A Policy Legitimization for a New Water Rule: A Contextualization of the Unregulated Effects of Industrial Farming, Nutrient Runoff and Costly Surface and Drinking Water Treatments

by

Nicholas P. Guehlstorf, Ph.D.
Professor
Tel (618) 650-3852; Fax (618) 650-3509
nguehls@siue.edu
Department of Political Science and Environmental Sciences
Southern Illinois University Edwardsville

Emma MacNeill
Undergraduate Research Associate
emacnei@siue.edu
Environmental Sciences Department and Chemistry Department
Southern Illinois University Edwardsville

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Abstract:

Current agricultural practices, which involve winter and spring applications of nutrient-dense fertilizers, are being questioned by rural stakeholders, legally and politically. This industrial farming practice leads to high amounts of unregulated nutrient runoff which has an established relationship with the occurrence of algal blooms, degradation, and impairment of waters. Impairment threatens potable water availability for communities that rely on it as a drinking water source, leaving treatment facilities to consider expensive solutions. Toxic cyanobacterial harmful algal blooms (HABs) are threatening the security of freshwater. While the Clean Water Act (CWA) requires states to report impaired waterbodies, local decision-makers have no science-based framework for on-lake monitoring metrics. The case study for this project is a watershed that has experienced HABs and been legally impaired but is now encouraging precision/conservation farming and uses multiple technical devices to manage treatment. While the law only requires monitoring at a single site, the purpose of this research is to examine the water quality of a more significant portion of the lake and the health of the entire 12,897-acre watershed in Southern Illinois. Through monthly sampling at various locations, we show that nutrient concentration and algal bloom occurrence differ within the lake. These findings provide management techniques that encourage more sustainable farming, and small-scale, cost-effective point-treatments.

Key Words:

Environmental Public Administration, Farm Policy, Algal Blooms, Safe Drinking Water Act

1. Introduction

According to the EPA, more than 15,000 water bodies and over 100,000 miles of moving water are classified as impaired (Stade 2018). The impaired waterway determination comes when the water either fails to meet water quality standards (WQSs), total maximum daily loads (TMDLs), or is otherwise degraded and can no longer be used for drinking or recreational activities. In fact, the EPA estimates that the tourism industry alone loses over a billion dollars a year from restricted recreational activities, such as boating and fishing, due to legal impairment (US EPA 2014). Nutrients, specifically nitrogen and phosphorus, are essential for biological processes, including plant health and growth, and because of this, nutrients play a significant role in agriculture. To increase yields, standard industrial agricultural practices involve frequent and concentrated fertilizer applications. This practice routinely results in excess nutrient application which means that after nutrient uptake occurs, significant amounts of nitrogen, phosphorous, and ammonia remain on land where they are exposed to weather conditions, contributing to runoff and subsequent aquatic ecosystem and surface water degradation. Nutrient pollution has negative implications for aquatic systems, human health, and socio-economic issues, but the most important for this study is the association with algal blooms.

Algal Blooms

While the presence of both nitrogen and phosphorus from runoff can have oxygen-depleting effects in freshwater systems, phosphorus is a limited and growth-limiting nutrient, and as such promotes eutrophication (West et al. 2013). Eutrophication is the naturally occurring enrichment of surface waters with nutrients; however, large amounts of nutrients from runoff can amplify this enrichment and when combined with ideal climate conditions, can lead to algal blooms. Serious consequences that threaten potable water quality and availability can arise from nutrient pollution in drinking water supply reservoirs. Due to the manner in which environmental policy and regulation is implemented, this type of pollution occurs with few meaningful management policies, leading to environmental issues with complex solutions.

Algal blooms can lead to hypoxic or low oxygen conditions in waters (Coffey et al. 2019). The exponential growth of blooms also means decomposition of dying algae; this process requires large amounts of oxygen and effectively depletes levels in waters which can kill fish, organisms, and plants. The composition of algal blooms is mainly dependent on nutrient ratios, but they are caused by several organisms, including phytoplankton, benthic algae, macro-phytes, and cyanobacteria, also known as blue-green algae. Some cyanobacterial blooms produce toxins, known as harmful algal blooms (HABs), threatening ecosystem health, public health, and the availability of safe water. In the past, many lakes and coastal waters in states such as Florida and Louisiana have seen HABs almost annually, with some being affected every year for months at a time (Florida Department of Health n.d.; Louisiana Department of Health 2019).

The warm and humid climate of southern states has been a commonly accepted explanation for frequent algal blooms; however, blooms are increasing in frequency and geographic distribution. States such as Wisconsin, Minnesota, and Ohio, which historically have not seen harmful blooms to an extreme, have recently experienced an increase in HABs in their lakes and streams, degrading water quality. Wisconsin experienced a widespread algal bloom in Lake Superior in the summer of 2018, and in 2014, northwest Ohio was severely impacted by a HAB in Lake Erie, which left more than 400,000 people without drinking water for three days (Tanber 2014). These increases, especially in states that have previously been less likely to experience HABs in the past, are evidence of a more significant nutrient pollution problem occurring throughout the country.

Legislation and Current Implementation: Clean Water Act

The purpose of the Clean Water Act (CWA), passed in 1972, was to restore waters and remediate previous damage. The CWA was passed with opposition and a fair share of controversy centered on the cost of implementation. The Office of Management and Budget completed a cost-benefit analysis on the CWA, specifically on surface waters, and found that the cost of regulation significantly outweighed the resulting benefits; however, the EPA concluded that some of the benefits are unmeasured (Keiser, Kling, and Shapiro 2019; Van Houtven, Brunnermeier, and Buckley 2000). This Act and its subsequent Amendments targeted industrial pollution sources, or point sources, that emit from a localized and established point and can easily be tracked to the source. Point source nutrient pollution, often from industries and other facilities, is regulated by the Clean Water Act (CWA) through a system of permits known as the National Pollutant Discharge Elimination System (NPDES). This regulation for point source pollution is not considered successful as it does not reduce pollution, rather it addresses problems through management technologies. However, it has achieved more success than attempts to regulate nutrient overload from non-point sources. (Liu, Bruins, and Heberling 2018). The success of the CWA is difficult to measure because the U.S. continues to have a significant number of waterways failing to meet WQSs. This is at least partially a result of non-point source pollution issues.

The almost fifty-year regulatory strategy of focusing on point source pollution ignores the larger sources of pollution: non-point sources. Non-point source (NPS) pollution resulting from agricultural and land management practices is a particular challenge to regulators. The nutrients “run off” from agricultural lands at many points, unlike the nutrient loads from municipal and industrial point sources. NPS pollution is not regulated by the CWA and continues to be a significant issue facing bodies of water in the United States (Reimer, Denny, and Stuart 2018).

Safe Drinking Water Act

The SDWA, passed in 1974, was a governmental attempt to better protect potable water quality, since many other earlier water policies failed to do so. Before the passage of the SDWA, drinking water quality was considered a state issue with no Federal guidelines and no local or community involvement. This resulted in inconsistent regulation and quality of drinking water, especially in rural communities. It was not until the early 1970s that the severe gaps in water quality regulation were examined and determined to be a threat to human health. In the years following the passage of the SDWA, significant amendments occurred into the late 1990s which stressed risk management, focused on pollution prevention (rather than abatement) and a comprehensive cost-risk-benefit assessment. Specifically, the 1996 amendment set forth a requirement for the EPA to determine the best management practices (BMPs) for the assessment and the protection of drinking water sources (Zarkin 2015). In addition, states were required to implement their own programs to monitor drinking water sources and were encouraged to implement programs at a local level to strengthen pollution prevention practices.

The 1996 amendment to the SWDA also focused heavily on ensuring public notification and participation, as well as encouraging education. The amendments introduced provisions that would require public notification about water quality, including violations, as well as implementation efficiency at the local and state level (Humphreys and Tiemann 2021). The purpose was to make information available to the public in hopes of encouraging participation and compliance. The amendment also addressed incorporation of risk-based science into water contaminant regulation. In addition to the language addressing National Primary Drinking Water

Regulations (NPDWRs) and the Maximum Contaminant Levels (MCLs), the 1996 amendment introduced the Unregulated Contaminant Monitoring Rule (UCMR).

The purpose of the UCMR is to initiate the monitoring of contaminants of possible concern to determine if an NPDWR is necessary to protect public health. Currently, the UCMR is the only way that any regulatory Federal monitoring for cyanotoxins is achieved.

The SDWA’s regulations apply to publicly and privately owned systems that provide piped water to at least 25 people regularly (Humphreys and Tiemann 2021). For the fifth UCMR, the program requirements were altered to require all Public Water Systems (PWSs) that serve between 3,300 and 10,000 people to monitor during particular UCMR cycles, and a representative sample of small PWSs serving 3,300 (EPA 2021; Office of Water 2020).

<u>Otter Lake Case Study</u>

Many Midwestern states with a large agricultural industry, including Illinois, are significant contributors of nonpoint source phosphorus, thus the need for increased research to understand progress toward the goals established by the Illinois Environmental Protection Agency (IEPA) under the Illinois Nutrient Loss Reduction Strategy (NLRS). In order to examine the challenges with the CWA, NPDWRs, UCMR and contemporary issues of water impairment, algal blooms, and cyanotoxins, a pilot waterbody was selected for examination. The case study for this project is the Otter Lake Watershed, a 12,897-acre watershed in Southern Illinois generally and rural Macoupin County, specifically. Since its formation in 1967, the Otter Lake Water Commission (OLWC) has expanded to provide drinking water to eight rural towns and villages and two rural water districts across three counties (Northwater Consulting 2018). In 1996 and 2002, algal blooms created regulatory impairments to recreational activities and drinking water supplies, including a continued impairment designation by the IEPA until 2004.

Starting in 2006, and lasting six years, the OLWC was unable to provide water for Public and Food Processing due to impairments relating to mercury, manganese, total phosphorus, and aquatic algae levels (Northwater Consulting 2018). While the manganese and aquatic impairments were addressed in 2014, the lake is still considered impaired for mercury and total phosphorous standards.

Given that phosphorus is a significant nutrient source for algae, it is unsurprising that Otter Lake has observed significant blue-green algae counts in recent years; testing frequently reveals high counts of the cyanobacterial compounds 2-methylisoborneal (MIB) and geosmin in the water. These compounds have effects on taste and odor within the water supply, leading to increased treatment expenses and consumer complaints. While maximum concentrations of total phosphorous have decreased since 2010, minimum concentrations have remained consistent and routinely exceed Illinois's water quality standard. The highest total phosphorous values occur between July and October, exceeding the standard 73% of the time (Northwater Consulting, 2018). It is helpful to note that the water is more likely to be impaired during this timeframe, but more specific data about the entire lake at different depths in needed to monitor any of the non-point source pollution comprehensively. Since Otter Lake is used as a drinking water supply for eight rural towns and two rural water districts, understanding the relationship between unregulated chlorophyll *a* (Chl *a*) and levels of phosphorus and nitrogen in surface water is a valuable advancement of scientific knowledge. In fact, the concentrations of pollution throughout the lake and data correlations may have practical implications for Otter Lake.

This paper seeks to further understand the correlation between current agricultural practices, which yield unregulated amounts of nutrient pollution, and policy treatments for surface waters. In addition, this paper examines the impact of NPS pollution on the quality of drinking water that can be provided to local communities, while looking for patterns of impairment to suggest possible small-scale point-treatments to improve water quality in a cost-effective manner.

2. Literature Review:

The monitoring of surface water quality is an action currently required by Federal and State law. This action becomes even more important when that water is the primary or initial drinking source for communities. However, multi-locational monitoring or analysis over time is not required by Federal or State law. Current natural science research into the impacts of agricultural practices on surface water quality often focus on the presence and number of pesticides, herbicides, and components of fertilizer, specifically nitrogen and phosphorus. While this approach is important for the determination of contamination, it does not always provide insight into probable and possible techniques to implement surface water pollution controls. In contrast, social science research on water facilities has largely focused on the ability to provide water, periods of non-compliance, and the ineffectiveness of the CWA and the SDWA. This approach frequently ignores recent scientific data and focuses on the policy aspect of water quality. While policy research is both necessary and beneficial, the current approach seems to ignore or dismiss realistic and affordable water management solutions. The gaps created by the focus on either natural or social science research has allowed for lapses in water policy that has resulted in an adverse effect on surface waters, the reliant communities, and the environment.

In 1998, Congress passed the Harmful Algal Bloom and Hypoxia Research and Control Act to address the issue of Harmful Algal Blooms (HABs) in ocean and coastal waters. Included in this legislation was a national research plan to understand the causes and impacts of HABs in coastal areas to improve the likelihood of successful management and mitigation. This legislation is specific to coastal waterways and does not apply to land locked waterways (National Plan for Algal Toxins and Harmful Algal Blooms 2005). A companion plan is needed for freshwater, as many communities rely on these sources for tourism, food, and drinking water. A large number of freshwater water sources throughout the United States are compromised by HABs. (Notable regions with compromised water sources include the Great Lakes, the fresh and brackish water portions of the Gulf of Mexico, the entire Western Coast, and the top half of the East Coast (Environmental Protection Agency 2003; National Plan for Algal Toxins and Harmful Algal Blooms 2005; U.S. National Office for Harmful Algal Blooms n.d.; Watson et al. 2016). However, there are also a significant number of smaller lakes across all 50 states that have untreated HABs.

Due to the lack of legislation regulating freshwater, many water managers rely on an outdated 1990 assessment strategy from the World Health Organization to mitigate HAB risk (National Plan for Algal Toxins and Harmful Algal Blooms 2005). Since impaired freshwater results in no legal ramifications, many water managers react after a bloom has formed which typically results in crisis management over crisis preparedness. Lack of data for each waterway to be protected is also a limitation, as we often have no baseline data for the health of an aquatic system. A typical scenario is after the freshwater source becomes compromised by a HAB, consumers report a foul taste or smell from their water. The water disbursement agency will then send a water sample to a testing facility. Testing can take anywhere from three to four days, during which time the water is still being distributed. If a bloom is detected, the governing agency will release a “Boil” or a “Do Not Drink” order. However, the contaminated water has already been distributed to residences as potable water, to industry as non-potable water, and to local farming industries as water for their livestock and crops.

Risk management practices are currently informed by bloom frequency, the dose-toxin relationship, and budget-effective management solutions (Freeman 2010; Holsinger et al. 2015). This information currently is deemed by the EPA as insufficient to influence environmental policy and to implement new regulation for Freshwater HABs (Holsinger et al. 2015). Much of the scholarship, as seen in the following literature review, is focused on how agricultural practices impact surface water quality, and less on how to better predict periods of impairment for treatment facilities.

Natural Science Literature:

Harmful Algal Blooms (HABs) consist predominately of blue-green cyanobacteria but can also include diatoms and dinoflagellates. These blooms are harmful when growth occurs rapidly in aquatic ecosystems and can harbor dangerous and toxic bacteria that affect human and animal health, aquatic ecosystem health, and aquatic economics. HABs produce toxins that permeate the water and travel through the ecosystem, infecting fish, shellfish, plants, and animals. They are typically caused by nutrient loading, in which an aquatic ecosystem experiences a large influx of phosphorous and nitrogen. Over time, water systems experience a gradual increase in these nutrients and will cycle through eutrophication (Watson et al. 2016). When nutrient runoff pollutes an aquatic system, this eutrophication cycle can be amplified. In warmer water temperatures, this combination can result in an algal bloom. If consumed, HAB's can cause adverse effects on humans, livestock, and pets, and in some instances, can cause death. While not all algal blooms are toxic, they are a major concern for water managers globally.

Researchers in Mauritania studied the seasonal occurrence of cyanobacteria and microcystins within the Foum-Gleita Reservoir (Sadegh et al. 2021). This research involved examination of not only limnological factors but also biological and environmental factors including nitrogen and phosphorus concentrations, water pH, and water temperature. The research was completed over 11 months with regular sampling at various depths, allowing for a more in-depth analysis of cyanobacteria both throughout the year and the water column (Sadegh et al. 2021). The researchers concluded that toxic cyanobacteria were found in higher concentrations closer to surface level, likely due to the amounts of available sunlight, and were positively correlated with warmer temperatures (Sadegh et al. 2021). While these findings are not unexpected based on previous understanding of algae growth; the sampling occurred during different weather patterns throughout the year which provides more insight into the previous understanding of algae growth. When it relates to nitrogen, phosphorus, and pH, the associations were dependent on the species of cyanobacteria, but for many of these factors there is a positive correlation with growth with toxic cyanobacteria and warmer temperatures. These findings are supported by research done on Lake Erie (2019) and in the East African Rift Valley in Lake Tanganyika (2007) (Jankowiak et al. 2019; Wever et al. 2007). More research is needed to identify the relationships between temperature, nitrogen, phosphorus, pH, and the occurrence of algal blooms. While some of these studies examined microcystin relationships, they did not provide useful solutions other than more frequent sampling. As such, our sampling design includes surface and one-meter depth sampling, at multiple sites, multiple times a year, and for numerous contaminants.

The protection of water quality in lakes, streams, and reservoirs is a difficult task due to many factors, many of which are anthropogenic. Predicting how anthropogenic factors affect a water system is difficult, largely due to a lack of understanding of their impact. For this reason, modelling techniques continue to be developed to help researchers, and eventually watershed managers, better predict periods of impairment and future risks due to changes in land-use and ecological factors (Erol and Randhir 2013; Rousso et al. 2020). Researchers studying Lake Egirdir, a freshwater lake in Turkey significantly impacted by both nonpoint and point source pollution, turned to ecological comprehensive modeling techniques in order to better protect the water system (Erol and Randhir 2013). Two approaches were used to develop the model. The first involves using geographic information system (GIS) mapping to spatially analyze how pollution moves through the water system, and the second examines the effectiveness of the best management practices (BMPs) in place for both point and nonpoint pollution sources (Erol and Randhir 2013). The conceptual model includes water quantity, water quality, watershed characteristics, and weather factors to create a baseline model of the water system (Erol and Randhir 2013). The empirical modeling involves simulating the watershed and altering point and nonpoint pollutions and examining how differing loads impact the system (Erol and Randhir 2013). Through modeling, researchers were able to predict how factors will impact the health of the watershed and were able to provide BMP recommendations (Erol and Randhir 2013).

The modeling of cyanobacterial harmful algal blooms (cyanoHABs) is becoming more common. Widescale adoption of this approach would allow publicly owned treatment works (POTWs) to more accurately predict the occurrence of cyanoHABs and better choose appropriate treatment and management techniques before they start. Both process-based (PB) and data-driven approaches to successfully model cyanoHABs have been used (Rousso et al. 2020). Data-driven models are frequently used as short-term predictors and are more commonly used, whereas process-based are typically used for long-term predictions (Rousso et al. 2020). Some are capable of differentiating species, some in 1D, 2D and 3D, some look at anthropogenic factors, and others look at water movement (Bruce et al. 2006; Ibelings et al. 2003; Rousso et al. 2020). Data-driven models include regression and indexes, artificial neural networks (nonlinear regressions), decision trees, genetic programming, and Bayesian networks (Rousso et al. 2020). Each of these approaches has strengths and weaknesses, which is why they are frequently used together. Researchers have found that using regressions and indexes works well when examining a number of outside factors and the combination is useful for showing correlations (J. Liu and Fang 2017; Rousso et al. 2020). The use of Bayesian probability models on well-studied water systems to examine the probability that an event may happen can also help in future management (Bertani et al. 2016; Rousso et al. 2020).

Modeling has the possibility to be a powerful tool in the POTW arsenal if proper sampling and monitoring are completed. When modeling relies on statistics and is conducted without proper data input from sampling and monitoring, the model will eventually become inaccurate and unable to aid in the prediction and pretreatment of HABs (Sheng et al. 2012). Current research supports the benefits of modeling but argues for stronger and more standardized monitoring in order to create a modeling system that can be applied to the majority of water systems (Erol and Randhir 2013; Rousso et al. 2020; Sheng et al. 2012).

Social Science Literature:

The other approach to examining surface water impairment comes from policy analysis: the examination of laws and policies to determine if they are effective at protecting surface waters as they are currently implemented. This approach often uses past scientific data, surveys, compliance, and economic assessments of how much consumers are willing to pay. Many researchers have pointed out the flaws in the SDWA and CWA, especially for small towns (Marcillo and Krometis 2019). The Harmful Algal Bloom and Hypoxia Research and Control Act, which gave the National Oceanographic and Atmospheric Administration (NOAA) authority to spearhead a plan to mitigate the Red Tide affecting the Gulf of Mexico. The plan established a task force which, 12 months from its inception date, would submit two separate environmental impact reports to Congress about HABs and to provide subsequent assessments each year about Hypoxia. Again, this Act specifically states it is only for coastal waters, Federal regulatory agency does not supersede state authority, and Federal enforcement authority is limited to what was already in the CWA.

In 2004, an amendment changed the frequency of the assessments and included the Great Lakes and all US freshwater locations. However, this change did not gain traction as the regulatory body for national water sources is the EPA and as of current the EPA has not begun to create a risk management or assessment plan for Freshwater HABs. Under the CWA and the SDWA, the EPA is responsible for ensuring that potable water is safe for human consumption and therefore must be the one to initiate a plan for Freshwater HABs. In 2005, enforcement authority for Coastal HABs was increased through the National Plan for Algal Toxins and Harmful Algal Blooms which addressed critical needs of the United States coastal waters by providing grants of study for "bloom ecology and dynamics; toxins and their effects; food webs and fisheries; and public health and socioeconomic impacts" (National Plan for Algal Toxins and Harmful Algal Blooms 2005). The funding from these grants allowed development of technology, models, and regulating equipment that successfully predicted the Red Tide in Maine in 2008, minimizing the economic loss of contaminated shellfish and providing residents early warning to be careful during this time period (Lippsett and Carlowicz 2008).

The SDWA requires the EPA to develop a list of contaminants that may occur in public water systems every five years, but these listed contaminants are not currently subject to regulations. Since the passage of the Harmful Algal Bloom and Hypoxia Research and Control Act, multiple strains of cyanobacteria, such as microcystin, have been added to the list as microbes of interest; however, no policy or regulatory determinations have been made. In 2015, the EPA published “Recommendations for Public Water Systems to Manage Cyanotoxins in Drinking Water,” however the document plainly states, “This document is not a regulation; it is not legally enforceable; and it does not confer legal rights or impose legal obligations on any party, including EPA, states, or the regulated community” (EPA 2012). There are currently no Federal regulations or programs for Freshwater HAB management. Marcillo and Krometis (2019) found that very small and isolated rural area community water systems (CWSs) experienced more frequent noncompliance with the SDWA than larger water systems. Relying on data from CWSs to determine water quality can lead to mixed results in terms of whether the policy is effective at meeting its intent, and this frequently is a result of under- and misreporting of sampling results, periods of contamination, and periods of impairment. The reliance on data from CWSs is a limitation of many of the policy evaluations and research currently published today. Environmental scholars assert that this noncompliance likely allows for harmful contaminants to enter water systems unnoticed, potentially posing a concern to human health (Marcillo and Krometis 2019; Oxenford and Barrett 2016). It is important to note that the SDWA does include provisions to help water systems reach compliance, but frequently rural and small CWSs are unable to complete requests for funding, and any funding received is often insufficient to manage issues faced by small CWSs (Marcillo and Krometis 2019; The United States Environmental Protection Agency 2017).

Other policy-focused research studies the methods by which implementation of remediation techniques occur. David Switzer, from Florida Atlantic University, studied the levels of implementation in different areas for CWSs (Switzer 2019). Switzer determined that violations occurred more frequently in Republican leaning areas than Democratic, suggesting that politics have an influence on policy implementation and the protection of water quality (Switzer 2019). This influence of politics on policy implementation and water quality protection is hard to examine but relevant to the case study for this paper.

Locally in Illinois, Otter Lake has received impairment status from the EPA multiple times. This lake, predominately surrounded by rented farmland, experiences excess nutrient loading during the growing season. The sedimentation is so severe that the depth levels of the northern parts of the lake have changed by 10 feet in less than half a century. This suggests significant amounts of runoff through the year; however, the EPA only requires one site to be tested and only an annual average is reported. This methodology does not reflect the threat posed by the cyclical nature of algal blooms, the growing season, or environmental factors.

Annual reporting may not protect consumers and is problematic as the Water Commission determines if additional testing is required to keep the lake safe (and then must pay for it). Instead of finding the willingness to pay thresholds of the watershed, the Otter Lake Water Commission took the approach of requesting EPA grants for financial assistance in equipment and conservation efforts to manage run-off and clean-up the nutrient pollution. An assessment by a consulting group, Northwater Consulting, concluded cleaning up and stabilizing the lake would cost $8.5 million in order to reach and remain unimpaired (Northwater Consulting 2018). This estimated cost exceeds what the commission has or can receive in risk management grants from the EPA. These factors are why Otter Lake was chosen as a case study for developing a test for new water rules for freshwater sources in the United States. Current literature acknowledges the cost barriers which make it difficult to provide necessary tools to improve water quality. There are advantages and disadvantages to using solely statistical, modeling, and direct water analysis. This research aims to attempt to find a median between different monitoring techniques to provide cost-efficient and manageable solutions so that water facilities are equipped with tools to best manage water. Changes within the industrial agricultural practices are required through policy changes; because without it, nutrient loading will not decline. The reluctance and inability to force these changes pushes the burden of protecting surface waters onto treatment facilities, making it even more important that they are properly equipped with the best tools to do so.

3. Methodology and Hypothesis:

The purpose of this study is to construct a multi-month timeline to show how the concentration of nutrient pollution and chlorophyll changes throughout the year. If the testing reveals a strong correlation between chlorophyll *a* and increased levels of phosphorous and nitrogen, there are strong policy and public health implications since the question of whether a treatment facility can effectively and economically treat a source of drinking water for cyanobacteria toxins is an essential one. If the water cannot be treated, a "Do Not Drink" order is required until the nutrients and algae are controlled. Our questions are as follows:

> *H (1): The occurrence and prevalence of nuisance algal blooms* (chlorophyll *a* >20 µg) *has a direct correlation with nutrient run-off* (total Kjedahl nitrogen (TKN), total Kjedahl phosphorus (TKP)) *stemming from excessive fertilizer application practices and weather conditions.*
>
> *H (2): Comprehensive testing (*more than 30 lake sites) *can be used to allow POTWs to predict and manage HABs because there are significant differences of pollutants (*ammonia, nitrite, nitrate) *by location of lake and time of year*.

This research is designed to provide scientific evidence to support watershed-level decision-making on treatment strategies and how drinking water quality and availability is impacted.

To test this first hypothesis, it was necessary to make this sampling as comprehensive as possible; to do so, geospatial techniques were used to determine an initial 33 sampling sites throughout the second portion of the lake, where the water is used for drinking (Appendix A).

Following the input from a solar-powered stratifier, an additional site was added as site *34*, close to site *33*.

Tests were conducted monthly from March – December 2021. At all sites, 1 L samples were collected in HPDE bottles at surface level, using the grab technique. At sites *1, 14, 16, 22, and 27*, additional 1 L samples were collected at 1-meter depth using a Van Dorn water depth sampler. These five sites also had one 50 mL sample collected in total for suspended solids (TSS) determination, and three 500 mL samples were collected in amber glass bottles for pesticide determination. All samples were kept on ice until return to the laboratory. During sampling, field data was collected at every site: turbidity was determined using "Hanna instruments" with Fast Tracker Technology (Model HI98703); pH, temperature, conductivity, and relative dissolved oxygen (RDO) was collected using *Thermo Scientific Orion Star A* following the procedures provided in the manual.

Following return to the laboratory, samples were split and preserved according to their individual procedures. A total of nine tests were completed with these samples: total Kjedahl nitrogen (TKN), total Kjedahl phosphorus (TKP), ammonia, nitrite, nitrate, total phosphorus (TP), and chlorophyll. First, the samples were filtered and stored for chlorophyll determination following EPA Methods 445, 446, and 447. Following filtration, the rest of the samples were preserved according to their methods. TKN (Quikchem Method 10-107-06-2-P), TKP (Quikchem Method 10-115-01-2-C), ammonia (Quikchem Method 90-107-06-3-A) and TP (EPA Method 365.3) were all preserved in acid while nitrite and nitrate (Quikchem Method 90-107-04-2-A), were preserved in base. TSS is to be completed within seven days according to the EPA Method 1684 and no preservation occurs. All liquid samples were then stored at 4 ºC and the filters for chlorophyll were stored at -20 ºC.

All analysis was completed according to the methods listed for each test and can be found in the Appendix. The TKN and TKP samples went through a sulfuric acid digestion to convert all forms of nitrogen and phosphorus into ammonium sulfate. The presence of the ammonium ion was determined using colorimetry. These digested samples were then analyzed using a Flow-Injection Analyzer (FIA) which uses wavelength filters to determine the concentration of ammonium ions. This instrument is also used for ammonia, nitrate, and nitrite analysis; however, these samples do not need to be digested and instead are ready for analysis following preservation. The chlorophyll determination technique was completed using Ultraviolet-Visible spectrophotometry, fluorescence spectrophotometry and high-performance liquid chromatography coupled with photodiode-array detection (HPLC-PDA). TP was completed using UV-Vis spectrophotometry. These results were then compiled into excel and "R Studio" where statistical analyses were completed.

4. Results:

Hypothesis One:

To test the first hypothesis (*H(1)*), a correlation analysis was used to test the relationships between the various physical, chemical, and biochemical variables collected from water samples. For the purposes of this study, the concentration of chlorophyll *a* was most important and was, therefore, considered the dependent variable (DV). Other variables that were measured from each of the water samples and which were included as independent variables (IVs) include, TKP, TKN, nitrate, pH, turbidity, RDO, and water temperature. Table 1 is a summary table that shows the correlational results for Chl *a* levels and the independent variables for testing conducted March through December of 2021.

Table 1. *Chlorophyll a correlations summary table for March-December 2021.*

	Chlorophyll *a* correlations									
IVs	**March**	**April**	**May**	**June**	**July**	**August**	**Sept.**	**Oct.**	**Nov.**	**Dec**
TKP	0.23471	-0.2201	-0.0253	0.26195	0.57612	0.2071	0.03927	0.51887	0.14646	-0.0774
TKN	0.46987	0.50395	-0.0443	0.69967	0.41496	0.10442	0.32309	0.53734	0.28383	0.39098
Nitrate	0.3249	-0.4118	-0.2518	-0.3056	0.49469	-0.1475	-0.0938	-0.1427	0.55592	-0.2041
pH	0.60572	0.55184	0.27659	0.56273	-0.0805	0.29424	0.40767	0.43278	0.48242	-0.5385
Turbidity	0.69878	-0.2982	-0.0771	0.57081	0.58939	0.28251	-0.2100	0.61240	0.62992	-0.0627
RDO	0.83952	0.47904	0.14475	0.70084	-0.0597	0.08470	0.19675	-0.1629	0.79873	-0.1711
Temp	NA	0.17659	NA	-1	0.83589	NA	NA	-0.0499	-0.6469	0.12284

The correlation analysis shows that, for March, Chl *a* concentrations had significant positive correlations of 0.8395, 0.6988, and 0.6057, with RDO, turbidity, and pH respectively (Table 1). These relationships are not as strong in April. The correlation between pH, RDO, and TKN remains positive, but with the exception of TKN, is weaker. The relationships continue to become weaker in the month of May, with only pH and RDO being weakly positively correlated. It is important to note that during the months of March-September, water temperature readings were not taken at every site. At the time of sampling during June, TKN's relationship with Chl *a* is stronger than previous months at a value of 0.6997. RDO also has a strong positive relationship, although less than what was seen in March, at 0.7008. In July, there is a strong positive relationship between temperature and Chl *a* concentrations, and slight positive relationships between Chl *a* and TKP, TKN, nitrate, and turbidity. August samples showed low positive correlations with all variables apart from nitrate.

During the fall season, September to November, the correlations remained relatively constant with a weak positive relationship between pH, but the remaining variables had different relationships between the fall months. In September, TKP had a correlation coefficient of 0.03927, while in October the relationship was stronger, although moderate, at 0.51887. In both October and November, Chl *a* had a negative relationship with temperature, but the relationship was much stronger in November at -0.6469. The full correlation results for all ten months can be found in the Appendix. These results show all of the relationships between the variables. Many months saw high correlations between nitrate concentrations and the turbidity of the water (Appendix B).

A time-series analysis using a Type III Sum of Squares estimation technique for fixed effects indicated that, not only was there a significant temporal effect, but also TKN, nitrate, pH, turbidity, and RDO were significant predictors of Chl *a* concentration. Although multicollinearity amongst the predictors makes it inappropriate to interpret the coefficients of the predictors, the inclusion of these terms in a model led to decent predictions of Chl *a* (R^2 = 70.2%). The relevant F values of correlation for this time-series analysis can be seen in Table 2.

Table 2. *Time Series Analysis of Variance (ANOVA)*

	Sum Sq	Mean Sq	NumDF	DenDF	F value	Pr(>F)
TKP	230.3083	230.3083	1	355.9512	0.627414	0.428833
TKN	10775.91	10775.91	1	358.8039	29.35614	1.11E-07
Nitrate	7130.209	7130.209	1	354.8937	19.42438	1.39E-05
pH	2860.277	2860.277	1	221.2477	7.792073	0.005706
Turbidity	4615.742	4615.742	1	293.0768	12.57437	0.000455
RDO	1518.969	1518.969	1	351.5972	4.13803	0.04268
Date	115844.2	12871.58	9	328.3194	35.06522	4.1E-43

Quite simply, this means that Hypothesis One, which predicts a *correlation with nutrient run-off* (*organic nitrogen and phosphorus*), appears to be accurate but is very dependent upon month (Table 1). Furthermore, it appears that while there is a moderate relationship with TKP and Chl *a* in July and October, this relationship is not as consistent as the relationship between TKN and Chl *a* (Table 1). This is further verified by the time series ANOVA, which indicates that TKN ($p < 0.01$), is much more important at predicting Chl *a* concentration than in TKP ($p = 0.43$), although the multicollinearity between the IVs in undoubtedly inflating the latter p-value and necessitates that the coefficients of the IVs should not be interpreted directly.

Table 3. *Average chlorophyll a concentrations per sampling trip for March through December 2021.*

Sampling Month	**Chl *a* Concentrations (μg/L)**
March	9.4387395
April	91.398517
May	78.5855017
June	61.512187
July	53.0481658
August	65.1693906
September	67.6435644
October	44.7834373
November	18.2976415
December	9.29887669

In addition to site-based correlational analysis, examining an aggregate interpretation of the overall behavior of the lake was done. To do this, the chlorophyll *a* averages per month, seen in Table 3, should be considered. From Table 3, a general pattern is that concentrations increase in late spring, summer, and fall, and decrease during late fall, winter, and early spring. March had an average of 9.44 μg/L, while April had an average of 91.40 μg/L. These concentrations remain higher in the summer, with June, at the time of sampling, having a concentration of 61.51 μg/L. This level stays relatively consistent for the remainder of the summer, peaking at 67.64 μg/L in September before steadily dropping to 9.30 μg/L in December.

Hypothesis Two:

To test *H(2)*, it was necessary to select sites that differ spatially. For this, sites *16*, *32,* and *33* were selected. *Site 32* is the furthest North site and is close to the overflow, where water enters from the upper portion of the lake. *Site 16* is within the main channel, about halfway between the other two sites. *Site 33* is the closest site to the water intake that was tested all ten months. From here, the concentrations of Chl *a*, phosphorus (TKP), nitrogen (TKN and nitrate), were compared along with pH, turbidity, and conductivity. These results can be seen in Table 4.

Table 4. *Monthly concentrations and field results for Sites 32, 16, and 33.*

Month	Site #	Chl *a* (µg/L)	TKP (mg P/L)	TKN (mg N/L)	Nitrate (mg N/L)	pH	Turbidity (NTU)*	RDO (mg/L)
March	32	15.41378	0.248	0.667	0.784	9	17.1	15.2
	16	11.35499	0.262	1.052	0.265	8.69	8.79	13.9
	33	4.061408	0.323	0.646	0.231	8.45	NA	12.4
May	32	46.7457	0.087	1.38	4.4528	8.91	23.4	13.5
	16	50.05855	0.052	1.563	2.2844	8.85	13.5	10.16
	33	87.23741	0.213	2.243	1.8836	8.43	10.9	9.96
July	32	81.74301	0.174	1.53	2.181	9.35	12.2	32.2
	16	58.21263	0.092	1.574	0.872	9.39	8.3	30.1
	33	47.8538	0.17	1.884	0.709	9.29	6.57	29.9
October	32	107.8614	2.17	7.758	0.0753	8.89	14.5	24.3
	16	30.02693	0.058	0.912	0.0839	7.85	7.09	23.5
	33	53.85615	0.022	0.661	0.094	7.69	10.2	23.2
December	32	8.735366	0	0.759	1.0027	7.92	15.9	10.93
	16	7.435258	0.067	0.771	0.6903	8	10.8	10.36
	33	12.75031	0.107	0.864	0.83	7.24	14.7	10.01

*Nephelometric Turbidity Units

In comparing the sites within the same month, variation is evident. In May, *site 32* had 46.7457 µg/L of Chl *a,* while *16* had 50.05855 µg/L and *33* had 87.237 41 µg/L (Table 4). This variation in Chl *a* continued between months as site *32* in October had 107.8614 µg/L, while *16* and *33* had 30.02693 µg/L and 53.8615 µg/L respectively (Table 4). The implication of the acceptance of the second hypothesis is that *Comprehensive testing (*more than a single sites) *can be used to allow the Water Board to predict and manage HABs because there are substantial differences of pollutants (*ammonia, nitrite, nitrate) *by location of lake and time of year*.

This variation was present within nitrate concentrations, with all months excluding October, having higher concentrations at *site 32* when compared to *16* and *33*. From Table 4, it is evident that pH largely remained consistent throughout the lake, but turbidity differed between sites and between months. While additional statistical analyses (e.g. multivariate classification techniques) are needed, the take home message of this study and its two hypotheses is that water quality is dependent upon both spatial and temporal factors, both of which are not always accounted for in current water quality monitoring requirements. These results explicitly demand the need for legislation on and about freshwater algal blooms in agriculturally rich Mid-Western states. The rationale of this study was to examine different monitoring techniques that provide cost-efficient and manageable solutions so that water facilities are equipped with the tools to best manage water.

5. Discussion:

The data shows significant relationships between concentrations of Chl *a* and nutrients. These relationships have stronger correlations in April through October. This time frame is understandable due to the application of nutrient-dense fertilizer and increased rates of precipitation in early spring, which then lead to the summer months where the temperatures are ideal for growth. Then in fall, as temperatures cool, the upper surface water cools and becomes denser which causes a lake turnover. This turnover results in mixing of the water column and brings nutrients, that had settled to the bottom, up to the surface, leading to algal blooms. This can be seen in the monthly Chl *a* averages, as April saw significant Chl *a* concentrations leading into the summer, as well as high concentrations in September and October, when the water column is mixing. The decrease in Chl *a* concentrations in July may be explained by the time frame from when a bloom occurred to when sampling was completed. The OLWC recorded an algal bloom a few days prior to sampling, this could mean that a significant algal die-off occurred, and therefore resulted in lower Chl *a* concentrations. The OLWC was able to continue to supply water during this time and it is unclear if the bloom was reported to consumers.

In addition to the monthly averages and correlations, the location of sampling has a significant impact on the amount of nutrient contamination and Chl *a* levels. This is especially seen April through October (Table 4). This is likely due to several factors, including, closeness to shore, depth, amount of water movement, and the nearby environment. Closeness to the shoreline means that the site is a more ‘direct’ target of nutrient runoff, and the shallower a site, the less variability due to temperature and less separation of contamination. The amount of water flow plays a significant role in algal growth as algae prefer stagnant water. The nearby environment has impacts on the type of pollution as well as the amounts that enter the water, the effects from roads, cattle, and cropland, are different. The findings of this research strongly suggest that these factors influence algal blooms. It was found that nutrient pollution and Chl *a* concentrations were higher closer to shorelines, in more shallow water and lower flow rates.

While the OLWC currently has some BMPs in place, like stone retaining walls to reduce erosion and water stratifiers throughout the lake, the lake still suffers from high amounts of nutrient runoff and algae. However, when looking at the annual water quality reports from the OLWC, this is not made clear. Currently, the IEPA requires POTWs to test water quality at one location of the lake every month, however the reported value can be the annual average. The findings of this research show that the concentrations of nutrients and chlorophyll a can differ greatly throughout the year, and therefore, the reporting of a single annual average does not give an accurate comprehensive representation of the water quality of a lake.

The prevalence of HABs and continued water impairment in the United States reveal the lapses in the CWA and SDWA when it comes to the protection of surface waters. The current environmental policies at the state and federal level are not properly equipped with the legislative or economic power to provide meaningful requirements, such as multi-site water testing or water agitators. However, the framework created under the Harmful Algal Bloom and Hypoxia Research and Control Act could be used as a model to facilitate scientific investigation of Freshwater HABs while holding local authorities and the EPA accountable for the regulation of potable water. Environmental impact reports are crucial in both monitoring and assessing the risk of our local water sources for recreation, drinking safety, and economy. Nationwide surveillance of the occurrence of cyanobacteria and its effects on human, animal and ecosystem health are needed at different times of year. This is especially important during agricultural growing seasons and warmer weather when most humans would interact with impaired water sources. Additional research is needed on the impact of inhalation of microscopic cyanobacteria, as exposure to contaminated water can occur recreationally, in agriculture, sewage systems, or irrigation. Research is also needed on dermal exposure, specifically the impact of small dosages on humans and animals due to constant exposure.

The EPA can work with Congress and its scientist lead task force to create regulations, education, recommendations, and policy in accordance with the Freshwater HAB Act. This would provide multiple levels of accountability at a private and state level. Protecting our water sources is an essential component of sustainable economic development of aquatic resources in the United States. There is currently no legislation for freshwater comparable to the Harmful Algal Bloom and Hypoxia Research and Control Act, which is targeted toward coastal and ocean waters, but similar legislation for freshwater would facilitate prevention and mitigation of HABs. Due to the construction of environmental regulation, this will need to come from the EPA first. The CWA and the SDWA give the EPA regulatory control over the nation's waters, and therefore they must be the initiator in policy implementation. Like the SDWA, new legislation can stipulate for competitive funding, to ensure qualified institutions are engaged in research. International institutions should be able to apply for funding, as many countries will experience the same types of blooms in their water systems due to agricultural runoff. The funding given to industry research could be used by the EPA to initiate policy determination within Congress.

Prevention research would require cooperation from the agriculture industry, as well as inter-governmental participation. To reduce nutrient loads in affected areas, a green chemistry approach to the utilization of phosphorous or synthetic alternatives could create a closed loop cycle minimizing nutrient runoff. While eutrophication is common in any natural lake, those used by humans must be closely monitored and protected from this process to minimize the risk of HABs and water impairment. One approach to minimize the chance of eutrophication is to use artificial water stratifiers to mix different water columns to help prevent hypoxic zones, thereby disturbing the HAB position in the water column and inhibiting its nutrient allocation, especially in locations of the surface waters that are more likely to see blooms. Research on the chemical and biological effects on cyanobacteria can help understanding of how this process can be utilized in lake management to prevent HABs, in addition to its primary function of oxygenation (Yu et al. 2015). Autonomous underwater drones can map and identify algae blooms during a bloom period to further understand ideal conditions for HAB growth. Expanded Polymerase Chain Reaction testing of algae could provide real-time species identification (Antonella and Luca 2013). In conclusion, the Clean Water Act could provide both public salience and government financial support to prevention research, as nutrient loading and toxic water systems fall under this legislation.

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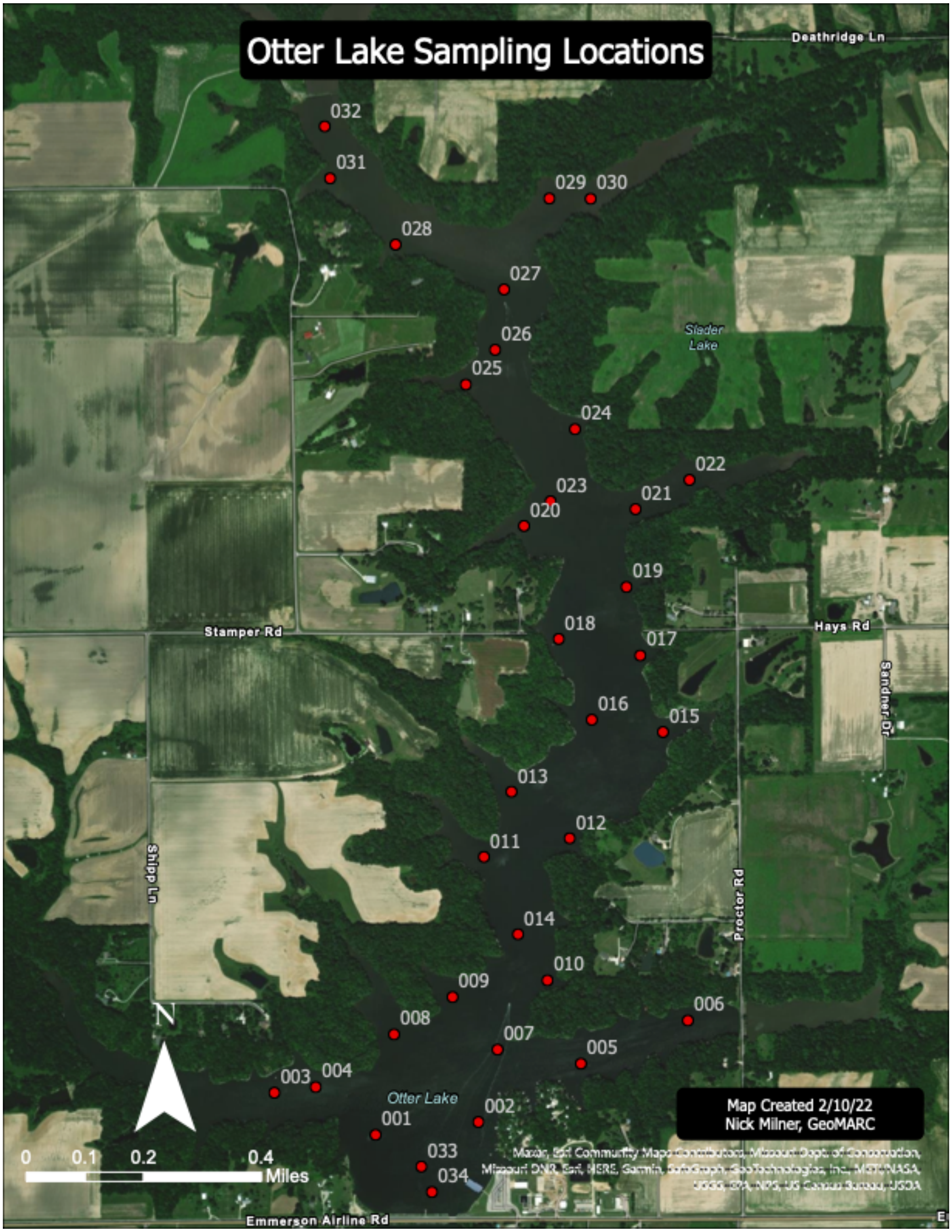
7. Appendix

Appendix A. *Site-Labeled Map of Otter Lake. Generated by Nick Milner at GeoMARC*



Otter Lake Sampling Locations

Map Created 2/10/22
Nick Milner, GeoMARC

Maxar, Esri Community Maps Contributors, Missouri Dept. of Conservation, Missouri DNR, Esri, HERE, Garmin, SafeGraph, GeoTechnologies, Inc., METI/NASA, USGS, EPA, NPS, US Census Bureau, USDA

Appendix B. *Statistical Correlations computed in "R Studio" for March through December on Otter Lake.*

3/4/2021	**Chl *a***	**TKP**	**TKN**	**Nitrate**	**pH**	**Turbidity**	**RDO**	**Temp**
Chl *a*	1.00000	0.23471	0.46987	0.32490	0.60571	0.69878	0.83952	NA
TKP	0.23471	1.00000	0.09017	0.06396	0.22501	0.25165	0.22791	NA
TKN	0.46987	0.09017	1.00000	0.36145	0.48924	0.55505	0.44223	NA
Nitrate	0.32490	0.06396	0.36145	1.00000	0.34305	0.73305	0.23444	NA
pH	0.60571	0.22501	0.48924	0.34305	1.00000	0.60746	0.57552	NA
Turbidity	0.69878	0.25165	0.55505	0.73305	0.60746	1.00000	0.65876	NA
RDO	0.83952	0.22791	0.44223	0.23444	0.57552	0.65876	1.00000	NA
Temp	NA	NA	NA	NA	NA	NA	NA	NA

4/13/2021	**Chl *a***	**TKP**	**TKN**	**Nitrate**	**pH**	**Turbidity**	**RDO**	**Temp**
Chl *a*	1.00000	-0.22013	0.50395	-0.41175	0.55184	-0.29819	0.47904	0.17659
TKP	-0.22013	1.00000	-0.09666	0.43746	-0.39349	0.39317	-0.37576	0.29177
TKN	0.50395	-0.09666	1.00000	-0.52700	0.57433	-0.47359	0.56499	-0.38355
Nitrate	-0.41175	0.43746	-0.52700	1.00000	-0.89321	0.92381	-0.77377	0.31154
pH	0.55184	-0.39349	0.57433	-0.89321	1.00000	-0.83414	0.89649	-0.07457
Turbidity	-0.29819	0.39317	-0.47359	0.92381	-0.83414	1.00000	-0.72844	0.32652
RDO	0.47904	-0.37576	0.56499	-0.77377	0.89649	-0.72844	1.00000	-0.05417
Temp	0.17659	0.29177	-0.38355	0.31154	-0.07457	0.32652	-0.05417	1.00000

5/7/2021	**Chl *a***	**TKP**	**TKN**	**Nitrate**	**pH**	**Turbidity**	**RDO**	**Temp**
Chl *a*	1.00000	-0.02525	-0.04431	-0.25178	0.27659	-0.07709	0.14475	NA
TKP	-0.02525	1.00000	0.55286	0.15175	-0.48803	0.18193	-0.15683	NA
TKN	-0.04431	0.55286	1.00000	-0.24364	-0.20449	-0.19794	-0.10676	NA
Nitrate	-0.25178	0.15175	-0.24364	1.00000	0.19593	0.84303	0.09948	NA
pH	0.27659	-0.48803	-0.20449	0.19593	1.00000	0.25060	0.32337	NA
Turbidity	-0.07709	0.18193	-0.19794	0.84303	0.25060	1.00000	0.16843	NA
RDO	0.14475	-0.15683	-0.10676	0.09948	0.32337	0.16843	1.00000	NA
Temp	NA	NA	NA	NA	NA	NA	NA	NA

6/4/2021	**Chl *a***	**TKP**	**TKN**	**Nitrate**	**pH**	**Turbidity**	**RDO**	**Temp**
Chl *a*	1.00000	0.26195	0.69967	-0.30558	0.56274	0.57081	0.70084	-1.00000
TKP	0.26195	1.00000	-0.15839	-0.23053	0.30412	0.09745	0.20020	-1.00000
TKN	0.69967	-0.15839	1.00000	-0.28549	0.50786	0.59545	0.58671	-1.00000
Nitrate	-0.30558	-0.23053	-0.28549	1.00000	-0.67052	0.03415	-0.48771	1.00000
pH	0.56274	0.30412	0.50786	-0.67052	1.00000	0.32395	0.60948	1.00000
Turbidity	0.57081	0.09745	0.59545	0.03415	0.32395	1.00000	0.39771	1.00000
RDO	0.70084	0.20020	0.58671	-0.48771	0.60948	0.39771	1.00000	-1.00000
Temp	-1.00000	-1.00000	-1.00000	1.00000	1.00000	1.00000	-1.00000	1.00000

7/7/2021	**Chl *a***	**TKP**	**TKN**	**Nitrate**	**pH**	**Turbidity**	**RDO**	**Temp**
Chl *a*	1.00000	0.57612	0.41496	0.49468	-0.08052	0.58938	-0.05972	0.83589
TKP	0.57612	1.00000	0.54514	0.48337	-0.19818	0.52196	0.15093	0.05609
TKN	0.41496	0.54514	1.00000	0.06733	-0.23410	0.27413	-0.33857	0.22450
Nitrate	0.49468	0.48337	0.06733	1.00000	0.07352	0.73971	0.61844	-0.90569
pH	-0.08052	-0.19818	-0.23410	0.07352	1.00000	-0.15599	0.33131	-0.47849
Turbidity	0.58938	0.52196	0.27413	0.73971	-0.15599	1.00000	0.28089	0.54348
RDO	-0.05972	0.15093	-0.33857	0.61844	0.33131	0.28089	1.00000	-0.90433
Temp	0.83589	0.05609	0.22450	-0.90569	-0.47849	0.54348	-0.90433	1.00000

8/13/2021	**Chl *a***	**TKP**	**TKN**	**Nitrate**	**pH**	**Turbidity**	**RDO**	**Temp**
Chl *a*	1.00000	0.20711	0.10442	-0.14752	0.29424	0.28251	0.08470	NA
TKP	0.20711	1.00000	0.39606	0.00267	0.12513	0.43445	0.17806	NA
TKN	0.10442	0.39606	1.00000	0.29296	-0.43774	-0.20240	-0.02596	NA
Nitrate	-0.14752	0.00267	0.29296	1.00000	-0.04400	-0.22538	-0.28687	NA
pH	0.29424	0.12513	-0.43774	-0.04400	1.00000	0.45237	0.14856	NA
Turbidity	0.28251	0.43445	-0.20240	-0.22538	0.45237	1.00000	0.45643	NA
RDO	0.08470	0.17806	-0.02596	-0.28687	0.14856	0.45643	1.00000	NA
Temp	NA	NA	NA	NA	NA	NA	NA	NA

9/10/2021	**Chl *a***	**TKP**	**TKN**	**Nitrate**	**pH**	**Turbidity**	**RDO**	**Temp**
Chl *a*	1.00000	0.03927	0.32309	-0.09376	0.40767	-0.21001	0.19675	NA
TKP	0.03927	1.00000	-0.78006	-0.07083	0.15672	-0.11536	0.07426	NA
TKN	0.32309	-0.78006	1.00000	-0.07287	0.04548	-0.03065	0.09712	NA
Nitrate	-0.09376	-0.07083	-0.07287	1.00000	-0.06698	-0.07321	-0.05752	NA
pH	0.40767	0.15672	0.04548	-0.06698	1.00000	0.10541	0.20266	NA
Turbidity	-0.21001	-0.11536	-0.03065	-0.07321	0.10541	1.00000	-0.07829	NA
RDO	0.19675	0.07426	0.09712	-0.05752	0.20266	-0.07829	1.00000	NA
Temp	NA	NA	NA	NA	NA	NA	NA	NA

10/8/2021	**Chl *a***	**TKP**	**TKN**	**Nitrate**	**pH**	**Turbidity**	**RDO**	**Temp**
Chl *a*	1.00000	0.51887	0.53734	-0.14274	0.43278	0.61240	-0.16287	-0.04993
TKP	0.51887	1.00000	0.99238	-0.02748	0.42178	0.42478	0.01441	0.05190
TKN	0.53734	0.99238	1.00000	-0.05197	0.46331	0.42768	-0.00852	0.06939
Nitrate	-0.14274	-0.02748	-0.05197	1.00000	-0.03320	-0.00880	0.91491	-0.00608
pH	0.43278	0.42178	0.46331	-0.03320	1.00000	0.45627	-0.02887	0.39211
Turbidity	0.61240	0.42478	0.42768	-0.00880	0.45627	1.00000	-0.09298	-0.24602
RDO	-0.16287	0.01441	-0.00852	0.91491	-0.02887	-0.09298	1.00000	0.20323
Temp	-0.04993	0.05190	0.06939	-0.00608	0.39211	-0.24602	0.20323	1.00000

11/12/2021	**Chl *a***	**TKP**	**TKN**	**Nitrate**	**pH**	**Turbidity**	**RDO**	**Temp**
Chl *a*	1.00000	0.14646	0.28383	0.55592	0.48242	0.62992	0.79872	-0.64693
TKP	0.14646	1.00000	0.16888	0.06099	0.03640	0.00721	0.07081	0.05659
TKN	0.28383	0.16888	1.00000	0.19481	0.14215	0.22957	0.24018	-0.26220
Nitrate	0.55592	0.06099	0.19481	1.00000	0.13837	0.46346	0.32423	-0.57236
pH	0.48242	0.03640	0.14215	0.13837	1.00000	0.05194	0.81989	-0.67061
Turbidity	0.62992	0.00721	0.22957	0.46346	0.05194	1.00000	0.31548	-0.38645
RDO	0.79872	0.07081	0.24018	0.32423	0.81989	0.31548	1.00000	-0.71882
Temp	-0.64693	0.05659	-0.26220	-0.57236	-0.67061	-0.38645	-0.71882	1.00000

12/3/2021	**Chl *a***	**TKP**	**TKN**	**Nitrate**	**pH**	**Turbidity**	**RDO**	**Temp**
Chl *a*	1.00000	-0.07741	0.39098	-0.20411	-0.53850	-0.06270	-0.17112	0.12284
TKP	-0.07741	1.00000	0.05550	-0.12724	-0.10380	0.24133	-0.17474	-0.12919
TKN	0.39098	0.05550	1.00000	-0.09692	-0.17227	-0.16737	-0.13700	-0.03279
Nitrate	-0.20411	-0.12724	-0.09692	1.00000	0.08705	0.37875	0.13592	-0.24366
pH	-0.53850	-0.10380	-0.17227	0.08705	1.00000	-0.21798	0.62289	0.17740
Turbidity	-0.06270	0.24133	-0.16737	0.37875	-0.21798	1.00000	0.02462	-0.08228
RDO	-0.17112	-0.17474	-0.13700	0.13592	0.62289	0.02462	1.00000	0.33939
Temp	0.12284	-0.12919	-0.03279	-0.24366	0.17740	-0.08228	0.33939	1.00000