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Summer phytoplankton nutrient limitation in Maumee Bay of Lake Erie during high-flow and low-flow years

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ABSTRACT

Algal production in Maumee Bay in western Lake Erie is highly affected by inputs of nitrogen (N) and phosphorus (P) from the Maumee River, which drains predominantly agricultural lands, leading to the formation of cyanobacterial blooms. In a 3-year study, precipitation and discharge ranged from relatively low (2012) to relatively high (2011) with corresponding changes in the size of the cyanobacterial bloom. This study aimed to quantify the relation between river discharge and algal nutrient limitation in Maumee Bay. During the summer growing seasons, 20 nutrient enrichment bioassays were performed to determine which nutrient (P or N) might limit phytoplankton growth; and ambient N and P concentrations were monitored. The bioassays suggested that phytoplankton growth shifted from P-limited to N-limited during summer of the low and intermediate discharge years (2012 and 2010, respectively), whereas during the high discharge year (2011) phytoplankton were nutrient-replete before becoming N-limited. Phosphorus-replete growth during the high discharge year likely was due to high P loads from the river and dissolved P concentrations greater than 1 μmol/L. Symptoms of N-limited growth occurred during August and September in all three years and during July of 2012 when NO_3^- plus NH_4^+ concentration was less than 7.29 μmol/L suggesting low or no correspondence between N-limitation and size of the cyanobacterial bloom. Occurrence of a relatively small cyanobacterial bloom in 2012 following the record-breaking bloom in 2011 suggests the possibility of fast-reversal of eutrophication in Maumee Bay if P loading from the watershed could be decreased.

Introduction

Land use is a major factor in determining nutrient export from watersheds to lakes (Carpenter et al., 1998). Nutrient export from agricultural watersheds can degrade water quality of lakes by increasing concentrations of potentially limiting nutrients (Tilman et al., 2001), and the rates of export are highly dependent on weather patterns because increases of rainfall accelerate nutrient loading (Haygarth et al., 1999). Phosphorus (P) has long been recognized as the main limiting nutrient in freshwater ecosystems (Reynolds, 2006; Schindler, 1977) and excessive P loading often results in symptoms of eutrophication including cyanobacterial blooms (Downing et al., 2001).

During the mid-1900s Lake Erie (North America) was eutrophic, with dense cyanobacterial blooms, due to excessive P loading (Davis, 1964; Matisoff and Ciborowski, 2005). Regulations set by the United States and Canada in the 1970s restricted P loads into the lake, and water quality quickly improved (DePinto et al., 1986). Cyanobacterial blooms were absent during the 1980s and early 1990s (Makarewicz, 1993). However, following the brief (~20 years) period of recovery and despite the ongoing P regulations, western Lake Erie has returned to eutrophic conditions (Conroy et al., 2005b), and harmful cyanobacterial blooms have been an annual occurrence since the mid-1990s (Millie et al., 2009). The return of cyanobacterial blooms has corresponded to a substantial increase in dissolved reactive P (DRP) loading from the Maumee River (Joosse and Baker, 2011). Agricultural non-point sources are considered to be the main contributor to re-eutrophication of Lake Erie (Richards et al., 2012).

Phytoplankton primary production in Lake Erie water generally has been considered to be P-limited throughout the improving P conditions of the 1980s (Hartig and Wallen, 1984) and into the early 2000s (Wilhelm et al., 2003). However, given the increasing rate of DRP loading over the past 15 years (Joosse and Baker, 2011), P-limitation may be decreasing and the importance of N may be increasing. Furthermore,

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high densities of exotic *Dreissena* mussels, which excrete ammonium and dissolved P at low N-to-P ratios, may also be driving a shift to N-limitation (Conroy et al., 2005a). Finally, nitrate concentration and the total N-to-total P (TN:TP) decline throughout summer (Chaffin et al., 2011) suggesting that N-limitation may become important in late summer. Conversely, *Microcystis aeruginosa* dominates the current cyanobacterial blooms in the western basin (Millie et al., 2009). *Microcystis* is not a N-fixer, and so its dominance would not necessarily suggest N-limitation. Because the above evidence indicates the possibility of both P and N limitation, a reassessment of Maumee Bay phytoplankton nutrient status is in order.

The Maumee River watershed is the largest watershed in the Great Lakes basin and is 87.8% agricultural (Han et al., 2011). The Maumee River loads high amounts of suspended sediments (Richards et al., 2008) and nutrients (Baker and Richards, 2002) into Maumee Bay in the southwest end of Lake Erie. Most of the nutrient export from the Maumee River occurs during large rainstorms (Richards et al., 2010), and large rainstorms that occur during the spring months can result in summer cyanobacterial blooms in Lake Erie (Stumpf et al., 2012).

The Maumee River loaded high amounts of P and N into Lake Erie during 2011 which resulted in a record-breaking cyanobacterial bloom (Bridgeman et al., 2013; Stumpf et al., 2012). In contrast, 2012 was a very dry year. There were two goals for this study, 1) compare nutrient limitation of phytoplankton in Maumee Bay during low-flow and high-flow years, and 2) determine the concentrations of N, P, and TN:TP ratios that will induce N or P limitation of phytoplankton growth. To test the hypothesis that increased nutrient loading via river discharge results in increased size of the cyanobacterial bloom in Maumee Bay, algal nutrient limitation was assessed over a 3-year period (2010–2012). Fortuitously, 2011 was one of the wettest and 2012 one of the driest on record (Stumpf et al., 2012). During the summers of 2010, 2011, and 2012 we monitored N and P concentrations, chlorophyll *a* levels, and light climate, and also conducted 20 nutrient enrichment bioassays with Maumee Bay water to determine nutrient limitation of phytoplankton growth.

Methods

Maumee River discharge and nutrient loading

The Maumee River cumulative discharge, cumulative total N load, and cumulative total P load were calculated for 1 March through 30 June for years 2010, 2011, and 2012 using the tributary loading tool provided by the Heidelberg University National Center for Water Quality Research (NCWQR) (downloaded from: http://www.heidelberg.edu/academiclife/distinctive/ncwqr/data, accessed 29 January, 2013). Cumulative discharge and loads were calculated from 1 March through 30 June because this time period was the best predictor of cyanobacterial bloom magnitude in Lake Erie (Stumpf et al., 2012). Total N load was calculated as the sum of total Kjeldahl N (TKN) and nitrate loads.

Field methods

This research was conducted at site MB18 (N 41°44′51″, W 83°24′5″) in Maumee Bay from early June to late September in 2010, 2011, and 2012. Site MB18 has a depth of 2.5 m. Water was collected over the entire water column using a metal-free, 2-meter long integrated tube sampler constructed from PVC tubing. Water for nutrient analysis was transferred to 250-mL acid-washed polyethylene bottles and kept on ice during transportation back to the laboratory. Water for nutrient enrichment bioassays was poured into 20-L acid-washed polyethylene containers and kept in a large dark cooler.

Vertical profiles of underwater photosynthetic active radiation (PAR) were recorded, as in Chaffin et al. (2011). The PAR profiles were used to determine the light attenuation coefficient (K_d). We then calculated mean PAR (Guildford et al., 2005) using K_d, the light intensity at the lake surface, and the depth of site MB18 (2.5 m) rather than the lesser mixing depth because site MB18 does not thermally stratify. Mean PAR is presented as percent of surface light (Guildford et al., 2005). During 2012 PAR profiles were not completed on every sample trip, but Secchi disk depth was measured on all trips. Mean PAR for these dates was calculated based on the relationship between Secchi disk depth and the light attenuation coefficient at site MB18 (Bridgeman unpublished data).

Nutrient analysis

Total phosphorus (TP) and total Kjeldahl nitrogen (TKN) concentrations were determined on unfiltered water. Dissolved inorganic nutrient [dissolved reactive P (DRP), NO_3^-, NH_4^+] concentrations were determined on water samples filtered through a 0.45-μm membrane filter. After filtering, all nutrient samples were stored at −20 °C until analyses at the National Center for Water Quality Research (NCWQR) at Heidelberg University (Tiffin, Ohio, USA) using USA Environmental Protection Agency protocols (Richards et al., 2010). Details on methods and minimum detection concentrations are available from NCWQR (at http://www.heidelberg.edu/academiclife/distinctive/ncwqr).

Bioassays

Phytoplankton nutrient limitation was determined monthly (June, July, August, September) in 2010 and 2011 and 12 times during 2012 by P- and N-enrichment bioassays (Schelske, 1984). For the incubations, 200 mL of lake water were poured into acid-washed 250-mL polycarbonate flasks. Treatments included the enrichment of 10 μmol/L P (+P; KH_2PO_4), 520 μmol/L N (+N; 500 μmol/L $NaNO_3$ and 20 μmol/L NH_4^+ [$(NH_4)_2SO_4$]), and combination P and N enrichment (+P&N). Controls were used in which only deionized water was added to lake water at a volume that matched the volume of nutrient additions. Each treatment was replicated in three separate flasks. Flasks were incubated in a growth chamber (Percival model: E-36HO, Fontana, Wisconsin, USA) at lake temperature (19.1 °C to 27.5 °C) at the time of collection under a light intensity of 300–350 μmol photon/m^2/s on a 12:12 h light:dark cycle. This light intensity approximates the mean PAR of western Lake Erie (Chaffin et al., 2011), and previous Lake Erie bioassays conducted in incubation chambers used similar light intensities (Moon and Carrick, 2007). Flasks were inverted several times to prevent settling and randomly rearranged in the growth chamber daily (Moon and Carrick, 2007).

Phytoplankton abundance was estimated as chlorophyll (chl) *a* on initial samples and after 48 h of incubation. During 2010 and 2011 chl *a* was extracted using dimethylsulfoxide and quantified by absorbance (Chaffin et al., 2012). During 2012 chl *a* was extracted from the filters using N-N-dimethylformamide and quantified by fluorometry (Speziale et al., 1984). These two methods gave very similar results from split water samples that were analyzed for chl *a* during 2007 and 2008 (Chaffin, 2009).

Microcystis biovolume

Microcystis biovolume was measured to compare bloom intensity among the three years. Data for 2010 and 2011 were accessed from Bridgeman et al. (2013) and data for 2012 were determined following the methods of Bridgeman et al. (2013).

Data analysis

Final chl *a* concentration of each nutrient enrichment experiment was subjected to a normality test. Normally distributed data were analyzed with a one-way analysis of variance (ANOVA) and post hoc Tukey test. Non-normally distributed data were first log transformed

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