

## Review

## The re-eutrophication of Lake Erie: Harmful algal blooms and hypoxia



Susan B. Watson<sup>a,\*</sup>, Carol Miller<sup>b</sup>, George Arhonditsis<sup>c</sup>, Gregory L. Boyer<sup>d</sup>, Wayne Carmichael<sup>e</sup>, Murray N. Charlton<sup>f</sup>, Remegio Confesor<sup>g</sup>, David C. Depew<sup>a</sup>, Tomas O. Höök<sup>h</sup>, Stuart A. Ludsin<sup>i</sup>, Gerald Matisoff<sup>j</sup>, Shawn P. McElmurry<sup>b</sup>, Michael W. Murray<sup>k</sup>, R. Peter Richards<sup>n</sup>, Yerubandi R. Rao<sup>a</sup>, Morgan M. Steffen<sup>l</sup>, Steven W. Wilhelm<sup>m</sup>

<sup>a</sup> Watershed Hydrology and Ecology Research Division, Environment and Climate Change Canada, Burlington, ON L7S 1A1, Canada

<sup>b</sup> Department of Civil and Environmental Engineering, Urban Watershed Environmental Research Group, Wayne State University, Detroit, MI 48202, USA

<sup>c</sup> Department of Physical & Environmental Sciences, University of Toronto, Toronto, ON M1C 1A4, Canada

<sup>d</sup> Department of Chemistry, State University of New York – College of Environmental Science and Forestry, Syracuse, NY 13210, USA

<sup>e</sup> Department of Biological Sciences, Wright State University, Dayton, OH 45435, USA

<sup>f</sup> Burlington, ON L7 T 3C9, Canada

<sup>g</sup> National Center for Water Quality Research, Heidelberg University, Tiffin, OH 44883, USA

<sup>h</sup> Department of Forestry and Natural Resources, Purdue University, West Lafayette, IN 47907-2033, USA

<sup>i</sup> Aquatic Ecology Laboratory, Department of Evolution, Ecology, and Organismal Biology, The Ohio State University, Columbus, OH 43212, USA

<sup>j</sup> Department of Earth, Environmental and Planetary Sciences Case Western Reserve University, Cleveland, OH 44106-7216, USA

<sup>k</sup> National Wildlife Federation Great Lakes Regional Center, Ann Arbor, MI 48104, USA

<sup>l</sup> Department of Biology, James Madison University, Harrisonburg, VA 22807, USA

<sup>n</sup> Department of Microbiology, The University of Tennessee, Knoxville, TN 37996-0845, USA

<sup>m</sup> Oberlin, Ohio 44074 USA

## ARTICLE INFO

## Article history:

Received 25 September 2015

Received in revised form 21 April 2016

Accepted 22 April 2016

## Keywords:

*Microcystis*

*Cladophora*

Cyanobacteria

Climate change

Internal loading

Nutrient management models

## ABSTRACT

Lake Erie supplies drinking water to more than 11 million consumers, processes millions of gallons of wastewater, provides important species habitat and supports a substantial industrial sector, with >\$50 billion annual income to tourism, recreational boating, shipping, fisheries, and other industries. These and other key ecosystem services are currently threatened by an excess supply of nutrients, manifested in particular by increases in the magnitude and extent of harmful planktonic and benthic algal blooms (HABs) and hypoxia. Widespread concern for this important international waterbody has been manifested in a strong focus of scientific and public material on the subject, and commitments for Canada-US remedial actions in recent agreements among Federal, Provincial and State agencies. This review provides a retrospective synthesis of past and current nutrient inputs, impairments by planktonic and benthic HABs and hypoxia, modelling and Best Management Practices in the Lake Erie basin. The results demonstrate that phosphorus reduction is of primary importance, but the effects of climate, nitrogen and other factors should also be considered in the context of adaptive management. Actions to reduce nutrient levels by targeted Best Management Practices will likely need to be tailored for soil types, topography, and farming practices.

© 2016 Published by Elsevier B.V.

## Contents

1. Introduction	45
2. Signs of impairment: harmful blooms and hypoxia	45
2.1. Harmful algal blooms (HABs)	45
2.1.1. Planktonic HABs	45
2.1.2. Benthic HABs	49

\* Corresponding author. Tel.: +1 905 336 4759.

E-mail address: [sue.watson@canada.ca](mailto:sue.watson@canada.ca) (S.B. Watson).

2.2.	Hypoxia	50
2.2.1.	Causes of hypoxia	50
2.2.2.	Climate change and hypoxia	51
2.2.3.	Effects on foodwebs	52
3.	Nutrient loading to Lake Erie	52
3.1.	External loading: nutrient management and models	52
3.1.1.	Advancements in our understanding of the role of phosphorus loading	52
3.1.2.	Recent modelling work in Lake Erie	53
3.2.	External loading: effectiveness of urban and agricultural BMPs in reducing phosphorus loads	54
3.2.1.	Urban BMPs	54
3.2.2.	BMPs in agricultural and rural environments	55
3.2.3.	Summary	56
3.3.	Coastal wetlands	57
3.4.	Internal loading: phosphorus	58
3.4.1.	Underlying mechanisms of internal loading and their importance in Lake Erie	58
3.4.2.	Inter-basin transfers	58
3.4.3.	Biological recycling	59
3.4.4.	Sediment P regeneration to the water column	59
3.4.5.	Implications of internal loading/recycling	60
4.	Conclusions	60
	Acknowledgements	60
	References	60

## 1. Introduction

Lake Erie, the southernmost, warmest, shallowest, and most biologically productive of the Great Lakes, supplies drinking water to more than 11 million consumers, processes millions of gallons of wastewater, provides habitat for economically, ecologically, and culturally important biota, and supports a substantial industrial sector, with annual income to tourism, recreational boating, shipping, fisheries, and other industries of over \$50 billion (LEIA, 2012). These and other key ecosystem services are now increasingly threatened by eutrophication, manifested by increases in the magnitude, duration, and extent of harmful algal blooms (HABs; Higgins et al., 2008; Michalak et al., 2013; Steffen et al., 2014) and hypoxia (Zhou et al., 2013, 2015; Scavia et al., 2014). In recognition of these ecosystem impacts and the need to develop a sustained restoration and management programme, the International Joint Commission (IJC) commissioned the Lake Erie Ecosystem Priority (LEEP) taskforce in 2012 to evaluate current conditions, identify knowledge and monitoring gaps, provide guidance for management targets, and engage public interest and support (International Joint Commission (IJC), 2014; Watson et al., 2013). This initial IJC review, and subsequent work by binational taskforces, led to commitments for remedial action in the recently renewed Canada-USA Great Lakes Water Quality Agreement (GLWQA). Nutrients, hypoxia and algal biomass are addressed under Annex 4 of this Agreement, with specific references to setting interim total phosphorus (TP) load and basin-specific concentration targets for Lake Erie. Similarly, the 2014 renewed Canada-Ontario Agreement (COA) specifically highlights Lake Erie in the objectives to set loading and concentration targets.

Scavia et al. (2014) reviewed and evaluated recent eutrophication-related trends in Lake Erie, and developed response curves to guide hypoxia-based loading targets. Following a binational remedial effort, central basin hypoxia and west-central phytoplankton biomass showed strong declines in the late 1970s and early 1980s, followed by a general increase in these parameters since the mid-1990s which the authors attributed to increased agricultural loading of soluble reactive phosphorus (SRP). The authors also concluded that reducing central basin hypoxic area to levels observed in the early 1990s (ca. 2000 km<sup>2</sup>) requires the reduction of TP loads by 46% from the 2003 to 2011 average or SRP loads by 78% from the 2005 to 2011 average, and that

those reductions would protect fish habitat. Based on an analysis of long-term records of climate and nutrient data, however, Zhou et al. (2015) reported that a record-breaking hypoxic event in 2012 followed a period of drought and low tributary flow, while in 2011, the largest cyanobacterial bloom of the decade (as measured by remote sensing; e.g., Michalak et al., 2013) was accompanied by mild hypoxia. These and other authors have concluded that both the extent and severity of central basin hypoxia and west basin planktonic algal blooms show strong, but fundamentally different relationships with the timing and volume of spring-summer river discharge and associated nutrient inputs (Rucinski et al., 2014; Stow et al., 2015). Furthermore there is evidence that together with P, nitrogen plays an important role in planktonic bloom composition and toxicity, while light and temperature have a major effect on benthic bloom development (Auer et al., 2010; Steffen et al., 2014; Davis et al., 2015). These and other studies demonstrate that P reduction is of primary importance, but the effects of climate, nitrogen and other factors may need to be considered when developing an adaptable response which may require a tailored approach, such as the application of targeted Best Management Practices (BMPs) that account for soil types, topography, and farming practices (McElmurry et al., 2013). A proliferation of scientific and other published material has led to a global awareness of the issues challenging this lake (e.g., a Scopus database search yields >300 documents for the terms 'blooms', 'nutrients' and 'hypoxia' in Lake Erie over the past 5 years) and widespread media and website coverage (e.g., [http://www.glerl.noaa.gov/res/HABs\\_and\\_Hypoxia/](http://www.glerl.noaa.gov/res/HABs_and_Hypoxia/); <http://www.cbc.ca/news/trending/algae-blooms-lake-erie-lake-st-clair-nasa-photoshow-1.3179298>). This review presents a synopsis of past, present and future nutrient loading, HABs and other impairments in this lake and options for mitigation.

## 2. Signs of impairment: harmful blooms and hypoxia

### 2.1. Harmful algal blooms (HABs)

#### 2.1.1. Planktonic HABs

In most regions of North America the majority of planktonic HABs are caused by cyanobacteria (cHABs), most often where surface waters receive high inputs of growth-limiting nutrients (phosphorus, P and nitrogen, N). Global climate and regional

weather patterns can also increase the risk of cHABs, which are generally favoured by nutrient-enriched runoff from intense rain events, warm temperatures, low flushing, water column stability and prolonged ice-free growing seasons (Paerl and Otten, 2013; Watson et al., 2015).

Recently, cHABs have become particularly problematic in Lake Erie, the shallowest, warmest and most populated of the Great Lakes, and most vulnerable to disturbance. Lake Erie has undergone some significant regime shifts over the past century, manifested in disturbing trends in the risk and severity of harmful algal blooms (HABs). Basin development, eutrophication, species invasion, increases in diffuse and point-source loading, climate change and restoration efforts have been accompanied by salient changes in plankton community structure and productivity (Allinger and Reavie, 2013). Analyses of long term plankton and sediment data have shown that algal blooms are not new to this Lake, but have changed dramatically in species composition and dynamics over the past fifty years (Allinger and Reavie, 2013; Steffen et al., 2014). Earlier eutrophication during the mid-late 1900s resulted in extensive cyanobacterial blooms across many of the Great Lakes, despite their size. These were occurring during a time when the threat of toxins was generally unrecognized, and the main concern was with aesthetics and taste-odour (Carmichael, 2008). Dense spring and fall blooms were driven by excessive wastewater-dominated nutrient loading, and were generally composed of eukaryotic algae (diatoms such as *Asterionella*, *Cyclotella*, *Fragilaria*, *Synedra*, *Aulacoseira*, along with chlorophytes and dinoflagellates), with outbreaks of  $N_2$ -fixing (*Aphanizomenon* spp.; *Anabaena*)<sup>1</sup> and non-fixing cyanobacteria (*Coelosphaerium*, *Anacystis*, *Oscillatoria* (syn. *Planktothrix*), *Merismopedia*, *Woronochinia*, *Microcystis*, *Pseudanabaena*) in the mid-summer (Munawar et al., 2008; Allinger and Reavie, 2013). Casper (1965), for example, described a severe surface bloom that followed a period of calm weather in late summer 1964, extending across some 2000 km<sup>2</sup> of the west basin. This bloom was dominated by an assemblage of cyanobacteria (*Anacystis cynea* (syn. *Microcystis aeruginosa*), *Oscillatoria* (syn. *Planktothrix*), *Aphanizomenon holsaticum* (syn. *Aph. flos-aquae*), *Anabaena circinalis*) and the green colonial flagellate *Carteria*.

To address the advanced eutrophication of Lake Erie and the other Great Lakes, binational restoration efforts were implemented in 1972 with the signing of the Canada-US GLWQA (Hasler, 1969). Within ten years, binational remediation, aimed largely at point source municipal and industrial inputs, reduced total P loading into Lake Erie to half the peak levels observed in 1968 (Makarewicz and Bertram, 1991; Charlton et al., 1993). Total phytoplankton biomass in the lake was also significantly reduced, but species of *Aphanizomenon*, *Anabaena*, *Planktothrix* and *Microcystis* remained present in these assemblages (Kane et al., 2009; Allinger and Reavie, 2013). Since the 1990s, several other changes have occurred which have had profound effects on Lake Erie. Most notably, widespread colonization by dreissenid mussels and other invasive species, increasing non-point agricultural loading of more bioavailable P (i.e., SRP) and climate change have engineered ecosystem-wide changes in the physical, chemical and biological regimes of the Great Lakes (Hecky et al., 2004; Conroy et al., 2005; Jiang et al., 2015). In Lake Erie, this has been accompanied by a resurgence of algal biomass, particularly in the west basin, and major shift in the dominant bloom species and their spatial and temporal dynamics starting in the early 2000s (Barbiero et al., 2006; Millie et al., 2009). Planktonic blooms in this lake have since been characterized by a rise in the predominance and severity of cHABs and the dominance of more toxic taxa such as *Microcystis*

(Munawar et al., 2008; Allinger and Reavie, 2013; Kane et al., 2014).

The majority of these blooms occur in the west basin of Lake Erie, where the mid-summer cyanobacterial peaks of the late 1900s have been replaced by severe cHABs which cover large areas and can persist throughout the summer and late fall (Michalak et al., 2013; Steffen et al., 2014). Extensive cHABs dominated by *Microcystis* and other potentially toxic cyanobacteria (*Dolichospermum*, *Planktothrix*) are now an annual event across the open waters of the basin (e.g., Rinta-Kanto et al., 2009a; Steffen et al., 2014). Biomass is generally lower in the more oligotrophic offshore regions of the central and east basins, although there is some indication of a recent increase in cyanobacteria in the summer plankton assemblages in these basins (Allinger and Reavie, 2013; Kane et al., 2014), while narrow shoreline bands of dense cHABs have arisen over the past several years; for example following the 2012 upwelling and extensive fish kill along the northern shore (Rao et al., 2014).

Brittain et al. (2000) first galvanized public attention to the serious implications of the rise in predominance of *Microcystis* with their report of significant levels of the hepatotoxin microcystin-LR (MC-LR) in the west basin of Lake Erie from a 1995 bloom of *Microcystis aeruginosa*, a widespread producer of these toxins in north temperate freshwaters (Wilson et al., 2005). MC-LR is among the most toxic of the >100 different microcystin varieties (congeners) characterized to date, and has been identified in numerous bloom samples from the Great Lakes along with microcystin-LA and -RR (Dyble et al., 2008). Molecular studies have identified toxic strains of *M. aeruginosa* across the lower Great Lakes, Lake St. Clair and Lake Michigan, demonstrating a capacity for this species to form blooms that are not localized to near-shore areas, but widely distributed across Lake Erie and connected waters (Rinta-Kanto et al., 2005; Dyble et al., 2008; Rinta-Kanto et al., 2009a; Davis et al., 2014). Dyble et al. (2008) identified distinct differences in genetic coding for MC production among *M. aeruginosa* populations in Lake Erie and Saginaw Bay (Lake Huron), which may affect the congeners produced by local blooms and their associated toxicity. These studies also show that current Lake Erie *Microcystis* is genetically indistinguishable from earlier populations and not an invasive new genotype, consistent with the persistence and opportunistic population surges of this taxon throughout historical regime shifts in response to environmental influences (Rinta-Kanto et al., 2009b). Other cyanobacteria may contribute to toxin levels, notably *Planktothrix suspensa/aghardhii*, a known source of MC toxins in Lake Erie, which can predominate under turbid conditions in the nutrient-rich Maumee River and particularly Sandusky Bay (Rinta-Kanto and Wilhelm, 2006; Kutovaya et al., 2012; Davis et al., 2015).

Although to date, no human fatalities have been directly attributed to cHABs in North America, some cyanobacterial toxins are irritants or suspected carcinogens (notably, MC) and the health effects of sub-acute chronic exposure are difficult to resolve and largely unrecorded (Boyer, 2007; Watson et al., 2008). Microcystins are chemically stable, and the most widely reported cHAB toxins in the Great Lakes. MC levels exceeding drinking and recreational water guidance values are now detected annually across Lake Erie and eutrophic inshore areas of Lakes Ontario, Huron and recently, Lake St. Clair (Dyble et al., 2008; Davis et al., 2014; Watson and Boyer, 2014). Other cyanobacterial toxins are less frequently monitored and rarely detected, despite the often-significant presence of cyanobacterial taxa that have been reported as toxic in other surface waters. The neurotoxin anatoxin-a, for example, has been reported intermittently in Lake Erie inshore areas and the embayments of Lake Ontario (Boyer, 2007; Perri et al., 2015). The toxin cylindrospermopsin has not been found in Lake Erie waters, despite the presence of species known to produce

<sup>1</sup> Planktonic forms are now reclassified as *Dolichospermum*; Wacklin et al., 2009.

this toxin in other lakes such as *Aphanizomenon flos-aquae* and the congeneric taxa *Cylindrospermopsis* (Conroy et al., 2005); strains of which also produce deoxy-cylindrospermopsin. Less is known about the occurrence of the paralytic shellfish toxins (PSTs) such as saxitoxin or its analogues produced by some freshwater cyanobacteria, although more recently, monitoring has become more intensive for all cyanotoxins (Watson and Boyer, 2014). During early surveys, PSTs were detected in only 0.2% of the mid- to late-summer samples collected from the lower Great Lakes basin in the mid-2000s and analyzed using HPLC techniques (Boyer, 2007). Since 2010, however, the US Ohio-EPA has been monitoring cyanotoxins in samples collected throughout the year from an extensive network of water treatment plant intakes and sources, including sites from Lake Erie ([http://wwwapp.epa.ohio.gov/gis/mapportal/HAB\\_Monitoring.html](http://wwwapp.epa.ohio.gov/gis/mapportal/HAB_Monitoring.html)). The data are based on ELISA analyses, and indicate that saxitoxin or its analogues may be present in as many as 40% of the samples, most at levels well below the Ohio EPA drinking water 'Do Not Use' threshold of 0.2 µg/L, including sites directly connected with or sampled from Lake Erie (average 0.022 µg/L). The cyanobacterial sources were not identified but could include either *Lyngbya* or *Dolichospermum* species, both of which are present in Lake Erie. As noted above, several species of *Dolichospermum* have been identified from many of the recent Lake Erie blooms, while *Lyngbya wollei* is often problematic in the south-west basin as dislodged or benthic mats; saxitoxin analogues (named *Lyngbya wollei* toxins, or LWTX) were identified in populations of this cyanobacteria from Lake St. Clair and the St. Lawrence River (Lajeunesse et al., 2012; Vijayavel et al., 2013).

Early detection of cHABs is fundamental to their effective management, and diagnostic and monitoring methods have been greatly enhanced by the advancement of microscopic, molecular, and imaging techniques. The increased spatial and spectral resolution of remote sensing techniques now allows earlier warning, targeted sampling and bloom forecasting, although currently, these methods are largely limited to tracking surface material and cannot represent the often-significant biomass dispersed through the water column (Binding et al., 2012; Obenour et al., 2014). Advanced molecular methods can source-track and diagnose phenotypic and physiological traits at different levels of community resolution. This approach has allowed scientists to quantify, for example, the abundance and expression of genes coding for nutrient assimilation, toxin and taste-odour production and other key metabolic processes, and identify major taxa in a mixed bloom (Kutovaya et al., 2012, 2013; Kutovaya and Watson, 2014; Steffen et al., 2015). Genetic sequencing has further elucidated both the structure and function of individual bloom community members (Steffen et al., 2012, 2015; Harke et al., 2015), and shown that toxin-producing *Microcystis* blooms are composed of a mix of toxic and non-toxic cells (Ha et al., 2008; Rinta-Kanto et al., 2009a).

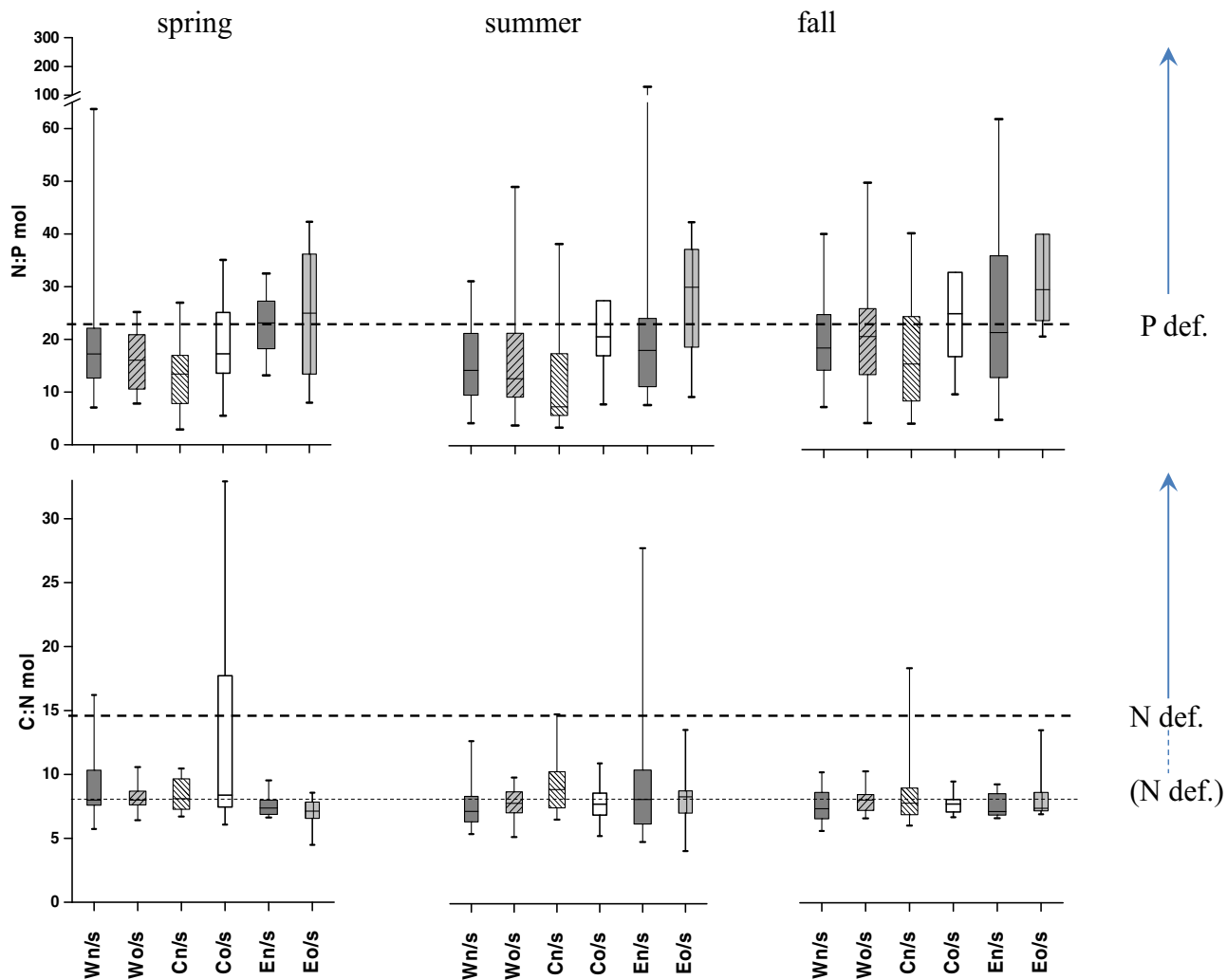
Eutrophication is widely acknowledged as a primary driver of high phytoplankton biomass. In particular, P is a key growth-limiting nutrient which reinforces cyanobacterial biomass and dominance in many freshwater systems including Lake Erie (e.g., Downing et al., 2001; Michalak et al., 2013). Nevertheless it is increasingly clear that cHAB management should consider how other factors qualify the response to P control, for example, the timing and bioavailability of this and other primary and trace nutrients, temperature, light, mixing and other physical factors, climate change, and top-down control from grazers and disease (e.g., Elser, 1999; Paerl and Otten, 2013). The recent shift in HAB characteristics has evolved with no discernible increase in total annual P loading, and instead, appears to be linked with the timing, sources and increased bioavailability of this load, coupled with other important changes (e.g. Obenour et al., 2014; Scavia et al.,

2014). Recent authors have linked the level of cHAB impairment in Erie to the volume of spring runoff in the Maumee basin, coupled with rising non-point SRP inputs from west basin tributaries (Stumpf et al., 2012; IJC, 2014; Kane et al., 2014). The rise in *Microcystis* dominance, also observed in other areas of the Great Lakes such as the Bay of Quinte (Lake Ontario), Green Bay (Lake Michigan) and Saginaw Bay (Lake Huron), has also been attributed to the enhanced water clarity, selective particle removal and soluble nutrient recycling associated with dreissenid filtering activity (Conroy et al., 2005; Munawar et al., 2008; Nicholls, 2011; Vanderploeg et al., 2013; De Stasio et al., 2014). Growing evidence now suggests that while P is a primary factor driving total phytoplankton biomass across most of Lake Erie, the supply and chemical speciation of N may play a key role in shaping the structure and abundance of this community (Dolman et al., 2012; Donald et al., 2013; Horst et al., 2014; Davis et al., 2015). The capacity to fix N<sub>2</sub> at low dissolved inorganic N supply is an important factor facilitating dominance by some cyanobacteria in P-enriched systems, and the basis for simple empirical models predicting cyanobacterial dominance at low N:P ratios (e.g., Smith, 1983). This paradigm has sparked significant debate, given that many cHAB taxa – including *Microcystis* – are not diazotrophs, while many heterotrophic Proteobacteria are, and benthic recruitment of *Microcystis* and other cyanobacteria from N-enriched sediments can facilitate cHABs even at low N:P ratios (e.g. Downing et al., 2001; Ståhl-Delbanco et al., 2003; Steffen et al., 2012; Davis et al., 2015). In Lake Erie, evidence points to the importance of N in cHAB events, for example, seston stoichiometric data for C:N:P indicates planktonic N limitation which varies among seasons, basins, nearshore and offshore zones, and years (Fig. 1). Taxon-specific bioassays also highlight the importance of both inorganic (NO<sub>3</sub>, NH<sub>4</sub>) and organic N (e.g., North et al., 2007; Chaffin and Bridgeman, 2014). Urea – increasingly used in fertilizers – can facilitate blooms of *Microcystis* and other cHAB taxa and influence toxicity (Davis et al., 2010, 2015; Finlay et al., 2010). Research also points to iron availability as a potential factor in cHABs events (Mioni et al., 2003; North et al., 2007; Molot et al., 2014) but little work has been done to evaluate this in Lake Erie.

Climate plays a large role in the inter-annual variance in the severity and spatial-temporal dynamics of these blooms (e.g., Michalak et al., 2013), affecting the timing and concentrations of nutrient loads, ice cover, storm events, water temperature, hydrodynamics and mixing patterns. Warmer temperatures have resulted in an earlier thaw or incomplete ice coverage on Lake Erie in recent years – although the record ice-cover in 2013–14 and 2014–15 highlights the significant annual variance of this cover. Higher temperatures also affect mixing, circulation and nutrient loading/internal processing, together with changes in precipitation, irradiance and storm events. This can alter aquatic community succession, and potentially facilitate extended blooms of cHAB species such as *Microcystis* (Davis et al., 2012; Koslow et al., 2013). Prolonged periods of stable thermal stratification and calm conditions can result in the rapid development of large surface blooms of large buoyancy-regulating cyanobacteria such as *Microcystis*, *Dolichospermum* and *Aphanizomenon* (Carey et al., 2012) which can become wind-blown aggregates representing a serious threat to inshore areas – as seen in the recent drinking water crisis in Toledo (Carmichael and Boyer, in press).

While non-HAB cyanobacteria such as picocyanobacteria (<2 µm) are essential food web components in more oligotrophic areas (Carrick and Schelske, 1997; Wilhelm et al., 2006), cHAB taxa are generally favoured by high nutrient levels. Current approaches reflect the historic focus on nutrients, but the science of cHABs in Lake Erie is evolving towards a more comprehensive study of bloom development and toxicity, and the importance of other factors such as climate change and other taxa. Blooms of species





**Fig. 1.** Box and whisker plots showing sestonic molar ratios of N:P (top), and C:N (bottom); plots arranged left to right in order of: West Basin Inshore, West Basin Offshore, Central Basin Inshore, Central Basin Offshore, East Basin Inshore, East Basin Offshore. Data from open water (May–Oct) surface samples (1 m), 2000–2010 (S. Watson unpublished, Environment and Climate Change Canada). Boxes represent 95th percentiles. Horizontal dashed lines are indicative of moderate (---) or severe (- - -) P or N deficiency (Ref. Healey and Hendzel, 1980). 'Offshore' defined as approximately 20 km from closest shoreline, not by depth contour.

from other major phytoplankton groups such as the winter diatom *Aulacoseira islandica* can not only impact water quality and food web structure but also may set the stage for successive plankton communities and cHAB development by altering nutrient cycling and deposition/recycling processes (Saxton et al., 2012; Twiss et al., 2012). For example, the accumulated biomass and heightened bacterial activity in early spring may fuel spring blooms and subsequent successional development, while the accumulated live and decaying diatom cells, bacteria and organic detritus at the sediment surface may periodically seed surface populations and exacerbate bottom nutrient recycling and hypoxia (Bižić-Ionescu et al., 2014; Wilhelm et al., 2014). Bacteria and viruses are also important components of HAB communities. Heterotrophic microbiota are involved in a range of physicochemical processes such as ice nucleation, nutrient cycling and oxygen drawdown, and influence both the development and decline of cHABs (Tang et al., 2010; Wilhelm et al., 2011; Louati et al., 2015). Metagenomic, metatranscriptomic and physiological studies are now revealing the significant roles that bacteria and phages play in cell mortality, along with the functional processes of N-fixation and metabolism, toxin production and breakdown, and colony formation (Steffen et al., 2012, 2015; D'Souza et al., 2013; Davis et al., 2015; Harke et al., 2015; Wang et al., 2015). Future cHAB research should focus on identifying

and evaluating the relative importance of all key biotic and abiotic driving forces behind bloom events. This approach will improve our modelling and predictive capacity for different nutrient and climate scenarios and provide a more comprehensive approach to cHAB management practices.

Harmful cyanobacteria blooms have significant socioeconomic effects (e.g. Steffensen, 2008; Smith et al., 2015), and have been reported from all states and provinces in the Great Lakes basin, with several regions recording animal illness and death, primarily of dogs and livestock. A recent review by Carmichael and Boyer (in press) evaluated the risk to human health across the Great Lakes from Federal, state and provincial case reports of cyanobacterial illness and death among animals and humans for all areas adjoining the Great Lakes. These data showed that only Lake Erie had probable or suspect cases of human and animal poisoning and that these were few in number. Ohio reported two probable and seven suspect cases of human illness associated with exposure to Lake Erie water in 2010, but none in the severe bloom year of 2011. No other provinces or states demonstrate cHAB-related health issues. All other Great Lakes experience periodic cHABs but no reports of human or animal illness, although there have been multiple beach postings and advisories, for example, in Hamilton Harbour (Watson et al., 2012).

In 2014, the first reported major impairment of Lake Erie drinking water supplies by cyanobacterial toxins affected a large population base. The source of the toxins was a *Microcystis* bloom in the west basin of Lake Erie, which although not exceptionally toxic or large in size, was driven towards the drinking water intake by strong winds from the north. As noted above, toxin-producing CHABs have occurred regularly in Lake Erie over the past 20 years dominated by *Microcystis* (Brittain et al., 2000; Watson et al., 2008; Steffen et al., 2014). Intermittently elevated toxin levels have also been reported from the west basin for more than a decade, for example, in 2000, toxin concentrations in *Microcystis* blooms exceeded 25 µg MC-LR/L (Brittain et al., 2000) and have since ranged from 1 to 200 µg MC-LR/L in surface water samples (Watson et al., 2008). While relatively minor compared to some CHAB-related drinking water events across the world (e.g., Lake Taihu; Qin et al., 2015), the 'Toledo incident' served to illustrate the challenges of predicting and managing such an event (Carmichael and Boyer, in press). In prompt response to the incident, there has been new US legislation which has included the passage of Ohio Senate Bill 1, regulating fertilizer application and the disposal of dredged lake sediments, and H.R. 212, *The Drinking Water Protection Act*, a United States federal bill which charges the US EPA to develop a strategic plan to assess and manage the risk associated with algal toxins.

### 2.1.2. Benthic HABs

The variety of physical habitat features in Lake Erie contributes to the rich diversity of benthic algae documented in Lake Erie since the late 1800s (>200 distinct taxa; Stewart and Lowe, 2008). Despite this apparent richness, detailed ecological study has been largely confined to two benthic HAB species responsible for the development of blooms that can foul beaches, shorelines and water intakes in Lake Erie: the chlorophyte *Cladophora glomerata* and cyanobacterium *Lyngbya wollei*. The spatial distribution of these blooms in Lake Erie is largely controlled by gradients in habitat features that exert control on growth. In the east basin, expansive areas of hard substratum, moderate to high water turbulence, high water clarity and adequate P supply, provide optimal conditions for *Cladophora* (Davies and Hecky, 2005; Higgins et al., 2008). In contrast, despite higher P levels in the west basin, the higher turbidity and relatively limited availability of hard substrate restrict the potential growth habitat of *Cladophora* to the splash zone. These conditions appear to provide favourable conditions for *L. wollei*, which achieves optimal photosynthetic rates at relatively low light (<50 µmol m<sup>-2</sup> s<sup>-1</sup>; Pinowska et al., 2007; Bridgeman et al., 2012) and is often found associated with unconsolidated substrates such as mixtures of dreissenid shell fragments and sand (Bridgeman and Penamon, 2010).

The negative economic, aesthetic and recreational use impacts from benthic HABs are well documented, and include the fouling of beaches and residential shorelines, clogging of industrial water intakes, and unpleasant odours associated with decaying organic matter (Carmichael et al., 1997; Higgins et al., 2008). Ecological impacts are not well characterized owing to their transient nature, but nonetheless exist. For example, expansive standing crops represent a significant nutrient sink over much of the growing season (Higgins et al., 2005; Bridgeman et al., 2012), yet neither *Cladophora* nor *Lyngbya* are considered ideal food resources for grazers (Dodds and Gudder, 1992; Hudon et al., 2012). Large accumulations of attached or drifting algal biomass can result in transient hypoxic conditions in shallow littoral regions (Gubelit and Berezina, 2010) which have deleterious impacts on invertebrate communities (Berezina and Golubkov, 2008). Potential risks to fish and other aquatic wildlife also exist, as some *Lyngbya wollei* populations are

reported to produce paralytic toxins (Carmichael et al., 1997; Lajeunesse et al., 2012; Vijayavel et al., 2013) and dense stands of *Cladophora* in the lake and deposited on beaches often harbour *Clostridium* bacteria that produce type E botulism toxins (Chun et al., 2015).

**2.1.2.1. Eastern Erie: *Cladophora*.** The severe blooms of *Cladophora* that were a persistent feature along much of the northern shore and isolated locations along the southern shore of the east basin through the 1950s to late 1970s were largely a symptom of eutrophication (Taft and Kischler, 1973; Mantai et al., 1982; Millner et al., 1982). Control of point-source P discharges to Lake Erie is generally credited with reducing the extent and severity of *Cladophora* blooms (Higgins et al., 2008), however, since the mid-1990s, expansive blooms of *Cladophora* have returned along much of the northern shores of the east basin (Higgins et al., 2005). Attached biomass observed in these contemporary blooms is generally lower than recorded historically, but growth extends deeper into the lake (Higgins et al., 2008). This change in growth pattern is thought to be largely driven by increased water clarity and an increase in suitable substrate following the advent of significant dreissenid mussel populations (both *Dreissena polymorpha* and *Dreissena bugensis*) in the east basin (Higgins et al., 2005). Mussels may also alleviate nutrient shortages through the excretion of metabolic wastes and/or increasing the amount of particulate matter available for re-mineralization through egestion of non-edible algae and other detritus in the benthic environment (Hecky et al., 2004). While it has been challenging to demonstrate this concept on a broad scale, recent in situ studies have demonstrated increases in SRP concentrations above mussel beds (Martin, 2010; Dayton et al., 2014), and P excretion rates are estimated to meet *Cladophora* growth requirements over the growing season (Ozersky et al., 2009). This is supported by observations of increased primary production rates by these macroalgae in the presence of mussels (Davies and Hecky, 2005) and a positive spatial association with dreissenid mussel abundance both within (Wilson et al., 2006) and across the lower Great Lakes (Depew et al., 2011).

**2.1.2.2. Western Erie: *Lyngbya*.** Blooms of *Lyngbya wollei* in the west basin are a more recently recorded phenomenon, with severe blooms first noted in 2006 (Bridgeman and Penamon, 2010; Bridgeman et al., 2012). While *Cladophora* in the Great Lakes is generally considered to represent one cosmopolitan species (Ross, 2006), it is unclear if recent blooms of *L. wollei* in the west basin are the result of a native strain or one recently introduced from elsewhere in North America (Bridgeman and Penamon, 2010). It is important to note, however, that *L. wollei* can often be mistaken for *Plectonema wollei* when based on morphological assessment (Stewart and Lowe, 2008), and *P. wollei* has been documented in Lake Erie since the late 1890s (Pieters, 1902; Taft, 1942; cited in Stewart and Lowe, 2008). Detailed study of *L. wollei* in western Lake Erie is limited, but it is most frequently found growing on substrates composed of dreissenid shell fragments and sand in close proximity to P-rich waters with high turbidity (e.g., Maumee Bay; Bridgeman and Penamon, 2010). This is consistent with conditions associated with other reported occurrences of this benthic cyanobacterium in the Great Lakes – St. Lawrence River system (Vis et al., 2008). In general, *Lyngbya* blooms are most common in nutrient enriched waters, including those with limited water exchange or flushing (Paerl et al., 2006). The recent increases in bioavailable P inputs to the western basin (Scavia et al., 2014) remains the most parsimonious explanation for their development in the western basin of Lake Erie at present (Bridgeman and Penamon, 2010; Bridgeman et al., 2012).

**2.1.2.3. Climate change and benthic HABs.** Forecasting the trajectory for benthic HABs under climate change in the Great Lakes region is challenging. The extent and severity of blooms under a warmer climate – and ultimately their ecological, economic and societal impacts – will also depend on both the species in question and the associated hydrologic changes that are likely to affect the physical–chemical environment (Bennington et al., 2010; Hayhoe et al., 2010) and biological processes such as metabolism and growth rate. While it is anticipated that warming water temperatures will facilitate an earlier start to growth and thus bloom formation, this is more likely to favour prokaryotes such as *Lyngbya*, because their growth rates are optimized at relatively higher temperatures (Speziale and Dyck, 1992) compared to eukaryotes such as *Cladophora* (Graham et al., 1982).

Insight into the anticipated response of benthic HABs to increased warming can be gleaned from existing studies of growth dynamics (Malkin et al., 2008), studies of algal communities impacted by artificially warmed habitats such as thermal effluent plumes (Kirby and Dunford, 1982), and application of growth models to warming scenarios (Malkin et al., 2008; Higgins et al., 2012). For example, in Lake Ontario, Malkin et al. (2008) observed *Cladophora* growth over two years with contrasting thermal regimes (average difference of 4.2 °C in water temperature between 2004 and 2005). Although attached biomass did not differ substantially between years (~120 g m<sup>-2</sup> dry wt in 2004, ~100 g m<sup>-2</sup> dry wt in 2005), peak biomass was attained much earlier (Julian Day 170) in the warmer year (2005) than in the cooler year (2004; Julian Day 204). Similar phenology has been observed when comparing *Cladophora* growth in the vicinity of thermal effluent plumes from power generating stations with growth observed remote from the thermal influence (Kirby and Dunford, 1982). Malkin et al. (2008) also applied the *Cladophora* Growth Model (CGM; Higgins et al., 2005) to simulate growth dynamics with up to an additional 2.5 °C of warming, but found only minor changes in predicted attached biomass (~+8%) with a warming of 1.0 °C, and no increases beyond 2.0 °C, as respiration rates increased to a greater degree than production rates. The applicability of these results to Lake Erie is uncertain given that this lake is on average, warmer than Lake Ontario. Nonetheless, both direct observation and modelling suggest that *Cladophora* is fundamentally constrained by P and light availability, so warming temperatures alone are not anticipated to have major impacts on biomass accrual unless accompanied by alterations in the timing of P availability and the benthic light climate.

**2.1.2.4. Summary.** The issue of benthic HABs in Lake Erie merits sustained integrated research and monitoring, because the symptoms of coastal impairment cannot be easily ignored. While it is generally accepted that the overabundance of algal species that constitute benthic HABs is primarily controlled by bottom-up processes (i.e. nutrient and light availability), the importance of top-down processes (i.e. grazers or physical detachment/sloughing) are not well characterized, particularly in the Laurentian Great Lakes. Given that the production and accrual of biomass can be quite variable over both temporal and spatial scales, definitive conclusions regarding the proximate factors responsible for the development and maintenance of contemporary benthic HABs in Lake Erie have been elusive, to a large extent because of infrequent and irregular measurements of standing crop (e.g., Higgins et al., 2005; Bridgeman et al., 2012) and an incomplete understanding of the impacts of recent ecological disturbance (e.g. impact of dreissenid mussels; *sensu* Hecky et al., 2004) and the changing nature of nutrient loads to the lake (Dolan and Chapra, 2012). Alternative monitoring and assessment approaches such as remote imaging (Shuchman et al., 2013) and acoustic survey methods (Depew et al., 2009, 2011) have recently demonstrated success at incorporating

natural variability into synoptic assessments, and offer the ability to extend assessment of benthic HABs to broader areas.

## 2.2. Hypoxia

The reduction of hypoxic conditions (i.e. <2 mg/L dissolved oxygen, DO) in the central basin of Lake Erie has long been a goal of international remedial efforts, which was first identified in the 1972 GLWQA, and re-stated in the recently renewed (2012) agreement. Concern with this issue was heightened during the late 1960s and early 1970s, with the publication of maps showing an anoxic area (i.e. <0.2 mg/L) in the bottom layer (hypolimnion) of the lake linked by the public media to the phrase “Lake Erie is Dead”, which became a common misconception (Sweeney, 1993; Ludsin et al., 2001). At the time, the scientific belief was that P loading had a significant and direct effect on the severity of hypoxia, and the 1972 GLWQA sought to bring about a large improvement in DO depletion by reducing P loads by approximately 50%. The initial post-remedial declines in hypoxia and anoxia during the early 1990s – which were associated with improved biological communities (Ludsin et al., 2001) – were thus attributed to these P abatement programs, but the response was less than initially desired (Charlton, 1980a; Bertram, 1993; Burns et al., 2005). To understand why hypoxia never fully disappeared and has recently re-emerged as an issue in Lake Erie (Hawley et al., 2006; Scavia et al., 2014), this section re-examines the causes of hypoxia, how it may be altered by climate change, and its documented and potential effects on food webs.

### 2.2.1. Causes of hypoxia

All temperate lakes that stratify during summer will have some degree of DO depletion in the hypolimnion, and in some cases, this can lead to hypoxic conditions that can negatively affect biota (Arend et al., 2011; Scavia et al., 2014). The bathymetry of Lake Erie renders it particularly prone to seasonal hypoxia. This is true of the central basin, which is deep enough to stratify during the summer (mean depth ( $Z_m$ ) = 18.3 m), and typically develops a “thin” hypolimnion with a small volume relative to the epilimnion owing to the establishment of the thermocline close to the lake bottom (generally <6 m above the sediment surface; Rosa and Burns, 1987). Even under relatively unproductive conditions (e.g., pre-settlement), the small volume of hypolimnetic water cannot accommodate the DO demand generated by a combination of the diffusion of reduced substances (e.g., methane) from the sediment and the degradation of organic detritus from the surface layer. The resulting DO depletion leads to hypolimnetic hypoxia, often established early in the summer and extending through to the fall turnover. Thus seasonal hypoxia has most likely occurred in the central basin of Lake Erie for thousands of years before the settlement of the basin (Delorme, 1982). By contrast, the shallow west basin ( $Z_m$  = 7.3 m) usually does not develop sustained hypoxia, owing to strong wind-induced mixing that keeps the basin from stratifying for long periods of time during summer, while the hypolimnetic volume of the deeper east basin ( $Z_m$  = 24.4 m) is large enough that there is insufficient DO depletion before fall turnover to cause hypoxia.

Owing to the bathymetry of central Lake Erie, the severity of hypoxia can vary both within and among years in response to meteorological conditions (e.g., wind, temperature). Local weather can drive physical processes such as mixing, within and between-basin circulation, and water-column stratification (strength, depth), all of which can influence the rate of DO depletion and concentrations by regulating hypolimnetic thickness, temperature, sediment-associated DO demand, and its transfer across the thermocline (Charlton, 1979, 1980a, 1980b; Cornett and Rigler, 1979; Burns et al., 2005; Mueller et al., 2012; Zhou et al., 2013; Rucinski et al., 2014). In this way, hypoxia can be problematic in

one year (e.g., a warm year with deep thermocline and thin hypolimnion) but not the next (e.g., a colder year with a shallower thermocline and thicker hypolimnion), regardless of allochthonous nutrient loading. To illustrate, no hypoxia was detected in central Lake Erie during 1975, a year of excessive P loading (Scavia et al., 2014), owing to the formation of an unusually thick, cold hypolimnion that was driven by meteorological conditions (Charlton, 1979). These phenomena can account in part for the presence of hypoxia in Lake Erie during pre-settlement times, and its high variance during periods of both eutrophication (1970s; Charlton, 1979) and oligotrophication (early 1990s; Bertram, 1993). Early oxygen data which pre-date nutrient controls suggested a large and direct effect of increasing P loads on hypoxia (Dobson and Gilbertson, 1971) and led to expectations of a large response to decreased P loads. Those data, however, represented intermittent and biased sampling, and the effect of increased P load was much less than previously believed, once hypolimnion thickness and temperature were accounted for (Charlton, 1979, 1980a). This is consistent with the persistence of hypoxia subsequent to P load reduction.

Despite the importance of weather-induced physical processes, humans have influenced the severity of hypoxia in central Lake Erie indirectly through excessive nutrient loading (i.e., cultural eutrophication; Rosa and Burns, 1987; Bertram, 1993; Burns et al., 2005; Rucinski et al., 2014). Historically, P has limited primary production in Lake Erie (Vollenweider et al., 1974), and large increases in loading have led to excessive production of phytoplankton and inedible cyanobacteria. For these reasons, a correlation between annual P loading and seasonal hypoxia has been documented, although as mentioned above, the full potential of hypoxia in Lake Erie is dependent on lake physical processes (Burns et al., 2005; Rao et al., 2008; Rucinski et al., 2014). This suite of findings highlights the need to consider both physical processes and allochthonous nutrient inputs when attempting to understand and predict hypoxia in Lake Erie. Failure to account for stochastic physical processes associated with weather can help to explain why nutrient loading alone is not an effective predictor of hypoxia and why hypoxia remained a sporadic problem both before and after TP loading targets (11,000 MTA) were met as a result of point-source P abatement programs under the 1972 GLWQA.

To illustrate the importance of considering both physical processes and allochthonous P loads when predicting hypoxia, DO depletion rates from 1970 to the present were analyzed (Fig. 2). These rates, based on the EPA monitoring data, were obtained by adjusting in situ DO depletion rates for differences in hypolimnion thickness and temperature, and DO transfer across the thermocline using the methods of Burns et al. (2005). The adjusted depletion rates capture trends in the organic decomposition in water and sediment, as well as the oxidation of reduced compounds in sediment, both of which relate to primary production, and in turn, to P loads. Using this approach, as higher temperatures increase bacterial decomposition rates, for example, the adjusted depletion rates would not change for a given P load (due to the  $Q_{10}$  temperature correction), but the in situ rates would increase and DO concentrations would be reduced. The adjusted data (Fig. 2) are reminiscent of a flattened version of the TP loading trend for Lake Erie (IJC, 2014; Scavia et al., 2014), showing an initial response to reduced loads up to about 1990, followed by variability and some tendency to increase later. In 1970, the P load was about 23,000 MTA and the adjusted oxygen depletion rate was 3.8 mg/L/month. By the period from 1990 to 2000, the depletion rate average was 3.1 mg/L/month at a nominal P load of 11,000 MTA. Following the regression in Burns et al. (2005), the depletion rate would be approximately 3 mg/L/month at a P load of 5000 MTA or a 21% decrease in oxygen depletion rate for a 75% decrease in P load from 1970. These new rates, which again consider many important

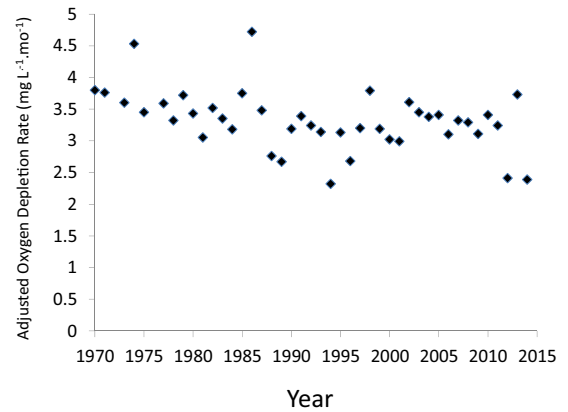


Fig. 2. Adjusted oxygen depletion rates in the central basin of Lake Erie (1972 and 1976, missing data). Data courtesy Great Lakes National Program Office, U.S. EPA.

physical processes in addition to oxygen concentrations, minimize the effects of stochastic variability and represent central basin DO depletion processes better than in situ DO depletion rates. In addition, the 1987–1996 post-remedial improvements and subsequent worsening of central basin hypoxia that were suggested by short-term measures of DO concentration (as in, for example, Scavia et al., 2014), may have been affected by other factors. For example, Burns et al. (2005) considered two of the improvement years with higher oxygen concentrations as anomalous, and perhaps due to hypolimnetic primary production. Further research is needed on how to derive and represent DO concentrations over time in a way that is not confounded by physical and other factors.

### 2.2.2. Climate change and hypoxia

Predictions made for other temperate freshwater ecosystems indicate that climate change may exacerbate the magnitude, duration, and frequency of hypoxia via a number of different mechanisms such as deeper and longer stratification, increased water temperature, and increased nutrient runoff during winter and spring (Kling et al., 2003; Ficke et al., 2007; Fang and Stefan, 2009; Jiang et al., 2012). To date, few modelling studies have attempted to predict the effects of climate change on Lake Erie hypoxia. For example, Schertzer and Sawchuk (1990) modelled the effect of doubling atmospheric CO<sub>2</sub> on stratification, predicting only slight hypolimnetic anoxia in the central basin of Lake Erie during late fall. Assuming that the warming and stratification trends in Lake Erie would be similar to those seen in Lake Ontario (Huang et al., 2012), the earlier development of a longer summer stratification period, with a later fall turnover is expected. As shown in other freshwater systems, these conditions would result in earlier bottom DO depletion, and hypoxic conditions are likely to be present over an extended time period (Fang and Stefan, 2009). Hypoxia could also be exacerbated by continued increases in multi-day storm events during winter and spring (Kunkel et al., 1999), which have been predicted to increase nutrient runoff in highly agricultural watersheds such as the Maumee River Basin during winter and spring (Cousino et al., 2015).

Water level is a key factor that is likely to affect future hypoxia formation, but much uncertainty exists in predicting the effects of climate change on Great Lakes water levels. For example, Angel and Kunkel (2009) predicted a decline of 0.61 m in Lake Erie water level, whereas MacKay and Seglenicks (2013) predicted much less drastic decreases in average water levels. If the projected water level reductions are realized (e.g., as in Angel and Kunkel, 2009) or reduced levels occur even sooner owing to natural climate variability (Gronewald and Stow, 2014), bottom hypoxia could be exacerbated further. This could happen because a reduced water



volume would allow the lake to warm more rapidly during spring and attain higher surface and hypolimnetic temperatures. In turn, the thermocline could develop closer to the bottom, resulting in intensified DO depletion.

### 2.2.3. Effects on foodwebs

Hypoxia can affect the structure and composition of the biological communities present in the hypolimnion. Most aquatic organisms require DO to meet their basal metabolic needs (Wu, 2002; Ekau et al., 2010; Pörtner, 2010), and sustained low DO levels can affect them directly by causing mortality (i.e., fish kills: e.g., Rao et al., 2014; benthic macroinvertebrate kills: Bridgeman et al., 2006) or indirectly by modifying movement and foraging behaviour, growth, and reproduction (Eby and Crowder, 2002; Wu, 2002; Ludsin et al., 2009; Domenici et al., 2013). Benthic invertebrates, particularly those with limited mobility (e.g., dreissenid mussels), would be expected to be affected negatively by hypoxia in the central basin, as boundary layer DO gradients may form close to the bottom under calm conditions. A 50-year historical dataset (Burlakova et al., 2014) supports this tenet by showing that dreissenids, which have dominated benthos in the west and east basins during the past few decades, were not detected in the offshore area of the central basin affected by hypoxia. For this reason, Burlakova et al. (2014) suggested that dreissenid density may be useful as an indicator of the severity of oxygen depletion, because of the limited tolerance of these organisms to hypoxia. Similarly, it has been suggested that hypoxia may regulate the distribution, abundance, and population dynamics of burrowing mayflies (*Hexagenia* spp.) in western Lake Erie (Bridgeman et al., 2006); however the central basin hypolimnion is too cold to allow development of a population of these organisms in the offshore areas, regardless of oxygen concentrations (Krieger et al., 2007).

Hypoxia may affect fish species or age classes differentially, and coupled with changes in predator-prey interactions, alter the food web structure and function (e.g., Breitburg et al., 1997; Brietburg, 2002; Eby and Crowder, 2002; Ludsin et al., 2009). It may also have a more generalized effect; acute, lethal effects of hypoxia have been documented, for example, in sporadic multi-specific fish kills associated with upwellings of hypolimnetic water on the north shore of Lake Erie (Rao et al., 2014). Nevertheless these acute effects are likely to be rare, given that fish can generally detect and avoid hypoxia (Wannamaker and Rice, 2000). Instead, non-lethal (indirect) effects would be expected to predominate and be manifested as changes in the behaviour of fish and their benthic and planktonic prey.

Towards the end of summer, bottom DO concentrations in the central basin of Lake Erie can exclude most fish from the hypolimnion and at the same time decrease the availability of zooplankton prey (Vanderploeg et al., 2009a,b; Pothoven et al., 2009). In turn, individuals may be exposed to increased predation risk (Vanderploeg et al., 2009a; Brandt et al., 2011) or excluded from optimal areas for foraging and growth (Roberts et al., 2009, 2012; Arend et al., 2011), thus forcing them to make energetically expensive foraging forays into the bottom sediments for food (Roberts et al., 2009, 2012). Some species are expected to be affected negatively by hypoxia (e.g., rainbow smelt, *Osmerus mordax*; round goby, *Neogobius melanostomus*; lake whitefish, *Coregonus clupeaformis*; Ludsin et al., 2001; Arend et al., 2011; Pothoven et al., 2009), others only affected in a minor fashion or not at all (e.g., yellow perch *Perca flavescens*; emerald shiner, *Notropis atherinoides*; Arend et al., 2011; Pothoven et al., 2009). Still others may even benefit indirectly from hypoxia (e.g., walleye, *Sander vitreum*; Brandt et al., 2011; Pandit et al., 2013). At present, however, the population-level effects of hypoxia on Lake Erie fishes remain largely hypothetical (Ludsin et al., 2001; Scavia et al., 2014)

which is consistent with the current level of understanding for nearly all other ecosystems in this respect (Breitburg et al., 2009). Given this uncertainty and the importance of sub-lethal effects of hypoxia, it is clear that more research is required to evaluate the short- and long-term effects on fish populations.

Lake Erie hypoxia encompasses considerable uncertainty in terms of both biological and nutrient control outcomes (Rucinski et al., 2014; Scavia et al., 2014). Nevertheless, if some of the predicted climate change scenarios are realized (e.g., warming, increased winter and spring precipitation), preventative nutrient controls will likely be needed simply to maintain the status quo in the lake, despite some uncertainty regarding the short term effects of these measures on hypoxia. Ongoing monitoring from different agencies provides data to meet the needs of the models with respect to hypoxia in the central basin; however, more research is required to quantify the sediment oxygen demand as nutrient load reductions are applied.

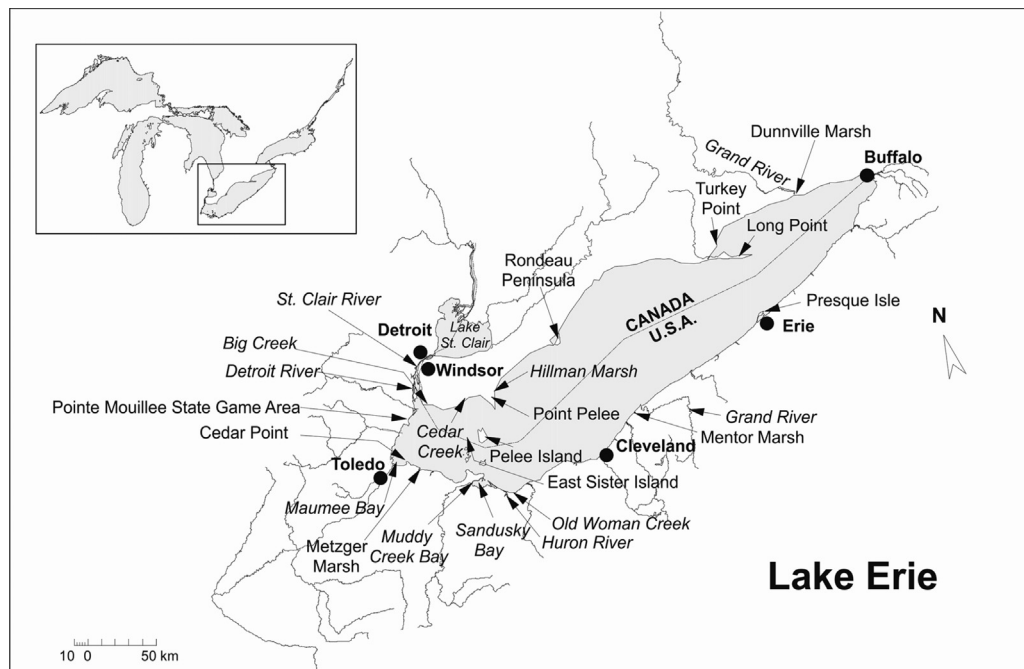
## 3. Nutrient loading to Lake Erie

### 3.1. External loading: nutrient management and models

#### 3.1.1. Advancements in our understanding of the role of phosphorus loading

As a result of the lake-wide remedial actions taken to meet the 1972 GLWQA objectives, TP inputs declined precipitously from 1972 to meet the annual target loading level of 11,000 metric tonnes per annum (MTA) by the early 1980s, and remained below this target in most years up to 2011 (Scavia et al., 2014). These reductions largely stemmed from the progress made with the control of point sources (e.g., upgrading of the wastewater treatment plants (WWTPs), restrictions in commercial detergents), whereas more recently, the non-point source contribution has become the primary external factor modulating the ecosystem dynamics of Lake Erie (e.g., Dolan and McGunagle, 2005; Stow et al., 2015). For example, in several monitored tributaries draining into the western and central basins, the most bioavailable P fraction (SRP) has more than doubled since the 1990s, rising from ~11% to 24% of the TP load, e.g., Maumee River, Sandusky River, Honey Creek, and Rock Creek (Baker et al., 2014; Stow et al., 2015; locations of major tributaries are shown in Fig. 3). Scavia et al. (2014) also estimated that the west, central, and east basins of the lake received approximately 60%, 30%, and 10% of the (2003–2011) lake-wide TP loads and ~68%, 24%, and 8% of the whole-lake SRP loads, respectively. Although these estimates do not consider the inter-basin nutrient transport, they do provide evidence that nutrient loads from several tributaries in the western half of the lake (particularly from Maumee, Detroit, Sandusky and Cuyahoga Rivers) are the predominant drivers of the severity of eutrophication symptoms in Lake Erie.

An updated estimate of the P inputs and outputs for U.S. and Canadian watersheds showed that human sources dominate in the Lake St. Clair, Clinton, Detroit, Huron, Cuyahoga, Ashtabula watersheds; agricultural fertilizers in Ottawa-Stony, Raisin, Maumee, Cedar-Portage, Sandusky, Huron-Vermilion, Cedar Creek input; and manure in the Grand (Ontario) and Thames loads (Scavia et al., 2014). In the same context, Daloğlu et al. (2012) and Bosch et al. (2013, 2014) used the Soil and Water Assessment Tool (SWAT) to evaluate P sources within several watersheds, and showed that approximately 40% of the TP load from the Maumee, Sandusky, and Cuyahoga watersheds originates from 25% of the basin, suggesting the need for designing BMPs to alleviate the impact of non-point source loading (see Section 3.2). It has been shown that the intensity of spring storms in the Lake Erie watershed has increased, and precipitation patterns are predicted to show more intense late-winter and early spring events in the



**Fig. 3.** Map of Lake Erie showing major tributaries and wetland complexes (the latter based on [Maynard and Wilcox, 1997](#)).

future ([Hayhoe et al., 2010](#); [Bartolai et al., 2015](#)). [Daloğlu et al. \(2012\)](#) demonstrated with a SWAT model of the Sandusky watershed that the more recent increase in intense spring storms, coupled with changes in tillage and fertilizing practices, is a likely driver of the increase in SRP loads from agricultural watersheds. The frequency of extreme rain events has increased since the early 1900s in this region, as has the duration of wet periods. [Bosch et al. \(2014\)](#) further showed that climate change will make load reductions even more difficult, highlighting the importance of adaptive management.

### 3.1.2. Recent modelling work in Lake Erie

A variety of empirical and process-based models has been developed to examine ecological mechanisms and predict the response to nutrient management actions for Lake Erie ([Kim et al., 2014](#)). The simplistic nature of empirical models typically does not allow the prediction of conditions induced by episodic events or internal loading ([Cheng et al., 2010](#)). As an alternative, process-based (or mechanistic) models mimic the role of individual ecological processes through mathematical description and realistic parameter specification to depict ecosystem dynamics. Based on a general set of equations, these models aim to describe key physical, chemical, and biological processes with site-specific parameters, initial conditions, and forcing functions which are then used to reproduce observed system dynamics and project future system response to different nutrient or climate scenarios ([Arhonditsis and Brett, 2004](#)). In Lake Erie, both types of models have been used to examine different ecological responses in the lakes. Some of the recent process-based models, however, are over-parameterized and therefore do not have the capacity to support robust predictive capability ([Kim et al., 2014](#)).

The selection of mathematical equations and parameters must be ecologically defensible and linked tightly to the contemporary understanding of the system. Nonetheless, differences in both mathematical assumptions and parameter specification of existing Lake Erie modelling constructs can conceivably yield significantly different projections under alternative management scenarios ([Zhang et al., 2008](#); [Leon et al., 2011](#); [Rucinski et al., 2014](#)). For example, models differ on their description of phytoplankton

functional groups and values for maximum growth rates, which are based on literature not necessarily derived from Lake Erie. The maximum growth rate is an influential parameter that aims to characterize the average patterns of diverse assemblages of phytoplankton species. The control of this maximum potential growth by resource (temperature, light, and nutrient) limitations determines the growth rates predicted by the phytoplankton simulations. In this regard, algal growth rates have been modelled as a single function of ambient nutrient concentrations, or as a two-step process that considers nutrient uptake rate in relation to the ambient supply, and subsequently, growth rate as a function of cell nutrient storage. Likewise, several of the existing modelling efforts have not explicitly simulated top-down processes ([Zhang et al., 2008](#); [Leon et al., 2011](#)). Both intracellular storage and top-down control are key factors to predicting phytoplankton responses to ambient nutrient variability, and may have profound implications in the predictive statements when examining lake dynamics under alternative management scenarios. Given that several of the existing mechanistic models have relevant modules ([Zhang et al., 2008](#); [Leon et al., 2011](#)), the impact of top-down processes will be addressed in the near future. The updated models have to accurately reflect the current understanding of the interplay between bottom-up and top-down forces in shaping phytoplankton dynamics in the system.

Many recent modelling studies have highlighted the need to include multiple nutrients along with the finer representation of phytoplankton communities to elucidate key issues such as cyanobacteria dominance. Nevertheless because of the unresolved ecology of plankton assemblages, the development of robust group-specific parameterizations to support such scenario-based predictions is highly uncertain ([Watson et al., 2008](#)) and the capacity of current models to simulate the dynamics of individual species or genera is far from established ([Anderson, 2005](#)). For example, the reproduction of seasonal phytoplankton succession is very sensitive and only occurs within a fairly narrow window of the model parameter space ([Zhao et al., 2008](#)). The effect of dreissenids is another critical aspect that requires robust modelling, yet existing sub-models lack the ability to predict changes in spatiotemporal mussel densities and age group distributions

(Bierman et al., 2005). As an alternative, statistical models provide straightforward relationships coupled with uncertainty estimates (e.g., response curves), founded upon data from the system. Despite their simplicity, these models offer a screening level insight into predicted Lake Erie response to external and internal changes, but they have major limitations outside the range of the dataset used.

Sediment diagenesis is a critical facet of nutrient biogeochemistry that has received little attention from both empirical (e.g., pore water analysis, P fractionation, organic matter profiles) and modelling points of view (McCulloch et al., 2013). Sediments are an important factor influencing lake water quality and recovery time, acting as a source or sink for numerous chemicals, including nutrients. Proximal to the sediment-water interface, intensive microbiological, geochemical and physical processes determine the release of organic matter, nutrients, and pollutants into the overlying water. Detailed knowledge of these processes is thus essential for the assessment of water quality, understanding the impacts of hypoxia, and managing surface water quality. Where measurements are impossible or expensive, diagenetic modelling is a valuable tool to investigate the interplay among the sediment processes, verify concepts, and potentially predict system behaviours (Boudreau, 1997). This approach can help to address questions such as whether sediment P retention can be predicted from sediment mineralogy, substance inputs, and catchment type and how this may be influenced by human activities and/or climate change.

In Lake Erie, models with different strengths and weaknesses offer a unique opportunity for synthesis and improvement of contemporary modelling. Collective Lake Erie mechanistic and empirical models can be integrated to guide the decision-making process, provide a framework for improvement of contemporary modelling practices, compare alternative ecological structures, challenge existing ecosystem conceptualizations, and synthesize different (and often conflicting) paradigms. One of the overlooked aspects of the existing local modelling work involves the rigorous assessment of model structural and parametric uncertainty (Arhonditsis et al., 2007). Recognizing the importance of the uncertainty problem, the recent model calibration practices tend to change from seeking a single “optimal” value for each model parameter, to seeking a distribution of parameter sets that all meet a pre-defined fitting criterion (Stow et al., 2007). These acceptable parameter sets may then provide the basis for estimating model prediction error associated with the model parameters.

### 3.2. External loading: effectiveness of urban and agricultural BMPs in reducing phosphorus loads

Although BMPs are most often employed to reduce peak flow and suspended solids, they are increasingly designed for pollutant reductions. This section focuses on BMPs that have been scientifically evaluated for their efficacy in P reduction, and highlights practices that have been implemented within the U.S. portion of the Great Lakes watershed.

#### 3.2.1. Urban BMPs

Urban P loads can represent a disproportionately large fraction of a drainage basin's TP loading. For example, the urbanized fraction of the Lake Champlain watershed constitutes only 3% of the area but is estimated to contribute 18% of the P load (Meals and Budd, 1998). Due to the multitude of land uses within urban watersheds, it is often difficult to pinpoint P loads from specific urban land cover, and holistic management/control of urban P loads with distributed BMPs at a majority of urban sub-watershed discharge points may be required (Winter and Duthie, 2000). The selection of BMPs for P removal in urban settings is often based on

general classification of perceived BMP utility (Gibb et al., 1999) and is used in the following discussion of urban BMPs.

**3.2.1.1. Non-structural (alternative behaviour/management) BMPs.** Non-structural BMPs include educational outreach and municipal ordinances focused on behaviour modification to reduce P loadings from activities such as the use of lawn fertilizers, better management of litterfall and pet wastes in urban environments. Educational outreach alone has demonstrated only minor improvements in reducing P loadings related to fertilizer use (Dietz et al., 2004). On the other hand, adoption of municipal ordinances that limit P-containing lawn fertilizer application resulted in significant reductions in TP and a trend of dissolved P reductions in Ann Arbor, MI (Lehman et al., 2009). Considerable reductions in P loads may also be realized through the use of soil chemistry testing to optimize fertilizer addition (Erickson et al., 2005). An additional practice that has been successful is the use of composted manure as a fertilizer, which provides a source of slow release P, and generally reduces TP loadings to urban streams compared to conventional commercial turf-grass sod maintained with inorganic P fertilizer (Richards et al., 2008). Nevertheless, results from these approaches have been mixed and are dependent on the scope of adoption.

**3.2.1.2. Non-point source structural or engineered BMPs.** Engineered BMPs include a full spectrum of approaches (Fig. 4; Table 1) such as porous pavements, media filters, bioswales, rain gardens, green roofs, detention biofiltration, and bioretention basins, constructed wetlands and various commercial devices. Structural or engineered BMPs typically employ filtration and/or detention (allowing settling of sorbed material), and many BMPs combine several different filtration and removal devices. The following section outlines the utility and limitations of these technologies as they pertain to urban P loadings (Table 2).

**3.2.1.3. Structural BMP treatment systems.** These BMPs cover a wide array of types and varying levels of performance (Table 1). Of the references reviewed, most focus on TP with only a few including dissolved P (DP) metrics. It is worth noting that none of the BMPs reviewed were found to have consistently high removal efficiency. To provide an illustrative example, over 6000 records from the International Stormwater BMP Database ([www.bmpdatabase.org](http://www.bmpdatabase.org)) were queried (downloaded Jan. 4, 2013) to evaluate the treatment efficiency of structural BMPs, using Event Mean Concentrations (EMCs) to quantify pollutant load. This exercise included over 216 different BMP installations. To measure the BMP performance, the fraction discharged after treatment ( $\beta$ ) was quantified by comparing the mass load of P being discharged from a BMP structure ( $M_{out}$ ) relative to the mass load entering the structure ( $M_{in}$ ) per individual storm event:

$$\beta = \frac{M_{out}}{M_{in}} \quad (1)$$

According to this equation, if the mass of P leaving the BMP structure is less than that entering, the fraction  $\beta$  is  $<1$  (i.e. the BMP is effective). When the mass of P leaving the structure is greater than the mass entering, the fraction  $\beta$  is  $>1$  and the BMP contributes to P loading. The logarithm of the fraction ( $\beta$ ) discharged following treatment is negative in cases of P removal, while a positive  $\log \beta$  indicates the BMP contributes to P loading.

TP was removed by detention basins in slightly more than half of the cases analyzed (Fig. 5), likely attributed to the removal of PP (while few cases, this curve plots to the left in the negative  $\log \beta$  range). In contrast, DP was removed in fewer than 40% of the cases



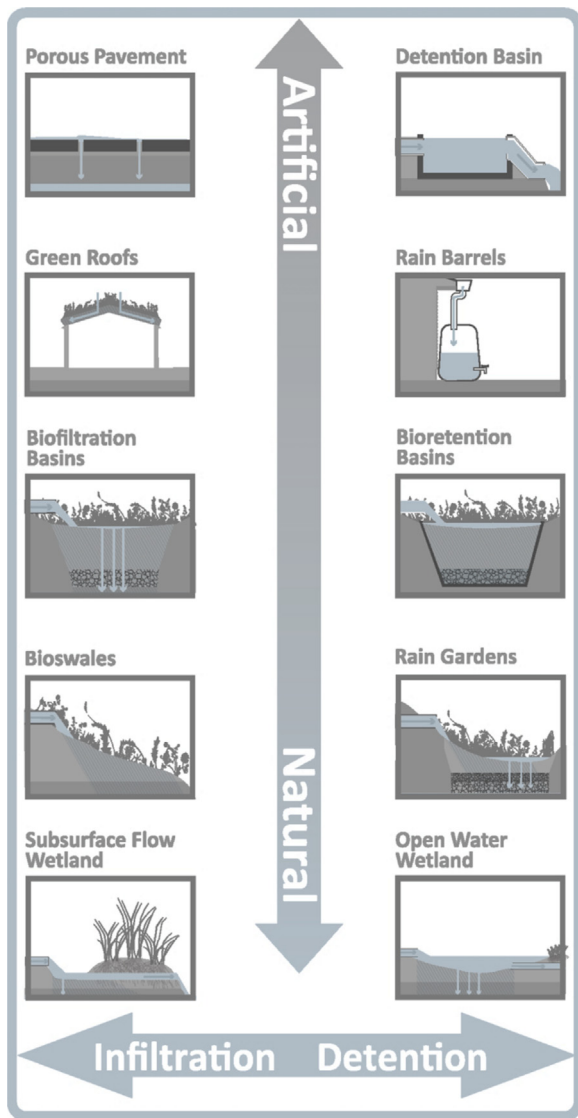


Fig. 4. Spectrum of urban structural BMPs.

analyzed. Detention basins, biofilters and wetland systems (basins and channels) resulted in notably different removal efficiencies associated with each P fraction, and the fraction of P removed tended to follow the order: PP > TP > DP.

Reductions in P discharges from large point sources are responsible for the majority of decreases in P loads to Lake Erie since the 1970s (Dolan and McGunagle, 2005). While large-scale WWTPs have been nearly 100% compliant since the 1990s, Lake Erie receives the largest municipal P load of the Great Lakes. Combined sewer overflows (CSOs) deliver some 90.4 MTA of TP to Lake Erie from Ohio alone (Ohio EPA, 2010). Nineteen CSOs discharge untreated sewage directly into the Lake, while 107 other CSOs discharge to receiving waters that empty into Lake Erie (e.g. Mill Creek, Cuyahoga River, Rocky River, Big Creek; Gomberg, 2007).

Urban BMP performance metrics can vary dramatically, depending on the metric used (Lenhart and Hunt, 2011). Evaluations based on concentrations alone may be misleading, because performance varies during and between stormwater runoff events (Lenhart and Hunt, 2011). On the other hand, percent removal is also problematic as a metric because it can vary with the influent concentration (Zhang et al., 2010) and does not account for

background water quality, eco-region differentiation, and background, or “irreducible” concentrations. Percent removal further assumes an association between influent and effluent concentrations, which may not exist (for example, due to the treatment media contributing rather than removing the pollutant) (McNett et al., 2011).

Regardless of the BMP type, three main mechanisms are responsible for P removal in stormwater: bio-uptake, sorption and precipitation. Ultimately P is retained via physical processes, by attaching to BMP materials (e.g. sorption to wetland plants) or by settling out – directly as a precipitate or indirectly, in association with biological material or suspended solids. Of these mechanisms, sorption reactions are the most common mechanism employed by BMPs. On average, ~70% of P in storm-water is removed by the elimination of particles >20 µm in diameter; and 90% of P is associated with particles (>0.45 µm) (Johnson et al., 2003). Nevertheless, because P partitioning between particulate and soluble forms varies with the solution chemistry and the amount and type of solids present, and furthermore can convert rapidly, BMPs need to address both P fractions in order to achieve consistent and high removal (Leisenring et al., 2010).

To assess the effectiveness of urban BMPs, appropriate and rigorous metrics should be used that include both dissolved and particulate fractions of P and flow-weighted EMCs for loading estimates, and account for the variability inherent to these dynamic systems. An important example is seen in the high spatiotemporal variance in water quality and nutrients measured in the Detroit River by a joint EC-USEPA sampling study (Burniston et al., 2009). Additionally, if systems are large or contain sufficient vegetation (e.g. constructed wetlands), an assessment will likely also need to consider the diurnal and seasonal variation induced by biological cycling of P.

### 3.2.2. BMPs in agricultural and rural environments

The Ohio Lake Erie Phosphorus Task Force (OH-LEPFT) provides agricultural BMP recommendations to reduce the amount of P, N and sediment exports to Lake Erie (Ohio EPA, 2010). BMP effectiveness is site-specific, and depends on local topography, climate, cropping systems maintenance, selection, and installation (Alfera and Weismiller, 2002). Several review articles have evaluated a suite of BMPs implemented simultaneously (Bishop et al., 2005) which makes it difficult to discern the effectiveness characteristics of an individual BMP within the suite. Many of the BMPs reviewed by these papers were outside the Lake Erie basin, but can be reasonably applied to the Lake Erie watershed. Assessments were done at different scales (plot, field, and watershed scale), using either field studies or simulation modelling. Most BMP assessments at watershed scale have been carried out as modelling studies or as a trend analysis of temporal changes in water quality parameters at the watershed outlet (e.g., Bosch et al., 2013). The BMP effectiveness tool created by Merriman et al. (2009; available online: Gitau, 2013) has been used in many of these BMP assessments.

Some of the agricultural BMPs cited by Sharpley et al. (2006) are now mandatory in Ohio, in particular western Lake Erie watersheds. An Ohio state law that took effect in July 2015 restricts manure and fertilizer application: (1) on frozen or snow-covered soil, (2) on moisture-saturated soil (top 2 in.), and (3) when a weather forecast calls for more than 50% chance of 1-in. precipitation in 12 h before commercial fertilizer application and 0.5-in. of precipitation in 24 h for manure. Subsurface application and incorporation of manure and fertilizer within 24 h of precipitation are exempted from this restriction.

In-field control BMPs minimize nutrient and sediment transport and include controlled drainage, conservation tillage, 2-stage ditch, grassed waterways, etc. BMPs designed to remediate and



**Table 1**  
Urban structural/constructed BMPs and associated removal effectiveness.

BMP	Removal	Comments
Porous pavements	TP: 60–71% <sup>1–3</sup>	Variable performance reported Inflow TP versus outflow TP not statistically different <sup>4</sup>
Media filters (sand filters, infiltration trenches, etc.)	TP: 43–82% <sup>3,5–9</sup>	Filter material selection critical Organic content of media does not appear to enhance P reductions <sup>10–12</sup> Statistically significant decreased observed for TP and PO <sub>4</sub> <sup>4</sup>
Filter strips/bioswales	TP (LSGFS): ~48% <sup>13,14</sup> TP (bioswales): 0–85% <sup>10,15–20</sup>	Length of LSGF is important <sup>21</sup> Based on EMC, removal still varies but appears to be around 50% <sup>22</sup> Increases in TP, PO <sub>4</sub> and DP, likely due to fertilizer application, are reported <sup>4</sup>
Green roofs	Initial increase in TP <sup>23</sup>	Limited data also suggests differences in short- versus long-term performance <sup>24</sup> More rigorous long-term monitoring required
Bioretention basins	TP: 20–90% <sup>3,7,16,18,19,25–28,29</sup> DP (wet ponds): ~51% <sup>30</sup> DP (wet ponds): ~60% <sup>13</sup>	Removal efficiency depends on design <sup>31</sup> Large difference observed between “wet” and “dry” basins <sup>13,30,32</sup> Better mechanistic understanding needed <sup>33</sup>
Wetlands	TP (constructed): 25–70%, <sup>22,34</sup> TP (channels): 51% <sup>35</sup> DP (channels): 39% <sup>35</sup>	Performance varies widely depending on surface or subsurface flow <sup>30,32</sup> Removal efficiencies in subsurface flow and open surface wetlands hampered by low oxygen conditions that induce P release <sup>36</sup> A better understanding of the geochemical processes in these systems is required
Commercial devices	TP (oil and grit separator): <10% TP <sup>3</sup> (sedimentation basins): ~50% removal <sup>37</sup>	Many commercial devices claim to remove P but lack rigorous external evaluation

References: <sup>1</sup>Hogland et al. (1987); <sup>2</sup>MWCOG (1983); <sup>3</sup>Young et al. (1996); <sup>4</sup>Leisenring et al. (2010); <sup>5</sup>Bell et al. (1995); <sup>6</sup>Horner and Horner (1995); <sup>7</sup>City of Austin (1990); <sup>8</sup>Welborn and Veenhuis (1987); <sup>9</sup>Maniquiz et al. (2010); <sup>10</sup>Claytor and Schueler (1996); <sup>11</sup>Stewart (1992); <sup>12</sup>Stormwater Management (1994); <sup>13</sup>Horner et al. (1994); <sup>14</sup>Reeves (1994); <sup>15</sup>Yousef et al. (1985); <sup>16</sup>Yu et al. (1993); <sup>17</sup>Yu and Kaighn (1995); <sup>18</sup>Yu et al. (1994); <sup>19</sup>City of Austin (1995); <sup>20</sup>Khan et al. (1992); <sup>21</sup>Winston et al. (2011); <sup>22</sup>Zhang et al. (2009); <sup>23</sup>Hathaway et al. (2008); <sup>24</sup>Berndtsson (2010); <sup>25</sup>Gain (1996); <sup>26</sup>Harper and Herr (1993); <sup>27</sup>Martin and Smoot (1986); <sup>28</sup>Yu and Benelouffok (1988); <sup>29</sup>Gibb et al. (1999); <sup>30</sup>Schueler (1999); <sup>31</sup>Hogan and Walbridge (2007); <sup>32</sup>Shaver and Maxted (1994); <sup>33</sup>Roy-Poirier et al. (2010); <sup>34</sup>US EPA (1993); <sup>35</sup>Schueler (1999); <sup>36</sup>Van de Moortel et al. (2009); <sup>37</sup>Zhang et al. (2010).

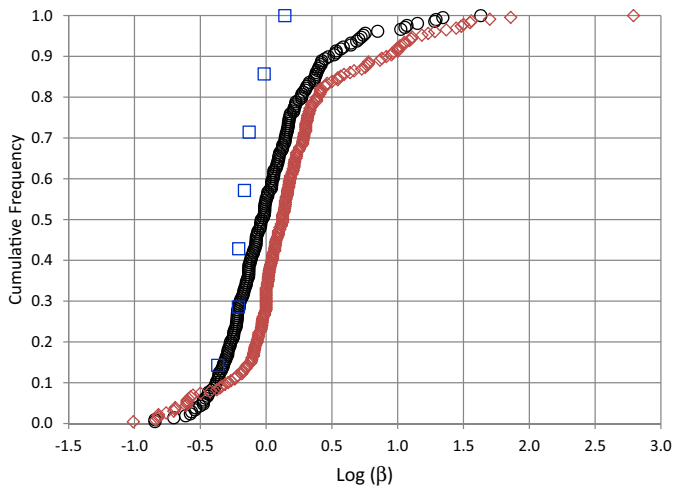
**Table 2**  
Agricultural phosphorus (P) – Transport Best Management Practices; TP: total phosphorus; PP: particulate P; SRP: soluble reactive P (i.e. PO<sub>4</sub>).

BMP	Overview	Loading reductions	References
Residue and tillage management (conservation tillage) practices	Soil surface remains partially covered with crop residue after tillage and planting to reduce soil erosion.	Up to 60–80%; TP and SRP reduction range widely (–390% to 91%)	Galloway et al. (1981), Mostaghimi et al. (1988), Schreiber and Cullum (1998)
Conservation cropping	Includes crop rotation, cover crops conservation, and double cropping.	Estimated 74% SRP, 73–88% NO <sub>3</sub> .	Bosch et al. (2009), Jiao et al. (2011)
Conservation buffers	Includes contour buffer strips of permanent, herbaceous vegetative cover, riparian forest buffers dominated by trees or shrubs adjacent to and up-slope of watercourses or water bodies, and filter strips of herbaceous vegetation.	2–93% TP, –103 to 93% SRP, 48–92% PP	McElmurry et al. (2013; Table 8)
Constructed/restored wetlands	Designed to treat wastewater and runoff primarily from agricultural processing, livestock, and aquaculture facilities, and to improve storm runoff or other flows lacking specific water quality discharge criteria.	Highly variable	Hoffmann et al. (2012), Rogers et al. (2009)
Grassed waterways	Shaped or graded vegetated channels designed to carry surface water at a non-erosive velocity to a stable outlet.	Not quantified	USDA-NRCS (2010)
Emerging technologies	Includes two-stage ditches; controlled drainage; and treatment of tile outlets with bioreactors, filters, etc.	Not quantified	Powell et al. (2007), Kroger et al. (2011), Nistor and Lowenberg-DeBoer (2007), McDowell et al. (2008)

intercept pollutants at the edge of field include filter strips, wetlands, sediment basins, bio-filters/reactors (US EPA, 2010; Miller et al., 2012). BMPs related to nutrient management have both environmental and economic benefits: off-farm nutrient transport is an investment loss for the farmers (Mullen et al., 2009). The effect of fertilizer application rate on P loss at a farm scale is related directly to application method, the hydrologic soil group, and crop type. The method of nutrient application is also related to tillage methods (Andraski et al., 1985). Nutrient management in combination with tillage and erosion practices may reduce TP loads by more than 80% but in some cases may increase the loads (Cestti et al., 2003). BMPs currently promoted in the Lake Erie U.S. watersheds are cover crops, conservation and no-tillage systems, the “4R” (right source, rate, timing, and application method) fertilizer principle, and drainage water management (McElmurry et al., 2013).

### 3.2.3. Summary

The design and evaluation of BMPs for P reduction in the Lake Erie watershed is complicated by the large variation in land use and soil types dispersed through the drainage basin, as well as by the multiplicity of governmental units (and duality of federal governments) responsible for water quality regulation. The BMPs recommended at one point in time may require adjustment as the exposed soil is modified in a dynamic fashion; for example in agricultural soils where the original organic matter and loam-rich topsoil have changed, the same BMPs applied to these areas may not have the same degree of effectiveness as in the past. The transition away from small family-owned operations to large-scale tenant farming and the associated changes in the way these farms are managed will also merit consideration. Until recently, most assessments of BMP effectiveness have focused on TP and sediment reduction, where a common assumption was that the



**Fig. 5.** Cumulative frequency of the fraction of P discharged after treatment ( $\beta$ ) for urban detention basins, reported in the International Stormwater Database (as of Jan. 4, 2013): total P ( $\circ$ ), dissolved P ( $\diamond$ ), particulate P ( $\square$ ).

majority of P losses occur as particulate P attached to sediments. As noted above, it is now increasingly apparent that more attention should be directed towards the concurrent removal and fate of dissolved P. Furthermore, P export via tile drains cannot be neglected in systematically-drained watersheds, such as the Maumee River watershed that drains to Lake Erie (King et al., 2014). The implications of this recent finding are not yet well-understood and pose additional challenges in controlling P exports.

The use of a suite of BMPs (or toolbox) may offer an appropriate approach to reducing P loading, however, as discussed above, separating the relative effectiveness of individual BMPs when applied in a suite is difficult (Bishop et al., 2005) and additional work will be required at varying scales (field to watershed) in order to provide an integrated assessment of multiple BMPs applied in concert. Modelling offers one potential approach to assess BMPs or suites of BMPs within a watershed. For example, as noted in an earlier section of this paper, Bosch et al. (2013) used the Soil Water Assessment Tool (SWAT) to evaluate the impacts of expanding the current use of filter strips, cover crops, and no-till BMPs in controlling runoff in six watersheds draining into western Lake Erie. Implemented singly and in combination, these BMPs reduced sediment and nutrient yields by 0–11%, and further reductions were observed by targeting BMPs in high source locations. Bosch et al. (2014) also evaluated the impacts of climate change induced alterations in precipitation and runoff and found that while BMPs could partially offset increased sediment and nutrient yields from agricultural watersheds, responses varied.

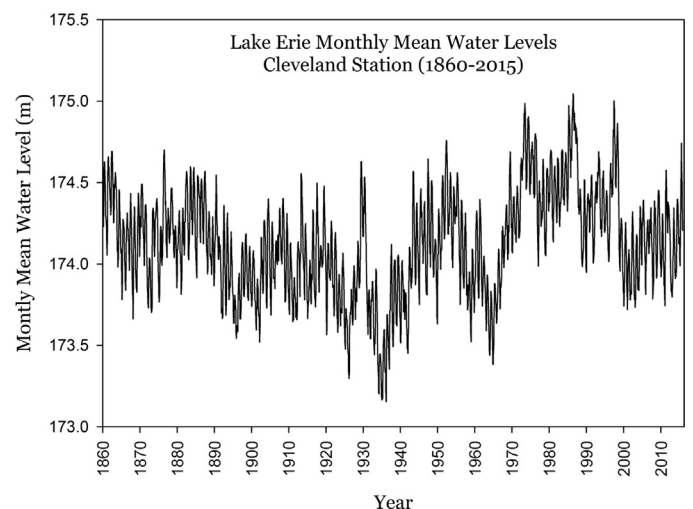
### 3.3. Coastal wetlands

Coastal wetlands affect nutrient cycling, offer important habitats for fish and wildlife, stabilize water supplies (both during floods and drought) and affect downstream water quality. This section briefly reviews coastal wetland issues in Lake Erie with emphasis on natural systems and wetland restoration; constructed wetlands are discussed in the BMP Section 3.2.1.2, and by McElmurry et al. (2013). Coastal wetlands can be categorized as lacustrine, riverine, and barrier-protected, based on the dominant water source, geomorphic position, and hydraulic connectivity to the lake (Albert et al., 2005). The Lake Erie watershed has a diversity of coastal wetland types, including

embayments, estuarine or drowned river-mouth (common in most of the tributaries entering the WB), and coastal lagoon wetlands (e.g., Presque Isle) (Herdendorf, 1992; Maynard and Wilcox, 1997), amounting to an earlier estimate of 530 km<sup>2</sup> of wetlands (Herdendorf, 1992; Maynard and Wilcox, 1997; Fig. 3). Coastal wetland loss in the Lake Erie watershed has been significant since European settlement, including ~90% in western LE/Maumee River basin (Maynard and Wilcox, 1997) and >97% for the Detroit River (Hartig et al., 2007). Key factors affecting Lake Erie coastal wetlands include water level fluctuations, seiches, sediment transport, upstream nutrient loading and artificial dikes (Mitsch, 1992), all of which can change over time.

Lake Erie water levels can vary short term (e.g. due to seiches), seasonally, annually and over longer periods with changes in net basin supplies (Quinn, 2002). Annual water level cycles typically show a peak in June (Keough et al., 1999), while longer term changes are driven in particular by climate factors (e.g. Wilcox et al., 2007; Fig. 6), though upstream contributions from the Detroit River (which supplies over 90% of the Lake Erie inflow; Herdendorf, 1992) are also affected by glacial isostatic rebound and alterations to bed geometry in the connecting rivers (IUGLS, 2009). Water level changes are important in determining wetland biotic communities, promoting higher plant diversity (e.g. Maynard and Wilcox, 1997; Wilcox and Nichols, 2008), while water level regulation (with lower year-to-year variation) can decrease wetland extent, diversity, and resilience (Maynard and Wilcox, 1997; Smith et al., 2008), and facilitate highly competitive species (e.g. cattails) and invasive species (e.g. purple loosestrife; Keough et al., 1999). Lake seiches have the potential to affect short-term wetland water balance and movement of dissolved and particulate constituents (Treibitz, 2006; Morrice et al., 2004; Bouchard, 2007). Human alterations to coastal environments affect water levels and/or nearshore processes more locally, for example through breakwalls and other structures affecting sediment supply, and artificial dikes (common along the southwest shore) that can affect water level patterns, nutrient transport and processing, plant diversity, and fish movement (Smith et al., 2008).

In addition to wetland extent, an additional important broad ecological consideration is wetland condition, which can be diminished by factors such as excessive loading of sediments and nutrients, contaminants, shoreline modification, diking, and invasive species (Maynard and Wilcox, 1997). Wetland impairment has occurred in Lake Erie with agriculture activity,



**Fig. 6.** Lake Erie monthly mean water levels measured at Cleveland (Station ID 9063063) (NOAA, 2015).

dredging, diking or other physical modifications, shoreline modification, and introduction of non-native species (e.g. common carp); the west basin has been particularly affected by these activities (Maynard and Wilcox, 1997). A recent assessment of Lake Erie wetlands overall showed “fair” scores (relating species-habitat specificity) and generally deteriorating conditions, and with Lake Ontario and the upper St. Lawrence River, common causes of degradation were water level control, nutrient enrichment, sedimentation, or some combination of these factors (EC and USEPA, 2014). A comprehensive Great Lakes basin-wide coastal wetland monitoring effort has been underway (<http://glri.us/projects/epa.html>), which will help to address gaps in condition assessment for coastal wetlands in the Lake Erie basin.

Wetland nutrient retention and processing is influenced by nutrient chemistry and inputs, hydrology, plant/microbial uptake, redox conditions and extent of oxidized and reduced soil layers. In part due to the wide diversity of wetland types, nutrient budgets (considering inflows, outflows, and internal cycling) can vary significantly and lead to variations in nutrient cycling both seasonally and between years within the same wetland. Earlier studies have identified several types of wetlands (including freshwater marshes, forested swamps) that acted as yearly, seasonal or inconsistent sinks, for P, N, or both (Mitsch et al., 1989; Mitsch and Gosselink, 2000). Intensive studies in Old Woman Creek National Estuarine Research Reserve (near Sandusky Bay) found variable DP and TP removal, which ranged up to 80% for SRP in one month (Heath, 1992) while overall TP removal was lower (~36%), especially during drought (Mitsch and Reeder, 1992). Reviews of this research have emphasized climate (including drought) and other drivers of hydrology that can influence wetland characteristics (e.g., transitioning from barrier-protected to embayment type) and nutrient transport (Heath, 1992), and the importance of considering loads and concentrations when evaluating nutrient reduction effectiveness (Krieger, 2003).

The potential for larger-scale wetland restoration to contribute to nutrient reduction for Lake Erie has been assessed in several studies. Mitsch et al. (1989) estimated that remaining wetlands in western Lake Erie retained up to 5% of nonpoint source P loading, and a broader restoration to 1000 km<sup>2</sup> of wetland coverage could increase this to 24–33%. Modelling also suggests the potential nutrient reduction benefits of large-scale wetland restoration (Wang and Mitsch, 1998; Mitsch and Wang, 2000); however, several issues need to be borne in mind. Wetlands need to be hydrologically connected to the waterbody, which may entail partial or full removal of dikes, with implications for wetland management (Wang and Mitsch, 1998). Many sub-basins are well upstream of Lake Erie (e.g. Han et al., 2012), and nutrient reduction would benefit by wetland restoration efforts closer to these sources. An additional issue is that restoration of historic wetland sites that had been converted to agriculture may mobilize nutrients, at least initially; for example, a Lake Michigan tributary showed up to 2.6 times higher SRP and TP levels downstream of restored agricultural system (Steinman and Ogdahl, 2011), while a restored North Carolina riverine wetland showed elevated SRP export and potential to continue for a significant period following flooding (Ardon et al., 2010). Additionally, there is the potential for conflict in multiple restoration objectives (Euliss et al., 2008) – for example, nutrient retention in the wetland may be achieved at the expense of improved fish and wildlife habitat. Indeed broader wetland restoration efforts often target excessive nutrient levels in the wetlands themselves through upstream activities (e.g. on agricultural fields) or in situ techniques (Wilcox and Whillans, 1999).

Climate change has important implications for nutrient cycling and coastal wetland condition, including winter storm damage with decreased ice cover, vegetation community changes or shifts with decreased lake levels or warmer temperatures, changes to barriers beaches, and increased spread of invasive species (Mortsch et al., 2006). Potential effects on wetland nutrient cycling include increased watershed nutrient export (and loading to downstream wetlands) via spring storms (Michalak et al., 2013), and P mobilization in coastal sediments from desiccation and inundation episodes (Steinman et al., 2012).

Overall, wetlands restoration has the potential to contribute to Lake Erie nutrient reduction goals. Such efforts would benefit from appropriate targeting, more systematic research and monitoring, and consideration of broader wetland management objectives. Several issues should be considered in wetland restoration aimed at nutrient reduction to Lake Erie, notably the potential for initial increased nutrient export, nutrient saturation, effects of climate change and the need to situate nutrient reduction amongst the broader suite of wetland services and values. Further research is needed, including identifying priority locations for wetland restoration and how restoration efforts can meet multiple objectives in the basin, including nutrient reduction.

### 3.4. Internal loading: phosphorus

#### 3.4.1. Underlying mechanisms of internal loading and their importance in Lake Erie

A better understanding of the nutrient dynamics within Lake Erie is required in order to forecast future water quality and to target remedial actions and Best Management Practices more effectively. In particular, one of the poorly known components of the nutrient dynamics is the amount of P that is recycled to the water column by internal loading. This unknown loading creates uncertainty in the determination of the quantity of P delivered to the lake, and in estimates of the lag time between reductions in tributary loading and expected improvements in lake water quality. Externally delivered P to any lake undergoes a complexity of in-lake processes, including physical transport, biological uptake and chemical transformations before deposition onto the sediment. Much of the P deposited on the bottom is regenerated by microbial degradation of organic matter and migrates upward to the sediment surface where it is partially retained on iron and manganese oxyhydroxides under aerobic conditions (Søndergaard et al., 2003) and partially fluxes into the water column. As a result, many lakes exhibit a delayed response to reductions in external loading (Søndergaard et al., 2003), and thus quantifying these recycling processes will provide a better understanding of system response times and expectations for recovery. There are three general types of internal P cycling in Lake Erie: inter-basin transfer, biological transformation and sediment regeneration. First, much of the externally loaded P is delivered to the west basin, a portion of which is cycled while the remainder is transported as loading to the central and eastern basins. Second, numerous biological transformations recycle P to various trophic levels, the water column and the sediment. Third, P may be released from anoxic sediment (Mortimer, 1941), or from sediments that have become resuspended due to wind, waves or anthropogenic activities such as dredging (e.g. for the Toledo Shipping Channel).

#### 3.4.2. Inter-basin transfers

There is currently insufficient information to evaluate the magnitude of these transfers within Lake Erie and whether they have changed over time. Sediment cores from the 1970s indicated a large discrepancy between net sedimentation rate in the west basin and the mass balance of external P inflows. It was proposed

that this reflected an unsampled flux of material along the bottom into the central basin (Burns et al., 1976), which has never been verified.

### 3.4.3. Biological recycling

This process is highly complex and may include uptake and excretion by all levels of the aquatic foodweb, as well as through decay, sedimentation, and bioturbation. Early empirical management models (e.g., Vollenweider, 1968) used simple linear model fits of annual or spring P concentrations or load, and seasonal average measures of total algal biomass. The unknown underlying processes of internal P recycling were integrated as a 'sedimentation coefficient'. These models were largely successful, albeit with a large uncertainty around their predictions, and none of the underlying complexities of internal loading prevented the success of 1970s nutrient load reductions. Since then, however, fundamental changes in the nature of the inputs (from point source to diffuse) and trophic structure of the Lake Erie biological community has invoked major changes in biological P sequestration and exchange and the recent resurgence of algal blooms has stimulated interest in the key details of P recycling, including sediment chemistry, to help evaluate the effects of load variations. In polymictic lakes with iron poor sediments, for example, internal P loading can lower the N:P ratio and may stimulate cyanobacterial blooms (Orihel et al., 2015).

Perhaps one of the most widely acknowledged changes in Lake Erie occurred with the widespread colonization by invasive dreissenid mussels, which profoundly altered the light and nutrient regimes, and the foodweb structure and response to nutrient inputs. Dreissenid veligers represent a relatively new sedimentation vector in Lake Erie which may also affect inter-basin transfers, although this process may not be easily discriminated. Dreissenids have increased the efficiency of internal P recycling and altered the inshore-offshore exchange of materials and nutrients, trapping these in the warmer and shallower nearshore zones in what is termed as the 'nearshore shunt' (Hecky et al., 2004). Zhang et al. (2011) modelled dreissenid P uptake and excretion and concluded that currently, Lake Erie may be more sensitive to P inputs than during the pre-mussel period. Mussel grazing can be equivalent to that of zooplankton but in addition, dreissenids also recycle detrital seston which would otherwise form sediment. Mussel excretion from colonized lake bottom areas can therefore return more P to the water than non-colonized substrates. Nevertheless, Zhang et al. (2008) asserted that algal productivity is "ultimately regulated by external P loads" (but see Section 2.1.2 on benthic HABs).

### 3.4.4. Sediment P regeneration to the water column

Sediment regeneration includes sediment release (as SRP) during hypolimnetic hypoxia (mainly in the central basin), aerobic decomposition of organic matter from the sediment-water interface, and resuspended material in the water column. Most interest has been in the first, because the degree of hypoxia is influenced by organic inputs from both external sources and lake productivity. Both are related to external P loading, one of the few controllable factors in this series. Benthic P release may also occur under oxic bottom water conditions, driven by resuspension events, organic matter degradation and biologically enhanced transport from greater sediment depths (Slomp et al., 1998; Meile and Van Cappellen, 2003; Katsev et al., 2006; Kim et al., 2013). In this case, benthic release is influenced by P retention in the underlying anoxic sediment (Gächter and Müller, 2003; Moosmann et al., 2006).

An early study by Burns and Ross (1972) concluded that the rate of hypoxic sediment P regeneration in the central basin can be as high as the rate of external loading over a two month period. Data

from later studies suggest, however, that while hypoxic sediment P regeneration in the central basin occurs, its significance and frequency is unclear and should be examined in detail. High SRP concentrations in hypolimnetic water are observed intermittently, but not commonly reported. Burns (1976) found about half of the central basin in 1970 was covered by an anoxic hypolimnion layer with up to 93  $\mu\text{g/L}$  SRP; long term monitoring data collected from samples 1–2 m above the bottom since then suggest that this degree of P release does not occur with high frequency (e.g. Environment Canada and EPA data; 1970–present). Less dramatic events may be identified with future efforts using probes that provide a greater temporal resolution. Such higher resolution sampling could be used to test for P release that may vary on a diurnal basis and/or may be rapidly dissipated into the water column by bottom currents (Rao et al., 2008) or rapidly consumed by algal and/or bacterial uptake. Incorporation of epilimnetic water into the hypolimnion or the advent of fall de-stratification can produce an oxygen flux that can re-precipitate iron-phosphorus complexes previously released as SRP and ferrous iron from anoxic sediment (Burns and Ross, 1972).

Under anaerobic conditions in the water overlying the sediment, the oxyhydroxides are reduced and release their sorbed P (Boström et al., 1998). A number of different techniques have been employed to quantify this internal P loading. For example, in Haringvliet Lake (The Netherlands) biogeochemical modelling showed that ~56% of the total P deposited on the sediment was returned to the overlying water through diffusion and bio-irrigation (the pumping of pore water by benthic macroinvertebrates through their burrows) (Canavan et al., 2006). Studies by Nürnberg et al. (2013) and Loh et al. (2013) used hypolimnion TP concentration changes and core incubations (respectively) and estimated the internal P loading in Lake Simcoe (Ontario) to be 45–89% of the external load. James (2012) incubated intact sediment cores from Lake of the Woods (Minnesota) under both aerobic and anaerobic conditions, and found that the diffusive P flux ranged from 8.3 to 12.5  $\text{mg m}^{-2} \text{d}^{-1}$  under anaerobic conditions and from 0.2 to 0.6  $\text{mg m}^{-2} \text{d}^{-1}$  under aerobic conditions. He concluded that the diffusive flux can represent an important contribution to the P cycling in this multi-basined lake. North et al. (2015) reported that under anoxic conditions internal P fluxes from Lake Diefenbaker, Saskatchewan are about 9 times higher than those under aerobic conditions, and that the internal loads in the winter are about the same as in the summer. The benthic P flux as a function of bottom water oxygen concentrations has also been measured using in situ bottom chambers (Noffke et al., 2012). In Lake Erie, Matisoff et al. (2016) used a variety of techniques to estimate the amount of P that is recycled to the water column by internal loading from the bottom sediments in the western basin. Fluxes under aerobic conditions at summertime temperatures averaged 1.35  $\text{mg P/m}^2/\text{d}$  and displayed spatial variability on scales as small as a centimetre. Using two different temperature correction factors the flux was adjusted to mean annual temperature yielding average annual fluxes of 0.43–0.91  $\text{mg P/m}^2/\text{d}$  and a western basin-wide total of 378–808  $\text{Mg P/y}$  as the diffusive flux from sediments. This is 3–7% of the 11,000  $\text{Mg P/y}$  IJC target load for P delivery to Lake Erie from external sources. Based on these average aerobic fluxes, the sediment contributes 3.0–6.3  $\mu\text{g P/L}$  as a background internal contribution that represents 20–42% of the GLWQA interim Target Concentration of 15  $\mu\text{g P/L}$  for the western basin. The implication is that this internal diffusive recycling of P is unlikely to trigger cyanobacterial blooms by itself but is sufficiently large to cause blooms when combined with external loads. This background flux may also contribute to a delayed response of the lake to any decrease in the external loading.

Other mechanisms may contribute to sediment P recycling in Lake Erie. Biologically enhanced transport of dissolved or adsorbed



PO<sub>4</sub> from sediment (i.e. bio-translocation) can play a significant and often overlooked role in sediment P exchange: migrating algae or cyanobacteria may incorporate P in deeper strata and transport it as particulate P (PP) into the epilimnion (Spears et al., 2007). Wind/wave action and resuspension of sediment P around the edges of the hypolimnion may produce prolonged increases in epilimnetic TP and DP in late summer, as occurred in 1970 (Burns, 1976) and 1968–1982 (Rosa and Burns, 1987). Sediment pore water SRP concentrations can reach over 1 mg/L or ~1000 times those in lake water (Azcue et al., 1996) while resuspended TP can be 40% soluble (Burns, 1976). The quantity of P released during resuspension depends on the frequency and depth of disturbance of the sediment bottom, the concentration and form of P in the sediment, partitioning between DP and PP, and whether the bottom waters are oxic or anoxic. Recently, Matisoff and Carson (2014) estimated that 68–86% of the sediment in suspension in the west basin is resuspended from the bottom. Under aerobic conditions, and assuming a phosphate exchange between the resuspended sediment and the lake water, this translates to an internal loading of about 300–400 MTA P (Hummel, 2014). Hummel (2014) obtained internal P loads as high as 9000 MTA under anoxic conditions and with sediments from the central basin. Aerobic decay of biological material can occur anywhere in the water column or in shallow sediments – as seen, for example, in the Bay of Quinte (Nicholls, 1999). As the process is somewhat temperature dependent, this would be a negative factor in the west basin if water temperatures increase under climate change.

#### 3.4.5. Implications of internal loading/recycling

Internal loading is sometimes discussed as a mechanism that would delay a response to costly nutrient controls, which can generate debate on the rationale for such controls and stall management action. Nevertheless an understanding of the expected response time of the lake to changes in external loadings is essential to evaluate the effectiveness of management actions. It can also be argued that prolonged delay in nutrient management is likely to further extend the lag before any sustained improvement occurs. Regardless of whether P is regenerated during hypoxia or by resuspension, excessive accumulation of sediment P can be limited through management of external nutrient loads. Once in the lake, there are no management options to eliminate P from a lake the size of Erie, except to wait for natural processes to take their course. Earlier remedial efforts under the GLWQA resulted in a 50% reduction in TP in Lake Erie within about 10 years, despite internal recycling. Burns et al. (1976) estimated that ~92% of the P entering Lake Erie is retained in the lake, but the proportion recycled as internal loading has yet to be directly quantified, hence it is difficult to estimate the likely delay in response to management actions. A simple calculation, assuming that 92% of the P is recycled each year, indicates that only 30–50% of the external P input will remain after about 10 years. A recent review showed that most European shallow lakes reached equilibrium with reduced P loads in 10–15 years (Jeppesen et al., 2007) – consistent with the response time of Lake Erie to the 1972 GLWQA loading reductions and with the simple calculation of P retention above. It may thus be reasonable to expect that the response of Lake Erie to P management actions would not be unduly delayed past 10–15 years, but regime shifts, changes in the nature and timing of the external loading and climate change may moderate the lake's response and prompt a re-evaluation of expected response times.

## 4. Conclusions

Lake Erie is the most vulnerable and impacted of the Great Lakes, and the re-eutrophication and extensive algal blooms and

severe hypoxia in this important waterbody have seen much national and international attention in recent years. In 2012, the International Joint Commission convened a task team to evaluate the conditions in Lake Erie and develop mitigation strategies (e.g. IJC, 2014). Although HABs and hypoxia in Lake Erie are influenced by the natural variability of hydro-meteorological conditions in the basin, it is clear that changes in agricultural practices have contributed to a significant increase in bio-available phosphorus loading to the lake. A suite of agricultural best management actions in the watershed may offer an appropriate approach to reducing P loading to the lake. Further, proliferation of dreissenid mussels and their influence on ecological changes in the lake requires further assessment. Most recently, the governments of Canada and the United States have adopted revised P loading targets for Lake Erie, designed to reduce the extent and severity of cyanobacterial blooms as well as the extent of hypoxia in the central basin (Binational.net, 2015). P targets designed to reduce *Cladophora* growth in the east basin remain unresolved (Binational.net, 2015) but once defined, may also be incorporated. The success story of the 1970s and 1980s for the Great Lakes provides a clear and inspirational example of effective, science-based remedial action, based on active engagement by all sectors at local, regional, national and binational levels. The recent changes in Lake Erie nevertheless illustrate a need for continued vigilance and an adaptive management strategy – which integrates monitoring, research, and modelling and is based on sound science and a standardized, targeted monitoring of key measures of ecosystem response.

## Acknowledgement

The authors wish to acknowledge the support and incentive provided by the IJC in the initial genesis of this review.[CG]

## References

- Albert, D.A., Wilcox, D.A., Ingram, J.W., Thompson, T.A., 2005. Hydrogeomorphic classification for Great Lakes coastal wetlands. *J. Great Lakes Res.* 31 (S1), 129–146.
- Alfara, L.K., Weismiller, R.A., 2002. *U.S. Experiences With Nutrient Management and Good Management Practices to Control Non-point Pollution From Agriculture*. ESSD Unit, The World Bank, Washington, DC 62 pp.
- Allinger, L.E., Reavie, E.D., 2013. The ecological history of Lake Erie as recorded by the phytoplankton community. *J. Great Lakes Res.* 39 (3), 365–382.
- Anderson, T.R., 2005. Plankton functional type modelling: running before we can walk? *J. Plankton Res.* 27 (11), 1073–1081.
- Andraski, B.J., Mueller, D.H., Daniel, T.C., 1985. Phosphorus losses in runoff as affected by tillage. *Soil Sci. Soc. Am. J.* 49 (6), 1523–1527.
- Angel, J.R., Kunkel, K.E., 2009. The response of Great Lakes water levels to future climate scenarios with an emphasis on Lake Michigan-Huron. *J. Great Lakes Res.* 36, 51–58.
- Ardon, M., Montanari, S., Morse, J.L., Doyle, M.W., Bernhardt, E.S., 2010. Phosphorus export from a restored wetland ecosystem in response to natural and experimental hydrologic fluctuations. *J. Geophys. Res.* 115, G04031.
- Arend, K.K., Beletsky, D., DePinto, J.V., Ludsin, S.A., Roberts, J.J., Rucinski, D.K., Scavia, D., Schwab, D.J., Hook, T.O., 2011. Seasonal and interannual effects of hypoxia on fish habitat quality in central Lake Erie. *Freshw. Biol.* 56, 366–383.
- Arhonditsis, G.B., Brett, M.T., 2004. Evaluation of the current state of mechanistic aquatic biogeochemical modeling. *Mar. Ecol. Prog. Ser.* 271, 13–26.
- Arhonditsis, G.B., Qian, S.S., Stow, C.A., Lamon, E.C., Reckhow, K.H., 2007. Eutrophication risk assessment using Bayesian calibration of process-based models: application to a mesotrophic lake. *Ecol. Model.* 208 (2–4), 215–229.
- Auer, M.T., Tomlinson, L.M., Higgins, S.N., Malkin, S.Y., Howell, E.T., Bootsma, H.A., 2010. Great Lakes *Cladophora* in the 21st century: same algae – different ecosystem. *J. Great Lakes Res.* 36, 248–255.
- Azcue, J.M., Rosa, F., Mudroch, A., 1996. Distribution of major and trace elements in sediments and pore water of Lake Erie. *J. Great Lakes Res.* 22 (2), 389–402.
- Baker, D.B., Confesor, R., Ewing, D.E., Johnson, L.T., Kramer, J.W., Merryfield, B.J., 2014. Phosphorus loading to Lake Erie from the Maumee, Sandusky and Cuyahoga rivers: the importance of bioavailability. *J. Great Lakes Res.* 40 (3), 502–517.
- Barbiero, R.P., Rockwell, D.C., Warren, G.J., Tuchman, M.L., 2006. Changes in spring phytoplankton communities and nutrient dynamics in the eastern basin of Lake Erie since the invasion of *Dreissena* spp. *Can. J. Fish Aquat. Sci.* 63, 1549–1563.

- Bartolai, A.M., He, L., Hurst, A.E., Mortsch, L., Paehlke, R., Scavia, D., 2015. Climate change as a driver of change in the Great Lakes St. Lawrence River basin. *J. Great Lakes Res.* 41, 45–58.
- Bell, W., Stokes, L., Gavan, L.J., Nguyen, T.N., 1995. Assessment of the Pollutant Removal Efficiencies of Delaware Sand Filter BMPs. City of Alexandria, Department of Transportation and Environmental Services, Alexandria, VA.
- Bennington, V., McKinley, G.A., Kimura, N., Wu, C.H., 2010. General circulation of Lake Superior: mean, variability, and trends from 1979–2006. *J. Geophys. Res.* 115, C12015.
- Berezina, N.A., Golubkov, S.M., 2008. Effect of drifting macroalgae *Cladophora glomerata* on benthic community dynamics in the easternmost Baltic Sea. *J. Mar. Syst.* 74 (S1), 80–85.
- Berndtsson, J.C., 2010. Green roof performance towards management of runoff water quantity and quality: a review. *Ecol. Eng.* 36 (4), 351–360.
- Bertram, P.E., 1993. Total phosphorus and dissolved oxygen trends in the central basin of Lake Erie, 1970–1991. *J. Great Lakes Res.* 19 (2), 224–236.
- Bierman, V.J., Kaur, J., DePinto, J.V., Feist, T.J., Dilks, D.W., 2005. Modeling the role of zebra mussels in the proliferation of blue-green algae in Saginaw Bay, Lake Huron. *J. Great Lakes Res.* 31 (1), 32–55.
- Binational.net, 2015. Recommended Phosphorus Loading Targets for Lake Erie. Annex 4 Objectives and Targets Development Task Team Final Report to the Nutrients Annex Subcommittee, <http://binational.net/2015/06/30/draftptargets-ciblesproposeesdep/>.
- Binding, C.E., Greenberg, T.A., Bukata, R.P., 2012. An analysis of MODIS-derived algal and mineral turbidity in Lake Erie. *J. Great Lakes Res.* 38 (1), 107–116.
- Bishop, P.L., Hively, W.D., Stedinger, J.R., Rafferty, M.R., Lojpersberger, J.L., Bloomfield, J.A., 2005. Multivariate analysis of paired watershed data to evaluate agricultural best management practice effects on stream water phosphorus. *J. Environ. Qual.* 34 (3), 1087–1101.
- Bižić-Ionescu, M., Amann, R., Grossart, H.-P., 2014. Massive regime shifts and high activity of heterotrophic bacteria in an ice-covered lake. *PLOS ONE* 9 (11), e113611, <http://dx.doi.org/10.1371/journal.pone.0113611>.
- Bosch, I., Makarewicz, J.C., Lewis, T.W., Bonk, E.A., Finiguerra, M., Groveman, B., 2009. Management of agricultural practices results in declines of filamentous algae in the lake littoral. *J. Great Lakes Res.* 35, 90–98.
- Bosch, N.S., Allan, J.D., Selegean, J.P., Scavia, D., 2013. Scenario-testing of agricultural best management practices in Lake Erie watersheds. *J. Great Lakes Res.* 39 (3), 429–436.
- Bosch, N.S., Evans, M.A., Scavia, D., Allan, J.D., 2014. Influence of climate change on the effectiveness of agricultural best management practices. *J. Great Lakes Res.* 40, 581–589.
- Boström, B., Andersen, J.M., Fleischer, S., Jansson, M., 1998. Exchange of phosphorus across the sediment-water interface. *Hydrobiologia* 170, 229–244.
- Bouchard, V., 2007. Export of organic matter from a coastal freshwater wetland to Lake Erie: an extension of the outwelling hypothesis. *Aquat. Ecol.* 41, 1–7.
- Boudreau, B.P., 1997. Diagenetic Models and Their Implementation: Modelling Transport and Reactions in Aquatic Sediments. Springer-Verlag, Berlin 430 pp.
- Boyer, G.L., 2007. The occurrence of cyanobacterial toxins in New York lakes: lessons for the MERHAB-Lower Great lakes program. *Lake Res. Manage.* 23 (2), 153–160.
- Brandt, S., Costantini, M., Kolesar, S., Ludsin, S.A., Mason, D.M., Rae, C.M., Zhang, H., 2011. Does hypoxia reduce habitat quality for Lake Erie walleye (*Sander vitreus*)? A bioenergetics perspective. *Can. J. Fish. Aquat. Sci.* 68, 857–879.
- Briertburg, D.L., 2002. Effects of hypoxia and the balance between hypoxia and enrichment on coastal fishes and fisheries. *Estuaries* 25, 767–781.
- Breitburg, D.L., Loher, T., Pacey, C.A., Gerstein, A., 1997. Varying effects of low dissolved oxygen on trophic interactions in an estuarine food web. *Ecol. Monogr.* 67, 489–507.
- Breitburg, D.L., Hondorp, D.W., Davies, L.W., Diaz, R.J., 2009. Hypoxia, nitrogen and fisheries: Integrating effects across local and global landscapes. *Ann. Rev. Mar. Sci.* 1, 329–350.
- Bridgeman, T.B., Penamon, W.A., 2010. *Lyngbya wollei* in western Lake Erie. *J. Great Lakes Res.* 36 (1), 167–171.
- Bridgeman, T.B., Schloesser, D.W., Krause, A.E., 2006. Recruitment of *Hexagenia mayfly* nymphs in western Lake Erie linked to environmental variability. *Ecol. Appl.* 16, 601–611.
- Bridgeman, T.B., Chaffin, J.D., Kane, D.D., Conroy, J.D., Panek, S.E., Armenio, P.M., 2012. From river to lake: phosphorus partitioning and algal community compositional changes in western Lake Erie. *J. Great Lakes Res.* 38 (1), 90–97.
- Brittain, S.M., Wang, J., Babcock-Jackson, L., Carmichael, W.W., Rinehart, K.L., Culver, D.A., 2000. Isolation and characterization of microcystins, cyclic heptapeptide hepatotoxins from a Lake Erie strain of *Microcystis aeruginosa*. *J. Great Lakes Res.* 26 (3), 241–249.
- Burlakova, L.E., Karatayev, A.Y., Pennuto, C., Mayer, C., 2014. Changes in Lake Erie benthos over the last 50 years: historical perspectives, current status, and main drivers. *J. Great Lakes Res.* 40 (3), 560–573.
- Burns, N.M., 1976. Temperature, oxygen, and nutrient distribution patterns in Lake Erie, 1970. *J. Fish. Res. Board Can.* 33 (3), 485–511.
- Burns, N.M., Ross, C., 1972. Oxygen-nutrient relationships within the Central Basin of Lake Erie. In: Burns, N.M., Ross, C. (Eds.), Project Hypo. Paper No. 6. Canada Centre for Inland Waters. USEPA Tech. Rept. TS-05-71-208-24, Burlington, Ontario, pp. 85–119.
- Burns, N.M., Williams, J.D.H., Jaquet, J.M., Kemp, A.L.W., Lam, D.C., 1976. A phosphorus budget for Lake Erie. *J. Fish. Res. Board Can.* 33 (3), 564–573.
- Burns, N.M., Rockwell, D.C., Bertram, P.E., Dolan, D.M., Ciborowski, J.J.H., 2005. Trends in temperature, secchi depth, and dissolved oxygen depletion rates in the central basin of Lake Erie, 1983–2002. *J. Great Lakes Res.* 31 (S2), 35–49.
- Burniston, D., McCrear, R., Klawunn, P., Ellison, R., Thompson, A., Bruxer, J., 2009. Detroit River Phosphorus Loading Determination. Internal Environment Canada, Water Quality Monitoring and Surveillance report # WQMS09-006, Contribution # 09-757 44 pp.
- Canavan, R., Slomp, C., Jourabchi, P., Van Cappellen, P., Laverman, A., van den Berg, G., 2006. Organic matter mineralization in sediment of a coastal freshwater lake and response to salinization. *Geochim. Cosmochim. Acta* 70, 2836–2855.
- Carey, C.C., Ibelings, B.W., Hoffmann, E.P., Hamilton, D.P., Brookes, J.D., 2012. Eco-physiological adaptations that favour freshwater cyanobacteria in a changing climate. *Water Res.* 46, 1394–1407.
- Carmichael, W.W., 2008. A world overview – one-hundred-twenty-seven years of research on toxic cyanobacteria – where do we go from here? In: Hudnell, H.K. (Ed.), *Cyanobacterial Harmful Algal Blooms: State of the Science and Research Needs*. Springer, New York, pp. 105–125.
- Carmichael, W.W., Boyer, G.L., in press. Estimating health impacts from cyanobacteria harmful algal blooms: the North American Great Lakes. *Harmful Algae* (in press).
- Carmichael, W.W., Evans, W.R., Yin, Q.Q., Bell, P., Moczydlowski, E., 1997. Evidence for paralytic shellfish poisons in the freshwater cyanobacterium *Lyngbya wollei* (Farlow ex Gomont) comb. nov. *Appl. Environ. Microbiol.* 63 (8), 3104–3110.
- Carrick, H.J., Schelske, C.L., 1997. Have we overlooked the importance of small phytoplankton in productive waters? *Limnol. Oceanogr.* 42 (7), 1613–1621.
- Casper, V.L., 1965. A phytoplankton bloom in western Lake Erie Proceedings of the 8th Conference of Great Lakes Research University of Michigan. *Great Lakes Res. Div.* 13, 29–35.
- Cestti, R., Srivastava, J., Jung, S., 2003. Agriculture Non-point Source Pollution Control: Good Management Practices – The Chesapeake Bay Experience. World Bank, Washington, DC 49 pp.
- Chaffin, J., Bridgeman, T., 2014. Organic and inorganic nitrogen utilization by nitrogen-stressed cyanobacteria during bloom conditions. *J. Appl. Phycol.* 26 (1), 299–309.
- Charlton, M.N., 1979. Hypolimnetic Oxygen Depletion in Central Lake Erie: Has There Been Any Change? Scientific Series No. 110. Inland Waters Directorate, Burlington Ontario.
- Charlton, M.N., 1980a. Oxygen depletion in Lake Erie: has there been any change? *Can. J. Fish. Aquat. Sci.* 37 (7), 72–81.
- Charlton, M.N., 1980b. Hypolimnion oxygen consumption in lakes: discussion of productivity and morphometry effects. *Can. J. Fish. Aquat. Sci.* 37, 1531–1539.
- Charlton, M.N., Milne, J.E., Booth, W.G., Chiochio, F., 1993. Lake Erie Offshore in 1990 – restoration and resilience in the central basin. *J. Great Lakes Res.* 19 (2), 291–309.
- Cheng, V., Arhonditsis, G.B., Brett, M.T., 2010. A reevaluation of lake-phosphorus loading models using a Bayesian hierarchical framework. *Ecol. Res.* 25 (1), 59–76.
- Chun, C.L., Oschner, U., Byappanahalli, M.N., Whitman, R.L., Tepp, W.H., Lin, G., Johnson, E.A., Peller, J., Sadowsky, M.J., 2015. Association of toxin producing *Clostridium botulinum* with the macroalgae *Cladophora* in the Great Lakes. *Environ. Sci. Technol.* 47 (6), 2587–2594.
- City of Austin, Texas, 1990. Removal Efficiencies of Storm Water Control Structures. Environmental Resource Division, Environmental and Conservation Services Department 36 pp.
- City of Austin, Texas, 1995. Characterization of Stormwater Pollution for the Austin, Texas Area. Environmental Resources Management Division, Environmental and Conservation Services Department, City of Austin, Austin, TX.
- Claytor, R.A., Schueler, T.R., 1996. Design of Stormwater Filtering Systems. The Center for Watershed Protection, Silver Spring, MD.
- Conroy, J.D., Edwards, W.J., Pontius, R.A., Kane, D.D., Zhang, H., Shea, J.F., Richey, J.N., Culver, D.A., 2005. Soluble nitrogen and phosphorus excretion of exotic freshwater mussels (*Dreissena* spp.): potential impacts for nutrient remineralisation in western Lake Erie. *Freshw. Biol.* 50 (7), 1146–1162.
- Cornett, R.J., Rigler, F.H., 1979. Hypolimnion oxygen deficits: their prediction and interpretation. *Science* 205, 580–581.
- Cousino, L.K., Becker, R.H., Zmijewski, K., 2015. Modeling the effects of climate change on water, sediment, and nutrient yields from the Maumee River watershed. *J. Hydrol.* 4 (Part B), 762–775.
- Dalóglu, I., Cho, K.H., Scavia, D., 2012. Evaluating causes of trends in long-term dissolved reactive phosphorus loads to Lake Erie. *Environ. Sci. Technol.* 46 (19), 10660–10666.
- Davies, J.M., Hecky, R.E., 2005. Initial measurements of benthic photosynthesis and respiration in Lake Erie. *J. Great Lakes Res.* 31 (S2), 195–207.
- Davis, T.W., Harke, M.J., Marcoval, M.A., Goleski, J., Orano-Dawson, C., Berry, D.L., Gobler, C.J., 2010. Effects of nitrogenous compounds and phosphorus on the growth of toxic and non-toxic strains of *Microcystis* during cyanobacterial blooms. *Aquat. Microb. Ecol.* 61 (2), 149–162.
- Davis, T.W., Koch, W., Marcoval, M.A., Wilhelm, S.W., Gobler, C., 2012. Mesozooplankton and microzooplankton grazing during cyanobacterial blooms in the western basin of Lake Erie. *Harmful Algae* 15, 26–35.
- Davis, T.W., Watson, S.B., Rozmarynowycz, M.J., Ciborowski, J.J.H., McKay, R.M., Bullerjahn, G.S., 2014. Phylogenies of microcystin-producing cyanobacteria in the Lower Laurentian Great Lakes suggest extensive genetic connectivity. *PLOS ONE* 9 (9), e106093, <http://dx.doi.org/10.1371/journal.pone.0106093>.
- Davis, T.W., Bullerjahn, G.S., Tuttle, T., McKay, R.M., Watson, S.B., 2015. Effects of increasing nitrogen and phosphorus concentrations on phytoplankton

- community growth and toxicity during *Planktothrix* blooms in Sandusky Bay, Lake Erie. *Environ. Sci. Technol.* 49 (12), 7197–7207.
- Dayton, A.I., Auer, M.T., Atkinson, J.F., 2014. *Cladophora*, mass transport, and the near shore phosphorus shunt. *J. Great Lakes Res.* 40 (3), 790–799.
- De Stasio, B.T., Schrimpf, M.B., Cornwell, B.H., 2014. Phytoplankton communities in Green Bay, Lake Michigan after invasion by dreissenid mussels: increased dominance by cyanobacteria. *Diversity* 6, 681–704.
- Delorme, L.D., 1982. Lake Erie oxygen: the prehistoric record. *Can. J. Fish. Aquat. Sci.* 39 (7), 1021–1029.
- Depew, D.C., Stevens, A.W., Smith, R.E.H., Hecky, R.E., 2009. Detection and characterization of benthic filamentous algal stands (*Cladophora* sp.) on rocky substrata using a high frequency echo sounder. *Limnol. Oceanogr. Methods* 7 (10), 693–705.
- Depew, D.C., Houben, A.J., Guildford, S.J., Hecky, R.E., 2011. Distribution of nuisance *Cladophora* in the lower Great Lakes: Patterns with land use, near shore water quality and dreissenid abundance. *J. Great Lakes Res.* 37 (4), 656–671.
- Dietz, M.E., Clausen, J.C., Filchak, K.K., 2004. Education and changes in residential nonpoint source pollution. *Environ. Manage.* 34 (5), 684–690.
- Dobson, H.H., Gilbertson, M., 1971. Oxygen depletion in the central basin of Lake Erie, 1929–1970. *Proc. Conf. Great Lakes Res.* 12, 743–748.
- Dodds, W.K., Gudder, D.A., 1992. The ecology of *Cladophora*. *J. Phycol.* 28 (4), 415–427.
- Domenici, P., Herbert, N.A., Lefrançois, C., Steffensen, J.F., McKenzie, D.J., 2013. The effect of hypoxia on fish swimming performance and behaviour. In: Palstra, A.P., Planas, J.V. (Eds.), *Swimming Physiology of Fish: Towards Using Exercise to Farm a Fit Fish in Sustainable Aquaculture*. Springer, Berlin, pp. 129–159.
- Dolan, D.M., McGunagle, K.P., 2005. Lake Erie total phosphorus loading analysis and update: 1996–2002. *J. Great Lakes Res.* 31 (S2), 11–22.
- Dolan, D.M., Chapra, S.C., 2012. Great Lakes total phosphorus revisited: 1. Loading analysis and update (1994–2008). *J. Great Lakes Res.* 38, 730–740.
- Dolman, A.M., Rucker, J., Pick, F.R., Fastner, J., Rohrlack, T., Mischke, U., Wiedner, C., 2012. Cyanobacteria and cyanotoxins: the influence of nitrogen versus phosphorus. *PLoS ONE* 7 (6), e38757, <http://dx.doi.org/10.1371/journal.pone.0038757>.
- Donald, D.B., Bogard, M.J., Finlay, K., Bunting, L., Leavitt, P.R., 2013. Phytoplankton-specific response to enrichment of phosphorus-rich surface waters with ammonium, nitrate, and urea. *PLoS ONE* 8 (1), e53277, <http://dx.doi.org/10.1371/journal.pone.0053277>.
- Downing, J.A., Watson, S.B., McCauley, E., 2001. Predicting cyanobacteria dominance in lakes. *Can. J. Fish. Aquat. Sci.* 58 (10), 1905–1908.
- D'Souza, N.A., Kawarasaki, Y., Gantz, J.D., Lee Jr., R.E., Beall, B.F.N., Shtarkman, Y.M., Kocer, Z.A., Rogers, S.O., Wilmshutte, H., Bullerjahn, G.S., McKay, R.M.L., 2013. Diatom assemblages promote ice formation in large lakes. *ISME J.* 7 (8), 1632–1640.
- Dyble, J., Fahnenstiel, G.L., Litaker, R.W., Millie, D.F., Tester, P.A., 2008. Microcystin concentrations and genetic diversity of *Microcystis* in the lower Great Lakes. *Environ. Toxicol.* 23, 507–516.
- Eby, L.A., Crowder, L.B., 2002. Hypoxia-based habitat compression in the Neuse River estuary: context-dependent shifts in behavioral avoidance thresholds. *Can. J. Fish. Aquat. Sci.* 59, 952–965.
- Ekau, W., Auel, H., Pörtner, H.-O., Gilbert, D., 2010. Impacts of hypoxia on the structure and processes in pelagic communities (zooplankton, macro-invertebrates and fish). *Biogeosciences* 7, 1669–1699.
- Elser, J.J., 1999. The pathway to noxious cyanobacteria blooms in lakes: the food web as the final turn. *Freshw. Biol.* 42 (3), 537–543.
- Environment Canada and U.S. Environmental Protection Agency (EC-USEPA), 2014. State of the Great Lakes 2011. Cat No. En161-3/1-2011E-PDF. EPA 950-R-13-002. Available at <http://binational.net>.
- Erickson, J.E., Cisar, J.L., Snyder, G.H., Volin, J.C., 2005. Phosphorus and potassium leaching under contrasting residential landscape models established on a sandy soil. *Crop Sci.* 45 (2), 546–552.
- Euliss, N.H., Smith, L.M., Wilcox, D.A., Browne, B.A., 2008. Linking ecosystem processes with wetland management goals: charting a course for a sustainable future. *Wetlands* 28 (3), 553–562.
- Fang, X., Stefan, H.G., 2009. Simulations of climate effects on water temperature, dissolved oxygen, and ice and snow covers in lakes of the contiguous United States under past and future climate scenarios. *Limnol. Oceanogr.* 54 (6), 2359–2370.
- Ficke, A.D., Myrick, C.A., Hansen, L.J., 2007. Potential impacts of global climate change on freshwater fisheries. *Rev. Fish Biol. Fish.* 17, 581–613.
- Finlay, K., Patoine, A., Donald, D.B., Bogard, M.J., Leavitt, P.R., 2010. Experimental evidence that pollution with urea can degrade water quality in phosphorus-rich lakes of the Northern Great Plains. *Limnol. Oceanogr.* 55 (3), 1213–1230.
- Gächter, R., Müller, B., 2003. Why the phosphorus retention of lakes does not necessarily depend on the oxygen supply to their sediment surface. *Limnol. Oceanogr.* 48 (2), 929–933.
- Gain, S.W., 1996. The Effects of Flow-Path Modifications on Urban Water-Quality Constituent Retention in Urban Stormwater Detention Pond and Wetland System. Florida Department of Transportation, Orlando, FL 49 pp.
- Galloway, H.M., Griffith, D.R., Mannering, J.V., 1981. Adaptability of various tillage-planting systems to Indiana soils. *Coop. Ext. Serv. Bulletin AY210*. Purdue University, West Lafayette, IN.
- Gibb, A., Kelly, H., Schueler, T., Horner, R., Simmler, J., Knutson, J., 1999. Best Management Practices Guide for Stormwater. Greater Vancouver Sewerage and Drainage District, Durnaby, BC. 240 pp. Available at: [http://www.gvrd.bc.ca/sewerage/management\\_guide.htm](http://www.gvrd.bc.ca/sewerage/management_guide.htm).
- Gitau, M.W., 2013. LMAV BMP Tool II. In: <http://www.box.net/shared/xuvt811rqa>.
- Gomberg, A., 2007. Sewage Overflow Billions of Gallons of Sewage Contaminate Lake Erie. Environment Ohio and Environmental Ohio Research and Policy Center, Columbus, OH 18 pp.
- Graham, J.M., Auer, M.T., Canale, R.P., Hoffman, J.P., 1982. Ecological studies and mathematical modeling of *Cladophora* in Lake Huron 4: photosynthesis and respiration as functions of light and temperature. *J. Great Lakes Res.* 8 (1), 100–111.
- Gronewald, A.D., Stow, C.A., 2014. Unprecedented seasonal water level dynamics on one of the Earth's largest lakes. *Bull. Am. Meteorol. Soc.* 95 (1), 15–17.
- Gubelit, Y.I., Berezina, N.A., 2010. The causes and consequences of algal blooms: the *Cladophora glomerata* bloom in the Neva estuary (eastern Baltic Sea). *Mar. Poll. Bull.* 61 (4–6), 183–188.
- Ha, J.H., Hidaka, T., Tsuno, H., 2008. Quantification of toxic *Microcystis* and evaluation of its dominance ratio in blooms using real-time PCR. *Environ. Sci. Technol.* 43 (3), 812–818.
- Han, H., Allan, J.D., Bosch, N.S., 2012. Historical pattern of phosphorus loading to the Lake Erie watersheds. *J. Great Lakes Res.* 38 (2), 289–298.
- Harke, M.J., Davis, T.W., Watson, S.B., Gobler, C.J., 2015. Nutrient-controlled niche differentiation of western Lake Erie cyanobacterial populations revealed via metatranscriptomic surveys. *Environ. Sci. Technol.*, <http://dx.doi.org/10.1021/acs.est.5b03931>.
- Harper, H.H., Herr, J.L., 1993. Treatment Efficiencies of Detention With Filtration Systems. Environmental Research and Design, Inc., Orlando, FL.
- Hartig, J.H., Zarull, M.A., Ciborowski, J.J.H., Gannon, J.E., Wilke, E., Norwood, G., Vincent, A. (Eds.), 2007. State of the Strait: Status and Trends of Key Indicators. Great Lakes Institute for Environmental Research, Occasional Publication No. 5, University of Windsor, Ontario, Canada, pp. 1715–3980.
- Hasler, A.D., 1969. Cultural eutrophication is reversible. *J. Biosci.* 19, 425–431.
- Hathaway, A.M., Hunt, W.F., Jennings, G.D., 2008. A field study of green roof hydrologic and water quality performance. *Trans. Am. Soc. Agric. Biol. Eng.* 51, 37–44.
- Hawley, N., Johengen, T.H., Rao, Y.R., Ruberg, S.A., Beletsky, D., Ludsin, S.A., Eadie, B.J., Schwab, D.J., Croley, T.E., Brandt, S.B., 2006. Lake Erie hypoxia prompts Canada-US study. *EOS, Trans. Am. Geophys. Union* 87 (32), 313–319.
- Hayhoe, K., VanDorn, J., Croley, T., Schlegal, N., Wuebbles, D., 2010. Regional climate change projections for Chicago and the US Great Lakes. *J. Great Lakes Res.* 36 (S2), 7–21.
- Healey, F.P., Hendzel, L.L., 1980. Physiological indicators of nutrient deficiency in lake phytoplankton. *Can. J. Fish. Aquat. Sci.* 37, 442–453.
- Heath, R.T., 1992. Nutrient dynamics in Great Lakes coastal wetlands – future directions. *J. Great Lakes Res.* 18 (4), 590–602.
- Hecky, R.E., Smith, R.E.H., Barton, D.R., Guildford, S.J., Taylor, W.D., Charlton, M.N., Howell, E.T., 2004. The nearshore phosphorus shunt: a consequence of ecosystem engineering by dreissenids in the Laurentian Great Lakes. *Can. J. Fish. Aquat. Sci.* 61 (7), 1285–1293.
- Herdendorf, C.E., 1992. Lake Erie coastal wetlands: an overview. *J. Great Lakes Res.* 18 (4), 533–551.
- Higgins, S.N., Howell, E.T., Hecky, R.E., Guildford, S.J., Smith, R.E.H., 2005. The wall of green: the status of *Cladophora glomerata* on the northern shores of Lake Erie's eastern basin, 1995–2002. *J. Great Lakes Res.* 31 (4), 547–563.
- Higgins, S.N., Malkin, S.Y., Howell, E.T., Guildford, S.J., Campbell, L., Liriart-Baer, V., Hecky, R.E., 2008. An ecological review of *Cladophora glomerata* (Chlorophyta) in the Laurentian Great Lakes. *J. Phycol.* 44 (4), 839–854.
- Higgins, S.N., Pennuto, C.M., Howell, E.T., Lewis, T.W., Makarewicz, J., 2012. Urban influences on *Cladophora* blooms in Lake Ontario. *J. Great Lakes Res.* 38 (S4), 116–123.
- Hoffmann, C.C., Heiberg, L., Audet, J., Schonfeldt, B., Fuglsang, A., Kronvang, B., Ovesen, N.B., Kjaergaard, C., Hansen, H.C.B., Jensen, H.S., 2012. Low phosphorus release but high nitrogen removal in two restored riparian wetlands inundated with agricultural drainage water. *Ecol. Eng.* 46, 75–87.
- Hogan, D.M., Walbridge, M.R., 2007. Best management practices for nutrient and sediment retention in urban stormwater runoff. *J. Environ. Qual.* 36 (2), 386–395.
- Hogland, W., Niemczynowice, J., Wahlan, T., 1987. The unit superstructure during the construction period. *Sci. Total Environ.* 59, 411–424.
- Horner, R.R., Horner, C.R., 1995. Design, construction, and evaluation of a sand filter storm water treatment system. Part II. Performance monitoring. Report to Alaska Marine Lines, Seattle, WA.
- Horner, R.R., Skupien, J.J., Livingston, E.H., Shaver, H.E., 1994. Fundamentals of Urban Runoff Management: Technical and Institutional Issues. Terrence Institute and United States Environmental Protection Agency, Washington, DC, USA.
- Horst, G.P., Sarnelle, O., White, J.D., Hamilton, S.K., Kaul, R.B., Bressie, J.D., 2014. Nitrogen availability increases the toxin quota of a harmful cyanobacterium, *Microcystis aeruginosa*. *Water Res.* 54, 188–198.
- Huang, A., Rao, Y.R., Zhang, W., 2012. On recent trends in atmosphere and lake variables in Lake Ontario. *J. Climate* 25, 5807–5816, <http://dx.doi.org/10.1175/JCLI-D-11-00495.1>.
- Hudon, C., Cattaneo, A., Tourville Poirier, A.-M., Brodeur, P., Dumont, P., Mailhot, Y., Amyot, J.-P., Despatie, S.-P., de Lafontaine, Y., 2012. Oligotrophication from wetland eutrophication alters the riverine trophic network and carrying capacity for fish. *Aquat. Sci.* 74 (3), 495–511.
- Hummel, S., 2014. Sediment resuspension of phosphorus flux to Lake Erie. (B.A. thesis) Case Western Reserve University, Cleveland, OH 71 pp.
- International Joint Commission (IJC), 2014. A Balanced Diet for Lake Erie: Reducing Phosphorus Loadings and Harmful Algal Blooms. Report of the Lake Erie Ecosystem Priority. International Joint Commission 96 pp.



- International Upper Great Lakes Study Board (IUGLS), 2009. Impacts on Upper Great Lakes Water Levels: St. Clair Rive, Final Report to the International Joint Commission. Available from: [http://www.iugls.org/Final\\_Reports](http://www.iugls.org/Final_Reports) (accessed 20.05.13).
- James, W.F., 2012. Estimation of internal phosphorus loading contributions to the Lake of the Woods, Minnesota. Report. Minnesota Pollution Control Agency. 42 pp. Available from: <http://www.lakeofthewoodsswcd.org/projects/lowtmdl/intpload.pdf>.
- Jeppesen, E., Søndergaard, M., Meerhoff, M., Lauridsen, T.L., Jensen, J.P., 2007. Shallow lake restoration by nutrient loading reduction – some recent findings and challenges ahead. *Hydrobiologia* 584, 239–252.
- Jiang, L.P., Fang, X., Stefan, H.G., Jacobson, P.C., Pereira, D.L., 2012. Oxythermal habitat parameters and identifying cisco refuge lakes in Minnesota under future climate scenarios using variable benchmark periods. *Ecol. Model.* 232, 14–27.
- Jiang, L., Xia, M., Ludsin, S.A., Rutherford, E.S., Mason, D.M., Jarrin, J.M., Pangle, K.L., 2015. Biophysical modeling assessment of the drivers for plankton dynamics in dreissenid-colonized western Lake Erie. *Ecol. Model.* 308, 18–33.
- Jiao, P.J., Xu, D., Wang, S.L., Zhang, T.Q., 2011. Phosphorus loss by surface runoff from agricultural field plots with different cropping systems. *Nutr. Cycl. Agroecosyst.* 90 (1), 23–32.
- Johnson, P.D., Pitt, R., Durran, S.R., Urrutia, M., Clark, S., 2003. *Metals Removal Technologies for Urban Stormwater*. Water Environment Research Foundation. WERF 97-IRM-2, Alexandria, VA ISBN 1-94339682-3. 701 pp.
- Kane, D.D., Gordon, S.I., Munawar, M., Charlton, M.N., Culver, D.A., 2009. The Planktonic Index of Biotic Integrity (P-IBI): an approach for assessing lake ecosystem health. *Ecol. Indic.* 9, 1234–1247.
- Kane, D.D., Conroy, J.D., Peter Richards, R., Baker, D.B., Culver, D.A., 2014. Reutrophication of Lake Erie: correlations between tributary nutrient loads and phytoplankton biomass. *J. Great Lakes Res.* 40 (3), 496–501.
- Katsev, S., Tsandev, I., L'Heureux, I., Rancourt, D.G., 2006. Factors controlling long-term phosphorus efflux from lake sediments: exploratory reactive-transport modeling. *Chem. Geol.* 234 (1–2), 127–147.
- Keough, J.R., Thompson, T.A., Guntenspergen, G.R., Wilcox, D.A., 1999. Hydrogeomorphic factors and ecosystem responses in coastal wetlands of the Great Lakes. *Wetlands* 19 (4), 821–834.
- Kim, D.-K., Zhang, W., Rao, Y.R., Watson, S., Mugalingam, S., Labenchi, T., Dittrich, M., Morley, A., Arhonditsis, G.B., 2013. Improving the representation of internal nutrient recycling with phosphorus mass balance models: a case study in the Bay of Quinte, Ontario, Canada. *Ecol. Model.* 256, 53–68.
- Kim, D.-K., Zhang, W., Watson, S., Arhonditsis, G.B., 2014. A commentary on the modelling of the causal linkages among nutrient loading, harmful algal blooms, and hypoxia patterns in Lake Erie. *J. Great Lakes Res.* 4 (3), 117–129.
- King, K.W., Fausey, N.R., Williams, M.R., 2014. Effect of subsurface drainage on streamflow in an agricultural headwater watershed. *J. Hydrol.* 519 (PA), 438–445.
- Kirby, M.K., Dunford, W.E., 1982. Attached algae of the Lake Erie shoreline near Nanticoke Generating Station. *J. Great Lakes Res.* 7 (3), 249–257.
- Khan, Z., Thrush, C., Cohen, P., Kulzer, L., Franklin, R., Field, D., Koon, J., Horner, R., 1992. *Biofiltration Swale Performance, Recommendations, and Design Considerations*. Publication 657, Municipality of Metropolitan Seattle, Water Pollution Control Department, Seattle, WA 16 pp.
- Kling, G.W., Hayhoe, K., Johnson, L.B., Magnuson, J.J., Polasky, S., Robinson, S.K., Shuter, B.J., Wander, M.M., Wuebbles, D.J., Zak, D.R., Lindroth, R.L., Moser, S.C., Wilson, M.L., 2003. *Confronting Climate Change in the Great Lakes Region: Impacts on our Communities and Ecosystems*. Union of Concerned Scientists, Cambridge, Massachusetts, and Ecological Society of America, Washington, DC., In: [www.ucusa.org/greatlakes](http://www.ucusa.org/greatlakes).
- Koslow, M., Lillard, E., Benka, V., 2013. Taken by Storm: How Heavy Rain is Worsening Algal Blooms in Lake Erie With a Focus on the Maumee River in Ohio. National Wildlife Federation, Ann Arbor, MI, pp. 23.
- Krieger, K.A., 2003. Effectiveness of a coastal wetland in reducing pollution of a Laurentian Great Lake: hydrology, sediment, and nutrients. *Wetlands* 23 (4), 778–791.
- Krieger, A.K., Bur, M.T., Ciborowski, J.J.H., Barton, D.B., Schloesser, D.W., 2007. Distribution and abundance of burrowing mayflies (*Hexagenia* spp.) in Lake Erie, 1997–2005. *J. Great Lakes Res.* 33 (S1), 20–33.
- Kroger, R., Moore, M.T., Farris, J.L., Gopalan, M., 2011. Evidence for the use of low-grade weirs in drainage ditches to improve nutrient reductions from agriculture. *Water Air Soil Pollut.* 221 (1–4), 223–234.
- Kunkel, K.E., Andsager, K., Easterling, D.R., 1999. Long-term trends in extreme precipitation events over the conterminous United States and Canada. *J. Climate* 12, 2515–2527.
- Kutovaya, O.A., Watson, S.B., 2014. Development and application of a molecular assay to detect and monitor geosmin-producing cyanobacteria and actinomycetes in the Great Lakes. *J. Great Lakes Res.* 40 (2), 404–414.
- Kutovaya, O.A., McKay, R.M., Beall, B., Wilhelm, S.W., Kane, D., Chaffin, J., Bridgeman, T., Bullerjahn, G.S., 2012. Evidence against fluvial seeding of recurrent toxic blooms of *Microcystis* spp. in Lake Erie's western basin. *Harmful Algae* 15, 71–77.
- Kutovaya, O.A., McKay, R.M.L., Bullerjahn, G.S., 2013. Detection and expression of genes for phosphorus metabolism in picocyanobacteria from the Laurentian Great Lakes. *J. Great Lakes Res.* 39, 612–621.
- Lajeunesse, A., Segura, P.A., Gélinas, M., Hudon, C., Thomas, K., Quilliam, M.A., Gagnon, G., 2012. Detection and confirmation of saxitoxin analogues in freshwater benthic *Lyngbya wollei* algae collected in the St. Lawrence River (Canada) by liquid chromatography–tandem mass spectrometry. *J. Chromatogr. A* 1219, 93–103.
- Lehman, J., Bell, D., McDonald, K., 2009. Reduced river phosphorus following implementation of a lawn fertilizer ordinance. *Lake Res. Manage.* 25 (3), 307–312.
- LEIA, 2012. Sustaining Healthy Waters for Lake Erie's Economy. Lake Erie Improvement Association Strategic Plan for Lake Erie Partners. Unpublished report. 57 pp. [www.lakeerieimprovement.org](http://www.lakeerieimprovement.org).
- Leisenring, M., Clary, J., Stephenson, J., Hobson, P., 2010. International Stormwater Best Management Practices (BMP) Database Pollutant Category Summary: Nutrients. Geosyntec Consultants, Wright Water Engineers, International Stormwater BMP Database. In: <http://bmpdatabase.org/BMPPerformance.htm>.
- Lenhart, H.A., Hunt, W.F., 2011. Evaluating four storm-water performance metrics with a North Carolina coastal plain storm-water wetland. *J. Environ. Eng.* 137 (2), 155–162.
- Leon, L.F., Smith, R.E.H., Hipsey, M.R., Bocaniov, S.A., Higgins, S.N., Hecky, R.E., Antenucci, J.P., Imberger, J.A., Guildford, S.J., 2011. Application of a 3D hydrodynamic–biological model for seasonal and spatial dynamics of water quality and phytoplankton in Lake Erie. *J. Great Lakes Res.* 37 (1), 41–53.
- Loh, P.S., Molot, L.A., Nürnberg, G.K., Watson, S.B., Ginn, B., 2013. Evaluating relationships between sediment chemistry and anoxic phosphorus and iron release across three different water bodies. *Inland Waters* 3, 105–118.
- Louati, I., Pascual, N., Debroas, D., Bernard, C., Humber, J.-H., Luloup, J., 2015. Structural diversity of bacterial communities associated with bloom-forming freshwater cyanobacteria differs according to the cyanobacterial genus. *PLOS ONE* 10 (11), e0140614.
- Ludsin, S.A., Kershner, M.W., Blocksom, K.A., Knight, R.L., Stein, R.A., 2001. Life after death in Lake Erie: nutrient controls drive fish species richness, rehabilitation. *Ecol. Appl.* 11, 731–746.
- Ludsin, S.A., Zhang, X.S., Brandt, S.B., Roman, M.R., Boicourt, W.C., Mason, D.M., Costantini, M., 2009. Hypoxia-avoidance by planktivorous fish in Chesapeake Bay: implications for food web interactions and fish recruitment. *J. Exp. Mar. Biol. Ecol.* 381, S121–S131.
- MacKay, M., Seglenicks, F., 2013. On the simulation of Laurentian Great lakes water levels under projections of global climate change. *Climat. Change* 117, 55–67.
- Makarewicz, J.C., Bertram, P., 1991. Restoration of the Lake Erie ecosystem. *Bioscience* 41 (4), 216–223.
- Malkin, S.Y., Guildford, S.J., Hecky, R.E., 2008. Modeling the growth response of *Cladophora* in a Laurentian Great Lake to the exotic invader *Dreissena* and to lake warming. *Limnol. Oceanogr.* 53 (3), 1111–1124.
- Maniquiz, M.C., Lee, S.Y., Kim, L.H., 2010. Multiple linear regression models of urban runoff pollutant load and EMC considering rainfall variables. *J. Environ. Sci.* 22 (6), 946–952.
- Mantai, K.E., Garwood, P.E., Peglowski, L.E., 1982. Environmental factors controlling physiological changes in *Cladophora* in Lake Erie. *J. Great Lakes Res.* 8 (1), 61–65.
- Martin, E.H., Smoot, J.L., 1986. Constituent load changes in urban stormwater runoff routed through a detention pond-wetlands system in central Florida: U.S. Geological Survey Water-Resources Investigations Report 85-4310 78 pp.
- Martin, G., 2010. Nutrient sources for excessive growth of benthic algae in Lake Ontario as inferred by the distribution of SRP. (M.Sc. thesis) University of Waterloo, Waterloo, ON 91 pp.
- Matisoff, G., Carson, M., 2014. Sediment resuspension in the Lake Erie nearshore. *J. Great Lakes Res.* 40 (3), 532–540.
- Matisoff, G., Kaltenberg, E.M., Steely, R.L., Hummel, S.K., Seo, J., Gibbons, K., Bridgeman, T.B., Seo, Y., Behbahani, M., James, W.F., Johnson, L.T., Doan, P., Dittrich, M., Evans, M.A., Chaffin, J.D., 2016. Internal loading of phosphorus in western Lake Erie. *J. Great Lakes Res.*
- Maynard, L., Wilcox, D., 1997. *Coastal Wetlands*. State of the Lakes Ecosystem Conference, 1996 Background Paper 103 pp.
- McCulloch, J., Gudimov, A., Arhonditsis, G., Chesnyuk, A., Dittrich, M., 2013. Dynamics of P-binding forms in sediments of a mesotrophic hard-water lake: insights from non-steady state reactive-transport modeling, sensitivity and identifiability analysis. *Chem. Geol.* 354, 216–232.
- McDowell, R.W., Sharpley, A.N., Bourke, W., 2008. Treatment of drainage water with industrial by-products to prevent phosphorus loss from tile-drained land. *J. Environ. Qual.* 37 (4), 1575–1582.
- McElmurry, S.P., Confesor, R., Richards, R.P., 2013. Reducing phosphorus loads to Lake Erie: best management practices literature review, Draft review paper 3, for International Joint Commission, Taking Action on Lake Erie. 52 pp. <http://www.ijc.org/files/tiny/mce/uploaded/BMP%20Review-FINAL.pdf>.
- McNett, J.K., Hunt, W.F., Davis, A.P., 2011. Influent pollutant concentrations as predictors of effluent pollutant concentrations for mid-atlantic bioretention. *J. Environ. Eng.* 137 (9), 790–799.
- Meals, D.W., Budd, L.F., 1998. Lake Champlain Basin nonpoint source phosphorus assessment. *J. Am. Water Res. Assoc.* 34 (2), 251–265.
- Meile, C., Van Cappellen, P., 2003. Global estimates of enhanced solute transport in marine sediments. *Limnol. Oceanogr.* 48 (2), 777–786.
- Merriman, K.R., Gitau, M.W., Chaubey, I., 2009. Tool for estimating Best Management Practice effectiveness in Arkansas. *Appl. Eng. Agric.* 25 (2), 199–213.
- Michalak, A.M., Anderson, E.J., Beletsky, D., Boland, S., Bosch, N.S., Bridgeman, T.B., Chaffin, J.D., Cho, K., Confesor, R., Daloglu, I., DePinto, J.V., Evans, M.A., Fahnenstiel, G.L., He, L., Ho, J.C., Jenkins, L., Johengen, T.H., Kuo, K.C., LaPorte, E., Steiner, A.L., Verhamme, E., Wright, D.M., Zagorski, M.A., 2013. Record-setting algal blooms in Lake Erie caused by agricultural and meteorological trends consistent with expected future conditions. *Proc. Natl. Acad. Sci. U. S. A.* 110, 6448–6452.
- Millie, D., Fahnenstiel, G., Dyble Bressie, J., Pigg, R., Rediske, R., Klarer, D., Tester, P., Litaker, R.W., 2009. Late-summer phytoplankton in western Lake Erie



- (Laurentian Great Lakes): bloom distributions, toxicity, and environmental influences. *Aquat. Ecol.* 43, 915–934.
- Miller, T.P., Peterson, J.R., Lenhart, C.F., Nomura, Y., 2012. The Agricultural BMP Handbook for Minnesota. Minnesota Department of Agriculture.
- Millner, G.C., Sweeney, R.A., Ray-Frederick, V., 1982. Biomass and distribution of *Cladophora glomerata* in relation to some physical–chemical variables at two sites in Lake Erie. *J. Great Lakes Res.* 8 (1), 35–41.
- Mioni, C.E., Howard, A.M., DeBruyn, J.M., Bright, N.G., Twiss, M.R., Applegate, B.M., Wilhelm, S.W., 2003. Characterization and field trials of a bioluminescent bacterial reporter of iron bioavailability. *Mar. Chem.* 83, 31–46.
- Mitsch, W.J., 1992. Combining ecosystem and landscape approaches to Great Lakes wetlands. *J. Great Lakes Res.* 18 (4), 552–570.
- Mitsch, W.J., Reeder, B.C., 1992. Nutrient and hydrologic budgets of a Great Lakes coastal freshwater wetland during a drought year. *Wetl. Ecol. Manage.* 1 (4), 211–222.
- Mitsch, W.J., Gosselink, J.G., 2000. *Wetlands*, 3rd ed. John Wiley & Sons, Inc., New York, NY 920 pp.
- Mitsch, W.J., Wang, N.M., 2000. Large-scale coastal wetland restoration on the Laurentian Great Lakes: determining the potential for water quality improvement. *Ecol. Eng.* 15 (3–4), 267–282.
- Mitsch, W.J., Reeder, B.C., Klarer, D.M., 1989. The role of wetlands in the control of nutrients with a case study of western Lake Erie. In: Mitsch, W.J., Jorgensen, S.E. (Eds.), *Ecological Engineering: An Introduction to Ecotechnology*. John Wiley & Sons Inc., New York, NY, pp. 129–158.
- Molot, L.A., Watson, S.B., Creed, I.F., Trick, C.G., McCabe, S.K., Verschoor, M.J., Sorichetti, R.J., Powe, C., Venkiteswaran, J.J., Schiff, S.L., 2014. A novel model for cyanobacteria bloom formation: the critical role of anoxia and ferrous iron. *Freshw. Biol.* 59 (6), 1323–1340.
- Moosmann, L., Gächter, R., Müller, B., Wüest, A., 2006. Is phosphorus retention in autochthonous lake sediments controlled by oxygen or phosphorus? *Limnol. Oceanogr.* 51 (1), 763–771.
- Morrice, J.A., Kelly, J.R., Trebitz, A.S., Cotter, A.M., Knuth, M.L., 2004. Temporal dynamics of nutrients (N and P) and hydrology in a Lake Superior coastal wetland. *J. Great Lakes Res.* 30 (S1), 82–96.
- Mortimer, C.H., 1941. The exchange of dissolved substances between mud and water in lakes. *J. Ecol.* 29, 280–329.
- Mortsch, L., Ingram, J., Hebb, A., Doka, S. (Eds.), 2006. *Great Lakes Coastal Wetland Communities: Vulnerability to Climate Change and Response to Adaptation Strategies*. Final report submitted to the Climate Change Impacts and Adaptation Program, Natural Resources Canada. Environment Canada and the Department of Fisheries and Oceans, Toronto, Ontario. 251 pp. + appendices. [http://environment.uwaterloo.ca/research/aird/aird\\_pub/Great\\_Lakes\\_Coastal\\_Wetlands\\_Report\\_2006.pdf](http://environment.uwaterloo.ca/research/aird/aird_pub/Great_Lakes_Coastal_Wetlands_Report_2006.pdf).
- Mostaghimi, S., Dillaha, T.A., Shanholtz, V.O., 1988. Influence of tillage systems and residue levels on runoff, sediment, and phosphorus losses. *Trans. ASAE* 31 (1), 128–132.
- Mueller, B., Bryant, L.D., Matzinger, A., Wue, A., 2012. Hypolimnetic oxygen depletion in eutrophic lakes. *Environ. Sci. Technol.* 46, 9964–9971.
- Mullen, R., Diedrick, K., Henry, D., 2009. Best Management Practices for Mitigating Phosphorus Loss from Agricultural Soils. Fact sheet, Agriculture and Natural Resources. AGF-509-09. The Ohio State University Extension.
- Munawar, M., Munawar, I.F., Fitzpatrick, M., Niblock, H., Bowen, K., Lorimer, J., 2008. An intensive assessment of planktonic communities in the Canadian waters of Lake Erie, 1998. In: Munawar, M., Heath, R. (Eds.), *Checking the Pulse of Lake Schweizerbart*, Champaign, IL, pp. 297–346.
- MWCOG, 1983. Urban Runoff in the Washington Metropolitan Area: Final Report, Urban Runoff Project. EPA Nationwide Urban Runoff Program, Washington, DC, Metropolitan Washington Council of Governments. In: [http://www3.epa.gov/npdes/pubs/sw\\_nurp\\_vol\\_1\\_finalreport.pdf](http://www3.epa.gov/npdes/pubs/sw_nurp_vol_1_finalreport.pdf).
- Nicholls, K.H., 1999. Effects of temperature and other factors on summer phosphorus in the Inner Bay of Quinte, Lake Ontario: implications for climate warming. *J. Great Lakes Res.* 25 (2), 250–262.
- Nicholls, K.H., 2011. Detection of regime shifts in multi-species communities: the Bay of Quinte phytoplankton example. *Methods Ecol. Evol.* 2 (4), 416–426.
- Nistor, A.P., Lowenberg-DeBoer, J., 2007. The profitability factor of controlled drainage implementation. *J. Soil Water Conserv.* 62 (6), 156a.
- National Oceanic and Atmospheric Administration (NOAA), 2015. Tides and Currents, Water Levels, Station 9063063, Cleveland, OH. <http://tidesandcurrents.noaa.gov/stations.html?type=Water+Levels> (accessed 21.12.15).
- Noffke, A., Hensen, C., Sommer, S., Scholz, F., Bohle, L., Mosch, T., Graco, M., Wallmann, K., 2012. Benthic iron and phosphorus fluxes across the Peruvian oxygen minimum zone. *Limnol. Oceanogr.* 57, <http://dx.doi.org/10.4319/lo.2012.57.3.0851>.
- North, R.L., Guildford, S.J., Smith, R.E.H., Havens, S.M., Twiss, M.R., 2007. Evidence for phosphorus, nitrogen, and iron co-limitation of phytoplankton communities in Lake Erie. *Limnol. Oceanogr.* 52 (1), 315–328.
- North, R.L., Johansson, J., Vandergucht, D., Doig, L.E., Liber, K., Lindenschmidt, K.-E., Baulch, H., Hudson, J.J., 2015. Evidence for internal phosphorus loading in a large prairie reservoir (Lake Diefenbaker, Saskatchewan). *J. Great Lakes Res.* 41 (Suppl. 2), 91–99.
- Nürnberg, G.K., LaZerte, B.D., Loh, P.S., Molot, L.A., 2013. Quantification of internal phosphorus load in large, partially polymictic and mesotrophic Lake Simcoe, Ontario. *J. Great Lakes Res.* 39 (2), 271–279.
- Orihel, D.M., Schindler, D.W., Ballard, N.C., Graham, M.D., O'Connell, D.W., Wilson, L.R., Vinebrooke, R.D., 2015. The “nutrient pump”: iron-poor sediments fuel low nitrogen-to-phosphorus ratios and cyanobacterial blooms in polymictic lakes. *Limnol. Oceanogr.* 60 (3), 856–871.
- Obenour, D.R., Gronewold, A.D., Stow, C.A., Scavia, D., 2014. Using a Bayesian hierarchical model with a gamma error distribution to improve Lake Erie cyanobacteria bloom forecasts. *Water Resour. Res.* 50, 7847–7860.
- Ohio EPA, 2010. Ohio Lake Erie Phosphorus Task Force Final Report. Division of Surface Water, Ohio Environmental Protection Agency, Columbus, OH. In: [http://epa.ohio.gov/portals/35/lakeerie/ptaskforce/Task\\_Force\\_Final\\_Report\\_April\\_2010.pdf](http://epa.ohio.gov/portals/35/lakeerie/ptaskforce/Task_Force_Final_Report_April_2010.pdf).
- Ozersky, T., Malkin, S.Y., Barton, D.R., Hecky, R.E., 2009. Dreissenid phosphorus excretion can sustain *C. glomerata* growth along a portion of Lake Ontario shoreline. *J. Great Lakes Res.* 35 (3), 321–328.
- Paerl, H.W., Otten, T.G., 2013. Harmful cyanobacterial blooms: causes, consequences and controls. *Microb. Ecol.*, <http://dx.doi.org/10.1007/s00248-012-0159-y>.
- Paerl, H.W., Valdes, L.M., Joyner, A.R., Peierls, B.L., Buzzelli, C.P., Piehler, M.F., Riggs, S.R., Christian, R.R., Ramus, J.S., Clesceri, E.J., Eby, L.A., Crowder, L.W., Luettich, R.A., 2006. Ecological response to hurricane events in the Pamlico Sound System, NC, and implications for assessment and management in a regime of increased frequency. *Estuar. Coast* 29 (6), 1033–1045.
- Pandit, S.N., Zhao, Y., Ciborowski, J.J.H., Gorman, A.M., Knight, C.T., 2013. Suitable habitat model for walleye (*Sander vitreus*) in Lake Erie: implications for inter-jurisdictional harvest quota allocations. *J. Great Lakes Res.* 39 (4), 591–601.
- Perri, K.A., Sullivan, J.M., Boyer, G.L., 2015. Harmful algal blooms in Sodus Bay, Lake Ontario: a comparison of nutrients, marina presence, and cyanobacterial toxins. *J. Great Lakes Res.* 41 (2), 326–337.
- Pieters, A.J., 1902. The plants of western Lake Erie, with observations on their distribution. *Bull. US Fish. Comm.* 21, 57–79.
- Pinowska, A., Stevenson, R.J., Sickman, J.O., Albertin, A., Anderson, M., 2007. Integrated Interpretation of Survey for Determining Nutrient Thresholds for Macroalgae in Florida Springs. Laboratory Experiments and Disturbance Study. Florida Department of Environmental Protection, Tallahassee, FL 97 pp.
- Pörtner, H.O., 2010. Oxygen- and capacity-limitation of thermal tolerance: a matrix for integrating climate-related stressor effects in marine ecosystems. *J. Exp. Biol.* 213, 881–893.
- Pothoven, S.A., Vanderploeg, H.A., Ludsin, S.A., Höök, T.O., Brandt, S.B., 2009. Feeding ecology of rainbow smelt and emerald shiner in Lake Erie's central basin. *J. Great Lakes Res.* 35, 190–198.
- Powell, G.E., Ward, A.D., Mecklenburg, D.E., Jayakaran, A.D., 2007. Two-stage channel systems: Part 1, a practical approach for sizing agricultural ditches. *J. Soil Water Conserv.* 62 (4), 277–286.
- Qin, B., Li, W., Zhu, G., Zhang, Y., Wu, T., Gao, G., 2015. Cyanobacterial bloom management through integrated monitoring and forecasting in large shallow eutrophic Lake Taihu (China). *J. Hazard. Mater.* 287, 356–363.
- Quinn, F.H., 2002. Secular changes in Great Lakes water level seasonal cycles. *J. Great Lakes Res.* 28 (3), 451–465.
- Rao, Y.R., Hawley, N., Charlton, M.N., Schertzer, W.M., 2008. Physical processes and hypoxia in the central basin of Lake Erie. *Limnol. Oceanogr.* 52 (5), 2007–2020.
- Rao, Y.R., Howell, T., Watson, S.B., Abernethy, S., 2014. On hypoxia and fish kills along the north shore of Lake Erie. *J. Great Lakes Res.* 40 (2), 187–191.
- Reeves, D.W., 1994. Cover crops and rotations. In: Harfield, J.L., Stewart, B.A. (Eds.), *Crops Residue Management. Advances in Soil Science*. Lewis Publishers, Boca Raton, FL, pp. 125–172.
- Richards, C.E., Munster, C.L., Vietor, D.M., Arnold, J.G., White, R., 2008. Assessment of a turfgrass sod best management practice on water quality in a suburban watershed. *J. Environ. Manage.* 86 (1), 229–245.
- Rinta-Kanto, J.M., Wilhelm, S.W., 2006. Diversity of microcystin-producing cyanobacteria in spatially isolated regions of Lake Erie. *Appl. Environ. Microbiol.* 72, 5083–5085.
- Rinta-Kanto, J.M., Ouellette, A.J.A., Boyer, G.L., Twiss, M.R., Bridgeman, T.B., Wilhelm, S.W., 2005. Quantification of toxic *Microcystis* spp. during the 2003 and 2004 blooms in western Lake Erie using quantitative real-time PCR. *Environ. Sci. Technol.* 39, 4198–4205.
- Rinta-Kanto, J.M., Konopko, E.A., DeBruyn, J.M., Bourbonniere, R.A., Boyer, G.L., Wilhelm, S.W., 2009a. Lake Erie *Microcystis*: relationship between microcystin production, dynamics of genotypes and environmental parameters in a large lake. *Harmful Algae* 8, 665–673.
- Rinta-Kanto, J.M., Saxton, M.A., DeBruyn, J.M., Juliette, L.S., Marvin, C.H., Krieger, K.A., Sayler, G.S., Boyer, G.L., Wilhelm, S.W., 2009b. The diversity and distribution of toxigenic *Microcystis* spp. in present day and archived pelagic and sediment samples from Lake Erie. *Harmful Algae* 8, 385–394.
- Roberts, J.J., Höök, T.O., Ludsin, S.A., Pothoven, S.A., Vanderploeg, H.A., Brandt, S.B., 2009. Effects of hypolimnetic hypoxia on foraging and distributions of Lake Erie yellow perch. *J. Exp. Mar. Biol. Ecol.* 381, S132–S142.
- Roberts, J.J., Greca, P.A., Ludsin, S.A., Pothoven, S.A., Vanderploeg, H.A., Höök, T.O., 2012. Evidence of hypoxic foraging forays by yellow perch (*Perca flavescens*) and potential consequences for prey consumption. *Freshw. Biol.* 57, 922–937.
- Rogers, J.S., Potter, K.W., Hoffman, A.R., Hoopes, J.A., Wu, C.H., Armstrong, D.E., 2009. Hydrologic and water quality functions of a disturbed wetland in an agricultural setting. *J. Am. Water Res. Assoc.* 45 (3), 628–640.
- Rosa, F., Burns, N.M., 1987. Lake Erie central basin oxygen depletion changes from 1929–1980. *J. Great Lakes Res.* 13 (4), 684–696.
- Ross, S., 2006. Molecular phylogeography and species discrimination of freshwater *Cladophora* (Cladophorales, Chlorophyta) in North America. (M.Sc. thesis)-University of Waterloo, Waterloo, ON, Canada 155 pp.
- Roy-Poirier, A., Champagne, P., Filion, Y., 2010. Review of bioretention system research and design: past, present, and future. *J. Environ. Eng.* 136 (9), 878–889.

- Rucinski, D., Scavia, D., DePinto, J., Beletsky, D., 2014. Modeling Lake Erie's hypoxia response to nutrient loads and meteorological variability. *J. Great Lakes Res.* 40 (3), 151–161.
- Saxton, M.A., Arnold, R.J., Bourbonniere, R.A., McKay, R.M., Wilhelm, S.W., 2012. Plasticity of total and intracellular phosphorus quotas in *Microcystis aeruginosa* cultures and Lake Erie algal assemblages. *Front. Microbiol.* 3 (3), 1–8.
- Scavia, D., Allan, J.D., Arend, K.K., Bartell, S., Beletsky, D., Bosch, N.S., Brandt, S.B., Briland, R.D., Daloglu, I., DePinto, J.V., Dolan, D.M., Evans, M.A., Farmer, T.M., Goto, D., Han, H., Höök, T.O., Knight, R., Ludsin, S.A., Mason, D., Michalak, A.M., Peter Richards, R., Roberts, J.J., Rucinski, D.K., Rutherford, E., Schwab, D.J., Sesterhenn, T.M., Zhang, H., Zhou, Y., 2014. Assessing and addressing the re-eutrophication of Lake Erie: central basin hypoxia. *J. Great Lakes Res.* 40 (2), 226–246.
- Schertzer, W.M., Sawchuk, A.M., 1990. Thermal structure of the lower Great Lakes in a warm year: implications for occurrence of hypolimnetic anoxia. *Trans. Am. Fish. Soc.* 119, 195–209.
- Schreiber, J.D., Cullum, R.F., 1998. Tillage effects on surface and groundwater quality in loessial upland soybean watersheds. *Trans. ASAE* 41 (3), 607–614.
- Schueler, T., 1999. Microbes and urban watersheds. *Watershed Prot. Tech.* 3 (1), 551–596.
- Sharpley, A.N., Daniel, T., Gibson, G., Bundy, L., Cabrera, M., Sims, T., Stevens, R., Lemunyon, J., Kleinman, P., Parr, R., 2006. Best Management Practices To Minimize Agricultural Phosphorus Impacts on Water Quality. U.S. Department of Agriculture, Agricultural Research Service. ARS-163 50 pp.
- Shaver, E., Maxted, I., 1994. Construction of wetlands for stormwater treatment. In: Kuo, C.Y. (Ed.), *Proceedings, Symposium on Stormwater Runoff And Quality Management*. Penn State University, University Park, PA, pp. 53–90.
- Shuchman, R.A., Sayers, M.J., Brooks, C.N., 2013. Mapping and monitoring the extent of submerged aquatic vegetation in the Laurentian Great Lakes with multi-scale satellite remote sensing. *J. Great Lakes Res.* 39 (S1), 78–89.
- Slopp, C., Malschaert, J.F.P., Van Raaphorst, W., 1998. The role of adsorption in sediment-water exchange of phosphorus in North Sea continental margin sediments. *Limnol. Oceanogr.* 43 (5), 832–846.
- Smith, L.M., Euliss, N.H., Wilcox, D.A., Brinson, M.M., 2008. Application of a geomorphic and temporal perspective to wetland management in North America. *Wetlands* 28 (3), 563–577.
- Smith, R., Sawyer, D., Bass, B., 2015. Algal Blooms: Estimating Costs to the Lake Erie Basin Economy. Internal Report, Environment Canada. 87 pp. Available from [brad.bass@canada.ca](http://brad.bass@canada.ca).
- Smith, V.H., 1983. Low nitrogen to phosphorus ratios favor dominance by blue green algae in lake phytoplankton. *Science* 221 (4611), 669–671.
- Søndergaard, M., Jensen, J.P., Jeppesen, E., 2003. Role of sediment and internal loading of phosphorus in shallow lakes. *Hydrobiologia* 506 (1–3), 135–145.
- Spears, B.M., Carvalho, L., Perkins, R., Kirika, A., Patterson, D.M., 2007. Sediment phosphorus cycling in a large shallow lake: spatio-temporal variation in phosphorus pools and release. *Hydrobiologia* 584 (1), 37–48.
- Speziale, B.J., Dyck, L.A., 1992. *Lyngbya* infestations: comparative taxonomy of *Lyngbya wollei* comb. nov. (cyanobacteria). *J. Phycol.* 28 (5), 693–706.
- Ståhl-Delbanco, A., Hansson, L.-A., Gyllström, M., 2003. Recruitment of resting stages may induce blooms of *Microcystis* at low n:P ratios. *J. Plankton Res.* 25, 1099–1106.
- Steffen, M.M., Li, Z., Effler, T.C., Hauser, L.J., Boyer, G.L., Wilhelm, S.W., 2012. Comparative metagenomics of toxic freshwater cyanobacteria bloom communities on two continents. *PLoS ONE* 7 (8), e44002, <http://dx.doi.org/10.1371/journal.pone.0044002>.
- Steffen, M.M., Belisle, S., Watson, S.B., Boyer, G.L., Wilhelm, S.W., 2014. Status, causes and controls of cyanobacterial blooms in Lake Erie. *J. Great Lakes Res.* 40 (2), 215–225.
- Steffen, M.M., Belisle, S.B., Watson, S.B., Boyer, G.L., Bourbonniere, R.A., Wilhelm, S.A., 2015. Metatranscriptomic evidence for co-occurring top-down and bottom-up controls on toxic cyanobacterial communities. *Appl. Environ. Microbiol.* 81 (9), 3268–3276.
- Steffensen, D.A., 2008. Economic cost of cyanobacterial blooms. *Cyanobacterial harmful algal blooms: state of the science and research needs. Adv. Exp. Med. Biol.* 619, 855–865.
- Steinman, A.D., Ogdahl, M.E., 2011. Does converting agricultural fields to wetlands retain or release P? *J. North Am. Benth. Soc.* 30 (3), 820–830.
- Steinman, A.D., Ogdahl, M.E., Weinert, M., Thompson, K., Cooper, M.J., Uzarski, D.G., 2012. Water level fluctuation and sediment-water nutrient exchange in Great Lakes coastal wetlands. *J. Great Lakes Res.* 38 (4), 766–775.
- Stewart, T.W., Lowe, R.L., 2008. Benthic algae of Lake Erie (1865–2006): a review of assemblage composition, ecology and causes and consequences of changing abundance. *Ohio J. Sci.* 108, 82–94.
- Stewart, W.S., 1992. *Compost Storm Water Treatment System*. W&H Pacific Consultants, Portland, OR Final Report.
- Stormwater Management, 1994. *Three Year Performance Summary of Stormwater Pollutant and Treatment – 185th Avenue, Hillsboro, Oregon*. Technical Memorandum. Stormwater Management, Portland, OR.
- Stow, C.A., Reckhow, K.H., Qian, S.S., Lamon, E.C., Arhonditsis, G.B., Borsuk, M.E., Seo, D., 2007. Approaches to evaluate water quality model parameter uncertainty for adaptive TMDL implementation. *J. Am. Water Resour. Assoc.* 43, 1499–1507.
- Stow, C.A., Cha, Y., Johnson, L.T., Confesor, R., Richards, R.P., 2015. Long-term and seasonal trend decomposition of Maumee River nutrient inputs to Western Lake Erie. *Environ. Sci. Technol.* 49 (6), 3392–3400.
- Stumpf, R.P., Wynne, T.T., Baker, D.B., Fahnenstiel, G.L., 2012. Interannual variability of cyanobacterial blooms in Lake Erie. *PLoS ONE* 7 (8), e42444, <http://dx.doi.org/10.1371/journal.pone.0042444>.
- Sweeney, R.A., 1993. “Dead” sea of North America?—Lake Erie in the 1960s and 70s. *J. Great Lakes Res.* 19, 198–199.
- Taft, C.E., 1942. Additions to the algae of the west end of Lake Erie. *Ohio J. Sci.* 42, 251–256.
- Taft, C.E., Kischler, W.J., 1973. *Cladophora* as Related to Pollution and Eutrophication in Western Lake Erie, No 332X, 339X. Water Resources Center, Ohio State University, and US Department of the Interior, Columbus, OH 103 pp.
- Tang, X.M., Gao, G., Chao, J.Y., Wang, X.D., Zhu, G.W., Qin, B.Q., 2010. Dynamics of organic-aggregate-associated bacterial communities and related environmental factors in Lake Taihu, a large eutrophic shallow lake in China. *Limnol. Oceanogr.* 55 (2), 469–480.
- Trebitz, A.S., 2006. Characterizing seiche and tide-driven daily water level fluctuations affecting coastal ecosystems of the Great Lakes. *J. Great Lakes Res.* 32 (1), 102–116.
- Twiss, M.R., McKay, R.M.L., Bourbonniere, R.A., Bullerjahn, G.S., Carrick, H.J., Smith, R.E.H., Winter, J.G., D'Souza, N.A., Furey, P.C., Lashaway, A.R., Saxton, M.A., Wilhelm, S.W., 2012. Diatoms abundant in ice-covered Lake Erie: an investigation of offshore winter limnology in Lake Erie over the period 2007 to 2010. *J. Great Lakes Res.* 38 (1), 18–30.
- USDA-NRCS, 2010. National Handbook of Conservation Practices. Conservation Practice Standard Code 412 (Grassed Waterway). USDA Natural Resources Conservation Service, Washington, DC. In: [http://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/technical/cp/nrcs/?cid=nrcs143\\_026849](http://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/technical/cp/nrcs/?cid=nrcs143_026849).
- US EPA, 1993. *Guidance Specifying Management Measures for Sources of Nonpoint Pollution in Coastal Waters*. EPA 840-B-92-002. U.S. Environmental Protection Agency, Washington, DC 822 pp.
- US EPA, 2010. *Guidance for Federal Land Management in the Chesapeake Bay Watershed*. EPA841-R-10-002. EPA, Washington, DC 848 pp.
- Van de Moortel, A., Rousseau, D., Tack, F., Pauw, N., 2009. A comparative study of surface and subsurface flow constructed wetlands for treatment of combined sewer overflows: a greenhouse experiment. *Ecol. Eng.* 35, 175–183.
- Vanderploeg, H.A., Ludsin, S.A., Ruberg, S.A., Höök, T.O., Pothoven, S.A., Brandt, S.B., Lang, G.A., Liebig, J.R., Cavaletto, J.F., 2009a. Hypoxia affects spatial distribution of pelagic fish, zooplankton, and phytoplankton in Lake Erie. *J. Exp. Mar. Biol. Ecol.* 381, S92–S107.
- Vanderploeg, H.A., Ludsin, S.A., Cavaletto, J.F., Höök, T.O., Pothoven, S.A., Brandt, S.B., Liebig, J.R., Lang, G.A., 2009b. Hypoxic zones as habitat for zooplankton in Lake Erie: refuges from predation or exclusions zones? *J. Exp. Mar. Biol. Ecol.* 381, S108–S120.
- Vanderploeg, H.A., Wilson, A.E., Johengen, T.H., Dyble Bessie, J., Sarnelle, O., Liebig, J.R., Robinson, S.D., Horst, G.P., 2013. Role of selective grazing by dreissenid mussels in promoting toxic *Microcystis* blooms and other changes in phytoplankton composition in the Great Lakes. In: Nalepa, T.F., Schlosser, D.W. (Eds.), *Quagga and Zebra Mussels: Biology, Impacts, and Control*. 2nd ed. CRC Press, Boca Raton, FL, pp. 509–523.
- Vollenweider, R.A., 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. Organisation for Economic Cooperation and Development Technology Report 159 pp.
- Vollenweider, R.A., Munawar, M., Stadelmann, P., 1974. A comparative review of phytoplankton and primary production in the Laurentian Great Lakes. *J. Fish. Res. Board Can.* 31, 739–762.
- Vijayavel, K., Sadowsky, M.J., Ferguson, J.A., Kashian, D.R., 2013. The establishment of the nuisance cyanobacteria *Lyngbya wollei* in Lake St. Clair and its potential to harbor fecal indicator bacteria. *J. Great Lakes Res.* 39 (4), 560–568.
- Vis, C., Cattaneo, A., Hudon, C., 2008. Shift from chlorophytes to cyanobacteria in benthic microalgae along a gradient of nitrate depletion. *J. Phycol.* 44 (1), 38–44.
- Wacklin, P., Hoffmann, L., Komárek, J., 2009. Nomenclatural validation of the genetically revised cyanobacterial genus *Dolichospermum* (Ralfs ex Bornet et Flahault) comb. nova. *Fottea* 9 (1), 59–64.
- Wang, N., Mitsch, W.J., 1998. Estimating phosphorus retention of existing and restored coastal wetlands in a tributary watershed of the Laurentian Great Lakes in Michigan, USA. *Wetl. Ecol. Manage.* 6, 69–82.
- Wang, W., Shen, H., Shi, P., Chen, J., Ni, L., Xie, P., 2015. Experimental evidence for the role of heterotrophic bacteria in the formation of *Microcystis* colonies. *J. Appl. Phycol.* 1–13, <http://dx.doi.org/10.1007/s10811-015-0659-5>.
- Wannamaker, C.M., Rice, J.A., 2000. Effects of hypoxia on movements and behavior of selected estuarine organisms from the southeastern United States. *J. Exp. Mar. Biol. Ecol.* 249 (2), 145–163.
- Watson, S.B., Boyer, G.L., 2014. Harmful Algal Blooms (HABS) in the Great Lakes: current status and concerns. State of the Lakes Ecosystem (SOLEC) report, In: [http://binational.net/home\\_e.html](http://binational.net/home_e.html).
- Watson, S.B., Boyer, G.L., Ridal, J., 2008. Algal and cyanobacterial taste and odour and toxins: a review of current impairment, prediction and management in the Great Lakes. *Can. J. Fish. Aquat. Sci.* 65, 1779–1796.
- Watson, S.B., Boyer, G.L., Newbold, B., Matthews, E., Yang, R., 2012. Algal Bloom response and risk management: evaluation of on-site toxin kits in Hamilton Harbour. NWRI Report. Canadian Centre for Inland Waters, Burlington, ON 24 pp.
- Watson, S.B., Carpenter, D., Casselman, J., Miller, C., Murray, M., Arhonditsis, G., Bosch, N., Boyer, G.L., Charlton, M., Confesor Jr., R., Depew, D., Dolan, D., DePinto, J., Higgins, S., Höök, T., Howell, T., Ludsin, S., McElmurry, S., Richards, P., Scavia, D., Smith, R., Steffen, M., Wilhelm, S., Wang, Z., 2013. Taking Action on Lake Erie (TAcLE). International Joint Commission Report. 101 pp. [http://ijc.org/en/\\_leep/Technical\\_Documents](http://ijc.org/en/_leep/Technical_Documents).

- Watson, S.B., Whitton, B.A., Higgins, S.N., Paerl, H.W., Brooks, B.W., Wehr, J.D., 2015. Harmful Algal Blooms. In: Wehr, J.D., Sheath, R.G., Kociolek, J.P. (Eds.), *Freshwater Algae of North America*. Academic Press, San Diego, CA, ISBN: 978-0-12-385876-4 pp. 873–920.
- Welborn, C., Veenhuis, J., 1987. Effects of runoff control on the quality and quantity of urban runoff in two locations in Austin, TX. USGS Water Resources Investigations Report, pp. 87–4004.
- Wilcox, D.A., Whillans, T.H., 1999. Techniques for restoration of disturbed coastal wetlands of the Great Lakes. *Wetlands* 19 (4), 835–857.
- Wilcox, D.A., Nichols, S.J., 2008. The effects of water-level fluctuations on vegetation in a Lake Huron wetland. *Wetlands* 28 (2), 487–501.
- Wilson, A.E., Sarnelle, O., Neilan, B.A., Salmon, T.P., Gehringer, M.M., Hay, M.E., 2005. Genetic variation of the bloom-forming cyanobacterium *Microcystis aeruginosa* within and among lakes: implications for harmful algal blooms. *Appl. Environ. Microbiol.* 71 (10), 6126–6133.
- Wilcox, D.A., Thompson, T.A., Booth, R.K., Nicholas, J.R., 2007. Lake-Level Variability and Water Availability in the Great Lakes, U.S. Geological Survey. Circular 1311 25 pp.
- Wilson, K.A., Howell, E.T., Jackson, D.A., 2006. Replacement of zebra mussels by quagga mussels in the Canadian nearshore of Lake Ontario: the importance of substrate, round goby abundance, and upwelling frequency. *J. Great Lakes Res.* 32 (1), 11–28.
- Wilhelm, S.W., Bullerjahn, G.S., Eldridge, M.L., Rinta-Kanto, J.M., Bourbonniere, R.A., 2006. Seasonal hypoxia and the genetic diversity of prokaryote populations in the central basin hypolimnion of Lake Erie: evidence for abundant cyanobacteria and photosynthesis. *J. Great Lakes Res.* 32 (4), 657–671.
- Wilhelm, S.W., Farnsley, S.E., LeCleir, G.R., Layton, A.C., Satchwell, M.F., DeBruyn, J.M., Boyer, G.L., Zhu, G.W., Paerl, H.W., 2011. The relationships between nutrients, cyanobacterial toxins and the microbial community in Taihu (Lake Tai), China. *Harmful Algae* 10 (2), 207–215.
- Wilhelm, S.W., LeCleir, G.R., Bullerjahn, G.S., McKay, R.M.L., Saxton, M.A., Twiss, M.R., Bourbonniere, R.A., 2014. Seasonal changes in microbial community structure and activity imply winter production is linked to summer hypoxia in a large lake system. *FEMS Microbiol. Ecol.* 87, 475–485.
- Winston, R.J., Hunt III, W.F., Osmond, D.L., Lord, W.G., Woodward, M.D., 2011. Field evaluation of four level spreader-vegetative filter strips to improve urban stormwater quality. *J. Irrig. Drain. Eng.* 137, 170–182.
- Winter, J.G., Duthie, H.C., 2000. Export coefficient modeling to assess phosphorus loading in an urban watershed. *J. Am. Water Res. Assoc.* 36 (5), 1053–1061.
- Wu, R.S., 2002. Hypoxia: from molecular responses to ecosystem responses. *Mar. Pollut. Bull.* 45, 35–45.
- Young, G.K., Stein, S., Cole, P., Kammer, T., Graziano, F., Bank, F., 1996. Evaluation and Management of Highway Runoff Water Quality. Federal Highway Administration Report No. PD-96-032. U.S. Department of Transportation, Washington, DC.
- Yousef, Y.A., Wanielist, M.P., Harper, H.H., 1985. Removal of highway contaminants by roadside swales. *Transp. Res. Rec.* 1017, 62–68.
- Yu, S.L., Kaighn, R., 1995. The Control of Pollution in Highway Runoff Through Biofiltration. Vol. II: Testing of Roadside Vegetation. Virginia Department of Transportation, Report No. FHWA/VA 95-R29, Richmond, VA.
- Yu, S.L., Benelmouffok, D.E., 1988. Field testing of selected urban BMPs in critical water issues and computer applications. In: Proceedings of the 15th Annual Water Resources Conference, American Society of Civil Engineers, New York, NY.
- Yu, S.L., Barnes, S.L., Gerde, V.W., 1993. Testing of Best Management Practices for Controlling Highway Runoff. Virginia Department of Transportation, Report No. FHWA/VA-93-R16, Richmond, VA 70 pp.
- Yu, S.L., Kaighn, R.J., Liao, S.L., 1994. Testing of Best Management Practices for Controlling Highway Runoff, Phase II. Virginia Department of Transportation, Report No. FHWA/VA-94-R21, Richmond, VA 62 pp.
- Zhang, H., Culver, D.A., Boegman, L., 2008. A two-dimensional ecological model of Lake Erie: application to estimate dreissenid impacts on large lake plankton populations. *Ecol. Model.* 214, 219–241.
- Zhang, H., Culver, D.A., Boegman, L., 2011. Dreissenids in Lake Erie: an algal filter or a fertilizer? *Aquat. Invas.* 6 (2), 175–194.
- Zhang, R., Zhou, W.B., Field, R., Tafuri, A., Yu, S.L., Jin, K.L., 2009. Field test of best management practice pollutant removal efficiencies in Shenzhen, China. *Front. Environ. Sci. Eng. China* 3 (3), 354–363.
- Zhang, R., Zhou, W.B., Li, J., Yu, S.L., 2010. Field evaluation of an innovative stormwater treatment device—the Stormvault™ system. *Environ. Monit. Assess.* 169 (1–4), 113–123.
- Zhao, J., Ramin, M., Cheng, V., Arhonditsis, G.B., 2008. Competition patterns among phytoplankton functional groups: how useful are the complex mathematical models? *Acta Oecol.* 33, 324–344.
- Zhou, Y., Obenour, D.R., Scavia, D., Johengen, T.H., Michalak, A.M., 2013. Spatial and temporal trends in Lake Erie hypoxia, 1987–2007. *Environ. Sci. Technol.* 47 (2), 899–905.
- Zhou, Y., Michalak, A.M., Beletsky, D., Rao, Y.R., Richards, R.P., 2015. Record-breaking Lake Erie hypoxia during 2012 drought. *Environ. Sci. Technol.* 49 (2), 800–807.