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Mass-balance modeling framework for simulating and managing long-term water quality for the lower Great Lakes

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ABSTRACT

A mass balance model is used to simulate total phosphorus (TP) concentrations for the lower Great Lakes based on measured and estimated historical TP loading time series. Although Lake Erie is the primary focus, Lake Ontario is included in order to provide information about its potential response to proposed Lake Erie TP load reductions. The results demonstrate that Lake Erie loading controls would have a measurable effect on both Lake Erie and Lake Ontario's offshore phosphorus concentrations.

Introduction

Model description

Just over thirty-five years ago, a parsimonious total phosphorus (TP) budget model was developed to assess the impact of population and land-use on Great Lakes eutrophication (Chapra, 1977; Chapra and Robertson, 1977; Chapra and Sonzogni, 1979). The framework was then used, along with several other models, to establish phosphorus loading targets for the 1978 Great Lakes Water Quality Agreement (International Joint Commission, 1978; Bierman, 1980). The TP loading and resultant spring, offshore concentration targets for the lower lakes are listed in Table 1.

A positive initial confirmation of the model's predictive ability was made for Lake Ontario in the early 1980s when reductions in detergent and wastewater phosphorus had induced a significant downward trend in that lake's TP concentration (Chapra, 1980; Stevens and Neilson, 1987). Additional confirmation of model performance was established for the entire system through 1987 (Lesht et al., 1991). We recently expanded (Chapra and Dolan, 2012) the time frame of the previous modeling considerably by extending the analysis to 2010 based primarily on data collected by Environment and Climate Change Canada (Dove and Chapra, 2015). In so doing, a more complete assessment was provided of whether the model adequately simulated the water-quality improvements observed from the mid-1970s to the present.

In this recent assessment (Chapra and Dolan, 2012), a number of refinements were adopted to improve model performance. First, a more detailed segmentation scheme was employed with finer resolution for the major Great Lakes embayments. A chloride budget model was developed to better parameterize transport with particular emphasis on quantifying the magnitude of mixing across open boundaries. Then, the TP budget model was used to compute concentration trends and compare them with modern TP data for the period from 1965 to the present.

In the present exercise, we use the same framework developed by Chapra and Dolan (2012) to evaluate the impact of total phosphorus loadings on the eutrophication of the lower Great Lakes. Although Lake Erie is our primary focus, we have included Lake Ontario in order to provide a more comprehensive assessment of the impacts of load reductions. In particular, we believe it is important to recognize that Lake Erie controls may have a measurable effect on Lake Ontario. Offshore total phosphorus concentrations in Lake Ontario are currently in

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Table 1
Great Lakes total phosphorus target loads and resultant spring, offshore concentrations. Note that the target loads include phosphorus inputs from upstream lakes.

Basin	Target TP load (MTA)	Target TP concentration (μgP L^{-1})
Lake Erie	11,000	
Western Erie		15
Central Erie		10
Eastern Erie		10
Lake Ontario	7000	10

the range 5–6 μgP L^{-1} (Dove and Chapra, 2015) which are well below the target concentration of 10 μgP L^{-1} and indicating low nutrient status (oligotrophy). Given the proposed 40% reductions for Lake Erie western and central basin total phosphorus loads, as well as spring soluble reactive phosphorus loads to the western basin (Objectives and Targets Task Team, 2015), it is prudent to understand how these proposed Lake Erie loading reductions will affect Lake Ontario.

Methods

Our model is designed to predict the annual average concentration of total phosphorus (TP) in the offshore waters of each lake as a function of external loadings. Whereas this approach is consistent with the time and space scales required for the establishment of target loads under the GLWQA, it is important to stress that the model does not attempt to resolve finer-scale temporal (e.g., diel or seasonal) or spatial (e.g., nearshore–offshore) variability.

As with the original model (Chapra, 1977), Lake Ontario is represented as a single well-mixed system, whereas Lake Erie is divided into its 3 major subsegments to better resolve horizontal gradients (Fig. 1). TP mass balances for each basin can be written as (Chapra, 1975, 1977, 1997)

Western Lake Erie (w):

$$V_w \frac{dp_w}{dt} = W_w + Q_h p_h - Q_w p_w + E_{wc}'(p_c - p_w) - v_{s,w} A_w p_w \tag{1}$$

Central Lake Erie (c):

$$V_c \frac{dp_c}{dt} = W_c + Q_w p_w - Q_c p_c + E_{wc}'(p_w - p_c) + E_{ce}'(p_e - p_c) - v_{s,c} A_c p_c \tag{2}$$

Eastern Lake Erie (e):

$$V_e \frac{dp_e}{dt} = W_e + Q_c p_c - Q_e p_e + E_{ce}'(p_c - p_e) - v_{s,e} A_e p_e \tag{3}$$

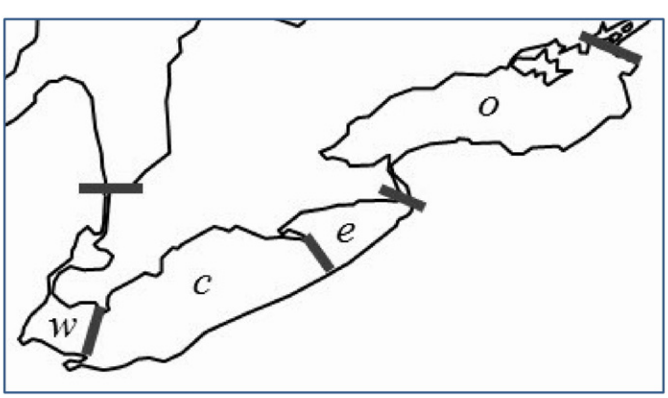

Fig. 1. Segmentation scheme for chloride and total phosphorus models for the lower Great Lakes.

Ontario (o):

$$V_o \frac{dp_o}{dt} = W_o + Q_e p_e - Q_o p_o - v_{s,o} A_o p_o \tag{4}$$

where V_i = volume of segment i (km^3), p_i = concentration of segment i (μgP L^{-1}), W_i = the mass loading rate to segment i (metric tonnes per annum or MTA), Q_i = advective outflow from segment i ($km^3\ yr^{-1}$), $E_{i,j}'$ = bulk horizontal eddy diffusion coefficient at interface between segments i and j ($km^3\ yr^{-1}$), $v_{s,i}$ = apparent sedimentation velocity for TP for segment i ($km\ yr^{-1}$), and A_s = the segment's bottom sediment surface area across which TP is permanently lost from the system (km^2). Note that by setting v_s to zero, Eqs. (1) through (4) apply to conservative substances such as chloride, for which case loadings and concentrations would be expressed in kMTA and mg L^{-1}, respectively. Note that although the apparent sedimentation velocity has velocity units, it should not be confused with an actual particle settling velocity. Rather, as originally defined by Chapra (1975) it is a phenomenological (hence, the modifier "apparent") and empirically-derived mass transfer coefficient representing all the varied mechanisms that ultimately contribute to the incorporation of water-column total phosphorus into a lake's bottom sediments.

The bulk mixing coefficient is also a phenomenological parameter that represents large-scale diffusive exchange across open boundaries. As described by Chapra (1979), this parameter accounts for all transport mechanisms over and above advective outflow from a lake or a segment of a lake. These include, but are not limited to, exchange due to large-scale eddy diffusion, and dispersion due to shear flow and spatial non-uniformities. The mixing is related to more fundamental quantities by Chapra (1979)

$$E' = \frac{EA_c}{\ell} \tag{5}$$

where E = horizontal eddy diffusion coefficient ($km^2\ yr^{-1}$), A_c = interface cross-sectional area (km^2), and ℓ = mixing length (km). The mixing length parameterizes the length of the zone defining the gradient between adjacent volumes (Chapra, 1997).

The complete system of differential equations provides a quantitative framework to analyze trends of chloride and TP. Given parameters, loading time series, and initial conditions, the equations can be integrated to obtain concentrations as a function of time. To do this, we use a constant-step, fourth-order Runge–Kutta method (Chapra, 2011). By centering the time derivative estimate, the fourth-order scheme tends to minimize the temporal component of numerical diffusion.

In addition to the time-variable solutions, a steady-state solution can also be generated. To do this, the derivatives of Eqs. (1) through (4) are set to zero and the resulting system of algebraic equations written in matrix form as

$$[A]\{p\} = \{W\} \tag{6}$$

where

$$[A] = \begin{bmatrix} Q_w + E_{wc}' + v_{s,w}A_w & -E_{wc}' & 0 & 0 \\ -E_{wc}' & Q_c + E_{wc}' + E_{ce}' + v_{s,c}A_c & -E_{ce}' & 0 \\ 0 & -E_{ce}' & Q_e + E_{ce}' + v_{s,e}A_e & 0 \\ 0 & 0 & 0 & Q_o + v_{s,o}A_o \end{bmatrix} \tag{7}$$

$$\{p\}^T = \{p_w \ \ p_c \ \ p_e \ \ p_o\} \tag{8}$$

$$\{W\}^T = \{\ (W_w + Q_h p_h) \quad W_c \quad W_e \quad W_o\ \}. \tag{9}$$

Eq. (6) can be solved for the steady-state concentrations

$$\{p\} = \left[A^{-1}\right]\{W\} \tag{10}$$

where $[A^{-1}]$ = the matrix inverse, also known as the "steady-state system-response matrix". The individualelements of this matrix, $a_{i,j}^{(-1)}$ represent the change in concentration of segment i due to a unit load change for segment j. The units of the elements of the matrix inverse are μgP L^{-1} per MTA for TP or mg L^{-1} per kMTA for chloride (i.e., when the v_s's = 0).

Model parameters and forcing functions

Morphometry

Morphological parameters (volumes, areas and lengths) for the lake segments and interfaces are listed in Tables 2 and 3. These are based on previously published estimates (e.g., Chapra and Sonzogni, 1979; Quinn, 1992; Chapra and Dolan, 2012).

Intersegment flows

From 1860 through 2010, the annual outflows for connecting channels are based on measurements reported by the Army Corps of Engineers (Keith Kompoltowicz, personal communication) and NOAA's Great Lakes Environmental Research Laboratory (Croley and Hunter, 1994; GLERL, 2011). Within-lake intersegment flows are derived from annual water balances consisting of measured tributary flows, changes in lake level, and estimates of net over-lake precipitation (precipitation minus evaporation). The flows prior to 1860 are set to constant values equal to the average flows for the period before the major Great Lakes diversions were implemented (1860–1899).

Loadings

Time series of annual chloride and TP loadings were determined for each model segment for the period from 1800 to early 2010 based on historical estimates (Chapra, 1977), reports to the International Joint Commission (IJC) by the Great Lakes Water Quality Board (as summarized for the State of the Lakes Ecosystem Conference, SOLEC, by Neilson et al., 1995), and recent direct measurements for the period from 1994 to 2010 (Dolan and Chapra, 2012).

Concentrations

The concentration data can be divided into two categories: historical (prior to the mid-1960s) and modern (after the mid-1960s). Historical chloride concentration data were taken from a number of sources as previously summarized by Chapra et al. (2009, 2012). Historical TP concentration data were taken from a number of published Great Lakes Water Quality Board reports (1974, 1975, 1976, 1977a, 1977b, 1978a, 1978b, 1979, 1987). For the historical data, values either represented single samples (especially for the earliest chloride data) or annual averages.

Beginning in the late 1960s, much more systematic data collection efforts were instituted under the auspices of Environment and Climate Change Canada (ECCC). Because they involved more rigorous quality control, this "modern" data set exhibits considerably less uncertainty than the historical data. Hence, it provides a better basis for separating long-term, emerging trends from interannual natural variability.

ECCC's Great Lakes Surveillance conducts ship-based cruises to collect water-quality samples on the lower Great Lakes. Each lake is generally monitored every second year, with multiple cruises conducted during that year. All regions (nearshore, offshore and major embayments) are monitored. Methods are described in Dove et al. (2009). The data collection and analysis methods used for chloride are also described in additional detail in Chapra et al. (2009, 2012) and for TP in Dove and Chapra (2015).

We have used latitude and longitude data to compute open-lake annual spring median values for both chloride and TP concentrations for each model segment. Medians, rather than average values, are chosen because the former is less sensitive to outliers. As described above, Lake Erie is divided into three basins to capture the west-to-east longitudinal water quality gradients. For Lake Ontario, we systematically excluded near-shore samples. This was done under the assumption that open-lake (offshore) waters best indicate long-term trends because they are less influenced by local pollutant discharges than shallower nearshore waters. It should also be mentioned that these segmentations are consistent with the current GLWQA. For example, the Agreement's TP concentration targets are basin-specific for Lake Erie and apply to Lake Ontario open-lake spring concentrations (Table 1).

Table 2
Segment parameters for the lower Great Lakes. The volumes and sediment areas represent measured values whereas the TP settling velocity is based on calibration. The higher settling velocities in parentheses are used starting in 1990.

Segment	Symbol	Volume km^3	Sediment area km^2	Settling velocity $m\ yr^{-1}$
West Erie	WE	28	3680	30 (50)
Central Erie	CE	274	15,390	22 (30)
East Erie	EE	166	6150	27 (32)
Ontario	ONT	1631	18,960	19 (29)

Model calibration and confirmation

The model calibration consists of adjusting the free parameters-the bulk mixing coefficients and the apparent sedimentation velocities-so that the final simulation results fit measured chloride and TP concentrations, respectively. Because the chloride simulations only depend on the former, the diffusion coefficients can be estimated by first adjusting them to fit the chloride data. These values along with the other parameters and the measured TP loadings can then be employed to estimate the apparent sedimentation velocities by adjusting them to fit the TP data. This sequential strategy of calibrating the transport first and then the transformations is commonplace in water-quality modeling (Chapra, 2003) and for the present case has the advantage of parsimony in that each calibration step involves adjusting a single free parameter (i.e., there is only one unknown parameter). Thus, there is only one degree of freedom at each step, and the mass-balance provides a framework to directly estimate each of the unknown parameters by difference.

Bulk mixing

Initial estimates of the intersegment eddy diffusion were based on previously published estimates (Chapra and Sonzogni, 1979; Chapra and Dolan, 2012). In addition, we also computed a first estimate for each boundary based on the Okubo (1971) diffusion diagram with the following equation:

$$E_{j,k} = 6,260\ell_{j,k}^{4/3} \tag{11}$$

where j and k designate the two segments straddling the open interface, $E_{j,k}$ = the horizontal eddy diffusion coefficient ($cm^2\ s^{-1}$), and ℓ = a characteristic length representing the approximate size of the eddies contributing to the turbulent mixing across the interface (km).

Using these values as a starting point, the diffusion coefficient was then adjusted to minimize the sum of the squares of the residuals between the measured and simulated chloride concentrations for the period from 1994 through 2008. The final values are listed in Table 3.

Of course, more refined hydrodynamic models as well as much higher resolution spatial and temporal data would be required to definitively describe the complex mixing regimes for such systems. Nevertheless, because they are primarily dependent on the observed annual (spring) median chloride concentrations, the values in Table 3 provide reasonable first-order estimates of annual mixing that are compatible with the present lumped, mean annual model.

Table 3
Interface parameters for the lower Great Lakes. The areas and lengths represent measured values whereas the diffusion coefficients are based on calibration.

Interface	Cross-sectional area km^2	Mixing length km	Diffusion 10^6 cm^2 s^{-1}	Bulk diffusion km^3 yr^{-1}
WE-CE	0.464	66.14	1	22.1
CE-EE	1.144	91.35	5	197.5

Apparent sedimentation velocity

As was the case for diffusion, initial estimates for some segments can be based on previous studies (Chapra and Sonzogni, 1979; Lesht et al., 1991). These values were then adjusted so that simulated TP concentrations were consistent with measured values.

In so doing, we again adopted a two-step approach. The model was first run for the period from 1800 to 2010 and the sedimentation velocities set at constant values in order that the model yielded TP concentrations consistent with observations collected just before loading reductions were implemented in the early 1970s. In this way, the subsequent model test (i.e., whether the model followed measurements after load reductions were implemented) starts with minimal initial condition error. For the main bodies of the lower lakes, the modeled and measured concentrations began to diverge after about 1990 with the most glaring discrepancy in Lake Ontario. We therefore recalibrated the model by increasing the post-1990 apparent sedimentation velocity in order to yield model output better matching current measured TP concentrations.

The resulting calibrated values are listed in Table 2 with the post-1990 values in parentheses. Although these values are quite variable, their range is not atypical of literature estimates. Data analyses by a number of individuals (e.g., Chapra, 1975; Dillon and Rigler, 1975; Thomann and Mueller, 1987) have determined that the sedimentation velocity can range over about an order of magnitude from about 3 to 30 m yr^{-1}. However, values have been reported from less than 1 to over 200 m yr^{-1}.

Final model output

The model output is expressed as predicted total phosphorus concentrations for each lake segment. These are compared with actual measurements, and load–response curves are provided for total phosphorus. The trophic indicator variables of summer chlorophyll *a* and Secchi disk depth are additionally predicted based on their empirical correlation with spring total phosphorus as described in Dove and Chapra (2015). Because the model is linear, we provide the response for each lake segment to each MTA loading reduction that may be implemented for Lake Erie. In this way, we can predict the response of Lake Erie water quality to total phosphorus load reductions, and additionally predict the (unintended) consequences for downstream Lake Ontario.

Results

The following summary of results focuses on total phosphorus concentrations for the lower Great Lakes model segments. Detailed results for the chloride simulations as well as for the major embayments (in the upper Great Lakes) are presented elsewhere (Schmitt Marquez, 2010; Dolan et al., 2011; Maccoux et al., 2016).

Total phosphorus concentration trends

Although the model simulations are generated for the entire post-settlement period (1800–2010), the results presented here are limited to 1950 to 2010. This was done in order to better visualize the behavior of the modern model output and data.

Fit statistics

In addition to the plots, we have also computed two fit statistics in order to quantify the residual error between the data points and the model simulations. These are an average absolute percent relative error and a root mean error.

The average percent absolute percent relative error is computed for each segment as

$$\text{Average Error} = \frac{\sum_{i=1}^{n}\left(\frac{|c_i(t_i)-c(t_i)|}{c_i(t_i)}\right)}{n} \times 100\% \tag{12}$$

where t_i = the time corresponding to the i^{th} TP concentration measurement; $c_i(t_i)$ = the i^{th} TP concentration measurement (μgP L^{-1}); $c(t_i)$= the model calculated TP concentration at time t_i (μgP L^{-1}), and n = the number of measurements.

The standard error is computed for each segment as

$$\text{RME} = \sqrt{\frac{\sum_{i=1}^{n}(c_i(t_i)-c(t_i))^2}{n-1}}. \tag{13}$$

Note that this formulation assumes that one degree of freedom is lost due to the fact that each segment's fit is primarily dictated by a single estimated parameter, the apparent sedimentation velocity. Of course, because the system is coupled, each segment's concentration is also dictated by other segment sedimentation velocities as well as the model's other estimated parameter, the open-interface bulk mixing coefficients. Nevertheless, we believe that Eq. (13) provides a reasonable estimate of the magnitude of the scatter around the best-fit line represented by the calibrated model.

As listed in Table 4, the results for most parts of the system indicate a good fit with root mean and percent relative errors on the order of 1 μgP L^{-1} and 20%, respectively. The highest errors occur for Lake Erie. This is a reflection of the fact that Lake Erie, of all the Great Lakes, demonstrates the greatest water quality variability (Chapra et al., 2009, 2012; Dove and Chapra, 2015) due to the large size of its inputs relative to its volume and its shallow depth (especially in the western basin) which causes variability due to resuspension events.

The trends and variability of the model simulations can also be directly assessed by examining the time series of the model output and data shown in Figs. 2 and 3. In order to relate the TP levels in the plots to trophic state, the present analysis uses values of 10 and 20 μgP L^{-1} as bounds for mesotrophy, as originally suggested by Dillon (1975) and confirmed for the Great Lakes by Chapra and Dobson (1981). In

Table 4
Calibrated model fit statistics (i.e., with higher post-1990 settling velocities). The average TP concentrations for 1970 and 2010 computed with the calibrated model are also included to provide a basis for assessing the RME.

			Average TP		Average
Segment	n	RME	1970	2010	Error
West Erie	43	5.4	39.6	15.5	26%
Central Erie	47	3.7	20.8	7.9	26%
East Erie	48	3.6	19.6	7.2	29%
Ontario	53	1.5	21.5	6.2	13%

addition, we have classified segments with TP < 5 μgP L^{-1} as being ultraoligotrophic (Homa and Chapra, 2011).

Lake Erie

After having experienced severe cultural eutrophication during the 1950s and 1960s, both the data and model simulations indicate that TP levels for all three basins of Lake Erie have dropped significantly over the following decades (Fig. 2). Nevertheless, the target concentration levels are still exceeded for some years. This is partly due to the fact that although Lake Erie's TP loadings have been reduced, they still sometimes exceed the established annual target loads (Dolan and Chapra, 2012). Furthermore, the exceedances are exacerbated by the fact that Lake Erie's response exhibits higher interannual variability than for the main body of Lake Ontario.

As noted previously, we have employed constant and elevated post-1990 sedimentation velocities for each of the basins. As depicted in Fig. 2, the range of the two scenarios is comparable to the range of the observations with the heightened case yielding somewhat better fits to the more recent 2010 data.

Lake Ontario

Whereas the foregoing results suggest that Lake Erie may have experienced stronger post-1990 TP assimilation, the simulations for Lake Ontario are much more conclusive. As depicted in Fig. 3, the observations indicate a much greater decline for Lake Ontario TP than predicted by the model with a constant apparent sedimentation velocity. Whereas the model predicts that the load reductions should bring the lake to oligo-mesotrophic levels, the data indicate that it is solidly oligotrophic bordering on ultraoligotrophic with a minimum mean annual TP concentration of approximately 5.4 μgP L^{-1} in 2002. In order to simulate this outcome, a higher post-1990 apparent sedimentation velocity must be employed. As listed in Table 2, an apparent sedimentation velocity of 29 m yr^{-1} after 1990 yields a model simulation that minimizes the sum of the squares of the residuals between the model output and measured values.

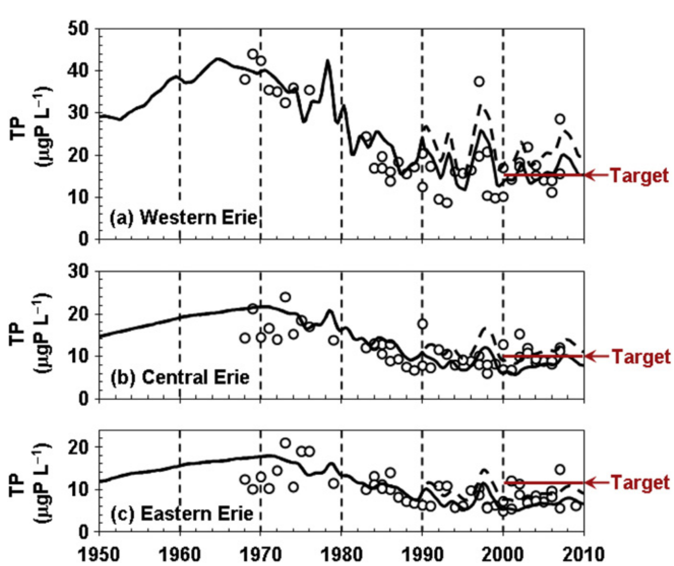

Fig. 2. Simulation results for TP concentration (μg P L^{-1}) versus year for the three basins of Lake Erie. Model simulations are designated by lines and measured data by markers. Two model simulations are shown for each basin. The upper dashed line on each plot employs a constant apparent settling velocity whereas the lower solid line uses a higher apparent settling velocity after 1990. The IJC target concentrations are indicated at the right of each plot.

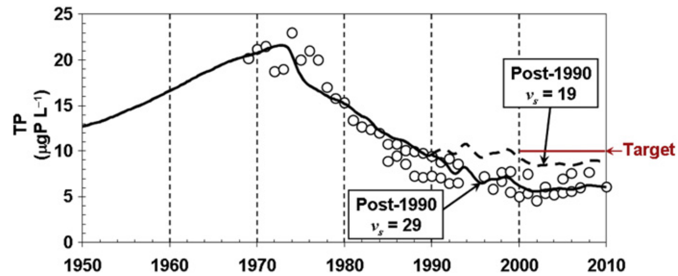

Fig. 3. Simulation results along with data for TP concentration (μg P L^{-1}) versus year for Lake Ontario. Model simulations are designated by lines and measured data by markers. Two versions of the model are indicated. The upper dashed line employs a constant apparent settling velocity of 19 m yr^{-1} for the entire simulation. The lower solid line uses the same value until 1990 when the settling velocity is increased to 29 m yr^{-1}. The GLWQA offshore target concentration is indicated at the right of the plot.

Application results

Load–response curves

The calibrated model was run at steady state using average flows and total phosphorus (TP) loadings (Dolan and Chapra, 2012) from water years 2001–2010. The average values in

Table 5 resulted. The model can then be used to compute the response of three trophic status indicators to total Lake Erie TP loads (Fig. 4).

Uncertainty/Sensitivity assessment

At this time, a full uncertainty/sensitivity assessment has yet to be conducted; and it is hoped that this would be implemented in the future. Nevertheless, a simple sensitivity analysis is provided by computing the steady-state system response matrix (Chapra and Sonzogni, 1979; Chapra et al., 1983) for TP. As depicted in Fig. 5, each element of the inverse $a_{i,j}^{(-1)}$ gives the response of lake i to a unit load to lake j (μgP L^{-1} per 1000 metric tonnes per annum, MTA). Thus, because the model is linear, each element provides the sensitivity of each basin's concentration response to a 1000 MTA load change for each other basin. For example, the plot indicates that if the loading to the western basin was lowered by 1000 MTA, then spring TP concentrations in the western, central and eastern basins of Lake Erie and the offshore of Lake Ontario would be lowered by 2.62, 0.722, 0.474, and 0.117 μgP L^{-1}, respectively.

Aside from such sensitivity calculations, the matrix can be employed to provide a detailed breakdown of how each lake's TP loads impact every other segment or lake as displayed in

Table 6. For example, the table indicates that about half ($= 4.07/8.15 \times 100\% = 49.9\%$) of the central Lake Erie's current concentration of 8.15 μgP L^{-1} is due to western Lake Erie loadings. Thus, reducing western basin loads would not only benefit that basin, but would induce significant water quality improvement in the central basin as well.

We can estimate the impact of the proposed loading reductions to offshore P concentrations using this sensitivity analysis. According to Maccoux et al. (2016), the 2008 (i.e., the base year for load reductions) load to the western basin from tributaries was 4457 MT and the load to the central basin from tributaries was 2321 MT. Assuming the 40%

Table 5
Average values for trophic state variables computed with the steady-state version of the model and current loading scenario. SD is Secchi disk depth.

Lake	Indices	TP (μg P L^{-1})	Chl *a* (μg L^{-1})	SD (m)
Western Erie	w (1)	16.00	2.63	1.55
Central Erie	c (2)	8.15	1.49	2.27
Eastern Erie	e (3)	7.03	1.31	2.46
Ontario	o (4)	6.37	1.21	2.60

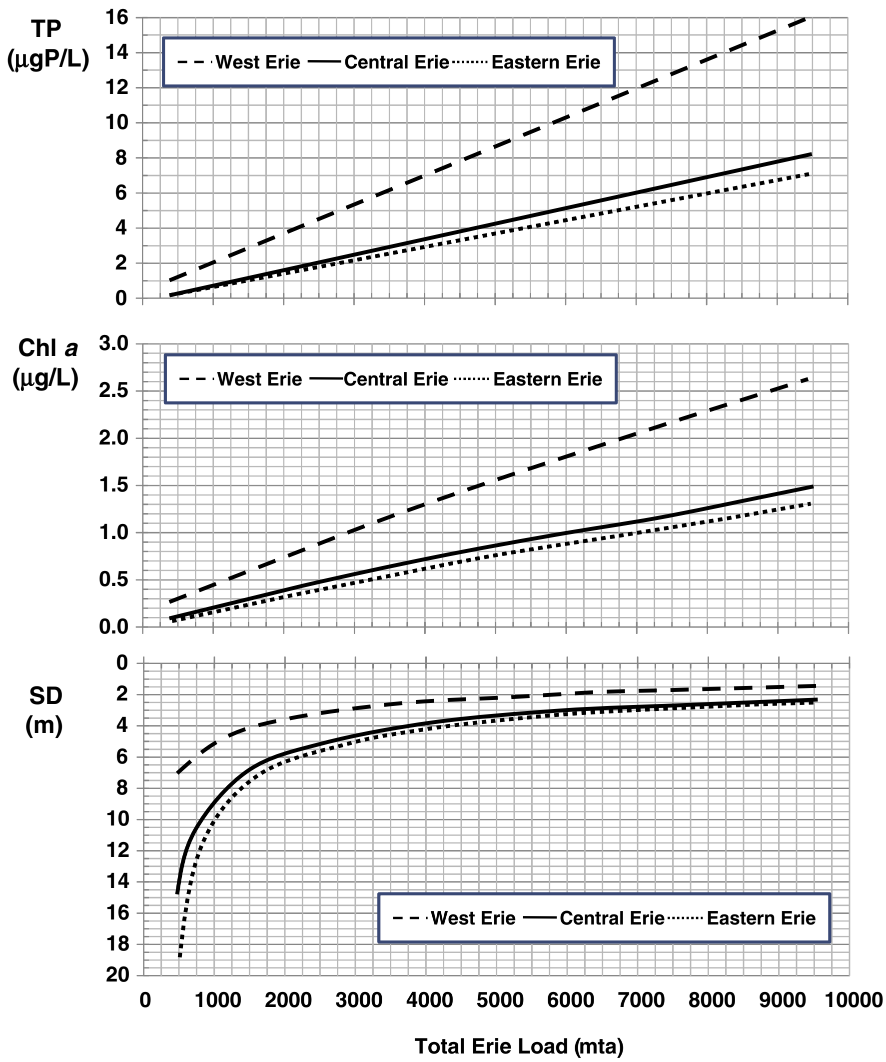

Fig. 4. Loading response plots for spring TP concentration, summer chlorophyll *a* concentration, and summer Secchi disk depth versus total Lake Erie TP loadings for the three basins of Lake Erie.

reduction in western and central tributary loadings is fully achieved, and the other loads remain unchanged, we have calculated the resulting spring offshore TP concentrations using the steady state system response matrix shown in Fig. 5. For example, for western Lake Erie, the result is

$$\begin{aligned} c_w &= 16-(0.4\times 4.457\times 2.6176)+(0.4\times 2.321\times 0.078302) \\ &= 16-(4.667+0.073)=16-4.74=11.26\ \frac{\mu\mathrm{gP}}{\mathrm{L}}. \end{aligned} \tag{14}$$

$$\begin{array}{ll} & \overbrace{\begin{array}{cccc}\text{West Erie} & \text{Cent Erie} & \text{East Erie} & \text{Ontario}\end{array}}^{\text{Loading (mta) to}} \\ & \begin{array}{cccc}\downarrow & \downarrow & \downarrow & \downarrow\end{array} \\ \text{Response } (\mu\text{gP/L}) \text{ of } \left[\begin{array}{r}\text{West Erie}\rightarrow \\ \text{Cent Erie}\rightarrow \\ \text{East Erie}\rightarrow \\ \text{Ontario}\rightarrow\end{array}\right. & \left[\begin{array}{cccc} 2.6176 & 0.078302 & 0.026319 & 0 \\ 0.72249 & 1.3737 & 0.46173 & 0 \\ 0.47378 & 0.90083 & 2.0049 & 0 \\ 0.11666 & 0.22182 & 0.49369 & 1.2744 \end{array}\right] \end{array}$$

Fig. 5. The TP steady-state system response matrix for the lower Great Lakes. Matrix predicts decrease of spring TP per 1000 MTA decrease in loads by model segment.

This and the other results are listed in Table 7. The greatest impacts remain in Lake Erie, where the western basin spring TP concentration would be reduced to approximately 11.3 μgP L^{-1} (or 30% of the present steady state concentration of 16 μgP L^{-1}) and the central basin mean concentration would be reduced to 5.6 μgP L^{-1}.

The impact in Lake Ontario is more muted than in Lake Erie, with a 6% decline in offshore spring TP concentrations predicted. However, this change may exacerbate declines that are already underway. Current spring TP concentrations in Lake Ontario's offshore are well below the target (Fig. 3). Using the same dataset we drew upon here, Dove (unpublished data) has updated the trends using the most current information and determined that spring, offshore TP in Lake Ontario over the period of 1998–2015 has been declining at a rate of approximately 0.06 μgP L^{-1} yr^{-1}, which is approximately 1% yr^{-1}. A further decline due to reduced loadings from Lake Erie via the Niagara River could exacerbate this oligotrophication already being observed.

The state of nutrient science in Lake Ontario is currently being considered by the Lake Ontario Nutrient Targets Task Team under GLWQA Annex 4. The impacts from reduced TP loads from Lake Erie will necessarily be considered in this assessment, and recommendations will need to be made in order to more fully assess the possible ramifications to the lake's nutrient status and its productivity.

Table 6
Detailed disaggregation of the impact of each major basin in the lower Great Lakes on the response (μg L^{-1}) of every other basin as determined with the steady-state version of the model.

	From Huron	Western Erie	Central Erie	Eastern Erie	Total Erie	From Erie	Ontario Direct	Total Ontario
Loads (mta) →	397.8	5633.3	2426.5	990.4	9448.0	1358.2	3641.2	4999.4
Western Erie	1.04	14.75	0.19	0.03	16.00	0.70		
Central Erie	0.29	4.07	3.33	0.46	8.15	0.54		
Eastern Erie	0.19	2.67	2.19	1.99	7.03	0.49		
Ontario	0.05	0.66	0.54	0.49	1.73	1.73	4.64	6.37

Discussion

Our finding that the apparent net "sedimentation velocity" determined through model calibration increased around 1990 warrants further discussion. This finding was originally reported and discussed in detail by Chapra and Dolan (2012). That work concluded that higher TP retention was absolutely necessary in order for the parsimonious TP budget model (i.e., with one phenomenological calibration parameter: the apparent sedimentation rate) to accurately match the post-1990 TP trends. This was especially true for Lake Ontario which exhibited the greatest divergence between TP measurements and model results after 1990.

Although it was noted that a definitive explanation for the increased TP trapping could not be based on such a simple phenomenological model, it was further noted that the increase in TP assimilation did coincide with the establishment of dreissenid mussels (viz., *Dreissena polymorpha* and *D. bugensis*) in the calcareous Great Lakes (i.e., excluding low calcium Lake Superior) in the early 1990s (Griffiths et al., 1991; Mills et al., 1993; Effler et al., 1996; Nicholls, 2001; Dove, 2009).

As suggested by others (notably, Hecky et al., 2004; Vanderploeg et al., 2010), dreissenid filtering provides enhanced transport of particulates to the lake bottom. If this retention is permanent (i.e., there is minimal subsequent recycle to the surface waters), this mechanism may account for the increased apparent sedimentation velocity required to match TP concentration trends over the past two decades. In particular, the filtering of the quagga mussels, which tend to colonize primarily soft sediments in deeper waters, may represent an efficient mechanism for permanently trapping TP in the profundal sediments.

Of course, for such a complex system, other ancillary mechanisms might arise to enhance and amplify the direct impact of the mussels on TP retention. For example, increased phosphorus uptake by bottom plants (e.g., both filamentous algae and macrophytes) in the nearshore might ultimately become sequestered in offshore bottom sediments when the plants die and slough. In addition, as the lakes become more oligotrophic, changes in phytoplankton community composition (e.g. a shift from buoyant cyanophytes or neutral chlorophytes to sinking diatoms) could also contribute to the observed increase in trapping. Finally, it is conceivable that a change in the nature of phosphorus loads (e.g. dissolved versus particulate, or particle size) could also affect TP retention.

Table 7
Predicted system response to a scenario of 40% reduction of 2008 central and western basin tributary loadings. Resulting TP spring, surface, offshore concentrations μgP L^{-1} are provided, with percent reduction from current steady-state values provided in parentheses.

	Western Erie		Central Erie
2008 Tributary load (mta) →	4457		2321
Reduced load scenario (mta)	2674		928
Western Erie		11.26 (30%)	
Central Erie		5.59 (31%)	
Eastern Erie		5.35 (24%)	
Ontario		5.96 (6%)	

Regardless of the underling mechanisms, it is unequivocal (Dove and Chapra, 2015) that most of the offshore waters of the Great Lakes have exhibited much greater response to phosphorus controls than had originally been predicted (e.g., in the original IJC model projections documented by Bierman, 1980). If the mussels and other ancillary mechanisms have in fact enhanced phosphorus trapping in Lake Ontario, it raises the further question of whether this mechanism is a temporary or a permanent, long-term (i.e., multi-decadal) phenomenon. It is well-known that invasive species often exhibit a heightened initial impact that is eventually attenuated by environmental feedbacks. If, over the following decades, the impact of dreissenid mussels becomes less pronounced, the apparent sedimentation velocity could eventually return to pre-1990 levels (~19 m/yr) in which case the trophic state of Lake Ontario should eventually rise to the oligo-mesotrophic state with (TP ≅ 10 μgP/L). Alternatively, and as has been the experience for the past 25 years, enhanced phosphorus trapping may be here to stay, and lake managers will need to take this sink into account when setting nutrient targets. The desire to ensure the continued productivity of Lake Ontario underscores the need for continued systematic monitoring of both loadings to the lake and in-lake concentrations over the coming years as well as the application of more sophisticated, process-based modeling approaches to better understand nutrient assimilation and permanent storage in lake sediments.

The magnitude of this sink may need to be further investigated given the differences between interlake transfers estimated using offshore Great Lakes values and those generated by the Great Lakes interconnecting channels monitoring data from ECCC. As observed and discussed by Chapra and Dolan (2012), starting in about 1990, there is a lack of correlation between the mid-Eastern Lake Erie values and the corresponding TP concentrations in Niagara River. In particular, the post-1990 Niagara River values are noticeably higher than the mid-lake estimates. Chapra and Dolan (2012) developed several hypotheses to explain the discrepancy; regardless, higher interlake transfers could have significance to our model predictions as well as for Lake Ontario nutrient management. In particular, if the Niagara River interlake transfer is higher than that estimated by the east Lake Erie concentrations, an even-higher apparent settling velocity would be required for Lake Ontario after 1990.

Summary and further steps

Although our model is parsimonious, it provides a long-term perspective that nicely complements the other finer scale models employed by the GLWQA Annex 4 ensemble modeling team to inform phosphorus load target-setting decisions for Lake Erie. Further, it is based upon updated and reliable observations of total phosphorus concentrations monitored by Environment and Climate Change Canada. Finally, the model provides a direct link to earlier attempts to set phosphorus loading in the 1978 Great Lakes Water Quality Agreement (International Joint Commission, 1978; Bierman, 1980).

There are two primary research and development needs for this model. First, it should be modified to predict hypolimnetic oxygen depletion in the central basin as has been done elsewhere (Chapra and Canale, 1991). Second, a complete and thorough uncertainty/sensitivity analysis of the results should be implemented.

Finally, an earlier effort was made to couple our modeling approach with economic data to generate an optimal least-cost strategy for Great Lakes phosphorus load reductions (Chapra et al., 1983). Although this scheme was never implemented, such system analysis or operations research methods could certainly still be applied and implemented in order to attain water quality objectives as inexpensively and effectively as possible.

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