RECOMMENDED

PHOSPHORUS LOADING TARGETS

FOR LAKE ERIE

Annex 4 Objectives and Targets Task Team Final Report to the Nutrients Annex Subcommittee

May 11, 2015

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1.   Executive Summary

The Annex 4 Objectives and Targets Task Team was created in September 2013 with 25 binational members and Co-chaired by Sandra George, Environment Canada, and Jeffrey Reutter, Ohio Sea Grant and Stone Lab, The Ohio State University. The goal of the Task Team was to recommend revisions to phosphorus target concentrations and loads to Lake Erie needed to achieve the Lake Erie Objectives prescribed by Annex 4 (Nutrients) of the Great Lakes Water Quality Agreement (GLWQA) Amendment of 2012. Annex 4 seeks revised target loads and objectives for all of the Great Lakes, and calls for Lake Erie to be evaluated first because of observed re-eutrophication beginning in the mid-1990s. The worsening condition has been manifested in three ways: a reoccurrence of cyanobacteria blooms primarily in the Western Basin and primarily composed of the genus *Microcystis*; significant hypoxia conditions in the Central Basin hypolimnion; and the reoccurrence of major *Cladophora* nuisance blooms (along the northern nearshore of the Eastern Basin). These problems have occurred even when the overall annual target load of 11,000 metric tons of total phosphorus to the lake, as determined in 1978, has been met.

The Task Team agreed that the best way to determine phosphorus load management recommendations in the available time frame was to convene a sub-team of modeling experts and compare and contrast the results from a suite of existing Lake Erie models to quantify phosphorus load and eutrophication response relationships for the Lake Erie ecosystem. Utilizing nine validated models allowed the Task Team to evaluate the impact of a combination of load reduction strategies on Western Basin cyanobacteria blooms, Central Basin hypoxia, and Eastern Basin *Cladophora*. However, the Task Team also discussed the role of factors other than phosphorus loads and associated in-lake concentrations in governing eutrophication indicators. These other factors included nitrogen loads and concentrations, Dreissenid densities and impacts, and variations in annual precipitation and tributary discharges. Additional considerations included the setting of in-lake phosphorus concentration objectives for nearshore areas and the role that phosphorus bioavailability plays in governing the response of eutrophication response indicators to phosphorus loads. The results of those discussions and considerations are also presented in this report.

The report is divided into eight sections:

Section 1:   Executive Summary
Section 2:   Introduction
Section 3:   Lake Erie Eutrophication and Nutrient Load Trends
Section 4:   Ensemble Modeling Summary
Section 5:   Task Team Recommendations for Eutrophication Response Indicator (ERI) Thresholds, Corresponding Loading Targets, and Other Considerations
Section 6:   Conclusions
Section 7:   References
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The Task Team discussed a number of eutrophication response indicators for the Western Basin cyanobacteria blooms and Central Basin hypoxia. For cyanobacteria blooms the Task Team selected a target phosphorus load designed to produce a mild bloom (<9600 metric tons algal dry weight), the size of that observed in 2004 or 2012, or smaller, 90% of the time.
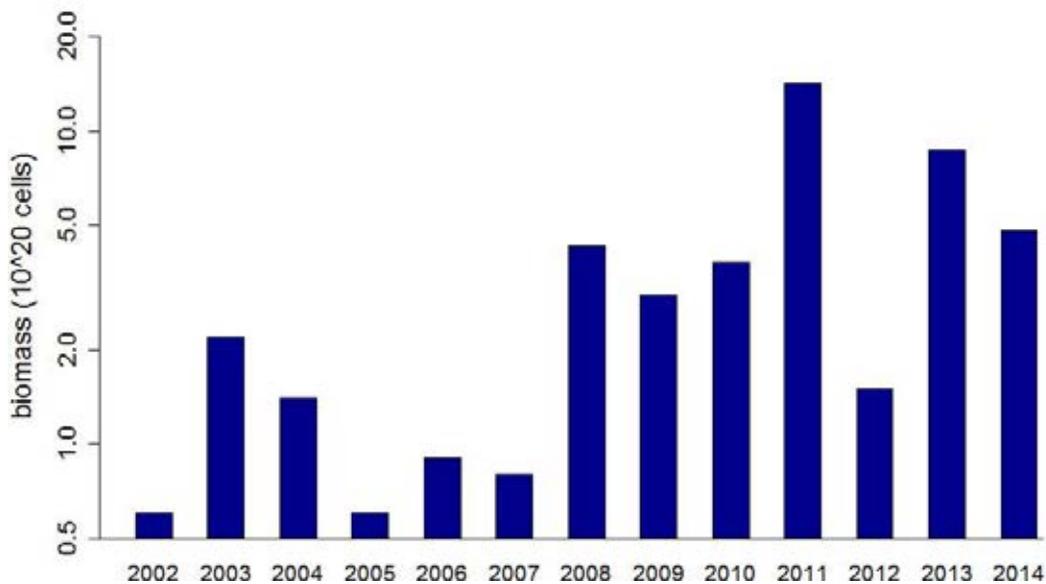
Cyanobacterial biomass for western Lake Erie based on measurement from satellite.
(Stumpf personal communication, updated from Stumpf et al., 2012).

The Task Team concluded that non-point source runoff from the Maumee River during the spring period of 1 March to 31 July each year was the best predictor of cyanobacteria bloom severity based upon the work of the modeling Sub-Team and loading data provided by the National Center for Water Quality Research at Heidelberg University. The Task Team used the 2008 water year as the base year for calculating reduction percentages. The scientific community considers phosphorus load measurements for 2008 accurate, and the 2008 whole-lake annual TP load was 10,675 metric tons per annum (MTA), which is very close to the Lake Erie target TP load of 11,000 MTA set in the 1978 Amendment to the GLWQA.

To achieve a bloom no greater than that observed in 2004 or 2012, 90% of the time, the Task Team recommends a total phosphorus (TP) spring load of 860 metric tons and a dissolved reactive phosphorus (DRP) load of 186 metric tons from the Maumee River. The 860 metric ton target is approximately a 40% reduction from the 2008 spring load of 1400 metric tons for TP and 310 metric tons of DRP, and the 2008 target load corresponds to a Flow Weighted Mean Concentration (FWMC) of 0.23 mg/L TP and 0.05 mg/L of DRP. Because discharge varies considerably from year to year, and because the discharge of the Maumee River was so large in 2008 that it has only been exceeded about 10% of the time in the last 20+ years, the Task Team expects that achieving a FWMC of 0.23 mg/L for TP and 0.05 mg/L for DRP will result in phosphorus loads below the targets (860 and 186 metric tons) 90% of the time (9 years out of 10), if precipitation patterns do not change.

 The Task Team also found that smaller cyanobacteria blooms have been observed from satellite imagery at the mouths of the Thames River (the 2011 phosphorus load from the Thames was 835 metric tons), River Raisin, Toussaint Creek, Portage River, and near Leamington. As a result, the Task Team concluded that, a 40% load reduction for each of those tributaries is also warranted. The 40% reduction target applies to all Western Basin tributaries unless there is a tributary program in place, which includes, monitoring, modeling, and/or management plans that demonstrate that the tributary and rivermouth nutrient conditions do not pose a cyanobacteria threat to adjacent nearshore water.

The Task Team agreed that bioavailable phosphorus was the most important target for reduction. DRP is 100% bioavailable and Particulate Phosphorus (PP) is 25-50% bioavailable. However, while reducing DRP will produce greater impacts than reducing PP, the models also showed that eliminating DRP with no reductions in PP would be insufficient to solve the problem. The Task Team also observed that since the mid-1990s, the load of DRP from the Maumee and other tributaries has increased approximately 150%.

For Central Basin hypoxia, the Task Team selected a target phosphorus load designed to raise the average August and September hypolimnetic dissolved oxygen concentration to 2.0 mg/L or higher. This is the threshold for hypoxia and should result in improvements to the Central Basin bottom habitat and reductions in internal loading of phosphorus from Central Basin bottom sediments during periods of anoxia.

The models demonstrated that the annual load of phosphorus was a driver of Central Basin hypoxia rather than just the spring load as was the case for Western Basin cyanobacteria blooms. Therefore, the targets for Central Basin hypoxia focus on annual loads from all tributaries to the Western Basin, including the Huron Erie Corridor (HEC), and the Central Basin tributaries.

The table below summarizes the recommendations of the Task Team for phosphorus load reduction targets in Lake Erie to achieve the desired eutrophication responses. More information on the approach and rationale for arriving at each of these recommendations is discussed in the body of this report.

Summary of Phosphorus Load Targets recommended to achieve desired thresholds for eutrophication response indicators in Lake Erie.

	Spring (Mar-July)	Annual
Western Basin Cyanobacteria – Bloom biomass less than or equal to 2004 or 2012 9 years out of ten, and/or reduce risk of nearshore localized blooms		
Maumee River		
Total Phosphorus load	860 MT#	
Dissolved Reactive Phosphorus load	186 MT#	
Other Western Basin Tributaries and Thames River		
Total Phosphorus load	40% reduction*	
Dissolved Reactive Phosphorus load	40% reduction*	
Central Basin Hypoxia – Aug –Sept Average Hypolimnetic Oxygen of 2 mg/L or more		
Total Phosphorus load to Western and Central Basins, including Detroit River and atmospheric load)		6000 MT**
Eastern Basin *Cladophora* – insufficient information to establish target		

#to be met 90% of the time based on inter-annual flow variability for the March-July period.
*Note: Percent reductions are based on 2008 loads
**This represents a 40% reduction of annual loads to the Western and Central Basins, including the Detroit River and atmospheric load.

For Eastern Basin *Cladophora*, the models demonstrated that open lake phosphorus concentrations in the Eastern Basin would be reduced by the efforts to address cyanobacteria blooms and hypoxia, but the models of *Cladophora*-phosphorus uptake and utilization were insufficiently developed to allow the Task Team to evaluate the impact of the reductions on *Cladophora* growth. The models also were not able to predict the impact of any proposed Eastern Basin tributary load reductions on nearshore growth. Therefore, while the Task Team believes improvements in Eastern Basin *Cladophora* densities may occur from reductions targeting cyanobacteria blooms and hypoxia, the Task Team could not reach consensus on further tributary reductions at this time. More research and model development will be required to allow us to address that issue.

Finally, the Task Team recognizes that there are intrinsic uncertainties associated with the estimated load-response relationships supported by our ensemble modeling analysis. These uncertainties arise from the natural variability of the system (hydrology, meteorology, ecological structure and function), and the fact that all models are simplifications of reality. The system response predicted by the models provides the best available estimate of the load reduction needed to meet Annex 4 objectives, and warrants the recommended action as a "no regrets" strategy. In recognition of the inherent uncertainties, the Task Team strongly endorses adoption of a carefully-designed adaptive management process to track the response of the system, evaluate the effectiveness of management efforts, and update management recommendations as we learn more about the processes underlying the system response. This effort will require a monitoring program capable of tracking loading trends over time and in-lake responses, as well as studies directed at learning more about processes that may be important, but are incompletely understood. The Task Team also recommends updating the models at regular intervals as part of the adaptive management process. The Coordinated Science and Monitoring Initiative (CSMI) cycle will provide opportunities for focused science and monitoring but a 5-year interval will not be sufficient for an effective adaptive management process. The monitoring program needed for the adaptive management approach must be capable of detecting critically important changes in precipitation patterns and phosphorus loads. If the frequency of severe storms increases in the future as predicted by climate change models, phosphorus loading to the lake will increase and the frequency and severity of cyanobacterial blooms will also increase, requiring larger phosphorus load reductions than those recommended in this report. The recommended adaptive management approach would detect these occurrences and allow for modification of target loads.

2.   Introduction

In the late 1970s, a series of contemporary Great Lakes eutrophication models were applied to establish target phosphorus loads for each of the Great Lakes and large embayments/basins. Those target loads were codified in the Control of Phosphorus Annex (Annex 3) of the 1978 Amendment to the Great Lakes Water Quality Agreement (GLWQA). The models applied for that analysis ranged from simple empirical relationships to kinetically complex, process-oriented models. In order of increasing complexity, these included: Vollenweider's empirical total phosphorus (TP) model (all lakes), Chapra's semi-empirical model (all lakes), Thomann's Lake 1 process model (Lake Ontario and Lake Huron), DiToro's process model (Lake Erie), and Bierman's process model (Saginaw Bay). The results of these model applications have been documented in the International Joint Commission (IJC) Task Group III report (Vallentyne and Thomas, 1978) and in Bierman (1980). The post-audit of several of these models in the mid-1980s confirmed that they had established a good relationship between total phosphorus loading to a lake/basin/embayment and its system-wide averaged TP and chlorophyll *a* concentration.

In 2006, as part of the Parties' (United States represented by U.S. EPA, cand Canada represented by Environment Canada) review of the Great Lakes Water Quality Agreement, a sub-committee of Great Lakes modelers (co-chaired by Joe DePinto, LimnoTech, and David Lam, Environment Canada) conducted an examination of the data and models that were used to support the phosphorus target loads specified in Annex 3 of the Agreement relative to the current status of the Lakes. The charge to this sub-group was to address three questions:

(1)   Have we achieved the target Phosphorus (P) loads in all of the Great Lakes?
(2)   Have we achieved the water quality objectives in all of the Great Lakes?
(3)   Can we define the quantitative relationships between P loads and lake conditions with existing models? Are the models still valid on a whole lake basis or have ecosystem changes to the P-chlorophyll relationship occurred such that new or updated models need to be run?

The findings of this sub-group were that those models were aimed at whole lake eutrophication symptoms as they were manifested at the time, but were now not sufficiently spatially resolved to capture the nearshore eutrophication being observed throughout the lakes and did not represent the process formulations to capture the impacts of ecosystem structure and function changes (e.g., Dreissenid impacts) relative to phosphorus processing and eutrophication responses in the lakes (DePinto *et al*., 2006). There was a general recommendation for a concerted research, monitoring, and model enhancement effort:

- to quantify the relative contributions of various environmental factors (total phosphorus loads, changes in the availability of phosphorus loads, hydrometeorological impacts on temperature conditions and hypolimnion structure and volume, *Dreissena*-induced alterations of nutrient-phytoplankton-light conditions and oxygen demand functions) to the nearshore re-eutrophication of the Great Lakes; and
- to develop a revised quantitative relationship between these stressors and the recently observed eutrophication indicators such as cyanobacteria blooms, enhanced hypoxia and nuisance benthic algal (e.g., *Cladophora*, *Lyngbya*) growth.

The recent publication of the 2012 Protocol amending the Great Lakes Water Quality Agreement (United States and Canada, 2012) includes an Annex 4 on nutrients, in particular on phosphorus control to achieve the following Lake Ecosystem Objectives (LEO) related to eutrophication symptoms:

1.   minimize the extent of hypoxic zones in the Waters of the Great Lakes associated with excessive phosphorus loading, with particular emphasis on Lake Erie;
2.   maintain the levels of algal biomass below the level constituting a nuisance condition;
3.   maintain algal species consistent with healthy aquatic ecosystems in the nearshore Waters of the Great Lakes;
4.   maintain cyanobacteria biomass at levels that do not produce concentrations of toxins that pose a threat to human or ecosystem health in the Waters of the Great Lakes;
5.   maintain an oligotrophic state, relative algal biomass, and algal species consistent with healthy aquatic ecosystems, in the open waters of Lakes Superior, Michigan, Huron and Ontario; and

6. maintain mesotrophic conditions in the open waters of the western and central basins of Lake Erie, and oligotrophic conditions in the eastern basin of Lake Erie.

The Annex set "interim" phosphorus concentration objectives and loading targets that are identical to the Annex 3 values established in the 1978 Amendment. However, it requires that the "Parties, in cooperation and consultation with State and Provincial Governments, Tribal Governments, First Nations, Métis, Municipal Governments, watershed management agencies, other local public agencies, and the Public, shall:

(1) For the open Waters of the Great Lakes:
   a. Review the interim Substance Objectives for phosphorus concentrations for each Great Lake to assess adequacy for the purpose of meeting Lake Ecosystem Objectives, and revise as necessary;
   b. Review and update the phosphorus loading targets for each Great Lake; and
   c. Determine appropriate phosphorus loading allocations, apportioned by country, necessary to achieve Substance Objectives for phosphorus concentrations for each Great Lake;
(2) For the nearshore Waters of the Great Lakes:
   a. Develop Substance Objectives for phosphorus concentrations for nearshore waters, including embayments and tributary discharge for each Great Lake; and
   b. Establish load reduction targets for priority watersheds that have a significant localized impact on the Waters of the Great Lakes.

The Annex also calls for research and other programs aimed at setting and achieving the revised nutrient objectives. The Parties are to take into account the bioavailability of various forms of phosphorus, related productivity, seasonality, fisheries productivity requirements, climate change, invasive species, and other factors, such as downstream impacts, as necessary, when establishing the updated phosphorus concentration objectives and loading targets. It also states that concentration objectives and loading targets are to be developed for other nutrients if required. Finally, it calls for the Lake Erie objectives and loading target revisions to be completed within three years of the 2012 Agreement entry into force.

To assist the Parties in developing and applying an approach for accomplishing these Annex 4 mandates, an Annex 4 Subcommittee was formed. This Subcommittee in turn formed three task teams: an Objective and Targets Task Team, an Agricultural Sources Task Team, and an Urban and Rural Sources Task Team. The charge to the Objectives and Targets Task Team was to initially accomplish the following goals for Lake Erie:

- By November 2014, review and update Substance Objectives (phosphorus concentrations) for offshore waters and develop Substance Objectives for nearshore waters
- By February 2015, review and update P loading targets for offshore waters and establish nearshore P load reduction targets necessary to achieve Substance Objectives and allocate by country.

Table 1. Summary of GLWQA commitments for Lake Erie. This table summarizes how the Nutrients Annex Sub Committee addresses the GLWQA commitments. It is important to note that the targets presented below, by basin, work in concert not in isolation. All tributaries to Lake Erie, including the Detroit River and the Huron-Erie Corridor contribute phosphorus loads to Lake Erie. In addition, the Western Basin loads contribute to the Central Basin loads which contribute to the Eastern Basin.

GLWQA Commitment	Recommended Target for Lake Erie	Comments

1. **for the Open Waters of the Great Lakes:**
  • **Minimize the extent of hypoxic zones associated with excessive phosphorus loading (1)**
  • **Maintain the levels of algal biomass below the level constituting a nuisance condition (2)**
  • **Maintain cyanobacteria biomass at levels that do not produce concentrations of toxins that pose a threat to human or ecosystem health (4)**
  • **Maintain mesotrophic conditions in the open waters of the western and central basins of Lake Erie, and oligotrophic conditions in the eastern basin of Lake Erie (6)**		

(a) review the interim Substance Objectives for phosphorus concentrations for each Great Lake to assess adequacy for the purpose of meeting Lake Ecosystem Objectives, and revise as necessary;	No new phosphorus concentration objectives for the open waters are recommended at this time.	With achievement of the loading targets, the following P concentrations for the open waters are expected: Western Basin -12 µg/L Central Basin - 6 µg/L Eastern Basin - 6 µg/L Flow-weighted mean concentrations at tributary mouths should be used as a benchmark to track progress in load reductions.
(b) review and update the phosphorus loading targets for each Great Lake;	**Target load to reduce cyanobacteria blooms in the Western Basin:** Reduce **spring TP** and **DRP** loads from Maumee by 40% from the 2008 spring loads. **Target load to reduce hypoxia in the Central Basin:** 6,000 metric tons TP annual loading to the Central Basin. This loading target represents a 40% reduction in loadings from 2008 levels. **Target load to reduce nuisance algal density in the Eastern Basin**: There is insufficient information/data at this time to recommend a loading target.	Achieving the Maumee target will reduce cyanobacteria blooms to non-severe levels (i.e. levels less than or equal to the 2004/2012 blooms) 90% of the time (i.e., nine years out of 10). While the models indicate the Maumee River spring loads drive the Western Basin bloom, we believe that when the Maumee loads are high the loads from other tributaries are also high and that they too contribute to the bloom. Therefore, we recommend a 40% reduction in spring TP and DRP loads to the other Western Basin tributaries, starting with the nearshore priority watersheds in the Western Basin. Achieving the 6000 MTA target will increase the average DO level in the hypolimnion (cold bottom layer) to greater than 2 mg/L Aug to Sept, thereby reducing hypoxia, increasing DO levels in surface sediment, reducing internal loading of phosphorus to the Central Basin, and improving fishery habitat.
(c) determine appropriate phosphorus loading allocations, apportioned by country, necessary to achieve Substance Objectives for phosphorus concentrations for each Great Lake	Allocation by country to be determined	Adaptive management will be used to evaluate the effectiveness of our targets and associated actions.

GLWQA Commitment	Recommended Target for Lake Erie	Comments

2. **for the Nearshore Waters of the Great Lakes:**
  • **Maintain the levels of algal biomass below the level constituting a nuisance condition (2)**
  • **Maintain algal species consistent with healthy aquatic ecosystems in the nearshore waters (3)**
  • **Maintain cyanobacteria biomass at levels that do not produce concentrations of toxins that pose a threat to human or ecosystem health (4)**		

(a) develop Substance Objectives for phosphorus concentrations for nearshore waters, including embayments and tributary discharge for each Great Lake	No nearshore phosphorus concentration objectives are recommended at this time.	Flow-weighted mean concentrations at tributary mouths should be used as a benchmark to track progress in load reductions
(b) establish load reduction targets for priority watersheds that have a significant localized impact on the Waters of the Great Lakes.	Priority watersheds and targets to reduce nearshore algal blooms: Reduce spring TP and DRP loading by 40% relative to the 2008 spring loads in the following priority watersheds: **Lake St. Clair** Thames River - Cdn **Western Basin** Maumee River - US River Raisin - US Portage River - US Toussaint Creek - US Leamington Tribs – Cdn **Central Basin** Sandusky River - US Huron River, OH – US	In the nearshore areas of the priority watersheds there is evidence that blooms occur on a regular basis. Specific target loads for the Maumee River are: • 860 MT TP spring • 186 MT DRP spring Similar 40% reductions in spring loading (TP and DRP) relative to 2008 levels, are recommended for the other priority watersheds. Load reduction targets for the remaining priority watersheds will be established in the domestic action plans. For the Maumee River, the 2008 target load corresponds to a Flow Weighted Mean Concentration of 0.23 mg/L TP and 0.05 mg/L DRP.

This report contains a summary of the findings and recommendations of the Objectives and Targets Task Team on its work to evaluate the interim phosphorus concentration objectives and load targets for Lake Erie and to propose an updating of those targets to meet the above LEOs in light of new research, monitoring and modeling of the lake.

The geographic area for the proposed targets includes the Huron-Erie Corridor from the outflow of Lake Huron to Lake Erie. Figure 1 below provides a geographic orientation of Lake Erie, including delineation of its three basins, Lake St. Clair, and locations of major tributaries.

Figure 1. Schematic map of Huron-Erie Corridor and Lake Erie, showing its three basins and major tributaries. (Scavia, D. and J.V. DePinto. 2015. Annex 4 Ensemble Modeling Report).

During the past year, the Objectives and Targets Task Team deliberated and analyzed the charge it was given. The Task Team had two 2-day meetings and three 3-day meetings at NOAA's Great Lakes Environmental Research Lab, and 13 conference calls. Much of the deliberation involved identifying revised target P loads to achieve the Lake Erie objectives indicated in Annex 4. However, the Task Team also worked to understand the role of factors in addition to phosphorus loads and associated in-lake concentrations in governing eutrophication indicators. Among these other factors were nitrogen loads and concentrations, Dreissenid densities and impacts, and inter-annual hydrometeorological conditions. Additional considerations included: setting in-lake phosphorus concentration objectives for nearshore areas, and the role that phosphorus bioavailability plays in governing the response of eutrophication response indicators to phosphorus loads. The results of those discussions and considerations are also presented in this report.

The Objectives and Targets Task Team agreed that the best way to determine phosphorus load management recommendations to meet its charge in the time frame required was to apply an ensemble modeling approach using a suite of Lake Erie models that had already been developed to quantify phosphorus load – eutrophication response relationships for the current Lake Erie ecosystem (LimnoTech, 2013). The process and results of this modeling effort are also summarized below. Lessons learned from the approach developed and applied to Lake Erie can inform subsequent development of nutrient objectives and targets for the other Great Lakes in meeting the 2012 Great Lakes Water Quality Agreement Protocol Annex 4 mandates.

- Note: one of the objectives of the Task Team was to "maintain cyanobacteria biomass at levels that do not produce concentrations of toxins that pose a threat to human or ecosystem health in the Waters of the Great Lakes." Cyanobacteria are capable of producing very dangerous toxins and are often called Harmful Algal Blooms. Due to the uncertainty of when toxins will be produced and the difficulty of measuring, or quickly measuring, toxin concentrations, prevention of cyanobacteria blooms is the most appropriate action.

In this report we will refer to these blooms as cyanobacteria blooms or cyanobacteria biomass.

3.   Lake Erie eutrophication and nutrient load trends

Because of its large, heavily populated and agriculturally intense watershed and its relatively small size and shallow depth (in the Western Basin and Central Basin), Lake Erie has had a long history of exhibiting the most troublesome eutrophication symptoms of any of the Great Lakes (Fuller *et al*., 1995). For this reason, Lake Erie became the prime concern and model for addressing eutrophication issues in the Great Lakes when the GLWQA was first amended in 1978.

At that time, state-of-the-science models showed that P reduction was the best approach to manage eutrophication. Target TP loads were established for each lake (11,000 MTA for Lake Erie) to achieve certain water quality goals (7 µg/L and 5 µg/L chlorophyll *a* for the western and central/eastern basins of Lake Erie, respectively; Task Group III 1978). For Lake Erie, major load reductions were achieved through phosphate detergent bans and municipal point source controls, with target TP loading goals largely achieved in Lake Erie by the early 1980s (Dolan 1993), as shown in Figure 2. Because setting a 1 mg/L P concentration standard for point sources discharging >1 Million Gallons per Day (MGD) was not sufficient to achieve target loads in years of high precipitation, best management practices (BMPs) were implemented on agricultural lands within the basin (DePinto *et al*. 1986). By 1992, 34% of the Ohio Lake Erie basin land used for corn and soybeans was being farmed using conservation tillage practices (Ohio Lake Erie Office 1993).
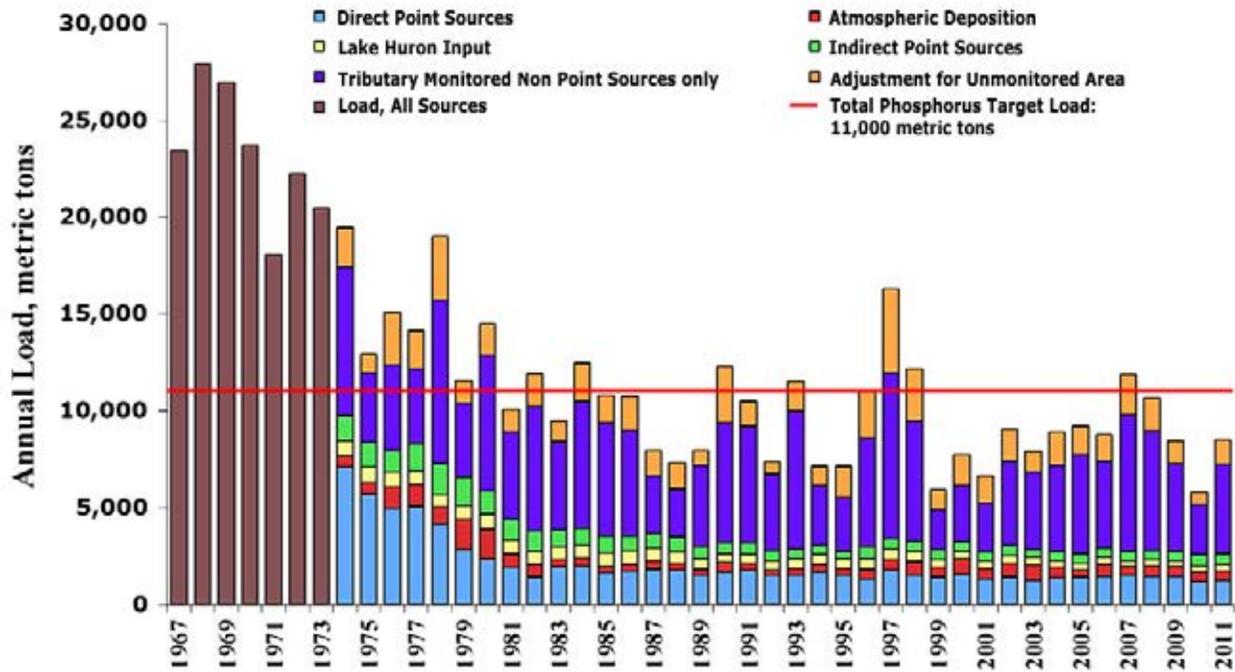
*Figure 2. Annual Total Phosphorus Load to Lake Erie. From Scavia **et al**. (2014).*

Response to P load reductions was rapid, profound, and close to that predicted by DiToro and Connolly (1980). A post-audit of their eutrophication model indicated it predicted concentrations of P, chlorophyll *a*, and central basin hypolimnion dissolved oxygen quite well (DiToro *et al*. 1987). Bertram (1993) reported that spring isothermal TP levels in the central basin dropped from ~20 µg/L in the 1970s to the target of 10 µg/L by 1987. Makarewicz (1993) reported a significant reduction in phytoplankton biomass in all three basins between 1970 and the mid-1980s. These values represented a 52-89% reduction in mean basin-weighted algal biomass from [1]970 values measured by Munawar and Munawar (1976). Delayed but only slight improvements in the degree of summer anoxia in the central basin had been observed (Bertram 1993; Charlton *et al*. 1993).

However, the models used to establish the 11,000 MTA load target in the 1970s were coarse-scale whole lake models and did not represent the process formulations to capture the impacts of ecosystem structure and function changes (e.g., Dreissenid impacts) on phosphorus processing and eutrophication responses (DePinto *et al.* 2006). Furthermore, the annual phosphorus load compilation (Figure 2) was not maintained on a routine basis after 1993; it was not until recently that the Lake Erie loads after that time were updated. Hence, the re-eutrophication of Lake Erie as described by a recent IJC report (IJC 2014) was not anticipated, nor was the relative importance of load changes versus ecosystem changes in causing these problems.

The re-eutrophication of Lake Erie over the past 20 years has been manifested in three ways: a reoccurrence of cyanobacteria blooms primarily in the Western Basin and primarily composed of the genus *Microcystis* (Bridgeman *et al.*, 2013) (Figure 3); the return of significant hypoxic conditions in the Central Basin hypolimnion (Zhou *et al.* 2013) (Figure 3); and the reoccurrence of *Cladophora* nuisance blooms along the northern nearshore of the Eastern Basin (IJC 2013).
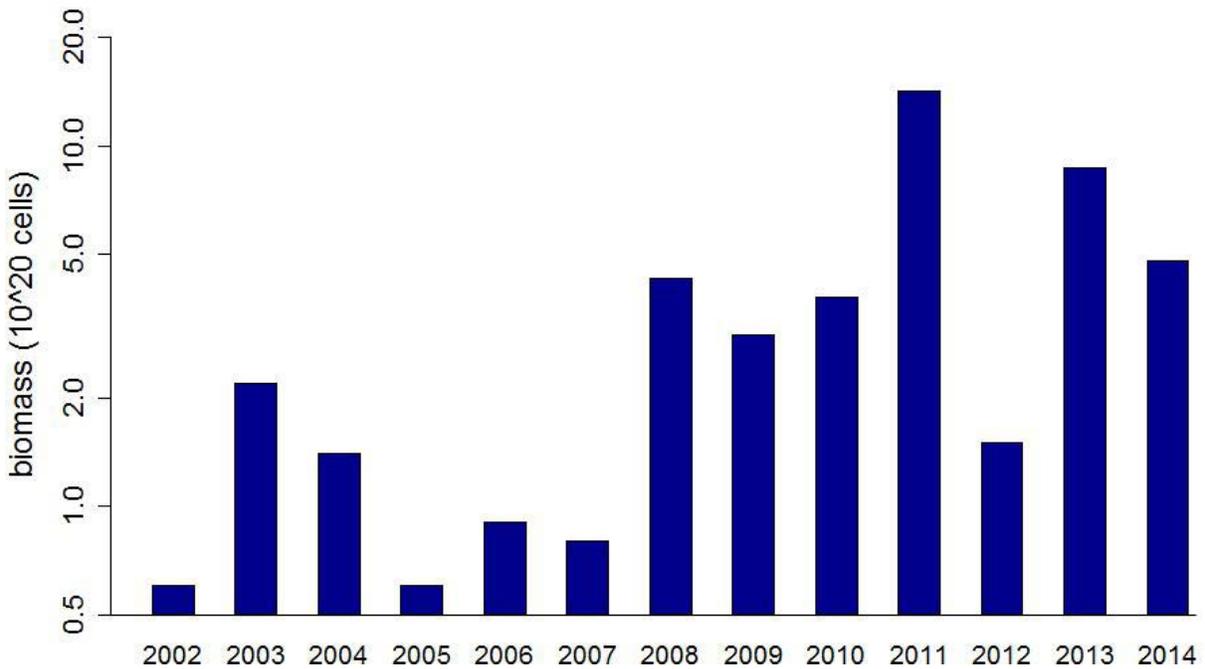
Figure 3. Cyanobacterial biomass for western Lake Erie based on measurement from satellite.
(Stumpf personal communication, updated from Stumpf *et al.* 2012)
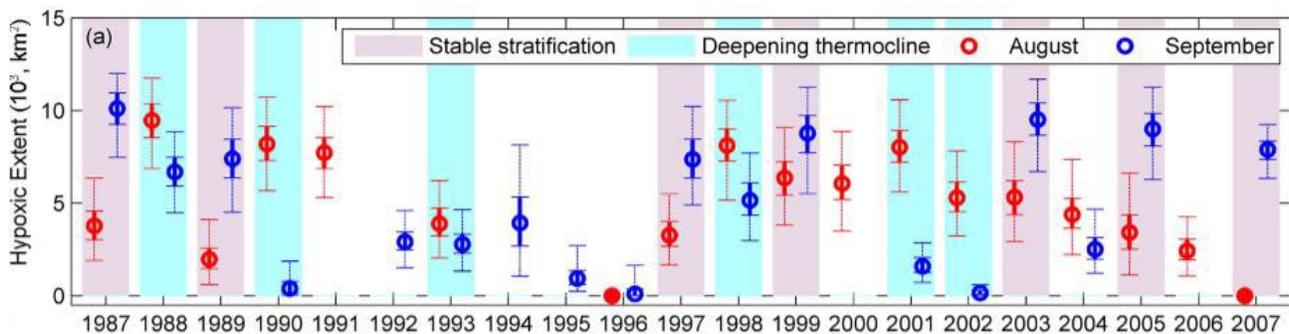
Figure 4. Estimates and 95% confidence intervals for the areal extent of Central Basin hypoxia (from Zhou *et al* 2013).

Characteristics and Trends of Phosphorus Loads to Lake Erie

After considerable research focused on these problems over the past 10-15 years, the scientific community identified two major drivers of these eutrophication symptoms: changes in the character of the phosphorus loads to the lake from its watershed and the ecological changes that resulted from the Dreissenid invasion. The phosphorus loading characteristics and trends are described below; Dreissenids are discussed later in section 5.

Phosphorus Loads to Lake Erie

Over the most recent ten years for which detailed data on Lake Erie TP loads are available (2002-2011), phosphorus from non-point sources and point sources upstream of tributary sampling stations, transported to the lake by runoff and rivers, contributed on average 78% of the total annual load to the lake (Dolan and Chapra 2012; Dolan, pers. comm. 2012). Because of the size of the Detroit River and the Maumee River, the Western Basin receives the majority of the TP load (Figure 5). Since 1994 the Western Basin has received 61% of the whole lake annual TP load, while the Central Basin and Eastern Basin received 28% and 11%, respectively.
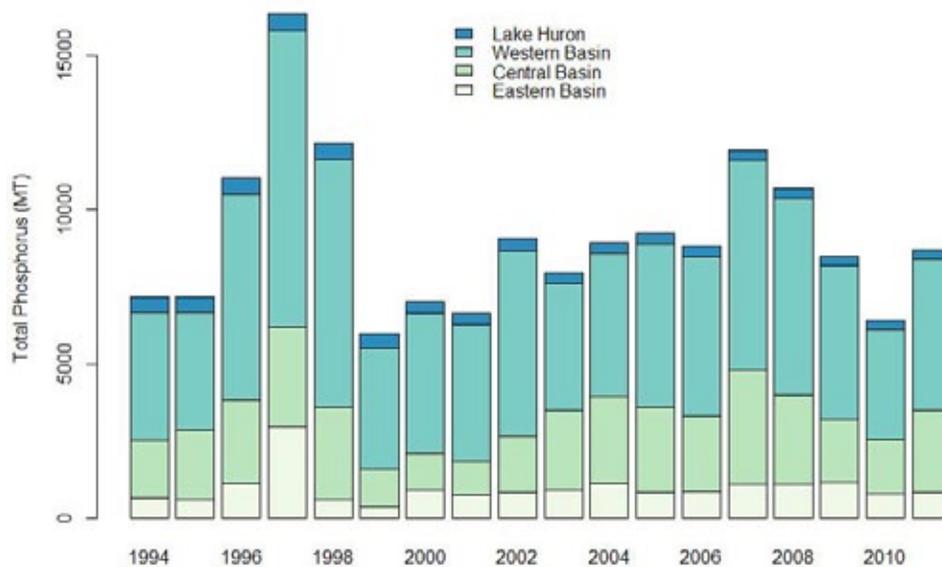
Figure 5. Total phosphorus loads to Lake Erie divided by basin for water years 1994-2011. A water year runs from October - September (sources: Dolan and Chapra 2012; Dolan, pers. comm., 2012). Note: the Lake Huron load is measured at the outlet of Lake Huron.

Figure 6 shows the breakdown of TP loads to the Western Basin for 2011-2013. During this time, an average of 41% of the annual TP load to the basin came from the Detroit River, with 47% from the Maumee River. The interannual variability of these percentages are evident from comparing these three years, when 2011 was a high runoff year for Western Basin tributaries, 2012 was a very dry year, and 2013 was midway between 2011 and 2012 with respect to Western Basin tributary flows. These variations in tributary flows lead to a variation in the percent contribution from the Detroit River, the flow from which does not vary much from year to year. It should also be noted that, because the flows from the Detroit River are so large (94% of flow into the Western Basin during this period) compared to the Maumee River flow (4% of the total flow), the concentration of TP in the Detroit River is 25 times smaller than the Maumee River (0.014 mg/L P versus 0.42 mg/L P respectively).
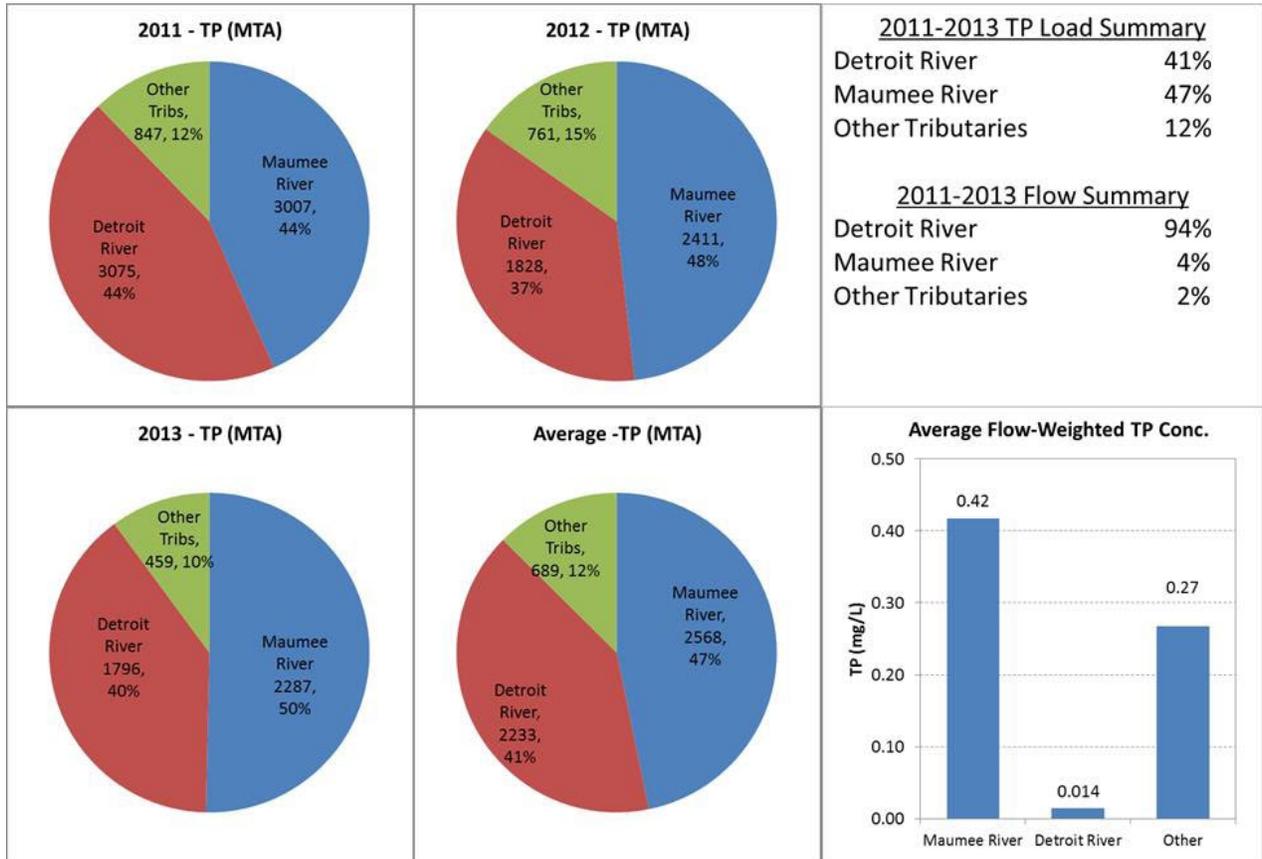
Figure 6. Source distribution of annual TP load (MTA) to the western basin of Lake Erie for 2011-13 water years, and the average TP concentration in those sources during that time (based on Maccoux, personal communication).

As discussed below, 2008 has been selected by the Objectives and Targets Task Team as a baseline year from which to compute recommended load reduction percentages. Therefore, it is worth looking at the loading characteristics for that year, based on the work of Dolan and Chapra (2012), and to note that it differs somewhat from the 2011-2013 averages shown in Figure 6. Figure 7 shows the significance of the Maumee River and Detroit River TP load. This pie chart indicates the distribution of the annual TP load to the lake by tributary, treating the Detroit River connecting channel load as a single source and accounting for all tributaries that individually contributed >150 MTA during that year.

2008 Total Annual TP Load to Lake Erie = 10,722 MTA

165
202
205
262
239
359
452
447
1101
1396
3812
1987

- Maumee River
- Detroit River
- Sandusky River
- Grand River, ONT
- Cuyahoga River
- Portage River
- Cattaraugus Creek
- Raisin River
- Huron River, OH
- Vermilion River, OH
- Grand River, OH
- Other
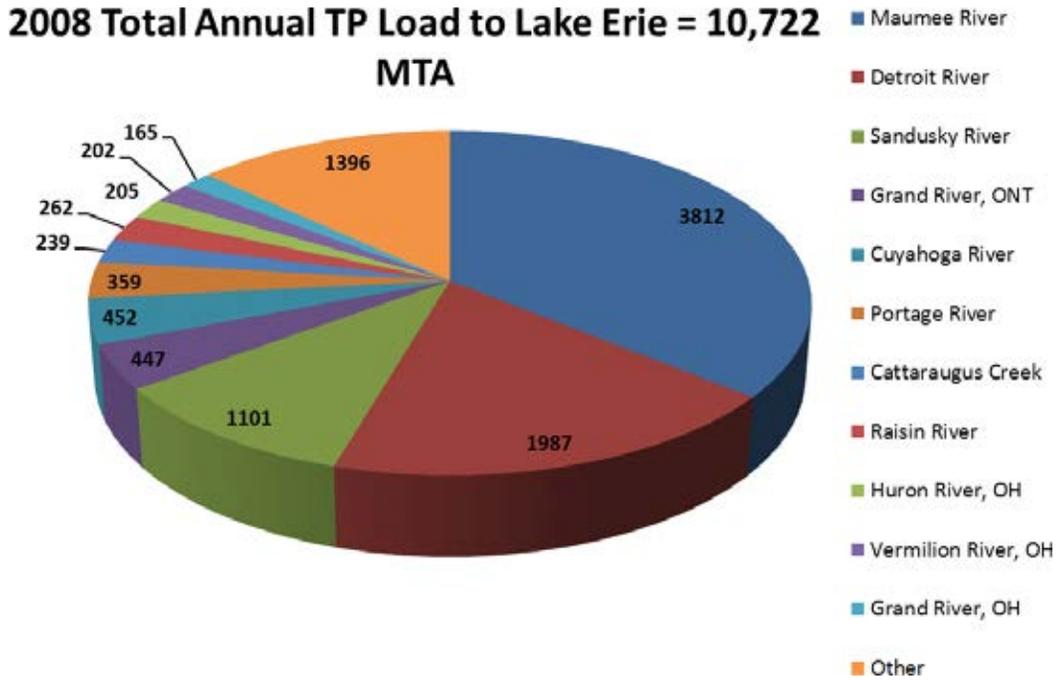
Figure 7. Annual TP load (MTA) to Lake Erie during the 2008 water year broken down by major tributaries and the Detroit River connecting channel (Maccoux, unpublished data).

Finally, it should be recognized that the loading sources for the two major phosphorus contributors differ. The distribution of source types for the Maumee and Detroit Rivers for 2008 is shown in Figure 8. Note that 94% of the 2008 annual TP load from the Maumee is nonpoint source (NPS) in origin, while 34% of the Detroit River load (includes the St. Clair River, Lake St. Clair and Detroit River sources) is from NPS, 16% is from Lake Huron, and the remaining 50% is from point sources (PS) (including the Detroit Wastewater Treatment Plant (WWTP)). In 2008, Dolan reported the Detroit WWTP load as 764 MTA; however, estimates for more recent years have been closer to ~500 MTA as reported by the Michigan DEQ (MDEQ, personal communication). The Thames River which flows into Lake St. Clair is lumped into the Detroit River load. However, it is important to note that 55% of the difference in annual Detroit River loads between 2008 (1987 MT) and 2011 (3077 MT) is due to changes in loads from the Thames River.

Maumee River

150 142 83

3812 MTA

3,437

- Monitored NPS
- Unmonitored NPS
- Indirect PS
- Direct PS

Detroit River

1987 MTA

321 360
167
109
764 266

- Monitored NPS
- Unmonitored NPS
- Indirect PS
- Direct PS
- Detroit WWTP
- Input from L. Huron
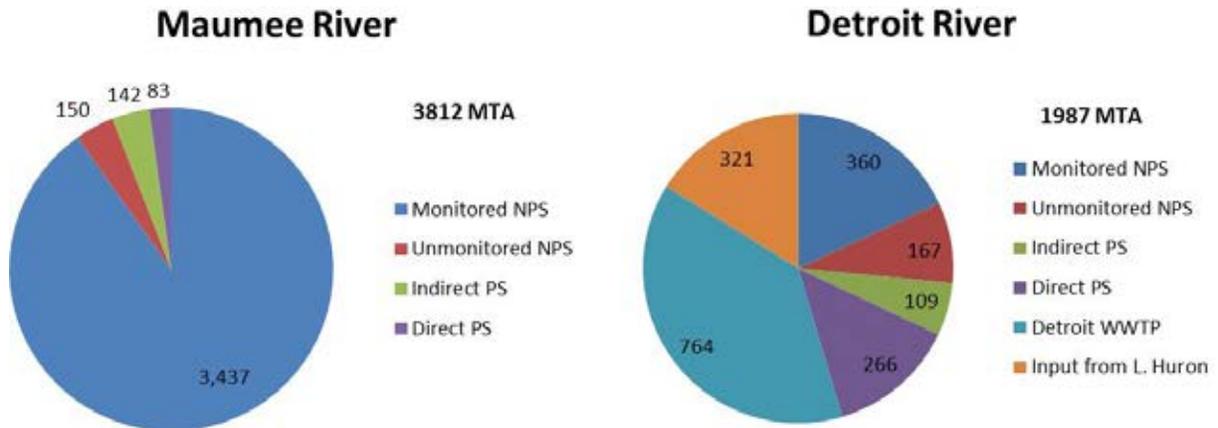
Figure 8. Distribution of annual TP load for 2008 from the Maumee and Detroit Rivers by source category (Dolan unpublished data).

Phosphorus Export from the Maumee River

With the success of point source phosphorus controls in the 1970s and the increasing importance of phosphorus from nonpoint sources, the Maumee River has become a dominant phosphorus source to Lake Erie (Figure 7). Thus, the characteristics and trends of inputs from the Maumee are important for assessing future management options to alleviate cyanobacterial blooms. Long-term patterns are revealed in the detailed tributary loading studies on the Maumee River that have been conducted by the National Center for Water Quality Research (NCWQR) at Heidelberg University and discharge data collected by the USGS.
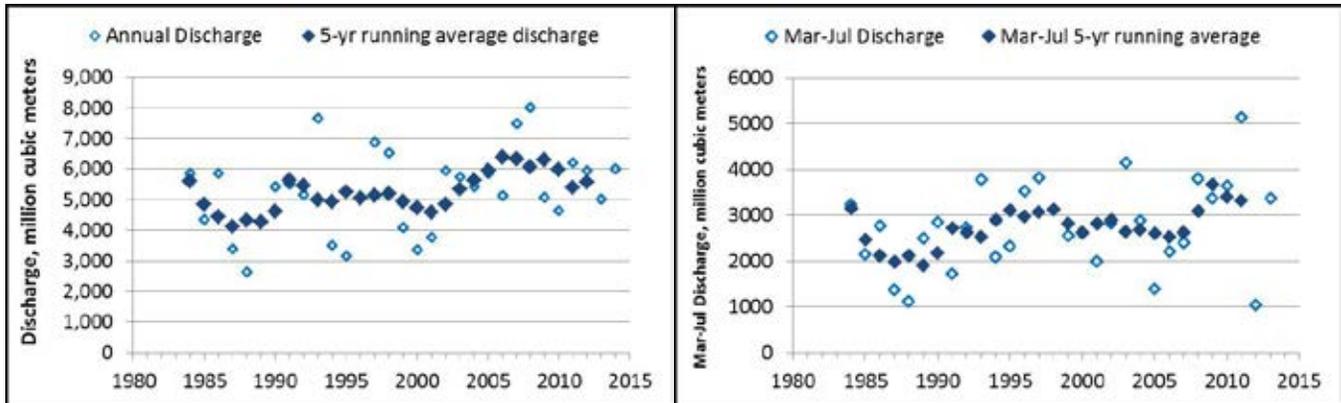
Figure 9. Maumee River annual discharge (left) and discharge during the March-July "critical period".
Open diamonds = raw values, solid diamonds = five-year running average.

Annual discharge from the Maumee is highly variable (Figure 9), due to variations in the intensity, amount, and timing of precipitation. This variability is an important factor leading to yearly differences in phosphorus loads. Similarly, discharge from spring to early summer (March-July) varies annually. Inter-annual variability during this period has been associated with variations in the size of the summer cyanobacteria bloom (Stumpf *et al* 2013, Obenour *et al* 2014), thus inputs during this "critical period" merit particular attention. Both the annual and critical period discharges suggest slightly increasing patterns since 1985, a factor that may be contributing to re-eutrophication.

Flow Weighted Mean Concentrations (FWMC) provide a useful means to address inter-annual variability by normalizing the phosphorus delivery from a tributary with respect to flow, so that year-to-year performance is not confounded by inter-annual variability in hydrology. FWMC can be calculated for tributaries by dividing the phosphorus load during a specified period (e.g., annually or March-July) by the cumulative flow during that period.

Total phosphorus concentrations also vary yearly (Figure 10) which, in combination with the yearly discharge variability makes annual total phosphorus loads highly variable. Similarly, total phosphorus concentrations and loads demonstrate inter-annual variability during the critical period, likely contributing to inter-annual differences in bloom size. During the period of record there are no clear trends in total phosphorus concentrations or loads.

In contrast, dissolved reactive phosphorus concentrations and loads exhibit a much clearer pattern over this time period (Figure 11). Concentrations and loads have steadily increased since the mid-1990s, both annually and during the critical period. These increases are particularly noteworthy because dissolved reactive phosphorus is readily bioavailable and thus may be a contributing factor to re-eutrophication observed in the 2000s.
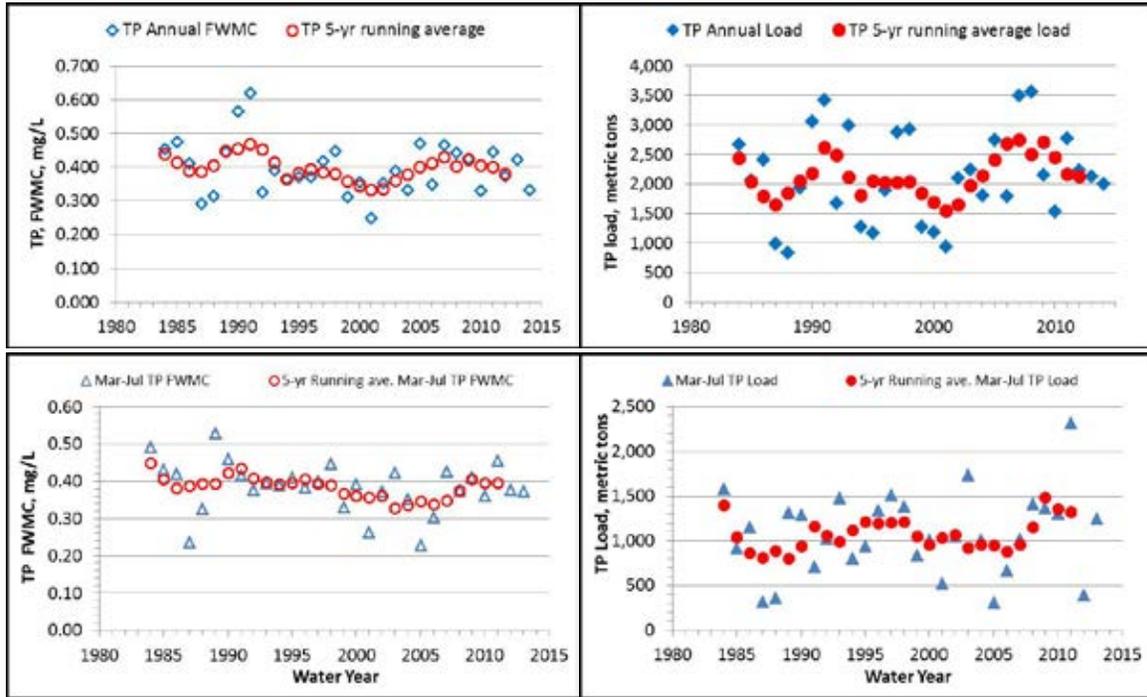
Figure 10. Maumee River flow-weighted mean total phosphorus concentrations (left panels) and loads (right panels). Top panels depict annual (water year) values, bottom panels depict March-July "critical period" values. Blue symbols = raw values, red symbols = five-year running averages.
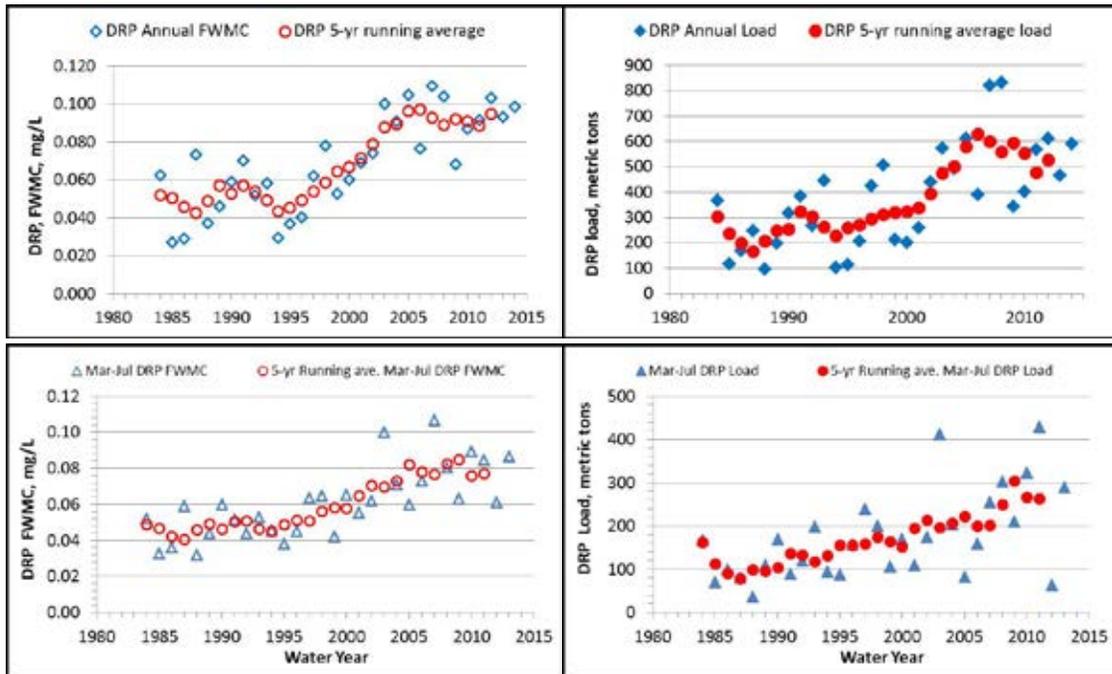
Figure 11. Maumee River flow-weighted mean dissolved reactive phosphorus concentrations (left panels) and loads (right panels). Top panels depict annual (water year) values, bottom panels depict March-July "critical period" values. Blue symbols = raw values, red symbols = five-year running averages.

Changes in other Maumee River constituents during this time are also notable. Annual nitrate concentrations exhibit recent decreases (Figure 12), although annual loads have been fairly steady, due to the simultaneous discharge increases. These annual patterns are also reflected during the critical period. Suspended solids concentrations have also decreased, although the annual decrease is not as pronounced as that of the critical period (Figure 13). Changes in suspended solids loads are less apparent, again due to the concurrent discharge increases.
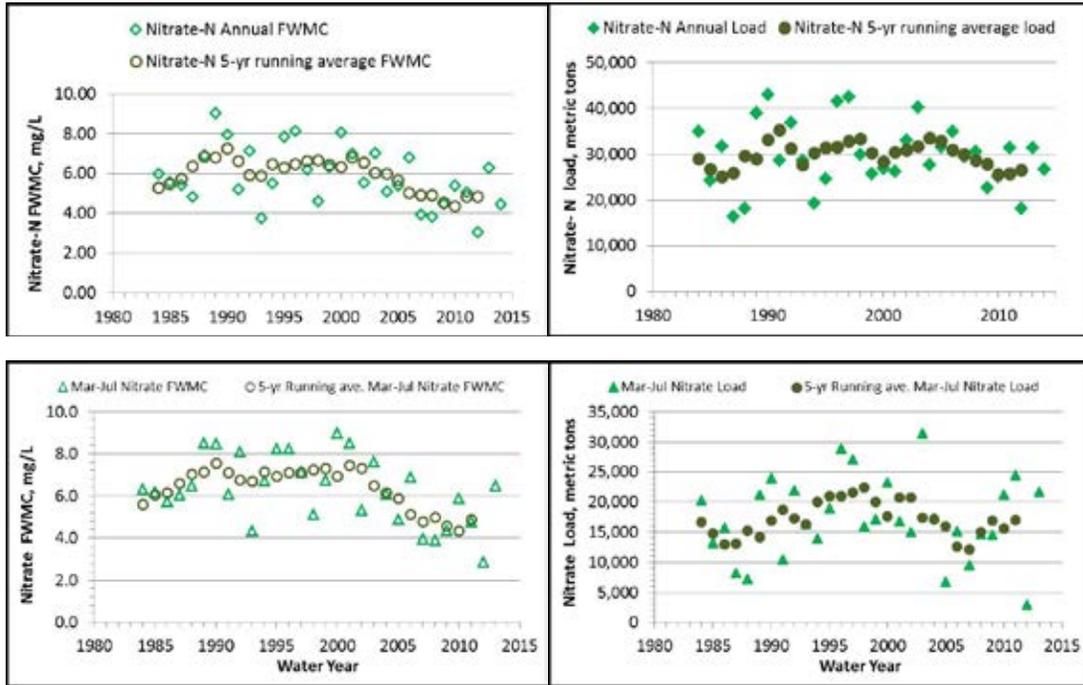
Figure 12. Maumee River flow-weighted mean total nitrate concentrations (left panels) and loads (right panels). Top panels depict annual (water year) values, bottom panels depict March-July "critical period" values. Light green symbols = raw values, dark green symbols = five-year running averages.
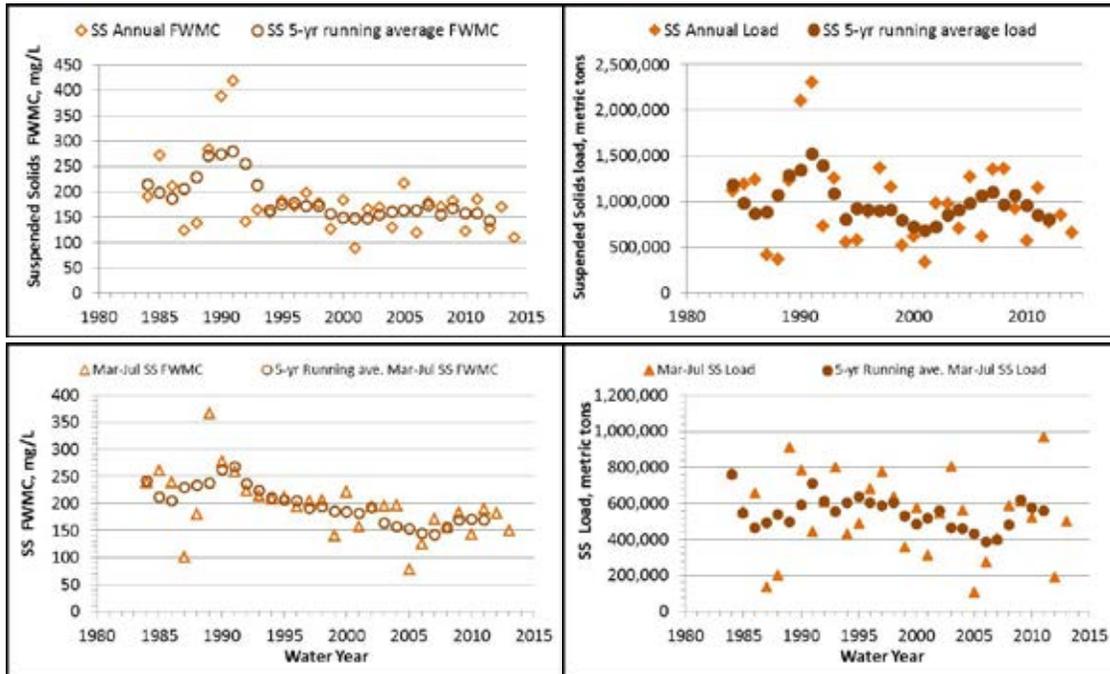
Figure 13. Maumee River flow-weighted mean total suspended solids concentrations (left panels) and loads (right panels).
Top panels depict annual (water year) values, bottom panels depict March-July "critical period" values.
Light green symbols = raw values, dark green symbols = five-year running averages.

In summary, Maumee River discharge increased from 1984-2013, a pattern that has been shown to be consistent with long-term precipitation increases (Stow *et al*. 2015). Concurrently, while total phosphorus inputs exhibited minimal net change over this period, dissolved reactive phosphorus concentrations and loads have shown a steady increase. Because dissolved reactive phosphorus is readily bioavailable this increase has the potential to be contributing to recent bloom problems (Baker *et al*. 2014). Contemporaneous decreases in both nitrate and suspended solids suggest that changing land-use practices are a likely factor influencing constituent concentrations and loads and must be considered when evaluating the causes of, and approaches to mitigating, ongoing eutrophication symptoms.

4.   Ensemble modeling summary

This section summarizes the modeling work undertaken by the Task Team to inform their recommendations which appear in Section 5. The Objectives and Targets Task Team reached consensus that the best way to inform the establishment of targets for Lake Erie was to use an ensemble modeling approach similar to the one used in the 1970's, but with a different suite of models that recognized the changes in the Lake Erie ecosystem, the directives of Annex 4, and the resulting need for a revision of targets. The modeling team undertaking this ensemble modeling approach worked between April and December, 2014 to accommodate a very compressed time scale. The team held two workshops, the proceedings for which are included in the Annex 4 Ensemble Modeling Report (Scavia and DePinto 2015). The first workshop, in April, 2014, served to assess the capabilities of existing models relative to a set of criteria to measure their ability to develop phosphorus load-response curves for the lake eutrophication response indicators (ERI). The second workshop, held in September, 2014, was used to allow the modelers to present the results of their load-response analysis conducted during the summer, to review and evaluate the results, and to discuss how best to synthesize the results to produce ensemble guidance.

Eutrophication Response Indicators

The Objectives and Targets Task Team, with input from the modeling team, selected four ERIs and defined one or more metrics for each in terms of how it is measured and what spatial and temporal scale will be used for that metric measurement. They are presented below:

(1)   **Overall trophic status**

- Basin-specific, summer average chlorophyll *a* concentration (µg/L)
- Basin-specific, spring TP concentration

These are traditional indicators of lake trophic status (i.e., oligotrophic, mesotrophic, eutrophic) in lakes. The analysis here involves determining if phosphorus load restrictions for meeting the other ERIs will reduce the trophic status to a level that cannot support the important Lake Erie fishery.

(2)   **Cyanobacteria blooms in the Western Basin**

- maximum 30-day Western Basin cyanobacteria biomass (metric tons (MT))

This metric gives an indication of the worst condition relative to cyanobacteria blooms in the Western Basin. It represents the 30-day period in the summer when the cyanobacteria biomass in the Western Basin is the highest. We assume that this will be when the largest impacts, such as production of microcystin, will be most severe for that summer.

(3)   **Hypoxia in hypolimnion of the Central Basin**

- Number of hypoxic days
- Average areal extent during summer
- Average hypolimnion dissolved oxygen (DO) concentration during August and September.

All three of these metrics are quantitatively correlated based on Central Basin monitoring and analysis, but they are different manifestations of the problem; and each has a bearing on the assessment of the impact on the ecosystem (especially fish communities) and on the relative impact of physical conditions and nutrient-algal growth conditions on the indicator.

(4)   **Cladophora in the nearshore areas of the Eastern Basin**

- Algal dry weight biomass
- Tissue Phosphorus concentration

While beach fouling by sloughed *Cladophora* is likely the most important metric, there is neither an acceptable monitoring program to measure and report progress, nor a scientifically credible model to relate it to nutrient loads and conditions. There are models that can relate *Cladophora* growth to ambient dissolved reactive phosphorus (DRP) concentration; however, there was insufficient time and resources available to develop a site-specific model that could relate soluble reactive phosphorus (SRP) concentrations along the north shore of the Eastern Basin to open-lake boundary conditions and land-side loads. Therefore, a generic calculation was done to estimate the potential direction and rough magnitude of the *Cladophora* response to load reductions driven the cyanobacteria and hypoxia ERIs.

Model Selection

During the first workshop, models capable of addressing each of the ERIs were identified. The following criteria were used to assess the ability of each modeling effort to address the goals of the analysis:

Ability to develop load-response curves and/or provide other output important for quantitative understanding of the questions/requirements posed in Annex 4: A key function of the models used in this effort was to establish relationships between phosphorus loads and the metric defined by the Annex 4 subgroup for each objective. As such, models were evaluated as to their ability to establish load-response curves as the highest priority. Other models were also evaluated as to their utility to provide additional information to help understand dynamics, justify relationships, or otherwise inform the response curves or targets.

Applicability to objectives/metrics provided by the Annex 4 subgroup: The models were evaluated as to their ability to address the specific spatial, temporal, and kinetic resolution characteristics of the objectives and metrics outlined by the Annex 4 subgroup. While models that address other objectives and metrics can be additionally informative, the highest priorities are those that can address them directly.

Extent/quality of calibration and confirmation: Calibration - Given the broad range in model type and complexity, a wide range of skill assessments was used. Models were evaluated as to their ability to reproduce state-variables that match the objective metrics, as well as internal process dynamics. Post-calibration testing – Models were also measured against their ability to replicate conditions not represented in the calibration data set.

Extent of model documentation (peer review or otherwise): Models were evaluated based on the extent of their documentation. Full descriptions of model kinetics, inputs, calibration, confirmation, and applications were expected. This could be done through copies of peer reviewed journal articles, government reports, or other documentation, but it was required to be in writing.

Level of uncertainty analysis available: Models were evaluated as to the extent they are able to quantify aspects of model uncertainty, including uncertainties associated with observation measurement error, model structure, parameterization, and aggregation, as well as uncertainty associated with characterizing natural variability.

The models selected on the basis of these criteria are summarized in Table 2 below and briefly described after that.

Table 2. List of ensemble models and the Eutrophication Response Indicator that each is capable of addressing.

Model	Eutrophication Response Indicators			
	Overall Trophic Status	**Western Basin cyanobacteria blooms**	**Central Basin hypoxia**	**Eastern Basin Cladophora (nearshore)**
NOAA Western Lake Erie cyanobacteria blooms (Stumpf)		X		
U-M/GLERL Western Lake Erie cyanobacteria blooms (Obenour)		X		
TP Mass Balance Model (Chapra, Dolan, and Dove)	X			
1-D Central Basin Hypoxia Model (Rucinski)	X		X	
Ecological Model of Lake Erie (EcoLE) (Zhang)	X		X	
9Box model (McCrimmon, Leon, and Yerubandi)			X	
Western Lake Erie Ecosystem Model (LimnoTech)	X	X		
ELCOM-CAEDYM (Bocaniov, Leon, and Yerubandi)	X		X	
Great Lakes *Cladophora* model (Auer)				X

Model Descriptions

Models used in the ensemble effort are described briefly here. See Appendix B of the full modeling report (Scavia and DePinto 2015) for more in depth information about each model application.

NOAA Western Lake Erie Cyanobacteria Blooms Model (Stumpf) - In Stumpf *et al.*(2012) the authors present an empirical regression between spring TP load and flow from the Maumee River and maximum 30-day cyanobacteria index (CI) for Western Lake Erie as calculated by the European space satellite, MERIS. This method applies an algorithm to convert raw satellite reflectance around the 681 nm band into an index that correlates with cyanobacteria density. Ten day composites were calculated by taking the maximum CI value at each pixel within a given 10-day period to remove clouds and capture areal biomass. The authors conclude that spring flow or TP load can be used to predict bloom magnitude. Maumee River TP load from March to July was used as a predictor of CI utilizing data from 2002 to 2013.

U-M/GLERL Western Lake Erie Cyanobacteria Model (Obenour) - A probabilistic empirical model was developed to relate the size of the Western Basin cyanobacteria bloom to spring bioavailable phosphorus loading (Obenour *et al*. 2014). The model is calibrated to multiple sets of bloom observations, from previous remote sensing and in situ sampling studies. A Bayesian hierarchical framework is used to accommodate the multiple observation datasets, and to allow for rigorous uncertainty quantification. Furthermore, a cross validation exercise demonstrates the model is robust and would be useful for providing probabilistic bloom forecasts. The model suggests that there is a threshold loading value, below which the bloom remains at a baseline (i.e., background level). However, this may be an artifact of the lack of sufficient cyanobacteria observations at low loads. Above this threshold, bloom size increases proportionally to phosphorus load from the Maumee River. The model includes a temporal trend component indicating that response for a given load has been increasing over the study period (2002-2013), such that the lake is now more susceptible to cyanobacteria blooms than it was a decade ago.

Total Phosphorus Mass Balance Model (Chapra, Dolan, and Dove) - Chapra and Dolan (2012) have developed an update to the original TP mass balance model that was used (along with other models) to establish phosphorus loading targets for the 1978 Great Lakes Water Quality Agreement. Annual TP estimates were generated from year 1800 to 2010. The model is designed to predict the annual average concentrations in the offshore waters of the Great Lakes as a function of external loading and does not attempt to resolve finer-scale temporal or spatial variability. For Lake Erie this model computes the basin-wide annual average TP concentration as a function of all external TP loads. Calibration data for this model were obtained from Environment Canada and the United States Environmental Protection Agency's Great Lakes National Program Office. The model can be expanded to include chlorophyll *a* and potentially Central Basin hypoxia through available empirical relationships with TP.

1-Dimensional Central Basin Hypoxia Model (Rucinski) - A one-dimensional model with 50 0.5-meter layers, calibrated to observations in the Central Basin of Lake Erie, was used to develop load-response curves relating chlorophyll *a* concentrations and hypoxia to phosphorus loads to the Western Basin and Central Basin. The model is driven by a 1D hydrodynamic model that provides temperature and vertical mixing profiles (Rucinski *et al*. 2010). The biological portion of the coupled hydrodynamic-biological model incorporates phosphorus and carbon loading, internal phosphorus cycling, carbon cycling (in the form of algal biomass and detritus), algal growth and decay, zooplankton grazing, oxygen consumption and production processes, and sediment interactions. The model has been applied to 19 years (1983 – 2003) of physical conditions to understand the relative contribution of stratification conditions versus phosphorus loading on the severity of hypoxia in the Central Basin.

Ecological Model of Lake Erie - EcoLE (Zhang) - Zhang *et al*. (2008) developed and applied a 2D hydrodynamic and water quality model to Lake Erie termed the Ecological Model of Lake Erie (EcoLE), which is based on the CE-QUAL-W2 framework. The purpose of the model application was to estimate the impact that Dreissenids are having on phytoplankton populations. The model was calibrated against data collected in 1997 and verified against data collected in 1998 and 1999. Model results indicate that Dreissenid mussels have weak direct grazing impacts on algal biomass and succession, while their indirect effects through nutrient excretion have much greater and profound impacts on the system (Zhang *et al*. 2011). The model can produce load-response curves for chlorophyll *a* concentrations and dissolved oxygen.

Nine-Box model (McCrimmon, Leon, and Yerubandi) - This model is a coarse grid (9-box) phosphorus mass balance model for quantitative understanding of Lake Erie eutrophication and related hypoxia (Lam *et al*. 1987; Lam *et al*. 2002). The model is extensively calibrated and validated against observations in the past. Re-calibrations were conducted for post-zebra mussel period and found that 9-box model is able to express offshore Lake Erie concentrations reasonably well. The model can be expanded to include empirically derived chlorophyll *a* relationships for given TP concentrations.

Western Lake Erie Ecosystem Model (WLEEM) (LimnoTech) - The Western Lake Erie Ecosystem Model (WLEEM) has been developed as a 3D fine-scale, process-based, linked hydrodynamic-sediment transport-advanced eutrophication model to provide a quantitative relationship between loads of water, sediments, and nutrients to the Western Basin of Lake Erie from all sources and its response in terms of turbidity/sedimentation and algal biomass (three different phytoplankton functional groups, including cyanobacteria are modeled separately). The model operates on a daily time scale and can produce time series outputs and spatial distributions of either total chlorophyll and/or cyanobacteria biomass as a function of loading. Therefore, it can produce load-response plots for several potential endpoints of interest in the Western Basin. The Western Basin model domain is bounded by a line connecting Pointe Pelee with Marblehead. It can also produce mass balances for the Western Basin for any one of its ~30 states variables; therefore, it has computed the daily loading of Western Basin nutrients and oxygen-demanding materials to the Central Basin as a function of loads to the Western Basin. This provided valuable information on how load reductions to the Western Basin impact hypoxia development in the Central Basin.

ELCOM-CAEDYM (Bocaniov, Leon, and Yerubandi) - ELCOM-CAEDYM is a three-dimensional hydrodynamic and bio-geochemical model that consists of two coupled models: a three-dimensional hydrodynamic model - the Estuary, Lake and Coastal Ocean Model (Hodges *et al.* 2000), and a bio-geochemical model - the Computational Aquatic Ecosystem Dynamics Model (Hipsey and Hamilton 2008). The ELCOM-CAEDYM model has shown a great potential for modeling of biochemical processes and it has been successfully used for in-depth investigations into variable hydrodynamic and biochemical processes in many lakes all over the world including the Laurentian Great Lakes. In Lake Erie it has been used to study nutrient and phytoplankton dynamics (Leon *et al.* 2011; Bocaniov *et al.* 2014), the effect of mussel grazing on phytoplankton biomass (Bocaniov *et al.*, 2014), the sensitivity of thermal structure to variations in meteorological parameters (Liu *et al.*, 2014) and even winter regime and the effect of ice on hydrodynamics and some water quality parameters (Oveisy *et al.* 2014). The application of the ELCOM-CAEDYM model to study the oxygen dynamics and understand the Central Basin hypoxia is a subject of ongoing work. In this ensemble modeling process, ELCOM-CAEDYM is being applied to develop load-response curves for total chlorophyll *a* in all three basins and Central Basin hypoxia.

Great Lakes Cladophora model (Auer) - The Great Lakes *Cladophora* Model (GLCM) is a revision of the original *Cladophora* model developed by Auer and Canale in the early 1980s in response to the need to understand the causes of large *Cladophora* blooms around the Great Lakes, especially in Lake Huron (Tomlinson *et al.* 2010). The new model reflects current understandings of *Cladophora* ecology and a new set of tools and software to allow others to quickly run the model and view output. The updated model was calibrated and verified against data from Lake Huron (1979) and new data collected by the authors in 2006 in Lake Michigan. The model allows users to simulate standing crop of *Cladophora* as influenced by environmental parameters such as depth, light, and phosphorus concentrations. For this ensemble modeling process, the model is being applied generically to conditions in the Eastern Basin of Lake Erie.

Modeling Results

The models were applied to produce load-response curves for the ERI metrics that they were designed to model. This section summarizes the results of those applications as well as the decisions made in synthesizing those results to support the target load recommendations of the Objectives and Targets Task Team. A more complete presentation of the results of this ensemble modeling process is presented in Scavia and DePinto (2015).

In an ideal ensemble modeling process, each model would have been calibrated to the same data sets and driven by the same inputs (nutrient loads, meteorological drivers, etc.) to afford the opportunity to "average" results in forming the ensemble. However, given the limited time and resources available for this effort, the team relied primarily on existing models that were built and tested with a range of conditions. In general each model was run to produce load-response curves using 2008 as a baseline year and appropriately applying TP and DRP load adjustments to the conditions of that year. Load adjustments included 0%, 25%, 50%, 75%, 100%, and 125% of the 2008 loads. 2008 was used as a baseline year because it is the last year for which a thorough lake-wide, tributary-specific load compilation was produced (Dolan and Chapra 2012). It also had the most complete representation of necessary model input data in recent years. Finally, the 2008 whole-lake annual TP load was 10,675 MTA, which is very close to the Lake Erie target TP load of 11,000 MTA set in the 1978 Amendment to the GLWQA.

Western Basin Cyanobacteria Blooms

Three models were applied to simulate the phosphorus load effect on cyanobacteria blooms in the Western Basin of Lake Erie – one regression model calibrated to satellite-derived observations (Stumpf), one regression model calibrated to both satellite-derived and in-situ observations (Obenour), and one fine-scale process-based model calibrated to a broad set of in-situ observations (WLEEM, DePinto). All three models had the ability to produce a prediction of cyanobacteria biomass as a function of loading from the Maumee River although WLEEM included the ability to analyze the role of other Western Basin source areas, including the Detroit River.

The load-response curves for the three models, plotted as the maximum 30-day average biomass versus the March – July TP load are presented in figures 14, 15 and 16 for Stumpf, Obenour, and DePinto models, respectively. The detail of how these curves were developed is presented in Scavia and DePinto (2015). The WLEEM analysis also showed that the maximum 30-day average cyanobacteria biomass in the Western Basin – the metric used for this ERI – was well-predicted by the March – July TP load from the Maumee River and was not sensitive to reductions in other Western Basin tributaries, especially the Detroit River load. In all three plots a threshold below which was judged to be a non-severe bloom (lower maximum 30-day biomass than was observed in 2012) by the Objectives and Targets Task Team is shown as a green dotted line. Where the load-response curve intersects that line could be judged as a load target to be reached to minimize severe cyanobacteria blooms.
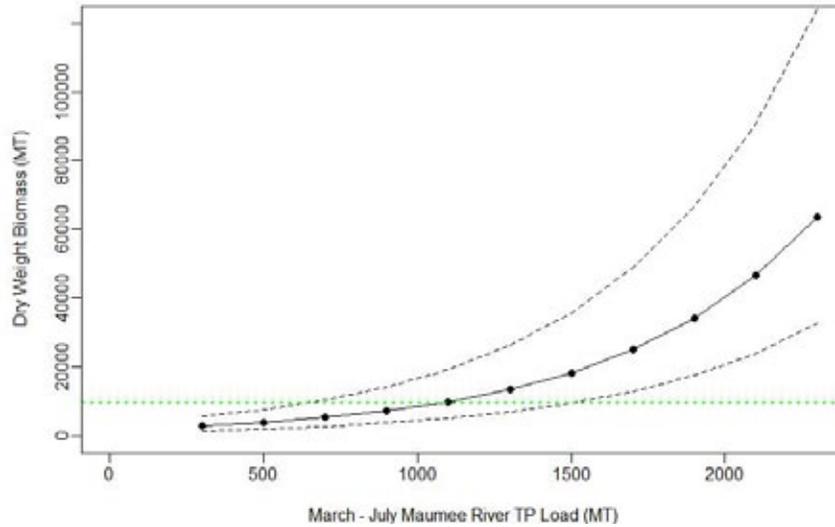
Figure 14. Cyanobacteria bloom size (peak 30-day average biomass) predicted by the Stumpf et al. (2012) model in the Western Basin as a function of spring Maumee River TP load. The solid line represents mean predictions, while dashed lines represent 95% prediction intervals. The green horizontal dotted line indicates the threshold for "severe" blooms.
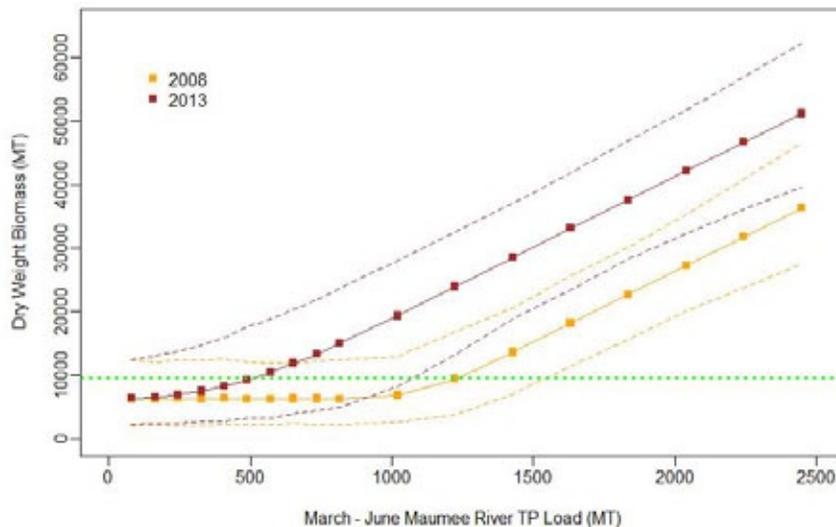
Figure 15. Cyanobacteria bloom size (peak 30-day average biomass) predicted by the Obenour et al. model in the Western Basin as a function of spring Maumee River TP load. Solid lines represent mean predictions under 2008 and 2013 lake conditions (see Appendix B-2 of Scavia and DePinto (2015) for details), while dashed lines represent 95% prediction intervals. The green horizontal dotted line indicates the threshold for "severe" blooms.
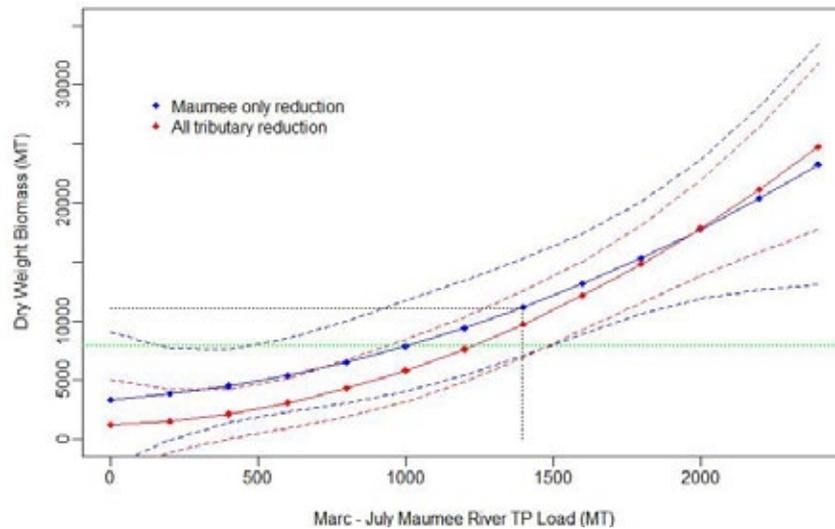
Figure 16. Cyanobacteria bloom size (peak 30-day rolling average biomass) as a function of March – July Maumee River TP load, predicted by WLEEM in the Western Basin in relation to changes in TP loads from all external sources (red lines) and from the Maumee River only (blue lines). Solid lines represent mean predictions, while dashed lines represent 95% prediction intervals around the regression fitted through results of model simulations using 2008, 2011-13 (Appendix B-7 of Scavia and DePinto (2015)). The black dotted line marks the 2008 TP load and corresponding bloom prediction. The green horizontal dotted line indicates the biomass level corresponding to a decrease in the 2008 bloom prediction equivalent to the percentage reduction of Stumpf`s 2008 predicted value down to the satellite data based value of 9600 MT. To account for this difference in calculation of this 30-day biomass value and to provide a bloom threshold consistent with WLEEM`s biomass estimates and equivalent to 9600 MT, we calculated the ratio of the Stumpf modeled 2008 biomass (our baseline year and also a severe bloom year) to the 9600 MT value, and then applied that ratio to the 2008 biomass predicted by WLEEM to obtain an "equivalent" threshold of 7990 MT cyanobacteria biomass for WLEEM (green line).

It should be noted that the Obenour model found an improved empirical fit to the observations if a time-dependent term was added to the regression. This model therefore suggests an increasing sensitivity of the Western Basin cyanobacteria to loading over time, hence the 2013 load-response curve in figure 16 being higher than the curve drawn for 2008. The Objectives and Targets Task Team felt that there was insufficient data to adequately assess the cause or persistence of this trend into the future, so the 2008-based forecast was used. However, this empirical trend could be significant and suggests target loads should be evaluated in an adaptive way as future data are added to the analysis.

Central Basin Hypoxia

Four models were used to develop load-response curves for the three metrics – number of hypoxic days, average areal extent during summer, and average hypolimnion dissolved oxygen (DO) concentration during stratification – determined to represent the Central Basin Hypoxia ERI. The models were: ELCOM-CAEDYM, Rucinski 1D hypoxia model, EcoLE, and the 9-box model.

All of these models and previous studies have shown that the hypolimnetic oxygen depletion rates in the Central Basin of Lake Erie are driven by both the sediment oxygen demand (SOD) and water column oxygen demand (WOD) and summer stratification. Since the full effect of nutrient load changes cannot be seen with short simulations of the models, SOD rates are adjusted to capture the nutrient load reductions (see Appendices B4-B8 in Scavia and DePinto, 2015).

To apply the hypoxia load-response curves, the hypoxia modeling team selected a 2000 km² threshold, the average extent of the Central Basin hypoxic zone, which represents a value typical of the early- to mid-1990s, which coincides with recovery of lessening hypoxia conditions. Based on the hypoxia-load response curve shown in figure 17 (the 9-box model was not incorporated into the hypoxic area ensemble because of its coarse spatial segmenting), this threshold corresponds to a range of loads between 3415 and 5955 MTA. Using the 2008 annual TP load of 9577 MTA, this target represents a range of 38 – 65% load reduction, depending on the model used.
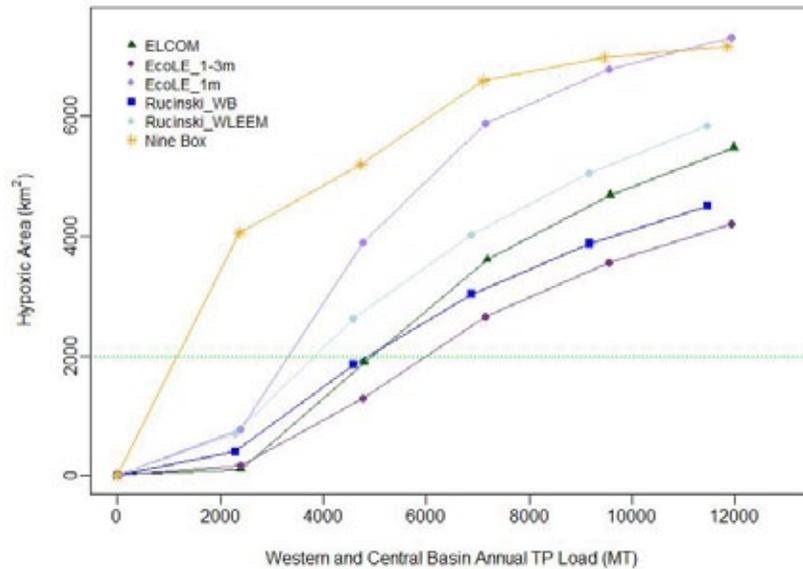
Figure 17. August-September average extent of the hypoxic area predicted by different models in the Central Basin as a function of annual TP loads to the Western and Central Basin. The green horizontal dotted line indicates a suggested threshold of 2000 km².

Figure 18 shows the response curves for hypolimnetic dissolved oxygen concentration from the various models. Each model shows a decreasing trend with increasing loads, although there is more separation among models at the lower loads. The models also estimated number of hypoxic days, but as discussed in the next section, the Objectives and Targets Task Team felt that the hypolimnetic DO concentration was a more appropriate metric for the range of hypoxia impacts of concern (Scavia and DePinto, 2015). The 9-Box model is the only model that was run for 3 consecutive years to approximate a steady-state response to the load reductions, and that could partially explain some of the differences in model results. Additional differences among model output include the fact that the Rucinski models and the 9-Box model report hypolimnetic averaged DO over the August-September, while the other two models report concentrations from the basins' bottom layer (0.5-1.0 m for ELCOM-CAEDYM; 1.0 and 1-3 m for EcoLE) for that period. Rucinski_WB and Rucinski_WLEEM use two representations of flux from the Western Basin to the Central Basin. In addition, all response curves were plotted as a function of Western + Central basin TP loads. Whenever necessary, whole lake loads originally reported by each modeler were converted to Western + Central basin loads based on the Western + Central load-to-whole lake load ratio recorded in the 2008 baseline year used by each modeler. As an example, we would anticipate a 6000 MTA load would produce mean DO concentrations between 2.2-3.5 mg/L.
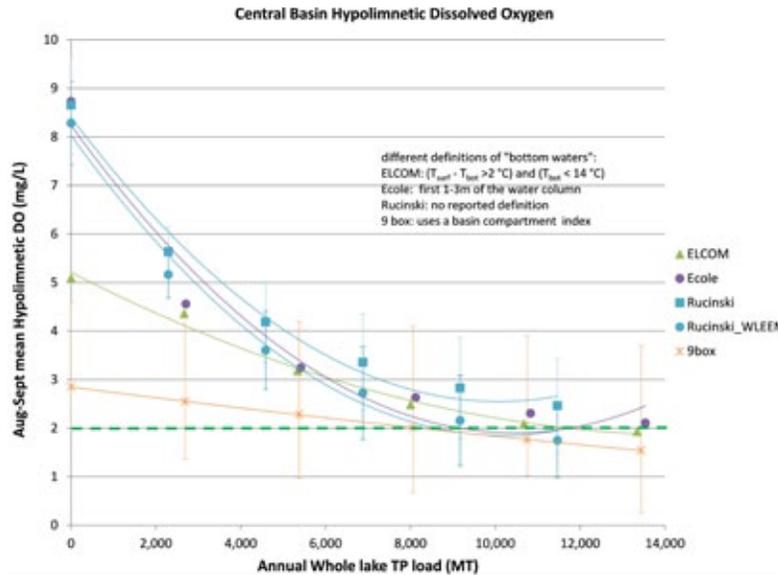
Figure 18. August-September average hypolimnetic dissolved oxygen concentration predicted by different models in the Central Basin as a function of annual TP loads. The green dotted line shows the annual whole-lake load to achieve a threshold of 2 mg/L for the various models.

Eastern Basin *Cladophora*

Despite the acknowledged difficulties of relating *Cladophora* biomass to Lake Erie phosphorus loads and the relative impact of open-lake boundary phosphorus concentrations and land-side phosphorus loads along the north shore of the Eastern Basin, a generic modeling analysis was performed to gain a relatively uncertain estimate of the relationship between TP loads and the *Cladophora* biomass metric (as described in Appendix B-9 of Scavia and DePinto, 2015). The analysis included an application of the Great Lakes *Cladophora* Model (GLCM) that relates standing stock *Cladophora* biomass to Eastern Basin spring DRP concentrations and two empirical relationships – one that relates DRP to TP concentrations, and one that relates Eastern Basin TP concentrations to TP loads. Given those relationships, a load-response curve relating *Cladophora* biomass to whole-lake TP annual load was developed (Figure 19). Auer, the *Cladophora* modeler on this ensemble team, suggested as an example a threshold value for the *Cladophora* biomass metric of 30 g/m^2 DW, the biomass below which nuisance algal growth in the Eastern Basin is eliminated. The model application indicates that an Eastern Basin spring DRP concentration of 0.9 µg/L P would be required to achieve this threshold. The empirical correlation analysis indicates that a 0.9 µg/L DRP concentration corresponds to a TP concentration of 6.3 µg/L P, and the Chapra TP model indicates that a 6.3 µg/L TP concentration in the Eastern Basin requires a total TP load for Lake Erie of 7000 MTA, or a 22% reduction from the 2002-2011 average. This total lake target represents a 35% reduction from the 2008 baseline year load of 10,675 MTA. While this computation suggests that cyanobacteria and hypoxia driven load targets are sufficient to achieve a desired reduction in *Cladophora* in the Eastern Basin, we are not confident in a number of relationships, including the DRP/TP ratio; hence the Objectives and Targets Task Team is not prepared to set a loading target for *Cladophora* in the Eastern Basin. Before doing that, it is strongly recommended that a site-specific model, including exchange with the open water, and load and transport of specific tributaries, including the Grand River (ON), and the role of Dreissenids on phosphorus cycling and availability in the nearshore, be developed for the north shore of the Eastern Basin to gain more confidence in this forecast.
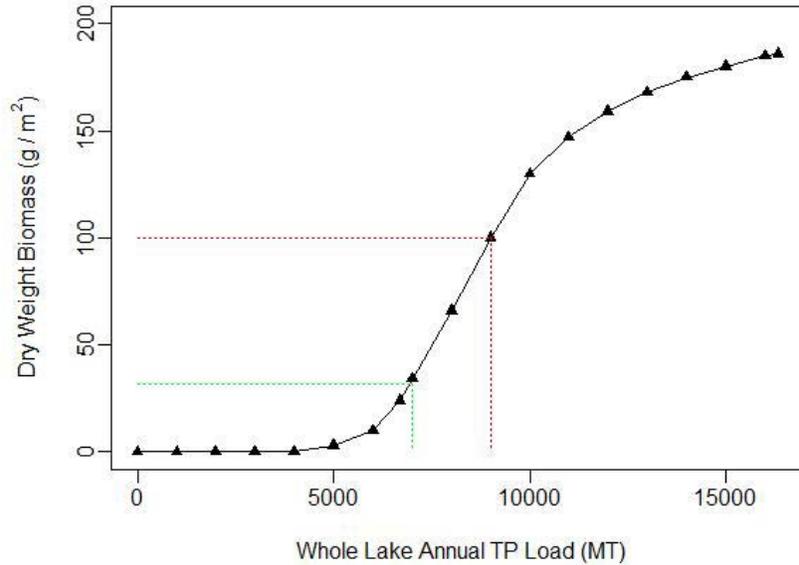
*Figure 19. **Cladophora** biomass predicted by the Great Lakes **Cladophora** model (GLCM) in the Eastern Basin as a function of annual TP load to the whole lake. The green dotted line represents the author's suggested TP loading target (7000 MT) to achieve a threshold biomass of 30 mg/L dry weight, while the red dotted line represents the average annual TP load to Lake Erie over the period 2002-2011 (9022 MT).*

Basin-specific Trophic Status

Three models – ELCOM-CAEDYM, EcoLE, and Chapra – provided load-response curves for TP and chlorophyll *a* for each of Lake Erie's three basins; WLEEM also made a similar computation for its model domain, the Western Basin. The Environment Canada 9-box model also provided basin-specific load-response curves for TP concentration as a function of TP load. In general, these models found a linear relationship between TP load and basinwide TP concentration, and an inverse hyperbolic relationship (i.e., a saturation function) between TP load and basinwide summer chlorophyll *a* concentration. Also, it was found that only the Western Basin required load reductions to assure that the basin would be in the mesotrophic range. However, the load-response curves for TP from the 9-box model and Chapra model are used to determine if the cyanobacteria and hypoxia load reduction recommendations produce basin-wide TP concentrations below the acceptable range for trophic level status recommended by the Great Lakes Fishery Commission to support the fishery.

5.   Task Team Recommendations for Eutrophication Response Indicator Thresholds, Corresponding Loading Targets, and Other Considerations

The Objectives and Targets Task Team derived the recommendations for phosphorus load management based on discussions and interpretation of the modeling load-response analyses. In doing so, the team decided that the phosphorus loading targets should be based primarily on the Western Basin cyanobacteria and Central Basin hypoxia ERI metrics. The Eastern Basin *Cladophora* and basin-specific trophic indicators have been used to check the response of these ERI metrics to the recommended load targets. The Objectives and Targets Task Team opted not to recommend targets for Eastern basin *Cladophora* due to insufficient information. The evaluation of Eastern Basin *Cladophora* and basin-specific trophic level metrics, as opposed to setting targets based on them, is due to lower confidence in modeled results and a concern over the potential impacts on the lake carrying capacity for a healthy fish community.

The Task Team discussed a number of eutrophication response indicators for the Western Basin cyanobacteria blooms and Central Basin hypoxia. For cyanobacteria blooms the Task Team selected a target phosphorus load designed to produce a mild bloom (<9600 MT), the size of that observed in 2004 or 2012, or smaller, 90% of the time.

The Task Team concluded that non-point source runoff from the Maumee River during the spring period of 1 March to 31 July each year was the best predictor of cyanobacteria blooms severity based upon the work of the modeling Sub-Team and loading data provided by the National Center for Water Quality Research (NCWQR). The Task Team used the 2008 water year as the base year for calculating reduction percentages. The scientific community considers phosphorus load measurements for 2008 accurate, and the 2008 whole-lake annual TP load was 10,675 MTA, which is very close to the Lake Erie target TP load of 11,000 MTA set in the 1978 Amendment to the GLWQA.

To achieve a bloom no greater than that observed in 2004 or 2012, 90% of the time, the Task Team recommends a total phosphorus spring load of 860 metric tons and a dissolved phosphorus load of 186 metric tons from the Maumee River. The 860 metric ton target is approximately a 40% reduction from the 2008 spring load of 1400 metric tons for TP and 310 metric tons of DRP, and the 2008-target load corresponds to a Flow Weighted Mean Concentration (FWMC) of 0.23 mg/L TP and 0.05 mg/L of DRP. Because discharge varies considerably from year to year, and because the discharge of the Maumee River was so large in 2008 that it has only been exceeded about 10% of the time in the last 20+ years, the Task Team expects that achieving a FWMC of 0.23 mg/L for TP and 0.05 mg/L for DRP will result in phosphorus loads below the targets (860 and 186 metric tons) 90% of the time (9 years out of 10), if precipitation patterns do not change.

 The Task Team also observed that smaller cyanobacteria blooms have been observed from satellite imagery at the mouths of the Thames River, River Raisin, Toussaint Creek, Portage River, and near Leamington. As a result, the Task Team concluded that while accurate loading data for all Western Basin tributaries are not available, it recommends a 40% load reduction for all of those tributaries as well.

 The Detroit River was not treated as a single source. Instead, the Task Team opted to address each of the tributaries flowing into the Huron-Erie Corridor (HEC) to avoid local cyanobacteria bloom issues at those tributary mouths, and, along with reductions to other sources in the HEC, including the Detroit WWTP, to address hypoxia in the Central Basin (discussed later in this report). The Detroit WWTP does not appear to contribute significantly to the cyanobacteria blooms in the Western Basin, but the Task Team encourages additional monitoring of the Detroit River loads and concentrations from each of the channels across the river from east to west. The Task Team recommends that all watershed coordinators in the HEC work to reduce tributary loads; however, because of their significant contributions to the total Detroit River load, priority should be given to reducing the loads from the Thames River and the Detroit WWTP.

While the discharge volume of the Maumee River is much smaller than the Detroit River, the concentrations of phosphorus coming out of the Maumee are often about 25 times greater than those of the Detroit River.

Table 3 below summarizes the recommendations of the Objectives and Targets Task Team for phosphorus load reduction targets in Lake Erie. The approach and rationale for arriving at each of these recommendations is discussed in the remainder of this section.

Table 3. Summary of phosphorus load targets recommended to achieve desired thresholds for Eutrophication Response Indicators in Lake Erie.

	Spring (Mar-July)	Annual
Western Basin Cyanobacteria – Bloom biomass less than or equal to 2004 or 2012 9 years out of ten, and/or reduce risk of nearshore localized blooms		
Maumee River		
Total Phosphorus load	860 MT[#]	
Dissolved Reactive Phosphorus load	186 MT[#]	
Other Western Basin Tributaries and Thames River		
Total Phosphorus load	40% reduction*	
Dissolved Reactive Phosphorus load	40% reduction*	
Central Basin Hypoxia – Aug –Sept Average Hypolimnetic Oxygen of 2 mg/L or more		
Total Phosphorus load to Western and Central Basins, including Detroit River and atmospheric load)		6000 MT**
Eastern Basin *Cladophora* – insufficient information to establish target		

[#]to be met 90% of the time based on inter-annual flow variability for the March-July period.
*Note: Percent reductions are based on 2008 loads
**This represents a 40% reduction of annual loads to the Western and Central Basins, including the Detroit River and atmospheric load.

Western Basin Cyanobacteria Blooms

The metric for the cyanobacteria ERI is the maximum 30-day Western Basin average cyanobacteria biomass. This metric was selected because it represents the worst conditions relative to cyanobacteria blooms in terms of its impact on aesthetics, recreation, fish community, and cyanotoxin production. Also, bloom sizes over the past dozen years have varied considerably and been closely related to external phosphorus loading to the Western Basin from the Maumee River in the March – July period. During that time, the smallest bloom (quantified by our cyanobacteria metric) occurred in 2012 when the March – July TP loading from the Maumee River was the smallest. No significant cyanobacteria related impacts were noted, with the exception of some bloom conditions in inner Maumee Bay. Hence, the Objectives and Targets Task Team determined that a reasonable threshold to limit the cyanobacteria metrics would be at a level below the 2012 biomass value, which was about 9,000 MT of cyanobacteria in the Western Basin for the same period of the year.

Based on the load-response curves from the models presented above, **the cyanobacteria biomass threshold would be met if the March – July TP load from the Maumee River was below 860 MT and the DRP load for the same period was below 186 MT, based on the average fraction of DRP/TP over the past ten years**. It is worth noting that this target is virtually the same as determined from both the Stumpf and Obenour models (relatively simple empirical models) and WLEEM (complex process-based model). The fact that two very disparate model frameworks give the same load-response relationship gives us added confidence in making this recommendation.

In meeting the TP load target, it is important to focus on reductions of both major fractions of TP, namely particulate phosphorus (PP) and dissolved reactive phosphorus (DRP). DRP is virtually 100% bioavailable and represents about 20% of the total load. PP is between 25-50% bioavailable and represents ~80% of the TP load. Therefore, reductions to DRP will be more efficient at reducing bioavailable P, which suggest that BMPs focusing on DRP should be given a higher priority than BMPs focusing on PP. However, the models conclude that totally eliminating DRP without changing the PP load will not by itself solve the problem. Hence, there is a need to reduce both DRP and TP loads; and it would be prudent to aim for equal percent reductions of both. Of course, it should be recognized that different nonpoint source reduction BMPs will emphasize different TP components; therefore, establishment of NPS reduction actions should account for this reality.

Because the spring flows (frequency and magnitude of high flow runoff events) in the Maumee River are so variable inter-annually, it is not reasonable to expect to meet the target TP load every year. Based on the spring flows and associated TP loads over the period from 1990 - 2013, the March – July flow has been below 4×10^9 m³ 90% of the time; only in 2011 (the unusually high spring load and corresponding unprecedented bloom) and in 2003 was the spring flow larger. Hence, we recommend that the spring Maumee TP load target be met for all cumulative spring flows of 4×10^9 m³ or less. **Meeting an 860 MT TP load target at a total March-July Maumee flow of 4×10^9 m³ amounts to a attaining a 40% Mar-July TP load reduction relative to 2008**. This would, of course, imply that at lower spring flows, lower TP loads would be expected.

While the WLEEM application confirmed that the Maumee River is the overwhelmingly dominant source of phosphorus causing cyanobacteria blooms in the Western Basin, the Objectives and Targets Task Team concluded that some benefits relative to the Lake Erie Objectives for a healthy nearshore algal community will be derived from P load reductions for the other tributaries to the Western Basin, including those that flow into the HEC. These tributaries are included in the load reduction target for Western Basin cyanobacteria blooms because: 1) they do contribute, albeit small quantities, to the Western Basin cyanobacteria biomass, 2) they have the potential to generate local river mouth cyanobacteria problems not part of the main Western Basin bloom but do represent nearshore algal problems, and 3) they contribute to the total Western plus Central basin load that drives the Central Basin hypoxia.

As discussed previously, the Detroit River is a blend of Lake Huron load, tributary and direct drainage loads, and point sources (including the Detroit Waste Water Treatment Plant). The Task Team concluded that the vast majority of the phosphorus load to the HEC comes from the Thames River, the Detroit WWTP and the outflow from Lake Huron (the Task Team does not address reduction of the load coming out of Lake Huron. The phosphorus load coming from Lake Huron was approximately 320 MT in 2008 out of a total Detroit River load of 1,987 MT). As a result of the large volume of water in the Detroit River, the concentration of TP and DRP in the Detroit River flow into the Western Basin is low – on the order of 20 µg/L TP – and is insufficient to cause cyanobacteria blooms (Downing, *et al*., 2001). Consequently, the Detroit River load is not treated as a separate single loading source in setting a Western Basin cyanobacteria target load. This makes it much more manageable because each of the loads to the corridor, including the Thames River which is a priority watershed, can be treated separately. Based on cyanobacteria blooms occurring at the mouth of the Thames River in Lake St. Clair, the Task Team recommends a 40% reduction in the spring load of phosphorus from the Thames relative to the 2008 spring load. The Task Team also recommends 40% reductions relative to 2008 in *annual* phosphorus loads from the Detroit WWTP.

Given the above considerations, the Objectives and Targets Task Team recommends the following target loads for the HEC and Western Basin tributaries:

> Apply 40% reduction relative to the 2008 loads (equivalent percentage to that determined for the Maumee River) for TP and DRP for Mar-July, unless there are local monitoring, modeling, and management plans in place that will mitigate/reduce/minimize risk of local blooms during the March through October growing season.

The target of 40% reduction will apply unless there is a tributary program in place that includes, monitoring, modeling and/or management plans. A tributary program should demonstrate that the tributary and river mouth nutrient conditions do not pose a cyanobacteria threat to adjacent nearshore water.

The purpose of allowing a program in place as an alternative to meeting a 40% reduction is to allow for flexibility in local conditions. The Task Team believes this approach acknowledges one or more of the following situations:

- Many small tributaries lack data that would identify the extent to which a tributary is (or is not) contributing to phosphorus loading. This approach may instigate data collection efforts to provide program managers with the information and data to determine if reduction efforts are necessary.
- Nutrient management efforts may be in place that are effectively managing nutrient runoff.

The Task Team recommends that the Domestic Action Plans to be developed by the Parties define the thresholds to allow for an alternative to the 40% reduction. The metrics defining a potential source could include the following: 1) a microcystin-LR concentration exceeding 1.0 ug/L; and, 2) a cyanobacteria cell count exceeding 20,000 cells/mL in any surface-collected water sample taken over the period of May to October, with a significant sampling effort between July and September. The river mouth zone will be defined as the confluence of either i) the tributary mouth, or, ii) tributary-connected embayment with the open waters Lake Erie (Chorus and Bartram 1999). Additional thresholds could be defined for nutrient data collection, modeling analyses and management plans.

Finally, the Objectives and Targets Task Team recommends using tributary flow-weighted mean concentrations (FWMC) as a benchmark to track progress in load reduction (see above discussion in the load characterization section). FWMC can be calculated for tributaries by dividing the phosphorus load during a specified period (e.g., annually or March-July) by the cumulative flow during that period. FWMC offers the advantage of normalizing the phosphorus delivery from a tributary with respect to flow, so that year-to-year performance is not confounded by inter-annual variability in hydrology. For the Maumee River 2008 water year, the FWMC for TP = 358 µg/L P and for DRP = 78 µg/L P. If the target load were achieved with proportional decreases to TP and DRP, the target FWMC would be TP = 215 µg/L P and DRP = 47 µg/L P. This comparable calculation can be made for all Western Basin tributaries assuming phosphorus loads and flows have been monitored.

Central Basin Hypoxia

Several of the models described above use differing approaches to compute the relationship between external phosphorus loading to Lake Erie and the magnitude of hypoxia in the Central Basin. The phosphorus loading effect on hypoxia is expressed both through a water column oxygen demand from decomposition of phytoplankton settling into the hypolimnion and through sediment oxygen demand from the deposition of oxygen-demanding organic material (e.g., dead phytoplankton) settled to the surface bottom sediments. All models and data suggest that the best load-response relationship is derived from the annual load to the Western Basin + Central Basin because of their combined effect on phytoplankton production in the Central Basin, regardless of when that load is input. The models also emphasize the importance of the physical factors (temperature, wind, currents) that have a significant impact on the stratification timing and magnitude and the volume of the associated hypolimnion. This creates potentially significant variability in hypoxia as a function of a given phosphorus loading profile. The effect of variable physical factors is demonstrated by the application of the Rucinski 1D hypoxia model to the actual physical conditions that were observed for the years of 1987 – 2005 (figure 20). This plot shows the variation of August-September hypoxic area for a given annual Western Basin + Central Basin TP load. This demonstrates approximately a factor of two variability in hypoxic area (also mean hypolimnion DO concentration – not shown) for a given TP load due to variation in physical conditions.
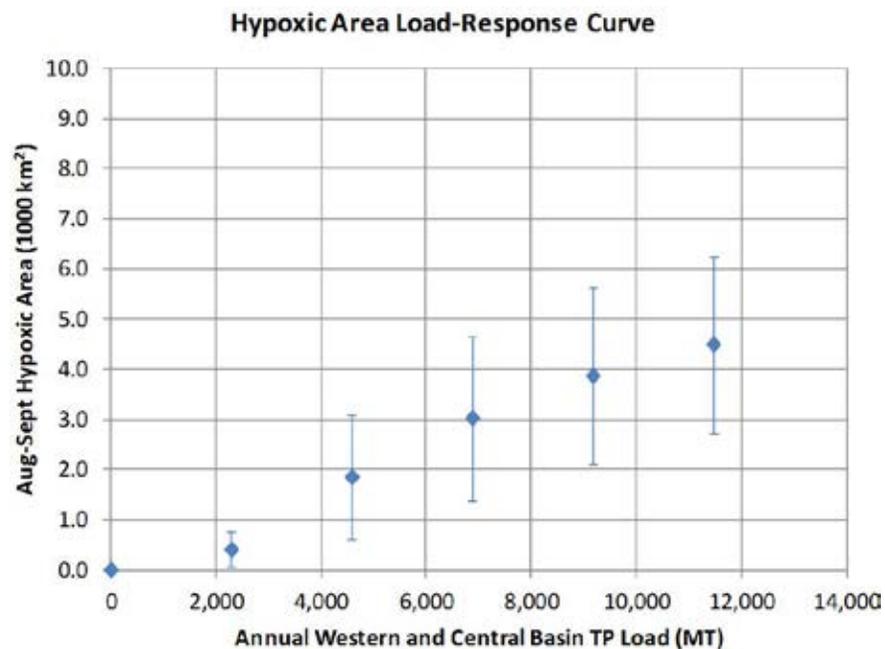
Figure 20. Aug-September average hypoxic area load-response curve computed by Rucinski 1D Hypoxia Model. Mean of 1987-2005 model estimates values shown as blue diamonds. Standard deviation of 1987-2005 values shown as vertical error bars.

As a result of the effects of physical factors and the difficult morphometric characteristics of the Central Basin (i.e., shallow basin yet deep enough to form a well-defined, yet thin hypolimnion during the stratified period), it is likely impossible to eliminate hypoxia and a given phosphorus load will lead to a range of hypoxia severity depending on weather and lake level. Our goal is therefore to "reduce" hypoxia while maintaining or improving fish production.

With that goal in mind, it was determined to establish a threshold for one of the hypoxia ERI metrics and then to estimate the TP load target that the ensemble models recommend for achieving that threshold. While the typical definition of hypoxia is for hypolimnion dissolved oxygen concentrations below 2 mg/L, Zhou *et al*. (2013) showed that statistically significant hypoxic areas begin for average bottom water DO concentrations below approximately 4.0 mg/L. However, the Objectives and Targets Task Team concluded that reaching a threshold hypolimnion average DO concentration of 4.0 mg/L could not be reliably attained because of the Central Basin morphometry and variability in physical conditions. Also, the Task Team felt that the load reduction necessary to achieve a DO threshold of 4.0 mg/L was too stringent and might impact the fish carrying capacity of the Central Basin. Therefore, the Objectives and Targets Task Team consensus was to set a threshold of hypolimnion August-September average DO at or above 2.0 mg/L. The data and models suggest that this threshold will have several benefits in the Central Basin, including:

   •   Reduction of internal phosphorus loading due to decrease of sediment anoxia
   •   Improvements to the benthic community in the Central Basin
   •   Benefits to drinking water intakes by reducing the release of taste impacting heavy metals such as manganese
   •   Reduction of cyanobacteria blooms in the Central Basin

As indicated in Figure 19 (DO load-response curves) above, this threshold is achievable with a less significant load reduction than that required to achieve a hypoxic area of 2000 km². This figure indicates that at a Western Basin + Central Basin annual TP load of 6000 MTA, all models will give >2.0 mg/L hypolimnion average DO. **Because the baseline 2008 annual TP load to the Western + Central Basin is 9577 MTA, application of a 40% annual load reduction relative to 2008 to all Western Basin (including those discharging into the Huron-Erie corridor) and Central Basin tributaries will meet the desired 6000 MTA load.**

Eastern Basin Cladophora

It is clear that phosphorus load reductions will be necessary to reduce nuisance levels of *Cladophora* in the nearshore waters of the Eastern Basin of Lake Erie. A combination of phosphorus from the offshore, phosphorus loading from tributaries and phosphorus recycled through Dreissenid mussels interact to determine the nearshore concentrations that sustain the current nuisance levels of *Cladophora* growth. At present, models capable of predicting the integrated phosphorus loading required to reduce *Cladophora* biomass to an acceptable level are not yet sufficiently developed. Nevertheless, *Cladophora* growth is fundamentally limited by phosphorus availability, and thus concentrations of DRP in the nearshore and offshore waters will need to be low. Therefore, a need to reduce Eastern Basin phosphorus loads from both tributaries (and the direct shoreline) and offshore inputs are anticipated. While reductions in phosphorus loads from Eastern Basin tributaries are expected to result in reductions of localized *Cladophora* blooms; the Task Team is unable to make specific recommendations for the size of reductions without additional research. Efforts to reduce P loads to the Western and Central basins will help to reduce Eastern Basin offshore concentrations, but additional reductions in loads from Eastern Basin tributaries will be necessary to reduce nearshore levels of P. Immediate efforts by watershed coordinators to reduce phosphorus loading from Eastern Basin tributaries (such as those being undertaken in the Grand River watershed) are highly encouraged and will be critical to achieving phosphorus reductions in the nearshore areas affected by local enrichment. Therefore, implementation of existing watershed plans should be expedited.

Additional research is recommended to determine the importance of watershed phosphorus sources versus the open lake boundary in relation to *Cladophora* growth. Three critical research questions have been identified to this end:

- Which features affecting the nearshore environment most strongly determine (and need be quantified to predict) the mass transport of DRP supporting *Cladophora* growth?
- What is the magnitude and relevance of DRP supply mediated by Dreissenids for *Cladophora* growth? What impact will P load reductions have on Dreissenids and will these feedbacks have to be factored into the modeled *Cladophora* response?
- What form should *Cladophora*-P load response models take considering the range of factors that may be important drivers of P supply (e.g., Dreissenid mussels, boundary layer effects, variable nearshore conditions)?

To support these science gaps, a monitoring, research, and modeling program should be begun to develop a site-specific load-response model for the north shore of the Eastern Basin. This program will not only address the above research questions but it will include monitoring information to track the severity and features of the *Cladophora* problem in the Eastern Basin. This information will be critical to assess the effectiveness of proposed P load reductions for the Western and Central basins, and any subsequently developed for the Eastern Basin tributaries. The overall outcome of this program would be to improve our predictive capability for *Cladophora*-P load responses, both in Lake Erie and in other lakes (Lake Ontario, Lake Huron, Lake Michigan) exhibiting similar problems.

A more detailed discussion of the Eastern Basin north shore *Cladophora* issue is presented in Appendix A.

Basin-Specific Trophic Status

A key concern regarding the implementation of the above loading targets to address Western Basin cyanobacteria blooms and Central Basin hypoxia are related to its effect on the lake's overall basin-specific trophic status and resulting carrying capacity for a healthy and diverse fish community. The Objectives and Targets Task Team consulted with the Great Lakes Fisheries Commission-Lake Erie Committee (LEC) and they offered the following preliminary perspective:

- First, the LEC does think there is the potential for hypoxia in the central basin to have direct negative impacts on coldwater fish like Lake Whitefish. Large anoxic areas, coupled with increased temperatures put these fish into a temperature-dissolved oxygen "squeeze". Therefore, dealing with hypoxia in the Central Basin can benefit the fish community there.
- A 40% reduction in tributary loads makes sense, as long as it does not cause in-lake P concentrations to be significantly below the mesotrophic target for the Western Basin and Central Basin preferred by the LEC (10-15 μg/L P).
- The LEC is strongly supportive of an adaptive management approach to understanding the effects of targets once they are implemented and revising targets if necessary.

We have applied the Chapra model to evaluate the impact of the recommended phosphorus load reduction strategy on basin-specific TP concentration as a metric of the lake basin trophic status. The Chapra model has been applied to compute the steady-state TP concentration for the three basins once the load targets have been achieved. A summary of that application is presented in Table 4 below.

Table 4. Estimated loads and TP concentrations for four different scenarios: 2008 load, 40% reduction in annual TP load (in metric tons per annum) to each Basin relative to 2008, 40% reduction in annual TP load for Western and Central Basins only, and 40% reduction in annual TP load from Western Basin tributaries, including tributaries to the Huron Erie Corridor.

Estimated Loadings by Basin (MTA)

Scenario	Huron	Western Direct	Western	Central	Eastern	Lake Erie Total
2008 Load	321	6368	6689	2888	1098	10675
40% reduction from all tribs	321	3821	4142	1733	659	6533
40% reduction from Western and Central Basin tribs	321	3821	4142	1733	1098	6973
40% reduction from Western Basin tribs	321	5071	5392	2888	1098	9378

Estimated TP concentrations by Basin (μg/L P)

Scenario	Western	Central	Eastern	Ontario
2008	19	9	8	6
40% reduction from all tribs	12	6	5	6
40% reduction from Western and Central Basin tribs	12	6	6	6
40% reduction from Western Basin tribs	15	8	8	6

These model results compare quite well with those of the Environment Canada 9-box model when similar load reductions are applied. Both models suggest that there will be no significant changes in trophic status of Lake Erie basins, except perhaps in the Central Basin if a 40% total basin load reduction is achieved for both the Western Basin and Central Basin. The Chapra model computes that under that loading regime the annual average TP in the Central Basin would be reduced from 9.4 to 5.9 μg/L P (a 38% reduction). The 9-box model computes a Central Basin April TP concentration of 5.7 μg/L P given the same load reduction applied by the Chapra model.

The work of Chapra provides the added analysis of the impact of phosphorus load reductions in Lake Erie on the Lake Ontario TP concentration. As indicated in the above table the TP concentration in Lake Ontario will decrease from a 2008 steady-state baseline value of 6.3 µg/L P to 5.5 µg/L P under a 40% annual TP load reduction to the entire Lake Erie. This is because under the baseline load conditions of 2008 for both lakes the Lake Erie load via the Niagara River contributes to about 30% of the Lake Ontario annual average TP concentration.

Priority Watersheds

The following table provides the priority watersheds identified by the Task Team by basin and Eutrophication Response Indicator.

Table 5. Priority Watersheds by Basin and Eutrophication Response Indicator

Tributary	Basin			Eutrophication Response Indicator			
	Western Basin	Central Basin	Eastern Basin	Cyanobacteria: 40% Spring P Reduction		Central Basin Hypoxia: 40% annual P reduction	Eastern Basin *Cladophora*: *insufficient science to establish P reduction target at this time
				Western Basin Open-water	Nearshore		
Detroit River	X					X	
Thames River	X				X	X	
Leamington Tribs	X			X	X	X	
River Raisin	X			X	X	X	
Maumee River	X			X	X	X	
Portage River	X			X	X	X	
Toussaint Creek	X			X	X	X	
Sandusky River		X			X	X	
Huron River (Ohio)		X			X	X	
Vermillion River		X				X	
Cuyahoga River		X				X	
Grand River (Ohio)		X				X	
*Grand River (Ontario)**			X				X
*Cattaragus Creek**			X				X

The Detroit River often brings about 90% of the water into Lake Erie and a large load of phosphorus. However, the Detroit River discharge is composed of the discharge from Lake Huron and discharges from a number of other tributaries and wastewater treatment plants along the Huron Erie Corridor (HEC). Overall the largest sources of phosphorus in the HEC are the Thames River and the Detroit WWTP. The phosphorus load out of Lake Huron is very low compared to the size of the discharge, resulting in a very low phosphorus concentration. For that reason, the Task Team chose not to prioritize the phosphorus load from Lake Huron. The Team believes priority should be given to reducing loads from the Thames River because of the size of the load and because of the cyanobacteria blooms at the mouth of the river that can be traced along the north shore of Lake Erie. The Detroit WWTP contributes to Central Basin hypoxia, but does not appear to be a significant contributor to Western Basin cyanobacteria blooms. Reductions in loads from other HEC tributaries are strongly encouraged, but those tributaries are not as high a priority.

The Sandusky River flows into Sandusky Bay that empties into the Central Basin. It carries a large phosphorus load and is an obvious priority to reduce Central Basin hypoxia. However, the cyanobacteria blooms that occur annually in Sandusky Bay start the earliest, last the longest, and reach the greatest algal cell densities of any in Lake Erie. For this reason, while not contributing to Western Basin cyanobacteria blooms, the Task Team believes that spring reductions in P loads should be a priority for this watershed in addition to annual reductions.

The Huron River (Ohio) flows into the Central Basin and does not contribute to Western Basin cyanobacteria blooms. However, satellite images show that a small cyanobacteria bloom forms at the mouth of this river. Therefore, spring load reductions should be considered for this tributary in addition to annual reductions.

While the Task Team has not established targets for Eastern Basin *Cladophora*, the team has identified the Grand River, Ontario and Cattaragus Creek as potential priority watersheds. As mentioned earlier, the Task Team anticipates that reductions to the eastern basin will be necessary to reduce *Cladophora* growth and these two watersheds are identified as contributing a significant portion of the total phosphorus load to the Eastern Basin and are likely to be targeted for reductions. The Task Team encourages watershed managers to reduce phosphorus loadings from these watersheds in advance of targets being established. The Task Teams also recommends that for the Grand River, Ontario further research and monitoring be a priority to better understand the relationship between the loadings from the Grand River and *Cladophora* growth along the north shore of the Eastern Basin.

While identifying priority wastewater treatment plants was not an objective of the Task Team, the Team supports the strategy of the Urban Task Team which has identified five U.S.-based wastewater treatment plants representing more than 95% of the sewage load to the Western Basin and prioritized them for action. The Team also supports a similar approach for the Central and Eastern Basins.

Other Considerations

In-lake Phosphorus Concentration Objectives

There is a need for nutrient guidelines because nutrients are a basic driver of the ecosystem, recognizing that nutrient concentrations are not the endpoint but rather a contributing factor to ecosystem endpoints of concern (e.g., cyanobacteria blooms, hypoxia, and fish community health). The Objectives and Targets Task Team has had considerable discussion regarding the challenges of establishing concentration targets for both nearshore and offshore areas. The discussion has focused on the following issues regarding in-lake phosphorus concentrations:

- Concentrations exhibit significant spatial and temporal variation throughout the lake, especially in nearshore river mouth areas during high flow events.
- During a bloom, dissolved orthophosphate (PO_4) is taken up quickly by algae such that, often in severe blooms, DRP concentrations will be undetectable.
- Verifying whether concentration targets are being met pose a significant challenge that would require intensive monitoring programs that can capture the spatial and temporal variation.
- In contrast, loading targets (for total and dissolved phosphorus) are measureable and meaningful and can be developed from current models that calculate the load-response relationship.
- While concentration targets can be derived from the loading calculations a clear rationale needs to be developed for assessing compliance. Because measurements in the open water will be spatially and temporally variable, an effective compliance assessment strategy needs to include consideration of the following: the spatial and temporal domains over which compliance with be applicable concentration objective, the sample statistic that will be used to assess compliance (such as the sample average), the number of samples that will be required to achieve an acceptable level of uncertainty in this sample statistic (Stow *et al*. 2014).

The interim concentration objectives in Annex 4 and specified in the Lake Erie LaMP have historical precedents in Lake Erie programs. The Objectives and Targets Task Team explored the feasibility of developing recommendations that do not include phosphorus concentration values. The Task Team believes that nearshore objectives should not be developed or viewed strictly in the context of the re-eutrophication endpoints dealt with in the current models because relationships between nutrients and other biological responses (e.g., macrophyte community, macroinvertebrate benthic community, fish community) become more complicated and costly to measure in locations close to the shoreline. For example, WLEEM has the capability to model the time series TP concentration in Maumee Bay as a function of load from the Maumee River (figure 21). This plot is a time series of modeled Maumee Bay-wide average TP for the 2011-13 loads from the Maumee River. This plot demonstrates the significant temporal and inter-annual variability of a typical nearshore area of Lake Erie, making measurement of this metric very expensive and difficult to interpret in terms of compliance with an objective. And the fine-scale whole lake TP model developed by Schwab *et al.* (2009) demonstrates that these significant gradients exist at the river mouth areas of all major Lake Erie tributaries.
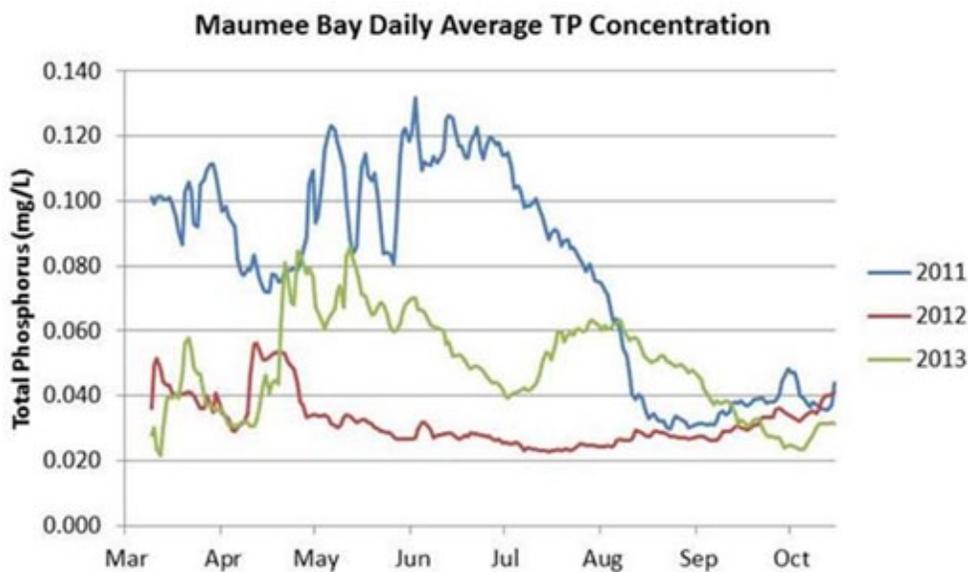
Figure 21. Maumee Bay daily average TP concentration simulated by WLEEM for 2011, 2012, 2013.

Nevertheless, nutrient concentrations play an important role everywhere in the lake and largely define trophic status in most habitats. Anticipated nutrient concentrations in the open waters of Lake Erie's three basins can be inferred from modeled loads, and we have checked that these are consistent with the Lake Erie Objectives.

We can set nearshore phosphorus concentration objectives that are consistent with the current eutrophication-related load-response relationships; however, in the spirit of setting a target for Lake Erie, it is the loads that must be controlled to manage the system. Nevertheless, other indicators of ecological concern with regard to coastal margins and the nearshore should also be considered. The need for such endpoints has been identified by SOLEC and the IJC to characterize overall ecological conditions of coastal margin and nearshore areas. Examples might be wetland vegetation or nearshore fish community characteristics, which reflect local nutrient conditions and water quality.

There is a need for research to develop and model the relationships between the condition of those ecological indicators and nutrient loads or concentrations in relevant habitat zones once developed. However, water levels, temperature, and wave action and other features are also important in these areas and must be incorporated into habitat-specific models.

For these reasons we do not recommend identifying phosphorus concentration objectives for the coastal margins or nearshore at this time; but rather we recommend implementation of a monitoring, research, and modeling effort that can set nearshore nutrient objectives in the context of the broader ecological framework noted above.

Bioavailable Phosphorus Considerations

Even though the loading targets recommended for Lake Erie are for TP loads, a considerable effort has been made by the Objectives and Targets Task Team to understand what we know and do not know about the relative bioavailable phosphorus content of the various Lake Erie sources. And this knowledge has been considered in making those recommendations. Presented below is summary of what we know and do not know about bioavailable phosphorus in Lake Erie and how that knowledge has been used in providing load target recommendations.

What we know

1. The only phosphorus compound that algae can use for growth is dissolved orthophosphate.
2. Algae can obtain dissolved orthophosphate from a number of phosphorus forms in water. These forms are considered to contain bioavailable phosphorus (BAP); that is, they have phosphorus forms that can be converted by various physical, chemical, or biological processes to orthophosphate.
3. TP as measured in a water sample consists of two broad categories: particulate phosphorus (PP) and TDP. TDP is measured by totally digesting and analyzing the phosphorus in a sample of water that passes through a 0.45 µm filter. PP is measured by subtracting the TDP from a TP measurement, which is measured by applying the same digestion and analysis procedure to the unfiltered water sample.
4. Both PP and TDP contain BAP, but with different fractions (Young and DePinto, 1982).
5. TDP, which contains soluble reactive phosphorus or SRP (also called DRP) and soluble non-reactive phosphorus (NRP), is normally assumed to be virtually 100% bioavailable. The SRP is virtually all orthophosphate and very labile dissolved organic phosphorus (i.e., molybdenum reactive without requiring the digestion step). The NRP in TDP contains colloid-bound (i.e., passing the 0.45 µm filter) plus non-reactive dissolved organic phosphorus. Under extreme orthophosphate limitation conditions, algae can produce alkaline phosphatase which promotes the release of dissolved orthophosphate from dissolved organic phosphorus. All of these forms can be converted quite readily to algal-available orthophosphate.
6. PP is generally composed of both organic and inorganic phosphorus-containing compounds adsorbed/bound to suspended solids (SS) in a water sample. The fraction of PP in a sample that is ultimately BAP depends on the source (soil-derived depending on land use and surficial geology or biotic produced solids (i.e., algae)) and associated characteristics of the SS and the extent to which the phosphorus is easily exchangeable with dissolved orthophosphate in the water.
7. The BAP of PP in Lake Erie tributaries can vary from about 15-50%, with tributaries draining the agricultural watersheds in northwest Ohio containing between 25-50% BAP). The non-BAP in PP is normally occluded (bound in the particle interstices) or exists as precipitated forms that do not readily dissolve (i.e., apatites).
8. Bioassays of the release of BAP from Ohio tributary suspended solids have shown that the conversion of the particle bound phosphorus to orthophosphate available for algal uptake and use occurs at a rate of approximately 0.1/day (or 10% of the ultimately available phosphorus on the solids per day)(DePinto 1982). So the immediate release of BAP depends on this rate relative to the rate of deposition of the solids to the bottom sediments once they enter the lake (DePinto, *et al*. 1981). Of course, even after depositing to the lake bottom sediments, the remaining ultimately bioavailable phosphorus can be released to the water column by resuspension and further desorption and/or by desorption in the sediments and release by pore diffusion to the water column.
9. Over the past twelve years from 2002-2013 the average spring (March-July) TP load from the Maumee River has been 1168 metric tons (MT), with 21% (246 MT) being DRP and 79% (922 MT) being PP. Assuming that DRP is 100% bioavailable and PP is 30% bioavailable, this average spring loading is 523 MT BAP, with 47% (246 MT) coming from DRP and 53% (277 MT) coming from PP. So, approximately half of the total BAP load comes from each measured form of the TP.
10. We know that tributary loading to Lake Erie is highly variable on an inter-annual basis, and that the variability is primarily controlled by varying precipitation-runoff driven hydrology. For example, the 2002-2013 spring TP load from the Maumee River, which averages 1168 MT has a standard deviation of 567 MT (or a coefficient of variation of 48.5%).
11. We know that various non-point source load control measures address the control of SRP versus PP differently. We also know that a combination of control measures, such as conservation tillage combined with fertilizer injection, can address both DRP and PP.
12. We can compute with our more complex models the in-lake cycling of BAP delivered from tributaries and relate that, along with other drivers such as light and temperature, to algal growth (DePinto, *et al*. 1986; LimnoTech 2010; LimnoTech, 2014).

What we do not know

1. We have some data on the PP and DRP fractions of TP in other Lake Erie tributaries, but for those not monitored by the Heidelberg University program the data are meager. We also have minimal data on the ultimately BAP fraction of the PP in those tributaries.
2. We have some information on the reduction of BAP (i.e., PP versus DRP reduction) by various BMPs, but this data set is sorely lacking.

The models being applied to develop the load-response curves that support setting target P loads either empirically or with process formulations account for the fraction of TP loads that is bioavailable and the conversion of TP to bioavailable forms within the lake. Therefore, it has been the decision of the Objectives and Targets Task Team to recommend loading targets for TP and DRP. It must be recognized, however, that there may be different combinations of DRP and PP load reductions that can achieve the same target load of TP and associated BAP. Even though a target is established for a TP load, we should attempt to employ load reduction actions that will be most effective at reduction BAP, both in DRP and PP.

A concrete plan must be developed to measure compliance with established target loads, and use the modeling-monitoring program to support an ongoing adaptive management process.

Role of Nitrogen Loads

One basic assumption made by the Objectives and Targets Task Team is that phosphorus (either total or dissolved reactive or some blend that represents bioavailable P) loading is the independent variable in all of these load-response relationships that developed. We recognize that there may be some reasons to consider nitrogen load control in addition to phosphorus. There are times when Lake Erie can become nitrogen limited, but that generally only occurs where and when the system is not limited by available phosphorus. Chaffin, *et al*.(2014) found that nitrogen limitation occurred when nitrate concentration dropped below 0.10 mgN/L (or total N was below 0.80 mgN/L). One concern is that too much emphasis on nitrogen control without significant phosphorus reduction will only lead to large blooms of nitrogen-fixing cyanobacteria (*Aphanizomenon* or *Anabaena*). Also, most NPS BMPs will reduce both P and N loss, although by different relative amounts for different BMPs. Therefore, it is not logical to target N reduction because most load-response analysis to date shows good quantitative relationships with P load, and there is no guarantee that N reduction alone will reduce cyanobacteria blooms or Central Basin hypoxia reduction. There is a school of thought that suggests that N control with P reductions will improve water quality more than P-only reductions. Hence, it is important that an adaptive management program examines the changes in both P and N as a P reduction program is implemented and how the cyanobacteria blooms might be responding to those changes.

Role of Dreissenids and other Invasives

There is no question that the invasion of Dreissenid mussels to the Great Lakes has altered the way the lakes have responded to nutrient loads by altering the way those loads are processed in the lake and the potential promotion of cyanobacteria and *Cladophora* blooms, including alteration of the load-response relationships developed in this analysis. While the biomass and species of Dreissenids have changed over the ~25 years since their initial invasion (Karatayev, *et al*. 2014), impacts on load-response relationships computed for this project have been done in a way that either implicitly or explicitly (depending on the model) accounted for the recent effects of Dreissenids. It is likely that Dreissenids have given cyanobacteria a competitive advantage in the Western Basin of the lake. Also, it is clear that Dreissenids in the Eastern Basin of the lake are having a significant effect on *Cladophora* growth and development in that part of the system. Again, it is important to include monitoring of Dreissenids and their ecosystem impacts, especially with respect to establishing nutrient load targets and/or concentration objectives to address over-abundance of *Cladophora*, as a part of an adaptive management program for Lake Erie nutrient management.

Role of Inter-Annual Hydrometeorology

Over the past 20 years external TP loads to Lake Erie showed large year-to-year variation but no clear long-term trend (Figure 1). Inter-annual variability is largely driven by hydro-meteorological conditions, which modulate the timing and magnitude of surface runoff and ultimately the amount of nutrients delivered to the lake by tributaries. For example, the large loads recorded in 1996-1998 have been associated with exceptionally high tributary loads due to increased precipitation (Dolan and Richards 2008). Over the most recent ten years for which detailed data on Lake Erie TP loads are available (2002-2011), phosphorus from non-point sources, transported to the lake by runoff and rivers, contributed on average 78% of the total annual load to the lake (Dolan and Chapra 2012; Dolan pers. comm. 2012). The inter-annual loading trends shown above for the Maumee River are greatly influenced by annual variability in flows. The Objectives and Targets Task Team has attempted to account for this confounding behavior in how it has identified a maximum flow below which the target load should be met and by recommending the use of FWMC's to track progress for any given tributary target load.

In addition to annual hydrology variability, variability in wind-driven circulation and mixing/resuspension, and temperature are also factors that contribute to the lake's response. And there seems to be increasing evidence that climate change in the Great Lakes will cause an exacerbation of the ERI's due to changes in all these factors. Therefore, monitoring these climate change impacts on the load-response relationships is a very important aspect of a good adaptive management program.

Watershed prioritization

The Objectives and Targets Task Team feels that the priority watersheds for ongoing management actions and intensive monitoring should be: Maumee, Sandusky, Thames, and Grand (ON). These have been identified because they have documented significant localized impacts. In fact, for some like the Sandusky, the impacts in confined Sandusky Bay may require even more than the 40% load reduction identified above.

6.   Conclusions

The recommendations made by the Objectives and Targets Task Team have addressed the six Lake Ecosystem Objectives (LEO) stated in Annex 4. Based on current knowledge and data, it is the conclusion of the Objectives and Targets Task Team that the recommendations will achieve the LEO. However, because of analysis and modeling uncertainties, data gaps, anticipated changes in the Lake Erie ecosystem from climate change and watershed development, the time necessary to achieve the target loads and reap their benefits in the lake will depend on how fast we take action to change current human behavior. Therefore, we consider it imperative that an adaptive management plan be devised to permit ongoing measurement of compliance with established target loads, and provide for the application of a monitoring/research/modeling program to support the adaptive management process.

A conceptual diagram of how an operational ecosystem modeling, monitoring, and research can be employed as part of an adaptive management process (Figure 22, (DePinto 2013)). We recommend, as shown in this figure, a process that includes annual routine monitoring of appropriate load, FWMC, and in-lake nutrient-eutrophication response indicators in conjunction with an intensive monitoring, research, and operational model application program every fifth year during the CSMI year for that lake.
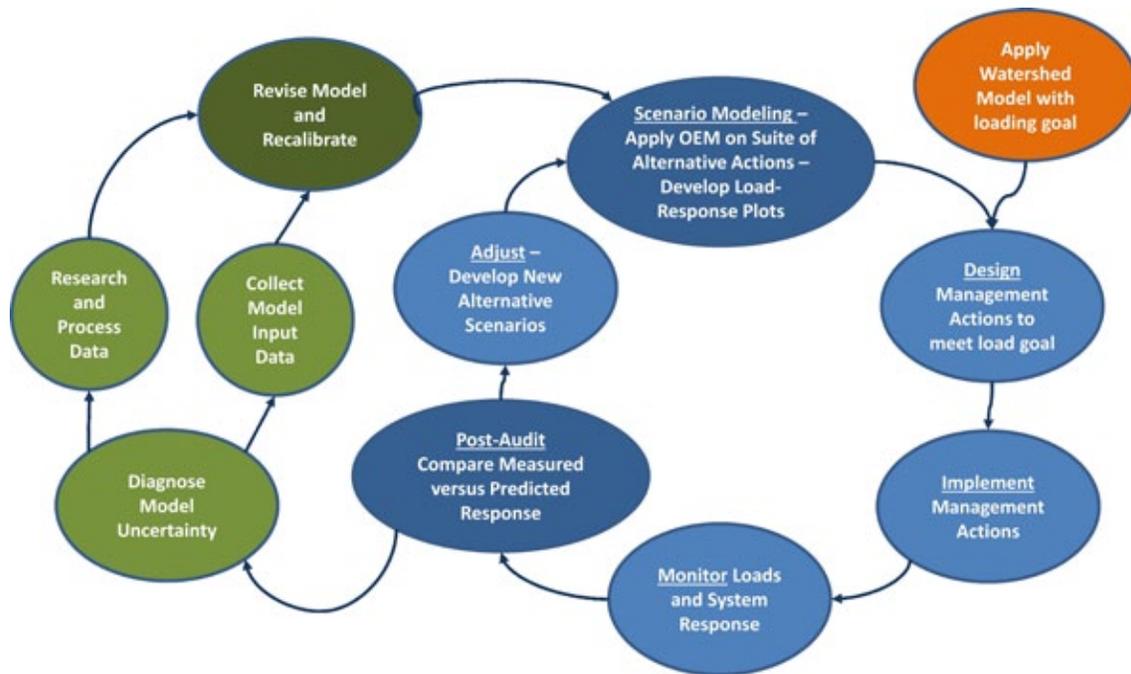
Figure 22. Adaptive management process employing an Operational Ecosystem Model (after DePinto 2013).

While the design of a comprehensive adaptive management program such as suggested above is beyond the scope of this report, we present below a list of research, monitoring, and modeling activities – in no particular order of priority – that could be included in such a program for Lake Erie. Included in parentheses are the pertinent LEO for each activity.

1. Annual and spring TP and SRP, N forms, and SS loading data for the top 12 tributaries (permanent monitoring); include calculation of FWMC equivalent. The top five tributaries, including the Thames, should have a high frequency data collection program such as that conducted for the Maumee River. Also, a continuation of the monitoring program on the Detroit River to refine information on its loads. (1,2,4,6)
2. Measure conditions at the mouth of tributaries in the Western Basin with reference to cyanobacteria blooms in order to prioritize and allow discontinuation of monitoring at lesser tributaries. (1,2,4,6)
3. Implement a *Cladophora* monitoring, research, and site-specific modeling program for the north shore of the Eastern Basin. (2,3)

4. Implement a monitoring and modeling interpretation program to determine required nearshore nutrient concentration objectives. This program should be aimed at quantifying important concentration-ecological response relationships, such as for fish and macroinvertebrate communities (develop in consultation with Annex 2 nearshore framework under development). (2,3)
5. Conduct research and monitoring to provide more data on BAP in tributaries and how it is expressed in the lake. These data can be combined with knowledge of which BMPs are being applied in the watershed to better understand the effectiveness of various BMPs for reducing DRP and PP. (1,2,3,4,6)
6. Measure ecological changes resulting from hypoxia reduction: P fluxes, benthic community, metrics for responses to the indicators, effects of temperature change. (1)
7. Monitor for cyanobacteria biomass and toxins related to human health concerns such as drinking water, possibility to manage for toxin production. (3,4)
8. Share research results and land management/lake responses. (1,2,3,4,6)
9. Evaluate monitoring programs to optimize sampling. (1,2,3,4,5,6)
10. Understand what the ecosystem response is to WB nutrient changes. (1,2,3,4)
11. Understand impact of changing AIS populations (eg: mussels) in order to improve the predictability of our models which influence future decisions. (1,2,3,4,6)
12. Develop a better understanding of nitrogen load and role in governing eutrophication responses in the lake. (1,2,3,4,6)
13. Develop a better understanding of the linkages between nutrients in the Thames R., Lake St. Clair, and Lake Erie. Extend the load-response models for the Western Basin cyanobacteria into the Huron - Erie Corridor. (1,2,3,4,6)
14. Project and measure climate change effects on cyanobacteria blooms, Hypoxia, and nearshore algae. (1,2,3,4,6)
15. Measure Niagara River nutrient load in outflow to Lake Ontario. (1,2,3,4,6)
16. Measure atmospheric nutrient loading. (1,2,3,4,6)
17. Measure nutrient internal cycling/sediment fluxes and net nutrient storage in each basin. (1,2,3,4,6)

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8. Appendix: Information Summary on *Cladophora*, a Benthic Algae Fouling Great Lakes Shoreline with Emphasis on Lake Erie

Report objectives:

This report was prepared for the Great Lakes Binational Annex 4 Loading and Nutrient Task Team to provide background information on the issue of nutrient-related overabundance of the alga *Cladophora* in the Great Lakes. The report provides a synopsis of information on the recent distribution of the alga, the ecological features influencing its abundance and a brief discussion on the models available to predict its response to environmental factors. The primary purpose of this document is to outline currently understood features of *Cladophora*, and the environment in which it grows with a focus on Lake Erie. These features will need to be considered in the development of efforts to:

(1) determine the extent of the nuisance algae problem;
(2) predict growth responses to environmental conditions on a broad geographic scale with established Great Lakes growth models;
(3) identify loading functions that express the linkages between *Cladophora* growth and land inputs of phosphorus; and
(4) determine the appropriate response for future abatement actions.

In addition, this report provides possible indicators and metrics that could be used to measure the status of the *Cladophora* problem and track the success of remediation measures with respect to reducing *Cladophora* abundance in the nearshore. At this time, the report does not attempt to make recommendations on the best approach to predict the growth response of *Cladophora* as a function of environmental nutrient loads although ultimately this will be required to support the development of nutrient loading targets and concentration objectives. The current uncertainty surrounding our ability to link the growth response of *Cladophora* to external nutrient loadings necessitates an in-depth discussion with the Annex 4 Task Team, as well as the larger community of specialists in this area to achieve consensus on appropriate indicators, metrics, and models. The outcome of this broader discussion as well as the outcome of the modeling scenarios is needed to develop specific recommendations for nutrient loading targets and concentration objectives aimed at reducing nearshore *Cladophora* blooms.

The report was prepared by Todd Howell[1], Veronique Hiriart-Baer[2] and David Depew[2].
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Introduction

Annex 4 (B. 2) of the renewed Great Lakes Water Quality Agreement of 2012 indicates that biomass of algae should be maintained below a nuisance level in the Great Lakes. When applied to benthic algae there is the question of what constitutes a nuisance level. Benthic algae problems are typically inferred from public complaints of fouling on the shoreline, which occurs when algae detach from the lakebed at points of growth and wash onto the shore. Public reaction to shore fouling is variable but can be strong. There are several examples in recent years of regionally based and community driven actions to investigate and promote abatement of shore fouling by nuisance benthic algae (e.g. Auer 2011; Bootsma *et al.* 2006; LOSAAC 2008; Great Lakes Mayors 2009)

The species of algae accounting for much of the shore fouling in the Great Lakes is *Cladophora glomerata*, a large-bodied filamentous green algae distinguished by strong holdfast cells that enable it to persist and flourish on high-energy coastlines. Recent occurrence features of *Cladophora* in the Great Lakes are summarized in Auer *et al.* (2010) and SOLEC (2014), while the ecology of *Cladophora* in the Great Lakes is well documented and reviewed by Higgins *et al.* (2008a) and others.

The distribution of *Cladophora* in the Great Lakes is broad with large tracts of shoreline where the algae achieves moderate to high abundance making it visually obvious on shallow lakebeds. Benthic lawns of *Cladophora* flourish widely in Lake Ontario, eastern Lake Erie, regions of Lake Michigan and within limited areas of Lake Huron and Lake Superior. The availability of hard substrate for filament attachment limits distribution in some areas.

Phosphorus is predominately the growth-limiting nutrient of *Cladophora* in the Great Lakes (Auer and Canale 1982; Higgins *et al*. 2008a). The promotion of *Cladophora* growth seen near point sources of phosphorus is also compelling support for the various lines of study (e.g. tissue nutrient stoichiometry, growth assays and field experiments) that have arrived at the conclusion that growth is phosphorus limited. Abatement of phosphorus loading to the Great Lakes in the 1970-80 is credited for a decline in *Cladophora* abundance in Lake Ontario in the 1980s (Painter and Kamaitis 1985). It is also suspected to be the reason for a similar decline in problems in eastern Lake Erie until the mid-1990s.

The level of photosynthetically active radiation (PAR) reaching the lakebed strongly shapes the abundance of *Cladophora* because of the alga's need for moderate to high light levels to flourish (Graham *et al*. 1982; Lorenz *et al*. 1991). Abundance typically falls sharply moving away from the shoreline into deeper water with the peak biomass levels found usually at depths of < 10 m. The importance of light regime for growth has been strongly demonstrated by the findings that increased water clarity over much of the Great Lakes (Lake Superior excepted) subsequent to the Dreissenid invasion likely accounts for an appreciable portion of the surge in *Cladophora* since the 1990s.

 A quantitative threshold to define "problem conditions" of *Cladophora* growth on the lakebed is challenging yet there is little difficulty in qualitatively concluding problem conditions exist when abundance levels are high. Biomass density of >50 g/m² (dry weight) has been suggested as the threshold for the onset of problem conditions (Auer *et al*. 2010).

Other elements of the benthic algae flora documented to foul shorelines do not appear to be as geographically wide-spread as *Cladophora*. Also, the distribution of nuisance problems are not as well known or investigated. The mat forming cyanobacteria *Lyngbya wollei* fouls shorelines on the south shore of the western basin of Lake Erie (Bridgeman and Penamon 2010) and recently in Lake St. Clair (Vijayavel *et al*. 2013). The green algae *Chara* and periphyton slurries dominated by diatoms foul shorelines in areas of eastern Lake Huron (Barton *et al*. 2013). Given that the focus of Annex 4 specifically identifies *Cladophora* as the nuisance macroalgae in Lake Erie for which targets are to be met, the discussion below will focus on *Cladophora*.

Historical and current Cladophora *conditions in Lake Erie*

Abundant growth of *Cladophora* leading to shoreline fouling has been an issue in eastern Lake Erie for decades (Milner and Sweeney 1982). Recently, the status of *Cladophora* and other benthic algae contributing to nuisance levels in Lake Erie was summarized and evaluated as part of the IJC TaCLE initiative (IJC 2013). Information on the distribution of *Cladophora* in Lake Erie is also given in the 2014 SOLEC report (see *Cladophora* indicator 2011) and in a status report on nutrients prepared for the Lake Erie LaMP (Hiriart-Baer *et al*. 2008a). There has been limited study of *Cladophora* in the western and central basins in comparison to the north shore of the eastern basin where optimal substrate for growth is widely distributed and the extent of past and present nuisance issues greatly exceeds other areas in the lake. The western and central basins are dominated by soft and unstable substrate providing limited scope for attachment of filaments which is critical for the success of *Cladophora*. Still, rocky shorelines of islands in the western basin (e.g. Lorenz *et al*. 1991) and shoreline structures throughout the lake provide habitat in an otherwise unfavorable physical environment.

Shore fouling on the north shore of the eastern basin can be tracked back to the 1960s based on descriptions in Ontario Ministry of the Environment reports from the early 1970s (OMOE 1973). Appreciable field studies were conducted over the period from the mid-1970s to mid-1980s by government agencies and academic researchers as well as Ontario Hydro who was the provincial power authority responsible for construction of a large coal-fired power plant adjacent to shoals optimal for *Cladophora* growth (Kirby and Dunford 1981). Although work appears to have ceased in the mid-1980s, presumably due to a perceived reduction of the fouling problem following management of phosphorus loads to the lake, a decline in fouling or *Cladophora* abundance was never documented. Recently, Brooks *et al*. (in press) analyzed historical Landsat imagery to determine temporal trends in distribution of *Cladophora* throughout the Great Lakes. They determined that its distribution declined slightly to the end of the 1980s and then increased on the Port Maitland (near the Grand River) shoreline of eastern basin through the 1990s. A second period of *Cladophora* study extending to present day began in 1995 with monitoring of extensive shoreline fouling and abundant growth of *Cladophora* in the shallows reported by Howell (1998) and Higgins *et al*. (2005a). Research and monitoring over the years since have confirmed both the fouling of shoreline by *Cladophora* at numerous locations and an abundance of *Cladophora* growing on hard substrate (Figure 1; Depew *et al*. 2011; Higgins *et al*. 2005a; Howell and Hobson 2003; OMOE 2009).

Information on the occurrence of *Cladophora* on the south shore of the eastern basin is sparse. There is limited hard substrate in the southern nearshore in comparison to the northern nearshore (Haltuch *et al.* 2000) where extensive bedrock shoals are distributed from the mouth of the Niagara River to the east of Long Point over approximately 110 km of shoreline. Shear and Konasewich (1975) and Mantai *et al.* (1982) make limited reference to occurrence levels of *Cladophora* on the south shore while recent studies (2009 to 2012) by Chris Pennuto Buffalo State College (personal communications) have examined the occurrence of *Cladophora* at sites along the south shore and associated water quality and biological conditions influential for growth. In these recent studies, *Cladophora* was detected at several sites in both the eastern and central basins reaching depths up to 10 m.

Cladophora growth patterns

A typical growth pattern for *Cladophora* in eastern Lake Erie consists of a rapid build-up of biomass over a period of weeks in early summer following a rise in water temperatures. High levels of biomass may persist for a time but often the initial build-up is followed by a rapid loss of biomass (a sloughing event) (see Higgins *et al.* 2006). Comparatively low biomass levels typically persist for the remainder of the summer and into the fall. The pattern is similar in Lake Ontario with the exception that there may not be a distinct sloughing event but a more gradual loss of biomass after peak biomass. Seasonal biomass trends reported by Tomlinson *et al.* 2010 at sites in Lake Huron and Lake Michigan indicate similarity in biomass levels over periods in the summer. Measured biomass of *Cladophora* on the lakebed as a growth indicator may be biased because of potential loss of biomass due to ongoing detachment of filaments from the lakebed or as catastrophic events. There are few direct measurements of the sloughing process making it difficult to know the degree to which observed biomass biases the interpretation of the accumulative growth in an area.

Studies documenting changes in areal biomass over annual growth cycles (e.g. Neil and Jackson 1982; Higgins *et al.* 2005a; Malkin *et al.* 2008) demonstrate the possible sensitivity of biomass measurements. The timing of data collection relative to annual growth peaks will affect inter-annual comparisons unless annual peak biomass levels are compared. To accomplish this, repeated biomass measurements are required targeting the period of expected maximal growth.

Modeling *Cladophora* growth: Model development, revision and application

Canale and Auer (1982) developed a mathematical model from first principles in the 1970s to characterize the response of *Cladophora* growth to environmental controls, i.e. light availability, phosphorus levels and temperature. At its most basic level, the model predicts the daily specific growth rate (μ; dS) as:

$$\mu= (GPP-R-S)*X \hspace{3cm} (1)$$

where GPP is daily specific gross production (d^{-1}), R is daily specific respiratory losses (d^{-1}), S represents daily physical sloughing or loss through detachment (d^{-1}), and X is the attached biomass (g/m^2). Each of these quantities is derived from a series of equations and the model requires only a few input variables: water temperature, PAR availability and SRP (or tissue P; QP). A full description of the model and its various formulae can be found in Canale and Auer (1982) and associated papers.

The initial application of the Canale and Auer model was largely successful in predicting a reduction in *Cladophora* growth and biomass in response to point source P control at Harbor Beach, Lake Huron (Canale and Auer 1982). In recent years, the Canale and Auer model has undergone several modifications and revisions and been re-cast as the "*Cladophora* Growth Model" (CGM; Higgins *et al.* 2005) and the "Great Lakes *Cladophora* Model" (GLCM; Tomlinson *et al.* 2010). Specific revisions to the model are discussed in detail by Higgins *et al.* (2005b) and Tomlinson *et al.* (2010). These revised models have been used primarily to evaluate the changes in *Cladophora* growth dynamics in the Great Lakes in the post-Dreissenid period (Higgins *et al.* 2006, 2012, Malkin *et al.* 2008, Tomlinson *et al.* 2010). A brief overview of these efforts is outlined below.

Lake Erie (east basin)

Higgins *et al*. (2005b) calibrated the CGM to a series of sites in Lake Erie and Higgins *et al*. (2006) evaluated the sensitivity of the CGM to changes in SRP concentration, light attenuation and temperature using data collected in 2002. Across sites, the natural variability in PAR attenuation in the water column (0.36 – 0.71 m^{-1}) induced significant changes in depth-integrated biomass. Variations in SRP among sites, particularly during the period of rapid growth, was determined to have the largest impact on depth integrated biomass resulting in up to a 3.5 fold difference among sites. Differences in water temperature among sites contributed little to differences in depth-integrated biomass, although temperature did affect the seasonality of growth patterns.

Lake Ontario (west basin)

Malkin *et al*. (2008) calibrated the CGM to a site along a section of highly urbanized shoreline in western Lake Ontario. The CGM substantially over-predicted both cumulative and attached biomass when bi-weekly measurements of SRP were used as a model driver. When using tissue P (QP), the model better tracked the observed growth dynamics. Malkin *et al*. (2008) suggested that loss processes (mainly respiratory losses) might be underestimated based on observed growth dynamics in the absence of catastrophic sloughing losses due to wind and wave action. Hindcast simulations using historical measures of QP and estimated light attenuation suggested that cumulative and attached biomass in 1972 and 1983 (pre-Dreissenid years) was 2.0 to 2.7 fold and 1.4 to 1.6 fold higher than contemporary measurements in 2004 and 2005. Like Higgins *et al*. (2006), small differences in water temperature yielded only small changes in attached and cumulative biomass.

Lake Ontario (multiple sites)

Higgins *et al*. (2012) used the CGM to model *Cladophora* growth at several sites in Lake Ontario in 2008. Increased light penetration at sites characterized by SRP concentrations < 0.5 µg/L were capable of two- fold biomass increases and mostly resulted from an expansion of growth in deeper waters (> 4 m). Model output at shallow depths was relatively unchanged, indicating the preponderance of P rather than light limitation. Inter-site variation in water temperatures (- 2.5 °C to + 2.5 °C) did not substantially impact depth integrated biomass, but small increases in SRP concentrations (+ 0.5 µg/L) were sufficient to increase both the depth-integrated biomass as well as the depth distribution of biomass. Similar to Malkin *et al*. (2008) the CGM over-predicted attached biomass by up to 8 fold when using a period median SRP concentration as a forcing variable. Model results using QP produced much improved predictions that better tracked observed biomass.

Lake Michigan (near Milwaukee)

Tomlinson *et al*. (2010) refined the original Canale and Auer (1982) model as the Great Lakes *Cladophora* Model (GLCM) and recalibrated the GLCM using the original data from Harbour Beach (Lake Huron), but also calibrated the model to more recent data from Lake Michigan. Tomlinson *et al*. (2010) reported an improvement over the original Canale and Auer (1982) model (measured by a reduction in root mean square error, RMSE) for the Harbour Beach dataset of 1979 and good performance at a deeper site in Lake Michigan (Atwater Beach, 9 m depth), where SRP was relatively invariant over the model domain. Tomlinson *et al*. (2010) did report that period average SRP levels from 1979 failed to capture the spatial heterogeneity in shallow sites that were impacted by the WWTP discharge to the lake.

Summary of modeling efforts

The general conclusions of these recent modeling efforts indicate that although water column levels of SRP have generally declined since the 1970s and early 80s, the increased light penetration (and area of hard substrate) in the post-Dreissenid era has increased the amount of suitable habitat for *Cladophora* relative to the pre-Dreissenid period. Based on model results, this is implicated as the most parsimonious explanation for the apparent increase in both attached *Cladophora* biomass and shoreline fouling experienced since the mid-1990s (Auer *et al.* 2010). While the changes in light penetration are generally attributed to the effects of particle removal by Dreissenid mussels, the relative contributions of mussel filtering and other factors that affect water clarity such as reduced P loading, whiting events and zooplankton grazing cannot be separated at present. Nonetheless, the models are in general agreement that contemporary *Cladophora* growth along affected shorelines is characterized by a modest reduction in the total depth-integrated and cumulative biomass compared to that observed (and predicted) for the 1970s and 80s, but extends deeper (and perhaps over a larger area) into the lakes.

The CGM and GLCM are data hungry models

Although the CGM and GLCM require only a few forcing variables (PAR, water temperature and measures of SRP and/or QP), the highly heterogeneous nature of physical, optical and chemical (and perhaps biological) properties of the Great Lakes nearshore poses challenges for modeling at local and regional scales. Both the CGM and GLCM performance is documented to suffer significantly when the most sensitive forcing variables (SRP and light) are not adequately characterized (Tomlinson *et al.* 2010; Higgins *et al.* 2012) and application of the models in environments known to experience substantial variability in light, nutrient and disturbance regimes (i.e. near major tributaries, diffusers) without suitable data may not yield results of value.

One approach presently underway to address this deficiency is via integration of the CGM (or GLCM) into larger more complex 3D hydrodynamic whole-lake models such as the Estuary and Lake Computer Model (ELCOM) coupled with the Computational Aquatic Ecosystem Dynamics Model (CAYDEM) and other formulations. These 3D models have demonstrated an ability to simulate the spatial and temporal variability of the major forcing variables required to describe the light, nutrient, thermal and disturbance regimes at large (2 km) and small spatial scales (100s of m) in Lake Erie and elsewhere (Leon *et al.* 2005, 2011, 2012; Schwab *et al.* 2009) and the potential for these models to overcome raw data limitations is impressive and needs to be investigated thoroughly. Recent updates to the ELCOM/CAYDEM formulation for Lake Erie include both modules for the CGM and a Dreissenid mussel component (Luis Leon, Environment Canada, personal communication), however, the success of these modeling approaches will ultimately rest with their ability to adequately simulate all relevant processes (both biological and physical) that are important for structuring the growth environment for *Cladophora*. Initial results, while encouraging, indicate that further work to parameterize important processes related to optical properties and nutrient dynamics are required before such modeling approaches can be used to effectively evaluate responses to P management scenarios (Luis Leon, Environment Canada, personal communication). Ongoing work to assess the P regime near the lakebed in proximity to *Cladophora* and efforts to develop and incorporate a lakebed P flux parameter into water quality models (e.g. Bocaniov *et al.* 2013) or the *Cladophora* growth model (see LimnoTech 2013) may provide an improved understanding and/or model performance.

SRP as a model currency in oligotrophic, Dreissenid dominated systems

The regions of the Great Lakes that experience the worst shoreline fouling and nuisance growth of *Cladophora* occur in waters generally classified as oligotrophic to meso-oligotrophic and colonized by Dreissenid mussels. The typically low levels of SRP in these phosphorus limited and biologically active environments pose both practical and conceptual questions on the use of SRP to represent biologically available P for *Cladophora* growth.

It is instructive to consider that the *Cladophora* models indicate that relevant environmental concentrations of SRP for *Cladophora* are only a few µg/L, and concentrations of SRP suggested being amenable to achieving a reduction in nuisance blooms of *Cladophora* are < 1 µg/L (Tomlinson *et al.* 2010). Even with excellent method detection limits (~0.5 µg/L), given the demonstrated sensitivity of the CGM and GLCM to comparably small changes of SRP (Higgins *et al.* 2005b, 2012; Tomlinson *et al.* 2010), small analytical errors or biases may nonetheless lead to very large and potentially misleading model results. Use of dialysis methods and attention to best practices for SRP sample processing and lab protocols should be able to provide measures that can resolve potentially problematic SRP values and uncertainties with analytical interpretation of data.

SRP, as it is presently measured and used within the models, is generally assumed to correspond to the available P, which in turn is considered entirely comprised of orthophosphate (PO_4^{-3}). It is well known that the conventional colorimetric SRP method (Murphy and Riley 1962) overestimates PO_4^{-3} concentrations in P limited lakes (e.g. Hudson *et al*. 2000), including the Great Lakes (Martin 2010). It is not currently known whether SRP as conventionally measured in nearshore habitats in the Great Lakes is the accurate measure of PO_4^{-3} that growth models for *Cladophora* assume it to be. Alternative analytical methods such as tracer techniques (Hudson *et al*. 2000), column chromatography (Taylor 2010) or pre-concentration approaches (e.g. MAGIC; Anagnostou and Sherrell 2008) are not simple enough for the kind of routine application needed to support regular monitoring of environmental concentrations.

Additional uncertainty regarding SRP dynamics in a Dreissenid dominated system is that SRP measured in the water column (as with most conventional sampling programs) may no longer be an appropriate measure of P availability to *Cladophora*. Recent studies in Lake Michigan and Ontario have documented the formation of an SRP concentration boundary layer within 15–35 cm of the lake bottom. SRP concentrations here can range between 2 – 8 fold higher than those measured at depths greater than 1–2 m off the bottom (Dayton *et al*. 2014; Martin 2010). Dayton *et al*. (2014) suggest that mussels, even at moderate densities can create ecologically significant P enrichment in the near bottom layer at intervals of sufficient duration to provide enough P to meet *Cladophora* growth requirements. Dayton *et al*. (2014) further suggest that these boundary layers are eroded when wave height exceeds a critical threshold (~ 0.15 m). Whether or not such conditions are formed at sufficient intervals to supply enough P to *Cladophora* or can form in shallower water (i.e. < 8 m depth) remains to be confirmed, however, these observations suggest that conventional sampling programs probably do not sufficiently characterize the P environment that *Cladophora* is now exposed to. Fortunately, the models allow for the use of tissue P (QP), which may provide a better approximation of the P available to *Cladophora* in the benthos rather than a point-in-time measurement of SRP that may or may not reflect SRP concentrations near the benthos. It is not known if the comparatively poorer performance of the CGM in Lake Ontario (Malkin *et al*. 2008; Higgins *et al*. 2012) when using SRP concentrations partly reflects such a phenomenon. A comparison of near bottom P concentrations and QP over a gradient of Dreissenid abundance may prove to be useful in resolving this discrepancy.

It will be important to pursue a greater understanding of SRP concentrations in close physical proximity to *Cladophora*, as concentrations at normal survey sampling depths (i.e. >1 m from the bed) may differ from those at the surfaces of the algal filaments. This will entail more specialized survey methods but should be feasible with existing technology.

Another uncertainty pertaining to P supply is the much less studied aspect of the role of dissolved organic P (DOP). Many freshwater algae (phytoplankton and benthic algae) produce enzymes (alkaline phosphatase; AP) that cleave phosphate from organic P substrates. Young *et al*. (2010) found that AP activity was not fully suppressed in fresh *Cladophora* from Lake Michigan, even when exposed to high PO_4 spikes (> 30 µg/L) for more than 10 d. Young *et al*. (2010) suggested that evolutionary adaptation to low P conditions has selected for constituent APA and the possibility that the use of DOP may be important at times of the year and should be evaluated.

Loss processes (sloughing) are relevant to growth prediction but difficult to quantify

Sloughing rates have been implicated as one of the most difficult parameters to accurately quantify in the models (Canale and Auer 1982). The most obvious impact of sloughing comes in the form of beach and shoreline fouling, however, proper characterization is also important for evaluating the utility of the growth models, particularly where calibration and validation are concerned. Malkin *et al*. (2008) noted increasing disagreement between model and observational data in the absence of catastrophic sloughing, implying either production was overestimated or respiratory losses were underestimated. Furthermore, it has been observed that the photosynthetic competency of *Cladophora* is quickly restored post-sloughing (Higgins *et al*. 2008b) so the opportunity for growth of additional cohorts (and therefore additional fouling) may be underestimated. Moreover, there is virtually no information on the fate of material that fails to wash up on beaches or shorelines, so the ecological significance of loss processes and possible re-growth cannot be fully assessed at present.

To date, sloughing has been modeled using a trigger mechanism, which may be based on metabolic balance at the base of filaments (Higgins *et al.* 2008b), water temperature (Canale and Auer 1982), or physical disturbance (Malkin *et al.* 2008, Tomlinson *et al.* 2010). Various levels of agreement between modeled output and observed data have been found using these triggers, but no one particular mechanism has been universally accepted or adopted, although it is generally agreed that the most catastrophic losses occur due to high wind and wave action (Malkin *et al.* 2008, Higgins *et al.* 2005b). This process generally occurs in mid-summer, as accrued biomass shades out cells at the base of the mat and respiratory losses exceed photosynthetic production. Severe declines in oxygenic photosynthesis coupled with high temperatures may further impair oxygen dependent electron transport required for metabolic integrity (Parr *et al.* 2002).

The flow of oxygenated waters into the mat may be impeded by accumulated algal biomass (attenuation coefficients between 0.29 and 0.96 m/m have been measured for Ulothrix, Dodds and Biggs 2002) and consumption of oxygen by respiring dreissenid mussels (Davies and Hecky 2005, Turner 2010) may further exacerbate oxygen stress. That these factors converge in mid-summer when water temperatures are generally peaking may explain the apparent correlation between sloughing and water temperature rather than some incipient lethal temperature threshold. This would be consistent with the rapid restoration of photosynthetic competence post-sloughing even though water temperature remains elevated (Higgins *et al.* 2008b).

Status of environmental drivers of *Cladophora* in eastern Lake Erie

The phosphorus regime of the offshore waters of the eastern basin is well documented and regularly tracked in monitoring programs by Canadian and US Federal agencies but less certain in nearshore areas. A summary of recent TP concentration trends based on federal monitoring is given in SOLEC (2014) and Hiriart-Baer *et al.* (2008). Spring concentrations have fluctuated around the basin objective of 10 µg/L while summer concentrations are lower with no obvious trend in the past 10 years. Median spring SRP concentrations over the last 10 years at offshore sites (to 2012) have varied from above detection to near 10 µg/L but have been <3 µg/L since 2008 (Dove and Chapra submitted).

The light regime of the north shore can vary widely both spatially and on intra and inter annual time scales complicating the interpretation of the growth patterns of *Cladophora*. The modelled growth projections of Higgins *et al.* (2005a, 2006) and Auer *et al.* (2010) for the eastern basin based on changes in light regime following the colonization by Dreissenid mussels in 1991 suggested that the depth distribution of *Cladophora* and seasonal biomass production increased post colonization due to stronger illumination of the lakebed. Supporting evidence for clearing of the water column after Dreissenid colonization in the form of diminished levels of phytoplankton in the nearshore of the eastern basin is reported by (Nichols *et al.*1993; Depew *et al*, 2006; North *et al.* 2012; Howell *et al.* 1996). Nonetheless, periods of high turbidity and reduced light penetration to the lakebed are frequent in the nearshore of the eastern basin (Higgins *et al.* 2005a; Howell and Hobson 2003).

Physical disturbance of the lakebed and shoreline, and discharge of turbid water from tributaries result in diminished PAR penetration to the lakebed. The extent to which local-scale factors affecting the light regime modify the distribution of biomass and productivity on inter-annual or spatial scales is uncertain. Considering the light requirements of *Cladophora* and the previous growth modelling in the eastern basin (Higgins *et al.* 2006; Auer *et al.* 2010) it is likely that variability in light regime will be a modifying factor in how the nutrient regime controls the biomass of *Cladophora* that develops in an area or in a given year.

The abundance of hard substrate in the eastern basin has contributed to the abundance of Dreissenid mussels (nearly all quagga mussels) which co-exist with *Cladophora*. Patterson *et al.* (2005) reports on the distribution of Dreissenid mussels in the eastern basin as of 2002. The subsequent invasion of the round goby, which feeds upon smaller Dreissenid mussels, is believed to have depressed mussel abundance (Barton *et al.* 2005), however, Dreissenid mussels remain widely disturbed and abundant on nearshore hard substrate. In 2010 Dreissenid densities were between 1000 to 3500 individual m-2 at 11 of 20 sites surveyed between the mouth of the Grand River and Port Colborne (OMOE unpublished data).

There is variability in water quality over the depths of optimal *Cladophora* growth on the north shore due to tributary discharge and land runoff. The Grand River is the largest tributary to the eastern basin, with a watershed area of approximately 7000 km², discharging nutrient enriched water which mixes with the lake over a wide area of habitat suitable for *Cladophora*. Land-use, hydrological and water quality features of the river are documented in a recent Grand River Conservation Authority (GRCA) report (draft 2013). Median river TP concentrations monitored from 2002 to 2011 at Dunnville just above the lake-effect zone were slightly above 100 µg/L, for both spring and summer periods. MacDougall and Ryan (2012) provide information on water quality, trophic state and biological features of the lower Grand River including near the point of confluence with the lake over seasonal surveys in 2003 and 2004. Water discharged to the lake can be characterized as eutrophic based on the results for the station nearest the lake (mean TP 90-100 µg/L; chlorophyll *a* (20-30 µg/L); total suspended solids 25-30 mg/L). However, SRP levels in the lower Grand River are not well described on a seasonal or inter-annual basis; studies underway by Environment Canada are expected to provide improved characterization of SRP in the lower river.

Despite past and ongoing monitoring and research on water quality in the adjacent nearshore, the scope of the river's influence on the nearshore remains vague. Total phosphorus levels at a drinking water intake located near the river mouth have been reported by Nichols *et al.* (2001) up until the year 1999. He at al. (2006) reported an extensive mixing area at the river mouth directed by alongshore currents. On occasion the influence of the river discharge on water quality can be detected >10 km east of the river mouth (Howell and Hobson 2003). Nicholls *et al.* (1983) examined the influence of Grand River on nearshore phytoplankton in 1979 and reported that during the summer, the influence of the river was confined to 5 to 10 km from the river mouth, similar to the findings of Painter and McCabe (1987).

Other tributaries to the north shore are comparatively small and the degree to which they impact the nearshore nutrient regime is unclear. After the Grand River, the next largest tributaries discharging to areas of rocky lakebed are Lynn Creek (288 km²), Nanticoke Creek (180 km²) and Sandusk Creek (158 km²); discharge from Nanticoke Creek and Sandusk Creek have been observed to affect water quality in the adjacent lake in areas of *Cladophora* habitat and locations with recurrent shore fouling (OMOE unpublished data).

Phosphorus management as a tool for controlling *Cladophora*

The relative importance of the pathways by which P is supplied to *Cladophora* from various discharges from the land, particularly over developed shoreline, continues to be a difficult question on which the approach to P management hinges. The question can be represented as a gradient in relative sources. At one end of this gradient the P in the water column of the lake sub-basin supplies the majority of P used for growth and P taken up by *Cladophora* is constantly replenished by lake circulation. At the other end of the gradient, the P supplied by local inputs to the shoreline enriches P concentrations over mixing zones, and these sources drive much of the *Cladophora* growth.

Teasing apart the relative influences on P supply supporting *Cladophora* growth over a given reach of shoreline has proven surprising difficult and recent studies have yielded contrasting interpretations. In a series of studies over the Oakville shoreline on Lake Ontario, Hecky *et al.* (2007) suggested that local supply of P from land inputs was not sufficient to account for the observed levels of *Cladophora* over the study area and inferred that *Cladophora* was sustained to a considerable extent by the basin P supply circulated inshore. These results are consistent with the observations of Bootsma *et al.* (2006) from studies over the Milwaukee shoreline of Lake Michigan. Similarly Depew *et al.* (2011) suggested that local enrichment of P supply seemed less likely to be sustaining the abundance of *Cladophora* then the abundance of Dreissenid mussels which was most strongly statistically related to *Cladophora* abundance among sites in Lakes Ontario, Erie and Huron. On the other hand, Higgins *et al.* (2012) suggested that regional differences in the biomass of *Cladophora* in Lake Ontario were due to higher P levels in the more urban area of the lake examined, while Auer (2011) suggested the P loading from a WWTP outfall was likely an important driver of the abundance of *Cladophora* over the Ajax shoreline of the Lake Ontario. It is obvious that the degree to which location-specific actions to manage P will be needed, will depend on the relevance of the site-specific P regime as it contrasts with that of the lake basin for growth. Likewise, the scale at which *Cladophora* modeling is best approached to support P management will depend on better resolution of this question.

Three generalized and interactive modes of P supply to *Cladophora* will be important for the interpretation of growing conditions:

(1)   water column concentrations of bioavailable P as a function of lake sub-basin trophic state and modes of onshore circulation;
(2)   water column concentrations of bioavailable P as a function of local and regional nearshore loadings and modes of alongshore and onshore-offshore circulation; and
(3)   enrichment of the near bed concentration of bioavailable P by flux out of the lakebed as a function of biological cover and near-bed circulation (and as affected directly or indirectly by 1) and 2)).

If Mode 1 Dominates

Mode 1 P supply would be the most straightforward to assess in that it aligns with the past approach to P management in the Great Lakes and is potentially predictable from basin-scale P loading. As noted earlier, modeling efforts to predict *Cladophora* growth are based on concentrations of SRP. Predicting dynamics of sub-basin SRP concentrations into the future below present day levels in the oligotrophic waters where *Cladophora* proliferates will require attention to model evaluation.

An approach to managing *Cladophora* based on modeling of sub-basin scale SRP concentration from basin or lake loads can likely work if three assumptions are met: 1) the largest fraction of total growth is fueled by P derived from the open-lake water column and ultimately offshore basin waters; 2) water column concentrations of SRP in the zone of *Cladophora* growth mirrors those of 4the broader epilimnion of the lake; and 3) changes in levels of SRP in the range relevant to the eastern basin and *Cladophora* growth can be reliably measured and modelled on the basis of integrated P loads to the lake and/or eastern basin. If these assumptions are met then existing *Cladophora* models can likely be used to predict growth, the accrual of biomass and the potential for shore-fouling based on estimates of basin-scale SRP over the growing season (informed by SRP concentration estimates derived from lake or basin scale loading models) and predicted by lake-basin P loading.

If Mode 2 Dominates or is Important

Some proportion of *Cladophora* growth will be driven by loading of P in proximity to where *Cladophora* is growing and exposed to locally elevated concentrations over discharge mixing areas. The principle that concentration gradients over mixing areas potentially enhance growth rates is well established (see Tomlinson *et al.* 2010). Area-focused studies have examined the impact of local conditions on *Cladophora* (e.g. Neil and Jackson 1982) and demonstrate the potential of local conditions to exacerbate *Cladophora* growth and shore fouling.

The overall significance of growth response to local conditions compared with Mode 1 growth will likely influence the approach to management. If an appreciable portion of the growth of *Cladophora* in the eastern basin of Lake Erie is dependent on P derived over local mixing areas, then this aspect of the P supply will need to be determined and included in the modeling of growth. The extent to which mode 2 growth is important is not understood nor is the P regime in the nearshore, as influenced by shoreline inputs, well described. The wide distribution of *Cladophora* along the north shore is suggestive of an overriding importance of the lake basin P supply, however, there are locations where discharges affect P levels in the nearshore and potentially affect the growth of *Cladophora* at the local scale. Considering studies around the Great Lakes there does not appear to be a basis to infer the relative importance of lake basin compared with local P supply, integrated over shoreline at either the basin or regional scales, for the growth of *Cladophora*.

Nevertheless, management of *Cladophora* under this scenario would dictate watershed remediation approaches such as the implementation of Best Management Practices (BMPs) or improved wastewater treatment, which would of course be watershed specific. A path forward may be to recognize that aspects of the *Cladophora* problem are likely best addressed at a local or regional scale using approaches that can identify the heterogeneity in the bioavailable P regime as influenced by area sources. As noted previously, the increasing use of fine-scale hydrodynamic and water quality models to evaluate water quality provides a basis to explore approaches capable of predicting driver variables needed for growth models at a local scale in the nearshore.

If Mode 3 Is Important

Areas heavily colonized by Dreissenid mussels focus P to the lakebed. This is evident by the qualitative increase in benthic biomass that is in large part sustained by the harvesting of organic material and nutrients from the water column by filter feeding Dreissenid mussels. Dreissenid covered lakebeds are typically rough, with much interstitial space, which increases the accumulation of particulate debris and provides habitat for periphyton and benthic invertebrates. The interstitial cavities are potential attractors of waste products of lakebed biota and of particulate material settling out of the water column especially during periods of runoff.

A hypothesis that has been developed by conceptual reasoning and the results of field and model-based studies suggests that leakage of bioavailable P from Dreissenid mussel beds into the base of the water column augments the nutrient supply to benthic algae beyond what would be inferred from water column concentrations. This has been suggested as a possible contributing factor in the resurgence of *Cladophora*. Mussel excretion and the breakdown of organic material accumulating on the lakebed results in a nutrient supply from the lakebed into the water column which *Cladophora* can access. The focus has been on the leakage of inorganic P, however, potential for release of labile organically-bound P as a nutrient source may also need consideration. There is evidence that mussel beds leak P (Ozersky *et al*. 2009) and as noted previously that there can be concentration gradients of SRP at the base of the water column over mussel beds (Martin 2010; Dayton *et al*. 2014). Dreissenid mussel beds (at depths of abundant *Cladophora*) have access to a substantial supply of P as phytoplankton and organic particles dispersed over the circulating lake epilimnion. Also, as land discharges pass over mussel beds, it is likely that some proportion of the particle-bound P contained in the mixing plumes settles out over the lakebed or is captured by the filter-feeding mussels. In either case watershed inputs are captured by the lakebed and may contribute P to the water column at later time.

It should be acknowledged that despite the intuitive reasoning and field studies suggesting a P subsidy to *Cladophora* from Dreissenid-colonized lakebed, there appears to be no direct evidence as yet that the growth of *Cladophora* utilizes or benefits from P derived from leakage from the lakebed. The successes of *Cladophora* growth modeling studies to date (as previously discussed) have been with models applications lacking any special consideration of Dreissenid interactions with P supply albeit in some cases the P inputs (i.e. QP or above lakebed SRP concentration data) may incorporate this effect. The relevance of Mode 3 remains a research question but one where the outcome may have considerable practical significance for the direction taken with management of the *Cladophora* problem.

If Mode 3 dominates, the management of *Cladophora* becomes a more complex task. With Modes 1 and 2, *Cladophora* responds to P concentrations in the water column, which in turn responds to phosphorus loads, either locally or lake-wide. Both of these P sources are manageable in the true sense of the word and actions required are relatively straightforward albeit potentially challenging to implement. In Mode 3, the P source is less directly manageable and the response to loading is mediated by interactions with Dreissenid mussels. To manage it and by extension *Cladophora*, the interaction with Dreissenid mussels may need to be included and in the extreme, directed at controlling Dreissenid mussel abundance. Phosphorus management in the watershed is still the only lever available, but interactions with Dreissenid mussels may affect response features and consequently the loading and concentration targets may not align with those derived from studies and modeling exercises that do not include key biological and physical interactions with the lakebed.

Dreissenid mussel growth can be limited by a number of factors including major ions (e.g. Ca2+), temperature and food quality and quantity (Baldwin *et al*. 2002; Stoeckmann and Garton 2001; Scheider *et al*. 1998). Temperatures, major ion concentrations and food quality required to limit mussel growth are not feasible management options in the natural environment. Food quantity may be the only management option available. A preliminary literature survey seems to suggest that mussel growth rates (d^{-1}; Baldwin *et al*. 2002) and shell length accrual (µm/d; Jantz and Neumann 1998) are lower or reduced at chlorophyll *a* concentrations less than 5 µg/L. However, recent data, at least in the surface mixed waters of the nearshore, suggest chlorophyll *a* concentrations may be lower than this suggested threshold. Dreissenid mussels have also been shown to be able to feed on bacteria, zooplankton and even supplement their energy requirements with dissolved organic matter uptake (Baldwin *et al*. 2002; Baines *et al*. 2007). Regardless of what dominates the food source of mussels in the nearshore of Lake Erie, once again we are faced with limited management options. Managing phytoplankton biomass, which has been shown to account for approximately 50% of the mussel diet in Lake Erie (Garton *et al*. 2005), may be the best available option for reducing the mussel P source. A better understanding of the drivers of phytoplankton biomass in the nearshore and the dynamics related to mussel feeding and growth may be required to identify P conditions in the nearshore which will cascade down to limiting *Cladophora* growth.

The reality is that all three modes are likely playing some role in the growth of *Cladophora* in the nearshore. Furthermore, their relative importance will also likely differ on local and/or regional scales. Questions evident when considering the environment of the north shore of the Lake Erie eastern basin and the management of *Cladophora* are similar to other regions of the Great Lakes where overabundance of *Cladophora* is a problem. There are areas where the possible role of local nutrients sources in fueling growth needs to be considered. The lawns of *Cladophora* carpeting much of the shallow nearshore co-exist with beds of Dreissenid mussels which may directly (leakage of P to the water column) or indirectly (through effects on basin P dynamics) moderate the P supply fueling *Cladophora* growth. For the most part, the overabundance of *Cladophora* occurs in a relatively low-P environment where water column concentrations in a broad sense are either at or below existing TP objectives of the Great Lakes Water Quality Agreement. The depth of understanding of the phosphorus pathways supporting *Cladophora* growth required to manage its overabundance is uncertain.

Cladophora **Indicators and Metrics**

Indicators

There are two broad points of reference when attempting to characterize a nuisance benthic algal problem: 1) biomass accumulating upon the shoreline (shore fouling); and, 2) growth on the lakebed (in-lake growth).

The assessment of shore fouling is challenging, the pattern of shore wash-up is the result of multiple factors including, alongshore currents, shoreline orientation and structure, weather, proximity to locations of algal growth as influenced by substrate and local to regional growing conditions (with nutrients as a key but not sole consideration). Consequently, fouling patterns are highly variable in time and space and most of the available information is qualitative. Furthermore, while there may be a high degree of fidelity from year to year in areas of shoreline subject to *Cladophora* fouling, the inter-annual variability in the timing and amounts of algal wash-up complicate the interpretation of temporal trends for this indicator. A further difficulty in assessing fouling is that there are variable types of algal material accumulating at the shoreline that are not easily quantified ranging from windrows of fresh material at the waterline to mats of putrid material undergoing anaerobic decomposition on the shore or in the swash zone.

Fouling of the shoreline is the outcome of growth (interpreted as net production resulting in biomass) by *Cladophora* and other benthic algae on the lakebed. In the case of *Cladophora*, a heavy bodied species (by micro algae standards) with high light requirements, its depth distribution is strongly skewed to shallow waters. Under typical present day nearshore conditions, biomass peaks occur in waters shallower than 6 m in Lake Erie and Lake Ontario but some growth extends to ca. 10m in Lake Erie and ca. 20m in Lake Ontario (Depew *et al*. 2011; Higgins *et al*. 2005a: Higgins *et al*. 2012; Malkin *et al*. 2008). On the other hand, the depth distribution of optimal growth appears to be deeper in Lake Michigan (Bootsma 2006). The distribution of *Cladophora* is also strongly affected by the availability of stable substrate for the attachment of filaments. The availability of stable substrate has increased tremendously in recent years through the stabilization of the lakebed by the colonization of Dreissenid mussels.

Possible Metrics for Indicators

Shore-Fouling

The assessment of shore fouling will be logistically challenging except when conducted over focused areas utilizing field observation or measurement. The fine spatial scale of shore fouling which is only a few metres wide and perhaps 10 to several hundred metres in length does not lend itself to most satellite imagery-based remote sensing approaches. Fouling events are often erratic in timing and variable in persistence. Several approaches have been used, from simple qualitative assessments over ranges of shoreline based on professional judgement to replicated collection of visual records of defined portions of shoreline to intense quantitative surveys of shore material (e.g. Barton *et al.* 2013). There does not appear to be a well-established method with a history of application or success. Studies conducted in the marine setting examining the accumulation of "wrack" debris on the shoreline (e.g. Gomez *et al.* 2013) may provide some insight into potential methods.

The peer-reviewed and gray literature provides information on a range of locations where shore fouling by *Cladophora* and other benthic algae has been observed, and in a number of cases on a repeated basis. Sentinel sites targeting sections of shoreline with documented fouling problems could be selected and used to track the incidence and severity of fouling over time. There are several possible parameters to track individually or collectively in the calculation of an integrative metric. These include percent cover by algal debris over a defined search area or median thickness of algal debris over a defined number of measurements over a search area. Repeated photographs over the search area could be used to assess cover, however, median thickness would require field measurements at multiple points over a seasonal growth cycle. Quantitative assessment of algal biomass on the shoreline would be a challenging option due to the logistical demands of collecting and separating algae from beach debris and given the multiple features of senescent *Cladophora* involved in fouling events.

In-Lake Growth

Physical sampling and remote observations have been used as techniques to assess the distribution and biomass of *Cladophora*. Physical sampling is usually based on stratified (depth, substrate etc.) random sampling and dry weight of algae removed from quadrats. Both depth-distributed sampling by divers over the nearshore zone (ca. to 10-20m depth) and surveys of the shoreline fringe (ca. 0-3 m depth) by snorkelling have been used. Visual assessments of the percentage of surface cover and height of *Cladophora* lawns have also been used to assess biomass distributions while others have used rakes in shallow waters to probe for the presence of *Cladophora*.

Remote sensing based on the interpretation of satellite imagery has been advocated recently as an assessment technique and methods have been developed and tested (Shuchman *et al.* 2013). The approach has potential for the assessment of areal distribution of *Cladophora* above the optical depth of the area of study. The methodology demonstrated by Shuchman *et al.* (2013) was capable of a semi-quantitative prediction of biomass based on three abundance regimes empirically related to field-determined biomass but this approach was unable to resolve intra-annual variability in growth beyond catastrophic sloughing. A related paper (Brooks *et al.*, in press) demonstrated how satellite imagery-based remote sensing could be used to evaluate time-series of areal extent at multiple locations in the Great Lakes. Their analysis identified large increases in the identifiable area of *Cladophora* and other submerged aquatic vegetation in the mid-1990s in Lakes Michigan, Erie, and Ontario. Shuchman *et al.* (2013) and Brooks *et al.* (in press) also examined strategies by which the analysis of satellite imagery may be used to derive areal-based biomass estimates. Sonar-based assessments have also been used successfully to assess the distribution and to estimate the cover, bed height and biomass of *Cladophora* (Depew 2009), but this approach has not yet been widely applied.

Surface cover by *Cladophora* can be high over a wide range of areal biomass. For this reason a metric of biomass based on measured biomass or inferred biomass from filament thickness (lawn height) over the lakebed will be more sensitive to change than percent cover.

With few exceptions Great Lakes studies have indicated that biomass is strongly depth-dependent with the highest biomass levels found at < 6 m. Presumably, the greatest scope for changes in biomass in response to environmental drivers will be in the shallows. Multiple measurements within a growth season will be required to determine near peak biomass, which in turn, will be needed to assess inter-annual variability. Spatial variability also presents a challenge in the field assessment of *Cladophora* biomass necessitating attention to site selection (i.e. substrate) and sample replication. Fine-scale variability in P and light regimes along reaches of shoreline, particularly at shallow depths were *Cladophora* is most abundant can be strong. Knowledge of site conditions relative to area influences on water quality will be essential for site selection.

Seasonal peak biomass on a depth-integrated basis for a unit length of shoreline would be an ideal metric but would be difficult to achieve on a broad geographic basis due to the field requirements for producing estimates using conventional methods (i.e. diver surveys). There may, however, be practical alternatives. Armed with knowledge of the depth ranges for optimal growth and periods of maximal biomass in regions, it may be possible to develop a survey strategy that targets the estimation of maximal biomass over optimal substrate and depth. This would require some level of field effort such as multiple surveys within a season and diver assisted data collection at depths. Preferably, data collection would target the same sentinel sites used for the shore fouling metric, and these would be selected to span a range of environmental influences affecting *Cladophora* growth.

Remote sensing assessments of the distribution of *Cladophora* as reported Shuchman *et al.* (2013) and Brooks *et al.* (in press) in combination with information collected at sites used for field-based assessments of areal biomass should be considered a means to providing a wider geographical coverage than would be possible with field monitoring alone.

The P content (QP) of *Cladophora* is a recognized indicator of P sufficiency and has been used to compare P sufficiency among areas and over time (e.g. Painter and Kamaitis 1985). Measurements of QP should be considered as a supplementary metric to areal biomass for the indicator 'in-lake growth'. Measurements of QP, in addition to other metric measures for areal biomass suggested above, would provide added insight on the overall growing conditions and when collected at times of high seasonal growth under high light conditions, would provide a measure of relative P sufficiency between years. The interpretation of field derived QP measurements is not without challenges given that it is a function of both external supply and internal utilization of P, and sometimes it is difficult to determine whether elevated QP is due to exposure to elevated P levels in the environment or insufficient light to support optimal growth and drawdown of cellular P. Hiriart-Baer (2008a) used fluorescence features of chlorophyll *a* to assess nutrient status of *Cladophora*. There is arguably scope for research to determine indicators of nutrient status, including but not limited to tissue P status that might be used to guide P management approaches.
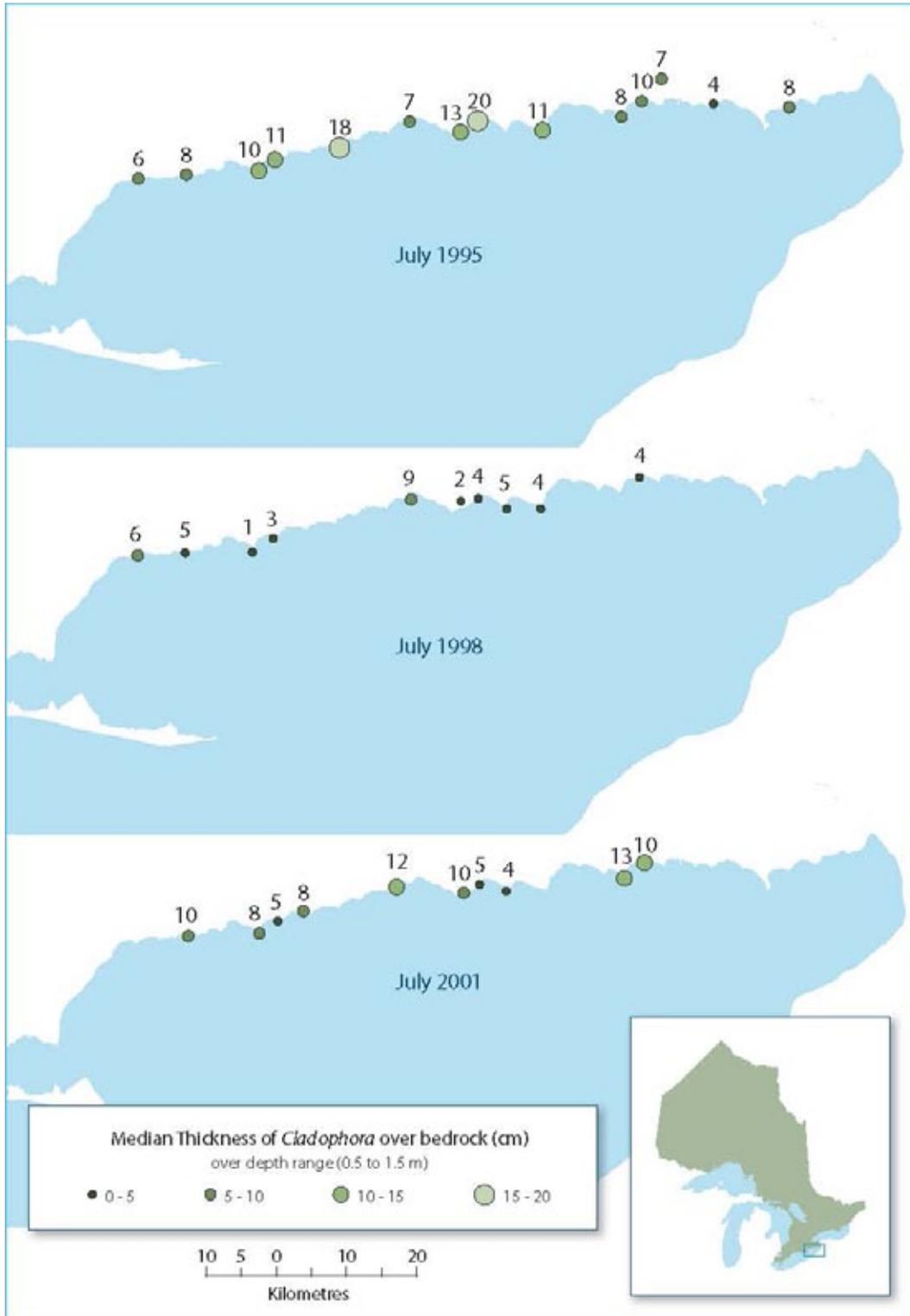
Figure 23. Median thickness of **Cladophora** growing over bedrock on the north shore of the Eastern Basin of Lake Erie. Figure extracted from Ontario Ministry of Environment Water Quality in Ontario report for 2008.

Recommended Indicators and Metrics

> Indicator
> 1) Shore Fouling
>
> Metrics
> a) percentage cover by algal debris over a defined area of shoreline
> b) median thickness of algal debris over a defined area of shoreline

The collection of shoreline sites, selected for the shore fouling surveys, should be chosen as a suite of sentinel sites with known and recurrent fouling by *Cladophora*. Survey locations within sites, should be identified as an area located at a fixed distance from the waterline, both into the lake and onto the shore, and extending a fixed distance along the shoreline.

> Indicator
> 2) In-lake Growth
>
> Metrics
> a) areal dry weight of *Cladophora* as the maximum seasonal biomass over the depth range of *Cladophora* growth
> b) as a more practical alternative to a), areal dry weight of *Cladophora* as the maximum seasonal biomass measured over a standardized depth strata that have been pre-identified as the zone of maximum growth over the specific sentinel shoreline segment.
> c) tissue concentrations of P (QP) in *Cladophora* collected at the time of seasonal peak growth and at the depth of maximal growth of *Cladophora*

The shoreline segments for survey should be selected as a suite of sentinel sites linked to known fouling of the shoreline by *Cladophora* and hence should correspond with the suite of sentinel sites selected for monitoring the shore fouling indicator.

Multiple surveys should be conducted in a year of survey to capture the period of maximal seasonal growth.

Remote sensing based analysis of surface cover of benthic vegetation based on the interpretation of satellite imagery should be attempted in the areas where sentinel sites are distributed.

Acknowledgments

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