Routine monitoring of Western Lake Erie to track water quality changes associated with cyanobacterial harmful algal blooms

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Abstract

The western basin of Lake Erie has a history of recurrent cyanobacterial harmful algal blooms (HABs) despite decades of efforts by the United States and Canada to limit phosphorus loading, a major driver of the blooms. In response, the National Oceanic and Atmospheric Administration (NOAA) Great Lakes Environmental Research Laboratory (GLERL) and the Cooperative Institute for Great Lakes Research (CIGLR) created an annual sampling program to detect, monitor, assess, and predict HABs in western Lake Erie. Here we describe the data collected from this monitoring program from 2012 to 2021. This dataset includes observations on physico-chemical properties, major nutrient fractions, phytoplankton pigments, microcystins, and optical properties for western Lake Erie. This dataset is particularly relevant for creating models, verifying and calibrating remote sensing algorithms, and informing experimental research to further understand the water quality dynamics that influence HABs in this internationally significant body of freshwater. The dataset can be freely accessed from NOAA National Centers for Environmental Information (NCEI) at https://doi.org/10.25921/11da-3x54 (Cooperative Institute for Great Lakes Research, University of Michigan; NOAA Great Lakes Environmental Research Laboratory, 2019).

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Introduction

Lake Erie is situated on the international boundary between the United States and Canada and is the smallest by volume of the five Laurentian Great Lakes. It is ecologically, culturally, and economically significant to the approximately 12.5 million people who live in the watershed. Each year Lake Erie supports nearly 14,000 tonnes of commercial and traditional fisheries, over 33,000,000 tonnes of freight, and over $1.5 million in recreation and tourism business (Sterner et al., 2020). Lake Erie has endured multiple anthropogenic stressors since European settlement in the area, most notably the draining of coastal wetlands for development of agricultural lands in the late 18th century (Allinger and Reavie, 2013). Currently, the ecological state of Lake Erie is considered poor, partially due to excess nutrient input that supports harmful algal blooms (HABs; ECCC and US EPA, 2022). These seasonal HABs are typically dominated by toxin producing cyanobacteria, causing concern for public and ecosystem health (Watson et al., 2016). Humans can be exposed to cyanotoxins through ingestion of contaminated fish and drinking water and through inhalation and dermal exposure during recreational events such as swimming and boating (Carmichael and Boyer, 2016; Buratti et al., 2017). Cyanotoxins can also cause illness and death in aquatic and terrestrial animals (Carmichael and Boyer, 2016). The economic cost of HABs impacts in Lake Erie is estimated to be hundreds of millions of dollars each year (Smith et al., 2019).

To combat the deteriorated state of Lake Erie water quality, bi-national water resource management policies alongside scientific research and water quality monitoring efforts have been underway for decades. The Great Lakes Water Quality Agreement (GLWQA), first signed in 1972, was a commitment between the US and Canada in response to degraded water quality throughout the Great Lakes ecosystem (GLWQA, 2012). Phosphorus was found to be the key nutrient that was promoting excess phytoplankton growth (Charlton et al., 1993), and thus the GLWQA sought to limit total phosphorus input to the lakes in an attempt to reduce

phytoplankton growth and biomass (Steffen et al., 2014). The 1972 Clean Water Act (CWA) was similarly enacted to regulate point-source pollution discharge, including phosphorus, into navigable waters in the United States. After the signing and implementation of the phosphorus load reduction practices outlined in the GLWQA and CWA, the water quality of Lake Erie improved and the lake experienced a period of restoration (Makarewicz and Bertram, 1991). This success was attributed to upgrades to sewage treatment plants and industrial discharges which reduced phosphorus loading from point sources by 50% within ten years of peak levels observed in 1968 (Charlton et al., 1993; Joosse and Baker, 2011; Steffen et al., 2014).

While the water quality of Lake Erie rebounded in the 1980s and early 1990s, by the mid 1990s and early 2000s annual HAB events were occurring in Lake Erie again, particularly in the warm, shallow western basin (Allinger and Reavie, 2013; Kane et al., 2015; Watson et al., 2016). Total phosphorus loading has been relatively stable in Lake Erie from the 1980s onward (Dolan and Chapra, 2012; Watson et al., 2016), and although point-source phosphorus loading controls had been a successful mitigation measure at one point, several anthropogenic stressors within the watershed were exacerbating the issue of poor water quality. An increase in agricultural sources of biologically available soluble nutrients, legacy phosphorus in the Lake Erie watershed, altered nutrient cycling by invasive dreissenid mussels, and climate change are thought to be primarily responsible for the HABs resurgence (Vanderploeg et al., 2001; Conroy et al., 2005; Bridoux et al., 2010; Michalak et al., 2013; Matisoff et al., 2016; Huisman et al., 2018; Van Meter et al., 2021).

The post-recovery period HABs have predominantly been composed of the cyanobacteria species *Microcystis aeruginosa* along with genera *Anabaena, Aphanizomenon, Dolichospermum,* and *Planktothrix* (Steffen et al., 2014; Watson et al., 2016). These cyanobacteria can produce an array of several types of phycotoxins, with the most common being a suite of hepatotoxins known as microcystins (MCs). Microcystins primarily affect the liver but can also cause adverse health effects on the kidneys, brain, and reproductive organs

(Carmichael and Boyer, 2016). Phycotoxins are commonly present during Lake Erie HABs, and in August 2014 the city of Toledo, OH drinking water supply was contaminated with MCs, leaving >400,000 without clean drinking water (Steffen et al., 2017).

To understand HAB events in US waterways, Congress authorized the Harmful Algal Bloom and Hypoxia Research and Control Act in 1998 (HABHRCA; Public Law 115-423) which mandated the National Oceanic and Atmospheric Administration (NOAA) to “advance the scientific understanding and ability to detect, monitor, assess, and predict HAB and hypoxia events". Under HABHRCA, the NOAA Great Lakes Environmental Research Lab (GLERL), NOAA National Centers for Coastal Ocean Science (NCCOS), and the Cooperative Institute for Great Lakes Research (CIGLR; formerly CILER - Cooperative Institute for Limnology and Ecosystems Research) developed an ecological forecast to predict HAB events in Lake Erie. Starting in 2008, researchers at these institutes began using remote sensing to monitor seasonal HABs, created a seasonal forecast system based on spring P loads, and developed models to predict short-term bloom changes to alert stakeholders and the public (Rowe et al., 2016). Products from these efforts, known as Lake Erie Harmful Algal Bloom Forecasts, are freely available during the bloom season at https://coastalscience.noaa.gov/research/stressor-impacts-mitigation/hab-forecasts/lake-erie/.

In-situ sampling of the bloom was necessary to calibrate and validate the remote sensing images and models as well as measure microcystin concentration. Sampling events were led by personnel at GLERL and CIGLR starting in 2008 and were designed to collect discrete samples within the extent of the bloom area. At first, samples were taken opportunistically within the bloom and sampling locations and analytical parameters were inconsistent. In 2009, regular sampling stations were identified based on spatial patterns of the bloom. From 2009 to 2011, in addition to opportunistic samples, nine main stations in the western basin of Lake Erie were sampled intermittently from June through October (Bertani et al., 2017; Rowland et al., 2020). While these sampling efforts initially began to complement

existing research products, the experimental nature of the 2008 to 2011 sampling cruises also provided insight into creating a regular monitoring program that would support critical research and product development related to western Lake Erie HABs.

In 2012, researchers at GLERL and CIGLR, with support from the Great Lakes Restoration Initiative (GLRI), formalized a sampling regimen to monitor the spatial and temporal variability of seasonal HAB events in western Lake Erie (WLE). The establishment of this monitoring program corresponded with increased federal emphasis on evaluating trends and drivers of WLE HABs and water quality. Four monitoring stations were identified and regular surface samples were collected from May to September and analyzed for nutrient, pigment, and particulate microcystin concentrations (Figs. 1 & 2). In following years, the monitoring program evolved and expanded. New stations were added to better characterize the bloom and complement other observing systems. Sampling parameters were adjusted and added based on the needs of current research (Table 1). Results of these sampling cruises were compiled and distributed informally upon request until 2019 when the data were organized and archived on the NOAA National Centers for Environmental Information (NCEI) open-access data repository (https://www.ncei.noaa.gov/).

Long term monitoring of WLE is fundamental to the continual assessment of water quality changes in response to both stressors and water quality management efforts (Hartig et al., 2009, 2021). The GLERL/CIGLR monitoring data has been used by numerous researchers to develop and assess models (Rowe et al., 2016; Weiskerger et al., 2018; Fang et al., 2019; Liu et al., 2020; Qian et al., 2021; Wang and Boegman, 2021; Hellweger et al., 2022; Maguire et al., 2022), to calibrate remote sensing algorithms (Sayers et al., 2016, 2019; Avouris and Ortiz, 2019; Bosse et al., 2019; Vander Woude et al., 2019; Pirasteh et al., 2020; Xu et al., 2022), and to elucidate ecological mechanisms and complement experimental data (Cory et al., 2016; Reavie et al., 2016; Berry et al., 2017; Steffen et al., 2017; Kharbush et al., 2019, 2023; Newell

et al., 2019; Den Uyl et al., 2021; Smith et al., 2021, 2022; Hoffman et al., 2022; Marino et al., 2022; Yancey et al., 2022a, b).

The objective of this paper is to inform users of the dataset "Physical, chemical, and biological water quality monitoring data to support detection of Harmful Algal Blooms (HABs) in western Lake Erie, collected by the Great Lakes Environmental Research Laboratory and the Cooperative Institute for Great Lakes Research since 2012" by describing the data generated from this monitoring program and detailing how samples were collected and analyzed. This paper contextualizes this long-term data set so that it can continue to be used to benefit our collective ecological knowledge of western Lake Erie.

Table 1. Description of stations sampled in western Lake Erie from 2012 to 2021. Latitude and longitude (decimal degree) coordinates for each station are target locations as the boat was allowed to drift at each site during *in-situ* sampling.

Station	**Latitude**	**Longitude**	**Avg. Depth (m)**	**Years Monitored**
WE02	41.762	-83.330	5.4	2012-2021
WE04	41.827	-83.193	8.4	2012-2021
WE06	41.705	-83.385	2.9	2012-2021
WE08	41.834	-83.364	4.8	2012-2021
WE09	41.718	-83.424	2.7	2016-2021
WE12	41.703	-83.254	6.6	2014-2021
WE13	41.741	-83.136	8.9	2014-2021
WE14	41.720	-83.010	9.3	2015
WE15	41.617	-83.009	4.5	2015-2017
WE16	41.660	-83.143	6.2	2018-2021

Methods

Study Site

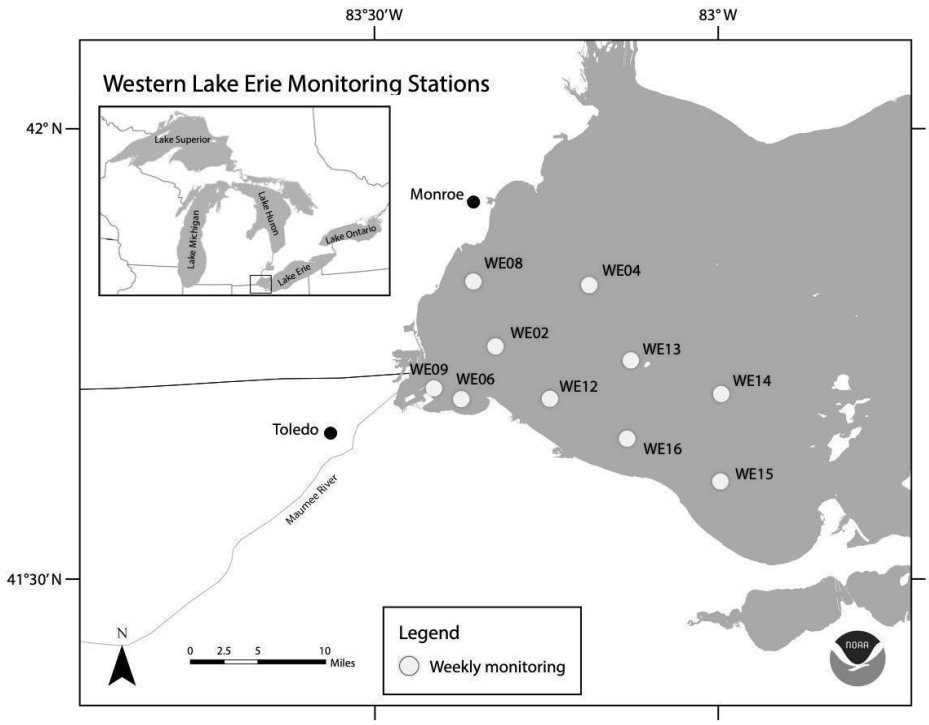
Based on the lake's bathymetry, Lake Erie can be divided into the eastern, central, and western basins which in turn influence physical and biological processes (Allinger and Reavie, 2013). The data presented in this paper were collected from the western basin, which encompasses the western part of the lake to Point Pelee, ON, Canada and Cedar Point, OH, USA (Fig. 1). The well-mixed western basin is the shallowest (maximum average depth of 11 m), warmest, and most productive of the three basins. Although it's typical for temperate WLE to have ice cover in the winter (Jan to Mar), summer (Jul to Sep) surface water temperatures often reach or exceed 25 °C. The western basin receives 95% of its hydraulic inflow from the Detroit River, which connects Lake Erie hydrologically to Lake Huron via the St. Clair River and Lake St. Clair (Cousino et al., 2015). Among the other tributaries to WLE (including River Raisin, Portage River, Ottawa River, Stony Creek, Swan Creek, and Sandusky River), the Maumee River discharges into the western basin near the city of Toledo, Ohio and contributes a significant amount of sediments and nutrients to the entire Lake Erie basin (Baker et al., 2014a, b; Rowland et al. 2020; see NCWQR 2022 for Maumee River water quality data). Nutrient and sediment loads from the Maumee River can vary with precipitation, where stormwater runoff can provide a pulse of nutrients into the basin, potentially altering cyanobacteria dynamics (Baker et al., 2014a; King et al., 2022). Land use in the Lake Erie watershed is 75% agricultural and 11% urban, both of which contribute to the large amounts of soluble reactive phosphorus into the basin (Mohamed et al., 2019; Myers et al., 2000).

This dataset includes water quality data from ten monitoring stations on the United States side of WLE that were sampled from 2012 to 2021 (Figs. 1 & 2, Tables 1 & 2). The

177 average depth of monitoring stations ranged from 2.7 m at WE9 to 9.3 m at WE14. These sites
178 were chosen to reflect the various nutrient and hydrologic inputs and gradients into WLE, as
179 well as represent areas of the basin that are prone to HABs. The Maumee River inflow was a
180 major consideration in determining these sites. The initial 4 stations sampled in this program
181 (WE02, WE04, WE06, and WE08) were selected because they were consistently within the
182 WLE blooms occurring at the time. Additional sites were later added to better represent the
183 spatial extent of HABs and to augment existing data provided by moored buoy continuous
184 monitoring systems, advanced monitoring technologies, such as Environmental Sample
185 Processors (Den Uyl et al., 2022), and other monitoring programs in WLE.

186 Field Sampling

187 Western Lake Erie discrete field sampling was accomplished using NOAA GLERL
188 research vessels. Sampling took place during ice-free months and aimed to quantify the
189 environmental conditions prior to, during, and at the end of the bloom (Fig. 2). Sampling stations
190 represent approximate locations (Table 1; Fig. 1); *in situ* measurements and sampling were
191 collected once the boat reached the targeted location and then proceeded to drift during
192 sampling. The frequency and timing of those cruises varied over the first few years but has been
193 consistent since 2017 (Fig. 2). Sampling was disrupted in 2020 due to the global COVID-19
194 pandemic and resulting public health restrictions. In 2020, sampling was initiated in mid-June at
195 a reduced number of sites for select water quality parameters. In July, sampling stations and
196 parameters were expanded and all stations and parameters were sampled and measured by
197 August 2020. The prior standard sampling schedule resumed in April 2021.



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199 Figure 1. Location of western Lake Erie water quality monitoring stations. This map was

200 provided by NOAA for use in this publication.

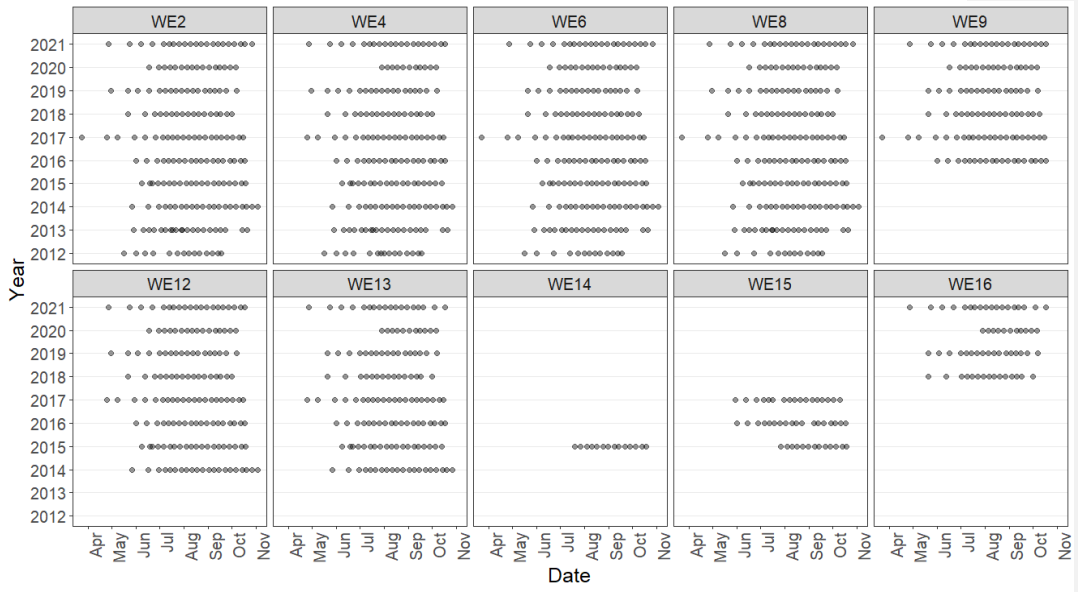

Figure 2. Sampling frequency for each monitoring station for years sampled between 2012 to 2021.

In-situ measurements for conductivity, temperature, dissolved oxygen (DO), beam attenuation, transmission, and photosynthetically active radiation (PAR) were taken with a Sea-Bird 19plus V2 conductivity, temperature, and depth (CTD) profiler attached to a hydraulic crane. Data were collected on the downcast and were reported as the mean of recorded values within ± 0.5 m of the discrete sample depth. In 2012, sample temperature was taken on the boat with a Vee Gee Scientific IP67-rated digital thermometer. Sky conditions were recorded at the discretion of the field technician at each station during the sampling cruise. A Secchi disk was lowered into the water on the shaded side of the boat at each station and the depth at which the Secchi disk was no longer visible was recorded (Wetzel and Likens, 2000).

Water column samples were collected using a 5 L vertical Niskin bottle (General Oceanics model 1010). Niskin casts were evenly distributed between one or more high-density

polyethylene bottles that were rinsed with site water and stored in a cooler. Three to four Niskin casts were used to fill the bottles, such that each bottle is a composite sample of the water collected. Surface samples were taken 0.75 m below the water's surface, mid-column samples were taken at approximately 4.25 m below surface, and benthic or bottom samples were taken at approximately 0.5 m above the lake bottom at each station. Surface samples were taken at all stations while mid-column and benthic sample collection varied between sites and years. Scum samples of dense cyanobacterial accumulation on the surface of the water were collected opportunistically using a 2 L modified Van Dorn water sampler. Sampling times were reported as Eastern Daylight Time (UT -4:00). Upon arrival at the laboratory, raw water samples were immediately subsampled and preserved until analysis.

Wind speed and wave height data were obtained from moored buoy continuous monitoring systems in proximity to sampling stations for a timestamp that corresponded to the time samples were collected at that station. Wave height data for all stations were obtained from the Toledo Intake Buoy (owned and maintained by Limnotech Inc.). Wind speed data for stations WE02, WE06, WE09, WE12, WE14, WE15, and WE16 were also collected from this buoy. Data for this buoy is available through the Great Lakes Observing System (GLOS; platform ID 45165, https://seagull.glos.org/data-console/71). Wind speed data for stations WE04, WE08, and WE13 were obtained from the Toledo Harbor Light no. 2 buoy (Station THLO1, owned and maintained by GLERL). Data for this buoy is available through NOAA's National Data Buoy Center (https://www.ndbc.noaa.gov/station_realtime.php?station=THLO1).

Laboratory analysis of samples

Water collected from WLE was subsampled to make a range of analytical measurements in the laboratory (Table 2).

Table 2. Summary of parameters reported in the dataset. Wind speed and wave height data are collected from moored buoy continuous monitoring systems which provide the data in Imperial units.

Parameter	Years monitored	Method
Surface samples (n=1296)	2012-2021	n/a
Mid-column samples (n=19)	2015	n/a
Benthic samples (n=512)	2015-2021	n/a
Station depth (m)	2012-2021	Sea-Bird 19plus V2 CTD profiler
Time of sampling (Eastern Daylight Time UTC -4:00)	2012-2021	n/a
Latitude (decimal degree)	2012-2021	n/a
Longitude (decimal degree)	2012-2021	n/a
Wind speed (knots)	2015-2021	Moored buoy continuous monitoring systems
Wave height (ft)	2012-2021	Moored buoy continuous monitoring systems
Cloud cover (sky)	2012-2021	Qualitative description
Secchi depth (m)	2012-2021	Wetzel and Likens (2000)
Sample temperature (°C)	2012	Vee Gee Scientific digital thermometer
CTD temperature (°C)	2013-2021	Sea-Bird 19plus V2 CTD profiler
CTD specific conductivity (µS cm^{-1})	2013-2021	Sea-Bird 19plus V2 CTD profiler
CTD beam attenuation (m^{-1})	2013-2021	Sea-Bird 19plus V2 CTD profiler
CTD transmission (%)	2013-2021	Sea-Bird 19plus V2 CTD profiler
CTD dissolved oxygen (DO; mg L^{-1})	2013-2021	Sea-Bird 19plus V2 CTD profiler
CTD photosynthetically active radiation (PAR; µE m^{-2} s^{-1})	2013-2021	Sea-Bird 19plus V2 CTD profiler

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Turbidity (NTU)	2013-2021	EPA Method 180.1
Particulate microcystins (µg L^{-1})	2012-2021	Wilson et al. (2008)
Dissolved microcystins (µg L^{-1})	2014-2021	Wilson et al. (2008)
Phycocyanin (µg L^{-1})	2012-2021	Horvath et al. (2013)
Chlorophyll *a* (µg L^{-1})	2012-2021	Speziale et al. (1984)
Total phosphorus (TP; µg L^{-1})	2012-2021	Standard Method 4500-P
Total dissolved phosphorus (TDP; µg L^{-1})	2012-2021	Standard Method 4500-P
Soluble reactive phosphorus (SRP; µg L^{-1})	2012-2021	Standard Method 4500-P
Ammonium-N (µg L^{-1})	2012-2021	Standard Method 4500-nh3-nitrogen (Ammonium)
Nitrate-N + Nitrite-N (mg L^{-1})	2012-2021	Standard Method 4500-no3-nitrogen (nitrate)
Urea-N (µg L^{-1})	2016-2017	Milvenna and Savidge (1992), Goeyens et al. (1998), Chaffin and Bridgeman (2014)
Particulate organic carbon (POC; mg L^{-1})	2012-2021	Hedges and Stern (1984)
Particulate organic nitrogen (PON; mg L^{-1})	2012-2021	Hedges and Stern (1984)
Colored dissolved organic material (CDOM; m^{-1})	2014-2021	Binding et al. (2008), Mitchell et al. (2003)
Dissolved organic carbon (DOC; mg L^{-1})	2012-2017	APHA Standard Method 5310 B
Total suspended solids (TSS; mg L^{-1})	2012-2021	APHA Standard Method 2540
Volatile suspended solids (VSS; mg L^{-1})	2012-2021	APHA Standard Method 2540

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Optical properties

Turbidity was measured on raw samples using a Hach 2100AN Turbidimeter following US EPA method 180.1 (1993). Colored dissolved organic material (CDOM, also defined as chromophoric dissolved organic matter) was determined by filtering lake water through an acid

rinsed 0.2 µm nuclepore polycarbonate filter into acid-washed and combusted borosilicate vials. Optical density of the filtered samples was then measured using a Perkin Elmer UV/VIS Lambda 35 spectrophotometer at wavelengths from 300-800 nm. CDOM absorption was calculated at 400 nm (Mitchell et al., 2003; Binding et al., 2008).

Dissolved organic carbon (DOC) concentrations were determined following American Public Health Association (APHA) Standard Method 5310 B. Briefly, lake water was filtered through 0.45 µm polyvinylidene difluoride membrane filters into combusted borosilicate glass vials and frozen at -20°C until analysis. The filtrate was acidified with HCl and sparged with air for 6 min before being analyzed on a Shimadzu total organic carbon analyzer.

Duplicate samples for particulate organic carbon (POC) and particulate organic nitrogen (PON) were collected onto pre-combusted glass fiber filters and analyzed following Hedges and Stern (1984) Samples were stored at -20 °C until analysis. The filters were then acidified by fumigation with 10% HCl and dried at 70°C for 24 h before being quantified on a Perkin Elmer 2400 or a Carlo-Erba 1110 CHN elemental analyzer.

Total suspended solids (TSS) and volatile suspended solids (VSS) were determined via gravimetric analysis following APHA Standard Method 2540. A known volume of lake water was filtered through a pre-combusted, pre-weighed Whatman GF/F glass fiber filter. The filters were then dried at 60° C for at least 24 h and reweighed. The difference in mass between the pre-weighed and processed filter was reported as TSS. Volatile suspended solids concentrations were quantified by combusting the filters used for TSS analysis at 450 °C for 4 h, weighing the combusted filters, and calculating the mass lost.

Nutrient fractions

Total phosphorus (TP) and total dissolved phosphorus (TDP) samples were collected in duplicate by subsampling 50 mL (2012 to 2019) or 20 mL (2020 to 2021) of lake water into acid washed glass tubes and by filtering 20 mL of lake water through a 0.2 µm membrane filter and

collecting the filtrate, respectively. Samples for TP and TDP were refrigerated until samples were digested with potassium persulfate solution and autoclaved at 121°C for 30 min, modified from APHA Standard Method 4500-P. Digested TP and TDP samples were stored at room temperature until concentrations were measured on a Seal QuAAtro continuous segmented flow analyzer (SEAL Analytical Inc.) from 2012 to 2019 and a Seal AA3 from 2020 to 2021 using the ascorbic acid molybdenum method as detailed by the instrument manual and APHA Standard Method 4500-P. Analytical detection limits for the analyses were taken from the instrument manufacturer's documentation.

Soluble reactive phosphorus (SRP), ammonium, nitrate + nitrite, and urea were each determined by filtering 12 mL of lake water through a 0.2 µm membrane filter into 15 mL centrifuge tubes during field sampling. Sample filtrates were stored at -20 °C upon receipt at the laboratory. Soluble reactive phosphorus, ammonium, and nitrate + nitrite concentrations were determined simultaneously on a Seal AA3 continuous segmented flow analyzer. Soluble reactive phosphorus concentrations, like TP and TDP concentrations, were measured using the ascorbic acid molybdenum method as detailed by the instrument manual and APHA Standard Method 4500-P. Ammonium concentrations were measured using Bertholet reactions according to the instrument manual and APHA Standard Method 4500-nh3-nitrogen. Nitrate + nitrite concentrations were measured using copper-cadmium reduction methods according to the instrument manual and APHA Standard Method 4500-no3-nitrogen. Analytical detection limits for these inorganic nutrient analyses were taken from the instrument manufacturer's documentation. Urea samples were measured by adding diacetyl monoxime and thiosemicarbazide to the filtrate and briefly vortexing to mix, followed by adding sulfuric acid and ferric chloride to the solution and briefly vortexing to mix. Samples were then incubated in the dark for 72 h at room temperature before absorbance at 520 nm was read on a Perkin Elmer UV/VIS Lambda 35 spectrophotometer. Urea concentrations were then quantified using a standard curve (Mulvenna and Savidge, 1992; Goeyens et al., 1998; Chaffin and Bridgeman,

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2014). The detection limit was calculated using the standard deviation of repeated measurements.

Photopigments and microcystins

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Particulate phycocyanin and chlorophyll *a* concentrations were determined by filtering a known volume of lake water under low vacuum (<200 mm Hg) onto 47 mm Whatman GF/F glass fiber filters (Cytiva Life Sciences). Particulate phycocyanin sample filters were stored in 15 mL conical polypropylene centrifuge tubes and chlorophyll *a* sample filters were stored in amber glass vials at -20 °C until analysis. Analysis methods for particulate phycocyanin were derived from Horváth et al. (2013) where 9 mL of phosphate buffer was added to sample tubes and samples were agitated using a shaker at 5 °C for 15 min at 100 rpm then vortexed for 10 s each. To encourage cell lysis, samples were subjected to three freeze/thaw cycles at -20 °C followed by sonication for 20 min using a Fisher FS110 H sonicator. Fluorescence of the extracted samples was measured using an Aquafluor 8000-010 fluorometer (Turner Designs) with excitation from 400-600 nm and emission filter of >595 nm. Particulate phycocyanin was calibrated annually against C-Phycocyanin material from Sigma-Aldrich. Analysis methods for chlorophyll *a* were derived from Speziale et al. (1984) where chlorophyll *a* was extracted from samples using dimethylformamide and placed into a 65 °C water bath for 15 min. Samples were then cooled to room temperature and vortexed for 15-20 s before being quantified using a 10 AU fluorometer (Turner Designs) with excitation filter of 436 nm and emission at 680 nm. Phycocyanin and chlorophyll *a* procedures were performed under low or green light to reduce pigment degradation within the cell.

Dissolved and particulate microcystins were quantified using a procedure adapted from Wilson et al. (2008). Dissolved microcystins (dMC) were determined through duplicate samples of ~ 2 mL filtrate that was passed through a 0.2 µm membrane filter and stored in glass vials at -20 °C until analysis. Particulate microcystins (pMC) were collected by filtering a known volume of lake water onto a Whatman GF/F glass fiber filter (2012 to 2015) or a 3 µm pore size

polycarbonate membrane filter (2016 to 2021). Particulate MC was then extracted from the filters. In sampling years 2012 to 2015, glass fiber filters were submerged in a glass vial containing a 75:25 methanol:water solution ($MeOH/H_2O$) and sonicated in an ice bath for 2 min. The samples were centrifuged for 15 min and the supernatant was transferred to a clean glass vial. An additional 5 mL of $MeOH/H_2O$ was added to the filter/precipitate and the sample was incubated at -20 °C for 5 h. The sample was then sonicated for 2 min, centrifuged, and the supernatant was removed and added to the first extract vial. The composite supernatant was then centrifuged under a vacuum until dry. The dried extract was then stored at -20 °C until analysis. Particulate MC concentrations were then determined by adding 1 mL of MIlliQ water to the sample and using sonication to dissolve the dried extract. For sampling years 2016 to 2021, filters were stored in 2 mL sterile microcentrifuge tubes at -20 °C until analysis. During analysis, pMC were extracted from the membrane filters by adding 1 mL of MilliQ water and subjecting samples to three freeze/thaw cycles at -20 °C followed by addition of Abraxis QuickLyse reagents according to the manufacturer (Eurofins/Abraxis). Particulate MC samples for all sampling years were analyzed immediately after extraction. For all sampling years, dMC and pMC concentrations were determined using a congener-independent enzyme-linked immunosorbent assay (ELISA) kit designed to detect and quantity microcystins and nodularins using the ADDA moiety (Envirologix brand used from 2012 to 2015; Eurofins/Abraxis microcystins/nodularins (ADDA) (EPA ETV) (EPA method 546), ELISA, 96-test kit used from 2016 to 2021). Analytical detection limits for the analyses were taken from the manufacturer's documentation.

Results and Discussion

This dataset demonstrates the temporal and spatial variability in water quality parameters in western Lake Erie from 2012 to 2021. Overall, sites closest to the Maumee River inflow (i.e., WE06 and WE09) had the highest median concentrations of nutrients, sediments, pigments, and microcystins compared to sites further out in the basin (i.e., WE02, WE04, and WE13; Table 3). Stations WE06 and WE04 were sampled since the initiation of the monitoring program and consistently represented the high and low extremes of water quality observations during a given time point, respectively, (Table 3) and select parameters for these two sites are represented in figs. 3 and 4. Supplemental figs. 1-16 display the same parameters as figs. 3 and 4 for the remaining stations.

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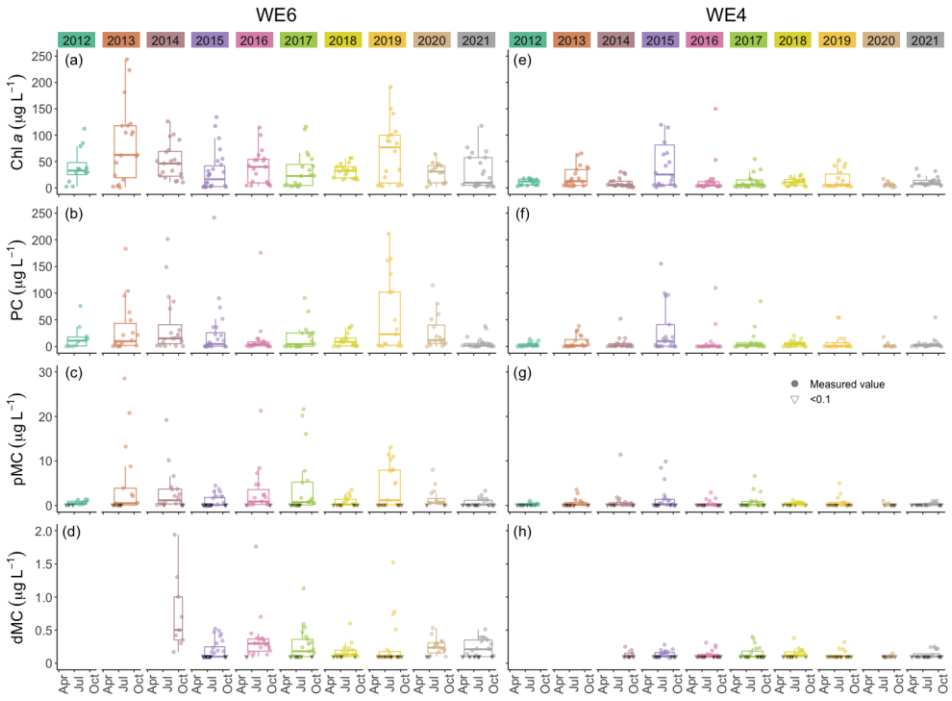

Fig 3. Comparison of chlorophyll *a* (Chl *a*), phycocyanin (PC), particulate microcystins (pMC), and dissolved microcystins (dMC) between stations WE04 and WE06 from 2012 to 2021. Boxplots represent the median and 25% and 75% quartiles with whiskers extending to the highest or lowest point within 1.5x the interquartile range. A scatterplot is overlaid on the boxplots.

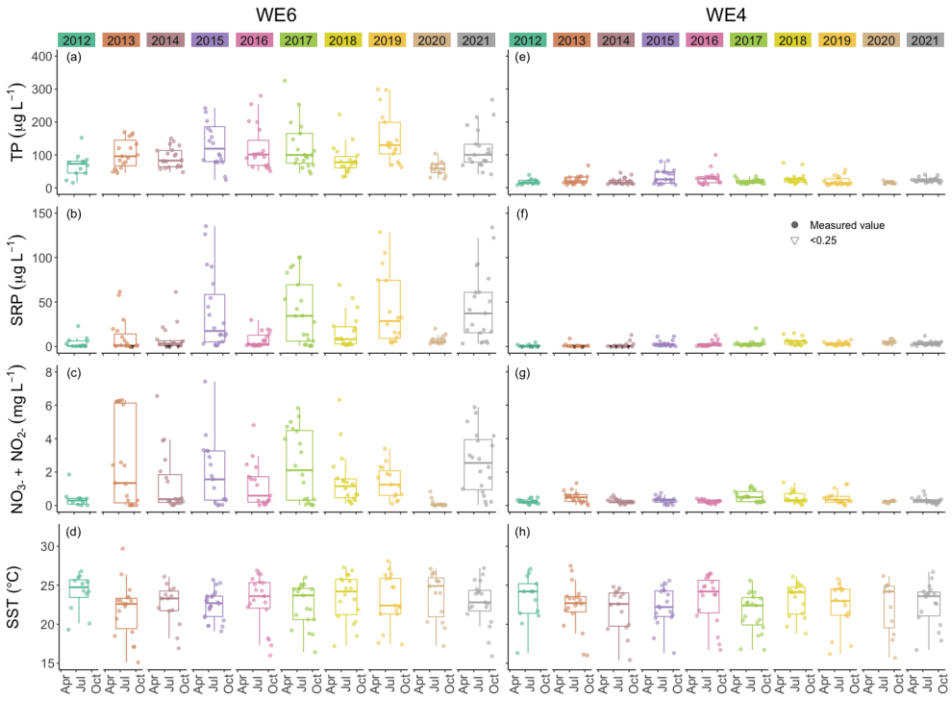

Fig 4. Comparison of total phosphorus (TP), soluble reactive phosphorus (SRP), nitrate plus nitrite (NO_3^- + NO_2^-), and sea surface temperature (SST) between stations WE04 and WE06 from 2012 to 2021. Boxplots represent the median and 25% and 75% quartiles with whiskers extending to the highest or lowest point within 1.5x the interquartile range. A scatterplot is overlaid on the boxplots.

379 Table 3. Median values of each parameter at each monitoring station for all surface samples

380 collected between 2012 to 2021.

	Secchi depth (m)	Temp. (°C)	Cond. (µS cm)	DO (mg L)	PAR (µE m s)	Beam Attenuation (m)	Transmission (%)	Turbidity (NTU)	Particulate MC (µg L)	Dissolved MC (µg L)	Phycocyanin (µg L)	Chl-a (µg L)	TP (µg L)	TDP (µg L)	SRP (µg L)	Ammonia (µg L)	Nitrate + Nitrite (mg L)	POC (mg L)	PON (mg L)	CDOM (m)
WE02	0.8	23.1	287	7.7	264	5.1	28.2	9.9	0.78	0.20	4.8	17.5	53.3	12.8	5.7	12.6	0.44	1.4	0.23	0.99
WE04	2.0	22.9	244	7.6	377	2.2	58.4	3.0	0.46	0.17	1.2	7.7	19.2	4.5	2.2	12.9	0.27	0.63	0.10	0.34
WE06	0.5	23.0	346	7.6	173	6.4	20.5	14.8	1.5	0.28	8.0	33.0	90.1	18.7	8.7	11.8	0.83	2.4	0.38	2.0
WE08	1.0	23.3	299	7.7	166	4.3	34.4	9.0	0.88	0.22	5.7	19.5	50.9	12.3	5.8	13.8	0.45	1.5	0.25	1.1
WE09	0.3	23.9	395	7.1	127	12.6	4.3	23.2	0.95	0.26	5.2	32.6	133	44.8	29.5	43.1	1.4	2.5	0.42	2.4
WE12	0.8	23.1	276	7.7	266	5.4	25.9	11.0	0.67	0.16	2.9	15.1	47.6	10.1	5.4	8.4	0.31	1.2	0.20	0.81
WE13	1.5	22.9	244	7.8	456	2.7	52.4	4.3	0.56	0.15	2.6	8.6	22.3	5.0	2.7	10.2	0.25	0.78	0.14	0.38
WE14	1.4	23.2	238	8.1	796	3.7	40.2	7.2	0.80	0.16	17.0	40.0	31.0	4.7	1.5	2.9	0.17	1.7	0.27	0.60
WE15	1.0	23.0	261	7.7	391	3.4	43.0	6.3	0.86	0.19	2.7	12.7	34.8	5.5	2.0	23.9	0.27	1.1	0.18	0.54
WE16	1.3	24.1	269	7.4	297	3.6	40.8	6.3	0.91	0.18	3.4	12.3	30.2	7.2	4.0	10.6	0.30	1.0	0.16	0.71

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Physicochemical properties

Median surface temperatures for all samples across all years ranged from 22.9 to 24.1 °C and median benthic temperatures ranged from 22.8 to 23.2 °C (Table 3, Fig. 4), indicating that WLE was thermally well mixed throughout the sampling period. A summary of the dataset indicates that 23.8% of surface temperatures were ≥25 °C, and these higher temperatures all occurred from mid-June through the end of September. Bloom forming cyanobacteria species in Lake Erie, including *Microcystis spp.,* often reach maximum growth rates at warmer temperatures (≥ 25 °C) than eukaryotic phytoplankton (Steffen et al., 2014; Huisman et al., 2018). Despite having warmer temperatures that promote recurring HABs, there was only one recorded instance of hypoxia (DO <2 mg L^{-1}) in the dataset and it occurred at WE13 on 08 July 2019. Median DO was 7.62 mg L^{-1} in all surface samples and 7.02 mg L^{-1} in all benthic samples from 2012 to 2021 (Table 3), again indicating minimal stratification in WLE during sampling. Median conductivity from 2012 to 2021 was highest at sites WE06 and WE09, which are closest to the Maumee River input, and lowest at sites WE04 and WE13 near the middle of the basin (Table 3). WE06 and WE09 were the only sites to have median conductivity values above 300 µS cm^{-1}.

Optical properties

Biotic and abiotic particulate concentrations and movement patterns in WLE are prone to spatial and seasonal variations and are heavily influenced by loading from the Maumee River (Prater et al., 2017; Maguire et al., 2022). Secchi depth, turbidity, and PAR measurements have been correlated with distance from Maumee Bay, where light penetration was lowest near the Maumee River (Chaffin et al., 2011). Variability in optical property measurements in WLE is also dependent on Maumee River inputs, and changes in optical properties can potentially be used

in remote sensing algorithms to detect changes in water quality (Sayers et al., 2019). Median Secchi disk depth over the entire dataset was highest at WE04 and lowest at WE06 and WE09, which are closest to the Maumee River (Table 3). Other optical properties, such as PAR, beam attenuation, and transmittance also followed this spatial pattern. In a summary of all samples, median PAR measured at 0.5 m below surface was highest at WE13 and WE14 and lowest at WE09; median transmittance was highest at WE04 and lowest at WE09; and median beam attenuation and turbidity were highest at WE09 and lowest at WE04 (Table 3). Median turbidity values at each site over the 2012 to 2021 period were within the range of previously reported values in the WLE basin (Barbiero and Tuchman, 2004). Median CDOM absorbance and DOC, TSS, and VSS concentrations were again highest at WE09 and lowest at WE04 (Table 3). CDOM gradients in WLE are likewise affected by loading from the Maumee River (Cory et al., 2016) and DOC and CDOM values from this dataset have been used as predictor variables in models estimating PAR attenuation variation in WLE (Weiskerger et al., 2018).

Nutrient fractions

The Maumee River is a major contributor of nutrients to Lake Erie (Steffen et al., 2014; Kast et al., 2021). Median TP concentrations in WLE from 2012 to 2021 were lowest at WE04 and highest at WE09 (Table 3, Fig. 4). Median concentrations at each station from 2012 to 2021 were above the GLWQA Annex 4 goals for TP concentration in open waters, which is 15 µg P L^{-1} for WLE. This goal was met in 92 of 1275 (7.2%) samples and these target values were primarily recorded from stations WE04 and WE13. Sites closer to the mouth of the Maumee River had higher median TP values. While TP loading from the Maumee River tributary declined between 1982 to 2018 (Rowland et al., 2020) the proportion of dissolved P has increased (Joosse and Baker, 2011; Stow et al., 2015). Median TDP values in the WLE dataset were lowest at WE04 and highest at WE09 (Table 3) with a highest recorded value of 274 µg P L^{-1} at

WE08 in 2015. Median SRP concentrations for each station in this dataset were lowest at WE14 and WE15 and were highest at WE09 (Table 3). The maximum recorded SRP concentration was 135.4 µg P L^{-1} at WE06 in 2015 (Fig. 4). Using this dataset, Newell et al. (2019) found that the Maumee River N loading has become more chemically reduced over time where ammonium and PON have increased. Median ammonium concentrations in WLE from 2012 to 2019 were lowest at WE12 and WE14 and highest at WE09 (Table 3) with a recorded maximum concentration of 2109 µg N L^{-1} at WE12 in 2017. Median nitrate + nitrite was lowest at WE13 and WE14 and highest at WE09 (Table 3), with a maximum recorded value of 9.5 mg N L^{-1} at WE09 in 2016. See Fig. 4 for a comparison of nitrate + nitrite concentrations between WE04 and WE06. Median PON concentrations were lowest at WE04 and highest at WE09 (Table 3) with a recorded max of 40.93 mg N L^{-1} at WE08 in 2015.

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Photopigments and microcystins

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Median extracted chlorophyll *a* concentrations in surface waters from 2012 to 2021 were lowest at WE04 and highest at WE06 (Table 3, Fig. 3). The highest recorded surface concentration of chlorophyll *a* was 6784 µg L^{-1} on 10 August 2015 at WE08 during the most severe bloom year in this dataset, according to the CI Index (Wynne et al., 2013; Lunetta et al., 2015). The highest measured levels of particulate phycocyanin, pMC, and TP were also recorded at WE06 on 10 August 2015. Other notably high chlorophyll *a* concentrations were measured during severe bloom years in 2017 (532 µg L^{-1} at WE09 on 04 August) and 2019 (593 µg L^{-1} at WE09 on 05 August). Similarly, median surface particulate phycocyanin concentration for 2012 to 2021 was highest at WE06 and lowest at WE04 (Table 3, Fig. 4). The highest recorded phycocyanin value was from WE08 on 10 August 2015 (8228 µg L^{-1}), followed by 3315 µg L^{-1} at WE06 in 2013 during another severe bloom year.

Particulate MC concentrations had highest median concentrations at WE06 and were lowest at WE04 (Table 3, Fig. 4), similar to particulate chlorophyll *a* and phycocyanin observations. The highest recorded particulate MC concentration in this dataset was from 10 August 2015 at WE08 during a severe bloom year (297 µg L^{-1}), followed by 289 µg L^{-1} at WE06 in 2017 during another severe bloom year according to the CI Index (Wynne et al., 2013; Lunetta et al., 2015). Median dMC concentrations were highest at WE06 and lowest at WE13 (Table 3). The maximum dissolved MC in the dataset was 8.19 µg L^{-1} at WE09 on 05 August 2019, which correlates with high chlorophyll *a* concentrations.

Although the United States does not federally enforce water quality criteria or regulations for cyanotoxins in drinking water, the US EPA has a recommended health advisory of 1.6 µg L^{-1} microcystins in drinking water for school-age children through adults (US EPA, 2015) while the WHO and the Ohio EPA use 1 µg L^{-1} microcystins as a guideline (WHO, 2020). From 2012 to 2021, 44.4% of pMC samples in this dataset exceeded the WHO guidelines and 34.1% exceeded the US EPA health advisory. Monitoring MC concentrations in western Lake Erie has become especially pertinent since August 2014 when the Toledo, OH drinking water treatment plant was contaminated with microcystins in excess of 1 µg L^{-1} and customers were alerted to not drink their tap water until toxin levels were decreased (Steffen et al., 2017). The pMC concentrations at our WLE monitoring stations varied from 1.2-10.1 µg L^{-1} on 04 August 2014 during this crisis.

Data Availability

The entire dataset detailed in this manuscript can be freely accessed through the NOAA National Centers for Environmental Information (NCEI) data repository at https://www.ncei.noaa.gov/. The data collection is titled “Physical, chemical, and biological water quality monitoring data to support detection of Harmful Algal Blooms (HABs) in western Lake Erie, collected by the Great Lakes Environmental Research Laboratory and the Cooperative Institute for Great Lakes Research since 2012”. The digital object identifier is https://doi.org/10.25921/11da-3x54. The data presented in this manuscript are available in three separate accession files within this collection including: 2012 to 2018 data is available under NCEI Accession 0187718 v2.2 at https://www.ncei.noaa.gov/archive/accession/0187718; 2019 data is available under NCEI Accession 0209116 v1.1 at https://www.ncei.noaa.gov/archive/accession/0209116; 2020 to 2021 data is available under NCEI Accession 0254720 v1.1 at https://www.ncei.noaa.gov/archive/accession/0254720 (Cooperative Institute for Great Lakes Research, University of Michigan; NOAA Great Lakes Environmental Research Laboratory, 2019). Future data will be added to this collection as it becomes available.

Conclusions

The western Lake Erie data collected and compiled by NOAA GLERL and CIGLR represent ten years of routine water quality monitoring to detect, track, and predict cyanobacterial HAB events in an area of the Great Lakes that has experienced significant environmental degradation. This ongoing program provides a service to the region and contributes data for investigating the nuanced dynamics of potentially toxic HABs fueled by excess nutrient loading into the WLE basin.

In addition to serving the need for generating data to improve remote sensing efforts, (see: Sayers et al., 2016, 2019; Stumpf et al., 2016; Avouris and Ortiz, 2019; Bosse et al., 2019; Vander Woude et al., 2019; Pirasteh et al., 2020; Xu et al., 2022) this monitoring program will continue to serve stakeholders and communities in the Laurentian Great Lakes. As this program has grown, so too has the scope of application of its dataset. For instance, this dataset has assisted in assessing progress toward binational nutrient loading reduction efforts on lake basin concentrations of phosphorus. It has also been used towards determining the significance of nitrogen in bloom formation and toxicity (Gobler et al., 2016; Newell et al., 2019; Hoffman et al., 2022). Other research groups have taken advantage of this dataset to investigate microbial and algal community dynamics (Berry et al., 2017; Kharbush et al., 2019; Smith et al., 2021; Marino et al., 2022; Smith et al., 2022), the genomic diversity of cyanobacteria in the WLE basin (Yancey et al., 2022), and hydrogen peroxide production and dynamics within blooms (Corey et al., 2016; Pandey et al., 2022). We anticipate this dataset will continue to be useful for addressing the complex relationships of abiotic and biotic factors contributing to WLE HABs. Long-term monitoring programs like this one provide consistent data which is useful for identifying patterns and variations within the ecosystem and in determining the root cause of those changes.

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As the sites and parameters of this monitoring program have already changed to adapt to the needs of research, this program will continue to evolve as we consider adding parameters that encompass broader aspects of bloom dynamics. For example, lake samples can be analyzed for genomic data that will provide insights on the ability of the current phytoplankton community to produce microcystins. This decadal history has already been an invaluable resource for the research community, and it will continue to enrich our collective scientific knowledge of water quality dynamics in western Lake Erie.

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Author Contributions

Anna G Boegehold prepared the manuscript. Ashley M. Burtner performed field sampling, laboratory processing, data processing, QA/QC and data management, manuscript revision, data curation. Andrew Camilleri performed field sampling, laboratory processing, manuscript revision. Glenn Carter performed field sampling, laboratory processing, data processing, methodology. Paul DenUyl performed field sampling, laboratory processing, manuscript revision. David Fanslow performed field sampling, laboratory processing. Deanna Fyffe Semenyuk performed field sampling, laboratory processing, manuscript revision. Casey Godwin was responsible for project administration, supervision, visualization, manuscript revision, methodology, field sampling, sample processing. Duane Gossiaux performed field sampling, laboratory processing, manuscript revision, methodology. Tom Johengen was responsible for project administration, supervision, field sampling, methodology. Holly Kelchner performed field sampling, laboratory processing, manuscript revision. Christine Kitchens performed field sampling, laboratory processing, data processing, manuscript revision. Lacey A. Mason was responsible for data curation, manuscript revision. Kelly McCabe performed field sampling, laboratory processing, manuscript revision, methodology. Danna Palladino performed field sampling, laboratory processing, data processing, manuscript revision. Dack Stuart performed field sampling, data processing. Henry Vanderploeg was responsible for project administration, supervision. Reagan Errera was responsible for project administration, supervision, Visualization, manuscript revision, methodology.

Competing Interests

The authors declare that they have no conflict of interest

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