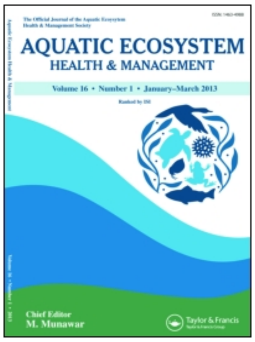

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Modelling phosphorus dynamics in Cootes Paradise marsh: Uncertainty assessment and implications for eutrophication management

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Cootes Paradise marsh, a hypereutrophic wetland draining into the western end of Hamilton Harbour, Ontario, has historically been considered an important regulatory factor of the severity of local eutrophication phenomena. In this study, we present a modelling exercise that aims to draw inference on the relative contribution of various external and internal flux rates to the phosphorus budget of Cootes Paradise. We first examined the capacity of a phosphorus mass-balance model, accounting for the interplay among water column, sediments and macrophytes, to reproduce the observed total phosphorus dynamics over a 17-year period (1996–2012). Water level fluctuations were one of the key challenges for balancing the phosphorus budget during model calibration. Our analysis shows that the model satisfactorily reproduced the average seasonal patterns, as well as the year-to-year total phosphorus variability (coefficient of determination = 0.20, relative error = 26.8%, root mean square error = 62.2 $\mu g\ P\ l^{-1}$, model efficiency = 0.15). However, our model failed to capture two years of the study period (1997 and 2007), when ambient TP levels significantly deviated from the typically prevailing conditions. Model sensitivity analysis identified the sedimentation of particulate material and diffusive reflux from sediments as two critical processes to characterize the phosphorus cycle in the wetland. Based on the current parameter specification, our model postulates that the sediments still act as a net sink, whereas macrophyte processes (respiration rates, nutrient uptake from interstitial water) appear to play a minor role. We conclude by discussing the various sources of uncertainty and additional remedial actions required in Cootes Paradise marsh to realize a shift from the current turbid-phytoplankton dominated state to its former clear-macrophyte dominated state.

Keywords: phosphorus modelling, nutrient recycling, sediment dynamics, Areas of Concern

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Introduction

In 1985, the International Joint Commission (IJC) recognized Hamilton Harbour, a eutrophic embayment at the western end of Lake Ontario, as one of 43 Areas of Concern (AOC) in the Laurentian Great Lakes (IJC, 1985). This designation was the outcome of a long history of eutrophication problems, including decades of waste inputs from industrial, agricultural, and municipal activities, wastewater treatment plants (WWTPs), and the City of Hamilton's combined sewer overflows (CSOs). Issues of concern for the Harbour included water quality deterioration, beach closures, toxic sediments, compromise of fish population integrity, and habitat losses (Hiriart-Baer et al., 2009). Having the mandate of restoring and protecting environmental quality and beneficial uses, the Hamilton Harbour Remedial Action Plan (HHRAP) was formulated by a wide variety of government, private sector, and community participants (Hall et al., 2006). The foundation of remedial measures originally proposed was based on the premise that the severity of eutrophication phenomena could be controlled by reducing ambient phosphorus concentrations. Following an analytical approach that incorporated data analysis, expert judgment and modelling, delisting targets for phosphorus loading, ambient total phosphorus (TP), chlorophyll α concentrations and Secchi disc depth were set at 142 kg day^{-1}, 17–20 μg l^{-1}, 5–10 μg l^{-1}, and 3.0 m, respectively (Charlton, 2001). To date, efforts to achieve these goals have involved substantial nutrient loading reduction from the sewage effluents discharging into Hamilton Harbour, which in turn led to a significant water quality improvement and aquatic macrophyte resurgence in most areas (Hiriart-Baer et al., 2009). However, the system still receives substantial loads of phosphorus, ammonia, and suspended solids from the Burlington and Hamilton sewage treatment plants, and therefore only moderate progress has been made since the mid-1990s (Gudimov et al., 2011).

Given the water quality gradient typically occurring in the western end of Hamilton Harbour, Cootes Paradise has been identified as a major loading source into the system (Hiriart-Baer et al., 2009). Cootes Paradise is a large marsh characterized by hyper-eutrophic conditions, stemming from the agricultural and urban development of the (previously forested) watershed along with the sewage effluent discharged into the marsh for over nine decades (Thomasen and Chow-Fraser, 2012). The vegetation cover in Cootes Paradise had receded to less than 15% by the 1990s, relative to >90% cover with very high plant diversity at the turn of the twentieth Century (Chow-Fraser, 2005). Coinciding with the vegetation decline, the fishery shifted from a desirable warm water fishery of Northern Pike and Largemouth Bass to one dominated by planktivorous and benthivorous species, such as Bullheads, invasive Common Carp, and Alewife. In particular, Common Carp, an exotic species introduced into Lake Ontario at the end of the nineteenth Century, accounted for up to 45% of the overall water turbidity (Lougheed et al., 2004). High turbidity had many detrimental effects across the entire food web, such as reducing light penetration to a level that was insufficient for submersed aquatic vegetation/periphyton growth, clogging filter-feeding structures of invertebrates, and affecting the behaviour and survival of visually hunting predators and mating fish (Thomasen and Chow-Fraser, 2012). To ameliorate the prevailing dire ecological conditions in the wetland, a number of restoration strategies have been implemented, such as Carp exclusion, nutrient loading reduction, and marsh planting. Nonetheless, while water quality has indeed improved, the biotic communities have not experienced a shift towards species representative of higher water quality conditions. There are many explanations for the apparent stability of the turbid-phytoplankton state, such as hysteresis (Scheffer et al., 2001), and the development of feedback mechanisms between abiotic and biotic factors (Suding et al., 2004). More importantly, high turbidity levels are expected to prevail in Cootes Paradise for the foreseeable future, and thus the re-establishment of a high richness community of submerged vegetation is unlikely (Thomasen and Chow-Fraser, 2012).

Recent empirical and modelling work has emphasized the importance of improving our understanding of the causal linkages between Cootes Paradise marsh and Hamilton Harbour (Gudimov et al., 2010, 2011). In particular, Gudimov et al. (2010) demonstrated that Cootes Paradise watershed ($\approx$290 km^2) represents the second most important exogenous loading source after the Woodward WWTP, accounting for 23% of TP variability in the Hamilton Harbour. In a follow-up analysis, Gudimov et al. (2011) hypothesized that the qualitative and quantitative features of the phytoplankton inoculum entering the western end of the Harbour, coupled with profound changes on the biogeochemistry and trophic functioning of the littoral zone induced by episodic pulses (summer storms), may potentially alter the contemporary growth and species competition patterns which in turn can be gradually propagated to the offshore sites (Schelske et al., 1995; Eadie et al., 2002; Schallenberg and Burns, 2004; Johengen et al., 2008). Notably, it was predicted that the short-term nutrient loading variability from Cootes Paradise and the by-pass of Woodward WWTP during the summer stratified period can potentially induce changes in the water quality, such as 5–15% increase of the chlorophyll a concentration and up to 30–50% increase of the relative cyanobacteria biomass, depending on the assumptions made about the intensity of the episodic event, the circulation patterns, and the concurrence of perturbations from the two major exogenous loading sources in the Hamilton Harbour. For these reasons, restoration of Cootes Paradise has been a focal point of the HHRAP efforts, and specific delisting targets are being recommended $<50\ \mu g$ TP l^{-1}, >1.5 m Secchi disk depth, and 20 μg chlorophyll $\alpha\ l^{-1}$ (Theÿsmeÿer, 2011).

In this study, our main objective is to examine the eutrophication patterns in Cootes Paradise and its potential to induce broader ecosystem changes to Hamilton Harbour. We present a modelling exercise that aims to draw inference on the relative contribution of various external and internal flux rates to the phosphorus budget of Cootes Paradise. We first examine the capacity of a phosphorus mass-balance model, accounting for the interplay among water column, sediments and macrophytes, to reproduce the observed total phosphorus (TP) dynamics over a 17-year period (1996–2012). The goal is to advance our understanding of the primary drivers of eutrophication in Cootes Paradise by elucidating the interplay between various phosphorus loading sources and the potential of macrophytes to become an integral part of the bottom-up forcing in the system. Our intent is to offer a much needed retrospective analysis of the impact of various restoration activities on the year-to-year variability of Cootes Paradise, and ultimately connect the lessons learned with on-going remedial measures in the Hamilton Harbour AOC.

Site description

Cootes Paradise marsh is approximately 4 km long, with a maximum width of 1 km, and a mean depth of 0.7 m. The surface area and volume can vary significantly according to water level fluctuations, reaching a maximum of 2.5 km^2 and $3.6 \times 10^6\ m^3$, respectively (Mayer et al., 2005). It is hydraulically connected to the west end of the Hamilton Harbour by a man-made channel, known as the Desjardins Canal. The marsh is an important migratory location for waterfowl, as well as a major fish nursery for Lake Ontario. At the beginning of the twentieth Century, 90% of the marsh was covered by a diverse community of emergent and submergent vegetation, which had declined down to 15% by the 1990s and was dominated by Cattails (*Typha* spp.) and exotic European Manna Grass (*Glyceria maxima*). Emergent vegetation loss has been attributed to sustained high water levels in Lake Ontario over the past 30 years and physical destruction by Carp (Lougheed et al., 2004). Submergent vegetation loss has been attributed to decreased water clarity associated with sediment resuspension from wind and Carp activity, as well as excessive inputs of nutrients from multiple sources, such as runoff from adjacent agricultural, residential, industrial, commercial and recreational lands, sewage effluent discharged from Dundas WWTP, and CSOs from the City of Hamilton (Thomasen and Chow-Fraser, 2012). Figure 1a shows the location of the marsh relative to the City of Hamilton along with the three main tributaries that drain the surrounding watershed: Spencer Creek, Chedoke Creek and Borer's Creek (Chow-Fraser et al., 1998). The restoration of aquatic vegetation in Cootes Paradise marsh was intended to reduce Carp biomass from an estimated average marsh-wide biomass of 80 to 5 tonnes km^{-2} (Wilcox and Whillans, 1999). The seasonal migration in the spring and fall enabled Carp exclusion via the construction of a barrier (or fishway) at the outlet of Cootes Paradise. The fishway became operational during the winter of 1997 and used 5-cm wide grating to physically exclude large fish, targeting Carp, from the marsh (Lougheed et al., 2004). This biomanipulation practice effectively prevented large Carp (>40 cm) from entering the marsh after February 1997.

In the Supporting Information section (available online), we provide detailed description of

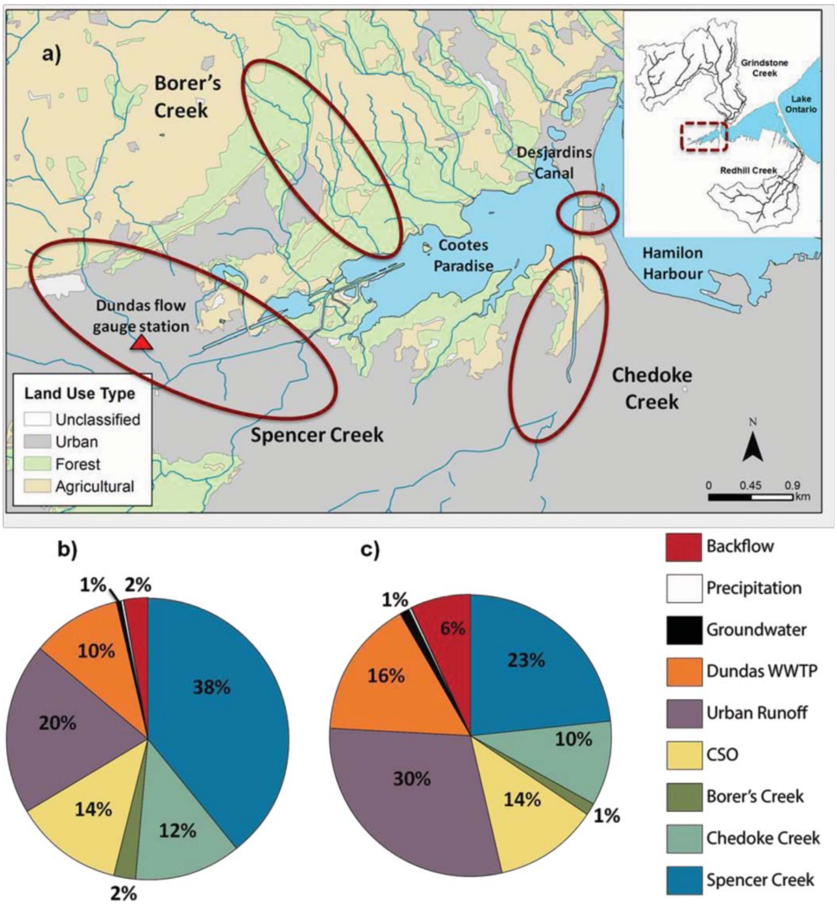

Figure 1. (*a*) Map and land use classification of Cootes Paradise and the surrounding watershed. (*b*) Annual and (*c*) growing season (May–October) daily *TP* loading to Cootes Paradise marsh during 1996–2012.

the methods used to estimate point/non-point source loading, calculate the water budget, assess the temporal trends of tributary flows, reproduce the processes pertaining to phosphorus cycle (including mathematical equations and parameter definitions; see Tables S1 and S2), and conduct sensitivity analysis.

Results

Annual daily loading to Cootes Paradise was estimated to be 38.62 kg day^{-1}. Tributary phosphorus loading demonstrated significant year-to-year variability, contributing an average of 52% of the total daily loading to the marsh (Figure 1b). Spencer Creek was the greatest phosphorus source representing an average of 73% of the tributary loadings and 38% of total loading, while Chedoke and Borer's creeks accounted for 12% and 2% of total daily loading, respectively. Urban runoff was the second most important non-point loading source accounting for 20% of total loading. Combined sewer overflows (CSOs) were the greatest point- and third greatest overall exogenous source representing 14% of daily loading, followed by the Dundas WWTP (10%). Backflow, precipitation, and groundwater collectively contributed less than 5% of the annual daily loading. Daily loading to Cootes Paradise averaged over the growing season (May-October) was estimated at 28.16 kg day^{-1}, representing 72.9% of total annual loading. The tributaries were again the greatest loading source contributing an average of 34% of daily loading of which, Spencer, Chedoke, and Borer's creeks accounted for 23%, 10%, and 1%, respectively (Figure 1c). Nonetheless, the relative TP loading contribution of the tributaries was distinctly lower during the growing season. Urban runoff was the second greatest contributor accounting for 30% of daily loadings, which may reflect the impact of summer storm events in conjunction with the ever-growing urban landscape in the area. The Dundas WWTP was the greatest point loading source contributing 16% of average daily loading, while CSOs accounted for 14%. Backflow from Hamilton Harbour contributed 6% of average daily loadings, while precipitation and groundwater combined accounted for less than 2%.

Considerable year-to-year variability characterized the flows in Spencer Creek (see Figure S1a in the online supplementary files). The peak flow rates were significantly lower in 1999, 2002 and 2004, but distinctly higher after 2010. Annual flow rate changes show how abruptly the magnitude of flow varied over time (Figure S1b in the online supplementary files). There was a moderate variation of the flow signal since 2000 with rates of change revolving around zero. Interestingly, there has been a shift to positive annual rates of change thereafter, with a global maximum found between 2008 and 2009 (0.6 m^3 s^{-1} $year^{-1}$). The strength of the association between flow and precipitation, as depicted by the time-series of β values, also varied dramatically over time with slightly decreasing pattern after 2000 (Figure S1c in the online supplementary files). The highest value was found in 1998 and the lowest one in 1999. The uncertainty of the β estimates has continued to increase over time, except in 1999, and was very high in 2012 and 2013, >60% (Figure S1d in the online supplementary files). This finding may imply that the continuous landscape changes and urbanization within the catchment may have obfuscated the relationship between tributary flows and precipitation.

Our mass-balance model closely reproduced the average monthly TP concentrations in Cootes Paradise (Figure 2a). The vector of calibration parameter values that provided the best fit is presented in Table S2 (in the online supplementary files). Calculating goodness-of-fit statistics with monthly averages, we found satisfactory RE (= 27.45%) and MEF (= 0.13) values, but fairly high RMSE (= 62.86 μg l^{-1}) largely reflecting the model's inability to capture ambient TP levels that deviated significantly from the typically prevailing conditions. Namely, the model was unable to capture the unusually low TP concentrations at Cootes Paradise marsh in 1997, resulting in an overestimation by approximately 50 μg TP l^{-1} on a monthly basis during the growing season. By contrast, the model significantly underestimated the substantially higher than average TP levels on a monthly basis observed in 2001 and 2007 (Figure 2a). When the model's predictive capacity was further examined at a coarser - seasonal – resolution, RE (= 18.13%) and RMSE (= 44.72 μg l^{-1}) values improved but MEF (= −0.007) became slightly negative. The model reproduced year-to-year variability of the seasonal TP average values, including the last five years of the study period (2008–2012) when TP was lower than the long-term seasonal average (Figure 2b). However, similar to the finer – monthly – resolution, the predicted seasonal TP differed significantly from the observed values in the marsh in 1997 and 2007. The model was able to capture month-to-month TP variability in the system, as derived after averaging model outputs and observed data across all years of our study period (1996 – 2012), including the TP peaks ($\approx$200 μg l^{-1}) typically experienced towards the end of summer/early fall (Figure 2c).

The five most influential parameters and/or external forcing functions on model outputs are identified for each of our three sensitivity analysis exercises in Table S3 (in the online supplementary files). The high r^2 values suggest a linear model response to input perturbations. The first analysis, involving the eight external forcing functions, demonstrated that the five most influential external drivers underlying water column, sediment TP dynamics, and total macrophyte biomass in Cootes Paradise are the loading from: (i) Spencer Creek; (ii) Dundas WWTP; (iii) CSOs from the City of Hamilton; and (iv) Chedoke Creek. Among these factors, Spencer Creek is by far the most influential external forcing function accounting for 38–53% of TP variability. The second exercise involving twenty model parameters showed differences in the ranking of the most influential model inputs for simulating TP concentrations in the water column and sediments, as well as macrophyte biomass. Diffusive reflux and sedimentation/particulate settling accounted for more than 80% of TP variability in the water column and $\approx$50% in the sediments (Table S3 in the online supplementary files). Sediment characteristics, such as porosity ($\approx$15%) and solid density ($\approx$12%), and burial rates linked to sediment deposition ($\approx$11%) were particularly influential on sediment TP dynamics. Notably, reflux and settling rates were not included within the five most influential parameters on macrophyte biomass. On the other

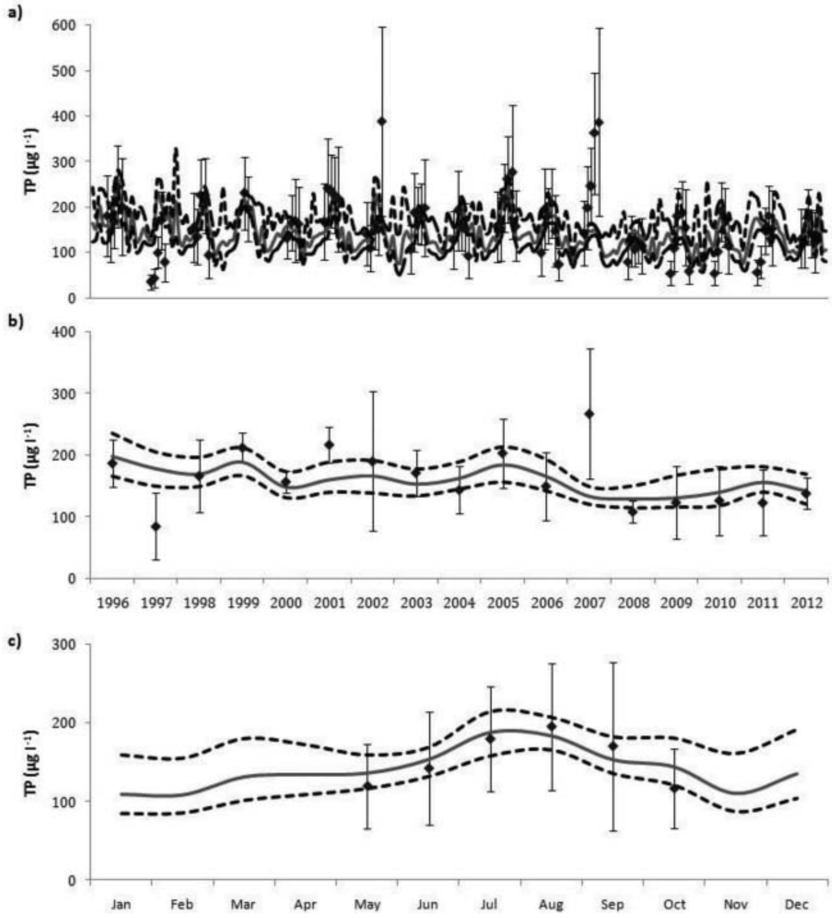

Figure 2. Comparison between observed and predicted *TP* concentrations on a (*a*) monthly and (*b*) seasonal basis (May–October) for each individual year. Panel (*c*) standardizes the within-year variability by averaging observed and predicted monthly *TP* concentrations across the entire study period (1996–2012).

hand, the most influential parameters related to our macrophyte biomass simulations mostly contained physiological features, such as growth and respiration rates, and resource (light and nutrient) limitations. The third exercise, focusing on the eight external forcing functions and five most influential parameters, demonstrated that reflux and settling rates, and inflows from Spencer Creek approximately accounted for 80% of TP variability in both water column and sediments. For total macrophyte biomass, maximum growth and respiration rates were predominantly influential, while the impact of the rest of the model parameters was practically negligible.

The phosphorus cycle simulated by the model is illustrated in Figure 3, where numbers correspond to the daily phosphorus flux rates for each process throughout the growing season (May–October) averaged across the entire study period (1996–2012). External TP loads contributed approximately 28.0 kg day^{-1} and were distinctly greater than internal loads (i.e. TP fluxes from sediment and macrophytes). In particular, internal loads stemming from sediment reflux and resuspension rates contributed to the water column approximately 16.2 kg day^{-1}, while those from macrophytes merely correspond to 0.28 kg day^{-1}. Our model suggests that approximately 12.9 kg day^{-1} are lost from the water column through sedimentation and 31.1 kg day^{-1} through outflows to Hamilton Harbour. The permanent loss of phosphorus from the system through burial to the deeper sediments accounts for 4.1 kg day^{-1}, and thus the net phosphorus flux related to the sediments (i.e. sedimentation + macrophyte mortality - sediment reflux/resuspension - macrophyte uptake - burial) is -7.34 kg day^{-1}. Based on

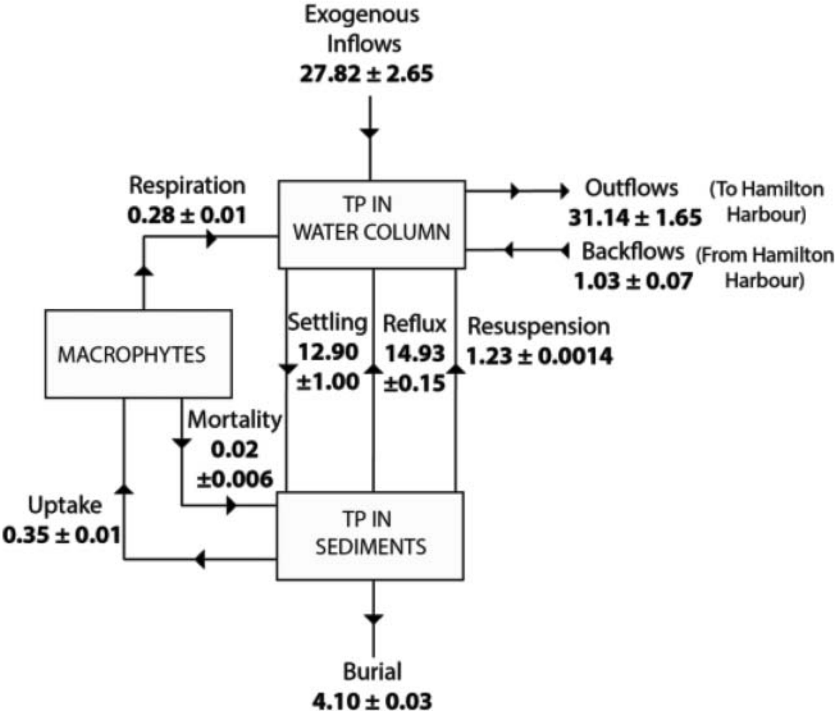

Figure 3. Average daily phosphorus flux (kg *TP* day^{-1}) corresponding to each simulated process during the growing season (May–October). Error estimates represent the 95% uncertainty zones derived from our Monte Carlo simulations.

their current abundance levels in the system, macrophytes appear to play a small role in the phosphorus cycle, removing a net of 0.33 kg day^{-1} from the sediments and contributing 0.28 kg day^{-1} into the water column.

Discussion

Management interventions aiming to induce shifts of degraded turbid, algal-dominated marshes to clear-water, macrophyte-dominated systems typically involve reduction of nutrient levels to decrease algal biomass, introduction of piscivores or removal of planktivores to control phytoplankton through trophic cascades, and elimination of benthivorous fish to increase water clarity (Whillans, 1996; Lougheed et al., 1998). Along the same line of reasoning, the overarching idea of remedial actions in Cootes Paradise marsh is that the system is capable of switching into a clearer state, although the prevailing conditions have not yet reached the critical levels that will allow such desirable shifts (Chow-Fraser, 2005). In 1997, for example, abnormally low spring temperatures caused a delay of fish migration into the marsh (including planktivores), which released the zooplankton population from predation, resulting in zooplankton-mediated improvement in water clarity and an expansion of submergent vegetation in previously unvegetated shallow locations (Lougheed et al., 2004). The mechanisms that led to this short-lived event offer a working hypothesis to delineate the optimal management actions in the area. In this context, modelling is a convenient methodological tool to examine critical conditions (e.g. nutrient loading levels, food web alterations) that could trigger abrupt (non-linear) changes and a rapid improvement of the ecological conditions (Arhonditsis and Brett, 2004). Before drawing such predictive statements though, our intent here was to conduct a retrospective analysis of the underlying drivers of the year-to-year variability in Cootes Paradise and obtain a better understanding of the system functioning.

What is the relative contribution of exogenous loading sources to the Cootes Paradise phosphorus budget?

Our analysis demonstrated that non-point sources from the tributaries contribute substantially more phosphorus to Cootes Paradise than point sources from CSOs, urban run-off and the Dundas WWTP. While Spencer Creek accounts for the greatest phosphorus export amongst the three tributaries, Chedoke Creek is responsible for approximately four times higher areal phosphorus loading (Theÿsmeÿer et al., 2009). Interestingly, our sensitivity analysis exercise indicated that the impact of TP loading from Chedoke Creek is twice as high relative to the TP loading emanating from urban runoff, although the latter is the second largest TP source. However, since the contribution of urban runoff was simply subtracted by the tributary nutrient loading (see Methods in the Supporting Information section on-line at www.taylorandfrancis.com) and thus the corresponding intra-annual variability was significantly muted, we caution that this particular result of our sensitivity analysis may not fully capture the actual impact of urban runoff on the dynamics of the system. Given also that Spencer Creek drains an agricultural watershed and Chedoke Creek watershed is predominantly urbanized, the reported trends deviate from the popular notion that nutrient exports from agricultural lands are greater, on an areal basis, than urban watersheds (Moore et al., 2004; Soldat and Petrovic, 2008; Soldat et al., 2009), but are on par with several recent studies in Southern Ontario (Winter and Duthie, 2000; Wellen et al., 2014a,b). Urban runoff was another major non-point loading source accounting for close to 50% of total tributary loads in the growing season. Furthermore, our analysis showed that on average, CSOs and the Dundas WWTP contribute similar amounts of phosphorus to Cootes Paradise and that both point sources exhibited a declining trend over the course of our study period. Specifically, a large improvement was observed after the development of the Main/King CSO tank in 1997, while average loading from the Dundas WWTP has remained below 4.0 kg day^{-1} after 2001; although there is still considerable year-to-year variability.

Nevertheless, in the context of eutrophication management, it is recognized that the sole consideration of the total annual TP loading may be misleading when applied to highly flushed systems, like the Cootes Paradise marsh (Minns et al., 2004; Kim et al., 2013). We thus calculated net loading from both point and non-point sources to weigh the different displacement of phosphorus due to the variability in the corresponding flow regimes induced. The net loading accommodates the idea that two equal total loads with opposite pairs of flow and concentration, high flow with low concentration or low flow with high concentration, could potentially have a very different effect on the trophic state of the receiving waterbody, and is simply calculated by multiplying the inflows from a particular source with the difference between inflow and outflow concentrations. We found that the "opposite pair" situation exists in regard to the contribution of point and non-point sources to the phosphorus budget at Cootes Paradise (Figure 4), as the net loading values clearly suggest that point sources always make a greater positive contribution, while the net tributary loads have mostly been negative. The latter loads are likely negative because the point source inputs and internal nutrient loading typically elevate TP concentrations in the system far above the tributary levels. Hence, tributary inputs actually allow carrying away some of the excess loading, thereby alleviating the impact of point source loads (Figure 4).

Interestingly, our models appears to reproduce accurately the relationship between the TP concentrations during the growing season and the corresponding flushing rates of the marsh (Figure 5; right panels), but less so the observed relationship between ambient TP and flow-weighted exogenous TP concentrations (Figure 5; left panels). The latter inability to reproduce one of the fundamental causal relationships underlying the eutrophication problem poses constraints in the capacity of the model to support predictions in the extrapolation domain (Ramin et al., 2011, 2012). Given the critical planning information associated with our loading assessment, we believe it is necessary to elaborate on the uncertainty associated with our estimates. As previously described, TP loading estimates for both Chedoke and Borer's creeks are based on the flow of tributaries with similar watershed characteristics, and as such, are associated with substantial uncertainty. The scarcity of concentration data for any of the tributaries outside of the growing season (November–April), contributes additional uncertainty to our tributary load estimates. For example, we found an average loading discrepancy of 88 kg P day^{-1} for the month of March between the year-specific rating curves and the rating curve developed after pooling all the data together from our 17-year study period. In order to reduce the uncertainty

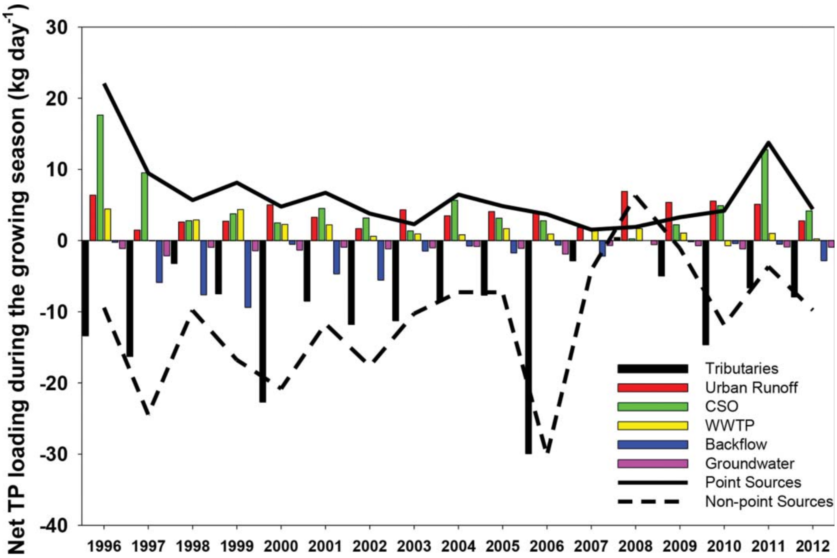

Figure 4. Temporal trends of exogenous *TP* loading to Cootes Paradise marsh during the study period (1996–2012): net loading (= $Flow_{exogenous} \times [TP_{exogenous} - TP_{Cootes}]$) from individual loading sources; and net point and non-point loading. Point sources include combined sewer overflows and wastewater treatment plants, and non-point sources contain tributaries, urban runoff, backflow, groundwater and precipitation.

associated with our loading estimates, we suggest collecting tributary nutrient data outside of the growing season to more accurately quantify the available phosphorus pool that can potentially fuel the biological activity after the physical limitations (solar radiation, temperature) are withdrawn in the spring. As the HHRAPTT (2010) points out, it may be an inefficient use of resources to monitor daily flow in both Chedoke and Borer's creeks; however, it would be beneficial to have bimonthly samples in order to consolidate the resemblance to the creeks used (Ancaster and Redhill creek) for establishing the corresponding estimates. Data that could provide better CSO volume estimates are urgently needed and nutrient concentration data collected more frequently would be beneficial to avoid loading estimates that are based on a single, average TP concentration.

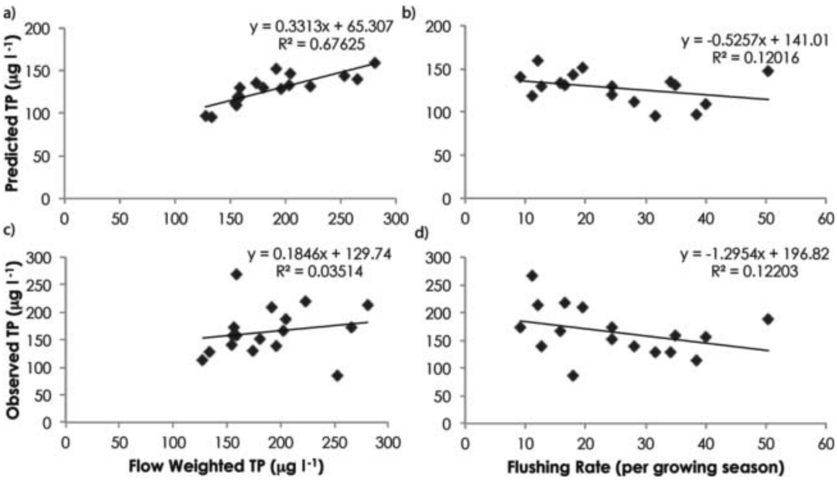

Figure 5. Predicted and observed relationships between *TP* concentrations against flow-weighted external *TP* and flushing rates during the growing season.

How critical is the role of internal nutrient loading?

Our internal loading estimates are substantially lower than those reported in previous modelling work by Prescott and Tsanis (1997), which suggested that 57% of phosphorus, or an average of 23.7 kg day^{-1}, in Cootes Paradise emanates from sediment reflux. However, this discrepancy can largely be explained by the sediment resuspension induced by Carp bioturbation, which were removed after the Prescott and Tsanis' (1997) study was conducted, reducing water turbidity and nutrients by up to 45% (Lougheed et al., 1998). Although studies in other shallow systems have shown similar reflux rates accounting for up to two thirds of TP loading (Søndergaard et al., 1990), field and lab experiments on the sediments of Cootes Paradise, following Carp exclusion, have not been indicative of such high internal loading rates (Kelton and Chow-Fraser, 2005; Mayer et al., 2005). For instance, after Carp exclusion, Kelton and Chow-Fraser (2005) demonstrated that internal loading due to resuspension, mineralization, and reflux contribute approximately 34% of the TP loading to Cootes Paradise, corroborating our internal loading estimate of 36% ([Reflux + Resuspension]/[Exogenous Inflows + Reflux + Resuspension] × 100, Figure 3). In the same context, Kelton and Chow-Fraser (2005) suggested that net mineralization and/or diffusion contributes an average of 8.8 kg P day^{-1}, while wind resuspension accounts for 4.2 kg P day^{-1}. Assuming Fickian diffusion, Mayer et al. (2005) estimated phosphorus release rates ranging from 0.27 to 5.25 mg m^{-2} day^{-1} at various sediment sites in Cootes Paradise marsh. Given that our estimate was somewhat higher (an average of 7.39 mg m^{-2} day^{-1}) during the growing season, we note the Mayer et al. (2005) recognized that their reflux rates may downplay the role of sediments, since the dependence of sediment diagenesis processes on temperature was not explicitly considered by their study. Both Kelton and Chow-Fraser (2005) and Mayer et al. (2005) also underscored the spatial variability of P reflux in the Marsh; a factor not currently considered that could potentially be incorporated in the next model iteration. It is also interesting to note that recent empirical evidence (Chris Parsons, University of Waterloo) suggests that the sediment reflux rates appear to be distinctly lower than those recorded in the Bay of Quinte towards the end of the fall. This finding invites further investigation of the actual mechanisms that drive seasonal and year-to-year variability in the Marsh.

According to our model projections (Figure 3), the TP contribution of internal sources (reflux, resuspension, macrophyte respiration) and sinks (sedimentation) appears to be significantly lower relative to the external sources (exogenous inflows) and sinks (outflows to Hamilton Harbour). However, our sensitivity analysis exercise showed that the intimate interplay between water column and sediment can profoundly modulate the TP dynamics in Cootes Paradise marsh (Table S3 in the online supplementary files). In particular, we found a positive relationship between the water column TP concentrations and the various characterizations of sediment diffusive reflux rate, effectively reaching a plateau after a threshold value of $\approx$50 kg day^{-1} is exceeded (Figure S2a in the online supplementary files). By contrast, a distinctly negative (and nearly monotonic) relationship exists between ambient TP and burial rates (Figure S2b in the online supplementary files). Given that the value assigned to our burial rate is largely based on radionuclide dating estimates of sediment deposition extrapolated from accumulation zones of the Bay Quinte (Minns, 1986), we caution that the parameterization of this process is highly uncertain and warrants further investigation. Specifically, according to our model, the rate at which sediments have been accumulating in the wetland was approximately equal to 1.2 mm year^{-1}, which is somewhat lower to the range suggested by Prescott and Tsanis (1997) in order for Cootes Paradise to keep pace with the water level rise (1.8–3.0 mm year^{-1}).

Release of phosphorus from actively growing submerged and emergent macrophytes is typically considered minimal, whereas decaying macrophytes may act as an internal phosphorus source adding considerable quantities of phosphorus into the water (Howard-Williams and Allanson, 1981; Granéli and Solander, 1988; Asaeda et al., 2000). In particular, emergent macrophytes usually possess large, perennial storage organs for carbohydrates and supporting tissues which are resistant to microbial attack, while submerged plants generally have only fine roots and do not contain much cellulose, and thus are more easily mineralized upon death (Twilley et al., 1986). Macrophytes also affect the chemical environment (oxygen, pH), which in turn may indirectly impact the phosphorus cycling (Granéli and Solander, 1988). Not surprisingly, our modelling analysis suggests that macrophytes play a minimal role in the phosphorus budget of Cootes Paradise, reflecting the fact that their abundance (e.g. biomass and density) is fairly low in its current state (Theÿsmeÿer, 2011; Thomasen and Chow-Fraser, 2012). Nonetheless, our sensitivity analysis also projects a potential increase of the macrophyte role when they will proliferate in the Marsh by the on-going restoration projects. Macrophyte growth and respiration rates are included in the five most influential parameters of the water column TP, although their contribution is still lower relative to the influence of water-sediment interactions and inflows from Spencer Creek.

Model performance–future model augmentations

Recognizing that models at both ends of the complexity spectrum have different strengths and weaknesses, we opted for a simple model structure as a first approximation to the phosphorus cycle in Cootes Paradise. Our model does not explicitly consider food web interactions among phytoplankton, zooplankton, and fish nor the competition between phytoplankton and macrophytes. By focusing solely on the hydro-chemical characteristics associated with the effects of water level fluctuations, exogenous nutrient loading, and internal recycling from the sediments, we were able to adequately reproduce TP dynamics in the Marsh. Nevertheless, there were two instances (1997 and 2007) in which extra complexity related to food web dynamics could have been useful. For example, while the ambient TP during the growing season in 1997 was approximately 85 μg l^{-1}, our model predicted a twice as high seasonal average of 168 μg l^{-1}. As previously mentioned, the spring of 1997 was characterized by unusually low spring temperatures that delayed the migration of spawning fish and released zooplankton from fish predation (Lougheed et al., 2004). The resultant increase of the grazing pressure on phytoplankton led to water clarity improvement that triggered extensive submerged macrophyte growth and proliferation in the Marsh. Hence, our model discrepancy reflects the amount of phosphorus typically sequestered in phytoplankton cells that was transferred to the upper trophic levels that year, following different pathways that ultimately changed its residence time in the water column. On the same note, although the clear-water phase was short-lived and Cootes Paradise switched back to a turbid state shortly after, this sequence of events provided evidence that the system is capable of an abrupt switch to a clearer alternative state. In particular, the critical importance of alleviating zooplanktivorous pressure is on par with Gudimov et al.'s (2011) assertion that any further remedial actions in the Hamilton Harbour system should be viewed in the context of a combined bottom-up (nutrient loading reduction) and top-down control.

A second discrepancy was found in 2007, when the model underestimated the high TP levels observed towards the end of summer-early fall. A plausible explanation for the latter mismatch could be a major pesticide spill that occurred near the end of July, releasing 900,000 litres of pesticide-contaminated douse water into Spencer Creek, which then moved downstream into Cootes Paradise (Theÿsmeÿer and Galbraith, 2007). The day after the spill occurred, diazinon, a synthetic organophosphate pesticide, was measured at 94 μg l^{-1} in Cootes Paradise - a level toxic to most small organisms. Theÿsmeÿer and Galbraith (2007) suggested that all small organisms, including aquatic invertebrates and small fish, were killed in the flow path of the contaminated douse water. It was estimated that 5−15 million fish died and likely sank to the bottom, where their tissue decomposition profoundly elevated nutrients into the system (Theÿsmeÿer and Galbraith, 2007). Moreover, the excessively high diazinon levels did not allow phytoplankton to capitalize upon that nutrient increase, leading to unusually clear water, devoid of the life typically experienced in Spencer Creek and most of Cootes Paradise. Thus, the two years with the greatest model misfit were induced by external factors, unaccounted for by the model, which in turn renders support to the general process characterization presented from our exercise (see P fluxes in Figure 3).

The apparent stability of the turbid-phytoplankton state in shallow systems has often been attributed to hysteresis – the inability of a system to be restored to its clear state once the external forcing factors that triggered a switch to a turbid state have been removed (Scheffer et al., 2001; Zhang et al., 2003). For instance, if phosphorus loading caused the forward switch to a degraded state, a switch back to a restored state will require achieving substantially lower phosphorus loading than what triggered the forward shift. It has been suggested that hysteresis is the reason Cootes Paradise has not responded desirably to restoration efforts, including Carp exclusion (Croft and Chow-Fraser, 2007). When we tested our model's response to loading reductions, TP in the water column responded in a linear fashion, with no indication of an abrupt (non-linear) shift to an alternative state. While this is likely the result of the simple model structure and its inability to capture important feedback mechanisms in the system, such as the competition forces between phytoplankton and macrophytes, we note that our characterization of the phosphorus cycle in Cootes Paradise marsh projects a response rate of 10−15 μg TP l^{-1} per kg TP day^{-1} or alternatively the external P loading should be approximately halved (<15–20 kg TP day^{-1}) in order to achieve TP concentrations lower than 100 μg TP l^{-1} during the growing season. The latter value was proposed as a minimum threshold for triggering a shift to a resilient alternative state (Thomasen and Chow-Fraser, 2012), provided that the on-going restoration efforts of plant community are successful (Theÿsmeÿer, 2011). Importantly, empirical evidence from Cootes Paradise suggests that competitive interactions between phytoplankton and macrophytes will likely shape these predictions (see Figure 7 in Lougheed et al., 2004). Except from the competition for nutrients, macrophytes may directly suppress algal growth through allelopathic effects (Wium-Andersen et al., 1982) or indirectly by providing refugia for large-bodied zooplankton grazers, such as Daphnia, promoting top-down control (Schriver et al., 1995). For example, Lougheed et al. (2004) showed that zooplankton at vegetated sites can withstand higher levels of fish predation and water is clearer relative to open water areas. Establishment of aquatic vegetation with sufficient density (>20 stems m^{-2}) may also maintain water clarity by stabilizing sediment and reducing sediment resuspension by wind in the Marsh (Lougheed et al., 2004).

Water level fluctuations are another critical factor that can profoundly modulate the interplay among physical, chemical, and biological components of the Cootes Paradise ecosystem. Lower water levels (and thus smaller water volumes) imply lower dilution and higher nutrient concentrations; a pattern consistent with our model predictions of higher ambient TP values towards the end of the summer-early fall, when the lower water levels in the Marsh occur. Further, with lower water levels, wind energy is more easily transmitted to the bottom sediments which in turn would accentuate the release of phosphorus due to stirring and mixing (Prescott and Tsanis, 1997; Chow-Fraser, 2005). The same mechanisms also appear to be the main drivers of the spatiotemporal variability of water turbidity, thereby influencing the illumination of the water column; especially, the light environment near the sediment surface in open-water sites which currently does not favour submerged macrophyte growth (Chow-Fraser, 2005). In the same context, two threshold water levels have been proposed for evaluating the resilience of submerged macrophytes; a maximum threshold, above which light availability becomes limiting, and a minimum threshold, below which conditions are excessively dry (Harwell and Havens, 2003). In fact, Chow-Fraser (2005) asserted that the disappearance of the submergent taxa in Cootes Paradise in 1999 was caused by the effects of desiccation as well as the increased reburial of propagules due to wind resuspension because of the low water levels. Given the importance of water level variability in the system, it is worth noting our difficulties in balancing the water budget with daily resolution data, predominantly due to the uncertainty of our outflow estimates that were clearly inadequate to capture the diurnal and semi-diurnal flow reversals between Cootes Paradise and Hamilton Harbour (Skafel, 2000).

Conclusions

In conclusion, we presented a modelling exercise that attempted to quantify all the major processes of the phosphorus cycle in Cootes Paradise marsh. Spencer Creek was the greatest phosphorus source representing an average of 73% of the tributary loadings and 38% of total loading, while urban runoff was the second most important non-point loading source accounting for 20% of total loading. Internal loads originating from sediment reflux and resuspension rates contribute to the water column approximately 16.2 kg day^{-1}, while the permanent loss of phosphorus from the system through burial to the deeper sediments accounts for 4.1 kg day^{-1}. Our phosphorus mass-balance model satisfactorily reproduced within- and among-year TP variability, whereas the two years with the greatest model error (1997 and 2007) were influenced by external events, unaccounted for by the model structure.

Based on the current parameter specification, our model postulates that the sediments still act as a net sink, whereas macrophyte processes (respiration rates, nutrient uptake from interstitial water) appear to play a minor role in the phosphorus budget. Water level fluctuations and competition forces between phytoplankton and macrophytes are two critical facets of the ecosystem functioning in order to draw credible predictions about the remedial actions required in Cootes Paradise marsh to realize a shift from the current turbid-phytoplankton dominated state to its former clear-macrophyte dominated state.

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Supplemental material

Supplemental data for this article can be accessed on the publisher's website.

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Modelling phosphorus dynamics in Cootes Paradise marsh: Uncertainty assessment and implications for eutrophication management

[Supporting Information]

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Methods

Point/Non-Point Source Loading

WWTP loading estimates were based on monthly flows and effluent *TP* concentrations from the Dundas *WWTP*, as reported by the City of Hamilton. *CSO* loading estimates into Cootes Paradise were based on two data sources: the Wastewater Treatment Facilities Annual Report(*HHWQTT*, 2007) and the Contaminant Loading Report(*HHRAPTT*, 2010).*CSO* volume estimates from the former source were available from 1996 to 2007, while measured *CSO* tank overflow volumes from the latter one were available from 2006 to 2012. When years with data from both sources were available and compared (2006 and 2007), we found that the measured *CSO* tank overflow values were approximately double estimates of the Hamilton Harbour *RAP* Technical Team (*HHRAPTT*, 2010). Given the City of Hamilton's data was based on continuous field measurements, we assumed they were more reliable and thus used the City of Hamilton's overflow values for 2006-2012 and doubled the *HH-RAP* report estimates for 1996-2005. Following the *HHRAPTT* (2010) approach, a fixed *CSO* concentration, based on the average measured *TP* concentrations from *CSO* tank influent, was used to calculate loading values.

Of the three main tributaries draining into Cootes Paradise, daily discharge data for the entire historical period were available only for Spencer Creek and were obtained from the Water Survey of Canada (*WSC*) Dundas flow gauge station. According to Theÿsmeÿer et al. (2009), a correction factor of 1.39 was used to adjust the areal ratio of drainage between the creek mouth and the Dundas monitoring site. Since data were not available for Chedoke and Borer's creeks, daily discharge had to be estimated by applying correction factors to tributaries with available data and similar watershed characteristics. Following Aquafor Beech Limited (2005), Chedoke Creek daily discharge was estimated by applying a correction factor of 0.49 to Redhill Creek discharge. However, since Redhill Creek discharge data were not available for the entire historical period, an equation was formulated to fill data gaps based on an established relationship between average precipitation from current and previous day and the available flow data. Daily discharge from Borer's Creek was estimated by applying a correction factor of 0.22 to Grindstone Creek discharge, following Theÿsmeÿer et al.'s (2009) methods. Discharge data for both Redhill and Grindstone Creeks were obtained from *WSC*.

Bi-monthly water quality data for the months of May to October were provided by the Royal Botanical Gardens for all three tributaries and were used to estimate daily *TP* loading from the tributaries.

Since *TP* concentrations were not available with a daily resolution, rating curves were utilized to estimate daily *TP* loads from tributary flows using the following equation:

$$\ln(L) = \beta_0 + \beta_1 \ln(Q)$$

where $L = C \cdot Q$ is the load, C is the *TP* concentration, Q is the flow discharge, β_0 and β_1 are regression coefficients. In this study, we opted for an average rating curve for the entire study period, as it provided more realistic tributary loading estimates relative to those derived from year-specific equations. Since the contributions of creeks and urban runoff are difficult to separate, it was assumed that urban runoff was already accounted for by creek nutrient loading to avoid double counting (*HHRAPTT*, 2010). Groundwater discharge and concentration measurements were used to estimate annual groundwater loading to Cootes Paradise (Ptacek and Blowes, 2004), following the methodology used by JEMSys Software Systems Inc. (2005). Daily loading was subsequently estimated by dividing the annual loading by the number of days in the year, and thus our groundwater loading estimates did not exhibit any intra-annual variability. Loading from precipitation was estimated using daily precipitation measurements from Environment Canada's Canadian Daily Climatic Data and a fixed *TP* concentration value (15 µg *TP* L^{-1}) originating from atmospheric loading estimates by Minns et al. (2004).

Water Budget

Daily water levels for Cootes Paradise were estimated using a predictive relationship that connects them with Lake Ontario water levels, as developed by Chow-Fraser et al. (1998). The latter predictor variable was provided by the *WSC* and was available for the entire study period at Burlington station. Daily water volumes were derived by the estimated daily water levels and bathymetric data for Cootes Paradise. Daily outflows from Cootes Paradise to Hamilton Harbour were estimated through regression analysis between daily discharge values measured at the Spencer Creek Hydat station and outflow data collected at the Desjardins Canal from April to November, 2009 (see Electronic Supplementary Material in Long et al., 2014). Because the water budget was not balanced when considering the derived outflows and measured inflows into Cootes Paradise marsh, we used an additional correction factor to account for this discrepancy. Available evidence demonstrates potential flow reversals between Cootes Paradise and Hamilton Harbour (Long et al., 2014), and thus the model considers both outflows and backflows. Depending on whether the correction factor was negative or positive, we postulated that backflows from Hamilton Harbour to Cootes Paradise or additional outflows from the marsh occur. Assuming that backflows brought water from the western end of the Harbour, we used *TP* concentrations from the Harbour's inner most monitoring station to calculate loading values. However, since data from this station were not available for the entire study period, we used regression analysis with the Harbour's centre monitoring station to estimate *TP* concentrations at the western station during the undocumented periods ($\ln[TP_{West}]=0.567 \cdot \ln[TP_{Center}]+1.586$, $\sigma_{error}=0.218$, n=78).

Temporal trends of tributary flows

Dynamic Linear Models (*DLM*s) were used to examine temporal trends of Spencer Creek flow, while explicitly accounting for the covariance between precipitation and tributary flow. Unlike conventional statistical approaches that use static parameters, *DLM*s use dynamic parameter values that reflect gradual shifts in the underlying processes over time (Lamon et al., 1998; Kim et al., 2013). Our *DLM* exercise consisted of an observation equation and a system equation, parameterized as follows:

Observation equation:

$$\ln[flow]_{ti} = level_t + \beta_t \cdot \ln[precipitation]_{ti} + \varepsilon_{ti} \quad \varepsilon_{ti} \sim N\,[0,\psi_t]$$

System equations:

$$level_t = level_{t-1} + rate_t + \omega_{t1} \quad \omega_{t1} \sim N\,[0,\, \Omega_{t1}]$$

$$rate_t = rate_{t-1} + \omega_{t2} \quad \omega_{t2} \sim N[0,\, \Omega_{t2}]$$

$$\beta_t = \beta_{t-1} + \omega_{t3} \quad \omega_{t3} \sim N\,[0,\, \Omega_{t3}]$$

$$1/\Omega^2_{tj} = \zeta^{t-1} \cdot 1/\Omega^2_{1j},\ 1/\psi_t^2 = \zeta^{t-1} \cdot 1/\psi_1^2 \quad t > 1 \text{ and } j = 1 \text{ to } 3$$

$$level_1,\ rate_1,\ \beta_1 \sim N(0,\ 10000) \quad t = 1$$
$$1/\Omega^2_{tj},\ 1/\psi_t^2 \sim G(0.001,\ 0.001)$$

where $\ln[flow]_{ti}$ and $\ln[precipitation]_{ti}$ represent the measured Spencer Creek flow and precipitation values in a logarithmic scale at day i of year t, β_t represents the year-specific regression coefficients between flow and precipitation, and $level_t$ denotes the year-specific intercepts when accounting for the relationship between the two covariates. Because the daily precipitation values were standardized prior to the analysis, the $level_t$ estimates could be interpreted as the geometric mean of annual flow. $Rate_t$ is the annual rate of change of the mean flow, ψ_{ti}, ω_{t1}, ω_{t2}, and ω_{t3} are draws from normal distributions with zero mean values and variances of ψ_t^2, Ω_{t1}^2, Ω_{t2}^2, and Ω_{t3}^2, respectively. The discount factor ζ represents the aging of information with the passage of time (Lamon et al., 1998; Sadraddini et al., 2011). The results reported here are based on a discount value of 0.95. $N(0,10000)$ is the normal distribution with mean 0 and variance 10,000; and $G(0.001,\ 0.001)$ is the gamma distribution with shape and scale parameters of 0.001. The prior distributions for the parameters of the initial year $level_1$, $rate_1$, β_1, $1/\Omega_{1j}^2$, $1/\psi_1^2$ are considered "non-informative" or vague.

Phosphorus model

We implemented a simple phosphorus mass-balance model to simulate the phosphorus budget in Cootes Paradise (Tables S1-2), based on a model originally developed by Minns and Moore (2004). Similar to Kim et al. (2013), the current study improves the original model structure by explicitly representing macrophyte dynamics and wind induced sediment resuspension. The model represents Cootes Paradise as a spatially homogeneous system with a hydraulic connection to Hamilton Harbour. Along with inflows and nutrient loading of the previously described point and non-point sources, water temperature, chlorophyll α, and macrophyte areal coverage data are used to force the model, as provided by the Royal Botanical Gardens monitoring program. Other forcing functions include solar radiation, day length, wind speed and direction, and evaporation, which are based on meteorological data from Environment Canada's daily climate database.

The ordinary differential equation dealing with ambient TP dynamics considers exogenous inflows, outflows, first-order sedimentation, phosphorus release via macrophyte respiration, TP reflux/diffusion and resuspension from the sediments. The role of macrophytes in the phosphorus cycle is accounted for by the dry-mass biomass submodel presented by Asaeda et al. (2000) and modified by Kim et al. (2013). However, unlike the previous models, we differentiate among three macrophyte functional groups: emergent, meadow, and submerged. Each equation considers macrophyte growth through uptake of dissolved inorganic phosphorus (DIP) from the interstitial water, respiration releasing phosphorus back to the water column, and mortality depositing phosphorus to the sediment pool. Since our model only explicitly considers sediment TP, interstitial DIP was calculated as a proportion of TP in the sediment mass. Temperature dependence was assigned to macrophyte mortality and respiration based on the Arrhenius equation (Chapra, 1997); whereas, growth temperature dependence was given an optimum level and is modeled by a function similar to the Gaussian probability curve (Cerco and Cole, 1994; Arhonditsis and Brett, 2005). Light availability was differentiated amongst macrophyte groups by assuming that the photosynthetic parts of emergent and meadow macrophytes were above the water surface and those of the submerged macrophytes were below the water surface. Therefore, the growth term for the emergent and meadow macrophytes is controlled by solar radiation reaching the water surface, while attenuated light modulates submerged macrophyte growth. For the latter process, Steele's equation coupled with Beer's law was used to scale photosynthetically active radiation to depth (Jassby and Platt, 1976). The extinction coefficient is determined as the sum of background light attenuation and attenuation due to chlorophyll a levels, which in turn were estimated using measured Secchi disk depth and chlorophyll a values in Cootes Paradise marsh (Jassby and Platt, 1976).

According to our model, sediment mass (kg) is defined as the product of the accumulation area (the product of the accumulation extent and the area of Cootes Paradise) multiplied by a "sediment factor".

The sediment factor is defined as a function of sediment thickness, porosity (i.e., sediment water content, %), and sediment solids density. The sediment *TP* equation considers additions of phosphorus to the sediment pool via settling and macrophyte mortality and losses through reflux and resuspension to the water column, burial to deeper sediments, and macrophyte uptake. Following Minns and Moore's (2004) strategy, the estimation of reflux (mg m^{-2} day^{-1}) from the sediment pool to the water column was done by fitting an exponential function, $R = a_{sed} \times e^{b_{sed} \times P_{sed}}$, to historical reflux data from the Bay of Quinte, Ontario. Since historical reflux data were not available for Cootes Paradise, we assumed reflux rates in the two systems were similar and adopted Minns and Moore's (2004) empirical coefficients as the default values for $a_{sed} (= 0.02)$and $b_{sed} (= 4.5)$. To validate the latter approach, we made comparisons between our estimated reflux rates and those reported by Mayer et al. (2005) in Cootes Paradise in the summer of 2001. Temperature dependence based on the Arrhenius equation was assigned to the reflux rates.

Following Kim et al. (2013), we used an empirical expression that postulates a linear relationship between sediment resuspension and excess bed shear stress (Mehta et al., 1982; Chao et al., 2008). It was assumed that the bottom shear stress associated with the near-bed current velocity was negligible compared to the near-bed wave velocity (Mian and Yanful, 2004). Quantification of the time-variant bed shear stress, as a function of wave characteristics (height, period length), water depth, wind speed, and fetch length followed the Sverdrup-Munk-Bretschneider (SMB) method for shallow water bodies (CERC, 1994). The burial process to the deeper sediments (effectively acting as a closure term in our model) is determined by the time-variant phosphorus mass in the sediments divided by a deposition coefficient. The latter defines the fraction of the sediment phosphorus pool disappearing each day through the burial process and is a function of the tributary sediment input divided by the previously defined sediment factor. The tributary sediment input was calculated using a fixed suspended sediment concentration in the tributary inputs, based on the work of Minns (1986) in the Bay of Quinte, divided by the average annual tributary flow across the entire historical period (1996-2012). Sediment input was then multiplied by the year-specific average daily tributary flow to accommodate the inter-annual variability.

Model evaluation-Sensitivity Analysis

Model goodness of fit was evaluated using the following summary statistics (Stow et al., 2003): (*i*) Root Mean Squared Error (*RMSE*) = $\sqrt{\frac{\sum_{i=1}^{N}(M_i - O_i)^2}{N}}$, (*ii*) Relative Error (*RE*) = $\frac{\sum_{i=1}^{N}|M_i - O_i|}{\sum_{i=1}^{N} O_i}$, and (*iii*) Modelling Efficiency (*MEF*) = $\frac{\sum_{i=1}^{N}(O_i - \bar{O})^2 - \sum_{i=1}^{N}(M_i - O_i)^2}{\sum_{i=1}^{N}(O_i - \bar{O})^2}$; where N is the number of observations; O_i is the *i*th of N observations; M_i is *i*th of N predictions; and $\bar{O}$ represents the average observed values. *RMSE* and *RE* are measures of the accuracy of model predictions and values close to zero are indicative of a close match between observed values and model predictions. *MEF* measures the predictive capacity of a model relative to the average observation. A *MEF* value of 1 corresponds to a perfect fit of model predictions and observed data; a value of 0 indicates that model predictions are as accurate as the average value of the observations; and values less than zero indicate that the average observation would be a better predictor than the model itself.

We also evaluated the sensitivity of model outputs to variations of model parameters and our exogenous phosphorus loading estimates. Independent perturbations were induced to each model input/parameter and subsequent changes in predicted state variables (*TP* concentrations and macrophyte biomass) were monitored to determine the influence of model input uncertainties. We implemented our sensitivity analysis in three stages. The first stage involved perturbations on eight external forcing functions (*TP* loads from backflow, groundwater, Dundas *WWTP*, urban runoff, *CSO* runoff, Borer's Creek, Chedoke Creek and Spencer Creek), while the second one involved perturbations (±15%) on eight model parameters. The final analysis involved perturbations on both the eight external forcing functions and the five most influential parameters. Gaussian distributions with mean and standard deviations equal to the rating curve mean estimates and standard error values were correspondingly assigned to the logtransformed daily phosphorus loading from the three tributaries (Spencer, Chedoke, and Borer's). Multiple regression models were developed for water column and sediment *TP* concentrations, and total macrophyte biomass with the relative influence of each input element ranked based on the squared semi-partial correlation coefficient (r^2_{spart}). Based on this assessment, the five most influential parameters were identified and combined with all the external forcing functions for the final stage of our sensitivity analysis.

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FIGURES LEGENDS

Figure S-1: Dynamic linear modeling analysis of temporal trends of flow rates in Spencer Creek between 1985-2013: (*a*) daily flow rates binned per year, with black line delineating the median annual flows; (*b*) annual rates of change of average flow rates; (*c*) time-series of the posterior regression coefficient β depicting the strength of the association between flow and precipitation, where the dashed lines correspond to 95% uncertainty intervals; and (*d*) annual coefficient of variation of the β estimates.

Figure S-2: Relationship between predicted *TP* concentrations in the water column against simulated (*a*) diffusive reflux rates from sediments, and (*b*) sediment burial rates into deeper layers in Cootes Paradise.

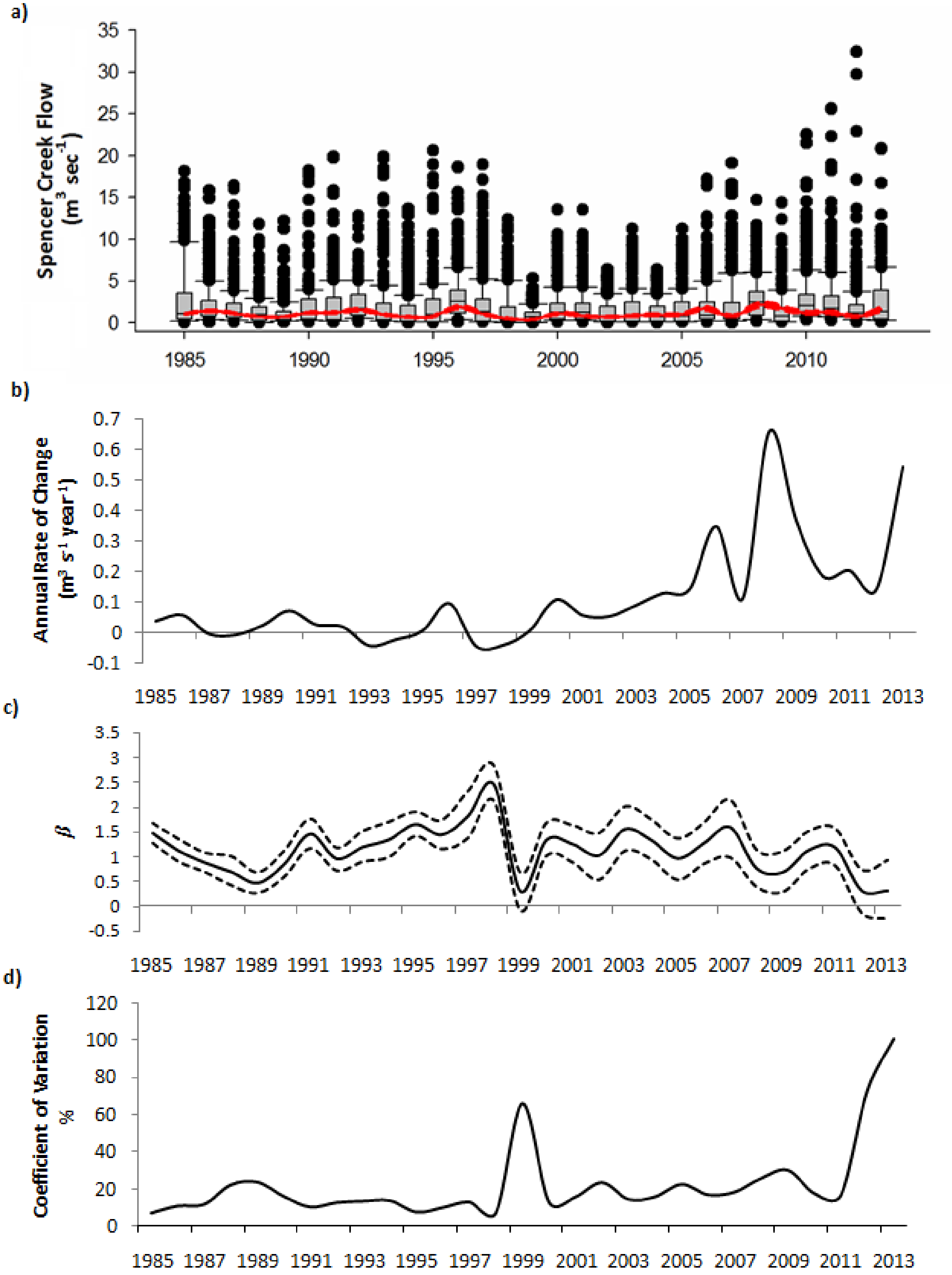
a)
Spencer Creek Flow
(m³ sec⁻¹)
35
30
25
20
15
10
5
0
1985
1990
1995
2000
2005
2010
b)
Annual Rate of Change
(m³ s⁻¹ year⁻¹)
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
-0.1
1985 1987 1989 1991 1993 1995 1997 1999 2001 2003 2005 2007 2009 2011 2013
c)
β
3.5
3
2.5
2
1.5
1
0.5
0
-0.5
1985 1987 1989 1991 1993 1995 1997 1999 2001 2003 2005 2007 2009 2011 2013
d)
Coefficient of Variation
%
120
100
80
60
40
20
0
1985 1987 1989 1991 1993 1995 1997 1999 2001 2003 2005 2007 2009 2011 2013

Figure S-1

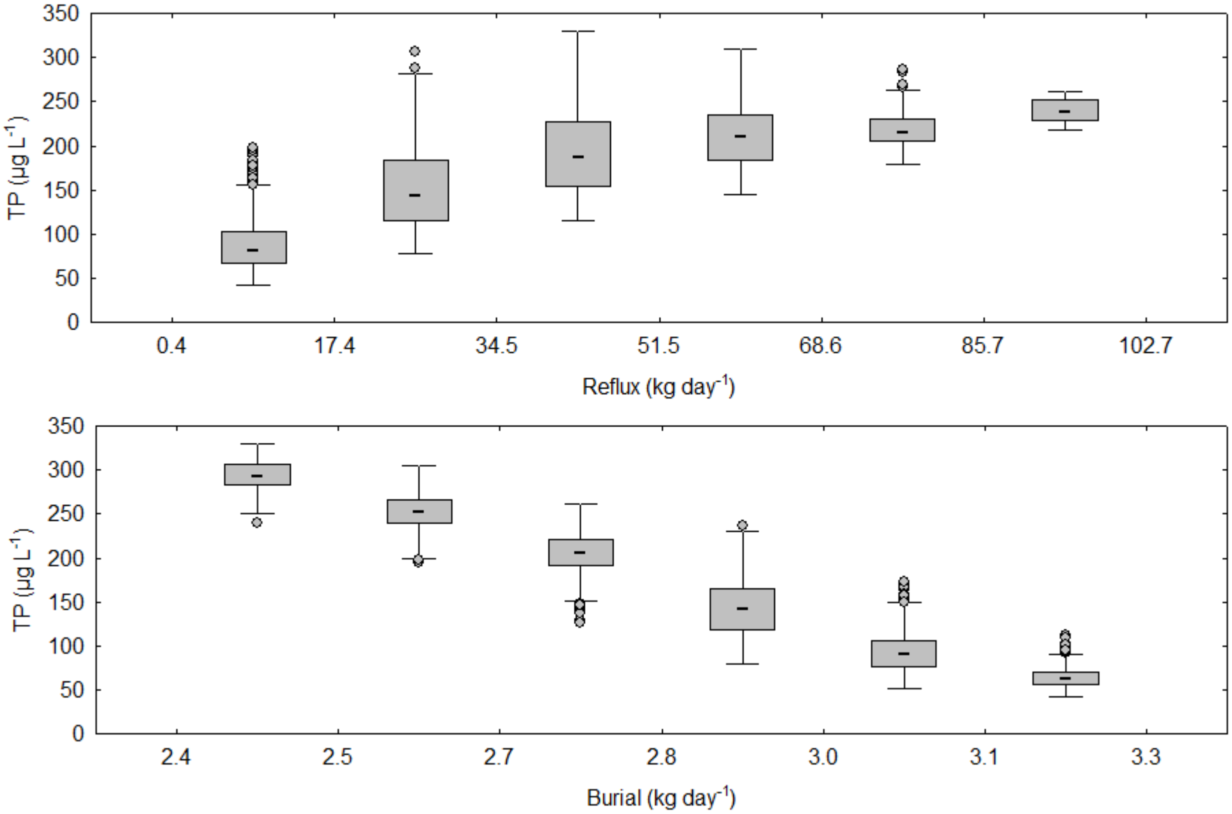
TP (µg L^{-1})
350
300
250
200
150
100
50
0
0.4
17.4
34.5
51.5
68.6
85.7
102.7
Reflux (kg day^{-1})
TP (µg L^{-1})
350
300
250
200
150
100
50
0
2.4
2.5
2.7
2.8
3.0
3.1
3.3
Burial (kg day^{-1})

Figure S-2

Table S-1: Mathematical equations of the total phosphorus model for Cootes Paradise marsh.

Process	***Symbol***	***Equation***
Water column	$\frac{\mathrm{dTP_w}}{\mathrm{dt}}$	$\frac{\mathrm{TP}_{in} + \mathrm{TP}_{Resus} + \mathrm{TP}_{Reflux} + \mathrm{TP}_{macR} - \mathrm{TP}_{out} - \mathrm{TP}_{wS} + [\mathrm{TP_{backflow}}]}{\mathrm{V_w}}$
	$\mathrm{TP}_{sedResus}$	$\mathrm{sed_A} \cdot \mathrm{R_{resus}}$
	$\mathrm{R_{resus}}$	$\mathrm{a}_{Resus}\left(\frac{\tau - \tau_c}{\tau_c}\right)^{b_{Resus}} \quad \text{if } \tau \geq \tau_c \text{ ,} \quad 0 \quad \text{if } \tau < \tau_c$
	$\mathrm{TP}_{sedReflux}$	$R_{reflux} \cdot A \cdot \theta^{T-20}$
	R_{reflux}	$a_{reflux} \cdot e^{TP_{sed} \cdot b_{reflux}}$
	$\mathrm{TP_{macR}}$	$\sum_{\mathrm{i}} \alpha_{\mathrm{mac_i}} \cdot \mathrm{R_{mac_i}} \cdot \mathrm{BP_{mac}} \cdot \mathrm{B_{mac_i}}$
	$\mathrm{TP_{wS}}$	$\frac{\mathrm{V_s}}{\mathrm{z}} \cdot \mathrm{TP_w}$
Macrophytes	$\frac{\mathrm{dB_{mac_i}}}{\mathrm{dt}}$	$(\mathrm{G_{mac_i}} - \mathrm{R_{mac_i}} - \mathrm{D_{mac_i}}) \cdot \mathrm{B_{mac_i}}$
	$\mathrm{G_{mac_i}}$	$\mathrm{P_{rmac_i}} \frac{\mathrm{DIP_{sed}}}{\mathrm{K_{pmac_i}} + \mathrm{DIP_{sed}}} \mathrm{f_L(t)} \cdot \mathrm{f_{temp_i}}$
	f_{temp_i}	$e^{-KTgr1_i(T-T_{opt})^2} \, when\ T \leq Topt_{mac_i}$
		$e^{-KTgr2_i(T_{opt}-T)^2} \, when\ T > Topt_{mac_i}$
	$\mathrm{f_L(t)}$	$\frac{2.718\ \mathrm{FD}}{\mathrm{K_{ext}\ Z_{mac}}}\{\mathrm{e^{-x1}} - \mathrm{e^{-x2}}\}$
	x1	$\frac{\mathrm{I_0 e^{-K_{ext} Z_{mac}}}}{\mathrm{FD\ I_{opt}}}$
	x2	$\frac{\mathrm{I_0}}{\mathrm{FD\ I_{opt}}}$
	$\mathrm{K_{ext}}$	$\alpha_1 + \alpha_2 \mathrm{chl}a$
	$\mathrm{R_{mac_i}}$	$\mathrm{R_{20mac_i}} \cdot \theta_{\mathrm{Rmac}}{}^{(\mathrm{T-20})}$

	D_{mac_i}	$\mathrm{D}_{20\mathrm{mac}_\mathrm{i}} \cdot \theta_{\mathrm{Dmac}}{}^{(\mathrm{T}-20)}$
Sediment	$\frac{\mathrm{dTP}_{\mathrm{sed}}}{\mathrm{dt}}$	$\frac{\mathrm{TP}_{\mathrm{wS}} - \mathrm{TP}_{Resus} - \mathrm{TP}_{Reflux} - \sum_\mathrm{i} \mathrm{G}_{\mathrm{mac}_\mathrm{i}} \cdot a_{\mathrm{mac}_\mathrm{i}} \cdot \mathrm{B}_{\mathrm{mac}_\mathrm{i}} \cdot \mathrm{BP}_{\mathrm{mac}} + \sum_\mathrm{i} \mathrm{D}_{\mathrm{mac}_\mathrm{i}} \cdot \alpha_{\mathrm{mac}_\mathrm{i}} \cdot \mathrm{B}_{\mathrm{mac}_\mathrm{i}} \cdot \mathrm{BP}_{\mathrm{mac}} - \mathrm{TP}_\mathrm{B}}{\mathrm{sed}_\mathrm{m}}$
	$\frac{dTP_B}{dt}$	$\frac{\mathrm{TP}_{\mathrm{sed}} \cdot \mathrm{sed}_\mathrm{m} \cdot \theta_{\mathrm{dep}}}{\text{average time}} - \mathrm{TP}_\mathrm{B}$
	θ_{dep}	$\frac{sed_{in}}{sed_f}$
	sed_{in}	$\frac{Q_{annual} \cdot sed_{dep} \cdot Q_{avg}}{365}$
	sed_m	$\mathrm{sed}_\mathrm{f} \cdot sed_A$
	sed_A	$A \cdot \left(1 - \frac{sed_{AE}}{100}\right)$
	sed_f	$sed_t \cdot sed_{SD} \cdot \left(1 - \frac{sed_{WC}}{100}\right)$

i = macrophyte group *A* (emergent), *B* (meadow), *C* (submergent)

Table S-2: State variables and parameters of the total phosphorus model for Cootes Paradise marsh.

Symbol	***Variables and Parameters***	***Value***	***Unit***
a_{reflux}	Reflux coefficient A	0.02	*mg* m^{-2} day^{-1}
a_{resus}	Resuspension A	8	*mg* m^{-2} day^{-1}
$\mathrm{B}_{\mathrm{mac}_\mathrm{A}}$	Emergent macrophyte biomass (dry weight)		*g* m^{-2}
$\mathrm{B}_{\mathrm{mac}_\mathrm{B}}$	Meadow macrophyte biomass (dry weight)		*g* m^{-2}
$\mathrm{B}_{\mathrm{mac}_\mathrm{C}}$	Submergent macrophyte biomass (dry weight)		*g* m^{-2}
b_{reflux}	Reflux coefficient B	5.0	*unitless*
b_{Resus}	Resuspension B	1	*unitless*
$\mathrm{BP}_{\mathrm{mac}}$	Phosphorus content in macrophyte biomass	0.0025	*g P g dry* $weight^{-1}$
$\mathrm{D}_{\mathrm{mac}_\mathrm{A}}$	Emergent macrophyte mortality rate		day^{-1}
$\mathrm{D}_{\mathrm{mac}_\mathrm{B}}$	Meadow macrophyte mortality rate		day^{-1}
$\mathrm{D}_{\mathrm{mac}_\mathrm{C}}$	Submergent macrophyte mortality rate		day^{-1}
$D_{20\mathrm{mac}_\mathrm{A}}$	Emergent macrophyte mortality rate 20°C	0.001	day^{-1}
$D_{20\mathrm{mac}_\mathrm{B}}$	Emergent macrophyte mortality rate 20°C	0.001	day^{-1}
$D_{20\mathrm{mac}_\mathrm{C}}$	Emergent macrophyte mortality rate 20°C	0.001	day^{-1}
$\mathrm{DIP}_{\mathrm{sed}}$	DIP in the sediment interstitial water		*µg* L^{-1}
$\mathrm{G}_{\mathrm{mac}_\mathrm{A}}$	Emergent macrophyte growth rate		day^{-1}
$\mathrm{G}_{\mathrm{mac}_\mathrm{B}}$	Meadow macrophyte growth rate		day^{-1}
$\mathrm{G}_{\mathrm{mac}_\mathrm{C}}$	Submergent macrophyte growth rate		day^{-1}
$\mathrm{I}_{\mathrm{opt}}$	Optimal solar radiation for macrophyte growth	15	*MJ* m^{-2} day^{-1}
$\mathrm{K}_{\mathrm{pmac}_\mathrm{A}}$	Half saturation constant of emergent macrophytes for phosphate in sediment pore water	3000	*µg* L^{-1}
$\mathrm{K}_{\mathrm{pmac}_\mathrm{B}}$	Half saturation constant of meadow macrophytes for phosphate in sediment pore water	3000	*µg* L^{-1}
$\mathrm{K}_{\mathrm{pmac}_\mathrm{C}}$	Half saturation constant of submergent macrophytes for phosphate in sediment pore water	1500	*µg* L^{-1}
$\mathrm{K}_{\mathrm{ext}}$	Extinction coefficient		m^{-1}
$KTgr1_A$	Effect of temperature below the optimal temperature on emergent macrophyte growth	0.004	C^{o-2}
$KTgr1_B$	Effect of temperature below the optimal temperature on meadow macrophyte growth	0.0005	C^{o-2}
$KTgr1_C$	Effect of temperature below the optimal temperature on submergent macrophyte growth	0.004	C^{o-2}
$KTgr2_A$	Effect of temperature above the optimal temperature on emergent macrophyte growth	0.0005	C^{o-2}
$KTgr2_B$	Effect of temperature above the optimal temperature on meadow macrophyte growth	0.004	C^{o-2}
$KTgr2_C$	Effect of temperature above the optimal temperature on submergent macrophyte growth	0.0005	C^{o-2}
$\mathrm{P}_{\mathrm{rmac}_\mathrm{A}}$	Maximum gross photosynthesis rate of emergent macrophytes	0.066	day^{-1}
$\mathrm{P}_{\mathrm{rmac}_\mathrm{B}}$	Maximum gross photosynthesis rate of meadow macrophytes	0.066	day^{-1}
$\mathrm{P}_{\mathrm{rmac}_\mathrm{C}}$	Maximum gross photosynthesis rate of submergent macrophytes	0.065	day^{-1}
$\mathrm{R}_{20\mathrm{mac}_\mathrm{A}}$	Emergent macrophyte respiration rate at 20°C	0.0181	day^{-1}

R_{20mac_B}	Meadow macrophyte respiration rate at 20°C	0.0180	day^{-1}
R_{20mac_C}	Submergent macrophyte respiration rate at 20°C	0.0180	day^{-1}
R_{mac_A}	Emergent macrophyte respiration rate		day^{-1}
R_{mac_B}	Meadow macrophyte respiration rate		day^{-1}
R_{mac_C}	Submergent macrophyte respiration rate		day^{-1}
R_{reflux}	Reflux rate		$mg\ m^{-2}\ day^{-1}$
R_{resus}	Sediment resuspension rate		$mg\ m^{-2}\ day^{-1}$
Q_{annual}	Year-specific whole bay tributary flow		$m^3\ year^{-1}$
Q_{avg}	Average annual (1996-2012) whole bay tributary flow		$m^3\ year^{-1}$
sed_A	Accumulation area		m^2
sed_{AE}	Accumulation extent	20	m^2
sed_{dep}	Sediment deposition		$g\ cm^{-2}\ year^{-1}$
sed_f	Sediment factor		$g\ cm^{-2}$
sed_{in}	Sediment input		$g\ cm^{-2}\ day^{-1}$
sed_{SD}	Sediment solid density	2.45	$g\ cm^{-3}$
sed_m	Sediment mass		*kg*
sed_t	Sediment thickness	20	*cm*
$Topt_{mac_A}$	Optimal temperature for emergent macrophyte growth		*ºC*
$Topt_{mac_B}$	Optimal temperature for meadow macrophyte growth		*ºC*
$Topt_{mac_C}$	Optimal temperature for submergent macrophyte growth		*ºC*
TP_B	Burial of phosphorus in the sediment		$g\ kg^{-1}\ day^{-1}$
TP_{in}	Total phosphorus fluxes from exogenous sources		$kg\ day^{-1}$
TP_{macR}	Total phosphorus fluxes from macrophyte respiration		$kg\ day^{-1}$
TP_{out}	Total phosphorus outflow fluxes		$kg\ day^{-1}$
TP_{sed}	Total phosphorus concentration in the sediments		$g\ kg^{-1}$
TP_{Reflux}	Total phosphorus fluxes from reflux		$kg\ day^{-1}$
TP_{Resus}	Total phosphorus fluxes from resuspension		$kg\ day^{-1}$
TP_w	Total phosphorus concentration in the water column		$\mu g\ L^{-1}$
TP_{wS}	Total phosphorus settling		$kg\ day^{-1}$
V_s	Settling rate	0.04	$m\ day^{-1}$
V_w	Volume as a function of time, determined by the water balance		m^3
α_{mac_A}	Emergent macrophyte areal coverage		m^2
α_{mac_B}	Meadow macrophyte areal coverage		m^2
α_{mac_C}	Submergent macrophyte areal coverage		m^2
α_1	Background extinction coefficient		m^{-1}
$\alpha_2 chla$	Phytoplankton self-shading effect		$m^2\ mg\ chla^{-1}$
θ_{Dmac}	Temperature dependence of macrophyte mortality	1.08	*unitless*

θ_{Rmac}	Temperature dependence of macrophyte respiration	1.08	*unitless*
A	Area		m^2
FD	Time fraction of daily solar radiation		*unitless*
I_o	Solar radiation on the surface		$MJ\ m^{-2}\ day^{-1}$
T	Water temperature		$^{\circ}C$
z	Water depth, determined by Lake Ontario water levels		*m*
Z_{mac}	Water depth from the surface to the top of macrophyte bed	0.4	*m*
τ	Sediment bed shear stress		$N\ m^{-2}$
τ_c	Critical sediment bed shear stress	0.03	$N\ m^{-2}$

Table S-3: Sensitivity results of the Cootes Paradise phosphorus model. Ranking was based on the squared semi-partial correlation coefficients (r^2_{spart}) values for the annual average of water column and sediment *TP* concentrations and total macrophyte biomass. Determination coefficients (r^2) of the respective multiple regression models are provided in parentheses.

	External forcing (0.995)	r^2_{spart}	Model parameters (0.965)	r^2_{spart}	Combined (0.991)	r^2_{spart}
Water column *TP*	Spencer Creek	0.532	Reflux coefficient B	0.605	Reflux coefficient B	0.499
	Dundas WWTP	0.172	Settling rate	0.228[a]	Settling rate	0.191[a]
	CSOs	0.159	Maximum macrophyte growth rate	0.029	Spencer Creek	0.136
	Chedoke Creek	0.104	Reflux coefficient A	0.027	Maximum macrophyte growth rate	0.047
	Urban runoff	0.043	Macrophyte respiration rate	0.022[a]	Macrophyte respiration rate	0.040[a]
	External forcing (0.988)	r^2_{spart}	Model parameters (0.995)	r^2_{spart}	Combined(0.997)	r^2_{spart}
Sediment *TP*	Spencer Creek	0.466	Reflux coefficient B	0.261[a]	Reflux coefficient B	0.358[a]
	CSOs	0.209	Settling rate	0.226	Settling rate	0.320
	Dundas WWTP	0.202	Sediment water content	0.151	Spencer Creek	0.146
	Chedoke Creek	0.089	Sediment solid density	0.116[a]	CSOs	0.035
	Urban runoff	0.038	Sediment deposition	0.114[a]	Dundas WWTP	0.033
	External forcing (0.853)	r^2_{spart}	Model parameters (0.787)	r^2_{spart}	Combined(0.922)	r^2_{spart}
Total macrophyte biomass	Spencer Creek	0.379	Maximum macrophyte growth rate	0.220	Maximum macrophyte growth rate	0.508
	CSOs	0.245	Macrophyte respiration rate	0.178[a]	Macrophyte respiration rate	0.445[a]
	Dundas WWTP	0.187	Half-saturation constant of light limitation for macrophytes	0.143[a]	Chedoke Creek	0.000
	Chedoke Creek	0.051	Sediment water content	0.135[a]	Reflux coefficient B	0.000[a]
	Urban runoff	0.028	Half-saturation constant of nutrient limitation for macrophytes	0.045	Borer’s Creek	0.000[a]

[a] Negative sign of the regression coefficients.