quagga-mussel on 11/20/20

Vanderploeg, H. A., Liebig, J. R., Carmichael, W. W. et al. (2001). Zebra mussel (Driessena polymorpha) selective filtration promoted toxic Microcystis blooms in Saginaw Bay (Lake Huron) and Lake Erie. *Can J Fish Aquat Sci*, 58, 1208-1221.

Visser, P. M., Ibelings, B. W., Bormans, M., & Huisman, J. (2016). Artificial mixing to control cyanobacterial blooms: a review. *Aquatic Ecology*, 50(3), 423-441.

von Stackelberg, N. 2016. Utah Lake nutrient model selection report. Submitted by Utah Department of Environmental Quality, *Water Quality*. pp. 27.

Watkins, J. M., Rudstam, L. G., Mills, E. L., & Teece, M. A. (2012). Coexistence of the native benthic amphipod Diporeia spp. and exotic dreissenid mussels in the New York Finger Lakes. *J Great Lakes Res*. 38(2), 226-235.

Weenink, E. F., Luimstra, V. M., Schuurmans, J. M., Van Herk, M. J., Visser, P. M., & Matthijs, H. C. (2015). Combatting cyanobacteria with hydrogen peroxide: a laboratory study on the consequences for phytoplankton community and diversity. *Frontiers in microbiology*, 6, 714.

Wetzel R. G. (2001). Limnology: Lake and river ecosystems. 3rd ed. San Diego (CA): Academic Press.

Winter, J. G., DeSellas, A. M., Fletcher, R., Heintsch, L., Morley, A., Nakamoto, L., & Utsumi, K. (2011). Algal blooms in Ontario, Canada: Increases in reports since 1994. *Lake Reserv Manage*, 27, 107-114.

Wu, X., Joyce, E. M., & Mason, T. J. (2011). The effects of ultrasound on cyanobacteria. *Harmful Algae*, 10(6), 738-743.

Yeager, N., & Carpenter, A. (2019). State approaches to addressing cyanotoxins in drinking water. *AWWA Water Science*, 1(1), e1121.

Yuan, J. (2020) Taxonomic and Genetic Diversities of Cyanotoxin Production. *Corpus J Vet Dairy Sci*.1:1013.

Zhang, H., Culver, D. A., & Boegman, L. (2008). A two-dimensional ecological model of Lake Erie: Application to estimate dreissenid impacts on large lake plankton populations. *Ecol Model*, 214(2-4), 219-241.

Zhang, Z., Sun, B., Johnson, B. E. (2015). Integration of a benthic sediment diagenesis module into the two dimensional hydrodynamic and water quality model - CE-QUAL-W2. *Ecol Model*, 297, 213-231.

Zhu, S., Zhang, Z., Liu, X. (2017). Enhanced two dimensional hydrodynamic and water quality model (CE-QUAL-W2) for simulating mercury transport and cycling in water bodies. *Water*, 9(9),22.

Endnotes

1 Representative Concentration Pathways (RCP) are standardized changes in the planet's solar radiative forcing based on greenhouse gas concentrations used for climate change analyses.

2 The ClimAID data were published in 2014 and used a historical base period of 1971 to 2000. Therefore, the 2020 time slice represents a change from the historical baseline.

3 Note that cyanobacteria are not true algae.

4 Any recreational restriction determined by NYSDOH is usually initiated through visual observation of a bloom, regardless of whether the toxin concentration is known. Restrictions to access are only enforced at permitted swimming areas, not at other areas of lakes. Please see: https://www.health.ny.gov/publications/2849/index.htm

5 The ecology of natural lakes and artificial reservoirs functions similarly and are considered interchangeable in this report.

6 While cyanobacteria are not technically a true alga, the term harmful algal bloom (HAB) has been applied to them for many years. Cyanobacterial has been added to be more precise. Sometimes blooms are simply referred to as HABs, cyanobacteria bloom, or other similar terms.

7 Paerl & Barnard, 2020; Wehr et al., 2015; Paerl et al., 2001; Paerl & Huisman, 2008; Paul, 2008; Jöhnk et al., 2008; Sandrini et al., 2015; Nalley et al., 2018; Visser et al., 2016; Lürling et al., 2017; Bui et al., 2018; Maliaka et al., 2018; Carey et al., 2012; Sandrini et al., 2016.

8 Davis et al., 2009; Paerl & Barnard, 2020; Wehr et al., 2015; Paerl et al., 2001; Paerl & Huisman, 2008; Paul, 2008; Jöhnk et al., 2008; Nalley et al., 2018

9 Stratification is the process by which lakes form thermal layers during warm weather that limits mixing and the transfer of oxygen and nutrients between the layers.

10 Chang et al., 2020; Wehr et al., 2015; Carey et al., 2012.

11 Chang et al., 2020; Wehr et al., 2015; Carey et al., 2012

12 Information on the waterbody inventory and priority waterbodies lists can be found at https://www.dec.ny.gov/chemical/36730.html

13 The City of Syracuse has a filtration avoidance waiver on its Skaneateles Lake drinking water supply due to the lake's high quality.

14 Gal et al., 2009; Hipsey et al., 2006; LimnoTech 2016; USACE 2015; Zhang et al., 2008

15 Gelda & Effler, 2007; Gelda & Effler, 2008; Owens et al., 1998)

16 The maximum length of open water wind can travel over a lake surface.

17 Wind direction is based on where the wind is blowing from.

18 The box shows the 25th and 75th percentiles with the median and mean in the middle. The whiskers are 10th and 90th percentiles with outliers shown as dots beyond the whiskers.

19 The Julian day is a 365-day count starting at January 1st through December 31st.

20 Median values were also plotted and exhibited nearly identical patterns as average values for TP and Chl-*a*.

21 Green algae can flourish in high temperatures up to approximately 30°C.

22 Refer to NYS DEC's Harmful Algal Blooms (HABs) Program Guide Section 6 for a description of methods for bloom prevention and control that are allowable in New York State. https://www.dec.ny.gov/docs/water_pdf/habsprogramguide.pdf NY

23 Nutrient sequestration using chemicals including alum is currently not allowed in the State due to permitting issues with discharging chemicals to waterbodies, fish toxicity in low pH lakes, and challenges with floc/sludge removal (NYSDEC, undated).

[24] Weenink et al., 2015; Burson et al., 2014; Matthijis et al., 2011

[25] Toxins inside cyanobacteria cells.

[26] Toxins that have been released from cells into the environment.

[27] Activated carbon effectiveness varies based on type of carbon, pore size, type of cyanotoxin, and other water quality parameters, such as NOM concentration.

[28] CyanoTOX® is a desktop application to calculate oxidant requirements for removal of cyanotoxins and can be downloaded here: https://www.awwa.org/Resources-Tools/Resource-Topics/Source-Water-Protection/Cyanobacteria-Cyanotoxins

NYSERDA, a public benefit corporation, offers objective information and analysis, innovative programs, technical expertise, and support to help New Yorkers increase energy efficiency, save money, use renewable energy, and reduce reliance on fossil fuels. NYSERDA professionals work to protect the environment and create clean-energy jobs. NYSERDA has been developing partnerships to advance innovative energy solutions in New York State since 1975.

To learn more about NYSERDA's programs and funding opportunities, visit nyserda.ny.gov or follow us on Twitter, Facebook, YouTube, or Instagram.

New York State Energy Research and Development Authority

17 Columbia Circle
Albany, NY 12203-6399

toll free: 866-NYSERDA
local: 518-862-1090
fax: 518-862-1091

info@nyserda.ny.gov
nyserda.ny.gov