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Application of a eutrophication model for assessing water quality in Lake Winnipeg

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ABSTRACT

A eutrophication model using Water Analysis Simulation Program (WASP) has been applied to Lake Winnipeg during the period from 2002 to 2007. The model includes two nutrient cycles (N and P) and three functional phytoplankton groups (non-cyanobacteria, N-fixing cyanobacteria and non-N-fixing cyanobacteria). The model also considers distinct features of the morphological, hydrological, and climate conditions of the South and North Basins. The calibrated and validated results of water quality variables are in good agreement with the observed data of TN, NO_3, NH_4, TP, PO_4, DO, and total chlorophyll-a. The model reproduced qualitative features of phytoplankton communities in space and time, such as cyanobacteria in the North Basin during the late summer and non-cyanobacteria in the South Basin during the spring. The non-N-fixing cyanobacteria showed an increasing trend, even though it occupied smaller percentage than N-fixer within total cyanobacteria. Multiple nutrient-reduction scenarios were examined to assess the potential influence of different N:P loading ratios on the lake ecosystem. A 10% reduction of phosphorus decreased the cyanobacteria percentage in both basins, and reduced peak values of chlorophyll-a concentration during late summer in the North Basin. However, model results indicate that this will promote growth of non-N-fixing cyanobacteria. A reduction of nitrogen and phosphorus loading by 10% will restrict non-N-fixing cyanobacteria. The averaged phytoplankton biomass (expressed as chlorophyll-a concentration) and phytoplankton components suggest that increasing N:P loading ratio (P reduction > 12% and N reduction < 7%) would be effective for improving water quality in Lake Winnipeg.

Introduction

In the last three decades, the accrued human activities in the Lake Winnipeg watershed have significantly impacted the health of the lake ecosystem. Eutrophication is one of the major current water quality issues of Lake Winnipeg (LWSB, 2006). An increased amount of nitrogen and phosphorus has reached the lake from a variety of sources, such as municipal sewage, agriculture, livestock waste, and industrial discharges. Satellite imagery shows widespread cyanobacteria (bluegreen algae) blooms in the North Basin of Lake Winnipeg (especially in the summers of 2003 and 2006), and more localized algal blooms in the South Basin (McCullough, 2009). Microfossil evidence from the sediment cores taken from both the basins of Lake Winnipeg also showed that the species composition in the upper sediments has changed to taxa that are more typically found in high nutrient waters (e.g., *Aulacoseira*, *Stephanodiscus*, and nitrogen fixing bluegreen algae *Aphanizomenon* and *Anabaena*), which was interpreted as an effect of increased anthropogenic eutrophication (Kling, 1998). Meanwhile, a comparison of dominant species composition between 1969 and 1994 indicates that the lake phytoplankton diversity decreased (Kling, 2007).

Water quality models have been widely developed to improve understanding of lake biogeochemical characteristics and to provide an essential framework for water quality management. A function of these models aims to estimate the nutrient load reduction required to reduce algal blooms and water column nutrient concentrations. Since the 1970s, some large shallow lakes with cultural eutrophication concerns, such as Lake Erie and Lake Okeechobee, have been modeled extensively to assess lake-wide responses to external nutrient load reductions and to analyse the internal nutrient processing within the lakes (Di Toro et al., 1973; James et al., 1997). These models were enhanced over time to improve their performances, including the addition of interaction between water column and sediment layer (e.g., the impact of sediment resuspension to the water column, James et al., 1997), increasing the number of major algal classes (such as adding N-fixing cyanobacteria, James et al., 2005), expansion in the number of spatial segments (Di Toro and Connolly, 1980), the impact of invasion species (e.g., quagga and zebra mussels in Lake Erie, Boegman et al., 2008), and the direct coupling of three-dimensional hydrodynamic model (León et al., 2006). Recently, a simple mass balance model considered nutrient loadings entering and leaving the lake to estimate the total P concentration of Lake Winnipeg and their trends from 1919 to 2006 (Stainton and McCullough, 2007). Lake Winnipeg has unique morphometric features with shallow depth, large surface area, and high drainage to surface area ratio of about 40 (Brunskill et al., 1980).

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The hydrological and geological properties and nutrient loading between the South Basin and the North Basin are also different. Because of the complex morphometry and biogeochemical structure of Lake Winnipeg, detailed eutrophication modeling studies are needed to obtain a better understanding of the lake ecosystem and to assist in developing long-term nutrient management goals for the lake.

In this paper, a spatially segmented eutrophication model of Lake Winnipeg was developed by applying the Water Quality Analysis Simulation Program (WASP) V7.3 (Ambrose et al., 1993). The main objective is to simulate two essential elemental cycles (N and P) and the dynamics of three functional phytoplankton groups (non-cyanobacteria, N-fixing cyanobacteria and non-N-fixing cyanobacteria) using nutrient loadings to the South and North Basins for the period from 2002 to 2007. Actual morphology, observed hydrology and climate conditions during this period have been used. The calibrated model was used to examine the response of key state variables under different nutrients reduction scenarios.

Methods

Model description

The WASP model has been modified and improved for over 30 years since it was derived from the Potomac Eutrophication Model. In the recent version (WASP7.3), the approach of aggregating the algae into several dominant functional groups (e.g., diatoms, greens, cyanobacteria, etc.) was applied in the simulation of phytoplankton dynamics (Wool et al., 2008a). This model has been used for setting the nutrient target loads in several lakes and rivers. The model for multiple-algal groups could enhance the capacity of model simulation on seasonal variations, and reflect the realistic relationship between nutrients and phytoplankton dynamics. Because of these reasons this model has been applied to address important issues, such as cyanobacteria blooms and phytoplankton diversity under different nutrient-reduction scenarios. Although the observations of nutrients and phytoplankton are inadequate to develop a three-dimensional model to describe the current state of the lake, efforts are being made by the authors to develop such a model for the lake. However, those studies are not intended for long-term simulations as carried out in this study.

Plankton community

The eutrophication model of Lake Winnipeg focuses on the simulation of two elemental cycles (N and P) and three phytoplankton groups (non-cyanobacteria, N-fixing cyanobacteria and non-N-fixing cyanobacteria) (Fig. 1). The non-cyanobacteria group (P1) represents phytoplankton species in Lake Winnipeg including Diatoms (*Aulacoseira islandica*), green algae (chlorophytes), and cryptophytes (Kling, 1996). The N-fixing cyanobacteria group (P2) denotes heterocystous bluegreens (*Aphanizomenon* and *Anabaena*). The non-N-fixing cyanobacteria group (P3) represents the non-heterocystous cyanobacterial taxa, such as *Mirocystis*, a particularly toxic bloom-former. Kling (1996) also suggested that the non-cyanobacterial algae usually dominate the spring and cyanobacteria species are abundant in mid to late summer. In the year-to-year variability, the non-cyanobacteria group have the advantage during the wet and cool years, whereas the cyanobacteria group yield massive blooms in warm and dry years (Kling, 1998). The size and magnitude of cyanobacterial blooms have been increasing during the last two decades, and mainly, various forms of the heterocystous N-fixers (P2 group) dominated those blooms. Recently, there has been a gradual increase of colony forming *Microcystis* species (P3 group) in Lake Winnipeg. In 2006, water samples data showed that very high concentrations of microcystin ($>2000\ \mu g\ L^{-1}$) have been occasionally detected during intensive offshore and in-shore algal bloom events although microcystin levels are usually quite low (less than $1\ \mu g\ L^{-1}$) in open water locations (Kotak et al., 2009).

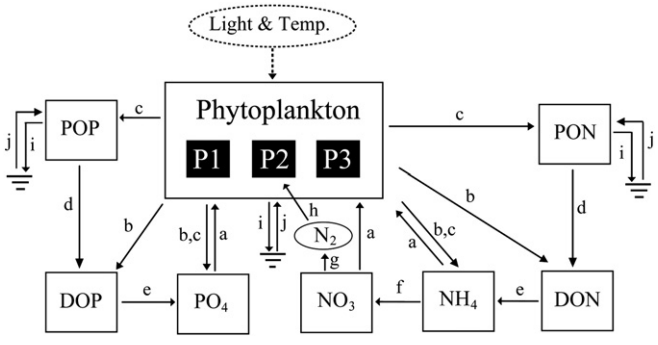

Fig. 1. Schematic of the phosphorus and nitrogen cycles of the Lake Winnipeg WASP model. The solid line arrows indicate flows of matter through the system. P1: Phytoplankton group 1 (non-cyanobacteria); P2: Phytoplankton group 2 (N-fixing cyanobacteria); P3: Phytoplankton group 3 (non-N-fixing cyanobacteria). (a) phytoplankton uptake; (b) phytoplankton respiration; (c) phytoplankton death; (d) detritus dissolution; (e) mineralization; (f) nitrification; (g) denitrification; (h) cyanobacteria N fixation; (i) settling; (j) resuspension.

The three phytoplankton functional groups are characteristic of their differing strategies for resource competition (nitrogen, phosphorus, light, and temperature) and metabolic rates as well as their morphological features (settling velocity and shading effects). Phytoplankton growth temperature dependence was chosen by the formulation which allows the predicted temperature adjustment factor to increase below an optimum level and to decrease above the optimum value (Cerco and Cole, 1994). Steele's equation was used to describe the relationship between photosynthesis and light intensity along with Beer's law to scale photosynthetically active radiation to depth (Smith, 1980). The computation of the total light extinction considered the impacts of algal shelf shading and non-algal light attenuation (such as solids and detritus). The non-cyanobacteria (P1) are modeled as r-selected organisms with relatively high maximum growth rates and higher respiration rates, strong phosphorus and weak nitrogen competitors, no capability of N fixation, low temperature optima, and high sinking velocities. By contrast, two cyanobacterial groups (P2 and P3) are modeled as K-strategists with lower maximum growth and lower metabolic rates, weak P and strong N competitors, low settling velocities, and high temperature optima. The model also reflected the differences in biological features between the two bluegreens. Aside from the ability to fix N_2, P2 group (*Aphanizomenon* and *Anabaena*) is configured to a higher maximum growth rate, lower phosphorus uptake affinities, higher abilities in nitrogen assimilation, and slightly lower preference temperature than P3 group (*Microcystis*) based on field and laboratory data (Dokulil and Teubner, 2000). In this model, silica requirements for non-cyanobacteria are not considered as the silica balance was not modeled. Because of lack knowledge of zooplankton seasonal variability, death rates are calibrated with somewhat higher values for non-cyanobacteria for favourable grazing from zooplankton. However, a lower death rate is adopted for the cyanobacteria groups to model less favourable predation from zooplankton (Lampert, 1987).

Nitrogen cycle

Four nitrogen variables are modeled: nitrate (NO_3), ammonium (NH_4), dissolved organic nitrogen (DON), and particulate organic nitrogen (PON) (Fig. 1). Both ammonium and nitrate are utilized by phytoplankton during growth. Algae generally have a preference for ammonium and the fraction of preferential uptake from ammonium is a function of its concentration and the concentration of the dissolved inorganic nitrogen ($DIN = NO_3 + NH_4$). Phytoplankton basal metabolism releases ammonium and organic nitrogen in the water column. A fraction of the particulate organic nitrogen hydrolyzes (dissolution) to dissolved organic nitrogen, and also PON sinks to sediment and