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Lake Erie's ecological history reconstructed from the sedimentary record

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ABSTRACT

We evaluated the recent ecological history of Lake Erie from diatoms and geochemistry in sediment cores. Two major transition points in the ecology of the western basin (WB; 1985 and 2008) and central basin (CB; 1935 and 1982) were defined. Changes in abundance of diatom eutrophic indicators and geochemical markers were interpreted as a degradation in water quality after 1935 due to the effects of increased population, agriculture, and industrialization until abatement measures were enacted in the 1970s and 1980s. Diatom indicators suggested modest recovery from eutrophic conditions in Lake Erie, however diatom-inferred total phosphorus suggested that despite abatement efforts total phosphorus was not reduced below pre-impact levels. The effects on diatoms of increased temperature and dissolved silica also became apparent in the 1980s, and in the WB recent shifts were likely caused by increased pollution and recent climatic warming. Based on stratigraphic changes since 1985, the diatom trajectory suggests the phytoplankton of Lake Erie will likely remain in a state of flux for the near future due to a variety of countervailing impacts including unknown effects of mitigation efforts, legacy pollution, climate change, and changing upstream conditions.

Introduction

Lake Erie has undergone dramatic swings in water quality over the past century prompting equally dramatic legislation to manage this freshwater resource. Pollution from expanding populations in the Lake Erie watershed was aggravated by inadequate water treatment facilities, the introduction of phosphate detergents, as well as increasing agriculture and industrialization in the first half of the twentieth century (Beeton, 1969; Hartman, 1973). This in turn led to cyanophyte and *Cladophora* blooms, increased algal biomass, and continued anoxia by the 1960s (Beeton, 1961; Davis, 1964; Taft and Taft, 1971). These deteriorating conditions led to a legislative response at the Federal level. The bi-national Great Lakes Water Quality Agreement (GLWQA) of 1972 (IJC, 1972) and amended in 1978 established point source P removal programs and P content reduction for detergents (IJC, 1978). Lake-wide targets for phosphorus (11,000 MT per year) set by the 1983 supplement to the GLWQA also included reductions from nonpoint sources (IJC, 1983).

There has been considerable, though unsystematic, monitoring of the algae to track changes in Lake Erie over roughly the past century (Allinger and Reavie, 2013). These studies of Lake Erie algae have been conducted by different researchers and agencies using different methods for different purposes (Ho and Michalak, 2015). Some studies linked the changes in algal assemblages and abundance with anthropogenic stressors (Davis, 1964; Beeton, 1969; Nicholls and Hopkins, 1993; Chaffin et al., 2011). Algal monitoring data in published research was intermittent prior to 1970 (e.g. Kellicott, 1878; Vorce, 1881; Snow, 1903; Chandler, 1940, 1942, 1944; Chandler and Weeks, 1945). The lack of study led to a need for proxy methods like paleolimnology to understand historical environmental conditions.

Examining paleolimnological data with a consideration of historic algal monitoring and environmental stressor data can provide resolution of the patterns and causes of algal succession through time, thereby improving our understanding of the lake's ecology. There have been two paleolimnological studies examining microfloral fossils obtained from sediment cores that were used to interpret changes in Lake Erie's algal ecology. Harris and Vollenweider (1982) used qualitative methods to examine a core from the central basin (CB) dated by pollen analysis and concluded that major eutrophication to the lake occurred in the twentieth century. Stoermer et al. (1987) following the Harris and Vollenweider (1982) study collected a core from the same CB location, but used newer dating methods. Stoermer et al. (1987) confirmed Harris and Vollenweider's (1982) conclusion that settlement in the Lake Erie basin had substantial impact on the lake's ecology; however, this occurred later than previously thought. Stoermer et al. (1987) interpreted that early settlement effects occurred from 1860 to 1900 and stated that disruptive change occurred from 1900 to 1920, then rapid change to eutrophic conditions from ~1920 to 1935. Moderate eutrophic conditions prevailed from 1935 to 1945 followed by gross eutrophication from 1945 until 1979, their coring date.

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Substantial ecological changes have occurred in the lake since Stoermer et al.'s (1987) study. After a period of apparent recovery through the 1980s (DePinto et al., 1986; Makarewicz and Bertram, 1991; Ludsin et al., 2001) the lake once again was beset by algal blooms and notable CB hypoxia (Bridgeman and Penamon, 2010; Millie et al., 2009; Michalak et al., 2013). The city of Toledo shut off water to 500,000 residents for two days in 2014 after their water treatment facility was overwhelmed by microcystin toxins (Wilson, 2014). Agricultural activities in the Maumee watershed were implicated in the development of blooms of *Microcystis* spp. (Baker et al., 2014; Kane et al., 2014). The impact from this ecological degradation was serious enough to prompt legislative action and best management farming practices were made mandatory by the state of Ohio (OH SB150 | 2013–2014 | 130th General Assembly). The International Joint Commission in 2015 recommended that the target TP load for the western and central basins be set at 4300 MT, a reduction of 46% from the observed average load from 2003 through 2011 (IJC, 2015). The expected TP concentration for the CB is 6 μg/L if current loading targets are reached (US EPA, 2015).

There are now renewed efforts to understand problems in the lake because of these recent events. We evaluated new sediment cores using stratigraphic indicators (diatom assemblages, inferred phosphorus, geochemistry) and integrated these results with stressor data acquired from historic records. We aimed to provide an update to the previous paleolimnological work and place the more recent ecology of Lake Erie in a longer historical context. We hypothesized that the recent changes from the 1970s and 1980s constituted a major transition point in the diatom assemblages relative to previous changes seen in the paleolimnological record. We analyzed the relationships among the diatom assemblages, water quality, and stressors (agriculture, urbanization, and population) to refine our understanding of these changes and forecasted the trajectory of the lake's pelagic ecology.

Methods

Lake Erie Western and Central Basins

The Lake Erie western basin (WB) eastern edge lies from the tip of Cedar Point, Ohio, on a line north to Point Pelee, Ontario (Fig. 1). It holds 5.1% of the volume of Lake Erie and covers 12.8% of the area of the lake. Most of the bottom is flat and lies between eight and 11 m in depth with the deepest sounding at 19 m. A number of small islands and shoals partially separate it from the CB.

The CB extends eastward from the WB and is separated from the eastern basin by a shallow sand and gravel bar extending from Erie, Pennsylvania, to the base of Long Point, Ontario. The CB contains 63% of the lake's volume and makes up 62.9% of the area. The bottom is flat except for the rising slope of glacial material extending southsoutheast from Point Pelee, Ontario. The basin has an average depth of 19 m and a maximum depth of 26 m (Bolsenga and Herdendorf, 1993).

The WB receives 95% of the total inflow to Lake Erie through the Detroit, Maumee, and Sandusky rivers (Hartman, 1973). The WB has a much higher particulate load than the CB because of these inflows and resuspension of bottom sediments. These rivers also carry most of the anthropogenic inputs to the lake (Marvin et al., 2007; Matisoff and Carson, 2014). The retention/replacement time for the water in all of Lake Erie is 2.6 years (Ludsin et al., 2001).

Sediment core collection and analysis

Sediment cores were collected from the WB (Lat. 41.892°, Long. −82.716°; 35 cm long) and from the CB (Lat. 41.998°, Long. −81.650°; 49 cm long) in July 2011 (Fig. 1). The WB core site was selected as an area likely to have a high sedimentation rate, and the CB site was selected to correspond with cores collected previously for paleolimnological studies by Harris and Vollenweider (1982) and Stoermer et al. (1987). Cores were collected with an Ocean Instruments model MC-400 multi corer. The surface sediment in the cores was stabilized with Zorbitrol polymer gel to minimize disturbance during extrusion and transport (Tomkins et al., 2008). The cores were sectioned at 0.5-cm intervals to 10 cm then at 1-cm intervals to the bottom of the core. Subsurface piston cores were also collected at both locations under the assumption that deeper sediment intervals may be needed to extend historical measurements. Diatom preservation was deemed very poor in these deeper cores, so we relied on multi-corer collections for this investigation.

Sediment cores were analyzed for ^{210}Pb activity to determine age and sediment accumulation rates for the past ~150 years (Electronic Supplementary Material (ESM) Appendix S1). ^{210}Pb activity was measured from its daughter product, ^{210}Po, which is considered to be in secular equilibrium with the parent isotope. Aliquots (0.5–3.0 g) of freeze-dried sediment were spiked with a known quantity of ^{209}Po (~4 pCi/g) as an internal yield tracer and the isotopes distilled at 550 °C after treatment with concentrated HCl. Polonium isotopes were then directly plated onto Au planchets from a 0.5 N HCl solution. Activity was measured for $1–3 \times 10^5$ s using an Ortec alpha spectrometry system. Supported ^{210}Pb was estimated by mean activity in the lowest core samples and subtracted from up core activity to calculate unsupported ^{210}Pb. Core dates and sedimentation rates were calculated using the constant rate of supply model (Appleby and Oldfield, 1978; Appleby, 2001). Dating and sedimentation errors represented first-order propagation of counting uncertainty (Binford, 1990). For the WB core, which did not contain supported background concentrations of ^{210}Pb at depth, gamma spectrometry was used to measure ^{137}Cs in the core (Ritchie and McHenry, 1973). Activity was measured using an Ortec-EGG (Oak Ridge, TN) High-Purity, Germanium Crystal Well, Photon Detector (Well Detector) coupled to a Digital Gamma-Ray Spectrometer

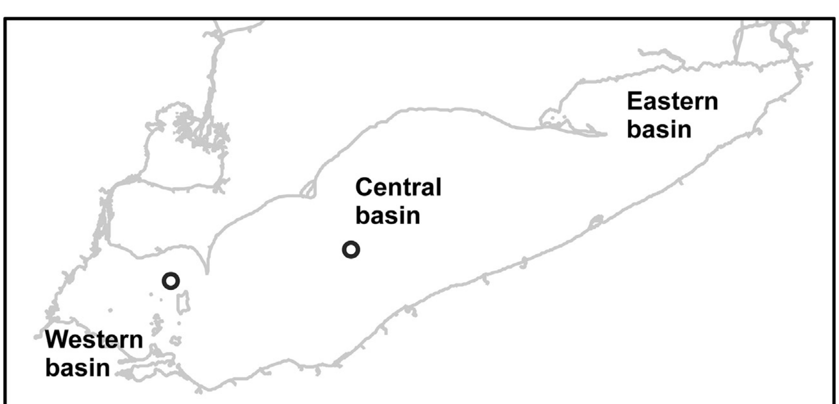

Fig. 1. Coring locations in the central and western and basins of Lake Erie.

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