RESEARCH ARTICLE

CrossMark

Water quality modeling of a prairie river-lake system

Nasim Hosseini[1] · Eric Akomeah[1] · John-Mark Davis[1,2] · Helen Baulch[1] · Karl-Erich Lindenschmidt[1]

Abstract

Eutrophication of an under-ice river-lake system in Canada has been modeled using the Water Quality Analysis Simulation Program (WASP7). The model was used to assess the potential effect on water quality of increasing inter-basin transfer of water from an upstream reservoir into the Qu'Appelle River system. Although water is currently transferred, the need for increased transfer is a possibility under future water management scenarios to meet water demands in the region. Output from the model indicated that flow augmentation could decrease total ammonia and orthophosphate concentrations especially at Buffalo Pound Lake throughout the year. This is because the water being transferred has lower concentrations of these nutrients than the Qu'Appelle River system, although there is complex interplay between the more dilute chemistry, and the potential to increase loads by increasing flows. A global sensitivity analysis indicated that the model output for the lake component was more sensitive to input parameters than was the model output of the river component. Sensitive parameters included dissolved organic nitrogen mineralization rate, phytoplankton nitrogen to carbon ratio, phosphorus-to-carbon ratio, maximum phytoplankton growth rate, and phytoplankton death rate. Parameter sensitivities on output variables for the lake component were similar for both summer (open water) and winter (ice-covered), whereas those for the river component were different. The complex interplay of water quality, ice behaviors, and hydrodynamics of the chained river-lake system was all coupled in WASP7. Mean absolute error varied from 0.03–0.08 NH_4-N/L for ammonium to 0.5 to 1.7 mg/L for oxygen, and 0.04–0.13 NO_3-N/L for nitrate.

Keywords Global sensitivity analysis · Cold region · River-lake systems · Under-ice processes · Water quality modeling

Introduction

It has been argued that the rate of cultural eutrophication of Canadian lakes is increasing as a result of agricultural and industrial activities and population growth (Schindler 2001). River and lake water quality modeling is widely used to predict changes in trophic state under different management strategies and occurrence of algal blooms (Liu et al. 2014; Brooks et al. 2016; Zhao et al. 2013). Several lake water quality models have been developed in the recent past to aid decision-making (Sagehashi et al. 2000; Gurkan et al. 2006; Komatsu

✉ Eric Akomeah
  eric.akomeah@usask.ca

[1] Global Institute for Water Security, University of Saskatchewan, 11 Innovation Boulevard, Saskatoon, Saskatchewan S7N 3H5, Canada

[2] Water Security Agency, 101-108 Research Drive, Saskatoon, Saskatchewan S7N 3R3, Canada

et al. 2007; Martins et al. 2008; Norton et al. 2009; Hu et al. 2010; Zhao et al. 2013; Liu et al. 2014). In the studies by Zhao et al. (2013) and Liu et al. (2014), the responses of Lakes Dianchi and Yilong to varied inflow magnitudes, timings, and load reduction strategies were assessed using complex lake eutrophication models. Komatsu et al. (2007) also used a lake water quality model to examine the response of aquatic ecosystem to long-term global warming. For these models, contributions from rivers connected to the lakes were modeled as boundary conditions. Others have also modeled the water quality of river and lake components of river-lake systems separately (Akomeah et al. 2015; Sadeghian 2017; Terry et al. 2018). While predominant effects along a lake-river system are from upstream to downstream, Akomeah et al. (2015) showed that lake backwater effects can impact water quality of riverine sections. Fewer studies (e.g., Lung and Larson 1995) have considered rivers and lakes together in a single model. Our study system, Qu'Appelle River (QR) in Canada, flows through a series of under-ice lakes. Ice cover has significant effects on the hydrology and water quality of watercourses and waterbodies (Prowse 2001). Several studies have shown that

aquatic ecosystems are biologically active under ice-covered conditions (Hampton et al. 2017; Hosseini et al. 2016, 2017a). Rivers and lakes in Saskatchewan are covered by ice for about one third to half of the year. Given these effects and the length of time of ice cover in many cold regions, it was imperative that the water quality model used for this study included components that reasonably represented under-ice water quality.

The objective of this study was thus to assess the feasibility of reasonably representing eutrophication dynamics in the upper QR and Buffalo Pound Lake by comparing the robustness of a river-only water quality model to a river-lake water quality model to predict the dynamics using the Water Quality Analysis Simulation Program (WASP7) (Wool et al. 2006). The second objective was to examine the response of the study system to flow augmentation from Lake Diefenbaker, an upstream lake. WASP has been used extensively for modeling river and lake pollution in the USA and Canada. In this study, the model was used to simulate eutrophication in the QR system. This was done using available empirical data in a calibration-validation process. The calibrated and validated model was then used to assess how a future water management scenario of increasing inter-basin transfer from Lake Diefenbaker to the QR is predicted to affect water quality in the Qu'Appelle watershed.

Sensitivity analysis is an important tool to evaluate the model output and identify dependencies within the structure and process descriptions of the model (Wagener and Kollat 2007; Razavi and Gupta 2015). We performed a global sensitivity analysis (GSA) to determine seasonal and spatial changes in sensitivity of the model response to various input parameters. GSA provides the means of identifying processes influencing a system (Razavi and Gupta 2015). Sensitivity of the model response enabled us to compare eutrophication processes between the river and lake components of the river-lake model to understand the impact of ice-cover on water quality of the system.

Study site

The Qu'Appelle River, located in Saskatchewan, Canada, flows from Lake Diefenbaker in southwestern Saskatchewan 430 km east to the Assiniboine River in Manitoba. The river flows through several lakes including Buffalo Pound Lake, the lake of interest in this study. Further downstream, the river flows through Pasqua, Echo, Mission, Katepwa, Crooked, and Round lakes. Our study area includes the upper QR and Buffalo Pound Lake (Fig. 1). The majority of the drainage
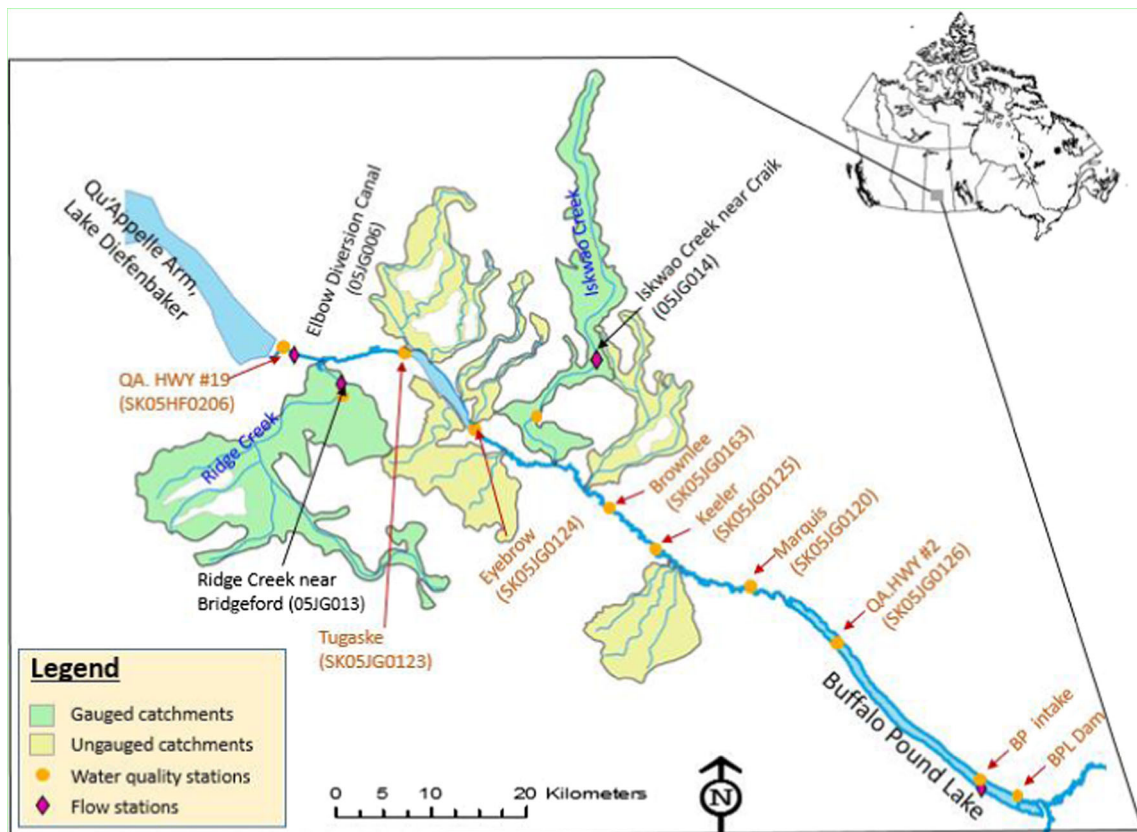

Fig. 1 Map of the upper Qu'Appelle River and Buffalo Pound Lake. The shaded gauged and ungauged areas represent the effective drainage area for those tributaries

area in the watershed is agricultural land (about 95%) (Hall et al. 1999), although much of the land is non-contributing with respect to runoff (Fig. 1). Buffalo Pound Lake has an average depth of about 3 m (Hall et al. 1999). Thermal stratification can occur for brief periods (hours to days) in this polymictic lake, (Hall et al. 1999, Baulch et al. unpubl. data). Flow in the upper QR is augmented from Lake Diefenbaker at the Qu'Appelle dam to maintain water levels in the QR system. The annual water level fluctuation in Buffalo Pound Lake is small (≤ 0.25 m). The important tributaries of the river upstream of Buffalo Pound Lake are Ridge Creek and Iskwao Creek, which flow intermittently. Buffalo Pound Lake provides drinking source water for the cities of Regina and Moose Jaw and surrounding communities and water for industrial uses such as potash mines. Buffalo Pound Lake is known as a naturally eutrophic prairie lake with frequent algal blooms and low Secchi depth readings (around 0.9 m; Hammer 1971). Buffalo Pound Lake is nutrient rich, and historically, the main source of external nutrient loads was from runoff in its watershed (Buffalo Pound Water Administration Board 2010). Warm temperatures, high pH, and low dissolved oxygen concentrations have been identified as major factors influencing lake sediment releases of phosphorus. High phosphorus release, in turn, can stimulate cyanobacteria blooms (Orihel et al. 2015; D'Silva 2017).

Methods

Model setup

Water quality model

The WASP7 program was developed by the United States Environmental Protection Agency (USEPA) (Wool et al. 2006; Ambrose et al. 1988). The program was designed to predict water quality responses to natural phenomena and anthropogenic pollution (Ambrose et al. 1988; Wool et al. 2006). For this initial work, a one-dimensional modeling approach was deemed sufficient to assess the feasibility of linking river and lake components.

In this study, the Eutrophication Module in WASP7 was used to simulate transport and transformation of dissolved oxygen (DO), total ammonia (NH_3-N + NH_4^+-N; hereafter NH_4-N), nitrate (NO_3-N), dissolved organic nitrogen (DON), orthophosphate (PO_4-P), dissolved organic phosphorus (DOP), and chlorophyll-*a* (Chl-*a*).

Geometric data and discretization

The upper QR (river component) was discretized into 165 longitudinal segments ranging in length from 600 to 800 m. The hydrodynamic data, including segment depth, width, velocity, and volume, were calculated from river morphology that was surveyed at approximately 770 locations along the 97-km stretch, from the Qu'Appelle Dam at Lake Diefenbaker to Buffalo Pound Lake.

For the Buffalo Pound Lake (lake component) segmentation, we used a digital elevation model (DEM) prepared in ArcGIS 10.2.2 (ESRI Inc., Redlands, CA, USA), which included sonar data of lake depth collected by boat in 2014, and a reservoir extent polygon and shoreline digital elevation data provided by the Saskatchewan Water Security Agency (WSA). Cross-sectional profiles of the lake were created based on the DEM using the River Bathymetry Toolkit (RBT) in ArcGIS and then converted to a HEC-RAS model for final bathymetry output. WASP7 allows up to 25 segments for ponds, which was deemed appropriate for Buffalo Pound Lake. Buffalo Pound Lake was discretized into 22 segments, each about 1400 m in length.

Hydrological and meteorological data

Daily flow rates at two locations, the Elbow Diversion Canal near the headwaters and Ridge Creek near Bridgeford (Fig. 1), were downloaded from Water Survey of Canada website. Estimated daily flows from Iskwao Creek and the other ungauged catchments were provided by WSA. Flow from ungauged catchments was distributed between the major drainage basins (shown in Fig. 1) based on their drainage area. On average, the main source of water to the upper QR and Buffalo Pound Lake is from Lake Diefenbaker. The high contribution of flow from Ridge Creek and the ungauged catchments primarily occurs during spring runoff. The contribution of flow from these creeks is also highly variable among years. One-dimensional kinematic wave routing in WASP7 was selected to simulate flows through the segments. This approach calculates flow wave propagation resulting in variations in water flow, volume, depth, and velocity throughout the stream network. The upper QR and Buffalo Pound Lake are typically covered by ice from some time in November until late April or early May. The period of ice cover was determined from the Water Survey of Canada symbols indicating ice cover. The first and last date of ice-cover symbols defined, respectively, ice-on and ice-off dates.

Daily meteorological data including air temperature and wind speed recorded at the Moose Jaw station were downloaded from the Environment Canada website. Daily solar radiation extracted from the National Centers for Environmental Prediction (NCEP) was provided by the NOAA/OAR/ESRL website at https://www.esrl.noaa.gov/. Model light extinction coefficients were based on the relationship between measured turbidity and measured extinction coefficients of PAR (photosynthetically available radiation) collected at multiple locations along the length of Buffalo Pound Lake by the WSA from 2015 to 2017. Based

on the study by Brown (1984), the relationship between turbidity and extinction coefficient for several reservoirs was found to be:

$$\text{Extinction coefficient} = C \times \text{Turbidity} \qquad (1)$$

where C was between 0.05 and 0.5.

The dashed-lines bounded the majority of the data collected at Buffalo Pound Lake falling within a C range of 0.15 to 0.5 (Fig. 2). In this study, 0.2235 was used as the constant "C" in the above equation (Fig. 2).

Tributaries

Water quality data were provided by WSA and Buffalo Pound Water Treatment Plant for six water quality stations along the upper QR and three stations along Buffalo Pound Lake for the period 2010 to 2016 (Fig. 1). The frequency of data for each water quality parameter varied from weekly to annually. The list of water quality stations and the number of measured data at each station for each water quality parameter are shown in Table 1. The furthest upstream station (HWY #19) and furthest downstream station (Buffalo Pound Lake dam) were used as boundary locations in the model. The period, 2012 to 2016, was selected for the model simulations to cover common forcing time series.

Nutrient concentrations in Ridge Creek and Iskwao Creek were provided by WSA for the time period of 2013 to 2016. Measured nutrient concentrations at Ridge Creek were used to estimate loads from the ungauged catchments.

Fluxes from the benthic layer can be an important source of nutrients to the water column. Initial model results showed that the system was sensitive to these fluxes. Ammonia and phosphate benthic fluxes were therefore calibrated to values of 24 mg/m^2/day and 1.5 mg/m^2/day, respectively. The values fall within measured benthic ammonia flux rate of 19–29 mg NH$_4$-N/m^2/

day at downstream lakes (unpublished data) and measured benthic phosphate efflux rates in incubation studies on the lake (0 to 40 mg/m^2/day) (D'Silva 2017).

Model calibration and validation

The model was calibrated and validated by adjusting kinetic parameters one-at-a-time until simulations matched observed data. The observed data sampled during the period 2014 to 2015 were chosen for the calibration exercise because more data were available during this time period. To validate the model performance, the calibrated parameter set was used to determine how well the simulations matched the observed data during the period of 2012 to 2013. Model performance was evaluated using mean absolute error (MAE), one to one regression plots of observed and modeled state variables, and probability plots.

Sensitivity analysis

A regional sensitivity analysis proposed by Hornberger (Hornberger and Robert 1981) was conducted to compare overall model response to stochastic perturbations of input parameters for two seasons, summer (May–October) and winter (November–April). OSTRICH (Optimization Software Tool for Research In Computational Heuristics; Matott 2008) was used to automatically run many WASP7 simulations. Thousands of parameter sets were selected randomly from uniform distributions within reasonable range of model parameter values. The parameter ranges were derived from previous studies (Hosseini et al. 2016, 2017a, b; Akomeah et al. 2015) of nearby catchments or adapted from the WASP7 model manual. The results were based on the normalized sum of squared error between simulated values from each run and the calibrated model outputs.

Scenario assessment of water management

The validated model was used to predict outcomes based on hypothetical changes to flow management, which involved increasing the magnitude of flow from Lake Diefenbaker. In summer (May–October), the flow released from the Qu'Appelle Dam was increased to 14 m^3/s, which is the present maximum capacity of the conveyance channel (Lindenschmidt and Sereda 2014). For winter months (November–April), the flow was increased to 6 m^3/s, which is the tested flow rate the river is able to accommodate without ice jamming or overbank flooding (Lindenschmidt and Davies 2014).
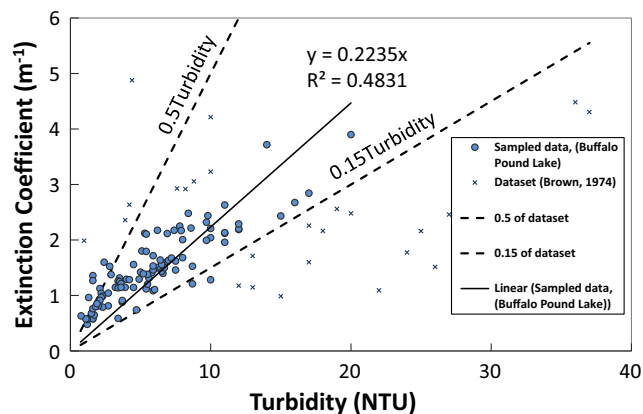

Fig. 2 Relationship between light extinction coefficient and turbidity for data collected on Buffalo Pound Lake between 2015 and 2017

Table 1 Water quality data availability used for this modeling exercise

Station names (location)	Number of observations from January 2010 to December 2016						
	DO	NH$_4$-N	NO$_3$-N	PO$_4$-P	Chl a	DON	OP
River sites							
QR. HWY #19	74	74	75	59	20	74*	58**
Tugaske Bridge	66	66	66	34	1	64*	32**
Eyebrown	7	9	9	5	–	9*	5**
Brownlee Bridge	55	5	5	5	2	3*	3**
Keeler Bridge	8	8	8	8	5	3*	3**
Marquis Bridge	83	84	84	43	7	83*	42**
Lake sites							
QR. HWY #2	19	20	21	21	19	21*	21**
BPL_intake	96	49	46	56	86	66	4
BPL Dam	3	4	4	4	–	4*	4**

*Number of calculated values from total Kjeldahl nitrogen and total ammonia

**Number of estimated values from total phosphorus and orthophosphate

DO dissolved oxygen, *NH4-N* ammonia + ammonium, *NO$_3$-N, PO$_4$-P* orthophosphate, *Chl a* chlorophyll-a, *DON* dissolved organic nitrogen, *OP* organic phosphorus. Data courtesy of the Saskatchewan Water Security Agency and Buffalo Pound Water Treatment Plant

Results and discussion

Calibration and validation

Model calibration and validation achieved reasonable qualitative and quantitative matches between modeled state variables and sampled records (Figs. 3, 4, and 5). The objective of calibration was to minimize the MAE. MAE provides a better metric for evaluating models than coefficient of determination (R^2), which just provides a pattern of the match between simulated and observed values rather than the values themselves (Wang and Ambrose 2015). A MAE of 1.5 is considered acceptable for water quality modeling (Akomeah and Lindenschmidt 2017). The MAE for DO, NH$_4$-N, and NO$_3$-N ranged from 0.03 to 1.7 with an average of 0.86 (Table 2). Based on qualitative and model performance metrics, the model simulation provided satisfactory match with monitored data especially for DO, NH$_4$-N, and NO$_3$-N for both the river and the lake components of the study system (Figs. 4 and 5). The model captured the average measured conditions (Fig. 3).

The model results generally followed the seasonal pattern of sampled data (Figs. 4 and 5). The seasonal trend in water quality differed between the river and lake components. In general, lower levels of DO and NO$_3$-N and higher levels of PO$_4$-P and NH$_4$-N concentrations were measured for the lake compared to the river. In the lake, DO and NO$_3$-N levels were lower in winter and higher in summer, while NH$_4$-N and PO$_4$-P increased in winter and declined in the summer months. Continued mineralization of deposited organic matter and dead plankton during winter and less nutrient uptake by

plankton community result in higher levels of NH$_4$-N and PO$_4$-P under-ice. In addition, model results suggest that lower nitrification rates occur as temperature decreases in winter. The decline in NH$_4$-N and PO$_4$-P concentrations during the ice-free period reflects uptake associated with active primary production by the plankton community during the summer. Increased DO during the summer is also reflective of primary productivity, as well as atmospheric exchange of DO once ice cover is lost.

Model predictions along the river stretch were better (Tugaske and Marquis Bridges) than in the lake (BP_intake) (Figs. 4 and 5) especially for ammonium and nitrate variables. The model also captured Chl-a concentrations well except for some algal bloom events in autumn. The shortcoming in Chl-a simulation might be due to the constant value assumed for carbon to chlorophyll ratio—the ratio is highly light sensitive and thus variable (Geider 1987; Sadeghian et al. 2018).

Sediment oxygen demand (SOD) was estimated to be higher in the lake than in the river. The SOD values were calibrated. SOD value of 1.87 g/m^2/day in the lake was in the same range reported by Terry et al. (2017) who also modeled SOD in Buffalo Pound Lake and by Veenstra and Stephen (1991) who showed that SOD values ranged between 1.49 and 4.08 g/m^2/day in oligotrophic to eutrophic lakes in the southwestern USA.

SOD has been reported to be velocity dependent because of changes in the diffusion coefficient and DO gradient under different flows (Nakamura and Heinz 1994). High flow velocity also decreases sedimentation rates (Nakamura and Heinz 1994). Oxygen consumption by sediments is known to be
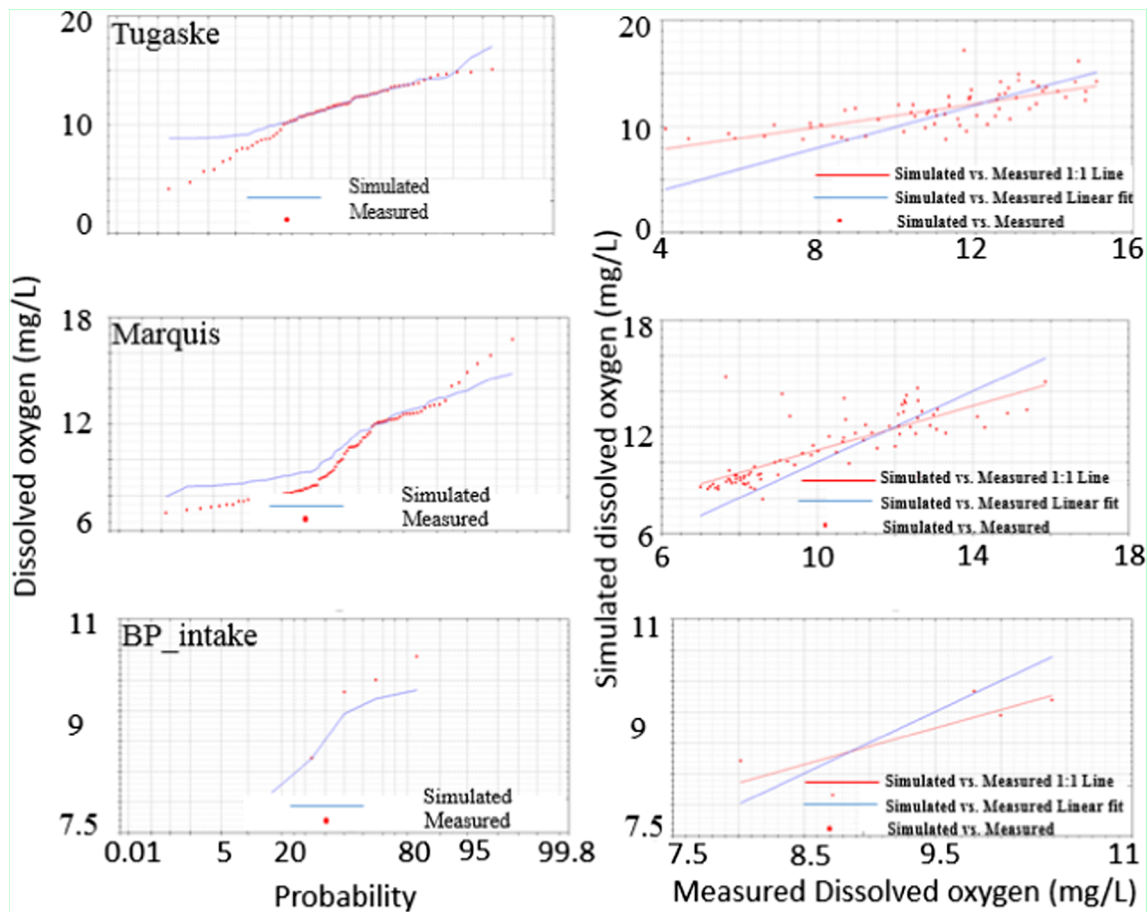
Fig. 3 Simulated dissolved oxygen (DO) calibration results at sampling stations: Tugaske, Marquis, and Buffalo Pond Intake. Left panel graphs represent variable exceedance probability plot between simulated and measured results. Right panel plots represent a one to one plot of simulated and measured data

greater in eutrophic lakes (Mathias and Barica 1980). Our results are consistent with findings of higher SOD rates in eutrophic standing water compared to the river section.

The reaeration rates estimated from the calibrated model in the river were significantly higher than the rates in the lake. Reaeration rate in lakes is a function of wind. The reaeration rate in rivers is controlled by depth and velocity. The expected reaeration rate based on the O'Connor-Dobbins formula at Marquis Bridge, as an example of a river segment, is shown by the red area in Fig. 6 published by Covar (1976). There was a large difference between model-estimated reaeration rates for the river and lake (Fig. 7). The high velocity associated with peak spring runoff and summer flows and the corresponding depth of flow may have contributed to elevated reaeration rates (Fig. 7) in the river (Ji 2008). Lake fluxes are lower, associated with the fact that wind-driven turbulence is the only major driver of gas exchange Fig. 8.

There were several potential sources of uncertainties in our model predictions, especially for the lake. There were fewer lake data with which to calibrate and validate the model, which led to aggregation of monthly data (e.g., fixed flux

rate). Hence, the model was unable to capture peak and low concentrations of sampled data. Greater frequency records improve the predictive power of water quality models. This is particularly the case when the sampling period captures transient behavior of the system. The second source of uncertainty in predictions for the lake component is that water quality data were measured at the intake of the water treatment plant. There is a short travel time from the lake to the plant through a pipe, meaning the water that may have been warmed could change some of the values from those in the lake. Third, the upstream part of the lake (upstream of the HWY #2 sample point) is shallow and has high macrophyte growth. This part of the lake is known to retain sediment and is presumed to also retain dissolved nutrients. However, we have no information on the amount of nutrients that may be retained, or how that might change seasonally. We assumed a direct transfer of substances from the river to the lake, but know that some of this is retained in the upper lake basin. This can lead to over prediction of state variables, as was the case for the winter 2014–2015 and summer 2015 when of PO_4-P concentrations were overestimated (Fig. 5).
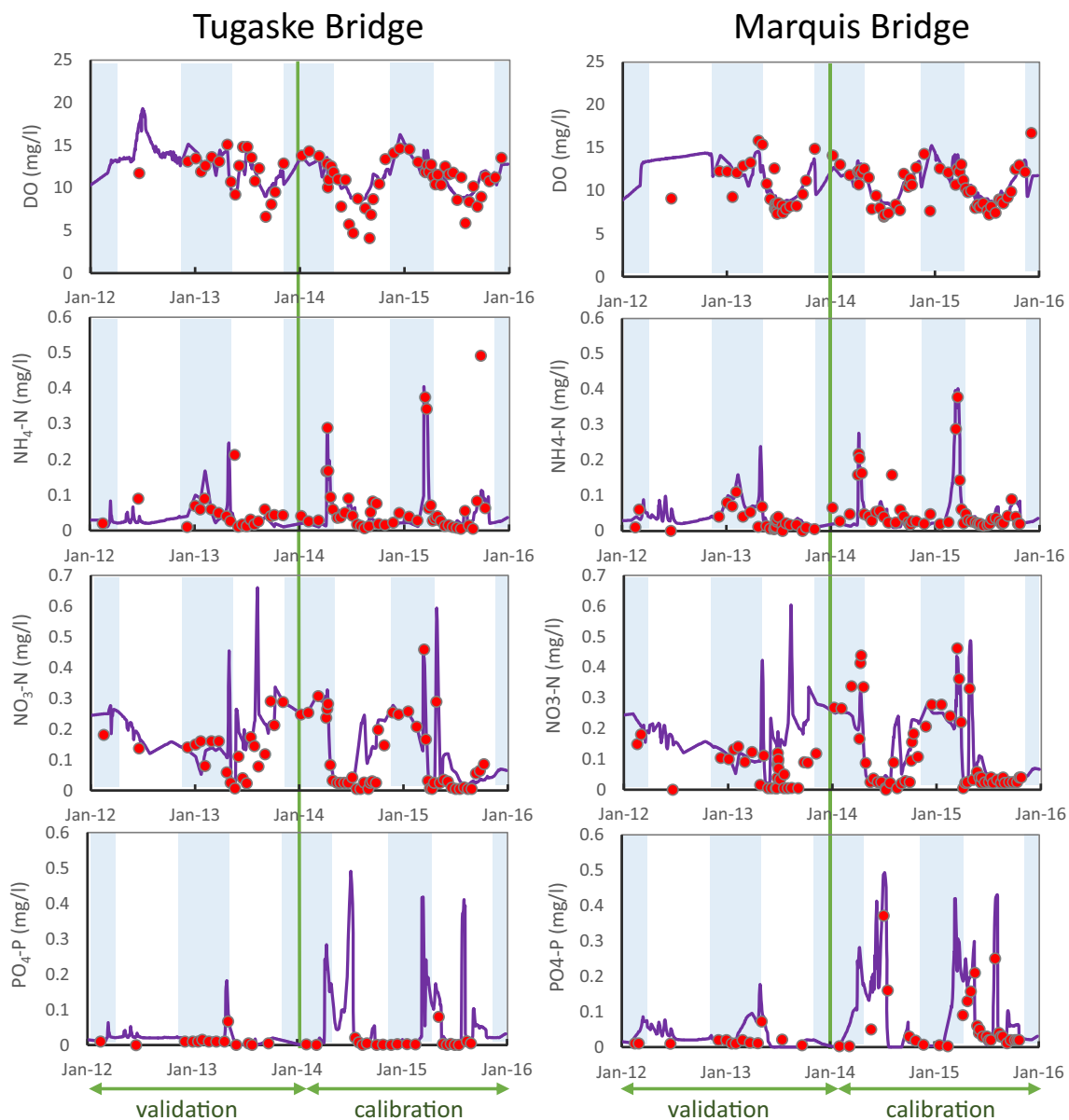
Fig. 4 Simulated versus observed concentrations of nutrients, and dissolved oxygen (DO) in the upper Qu'Appelle River. The blue bars represent ice cover periods

Snow tends to be the dominant driver of the light environment in winter. DO was underestimated at the BPL_intake in winters 2011–2012 and 2014–2015. Comparison of the DO concentrations in winter (ice-cover period) and snow thickness (Fig. 9) showed winter DO underestimation occurred in low snow year (2012, 2015). Global low DO levels were predicted for the winter of 2015–2016 (Fig. 9). Although high snow depth (Fig. 9) was recorded in the fall of 2015, snow depth in the winter was very low. Similarly, extremely lower Chl-*a* levels were predicted for the winters of 2012–2013 and 2014–2015. Pernica et al. (2017) found a strong inverse relationship between light penetration and snow depth. They stated a snow depth of 13.5 cm was the critical depth affecting

phytoplankton growth. Low snow winters or winters in which the ice cover remains snow free result in higher light penetration. However, since WASP7 includes ice data and not snow data, DO and Chl-*a* were underestimated—light thus becomes limited during winter due to restrictions by ice cover without low snow depth in WASP routines.

Comparison between river-only and river-lake model configuration

To gain insights into the influence of domain configuration of the study system on kinetic parameters, the calibrated WASP river-lake model in the current study was compared to a
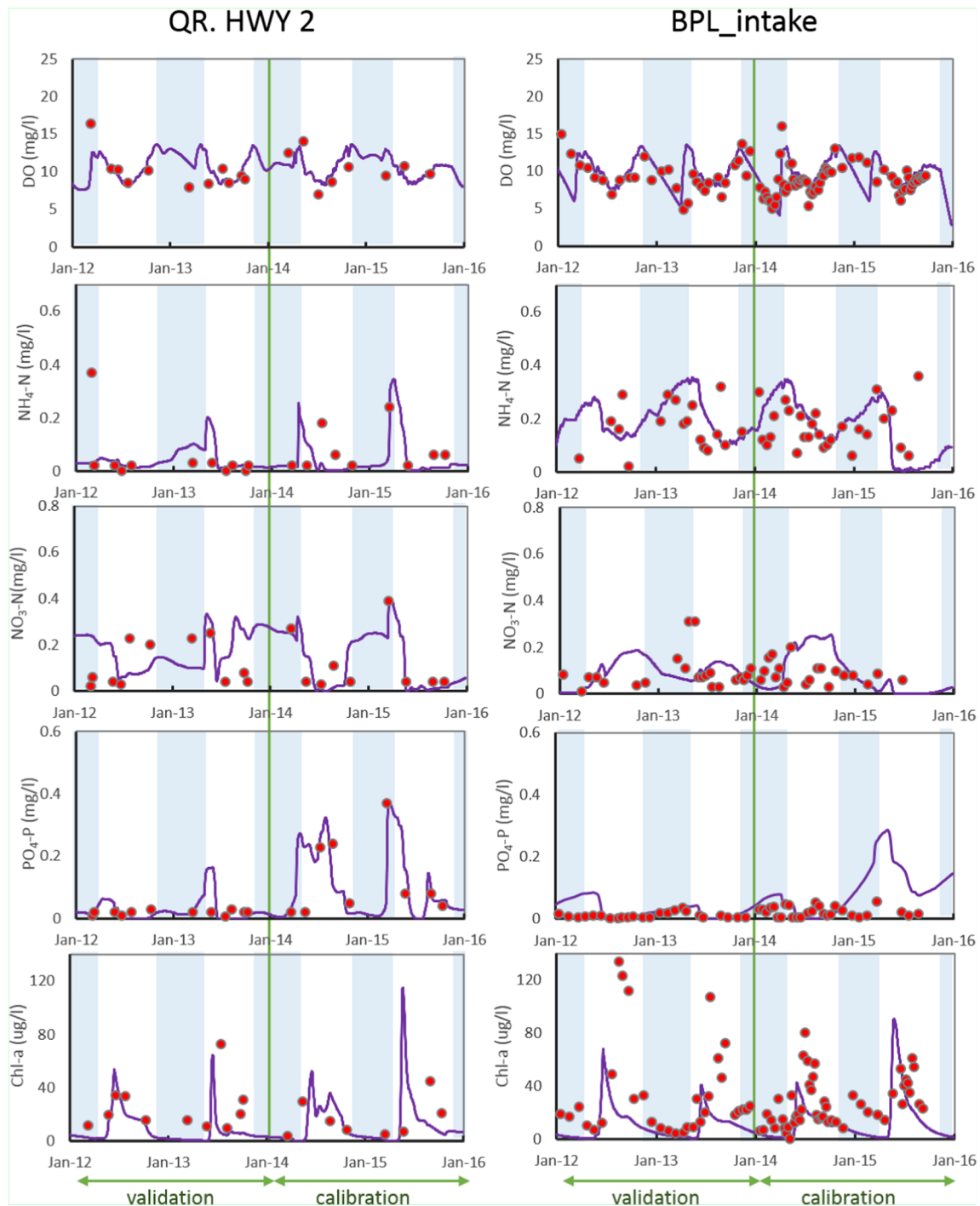
Fig. 5 Simulated versus observed concentrations of nutrients, dissolved oxygen (DO), and chlorophyll-*a* in Buffalo Pound Lake. The blue bars represent ice cover periods

WASP water quality model developed for the riverine section of the study system in a related study (Hosseini et al. 2017b) (Table 3).

The comparison was made to analyze the magnitude of spatial variation in calibrated parameters. The river-lake model was calibrated based on the observations along both the river and lake components, whereas the calibration of the river model was based solely on observations along the river portion.

The calibrated water quality parameters in the river-lake model were generally higher than calibrated parameters in the river-only model (Table 3). In the river-only model, most kinetic processes influencing coupled river-lake water quality, such as backwater flow, are lumped in

Table 2 Model performance evaluation metric for DO, NH$_4$-N, and NO$_3$

Gauge station	Dissolved oxygen (mg/L)		Ammonium (NH$_4$-N/L)		Nitrate (NO$_3$/L)	
	R^2	Mean absolute error	R^2	Mean absolute error	R^2	Mean absolute error
Tugaske Bridge	0.8	1.41	0.6	0.03	0.6	0.05
Marquis	0.6	1.2	0.4	0.03	0.43	0.07
HWY 2	0.1	1.7	0.4	0.05	0.05	0.13
BPL_intake	0.8	0.5	0.1	0.08	0.4	0.08

the model development. This may have contributed to the lower estimated parameters in the river water quality model (Table 3).

Nitrification rate in the river-lake model was higher than that in river-only model. Buffalo Pound has greater residence time than the river, which means nutrients spend more time in the lake and have greater opportunity to recycle within the lake and between sediment and the water column. This recycling process increases denitrification and nitrification rates. The accumulation of NH$_4$-N in the lake, especially, during the winter months may have contributed to the reason why the model predicts accelerated nitrification and denitrification processes during summer.
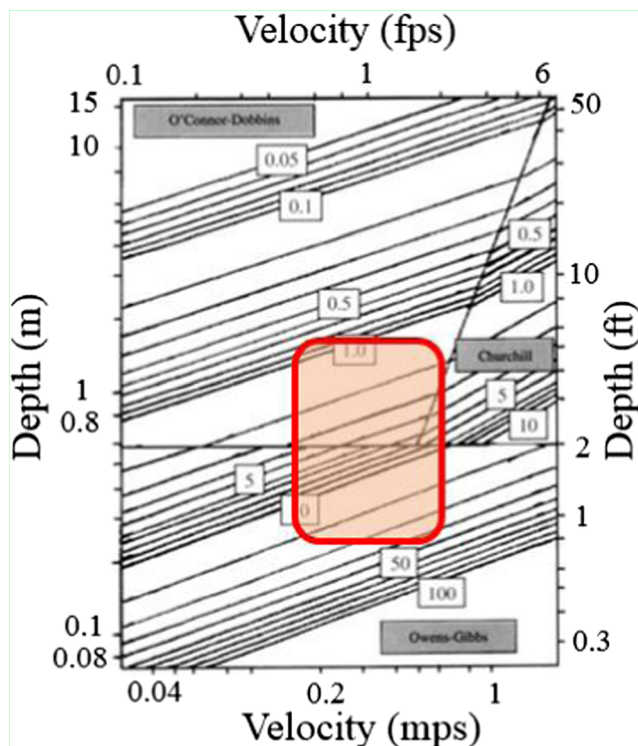

Fig. 6 Reaeration rate versus depth and velocity (Covar 1976). The red box represents estimated reaeration rate by O'Connor-Dobbins formula at Marquis Bridge

Parametric sensitivity of the river-lake model

Knowledge of model parametric sensitivity plays a key role in understanding which parameters and processes influence aquatic system behavior (Wagener and Kollat 2007; Razavi and Gupta 2015). In this study, global sensitivity analysis was undertaken to identify key parameters driving eutrophication in the river-lake model. This river section looks at global sensitivities of the parameters with only the river data, lake data, and the combined river-lake as objective function.

The model outputs from the lake were generally sensitive to more parameters than the model outputs from the river (Table 4). The results were based on the significance level (α) of difference between the 10% best parameter sets and the entire parameter sets. The 10% best parameter sets represent those with lowest sum of squared error. High significance was attributed to parameters with $\alpha < 0.001$, medium significance was attributed to parameters with $0.001 < \alpha < 0.01$, and no significance was attributed to $\alpha > 0.01$.

In general, parameter sensitivity results for the lake were similar for both winter and summer. Dissolved organic nitrogen mineralization rate, phytoplankton nitrogen to carbon ratio, phosphorus-to-carbon ratio, maximum phytoplankton growth rate, and phytoplankton death rate (Table 4) were found to be the most sensitive parameters for predictions. High sensitivity of the model response to phosphorus-to-carbon ratio and nitrogen-to-carbon ratio may be due to the higher amount of nutrients in the lake (Chapra 2008). Dissolved organic nitrogen mineralization
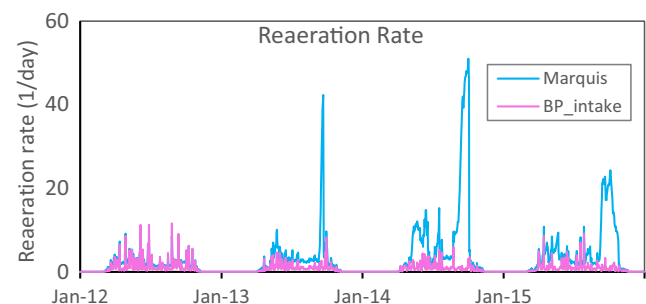

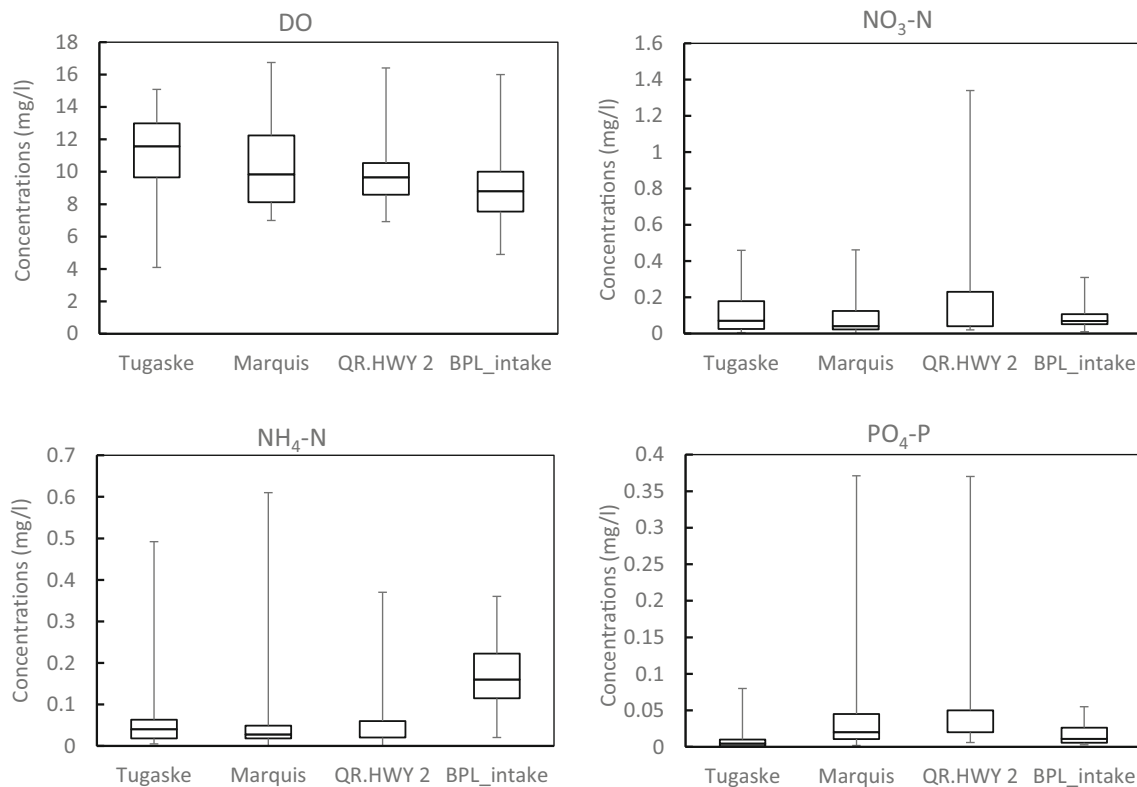**Fig. 7** Calculated reaeration rate at Marquis Bridge and BP_intake

Fig. 8 Concentration distribution of measured water quality parameters at four water quality stations along the upper Qu'Appelle River and Buffalo Pound Lake during 2012–2015

rate, maximum phytoplankton growth rate, and phytoplankton death rate were also reported to be the dominant control parameters for predictions of water quality parameters in the South Saskatchewan River (Hosseini et al. 2016, 2017a).

There were some slight differences in the parameter sensitivities of the river in summer and winter. The sensitivities were more pronounced in summer, particularly the parameters related to the death and mineralization of algae (K71C, K83C, K1R, and K1D; see definitions in Table 3). In winter, K71C (organic nitrogen mineralization rates) and SOD were found to have the most impact on organic nitrogen and dissolved oxygen. K12C, K1C, and KMNG still had a slight impact on the model predictions in winter.
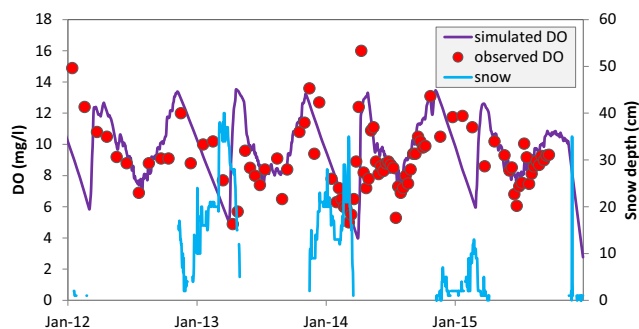

Fig. 9 DO concentration and snow thickness at BPL_intake

The high sensitivity of model results to K71C (organic nitrogen mineralization at 20 °C) may be explained by the limitation factors for phytoplankton growth. Figure 10 shows nutrient and light limitations for phytoplankton growth at Marquis Bridge and BP_intake. It is recognized that use of dissolved nutrients to infer limitation is problematic (Dodds 2003), that immediate nutrient availability is more closely related to physiological deficiency rather than limitation per se (Davies et al. 2010; Schindler 2012), and that co-deficiency of nitrogen and phosphorus (Davies et al. 2004) and nutrients and light (Healey 1985) occur. The model results presented in this study are meant to provide a general assessment of the relative availability of each and interpretation of results must be considered in this context; determination of limitation factors in the QR and Buffalo Pound would requires further work. Within the model, nutrient "limitation" is a function of dissolved inorganic phosphorus or nitrogen and varies between 0 (complete limitation) and 1 (no limitation). As shown in Fig. 10, the modeled growth is often predicted to be nitrogen limited at the beginning of summer and phosphorus limited at the end of the season. However, for some years, the phosphorus supplies were high resulting in nitrogen being present at a lower relative proportion and therefore considered the limiting factor by the model throughout the season.

The lake component of the model was insensitive to SOD but slightly sensitive to the SOD temperature coefficient

Table 3 Calibrated model input parameters

Parameter	Calibrated river-only model	Calibrated river-lake model	Units	Description
KMPG	0.0003	0.0048	mg P/L	Half-saturation constant for phytoplankton phosphorus uptake
KMNG	0.0499	0.26	mg N/L	Half-saturation constant for phytoplankton nitrogen uptake
K1D+K1R	0.19	0.032	1/day	Phytoplankton death rate
K1C	2.9792	3	1/day	Maximum phytoplankton growth rate
K12C	0.0068	0.01	1/day	Nitrification rate at 20 °C
NCRB	0.43	0.131	mg N/mg C	Nitrogen-to-carbon ratio
PCRB	0.0031	0.05	mg P/mg C	Phosphorus-to-carbon ratio
fON	0.58	0.95	–	Fraction of dead and respired phytoplankton recycled to *organic nitrogen*
K71C	0.0025	0.0001	1/day	*ON*-Mineralization rate at 20 °C
K83C	0.00043	0.011	1/day	*OP*-Mineralization rate at 20 °C
fOP	0.83	0.95	–	Fraction of dead and respired phytoplankton recycled to *organic phosphorus*
CCHL	59.94	20	mg C/mg Chl-a	Carbon-to-chlorophyll-a ratio
K2D	0.035	0.09	1/day	Denitrification rate at 20 °C
SOD_river	0.41	0.41	g/m^2 day	Sediment oxygen demand for river
SOD_lake	–	1.87	g/m^2 day	Sediment oxygen demand for lake
SOD$_T$	1.11054	1.13	–	Sediment oxygen demand temperature coefficient

(Table 4). SOD temperature became an important parameter to capture seasonality as the lake had a wide range of temperature variation (minimum = 0.3 °C to maximum = 24.9 °C). The higher sensitivity of SOD to the model outputs in the river

Table 4 Parameter sensitivity in winter and summer in the river-lake model

| Parameters	Description	Winter River Component	Winter Lake Component	Winter River-Lake	Summer River Component	Summer Lake Component	Summer River-Lake
SOD_r	Sediment Oxygen Demand for river		-		●	-	
SOD_l	Sediment Oxygen Demand for lake	-			-		
SOD_temp	Sediment Oxygen Demand Temperature coefficient		◉			◉	◉
K12C	Nitrification rate at 20°C				◉		
K20C	Denitrification rate						
K71C	ON-mineralization rate at 20°C	●	●	●	●	●	●
K83C	OP-mineralization rate at 20°C	●					
NCRB	Nitrogen-to-carbon ratio		◉	●		●	●
PCRB	Phosphorus-to-carbon ratio		●	●		●	●
CCHL	Carbon-to-chlorophyll-a ratio						
K1C	Maximum phytoplankton growth rate		●	●	◉	●	●
K1R	Phytoplankton death rate	●	●	●		●	●
K1D	Phytoplankton death rate	●					
KMNG	Half-saturation constant for phytoplankton nitrogen uptake				◉		
KMPG	Half-saturation constant for phytoplankton phosphorus uptake		●	●			
fON	Fraction of dead and respired phytoplankton recycled to Organic Nitrogen						
fOP	Fraction of dead and respired phytoplankton recycled to Organic Phosphorus						
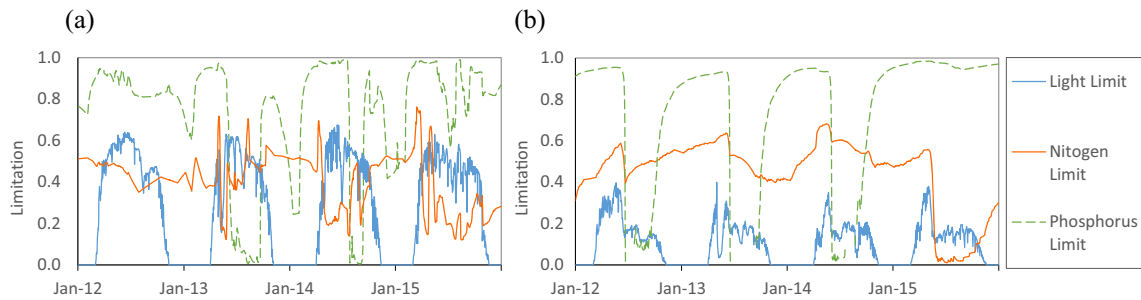
Fig. 10 Simulated nutrient and light limitation at **a** Marquis Bridge and **b** BPL_intake

compared to the model outputs in the lake component may be explained by the lower volume of water that exists in the river component compared to the lake component.

Effect of increasing flow on Buffalo Pound Lake water quality

The simulation of increased flow from Lake Diefenbaker was conducted over the years of the study, that is, the model was re-run for the same years but with greater outflows from Lake Diefenbaker. Understanding the effect of water transfer from Lake Diefenbaker on water quality in the QR watershed is important to better understand implications of water management

strategies for this system. It has potential implications to the water quality in Buffalo Pound, which serves as the source water for the province's capital city, Regina, the city of Moose Jaw and surrounding towns, and municipalities. Treatment efficiency at the water treatment has been affected by the lake's algal blooms (Kehoe et al. 2015) and high dissolved oxygen levels, and there are ongoing concerns related to dissolved organic carbon, which serve as precursors to disinfection byproducts. Lake Diefenbaker has lower nutrient, organic carbon, and salinity than Buffalo Pound. Thus, in theory, greater volumes of water transfer should improve downstream water quality, although the realization of this has not been obvious since water transfer started in the late 1950s and increased after construction of Lake Diefenbaker in
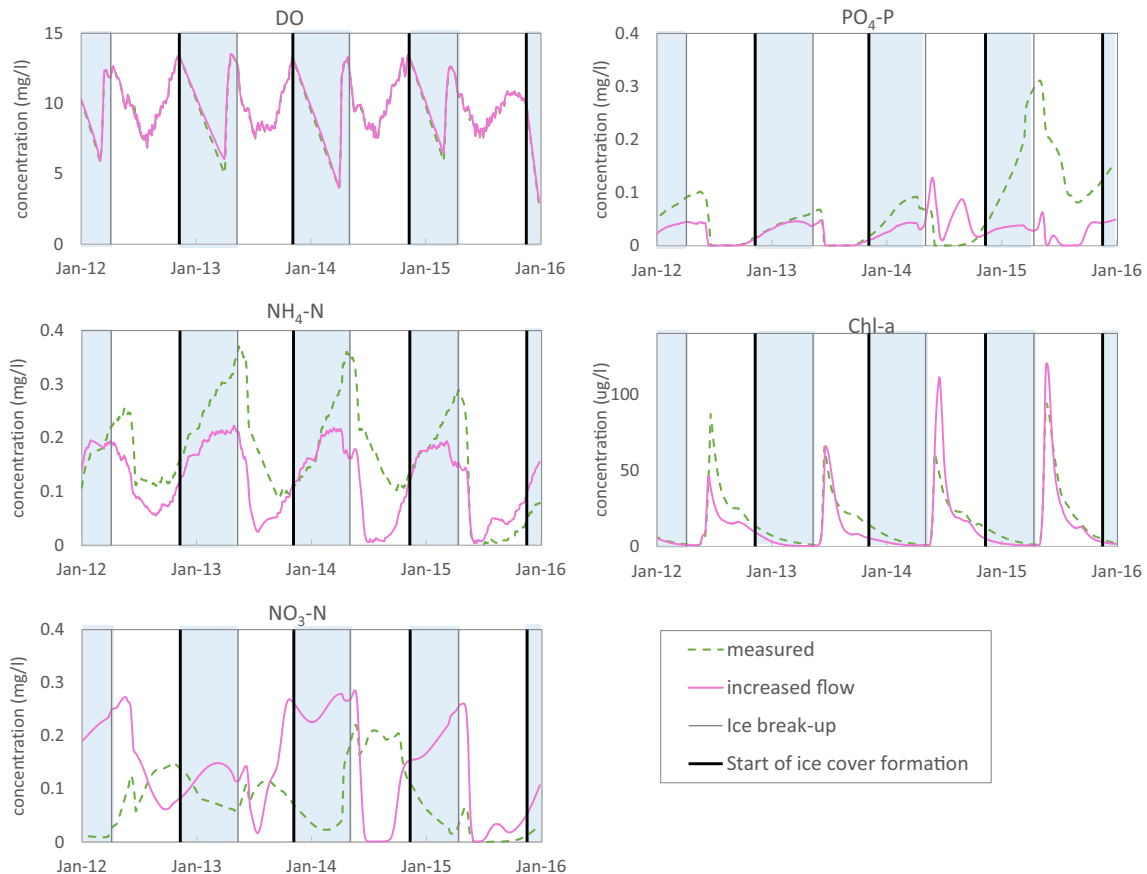

Fig. 11 Water quality parameter concentrations at BP_intake based on the period of measured flow (measured) versus the same period with simulated increased flow (increased flow)
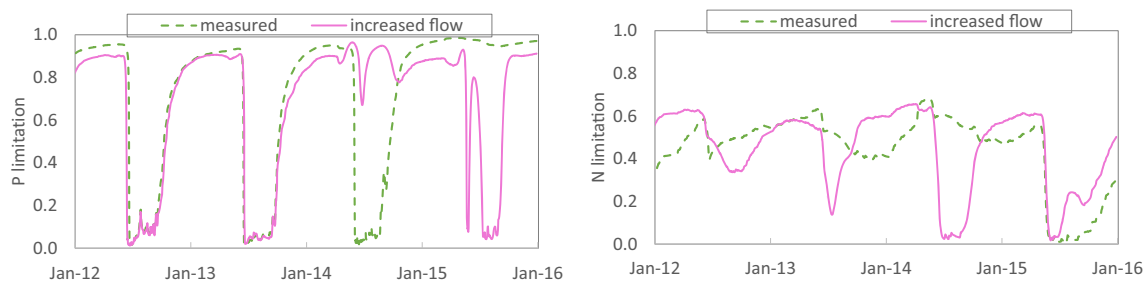
Fig. 12 P and N limitation based on the period of measured flow (measured) versus the same period with simulated increased flow (increased flow)

1967 (Davis and Sauer 1988). Water quality in Buffalo Pound is also affected by in-lake processes (D'Silva 2017) so that changes to water quality may be muted or substantially delayed. Furthermore, the QR watershed is relatively flat so water levels along the system must be carefully considered when water is transferred. Thus, in years with high runoff from the local watershed, there is less capacity to divert water from Lake Diefenbaker. When this occurs, especially if it occurs in successive years, the inflow to Buffalo Pound will have greater nutrients and organic carbon than in years where the proportion of flow from Lake Diefenbaker is greater (Lindenschmidt and Davies 2014).

This modeling test was conducted to evaluate the effect of increased diversion of water from Lake Diefenbaker, which is expected to occur in drier years when there are low volumes of runoff from the local watershed. Nutrient concentration results for two scenarios—with and without the flow increase—were significantly different: $\rho < 0.0001$, based on two-sample t test (Fig. 11). Since one of the main sources of NH_4-N and PO_4-P concentrations in Buffalo Pound Lake is from sediment flux, higher flow can lead to dilution of these constituents (Fig. 11). However, higher flows will also increase total loads to the lake, and could influence internal recycling processes. An increase in P limitation (Fig. 12) was simulated as a result of larger water transfers from Lake Diefenbaker. Lake Diefenbaker has comparatively low phosphorus concentrations compared to the QR system and Lake Diefenbaker is considered to be phosphorus limited, rather than shifting in limitation, as appears to be the case here (Hosseini et al. 2017b; North et al. 2015). Increased flows resulted in a decrease in simulated Chl-a both in summer and winter, except in summer 2014 when PO_4-P peaked at 0.09 mg/l. Despite the decline in PO_4-P and Chl-a concentrations, model simulations suggest the lake would still remain eutrophic.

both the river and the lake components. However, better simulation results were obtained for the river, which may reflect the transient importance of thermal stratification in this polymictic lake, and impacts on processes such as internal loading. We conclude that the model can be used to scale up to larger scale systems with multiple river-lake chains, for example, the entire Qu'Appelle River. Organic nitrogen-mineralization rate, maximum phytoplankton growth rate, and phytoplankton death rate were found to be the parameters which had the greatest effect on the model output. This suggests that hydrodynamics play a key role in the water quality of the study system. A 2D modeling would have been ideal, but data constraints limit such approach. The sensitivity results varied slightly between the river and lake components. The model outputs of the lake component were sensitive to more input parameters and no significant change was observed between summer and winter. In contrast, the model outputs for the river component were more sensitive to input parameters in summer. The scenario of the increased flow indicates that flow augmentation from Lake Diefenbaker could decrease NH_4-N and PO_4-P concentrations but Chl-a peaks did not decrease, which is reflective of the high nutrient status of Buffalo Pound. There is, however, a need to quantify the uncertainties associated with river flow releases to fully understand the impact of flow.

Conclusion

We assessed the feasibility of water quality modeling of a linked river-lake system using WASP7. The model was used to successfully simulate seasonal trends of water quality parameters (dissolved oxygen, dissolved nutrients, and chlorophyll-a) in

References

Akomeah E, Lindenschmidt KE (2017) Seasonal variation in sediment oxygen demand in a northern chained river-lake system. Water 9(4): 254

Akomeah E, Chun KP, Lindenschmidt KE (2015) Dynamic water quality modelling and uncertainty analysis of phytoplankton and nutrient

cycles for the upper South Saskatchewan River. Environ Sci Pollut Res 22(22):18239–18251

Ambrose RB, Wool TA, Connolly JP, Schanz RW (1988) WASP4, a hydrodynamic and water-quality model - model theory, user's manual, and programmer's guide. Environmental Protection Agency, Athens, GA (USA). Environmental Research Lab. PB-88-185095/XAB; EPA-600/3–87/039

Brooks BW, James ML, Meredith DAH, Mari-Vaghn V, Johnson Steve LM, Dawn AKP, Euan DR, Geoffrey IS, Stephanie AS, Jeffery A (2016) Are harmful algal blooms becoming the greatest inland water quality threat to public health and aquatic ecosystems? Environ Toxicol Chem 35(1):6–13. https://doi.org/10.1002/etc.3220

Brown R (1984) Relationships between suspended solids, turbidity, light attenuation, and algal productivity. Lake Reserv Manag 1:198–205

Buffalo Pound Water Administration Board (2010) Buffalo Pound Water Administration Board 2010 Annual Report

Chapra SC (2008) Surface water-quality modeling. Waveland Press, Long Grove

Covar AP (1976) Selecting the proper reaeration coefficient for use in water quality models. Washington, DC: In Proceedings of the Conference on Environmental Modeling and Simulation. Cincinnati, OH, EPA-600/9-76-016, Environmental Protection Agency

D'Silva LP (2017) Biological and physicochemical mechanisms affecting phosphorus and arsenic efflux from prairie reservoir sediment, Buffalo Pound Lake, SK, Canada (Doctoral dissertation)

Davies JM, Nowlin WH, Mazumder A (2004) Temporal changes in nitrogen and phosphorus codeficiency of plankton in lakes of coastal and interior British Columbia. Can J Fish Aquat Sci 61:1538–1551

Davies JM, Nowlin WH, Matthews B Mazumder A (2010) Temporal discontinuity of nutrient limitation in plankton communities. Aquat Sci 72:393–402

Davis E, Sauer EK (1988) Water deterioration during interbasin transfer in Saskatchewan. CWRJ 13(3):50–59

Dodds WK (2003) Misuse of inorganic N and soluble reactive P concentrations to indicate nutrient status of surface waters. J N Am Benthol Soc 22:171–181

Geider RJ (1987) Light and temperature dependence of the carbon to chlorophyll a ratio in microalgae and cyanobacteria: implications for physiology and growth of phytoplankton. New Phytol 106(1):1–34

Gurkan Z, Zhang J, Jørgensen SE (2006) Development of a structurally dynamic model for forecasting the effects of restoration of Lake Fure, Denmark. Ecol Model 197(1–2):89–102

Hall RI, Peter RL, Aruna SD, Roberto Q, John PS (1999) Limnological succession in reservoirs : a paleolimnological comparison of two methods of reservoir formation. Can J Fish Aquat Sci 1121:1109–1121

Hammer UT (1971) Limnological studies of the lakes and streams of the upper QR system, Saskatchewan, Canada I. Chemical and physical aspects of the lakes and drainage system. Hydrobiologia 37(3–4):473–507

Hampton SE, Aaron WEG, Stephen MP, Batt D, Stephanie GL, Noah RL, Emily H (2017) Ecology under lake ice. Ecol Lett 20:98–111. https://doi.org/10.1111/ele.12699

Healey FP (1985) Interacting effects of light and nutrient limitation on the growth rate of Synechococcus linearis (Cyanophyceae). J Phycol 21:134–146

Hornberger G, Robert S (1981) An approach to the preliminary analysis of environmental systems. J Environ Manag 12:7–18

Hosseini N, Chun KP, Lindenschmidt KE (2016) Quantifying spatial changes in the structure of water quality constituents in a large prairie river within two frameworks of a water quality model. Water 8(4)

Hosseini N, Chun KP, Wheater H, Lindenschmidt K-E (2017a) Parameter sensitivity of a surface water quality model of the lower South

Saskatchewan River—Comparison between ice-on and ice-off periods. Environ Model Assess 22:291–307

Hosseini N, Johnston J, Lindenschmidt KE (2017b) Impacts of climate change on the water quality of a regulated prairie river. Water 9(3):199

Hu L, Hu W, Zhai S, Wu H (2010) Effects on water quality following water transfer in Lake Taihu, China. Ecol Eng 36(4):471–481

Ji ZG (2008) Hydrodynamics and water quality: modeling rivers, lakes and estuaries, 1st edition, John Wiley, New Jersey, pp 676

Kehoe MJ, Chun KP, Baulch HM (2015) Who Smells? Forecasting taste and odor in a drinking water Reservoir. Environ Sci Technol 49(18):10984–10992

Komatsu E, Fukushima T, Harasawa H (2007) A modeling approach to forecast the effect of long-term climate change on lake water quality. Ecol Model 209(2–4):351–366

Lindenschmidt KE, Davies JM (2014) Winter flow testing of the upper Qu'Appelle River. LAP LAMBERT Academic Publishing

Lindenschmidt K-E, Sereda J (2014) The impact of macrophytes on winter flows along the Upper Qu'Appelle River. Can Water Resour Journal/Revue Can Ressour Hydr 39:342–355

Liu Y, Yilin W, Hu S, Feifei D, Rui Z, Lei Z, Huaicheng G, Xiang Z, Bin H (2014) Science of the total environment quantitative evaluation of lake eutrophication responses under alternative water diversion scenarios : a water quality modeling based statistical analysis approach. Sci Total Environ 468–469. Elsevier BV:219–227. https://doi.org/10.1016/j.scitotenv.2013.08.054

Lung WS, Larson CE (1995) Water quality modeling of upper Mississippi River and Lake Pepin. J Environ Eng 121(10):691–699

Martins G, Ribeiro DC, Pacheco D, Cruz JV, Cunha R, Gonçalves V, Brito AG (2008) Prospective scenarios for water quality and ecological status in Lake Sete Cidades (Portugal): the integration of mathematical modelling in decision processes. Appl Geochem 23(8):2171–2181

Mathias JA, Barica J (1980) Factors controlling oxygen depletion in ice-covered lakes. Can J Fish Aquat Sci 37(2):185–194

Matott LS (2008) OSTRICH : an optimization software tool ; documentation and user's guide. University at Buffalo, Buffalo

Nakamura Y, Heinz GS (1994) Effect of flow velocity on sediment oxygen demand: theory. J Environ Eng 120(5):996–1016

North RL, Johansson J, Vandergucht DM, Doig LE, Liber K, Lindenschmidt KE, Hudson JJ (2015) Evidence for internal phosphorus loading in a large prairie reservoir (Lake Diefenbaker, Saskatchewan). J Great Lakes Res 41:91–99

Norton DJ, Wickham JD, Wade TG, Kunert K, Thomas JV, Zeph P (2009) A method for comparative analysis of recovery potential in impaired waters restoration planning. Environ Manag 44(2):356–368

Orihel DM, David WS, Nathaniel CB, Mark DG, David WOC, Lindsey RW, Rolf DV (2015) The 'Nutrient Pump :' iron-poor sediments fuel low nitrogen-to- phosphorus ratios and cyanobacterial blooms in polymictic lakes. Limnol Oceanogr 60:856–871. https://doi.org/10.1002/lno.10076

Pernica P, North R L, Baulch HM (2017) In the cold light of day : the potential importance of under-ice convective mixed layers to primary producers. Inland Waters 2041 (May). Taylor & Francis, pp 1–13. https://doi.org/10.1080/20442041.2017.1296627

Prowse TD (2001) River-ice ecology. I: hydrological, geomorphic, and water-quality aspects. J Cold Reg Eng 15(1):1–16

Razavi S, Gupta HV (2015) What do we mean by sensitivity analysis? The need for comprehensive characterization of '"Global"' sensitivity in earth and environmental systems models. Water Resour Res 51(5):3070–3092. https://doi.org/10.1002/2014WR016527.Received

Sadeghian A (2017) Water quality modeling of Lake Diefenbaker. (Doctoral dissertation) University of Saskatchewan

Sadeghian A, Chapra S, Jeff H, Howard W, Lindenschmidt KE (2018) Improving in-lake water quality modeling using variable chlorophyll a/algal biomass ratios. Environ Model Softw 101:73–85

Sagehashi M, Sakoda A, Suzuki M (2000) A predictive model of long-term stability after biomanipulation of shallow lakes. Water Res 34(16):4014–4028

Schindler DW (2001) The cumulative effects of climate warming and other human stresses on Canadian freshwaters in the new millennium 1. Can J Fish Aquat Sci 58: 18–29

Schindler DW (2012) The dilemma of controlling cultural eutrophication of lakes. Proc R Soc B 279:4322–4333

Terry JA, Sadeghian A, Lindenschmidt KE (2017) Modelling dissolved oxygen / sediment oxygen demand under ice in a shallow eutrophic prairie reservoir. Water 9(2):131. https://doi.org/10.3390/w9020131

Terry JA, Sadeghian A, Baulch HM, Chapra SC, Lindenschmidt KE (2018) Challenges of modelling water quality in a shallow prairie lake with seasonal ice cover. Ecol Model 384:43–52

Veenstra JN, Nolen SL (1991) In-situ sediment oxygen demand in five southwestern US lakes. Water Res 25:351–354

Wagener T, Kollat J (2007) Numerical and visual evaluation of hydrological and environmental models using the Monte Carlo analysis toolbox. Environ Model Softw 22:1021–1033. https://doi.org/10.1016/j.envsoft.2006.06.017

Wang CBJ, Ambrose RB (2015) Development and application of mathematical models to support total maximum daily load for the Taihu Lake's influent rivers, China. Ecol Eng 83:258–267

Wool TA, Ambrose RB, James LM, Edward AC (2006) Water quality analysis simulation program (WASP) version 6.0 manual. DRAFT: user's manual

Zhao L, Yuzhao L, Rui Z, Bin H, Xiang Z, Yong L, Junsong W (2013) A three-dimensional water quality modeling approach for exploring the eutrophication responses to load reduction scenarios in Lake. Environ Pollut 177. Elsevier Ltd:13–21. https://doi.org/10.1016/j.envpol.2013.01.047