DEVELOPMENT OF THE WESTERN LAKE ERIE ECOSYSTEM MODEL (WLEEM): APPLICATION TO CONNECT PHOSPHORUS LOADS TO CYANOBACTERIA BIOMASS

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Abstract

Since the mid-1990s, Lake Erie has experienced re-eutrophication symptoms including harmful algal blooms in the Western Basin and summer hypoxia in the Central Basin. The 2012 Protocol for the Great Lakes Water Quality Agreement (GLWQA) required phosphorus objectives and management recommendations to be set for all the Great Lakes, beginning with Lake Erie. To inform setting revised loading targets for the Lake Erie portion of the GLWQA, modeling was performed. The development and application of one of those models, the Western Lake Erie Ecosystem Model (WLEEM), is described here. WLEEM is a three dimensional, fine-scale, process-based model that links hydrodynamic, sediment transport, and in-lake biogeochemical and ecological processes. WLEEM was applied here to assess system sensitivity to a range of variables, and ultimately to develop a robust phosphorus load – cyanobacteria response relationship to determine a maximum load of total phosphorus from the Maumee River during the period of March – July that would produce a mild cyanobacteria bloom (<7830 MT cyanobacteria biomass) in Western Lake Erie. The maximum total phosphorus load from the Maumee River for that period to produce a mild bloom was determined to be 890 metric tons. Given the natural variability of systems like this, tools like WLEEM used in a dynamic operational modeling mode, consistent tributary and lake monitoring, and ongoing research will be essential components of effective mitigation and science-based adaptive management of eutrophication in Lake Erie and other nutrient-impacted water bodies.

Keywords: cyanobacteria, Lake Erie, phosphorus, eutrophication, ecosystem model

Introduction

Being relatively shallow and having a large watershed to surface area ratio, Lake Erie is especially susceptible among large lakes to eutrophication symptoms resulting from excess nutrient loading. Through the early 1970s, excess phosphorus loadings to Lake Erie (DePinto et al., 1986; Dolan, 1993) produced significant eutrophication symptoms, such as nuisance phytoplankton blooms (Makarewicz et al., 1993) and hypoxia in the Central Basin (Burns et al., 2005; Bertram, 1993).

By the mid-1980s, the improvement of Lake Erie's eutrophication problems in response to the objectives of the Great Lakes Water Quality Agreement (GLWQA) was hailed as a major success story (DePinto et al., 1986; DiToro et al., 1987; Dolan 1993). However, in the mid-1990s Lake Erie began to experience what many called re-eutrophication symptoms. Central Basin hypoxia began to become more severe (Scavia, et al., 2014). Large and increasingly toxic blooms of cyanobacteria began to be observed in Lake Erie in 1998 and have continued since then (Stumpf et al., 2012). The largest cyanobacteria bloom on record was observed in 2011 (Michalak et al., 2013; Bridgeman et al., 2013), and the 2015 bloom recently eclipsed the 2011 record (Stumpf et al., 2016). A number of hypotheses have been offered as to the cause of the cyanobacteria bloom reoccurrence in Lake Erie (Michalak et al., 2013; Paerl and Otten, 2013; Smith et al., 2015); however, it seems that the major driver in Western Lake Erie is the spring load of bioavailable phosphorus from the Maumee River that is derived primarily from agricultural nonpoint sources (Baker et al., 2014a; Baker et al., 2014b; Stumpf et al., 2012; Obenour et al., 2015).

The 2012 Protocol for the GLWQA (Canada and United States, 2012) includes Annex 4 on Nutrients, in particular calling for phosphorus control to achieve ecosystem objectives related to eutrophication symptoms. To accomplish this charge for Lake Erie by the end of 2015, an Annex 4 Objectives and Targets Task Team was formed in September, 2013 to focus on Lake

Erie. 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 to 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 (Scavia et al., 2016).

This paper describes the development and application of a model which was developed to address nutrient problems throughout the Great Lakes, including eutrophication and hypoxia issues in Green Bay, Saginaw Bay, and Western Lake Erie. These types of models need to be able to link the water quality model frameworks to a fine-scale hydrodynamic model using the same grid resolution as the water transport model. Several other modeling groups have used a similar concept in previous Great Lakes modeling work (Schwab et al., 2009; Pauer et al., 2011; Leon et al., 2011; Leon et al., 2012; Bocaniov et al., 2014). The Western Lake Erie Ecosystem Model (WLEEM) was developed to provide a framework that could quantitatively relate the response of water quality conditions in Western Lake Erie to the full range of stressors including hydrology, sediment and nutrient external loads from internal and external sources, and wind and other climatic stressors. For example, a question exists as to the importance of the Detroit River, which has an annual load similar to that of the Maumee River, in stimulating western basin cyanobacteria blooms. Also, there are questions whether control of the loading of soluble reactive phosphorus (SRP) alone is sufficient to achieve a desired reduction in those cyanobacteria blooms. The process-based WLEEM can address these and other questions.

This paper presents the development of the WLEEM and calibration/confirmation of the model to available data for Western Lake Erie. Also presented in this paper is an application of WLEEM to develop a load-response relationship between nutrient loads to Western Lake Erie and the cyanobacteria bloom biomass Ecosystem Response Indicator (ERI) in support of the Annex 4 evaluation and the multiple model analysis (Annex 4 Objectives and Targets Task

Team, 2015). WLEEM was the only process-based model applied to simulate explicit nutrient links to harmful algal blooms (HABs). The other two cyanobacteria biomass models are statistically-based empirical models, also presented in this special issue (Stumpf et al., 2016; Bertani et al., 2016).

Methods

Study Site & Field Data

The study area comprises the western basin of Lake Erie, which includes an area bounded by a line from Point Pelee in Ontario to the end of the Marblehead Peninsula in Ohio and all waters of Lake Erie to the west. The surface area of this region of the lake is 2,980 km² with an average depth of 7.5m. Major tributaries within this part of the lake include the Detroit River, which carries additional outflow from the upper Great Lakes and St. Clair River/Lake system (including the Thames River in Ontario), and the Maumee River. The City of Toledo, Ohio is located at the mouth of the Maumee River, while the City of Detroit is located on the north end of the Detroit River. Water residence times within western Lake Erie are largely driven by the rather steady outflow from the Detroit River, which averages 5,324 m³/s. The average residence time of water within the western basin is approximately 50 days. The shallow nature of Western Lake Erie, particularly the shallow areas near the mouth of the Maumee River, makes it susceptible to the occurrence of HABs during the warm summer months (Wynn and Stumpf, 2015).

The primary dataset used for model calibration was obtained from the University of Toledo (UT). Data were collected by the University of Toledo during 2008, and from 2011 to 2014 by Bridgeman (2013). Coordinates of monitoring stations are available in Bridgeman (2013) with the locations shown on Figure 1. Depth-integrated observations for TP, SRP,

NO₂+NO₃, chlorophyll a, and cyanobacteria biovolume were compared against model results.

Biovolume was converted to biomass using the methods found in Bridgeman et al. (2013).

Model Description

WLEEM is a three dimensional, fine-scale, process-based, linked hydrodynamic-sediment transport-advanced eutrophication model that computes the time- and space-variable relationships between hydrometeorological conditions, inputs of tributary flow, nutrients, suspended solids, nutrient concentrations and mass budgets, phytoplankton biomass distributed among three functional groups, dissolved oxygen, and light extinction in the western basin of Lake Erie. WLEEM is composed of the following four linked sub-models: Simulating Waves Nearshore (SWAN) for the wind-wave sub-model, Environmental Fluid Dynamics Code (EFDC) for the hydrodynamic and temperature sub-model, Sediment Transport Model (SEDTRAN) algorithms for the sediment transport sub-model, and Advanced Aquatic Ecosystem Model (A2EM) for the water quality sub-model. The linked modeling framework comprised of sub-model applications of SWAN, EFDC, SEDTRAN, and A2EM, collectively referred to as WLEEM, provides a powerful and flexible tool for evaluating hydrodynamic, wind-wave, sediment transport, and nutrient and phytoplankton processes at a variety of temporal and spatial scales. Figure 2 shows the relational nature of the sub-model connections. Each sub-model is explained in further detail below.

SWAN is a numerical model for predicting wave conditions in coastal areas, lakes, and estuaries based on site-specific wind, depth, friction, and water velocity conditions (Young, 1999; Booij et al., 1999). As part of the WLEEM development effort, SWAN was linked to the EFDC and SEDTRAN sub-models to provide wave forcings for circulation and wind-driven resuspension impacts. The SWAN-EFDC-SEDTRAN linkage involved two steps: 1) water level and velocity results generated by the EFDC model were processed and input as forcing functions to the SWAN wind-wave simulations, and 2) SWAN results for wave height, period, and

direction were fed as input forcing functions to the SEDTRAN and A2EM sub-model to inform calculations of bottom shear stress, which support the simulation of wind-driven resuspension of bottom sediment and associated organic matter and nutrients.

EFDC is a state-of-the-art finite difference model that can be used to simulate hydrodynamic and thermodynamic behavior in one, two, or three dimensions in riverine, lacustrine, and estuarine environments (TetraTech 2007a, 2007b). The EFDC model input parameters include variable hydrologic inflows, water level boundary conditions, and atmospheric conditions such as air temperature, relative humidity, solar radiation, and wind speed. EFDC represents the model grid using a curvilinear orthogonal grid for the horizontal dimensions and the generalized vertical coordinate (GVC) system for the vertical dimension. The GVC system utilizes a varying number of layers in the vertical dimension depending on horizontal grid location; and therefore, it is well-suited for accurately describing vertical gradients in systems with rapidly changing bathymetry, such as occurs near the Toledo Harbor navigation channel.

The version of the SEDTRAN model used here incorporates a custom sediment transport sub-model based on the SEDZLJ model algorithms developed by Jones and Lick (2001). The SEDTRAN sub-model can be used to simulate sediment transport in one, two, or three dimensions. It provides a flexible set of options for simulating sediment settling/deposition, resuspension/erosion, bed armoring, and three-dimensional transport of multiple cohesive and non-cohesive sediment classes. The integration of the SEDZLJ code into LimnoTech's in-house version of the Row-Column AESOP (RCA) model code framework and associated code testing was accomplished previously (LimnoTech, 2010; LimnoTech, 2013).

A2EM is an aquatic ecosystem eutrophication simulation model customized by LimnoTech, which is based on a publicly available version of the RCA model developed and documented by HydroQual, Inc. (HydroQual, 2004). A2EM is capable of simulating water

quality dynamics on a fine-scale, multi-dimensional computational grid based on linkage to EFDC. The basic model framework includes a suite of state variables to represent carbon, nitrogen, phosphorus, oxygen, and phytoplankton dynamics. All the parameter coefficients used in WLEEM are available in Electronic Supplementary Material (ESM) Table S1.

LimnoTech has built upon the publicly available RCA model and added the capability to simulate up to five phytoplankton functional groups, three of which are used in this model application (blue-greens, summer assemblage, and winter assemblage). Three zooplankton functional groups (documented in LimnoTech, 2009), two dreissenid mussel classes and their effects on nutrient cycling (Bierman et al., 2005), and a benthic algal class based on the Great Lakes *Cladophora* Model (Auer et al., 2010) are also included in the model framework. For this application in Western Lake Erie, the zooplankton and Dreissenids submodels were activated; but the benthic algal model was not used given the high turbidity and low abundance of benthic algae in the western basin. Refinements were also made to particulate phosphorus processes to allow WLEEM to accurately represent phosphorus adsorption/desorption dynamics and conversion of particulate phosphorus to bioavailable forms (DePinto et al., 2009; LimnoTech 2010; LimnoTech 2013). The model framework also includes a coupled sediment diagenesis sub-model based on DiToro (2001) that simulates the cycling of detrital material and nutrients in the surface sediments as well as related impacts on near-bed sediment oxygen demand and release of dissolved nutrients, including dissolved inorganic phosphorus.

A custom linkage was created to connect output from the hydrodynamic (EFDC) and sediment transport (SEDTRAN) models to the water quality (A2EM) sub-model, allowing for a progressive simulation of circulation, temperature, sediment dynamics, water quality and eutrophication dynamics at a common spatial scale. The capabilities of A2EM are more sophisticated than most other nutrient/phytoplankton/zooplankton (NPZ) models because it pays particular attention to nutrient cycling of resuspended material and the potential influence of

dreissenids, benthic algae, and sediment diagenesis and sediment-water diffusion processes on the nutrient mass balance.

Model setup

The model domain and study area includes Western Lake Erie, which is delineated by a line from Point Pelee, Ontario to Marblehead, Ohio. The model computational grid developed for WLEEM, depicted in Figure 1, is based on a curvilinear orthogonal design. Particular attention to grid orientation and resolution was given to the Toledo Harbor navigation channel to ensure that the dredged channel width and depth are well represented in the model geometry. Grid cells on the eastern end of the domain are approximately 2km wide, while cells in Maumee Bay are approximately 300m wide. There are a total of 3,999 horizontal grid cells in the model domain. In the vertical dimension, one layer is added per three meters of depth up to a maximum of four layers.

Model Boundary Conditions

Three types of external forcings were applied as boundary conditions to the hydrodynamic model, including: water level and temperature at the open east boundary of the model domain; flow and temperature for the Maumee River, Detroit River, and six minor tributaries; and atmospheric forcing functions (such as wind speed and air temperature) applied over the entire model domain.

A dynamic water level boundary was applied as a forcing condition at the interface between the central and western basins of Lake Erie. Hourly data from NOAA station 9063079 (Marblehead, Ohio) were applied uniformly across the open boundary. Tributary inflows were developed using available data from the United States Geological Survey (USGS). In many cases, the location of the USGS gauge within a tributary did not represent the entire drainage area of a given tributary, nor were gauged data available for all watersheds. Therefore, simple drainage area ratios were used to scale the daily measured flow time series to estimate daily

flow for the entire watershed. The USGS gauge locations and the flow scaling factors used are shown in Table 1.

Atmospheric conditions were extracted from the Great Lakes Observing System (GLOS) point query tool website. This tool allows a user to extract model inputs from the NOAA Great Lakes Environmental Research Laboratory supported Great Lakes Coastal Forecasting System (GLCFS), which interpolates measured weather data across the open waters of the Great Lakes at a 2 km resolution (Schwab et al., 1999). Hourly average wind time series were extracted from ten locations within the WLEEM domain. A Thiessen polygon analysis was then performed on the model grid and the wind forcing locations so that each grid cell in the WLEEM grid was attributed with weighting factors for the neighboring wind forcing locations. Hourly average air temperature, dew point, barometric pressure, and cloud cover were also extracted from the GLCFS for one location near the center of the model domain. Cloud cover and day of year information were used to estimate solar radiation based on the formulation found in Blumberg and Mellor (1987), which is also the same method used by Beleskey et al. (2013).

Suspended solids concentration boundary conditions were developed for each tributary and the open boundary of Western Lake Erie. Initial organic carbon and nutrient concentrations and other parameters were also developed for the sediment bed represented in the model. An extensive suspended solids dataset is available for the Maumee River at Waterville, Ohio, based on long-term research conducted by Heidelberg University's National Center for Water Quality Research (NCWQR) (Baker et al., 2014a). Several other tributary inflows are represented in WLEEM in addition to the Maumee River, including the Detroit River, Ottawa River, River Raisin, Huron River, Stony/Otter Creek, Portage River, and Cedar Creek (see Table 2). In addition, nutrient loading from the Toledo Bay View Wastewater Treatment Plant (WWTP) is represented. The boundary condition for the Detroit River was set at a constant value of 10 milligrams per liter (mg/L) based on a review of available data for this Great Lakes connecting

channel (MDEQ 2006). The open boundary condition at the interface between the Western and central Lake Erie basins is characterized with a constant concentration of 10 mg/L based on available monitoring data from the International Field Year in Lake Erie datasets (Hawley and Eadie, 2007).

The A2EM uses the same open boundary and eight tributary boundary locations as the hydrodynamic and sediment transport sub-models. Daily estimated concentrations of major nutrients, dissolved oxygen, and phytoplankton are applied at every boundary location. Nutrient concentrations associated with Maumee River flow were derived from daily measurements made by NCWQR.

Nutrient concentrations for the Detroit River were based on monitoring by the Michigan Department of Environmental Quality (MDEQ). The latest published report of observations summarizes monitoring data through 2008 (MDEQ 2013). The Detroit River loads estimated by this method were consistent with the load estimates of Dolan and Chapra (2012) and Maccoux et al. (2016). Concentrations of nutrients in other minor tributaries were based on loading estimates made by Dolan and Chapra (2012) and Maccoux et al. (2016). A table of the flow weighted mean concentrations for the major tributaries included in the model for total suspended solids (TSS), total phosphorus (TP), soluble reactive phosphorus (SRP), nitrate plus nitrite ($\text{NO}_2 + \text{NO}_3$), and total Kjeldahl nitrogen (TKN) concentration are included in Table 3. The annual average TP load, and percent of the total load western basin load in parentheses, from the Detroit River, Maumee River, and other tributary inflows is 2,653 MT (48%), 2,315 MT (42%), and 555 MT (10%), respectively for the 2008, and 2011 to 2014 simulation period. Although the Detroit and Maumee rivers account for the majority of TP delivered to Western Lake Erie, the average flow from the Detroit River is significantly larger than that of the Maumee. This large flow difference is the reason the flow-weighted mean concentration of TP for the Maumee River is 400 $\mu\text{gP/L}$ while the Detroit River concentration is 20 $\mu\text{gP/L}$. Monitoring data from USEPA's open-lake

limnology program, extracted from the Great Lakes Environmental Database, were used to define nutrient concentrations at the open boundary.

Zooplankton boundary and initial conditions for cladocerans, copepods, and microzooplankton were developed using approximations from site specific data collected from Saginaw Bay during a previous A2EM application to Saginaw Bay. Zooplankton densities generally followed observations by Pothoven et al. (2013). Initial dreissenid densities across Western Lake Erie range between 0 gC/m²) and 0.25 gC/m². These ranges were based on dreissenid data contained in Karatayev et al. (2014). Mussel density was interpolated across Western Lake Erie based on sediment type and bathymetry data presented in LimnoTech (2010).

Model Simulation

The model simulation periods included March through October of 2008, 2011, 2012, 2013, and 2014. The 2008 year was chosen by the Annex 4 modeling subcommittee to represent an average HAB severity with a whole lake TP loading equivalent to the 1978 GLWQA target of 11,000 MTA, while 2011 to 2014 represent the latest years for which a complete set of tributary boundary conditions and in-situ monitoring data were available. All of the models within the WLEEM framework, including hydrodynamics, waves, temperature, sediment transport, and water quality were run for these periods.

Model Calibration and Confirmation Approach

Calibration of WLEEM was performed using a weight-of-evidence approach that consisted of multiple model comparisons, both graphical and statistical, to assess model performance (Thomann and Mueller, 1982). Model calibration focused first on physical process parameters (hydrodynamics, temperature, suspended solids), followed by chemical process parameters (TP and SRP, nitrogen species), and finally biological process parameters (chlorophyll *a* and cyanobacteria biomass). The calibration involved iteration and coefficient

adjustments between chemical and biological parameters, as phytoplankton growth affects nutrient concentrations in the water column and vice versa.

Monitoring data from 2011-2013 were used initially for calibration. These three years represented a wide range of environmental forcing conditions including high and low flow and nutrient loads, and varying wind patterns. Flow, nutrients, and biomass were very high in 2011, low in 2012, and moderate in 2013. Independent model confirmation was achieved by simulating 2008 and 2014 conditions without any change in model coefficients. Year 2008 was chosen given its significance to the Annex 4 modeling group as a baseline year, and 2014 was selected due to the relatively large size of the predicted bloom and associated impacts on water treatment plants for that year. Taken together, the simulation of five years of varying meteorological and tributary loading conditions in Western Lake Erie provides a corroboration of the model's skill in reproducing observed system conditions under a wide range of forcing conditions. All model runs were conducted for the period of March 1 – November 30 of a given year.

Results

Calibration Results

As described in the model approach section above, WLEEM was parameterized to simulate conditions in Western Lake Erie for March through November of 2008 and 2011 to 2014. Figures 3 to 8 provide visual comparisons of model predictions against monitoring data collected during 2008, and 2011 to 2014 at the University of Toledo's stations MB18, 8M, and GR1 for water temperature, TP, SRP, $\text{NO}_2 + \text{NO}_3$, total chlorophyll *a*, and cyanobacteria biomass. These three stations represent a gradient in conditions moving from the mouth of the Maumee River out to the open waters of Western Lake Erie.

Water temperature calibration was achieved by choosing representative forcing conditions for air temperature and solar radiation. Given the large surface area of the Western Basin, air temperature and solar radiation are two key model input time series that

directly affect the simulated water temperature. Figure 3 shows that the model reproduces measured water temperature very closely. The model accurately reproduces increases in water temperature in the spring and early summer, timing of peak temperatures, and decreases in the late summer and early fall.

Calibration of the sediment transport sub-model focused on confirming that the model representation of settling and deposition rates for suspended solids and the simulation of regional-scale resuspension events produce results consistent with available suspended solids data. Data and model results typically demonstrate an expected trend of elevated suspended solids concentrations in the vicinity of the Maumee River mouth, with concentrations declining with increasing distance from the mouth. Beyond Maumee Bay turbidity observations demonstrate lessening influence from the Maumee River and other tributaries and evidence of periodic wind-driven resuspension events.

Calibration of phosphorus and nitrogen first focused on ensuring accurate time-variable tributary loads were developed for tributaries with a small number of nutrient observations. These included the Detroit, Huron, Stony-Otter, Ottawa, Portage, and Cedar rivers. Western Lake Erie has a relatively short retention time of 30 to 60 days (Michalak et al., 2013), especially during high flow events, and tributaries can strongly influence conditions in Maumee Bay. Nutrient concentrations within the sediment bed were also parameterized based on available data (LimnoTech, 2014) and modified spatially to be more consistent with known variability in bottom sediment type. Adjustments to initial sediment TP concentrations were made so that the relative contributions of resuspended sediments and pore water diffusive flux from the sediments to water column TP concentrations were reasonable. Further calibration focused on adjustments in settling and resuspension rates of particulate P and N, which can decrease or increase TP and TN concentrations, respectively, in the water column. Calibration of SRP and $\text{NO}_2 + \text{NO}_3$ was

conducted coincident with phytoplankton calibration, as the uptake of SRP and NO_2+NO_3 by phytoplankton is the primary loss mechanism for these constituents within WLEEM.

The evaluation of nutrient state variables focused on comparing model predictions and monitoring data for TP, SRP, and NO_2+NO_3 . Time series plots for 2011, 2012, and 2013 for these parameters are shown below for stations MB18, 8M, and GR1 (Figures 4 through 6). Figure 4 shows the TP response to high flow spring loading events, with concentrations being attenuated (diluted) with distance from the Maumee River mouth. Figure 6 demonstrates that the model reasonably reproduces behaviors shown in the observed SRP data, with high load-driven spring SRP concentrations followed by low summer SRP concentrations driven by algal uptake. Similarly, Figure 6 shows that the model captures the phenomenon of high water column NO_2+NO_3 concentrations in response to elevated tributary loading in the spring followed by a gradual decline over the summer months.

Calibration of chlorophyll *a* and phytoplankton biomass focused on ensuring growth of each major phytoplankton functional group (cyanobacteria, winter assemblage, and summer assemblage) occur at appropriate times of the year. Diatoms peak in the spring, greens peak in the early summer, and cyanobacteria peak in the middle to late summer. The chlorophyll *a* data provided a bulk estimate of the total phytoplankton biomass (i.e., across all three functional groups), while cyanobacteria biomass data were used to ensure that the model captured the size and magnitude of the harmful algal bloom (HAB).

Figure 7 shows the pattern of total chlorophyll *a* concentration for the system. Overall the model reproduces the temporal and spatial (i.e. higher concentrations in the inner bay and lower concentrations in the outer bay) patterns of the observed data. Due to a lack of spring (Mar-May) sampling data, model predictions of diatom-dominated chlorophyll *a* concentrations during this period are not well known. The higher late summer – early fall chlorophyll *a* levels, which are

dominated by cyanobacteria, are reflected in the measured data and simulated reasonably well by the model.

Comparisons of model predicted and measured cyanobacteria biomass are shown in Figure 8. These comparisons demonstrate that WLEEM captures the relative differences among stations and years; however, it over-predicts peak densities during the low spring load year (2012). The model also tends to initiate HAB development earlier than monitoring data suggest by approximately four weeks. This represents a gap in research more than a deficiency of the model, as water temperatures are warm enough and sufficient nutrients are available to fuel a bloom during the earlier period. The deficiency could also be attributed to a poor understanding of phytoplankton population dynamics prior to the initiation of a HAB. For example, the competition of green and cyanobacteria species for light and nutrients could play a role in delaying bloom development.

A subset of final calibrated model coefficients is presented in Table 4 for each algal group. Calibration focused on altering the first four constants related to shaping the optimal temperature curve as well as the phosphorus half-saturation constant that governs nutrient limited growth. In addition, the base phytoplankton settling rate was adjusted to reflect current scientific understanding of the buoyancy of cyanobacteria (Chu et al., 2007).

Confirmation Results

Following model calibration, a model confirmation process was conducted by applying the calibrated model, with no change in model coefficients, to the 2008 and 2014 confirmation years. The model-data comparisons for these two years (2008 and 2014), along with the calibration years (2011-13) discussed previously, are shown in Figures 3 to 8. Many of the observations described in the calibration results sections can be reiterated for the performance of WLEEM for the confirmation years. The model predicts the temporal patterns and peak water

temperatures very well. On a whole, the model reproduces water column TP and SRP concentrations reasonably well, but tends to under-predict concentrations during the August-September period. WLEEM reproduces the pattern of declining $\text{NO}_2 + \text{NO}_3$ concentrations from spring to summer, but tends to over-predict concentrations during the August-September period, while the UT data suggest that concentrations continue to decline. Although the model over-predicts cyanobacteria biomass at the UT stations for both 2008 and 2014, including bloom formation about one month early, it matches the spatial gradient in peak biomass densities across the monitoring stations.

As an additional model confirmation test, a seasonal average cyanobacteria biovolume statistic was computed and compared against data-based estimates. The method used to estimate the annual biovolume followed the approach described in Bridgeman et al. (2013). The result of those computations were then averaged across the University of Toledo's 4P, 7M, 8M, and GR1 stations (see Figure 1 for the station locations) to obtain the final seasonal estimate of cyanobacteria biovolume. Figure 9 compares the annual cyanobacteria biovolume computed by WLEEM against estimates computed from data provided by UT. Model comparisons were not available for 2009 and 2010, as WLEEM simulations have not been developed for those periods. The overall model-data comparison for these five years corroborates the model's ability to capture inter-annual variability of cyanobacteria biovolume across a wide range of conditions.

Quantification of calibration/confirmation error

Statistical evaluations were conducted to quantify the capability of the model to reproduce water quality and algal observations. Although statistical calibration targets have been developed and widely used for watershed and riverine modeling (Moriasi et al., 2007), there are generally no accepted guidelines for what constitutes a "statistically acceptable" model for large coastal systems such as Western Lake Erie. The development and calculation of representative

model performance metrics for water temperature, nutrients, and biological parameters are challenging for large water bodies where monitoring data tend to be sporadically available at a limited number of locations. Monitoring data collected bi-weekly by UT represent relatively infrequent snapshots of nutrient concentrations and cyanobacteria biomass at one location in space and time, and therefore cannot be expected to capture all of the complex spatial and temporal variability that might be occurring between stations and sampling trips. In addition, the hydrodynamic, sediment and nutrient transport, and biological processes represented in the model are only a simplification of complex interactions occurring in the actual system. Therefore, both data representativeness and model complexity limit the capability of a model to precisely reproduce observed concentrations. These limitations must be taken into consideration when interpreting statistical measures of model performance against sporadic grab sample data.

The statistics chosen to evaluate model performance were percent bias (PBIAS), relative percent difference, and relative percent error. Equations for each of these statistics are provided below in equations 1 to 4. Moriasi et al., (2007) recommended the following performance ratings for simulated nitrogen and phosphorus: $PBIAS < \pm 25$ as very good, $\pm 25 \leq PBIAS < \pm 40$ as good, $\pm 40 \leq PBIAS < \pm 70$ as satisfactory, and $PBIAS \geq \pm 70$ as unsatisfactory. For water temperature, Donigan (2000) cites the following performance ratings for relative percent difference between simulated and recorded values: $\leq 7\%$ as very good, 8-12% as good, and 13-18% as fair. For nutrients, Donigan (2000) specified the following relative percent difference targets: $\leq 15\%$ as very good, 15-25% as good, and 25-35% as fair.

$$PBIAS = \left[\frac{\sum_{i=1}^n (O_i - S_i) * (100)}{\sum_{i=1}^n (O_i)} \right] \quad (\text{Equation 1})$$

$$\text{Mean Relative Percent Difference} = \frac{1}{n} \sum_{i=1}^n \frac{S_i - O_i}{\frac{1}{2}(S_i + O_i)} \times 100 \quad (\text{Equation 2})$$

$$\text{Mean Relative Percent Error} = \frac{1}{n} \sum_{i=1}^n \frac{S_i - O_i}{O_i} \times 100 \quad (\text{Equation 3})$$

$$\text{RMSE} = \sqrt{\frac{1}{n} \sum_{i=1}^n (O_i - S_i)^2} \quad (\text{Equation 4})$$

A summary of model performance based on the metrics described above is provided in Table 5. These statistics represent annual values across all six UT stations with the exception of cyanobacteria biovolume, which was computed from the four UT stations listed in Figure 9. UT stations MB18 and MB20 did not have sufficient sampling of cyanobacteria in 2011 and therefore were excluded from the analysis. By using the annual average measurements from all monitoring stations and dates to compare against the corresponding simulated values, these statistics represent both a temporal and spatial aggregation of model performance for the combined calibration and confirmation periods. For many parameters and statistics shown in Table 5, model predictions fall within the “very good” range according to Moriasi et al. (2007) and Donigan (2000), and all comparisons indicate that the WLEEM calibration is acceptable. Combined with the visual comparisons shown above in the calibration and confirmation results sections, these statistical comparisons provide further evidence that WLEEM simulates physical, chemical/nutrient, and biological state variables to a level of accuracy that warrants the model’s use for developing load-response relationships for Western Lake Erie.

Model Application and Discussion

2008 baseline and P load reductions

The calibrated and confirmed WLEEM was used to evaluate the response of cyanobacteria to external nutrient loading to Western Lake Erie. Presented below are a description of the process used to develop load-response curves, the results of the model application to produce those curves, and a brief diagnostic analysis of the model results relative to the factors governing the load-response curves.

Load-response curves were developed by executing multiple simulations where the only difference from the baseline simulation was a scaling of the daily external phosphorus loads (accomplished by adjusting tributary concentrations, but not flow). Five baseline simulation years (2008 and 2011 through 2014) were used to generate the load response curves to represent a range of annual load and bloom conditions. The independent variable (x-axis) on the load-response curves was presented as either the annual total phosphorus load to Western Lake Erie from all sources or March through July phosphorus load (either TP or SRP load) from the Maumee River. The dependent variable (y-axis response variable) evaluated for the cyanobacteria blooms in Western Lake Erie was the basin-wide maximum 30-day rolling average cyanobacteria biomass in units of metric tons. This metric is a measure of the most severe 30-day bloom period. In addition to indicating the most severe impact of the bloom, this metric affords a comparison between WLEEM results and those produced by the empirical models (Stumpf et al., 2016; Bertani et al., 2016) that base their dependent variable on the cumulative biomass from thirty days' worth of satellite images.

The 2008 simulation year was used as the primary baseline for development of cyanobacteria load-response curves for Annex 4 because it represents the most recent year for which a thorough lake-wide, tributary-specific load compilation was produced (Dolan and Chapra 2012) and because the annual load for this year is very close to the Lake Erie target TP load of 11,000 MTA set in the 1978 Amendment to the GLWQA.

Maumee River spring (March – July) TP load adjustments to 2008 included six load adjustment scenarios representing 0%, 25%, 50%, 75%, 100%, and 125% of the baseline load. These load scenarios were developed by multiplying each daily load for 2008 by the same fixed percentage. A time series plot of cyanobacteria basin-wide, depth-averaged concentration ($\text{mg (dw)}/\text{m}^2$) for these six scenarios is presented in Figure 10. Due to the load scaling approach and the fact that none of the other forcing functions were changed, these time series plots have

the same shape, with the magnitude determined by the P load adjustment. It is evident that the modeled peak cyanobacteria biomass in 2008 occurred in early August, with the duration of the cyanobacteria dominance spanning roughly mid-July through mid-October. While the magnitude of the bloom is driven by phosphorus loading from the Maumee River, the inception and duration of the bloom is influenced by water temperature and circulation.

Load-Response Relationships

Three of the five years simulated by WLEEM for the calibration/confirmation (2011, 2013, and 2014) had higher cyanobacteria biovolume than 2008. One of the years (2012) had lower cyanobacteria biovolume than 2008 due to its significantly lower spring Maumee River load. Therefore, all of these simulation years were considered in the development of the WLEEM load-response curve to cover a greater potential range of loading conditions driving the cyanobacteria bloom. A plot of the maximum 30-day average biomass versus the spring (March – July) Maumee River TP load for all five years was developed and inspected (Figure 11). A strong linear relationship is evident, and it suggests two things: 1) the other drivers (i.e., wind, temperature, hydrology, and hydrodynamics) play a lesser role in determining the maximum bloom biomass relative to the spring Maumee River TP load; and 2) the spring Maumee TP load is a good choice for the independent variable in our load-response curve analysis.

Given the strong linear relationship shown in Figure 11 based on a wide range of Maumee River spring loads, it was reasonable to use all five years rather than solely using 2008 to construct a load-response curve to support the Annex 4 target P load recommendation to address cyanobacteria bloom reduction goals in Western Lake Erie. The load-response regression is based on WLEEM simulations using the 2008 baseline with adjusted Maumee River spring TP loads at 0%, 25%, 50%, 75%, 100%, and 125% and the 2011 – 2014 baseline conditions simulated with the loads from the Maumee River set at 50% and 100% of the actual load for that year. All other

forcing functions, including external loads from other WesternLake Eritributaries and the Detroit River, were maintained at the appropriate baseline condition for each of the five years.

The product of these model scenarios was the load-response curve shown in Figure 12. Like the load-response curve developed using only the baseline simulations (Figure 11), the load-response curve that includes TP load-adjustment scenarios demonstrates a strong linear relationship. In order to use this curve to produce a recommended target P load for achieving non-severe cyanobacteria blooms, the satellite-derived cyanobacteria index (CI) threshold of 2 (or 9600 MT biomass) suggested by Stumpf et al. (2016) was used. An adjustment was made to the 9600 MT threshold to account for differences in the calculation of the maximum 30-day average biomass between WLEEM and the Stumpf satellite-driven model. An equivalent biomass threshold of 7830 MT for WLEEM was determined by computing the ratio of the 9600 MT threshold to the biomass predicted for the 2008 Mar-Jul TP load by the Stumpf et al. (2016) empirical model and then applying that ratio to the biomass predicted for the 2008 Mar-Jul TP load by the WLEEM load-response curve.

With a slope of 8.37 MT of cyanobacteria biomass per MT of spring TP load from the Maumee River, the WLEEM load-response curve intersects the 7830 MT biomass threshold at a TP load of approximately 890 MT (Figure 12). This represents the P load target for achieving non-severe cyanobacteria blooms in WesternLake Erie. To achieve a bloom with a maximum 30-day average biomass equivalent to the 2012 bloom as predicted by WLEEM, a much lower Maumee River Mar-Jul TP load of approximately 400 MT would be necessary (also shown in Figure 12).

In addition to supporting management decisions, such as those discussed above, WLEEM, being a sophisticated process-based model, can be used to understand how the system is functioning and the relative role of various factors in affecting the model's response to

environmental and loading conditions. Presented below are some examples of the unique value provided by WLEEM and other similar process-based models.

Questions that are often asked about the load-response relationship presented above include: Why is only the Maumee River load included as the independent variable? What about the Detroit River load, which is approximately equal to the Maumee load? Why is only the March – July Maumee River load considered in the analysis?

For example, the first question regarding why the independent variable is based on the Maumee River load alone can be addressed by comparing the load-response curve in Figure 12 with one produced by reducing all external sources of phosphorus to Western Lake Erie, including the Detroit River. To conduct this diagnostic analysis, an additional suite of simulations was conducted to compare the response of cyanobacteria biomass in Western Lake Erie when all tributary TP loads are adjusted against the response when only the Maumee River TP loads are adjusted. The slope of this regression line is 9.19 MT of cyanobacteria biomass per MT of spring phosphorus (MT/MT), compared with 8.37 MT/MT for load reductions in the Maumee River alone. Model results indicate that although reducing the TP loads from all tributaries as opposed to only the Maumee River TP load will result in a lower cyanobacteria biomass for a given Mar-Jul Maumee River TP load, the similarity in slope suggests that the added benefit is not substantial.

Another question that is often raised is why not just focus exclusively on reducing the SRP load from the Maumee River rather than TP, given that the SRP load that has increased significantly over the past 20 years (Baker et al., 2014b) and given that it has a higher bioavailability than TP (DePinto et al., 1981; Young and DePinto, 1982; Young et al., 1985). To address this and other load reduction questions, WLEEM scenarios were run for various different load source reductions to produce several load response relationships for the 2008 baseline year (Figure 13). The same response (dependent) variable as detailed above was used for this load-

response curve; however, the independent (x-axis) variable is the total annual Western Lake Erie TP load in order to make valid comparisons among the various source reductions. Figure 13 illustrates that only small additional benefits with respect to the cyanobacteria biomass metric are achieved by reducing all tributaries rather than just the Maumee River. It also shows (based on the low slope of the line) that reducing only the Detroit River TP load achieves very little progress toward the desired cyanobacteria bloom reduction. In fact, a reduction of greater than 75% for the 2008 annual Detroit River TP load would be required to achieve the “mild” bloom threshold.

The plot in Figure 13 does show that, because it is 100% bioavailable, reducing just the annual SRP load (i.e., with no change in particulate phosphorus load) from the Maumee River provides a bigger “bang for the buck” than reducing TP load. However, a 60% reduction of SRP only load is necessary to reach the “mild” bloom threshold in comparison to a 25% reduction required for the Maumee River annual TP load. These results demonstrate why the recommendation of the Annex 4 Objectives and Targets Task Team is to reduce TP load from the Maumee River, with equal focus on SRP and particulate P loading.

Conclusion

Management decision-support tools, including the model presented here, have been developed for Lake Erie to determine nutrient load limits to reduce the occurrence of HABs. The models on which these tools are based have become increasingly sophisticated over time, incorporating advances in understanding of biogeochemical processes and advances in computing power and model code formulation (Gilbert et al., 2010).

WLEEM, the model presented in this paper, is an example of a comprehensive, fine-scale process-based model developed and applied to the waters of Western Lake Erie. It has been applied here to develop an external phosphorus load – cyanobacteria biomass response

relationship to support the establishment of a maximum load of TP from the Maumee River during the period of March – July to achieve what is considered a mild cyanobacteria bloom in Western Lake Erie. WLEEM identified the maximum TP load for that period to be 890 metric tons. WLEEM also showed that reductions in TP from all other tributaries can also reduce the severity of a bloom, but without concurrent reductions in the Maumee TP load large blooms will still occur within Maumee Bay. The model further demonstrates that while reductions in SRP are the fastest way to reduce bloom severity, a sizeable bloom could still occur even if 100% of the SRP was removed from the Maumee River. Figure 13 demonstrates what the size of the bloom would be if 100% of the Maumee River SRP load was removed, which is represented by the line with diamond points on the far right. The bloom still occurs because other forms of P are still ultimately bioavailable and over time within the system can be converted to available forms to contribute to a bloom.

The WLEEM simulation outcomes described here contribute to one of the multiple model recommendations for setting loading targets under Annex 4 of the GLWQA. The final recommendations of the Annex 4 sub-committee are presented in the report to the International Joint Commission titled *Recommended Phosphorus Loading Targets for Lake Erie* (Annex 4 Objectives and Targets Task Team, 2015). One of the recommendations – consistent with concepts in the GLWQA – is to undertake an Adaptive Management process to permit ongoing monitoring of progress relative to the goals driving the load reduction recommendations and adjustment of targets as necessary. Any meaningful Adaptive Management process includes an iterative process of monitoring, operational modeling, and ongoing research. Conceptual models have been developed related to how operational ecosystem modeling, monitoring, and research can be employed as part of an adaptive management process (DePinto, 2013; Gilbert et al., 2010; Fitzpatrick et al., 2015). In 2014, NOAA released its five-year vision for an Ecological Forecasting Roadmap, which outlined a framework for development of a national NOAA-wide

capability to provide dependable operational ecological forecasts. Harmful algal bloom forecasting was identified as one of four priority areas. WLEEM is ideally suited to be part of that process.

One of the important things to track adaptively is progress in load reduction in response to conservation actions implemented in the Maumee River Watershed. Because of the high inter-annual variability and precipitation and runoff in river systems such as the Maumee, one of the best ways to track this indicator is by normalizing the trend of March – July load by dividing it by the total river flow for that period to obtain a Flow-Weighted Mean Concentration (FWMC) of TP at the mouth of the Maumee River. The FWMC for the Maumee River varies between 0.300 and 0.400 mg TP/L for the years simulated. To meet the target of 890 MT of TP from the Maumee, the FWMC would need to be decreased to approximately 0.250 mg TP/L. Continued investment in model development and refinement will be essential to achieve desired outcomes and to reach integrated, holistic, and sustainable solutions to the growing global problems of eutrophication in lakes, rivers, and reservoirs.

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Figure Captions

Figure 1. Map of model grid, bathymetry, tributary locations, and monitoring stations.

Figure 2. Illustration of linkage and data connections between models.

Figure 3. Water temperature (°C) model (line) and data (dots) comparison at University of Toledo stations MB18 (top), 8M (middle), and GR1 (bottom) for 2008 and 2011-2014 (Mar.-Oct.).

Figure 4. Total phosphorus (mg/L) model (line) and data (dots) at University of Toledo stations MB18 (top), 8M (middle), and GR1 (bottom) for 2008 and 2011-2014 (May to Oct.).

Figure 5. Soluble reactive phosphorus (mg/L) model (line) and data (dots) at University of Toledo stations MB18 (top), 8M (middle), and GR1 (bottom) for 2008 and 2011-2014 (May-Oct.).

Figure 6. Nitrate plus nitrite (mg/L) model (line) and data (dots) at University of Toledo stations MB18 (top), 8M (middle), and GR1 (bottom) for 2008 and 2011-2014 (May-Oct.).

Figure 7. Chlorophyll a (µg/L) model (line) and data (dots) at University of Toledo stations MB18 (top), 8M (middle), and GR1 (bottom) for 2008 and 2011-2014 (May-Oct.).

Figure 8. Cyanobacteria biomass (mg/m²) model (line) and data (dots) at University of Toledo stations MB18 (top), 8M (middle), and GR1 (bottom) for 2008 and 2011-2014 (May-Oct.).

Figure 9. Average annual cyanobacteria biovolume (mL/m²/yr) model and data at four stations (4P, 7M, 8M, GR1) for 2008 – 2014. Only model results are shown for simulated years.

Figure 10. Time series plot of cyanobacteria depth-averaged biomass (mg/m^2) for 2008 baseline (solid line) across the entire Western Basin model domain and sequential scaling of the Maumee River TP spring (March-July) loads (dashed lines).

Figure 11. Maximum 30-day average cyanobacteria biomass and Maumee River spring (March-July) TP load for 2008, 2011-14.

Figure 12. Load-response curve for cyanobacteria maximum 30-day average biomass for the whole Western Basin as a function of the spring (March – July) TP load from the Maumee River. Baseline simulation years are labeled, while reductions from baseline are shown with the same symbol and color as the baseline.

Figure 13. Load-response curves for cyanobacteria maximum 30-day average biomass for the whole Western Basin as a function of the 2008 annual TP load from all tributaries represented in the WLEEM domain. Each curve was developed by making the following adjustments to the 2008 baseline load for the specified source: 0%, 25%, 50%, 75%, 100%, and 125%.

List of Tables:

Table 1. Tributary locations of USGS gages and flow scaling factor to account for additional flow below gage and average annual flow (2008 and 2011-2014).

Table 2. Suspended solids concentrations used to derive loads for western basin tributaries.

Table 3. Table of flow weighted mean concentration (mg/L) for major tributaries in western Lake Erie.

Table 4. Key phytoplankton functional group calibration coefficients used for WLEEM application.

Table 5. Annual model performance statistics for the combined calibration/confirmation periods at UT monitoring stations for water temperature, nutrients, and biological parameters.

Table 1. Tributary locations of USGS gages and flow scaling factor to account for additional flow below gage and average annual flow (2008 and 2011-2014).

Tributary	USGS Gage	Location	Scale Factor	Drainage Area (mi²)	Average Flow (cfs)
Detroit River	04165710	Detroit, MI	1.00	222,400	179,700
Maumee River	04193500	Waterville, OH	1.04	6,583	7,168
River Raisin	04176500	Monroe, MI	1.02	1,063	950
Huron River	04174500	Ann Arbor, MI	1.23	897	756
Stony & Otter Cr.	04176500	Monroe, MI	0.43	448	400
Ottawa	04177000	Toledo, OH	1.57	236	223
Portage River	04195820	Elmore, OH	1.47	726	1032
Cedar Creek	04195820	Elmore, OH	0.49	242	344

Table 2. Suspended solids concentrations used to derive loads for western basin tributaries.

Flow Source Description	Flow-Based Regression ^a
Detroit River	$C_{TSS} = 10$
Maumee River	$C_{TSS} = \text{monitored conc.}$
Swan Creek	$C_{TSS} = 0.085*Q + 30.52$
Ottawa River	$C_{TSS} = 0.13*Q + 24.81$
River Raisin	$C_{TSS} = 0.0415*Q + 10.60$
Portage River + Cedar River	$C_{TSS} = 0.0406*Q + 20.42$
Bay View WWTP + Direct Drainage	$C_{TSS} = 10$
Lake Erie (open boundary)	$C_{TSS} = 10$

Table 3. Table of flow weighted mean concentration (mg/L) for major tributaries in western Lake Erie.

Tributary	TSS	TP	SRP	NO₂+NO₃	TKN
Detroit River	10	0.017	0.010	0.40	0.20
Maumee River	147	0.394	0.102	4.30	1.50
River Raisin	41	0.144	0.044	3.50	1.00
Huron River	13	0.067	0.016	0.66	0.92
Stony-Otter Cr.	51	0.114	0.023	1.90	1.50
Ottawa River	24	0.112	0.022	1.90	1.50
Portage River	24	0.234	0.159	1.88	1.50
Cedar Creek	24	0.234	0.159	1.88	1.50
Open Boundary (Lk Erie)	5	0.010	0.003	0.25	0.40

Table 4. Key phytoplankton functional group calibration coefficients used for WLEEM application.

Description	Symbol	Units	Cyanobacteria	Diatoms	Greens
Optimal growth temperature	TOPT	°C	25	15	22
Saturated phytoplankton growth rate at TOPT	KC	/day	2.0	1.6	1.8
Temperature correction effect on growth rate below TOPT	KBETA1	(°C) ²	0.050	0.005	0.008
Temperature correction effect on growth rate above TOPT	KBETA2	(°C) ²	0.002	0.008	0.012
Saturating algal light intensity	IS	ly/day	300	150	150
Half saturation constant for nitrogen	KMN	mg-N/L	0.010	0.020	0.020
Half saturation constant for phosphorus	KMP	mg-P/L	0.002	0.001	0.001
Carbon to chlorophyll ratio	CCHL	mg-C/mg-Chla	33	50	33
Carbon to phosphorus ratio (non P-limited)	CRBP1	mg-C/mg-P	40	40	40
Carbon to phosphorus ratio (P-limited)	CRBP2	mg-C/mg-P	90	90	90
Coefficient determining range of P limitation	CRBP13	L/mg-P	50	200	50
Carbon to nitrogen ratio (non-N-limited)	CRBN1	mg-C/mg-N	5.68	5.68	5.68
Carbon to nitrogen ratio (N-limited)	CRBN2	mg-C/mg-N	8.50	8.50	8.50
Coefficient determining range of N limitation	CRBN3	L/mg-N	10	10	10
Base algal settling rate	VSBAS	m/day	-0.50	0.25	0.10
Endogenous respiration rate at 30 deg. C	K1RB	/day	0.1	0.1	0.1
Temperature coefficient for K1RB	K1RT		1.09	1.07	1.08
Nutrient Coefficients	Symbol	Units	Phosphorus	Nitrogen	Carbon
Algal Recycle Fraction					
Refractory Particulate	FRPOCZ	%	0.03	0.15	0.25
Labile Particulate	FLPOCZ	%	0.07	0.25	0.5
Refractory Dissolved	FRDOCZ	%	0.2	0.1	0.125
Labile Dissolved	FLDOCZ	%	0.2	0.1	0.125
Dissolved Inorganic	FPO4TZ	%	0.5	0.4	--
Hydrolysis					
Refractory Particulate to Dissolved	K57C	1/day	0.01	0.05	0.01
Labile Particulate to Dissolved	K68C	1/day	0.25	0.12	0.1
Mineralization					
Refractory Dissolved Organic	K79C	1/day	0.05	0.05	0.01
Labile Dissolved Organic	K89C	1/day	0.1	0.1	0.1

Table 5. Annual model performance statistics for the combined calibration/confirmation periods at UT monitoring stations for water temperature, nutrients, and biological parameters.

Statistic	Water Temperature	TP	SRP	NO ₂ +NO ₃	Chlorophyll	Cyanobacteria biovolume
Number of samples	1986	321	314	300	301	195
PBIAS	3.1%	17.2%	14.3%	-6.7%	18.8%	-14.4%
Relative Percent Difference	-3.2%	-20.9%	-20.7%	10.2%	-19.4%	21.9%
Relative Percent Error	-3.1%	-18.3%	-16.7%	11.5%	-15.7%	31.5%
Root Mean Square Error	0.8 °C	13.1 µg-P/l	3.1 µg-P/l	0.09 mg-N/l	6.3 µg/l	6,576 mL/m ² /yr

Figure 1

Figure 2

Figure 3

Figure 4

Figure 5

Figure 6

Figure 7

Figure 8

Figure 9

Figure 10

Figure 11

Figure 12

Figure 13