Article

Phosphorus dynamics across intensively monitored subcatchments in the Beaver River

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Abstract

We report results from a spatially intensive monitoring and modelling study to assess phosphorus (P) dynamics in the Beaver River, a tributary of Lake Simcoe, Ontario. We established multiple monitoring stations (9 flow and 24 water quality stations) from headwaters to near the outflow that were operated for 2 field seasons, complementing longer term data from a flow monitoring site and water chemistry monitoring site. We applied the Branched-INCA-P model, which allows fully distributed simulations supported by highly distributed monitoring data. Using spatially distributed data helped better understand variable P and sediment dynamics across the catchment and identify key model uncertainties and uncertainties related to catchment P management. Measured and modelled total P concentrations often exceeded provisional water quality thresholds in many areas of the catchment and highlight the value of studying water quality across multiple subcatchments rather than at a single site. Total P export coefficients differed widely among subcatchments, ranging from 2.1–21.4 kg km^{-2} y^{-1} over a single year. Export coefficients were most strongly (negatively) related to the proportion of wetland cover in subcatchments. The INCA-P model captured spatial variation in P concentrations relatively well, but short-term temporal variability in the observed data was not well simulated across sites, in part due to unmodelled hydrological phenomena including beaver activity and unknown drivers of P peaks that were not associated with hydrological events.

Key words: biogeochemical modelling, eutrophication, export coefficient, Lake Simcoe, nutrient, phosphorus stream

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Introduction

Controlling the input of nutrients into aquatic ecosystems is a management focus across much of the world. Nutrient management is essential to prevent or help mitigate harmful effects of eutrophication, an environmental problem that has caused billions of dollars in economic harm in the United States alone (Dodds et al. 2009). Advanced treatment of wastewater has led to significant reductions in nutrient loads to many ecosystems; however, in many cases, control of point sources alone is inadequate, and nonpoint sources must also be addressed.

Lake Simcoe is located within Canada's most populous province, just north of its largest city, Toronto, and is the largest lake in Ontario, with the exception of the Laurentian Great Lakes. Like most inland waters of northern North America, Lake Simcoe is phosphorus (P) limited, and like many lakes, it is subject to elevated P loads from both urban and agricultural inputs. Elevated P inputs have led to increased algal productivity and hypolimnetic oxygen decline (Winter et al. 2007, OMOE 2009), threatening the recreational fishery on the lake, which is among the most important in the province (Hogg et al. 2009). Concern about the degraded state of the lake and economic impacts to the fishery led to development of ambitious management plans to facilitate lake restoration. The Lake Simcoe Phosphorus Reduction Strategy requires a 40% decrease in P loads to the lake by 2045 (OMOE 2010). Given the largest source of nutrients to the lake is its tributaries (LSRCA and OMOE 2009), achieving P load reductions will require major reductions in P export from its numerous river inflows.

Mathematical modelling has proven to be a useful tool in understanding nutrient fluxes at large spatial scales and in helping to inform appropriate management, including prescribing necessary load reductions to protect critical habitat, understanding sources in the landscape, and assessing potential remediation options under current and future climate scenarios. Different approaches to modelling nutrient dynamics range in complexity from simple steady-state empirical modelling using annual timesteps (e.g., export-coefficient approaches) to highly parameterized process-based dynamic modeling using much shorter timesteps (Radcliffe et al. 2009). These approaches differ not only in their data needs and temporal scale, but also in their effectiveness in different situations. Steady-state approaches have limited utility in assessing scenarios of future conditions, such as changes in climate and land use, while major concerns associated with dynamic models tend to focus on their extensive data requirements. Finally, the spatial scale at which nutrient models are typically implemented also ranges widely, from field scale to watershed and continental scales.

Typically, water quality monitoring programs focus on the main stem of rivers, and as a result, the focus of model-based studies has been on representing nutrient dynamics within the main stem of rivers. An improved understanding of the spatial pattern of nutrient fluxes is beneficial, however, particularly in understanding non-point-source P loading. In this study we applied the integrated catchment model of phosphorus dynamics (INCA-P) to support the first spatially intensive test of a new model structure. Our objective was to improve understanding of the spatial variability of different P sources in the catchment and variability in P concentrations in the river. Ultimately, we aim to build a more comprehensive process-based assessment of potential effects of climate and land use change in different areas of the Lake Simcoe catchment (Whitehead et al. 2011, Crossman et al. 2013) and better understand how ambitious nutrient load reduction targets may be achieved. In addition to this process-based modelling work, we report results of our spatially intensive sampling campaign and characterize P concentrations, sediment loads, and export coefficients for 23 different areas of the catchment. These results will support future comparison to other, less data intensive modelling approaches.

Study site

For much of its length, the Beaver River is a wide, meandering, shallow, slow flowing stream located in a gently sloping till plain (Fig. 1; LSRCA 2010). Parts of the 327 km^2 catchment are characterized by productive loamy soils that have been farmed since the 1820s. The river is underlain by organic soils and flows through an extensive wetland complex (LSRCA 2010).

The Beaver River catchment is representative of the mixed agricultural land uses in the Lake Simcoe area. Current land cover includes agriculture (28.4% of land area in intensive agriculture and 37.2% in nonintensive agriculture), wetland (19.1%), forest land cover (10.5%), and a relatively small urbanized area (4.8% of catchment area; Fig. 1; GIS data based on Ecological Land Classification of Ontario data provided by the Lake Simcoe Region Conservation Authority; LSRCA 2006). Major crops in the catchment were alfalfa, corn (primarily for grain), soybeans, and winter wheat (Statistics Canada 2008). Livestock raised in the catchment were primarily poultry (total number of hens and chickens: ~140 000) and cattle (~9000 cattle and calves). Census data indicate ~2500 pigs were in the catchment, in addition to ~2500 sheep and lambs.

Because of greenbelt legislation intended to protect current agricultural land uses and natural environments from urbanization across a large area of southern Ontario

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(Greenbelt Act 2005), anticipated future changes in land use in the Beaver River catchment are small. The areas of hay and pasture, cropland, forest, wetland, quarry, turf-sod, and unpaved roads are expected to vary by <1%. Although the area of high intensity development is expected to increase by 51% in the Beaver River catchment, the current area of high intensity development is so small that this represents a change from ~2 to 3% of the catchment area by 2031 (Louis Berger Group Inc. 2010). Phosphorus concentrations in this tributary to Lake Simcoe are among the lowest of Simcoe's tributaries and are declining (Winter et al. 2007).

Methods

Monitoring flow, nutrients, and suspended solids

Long-term flow data were available from a single site, Environment Canada gauge 02EC011, which provides year-round data and is located near BVR05 (Fig. 1). These data were supplemented by flow monitoring in our project at an additional 8 sites (BVR02, BVR04, BVR07, BVR09, BVR10, BVR11, BVR13, and BVR27), using continuous stage monitoring (Trutrack capacitance rod data loggers). Data from these sites collected during winter months were not used due to concerns regarding the effects of ice on equipment and stage measurements.

We used longer-term water quality monitoring data from a single station in the Beaver River catchment (upstream of BVR01). Sampling was typically performed 2–4 times per month, with some additional event-based sampling. Total phosphorus (TP) was analysed colorimetrically following a digestion on the whole water sample (OMOE 1983). Total suspended solids (TSS) were analysed gravimetrically after filtering a known volume of water through a glass fibre filter (1.5–2.0 µm pore size, 47 mm diameter; OMOE 1983). Monitoring at this site has been ongoing for more than 15 years; however, only data from 2006 onward are presented here.

To support catchment scale assessment of nutrient dynamics and modelling, we initiated a monitoring program that included sampling of 24 sites (including

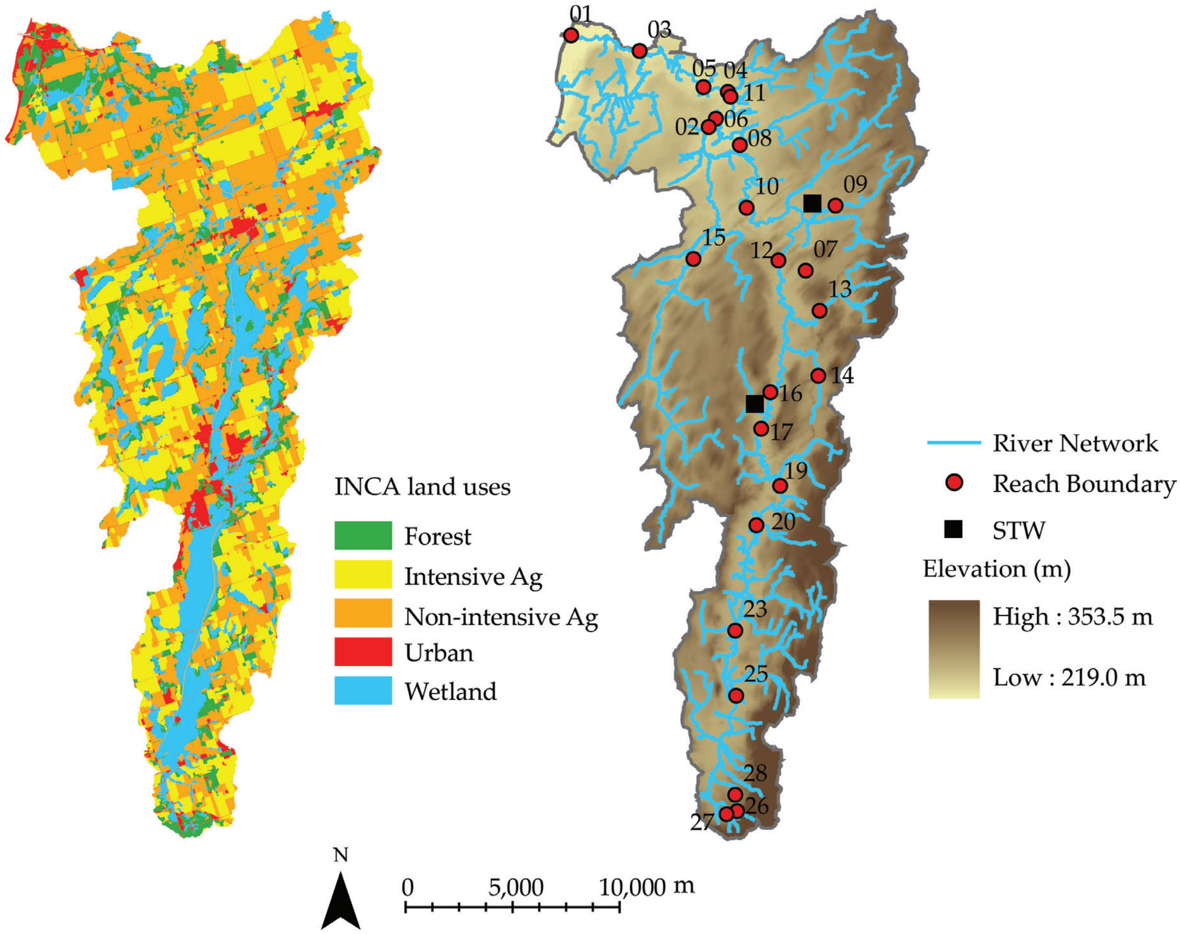

Fig. 1. Land use data, topography, and reach boundaries (including sampling locations) within the Beaver River catchment. Sewage treatment works (STW) are also indicated. Site numbers shown on the map are indicated in the text according to number, with the prefix BVR.

Table 1. Area and land use in study subcatchments (areas are cumulative and include nested subcatchments).

Site ID (BVR)	Subcatchment area (km^2)	% Land use				
		Urban	Intensive agriculture	Nonintensive agriculture	Wetland	Forest
01	325.3	4.9	28.3	37.0	19.3	10.5
02	75.0	1.9	31.7	39.0	17.8	9.6
03	322.6	4.6	28.5	37.1	19.4	10.4
04	3.4	4.2	32.4	25.5	9.5	28.3
05	299.6	4.7	30.1	35.6	19.9	9.7
06	0.1	0.0	0.0	73.2	3.6	23.2
07	0.2	0.0	0.0	61.6	32.5	5.8
08	185.4	6.5	26.0	34.5	23.0	10.0
09	8.3	2.8	14.5	58.9	19.5	4.4
10	178.0	6.7	26.1	33.4	23.6	10.2
11	3.0	0.4	65.2	32.5	0.8	1.1
12	136.8	6.7	26.6	29.3	26.1	11.3
13	10.1	5.4	31.8	30.8	17.5	14.5
14	5.2	4.0	37.8	23.2	20.0	14.9
15	48.4	1.8	40.9	29.0	18.8	9.4
16	100.4	7.8	29.3	26.5	25.4	11.0
17	93.2	7.9	30.1	25.4	25.4	11.1
19	77.2	5.9	31.5	25.3	26.0	11.2
20	71.3	4.7	32.7	25.7	26.1	10.8
23	40.9	5.0	30.3	25.8	24.9	14.0
25	26.7	6.8	30.4	22.0	24.9	15.9
26	0.4	12.7	2.9	6.8	11.6	66.1
27	0.6	5.1	0.9	5.8	19.3	69.0
28	1.1	6.0	28.9	26.7	7.3	31.1

BVR01) within the Beaver River catchment (Fig. 1; Table 1) from ~July 2009 to December 2011 (exact periods differed by sampling site). We sampled sites along 7 tributaries and multiple sites along the main stem. Samples were obtained ~1–2 times per month and analysed for TP and total dissolved phosphorus (TDP; filtered through 0.45 μm filter) using methods noted previously. Sampling did not specifically target events, although some were captured in the monitoring program. Long-term monitoring data at BVR01 were used interchangeably with our targeted monitoring program. We measured turbidity of all water samples (Man-Tech, PC-Titrate System Plus; Turbidity Assay plus) and TSS for a subset of samples, then developed turbidity–TSS relationships (linear regression; minimum r^2 of 0.81, average r^2 of 0.91) specific to 9 sites (BVR01, BVR05, BVR09, BVR10, BVR11, BVR19, BVR20, BVR26, and BVR28). These relationships were used to estimate TSS based on more frequent monitoring of turbidity.

Export coefficients

Export coefficients for TSS and TP were calculated using measured water chemistry (monthly average) and modelled flow data for each subcatchment within the Beaver River basin. We assessed relationships between land use (% of catchment) and P export coefficients using best subset regression and least-squares regression in SYSTAT 13.0 (Systat Software, Inc., Chicago, USA). We excluded % forested area from analyses to avoid multicollinearity. In addition, we analysed data only for sites where there were no monitored inflows to avoid noninde-

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pendence in data (as upstream sites influence results at more downstream sites). The best models were selected according to adjusted r^2, Akaike's information criterion (AIC), and Schwarz's Bayesian Information Criterion (BIC). Both AIC and BIC are measures of goodness of fit designed to avoid overfitting models by imposing penalties based on the number of model parameters.

Modelling

We used the dynamic, semidistributed, process-based model INCA-P to simulate water quality. INCA produces estimates of discharge and P concentrations on a daily time scale, across multiple points within a catchment and a river channel. Sediment dynamics are also simulated on a daily timestep within INCA-Sed (Lazar et al. 2010, Whitehead et al. 2010), which is a module of INCA-P. Particle P and soluble P dynamics are simulated in recently revised equations described by Wade et al. (2009). The most recent development in INCA-P is a revision of the model structure to allow simulation of fully branched river networks, allowing assessment of changes in nutrient concentrations and nutrient loads from the headwaters of multiple tributaries to a river main stem and ultimately its outflow (Whitehead et al. 2011). In the Black River (another Lake Simcoe tributary), the revised model structure was able to simulate main stem P dynamics reasonably well, but monitoring data were not available to support detailed assessment of P dynamics and model performance across the study catchment (Whitehead et al. 2011).

The first phase of modelling work in this study was the representation of hydrological conditions to support simulation of biogeochemical processes within INCA. Variables required by INCA include hydrologically effective rainfall and soil moisture deficit (SMD). Hydrologically effective rainfall is the net precipitation (precipitation in excess of evapotranspiration) available for runoff or recharge; SMD is the difference between soil moisture capacity and actual soil moisture. We simulated these parameters using a version of the model HBV (Lindstrom et al. 1997, Menzel and Burger 2002) developed in structural thinking experimental learning laboratory with animation (STELLA) modelling software, a version used successfully in the Beaver River and other Lake Simcoe catchments (Oni et al. 2011, 2012). Model simulations were performed for the period September 2006 to October 2011. HBV was calibrated using flow and weather data (weather data from a nearby monitoring site; see Jin et al. 2013; Fig. 1). Flow data for the duration of the modelling period were available from a single site in the lower catchment (Environment Canada gauge 02EC011, near BVR05), supplemented by 2 field seasons of data from the 8 additional sites previously noted.

Land use within the catchment (noted previously) was grouped into 5 types of land use, and a reach structure was defined (Fig. 1) to account for variation in terrain, land use, and locations of water quality and flow monitoring sites, and point source inputs. A large amount of information is required to set-up INCA-P (see details in Wade et al. 2002, 2009). Briefly, soil P levels on agricultural lands were estimated based on P budget data provided by Agriculture and Agrifood Canada using data associated with the Indicator of Risk of Water Pollution from P (IROWC_P; Eric van Bochove and Jean-Thomas Denault, Agriculture and Agri-Food Canada, June 2012, pers. comm.). This method uses data collected in the Census of Agriculture, environmental management surveys, and databases of climatic and hydrological variables to assess the risk of water pollution from both particulate and dissolved P. Inputs to land, regional P balances, and estimates of water soluble P and soil test P are calculated (see detailed methods in van Bochove et al. 2006, 2011). Data used in these analyses were estimated from data for soil landscapes of Canada polygons based on the proportion of different soil polygons within the study catchment.

These catchment P data derived from IROWC_P incorporate several sources of error, including survey error (e.g., IROWC_P data rely upon statistical data from the Census of Agriculture), error associated with method assumptions (e.g., regarding crop P requirements, animal P production), and spatial error (associated with different scales used in the Census of Agriculture, Soil Landscapes of Canada, and the study catchment). We used IROWC_P estimates of water soluble P to estimate the degree of P saturation, then estimated equilibrium P concentrations (EPC_0) using equations of Pothig et al. (2010). The resulting estimate of EPC_0 was 0.01 mg P L^{-1}.

We initially modelled P inputs to the catchment based on IROWC_P values (the most recent data available were 2006) and recommended fertilization rates based on crops within the catchment (OMAFRA 2012a). Based on initial simulations where the model significantly under-predicted P concentrations, we increased fertilization rates to intensive agriculture to 48 kg P ha^{-1} y^{-1} (more than double our estimation of recommended rates) while maintaining applications to nonintensive agricultural lands at 9.1 kg P ha^{-1} y^{-1} (estimated for forage crops; OMAFRA 2012b).

We assessed recent changes in the catchment P balance using the time series data available (based on 5-year intervals from 1981–2006, corresponding with Census of Agriculture data) from IROWC_P. We performed linear regression on manure P inputs, fertilizer P inputs, the P balance, estimated soil test P, and water soluble versus time (using IROWC_P data for the catchment). Data were transformed (log or exponential transformations) as required to conform to assumptions of the analysis. All statistical analyses were performed in SYSTAT 13.

In the model, we assumed all fertilizer application to intensive agricultural lands occurred over a 60 d period starting in late April. Although periods of fertilizer application may vary widely with crop and field conditions, this period is consistent with recommended practices (OMAFRA 2012c–e) for several crops. Inputs to nonintensive agriculture were assumed to occur over a 120 d period from late April to late August. We assumed the land was not irrigated (<0.5% of the catchment area is irrigated; Statistics Canada 2008).

The population of the catchment is estimated at ~11 000, the majority of whom (8000) are served by private septic systems. We estimated P inputs to septic beds based on rates of P excretion and detergent release and then estimated P loads to the catchment based on septic efficiency (a metric of the proportion of P retained in septic beds; Stephens 2007, Whitehead et al. 2011). This P was added to P input to nonintensive agricultural lands (0.5 kg P ha^{-1} y^{-1}). There are 3 sewage treatment works (STWs) in the Beaver River catchment, 2 of which discharge into the Beaver River. The third STW (Beaverton) discharges to Lake Simcoe. The Cannington plant (near BVR10) discharges 96–145 kg P y^{-1} (2006–2008 data; mean effluent concentrations of 0.27–0.30 mg P L^{-1}; XCG Consultants Ltd. 2010). Discharges from the Sunderland plant (near BVR16) are lower at 8–34 kg P y^{-1} (mean concentrations of 0.11–0.2 mg P L^{-1}; 2006–2008 data; XCG Consultants Ltd. 2010). Discharges from Sunderland and Cannington were input into the model as a constant value set at the mean reported discharge for each plant over the 2006–2008 period (XCG Consultants Ltd. 2010). Discharges from Beaverton were excluded from this catchment modelling exercise. Data sources and influential parameters used in modelling are summarized in Table 2.

Management scenarios

As a part of efforts to guide P reduction strategies, a series of agricultural best management practices (BMP) were identified by the Ontario Ministry of Agriculture Food and Rural Affairs, Ontario Ministry of the Environment, and the Lake Simcoe Region Conservation Authority, and the achievable implementation for these BMPs was estimated for the Beaver River catchment (see Louis Berger Group Inc. 2010 for details of scenario development). For intensive agriculture (row crops), 5 BMPs were identified: cover crops (potential for 5% of area), crop residue management (20%), crop rotation (10%), nutrient management plans (30%), and strip cropping (5%; Louis Berger Inc. 2010). Nutrient management planning was also identified as a BMP option for nonintensive agriculture (hay and pasturelands); however, these effects were not simulated due to the relatively small area on which these plans could be added (~5%). Streamside BMP opportunities include developing vegetated buffer strips (25%) and fencing for cattle (30%; Louis Berger Inc. 2010).

The effects of BMPs remain difficult to predict, and, even in intensively monitored catchments, BMP impacts can be difficult to quantify due to high seasonal and interannual variability in hydrology and nutrient export. To provide preliminary understanding of how BMPs might help achieve P load reductions, we explored scenarios based on these estimates of potential adoption (Louis Berger Group Inc. 2010) and changes to model parameterization designed to simulate the potential range of BMP effects (Table 3). These changes in model parameterization were based on past work (Whitehead et al. 2011) and expert knowledge; however, prediction of BMP effects remains highly uncertain due to relatively sparse research quantifying BMP effectiveness and issues of uniqueness of place affecting the generalization of what research has been performed.

In additional to our assessment of agricultural BMPs, we examined a scenario associated with cessation of sewage loading. This analysis assumed that all other conditions matched the baseline period (Sep 2006–Oct 2011) to which we compared scenario results. All scenarios assumed that land use in the catchment remains unchanged, which is consistent with the longest-term regional projections available (to 2031; Louis Berger Group Inc. 2010). These scenario analyses also allowed a sensitivity assessment of the model to the aforementioned model parameters.

Results

Observed data

The hydrograph at the Environment Canada site 02EC011 (the flow monitoring site with year-round data; near BVR05) followed expected patterns reflecting annual discharge dominated by snowmelt-driven spring runoff, with increases in autumn flow and periodic summer storm events. Other study sites showed similar trends in flow (Fig. 2; note modelled data). The monitoring network did not sample all runoff sources in the study catchment (Fig. 3).

The highest mean monthly TP concentrations were observed at BVR28 (0.046 mg P L^{-1}), BVR11 (0.045 mg P L^{-1}), BVR15 (0.035 mg P L^{-1}), BVR02 (0.035 mg P L^{-1}), and BVR07 (0.032 mg P L^{-1}; Fig. 4; means are for the

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Table 2. Major data sources used in model calibration.

Data	Data description	Data source
Hydrology		
Flow	Daily time series	Environment Canada monitoring (BVR05) Additional monitoring of 9 sites in support of this project
Precipitation, temperature and soil-moisture	Daily time series	Environment Canada weather data. Soil moisture deficit modeled using HBV
Water chemistry		
Stream chemistry	TP, TDP, TSS	Long-term monitoring data for site BVR01 (TP and TSS only). Additional sampling in support of this project (9 sites for turbidity+TSS, 24 for TP, TDP).
Catchment characterization		
Land use	Land use data divided into 5 categories	GIS - Ecological Land Classification of Ontario
Soil grain size	Grain-size distribution is assumed to be uniform across the catchment. Distribution is consistent with clay-loam soils (40% clay, and 20% each of mine, medium and coarse sand).	
Soil EPC_0	Water soluble phosphorus estimates were derived from IROWC-P data (adjusted to catchment area from soil landscapes of Canada polygons, and assuming 2006 conditions). These data were used to estimate equilibrium phosphorus concentrations using published equations (Pothig et al. 2010).	IROWC-P data, calculated EPC_0.
Phosphorus inputs		
Nonintensive agricultural lands	Manure application to nonintensive agricultural lands at rates consistent with moderate soil-test P results for forage crops. Septic inputs are added to these values.	(OMAFRA 2012a)
Septic inputs (nonintensive agricultural lands)	We assumed P excretion rates of 1 g per capita d^{-1} and adjusted values for detergent use, assuming 57% efficiency of septic tanks in P removal (Stephens 2007). These inputs are added to phosphorus inputs on nonintensive agricultural lands.	Population served by septics from GIS data (Stephens 2007)
Intensive agricultural lands	Initially calculated inputs to the land to obtain high response from fertilizer addition, based on estimate soil test phosphorus concentration. Inputs were then increased above this level in the model calibration process.	(OMAFRA 2012a)
All land	Atmospheric deposition rates were assumed constant across all land uses, with estimates based on monitoring data.	Nearest monitoring station (Ramkellawan et al. 2009)
Sewage inputs to the stream	Sewage inflows from Cannington and Sunderland STWs modelled based on annual loads, mean concentrations for 2006-2008. Beaverton STW is not included in catchment model (it is considered a direct-to-lake effluent).	(XCG Consultants Ltd. 2010)

Table 3. BMP scenarios. Description of BMP adoption potential; watershed BMPs are based on Louis Berger Group Inc. (2010). Scenario IDs are abbreviated as: intensive agriculture (IA), nonintensive agriculture (NIA), and sewage treatment works (STW). Representation of BMPs within the model are based on approaches of Whitehead et al. (2011), and expert knowledge.

Scenario ID	Description	Local BMP potential	Scenario modelling rationale	Representation in model
IA1	Nutrient management on intensive agricultural lands	30% in area with nutrient management plans	Decrease fertilizer application rates to bring catchment to close to a P balance	P applications to intensive agriculture are reduced by 20%
IA2			Cease fertilizer applications to help reverse trend of increasing soil P, and mine the residual P available in the soils. This is not likely to be politically, agronomically, or economically feasible over time.	Cease: P applications to intensive agriculture are 0 kg fertilizer/y
Erosion1	Erosion management on land	See text for % increases anticipated for cover crops, crop residue management, crop rotation, strip cropping.	These practices minimize sediment transport and filter direct runoff. Exact effects depend strongly on local conditions.	Assumes a 30% reduction in the transport capacity parameter of INCA-P to represent combined effects of these BMPs (based on the approach of Whitehead et al. 2011).
Erosion2	Streamside BMP opportunities	BMP opportunities include: developing vegetated buffer strips and fencing for cattle (see text for % changes anticipated)	These BMPs have the potential to reduce soil erosion and disturbance of the streambed, filter direct runoff and reduce livestock excretion directly into the stream. These effects are highly uncertain. Only sediment-related changes are represented here.	Assumes a 10% reduction in the transport capacity parameter of INCA-P across nonintensive agricultural land to represent combined effects of these BMPs (based on the approach of Whitehead et al. 2011).
NIA	Nutrient management			Assume 50% reduction in fertilizer application to nonintensive agricultural lands.
STW	Changes in sewage treatment	Upgrade to mechanical treatment with tertiary filtration (maximum effluent concentration objective of 0.10 mg P L^{-1}). Sewage diversions are not under management consideration	Assess effects of sewage diversion to another watershed to understand maximum reductions.	Removed point-source inputs to model.
Best case	Best case scenario of BMPs and point-source load reductions			IA1+Erosion1+NIA+STW

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period Jan 2010–Dec 2011, which is the period of most intense monitoring, although some sites lack data for 1–2 month periods in this span). Of 968 samples analyzed between 2010 and 2011, we measured 224 samples across the catchment that exceeded TP provisional water quality thresholds (0.026 mg L^{-1}; Chambers et al. 2011). The average TP concentration for this period across sites was 0.023 mg L^{-1}, and at several sites, elevated TP concentrations were observed in summer months (Fig. 5; e.g., BVR02), while short-term maxima were often seen in winter (Fig. 5; e.g., BVR27, BVR19).

The sites with highest mean monthly TDP concentrations were many of the same sites that showed elevated TP (BVR15, 0.029 mg P L^{-1}; BVR02, 0.024 mg P L^{-1}; BVR07, 0.021 mg P L^{-1}; and BVR04, 0.017 mg P L^{-1}; Fig. 4). The average monthly TDP concentration for this period (across sites) was 0.012 mg P L^{-1}. Looking at the evolution of TP concentrations through the catchment, the highest values were often observed in upper reaches and small inflows, consistent with patterns for TDP.

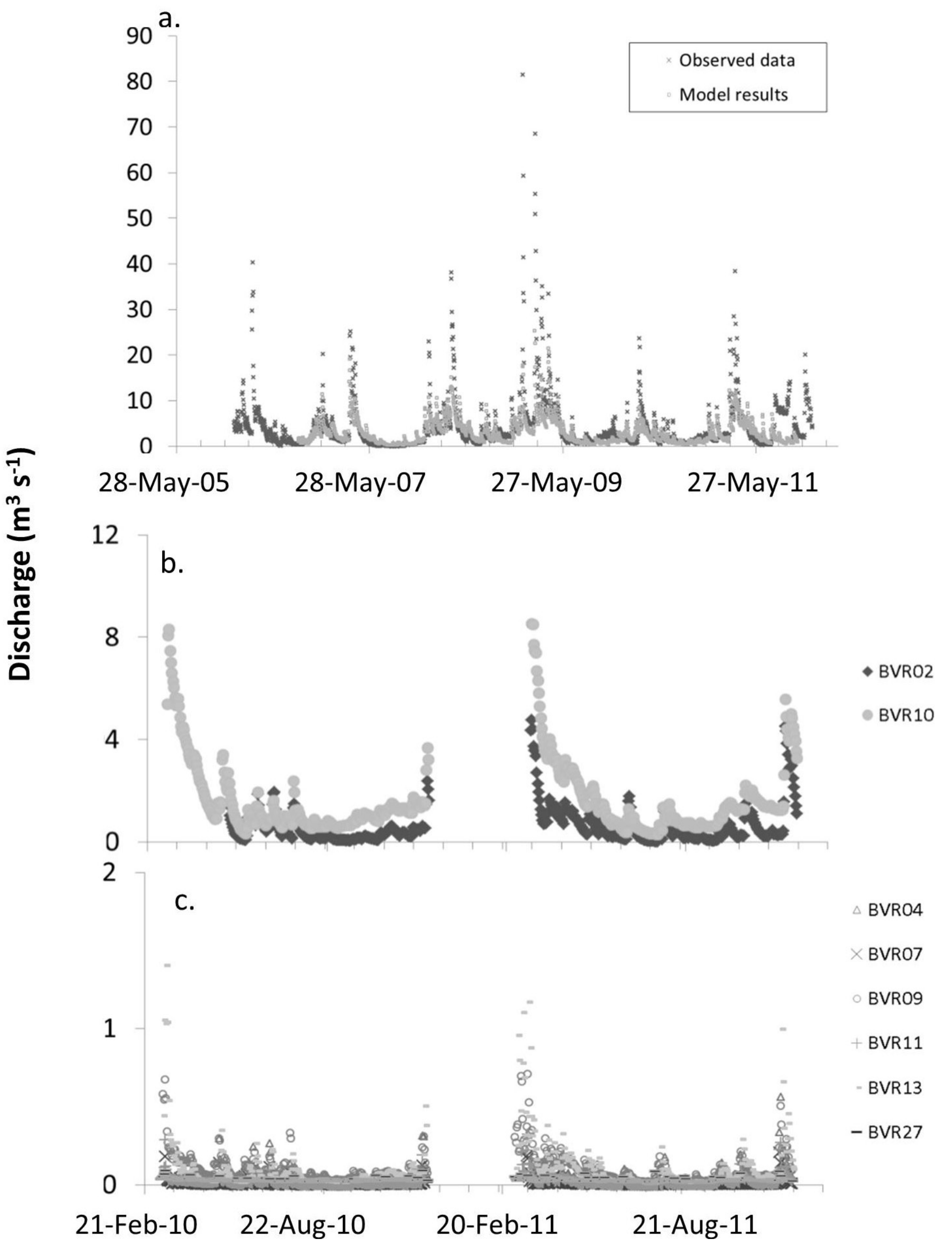

Fig. 2. (a) Modelled and measured discharge at the long-term Environment Canada monitoring site 02EC011 (labeled as BVR05; the nearest sampling site in this study); (b and c) discharge at all other monitoring sites (note: different scales). Discharge was not measured at these sites during winter months.

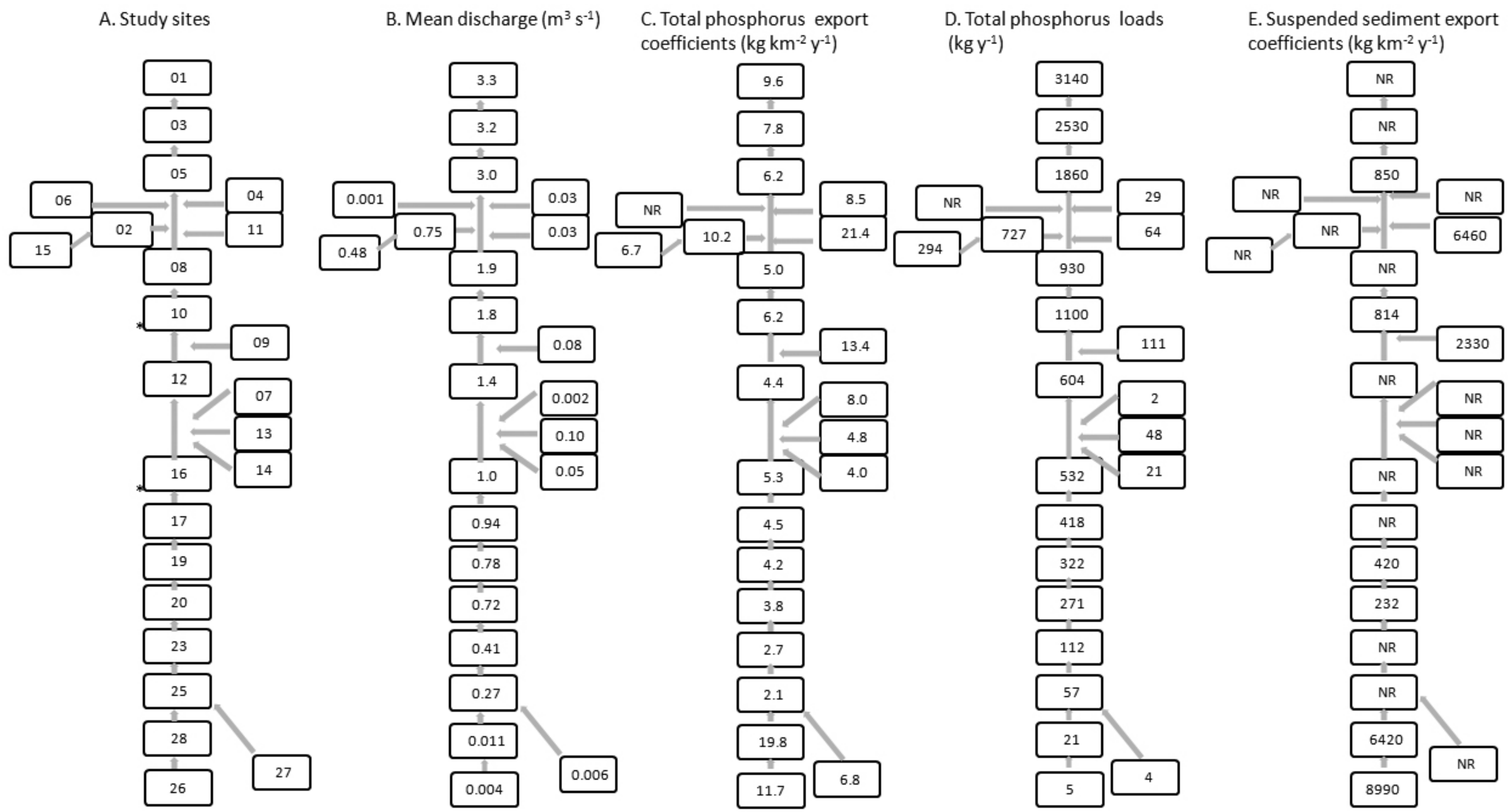

Fig. 3. Stream monitoring network diagram showing (a) study site codes, (b) mean modelled discharge from 1 September 2006 to 31 August 2011, (c) mean TP export coefficients (using monthly average measured P data and monthly average modelled discharge from April 2010 to March 2011), (d) loads across the catchment (calculated using same data as Fig. 3c); and (e) mean modelled TSS export coefficients from 1 September 2006 to 31 August 2011. Network is not to scale, and not all inflows were monitored or modelled. The generalized direction of catchment drainage is indicated by arrows (generally from south to north). NR indicates data are not reported, due to missing monthly concentration measurements or unmonitored sites. Site numbers are indicated in the text according to number, with the prefix BVR.

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TSS was less intensively monitored, but existing data showed high variation in mean concentrations across the catchment. Mean monthly TSS concentrations (calculated from turbidity) for 9 of the study sites showed elevated TSS at several sites at or near the headwaters (BVR11, BVR26, and BVR28; with mean monthly concentrations of 17.3, 18.8, and 18.9 mg L^{-1}, respectively) and at the outflow (BVR01, 11.2 mg L^{-1}; Fig. 4), a site subject to frequent disturbance associated with development, and boat activity. The average monthly TSS concentration across these 9 sites for the period was 8.8 mg L^{-1}.

Trends in catchment P inputs

Calculated P inputs (both manure P and fertilizer P) declined over time (r^2 = 0.77 and 0.70, respectively) during the period that census data were available (1981–2006). Despite this, the P balance remained positive (P inputs exceeding uptake) throughout the data record, and as a result, the cumulative P balance continued to grow; however, the magnitude of P imbalance for a given year declined over time (r^2 = 0.75). Both the estimated soil test P and estimated water extractable P increased through time (r^2 = 0.97 for both parameters).

Export coefficients

We calculated TP and TSS export coefficients for each intensively monitored study site. TSS export coefficients increased markedly in the most downstream reaches (downstream of BVR08) in parallel with increases in flow (Fig. 3).

Phosphorus export coefficients were extremely variable, ranging from 2.1 to 21.4 kg km^{-2} y^{-1} (Fig. 3). The average TP export coefficient across sites was 7.7 kg km^{-2} y^{-1}. TP loads increased from upstream to downstream (with the exception of a small decrease at BVR08) and showed increases in lower reaches. Using best subset

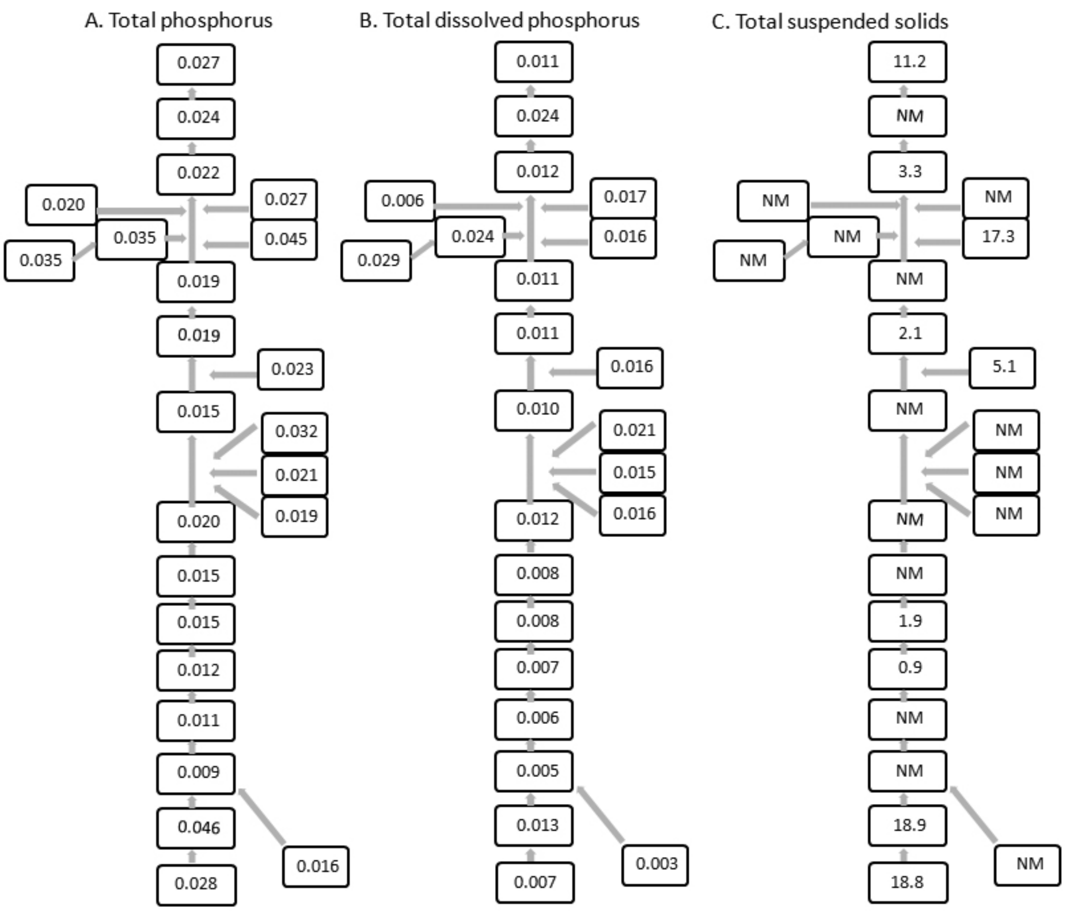

Fig. 4. Stream monitoring network diagram showing mean monthly measured (a) total phosphorus concentrations (mg L^{-1}), (b) total dissolved phosphorus concentrations (mg L^{-1}), and (c) total suspended solids concentrations (mg L^{-1}). Data are for the period of most frequent monitoring activity (Jan 2010–Dec 2011), although data are not available for all months and sites. Network is not to scale, and not all inflows are shown. Sites where no monitoring data are available are indicated with NM. The generalized direction of catchment drainage is indicated by arrows and is generally from south to north. Study site codes shown in Fig. 3.

regression, we identified 2 candidate models for prediction of export coefficient using land use data. The best model using adjusted r^2, AIC, and Schwarz's BIC included wetland, intensive agriculture, and nonintensive agriculture, while the best model using AIC (corrected) included % wetland area alone. Univariate correlations revealed fairly weak relationships between these variables and P export coefficient (export coefficient: wetland: $r = -0.64$; intensive agriculture: $r = 0.35$; nonintensive agriculture $r = 0.18$), but the combined model had an adjusted r^2 value of 0.64 ($p = 0.046$); F ratio 5.7, mean square $(MS)_{regression}$ 57, 3 degrees of freedom (df), $MS_{residual}$ 10, 5 df. The adjusted r^2 for the regression using only % wetland area was 0.36, $p = 0.052$ ($MS_{regression} = 96.5$, 1 df; $MS_{residual}$ 17.8, 7 df). Using data at BVR01, the most downstream sampling site, our results indicated a high degree of interannual variation in export coefficients, ranging almost 3-fold at the BVR01 over the period September 2006 to August 2011. The mean export coefficient during this period was 9.7 kg km^{-2} y^{-1}.

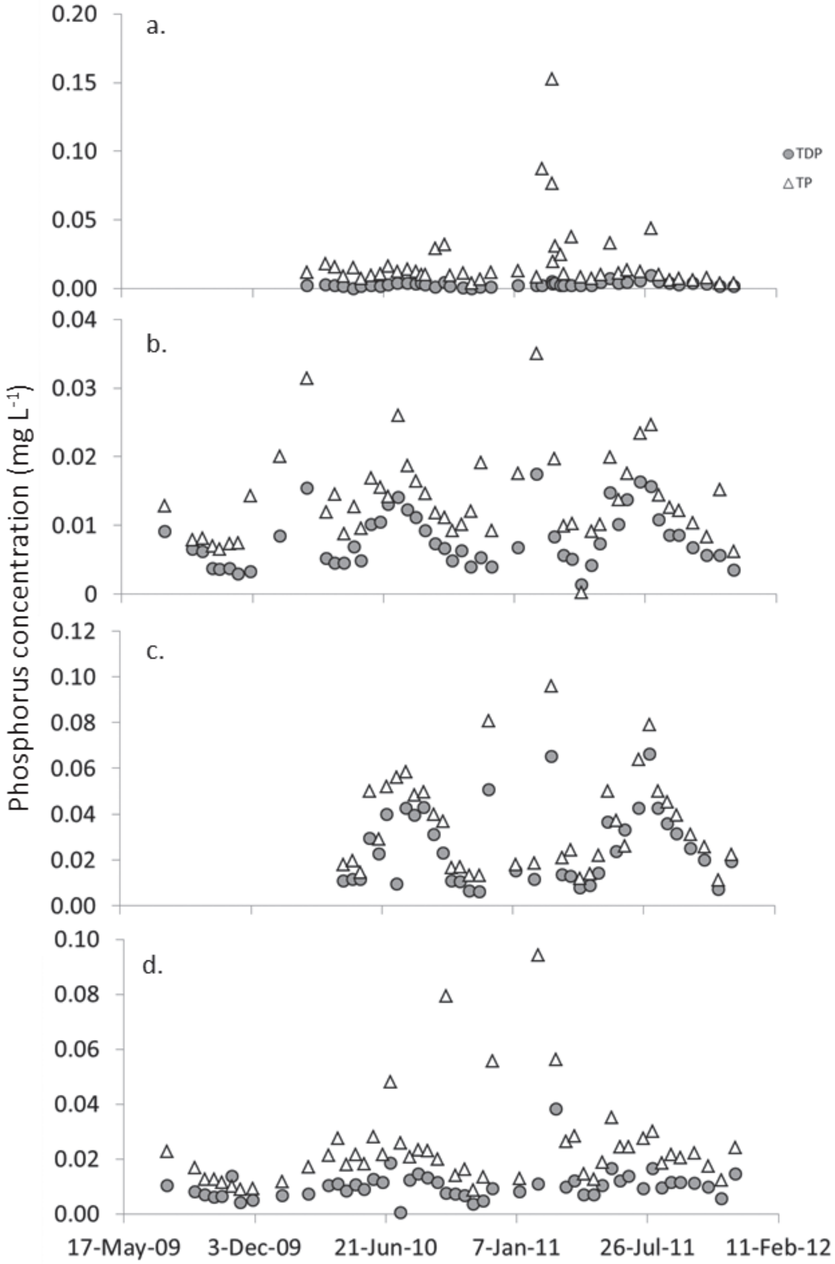

Fig. 5. Seasonal cycles of total phosphorus and total dissolved phosphorus at 4 monitoring sites: Panel (a) BVR27; (b) BVR19; (c) BVR02; and (d) BVR01. All data are measurements. Figure legend for all panels is shown in panel (a).

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Table 4. Effects of scenarios on TP concentrations (mean concentration over simulation period) compared to baseline scenarios for the modelling period September 2006–October 2011. Average changes across sites (%), and the maximum reduction (%) are reported. Negative signs indicate a reduction relative to baseline conditions.

	Concentration			Volume-weighted concentration		
	Average change	Maximum change	Effect at BVR01	Average change	Maximum change	Effect at BVR01
IA1	−0.5	−1.4	−0.4	−0.6	−1.4	−0.6
IA2	−2.6	−6.4	−2.2	−2.7	−6.9	−2.5
Erosion1	−0.4	−2.9	−0.9	−0.4	−2.7	−1.3
Erosion2	−0.1	−1.1	−0.4	−0.2	−1.1	−0.5
NIA	−1.1	−2.1	−0.9	−0.9	−2.0	−0.9
STW	−1.3	−11.5	−3.9	−0.9	−8.3	−3.2
Best case	−3.0	−12.7	−5.7	−2.6	−9.8	−5.2

Modelling results

Although the INCA-P model was able to simulate general patterns in hydrology, model fit was not strong. We calculated the mean r^2 between modelled and measured data (over time, for 9 sites with flow data). The best fit was $r^2 = 0.63$ for the site BVR10, while across all sites the mean r^2 was 0.41. Model simulations represented water quality at some sites well, although on the whole, agreement between model results and observed data was relatively weak. On average, across all sites, the model captured 14% of variation in measured TSS concentrations, with a maximum r^2 of 0.65. TDP and TP simulations captured <10% of the daily variability in the observed data for these parameters (across all sites). Maximum r^2 observed were 0.27 for TDP and 0.24 for TP. The difference between measured and modeled values at BVR01 (across 205 samples from 2006 to 2011, averaging both overestimates and underestimates) was 0.004 mg L^{-1}. The absolute value of this difference was much higher at 0.018 mg L^{-1}. These differences may reflect lake effects on this most downstream sampling location.

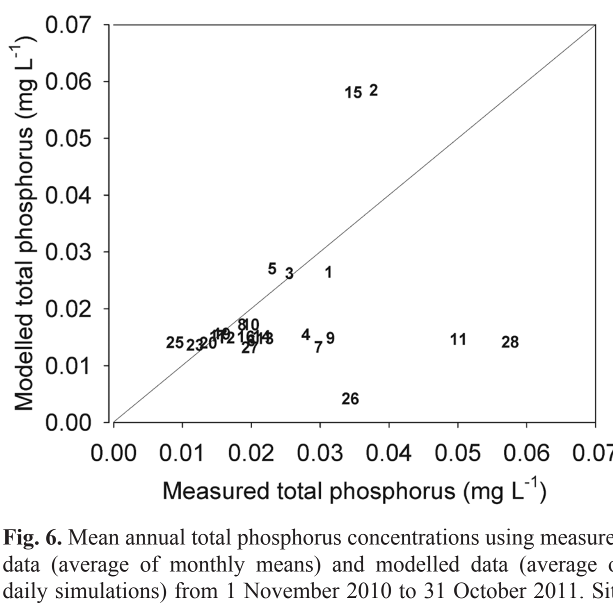

Fig. 6. Mean annual total phosphorus concentrations using measured data (average of monthly means) and modelled data (average of daily simulations) from 1 November 2010 to 31 October 2011. Site numbers are shown in the plot, excluding the prefix BVR.

Despite relatively poor model statistics, the model represented mean P conditions well at some sites (Fig. 6), although maxima were often not simulated (e.g., Fig. 7). This was a common issue when elevated values were not associated with a hydrological event, making the driver of these changes unclear. The model tended to under-predict stream TP concentrations (Fig. 6), despite the elevated P loading rate to intensive agriculture used in the model. Downstream sites showed relatively good fit between mean modeled and measured conditions (BVR01, BVR03, and BVR05; Fig. 6). The sites BVR 11, BVR26, and BVR28 had much higher P concentrations than predicted in the model, raising questions about the potential for hotspots of P transfer that may be associated with riparian land use, or other agronomic factors.

Average observed data were not directly comparable to mean simulated data because monitoring data may disproportionately reflect event or low-flow sampling conditions, and monitoring did not occur across all sites for the full model time period. Modelled TP at some sites was similar to the observed data (Fig. 6). The average model TDP concentrations exceeded measured concentrations at most sites, suggesting that particulate P export may be underestimated, an issue that may reflect difficulty simulating peak flows (Fig. 2a). The average simulated P concentrations (over all sites, and simulation dates from 1 Sep 2006 to 2 Nov 2011) were 0.020 mg P L^{-1} for TP and 0.019 mg P L^{-1} of TDP.

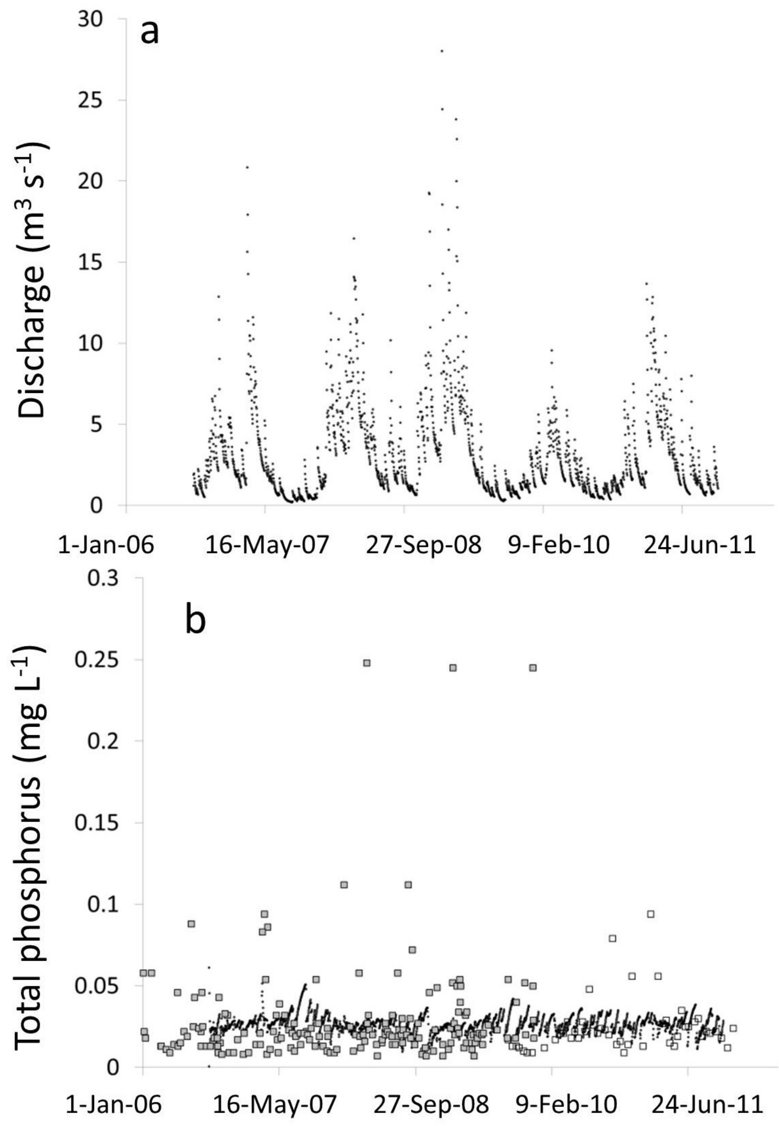

Fig. 7. Site BVR01 (a) simulated discharge and (b) simulated total phosphorus (TP) concentrations (small dots) and measured TP concentrations. Long-term monitoring data are shown in grey squares, while our recent sampling program is shown in open squares.

Maximum TP concentrations were simulated at 0.14 mg P L^{-1}, while minima were <0.001 mg P L^{-1}. The mean simulated TSS concentration was low (1.4 mg L^{-1}), although peaks as high as 9.2 mg L^{-1} were simulated, associated with high runoff. Minimum TSS concentrations, like TP concentrations, were extremely low (0.029 mg L^{-1}). Over the model simulation period from 1 September 2006 to 2 November 2011 across the 30 sites simulated (>56 000 days × sites), exceedances of provisional water quality objectives (0.026 mg TP L^{-1}; Chambers et al. 2011) were simulated on 9220 sites and dates (equating to 16% of simulations). This frequency of exceedance is lower than the 23% from observed data in the monitoring data (from recent years).

Management scenarios

Scenarios had a modest effect on P concentrations relative to the baseline scenario. The best case scenario achieved a 3.0% reduction on average across sites, with a maximal reduction of 12.7% (Table 4; unweighted concentrations). The IA2 scenario contributed to a 2.6% reduction in con-

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centrations across sites, while the STW scenario led to only a 1.3% reduction in concentrations across sites. Notably, however, at the outflow to the lake (BVR01) this reduction was 3.9%, while the maximum reduction scenario led to a reduction of 5.7% at this site. Effects of erosion control were small (average 1.1% reduction for scenario Erosion2, and 2.9% reduction for Erosion1; Table 4). Flow-weighted concentration changes at BVR01 were less than unweighted concentration changes at BVR01 for the best case scenario, while individual scenarios varied in their relative effects using flow-weighted and raw concentration data.

Discussion

Observed data and export coefficients

Although the long-term operation of spatially intensive monitoring networks is expensive, we suggest that at a minimum, some targeted monitoring across subcatchments is useful in modelling exercises such as this to help characterize uncertainty in spatial representation of nutrient dynamics. In addition, this spatially distributed monitoring and modelling is useful in assessing the frequency of water quality exceedances across a stream network, from headwaters to the outflow, because monitoring sites may not be representative of ecological conditions across a catchment. Our results clearly indicate that the stream ecology will be sensitive to current conditions because interim water quality thresholds (Chambers et al. 2011) are frequently exceeded in many reaches of the stream network, and high P concentrations often occur in summer, a period of maximum ecological sensitivity (due to lower flow, low oxygen solubility, and high water temperatures associated with maximal microbial activity). Peaks observed during winter months may reflect melt events or, at stable flow, may reflect low biotic demand for P, coupled with the potential for groundwater and sediment P inputs to strongly affect P concentrations when flow is low.

Although process-based biogeochemical models are being widely adopted, export coefficients are still frequently used in characterizing landscape nutrient export and understanding effects of land use on P export. Consistent with past studies across the Lake Simcoe basin (Winter et al. 2002), our results indicate a high degree of temporal variation in export coefficients, ranging almost 3-fold at the BVR01 site over the period September 2006–August 2011. The mean export coefficient during this period was lower than the mean value of 12.4 kg km^{-2} y^{-1} reported for this catchment during the period 1990–1998 (Winter et al. 2002).

Our results show several clear hotspots of P export, most notably, at BVR28 and BVR11. These subcatchments represent agriculturally dominated landscapes, and as such, export coefficients are not atypical and may even be considered low relative to the same dominant crops in other regions (Beaulac and Reckhow 1982). Interestingly, the minimum export coefficient was reported for a subcatchment (BVR25) that also has significant agricultural activity (Table 1), which could reflect the use of good nutrient management practices, or alternatively, could reflect limited connectivity between fields and the stream. Across the monitored tributaries of Lake Simcoe, export coefficients show much greater variability in space, ranging from 5.8 kg km^{-2} y^{-1} to >109 kg km^{-2} y^{-1} when hotspots of P export such as cultivated polders are considered (Winter et al. 2002). Among the Beaver River subcatchments, wetland area was the strongest predictor of variation in TP export coefficients, which may reflect a role of wetlands in P retention (Reddy et al. 1999) or a decrease in agricultural land use in wetland-dominated catchments (e.g., correlations for % wetland to % intensive agriculture $r = -0.63$, $p = 0.07$). Sewage loads are a small proportion of P export. In the lower reaches of the river (BVR05, BVR03, and BVR01), large increases in export coefficients and P loads were observed despite small increases in flow, suggesting this may be a high priority area for remediation work or study of the potential for P release from sediments.

Modelling results

This study is the first detailed application of the branched model of INCA-P to a highly distributed monitoring network. Our results suggest that this approach can help identify critical unknown parameters in space and identify potential hotspots of nutrient transport and unusual hydrology within the catchment. For example, high P concentrations in summer months are often representative of point source inputs (Bowes et al. 2008). In these catchments, sewage releases do not explain seasonality because sewage inputs are small and episodic (typically spring and autumn releases from lagoons); however, high summer P concentrations observed at several sites can also be indicative of sediment P release (simulated here using sediment water P-exchange rates, and equilibrium P concentrations) or may reflect high inputs of P from groundwater and septic plumes (Brett et al. 2005, Arnscheidt et al. 2007).

Significant uncertainty in our model simulations is associated with the rates of P fertilization. To simulate observed in-stream P concentrations, we increased fertilization rates considerably above those derived from recommended practices and IROWC_P estimates. As a result, fertilization rates that we used would be consistent with application rates in excess of recommendations; however, these elevated numbers may also be necessary to reflect disproportionate impacts of agriculture in riparian areas (e.g., Miles 2012). They may reflect a mismatch between scales associated with IROWC_P data and our much smaller study subcatchments, and likely reflect a limitation of the modeling approach, where homogenous fertilization rates and timing are assumed within a land use category, when farmer practices may vary widely. Scenario analyses can help provide insight into model sensitivity. Changing phosphorus inputs to intensive agricultural lands (Table 4) did affect P concentrations in our model scenarios. Sensitivity to this parameter varied across the watershed, but changes were relatively small over the short-term period assessed in these scenarios.

Model calibration was performed using best available catchment information, and parallel (though less spatially intensive) use of this model has resulted in much better model fit (Whitehead et al. 2011). This suggests either that the model is unable to capture some dynamics specific to this study catchment, and may be related to the importance of wetlands throughout much of the catchment, or that current understanding of land use practices and other factors in the catchment is inadequate. Although spatially intensive sampling has helped highlight limitations in our understanding of catchment P dynamics, our results suggest this type of sampling may not be necessary for model calibration.

Our modelling results did not simulate the importance of tile drains, which have been estimated to contribute 12.5% of P to runoff in this catchment (LSRCA 2012) and are likely to have significant impacts upon hydrology. Work is currently underway to better understand sediment P dynamics in this catchment and assess the potential for internal P regeneration, and further work is required to assess oxygen dynamics in the river. A parallel study on the Beaver River (Miles et al. 2013) has reported that strong relationships exist between P concentrations and the proportion of nonintensive agriculture in a 50 m buffer zone adjacent the stream. This suggests that even our spatially distributed model structure may not be adequate to capture near-stream source dynamics and indicates that monitoring work in near-stream agricultural areas may be merited.

Although modelling fine-scale processes may be seen as a requisite step to better parameterization of larger scale models, it may also be that model errors tend to be dampened out at larger spatial scales. As a result, this type of detailed process-based model may not be suited to spatially intensive use. Alternatively, further spatial disaggregation in our designation of subcatchments may be required to better represent very fine scale spatial variation reported by Miles et al. (2013) and to address issues associated with hotspots or critical source areas of nutrients in the catchment (Pionke et al. 2000).

In several subwatersheds, the observed hydrology was a relatively poor fit to modelled data. This catchment has numerous hydrological alterations characteristic of many human-dominated landscapes, including a large number of constructed ponds in the landscape, some of which are located in near-stream areas (e.g., BVR28). Perhaps more important, beaver activity is known to have altered the hydrological regime, and direct effects have been observed (e.g. BVR15). This beaver activity is also likely to impact the nutrient chemistry; however, the type of impact seems to depend on characteristics of the beaver dams constructed (Devito and Dillon 1993, Klotz 1998, Correll et al. 2000, Fuller and Peckarsky 2011). In other study sites (e.g., BVR27), it was not possible to isolate the driver of unusual hydrology, but upstream culverts, the potential for beaver activity, and field-based observations of high flow events that were not associated with rainfall (suggesting discharge of an upstream reservoir, potentially including a beaver pond or some type of on-farm water storage) may have influenced our ability to model catchment hydrology.

Lack of fit between modelled and observed data can help to identify shortcomings in our conceptual understanding of catchment biogeochemical processes (Ledesma et al. 2012). Using highly distributed monitoring data, it is possible to evaluate the hypotheses about environmental processes incorporated in process-based models. In our study, nutrient dynamics in mixed land use catchments are controlled by a variety of factors. Land cover, such as the balance between wetlands and arable land, is a good predictor of P fluxes. Fertilizer and sewage inputs also strongly influence P dynamics, but the discrepancy between reported P application rates and in-stream P concentrations indicates a need for more scrutiny of fertilizer application rates or the mechanisms responsible for delivery of fertilizer P to surface waters. The poor model fit for summer TP concentrations in some reaches is a cause for concern because it suggests that there is either an unmonitored source or another driving process, such as redox-related P release from stream sediments or significant groundwater P inputs.

Hydrological simulations provide the basis for simulation of nutrient chemistry (Wade et al. 2004) and, as a result, are a major limitation in our ability to simulate nutrient dynamics on a daily timestep. Although this type of highly distributed modelling and monitoring network can potentially lead to new insights into spatial dynamics of nutrient export, small scale impacts, such as variation in groundwater hydrology, beaver activity, and farm-water

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management, have a major impact on model representations at the spatial scale used. At larger scales, these impacts and other effects, such as cattle access to streams and large livestock operations near waterways, may tend to dissipate as water chemistry influences are aggregated in space and time. As a result, although modelling at small spatial scales may be valuable in assessing potential hotspots of nutrient loading in a catchment, it is unlikely that model fit will be as strong as fits observed in larger catchments, and in particular, in catchments with much higher nutrient concentrations, where impacts of sewage loading are more pronounced, and nutrient modelling has been more commonly applied.

Management scenarios

While significant progress has been made in reducing point-source P loads in many areas of North America, control of nonpoint P sources has remained a major challenge. In the Lake Simcoe watershed, efforts are underway to control both point- and nonpoint-source loads. Our exploratory scenario results suggest that efforts to control both point and nonpoint sources could have measureable effects on P concentrations, but these effects will be relatively small in contrast to ambitious targets for reducing P loads in the lake and necessary reductions to meet provisional water quality guidelines. Although these scenarios are grounded in detailed consultation and reflect the potential for agricultural BMP adoption within the watershed (Louis Berger Group Inc. 2010), complete cessation of STW inputs is not realistic. There are no current plans to divert sewage, and this scenario would remain unlikely even in the long term due to eutrophication-related stressors across much of southern Ontario. Upgrades to the STW are technically feasible but have high incremental costs of C$6014 to C$8033 kg TP^{-1} based on 25-year life cycle cost estimates (XCG Consultants Ltd. 2010). For context, between 2006 and 2008, the cost of P removal at Cannington ranged between C$4.62 and C$8.21 kg TP^{-1} (XCG Consultants Ltd. 2010).

In addition, scenario IA2 would not support continued high agricultural productivity, and as such is not considered realistic. Our maximum reduction scenario, which embeds a realistic reduction in P loads to intensive agriculture, still likely exceeds best case opportunities for short-term P load reductions because of assumptions regarding the cessation of STW loads. Scenarios reflect the mean change over a 6 year period in comparison to the baseline scenario, however, and greater reductions may be achieved in the longer term.

Load reductions at the Lake Simcoe (Beaverton) STW, the largest STW within the Beaver River catchment, could also contribute to efforts to reduce P loads. Effluent from this plant is released directly to the lake, so it is excluded from our modelling results. The achievable reduction (based on use of additional clarifiers to reduce effluent concentrations to 0.10 mg P L^{-1}) has been estimated as 164 kg P y^{-1}, with considerably lower incremental costs of C$208 kg P^{-1}. Other efforts to model the potential for P load reductions in the catchment using the model CANWET suggest reductions of ~14% are attainable (LSRCA 2012).

Because the quantitative effects of agricultural BMPs are difficult to predict and in many cases specific to conditions within a watershed, modelling has emerged as an important tool in predicting BMP impacts. Modelling future BMP scenarios, however, has significant inherent uncertainty associated with predicting future changes in land use practices, representation of the effects of land use change within the model, and uncertainty associated with the model structure and parameterization. Our scenario results suggest that ambitious efforts to reduce P loading are merited due to difficulty achieving short-term improvements in water quality, even in our most ambitious and optimistic scenario. We emphasize that uncertainty in this assessment is high. In light of concerns regarding model fit, we suggest that watershed managers may wish to aim for maximum feasible P reductions, given uncertainty in the effectiveness of individual P mitigation activities.

Conclusions

One objective of this work was to help constrain P dynamics in this catchment to assist in guiding management actions. Our findings indicate a need for remediation actions to help protect stream ecology, in addition to the need to reduce P loads from all catchments of Lake Simcoe to help protect its valuable coldwater fishery. Efforts to reduce P loads from Lake Simcoe face several challenges. The first is that both point- and nonpoint-source loads will have to be reduced (OMOE 2009). Control of point sources is often expensive, while control of nonpoint source loading is very challenging, and improvements in water quality may be difficult to achieve in the short-term. Despite extensive work to assess the potential for implementation of BMPs in the catchment, a great deal of uncertainty remains regarding the effectiveness of BMPs in reducing nutrient loads, observing timescales for nutrient reductions, and representing potential BMP impacts within modelling exercises. Our analysis of IROWC_P data suggests that further reductions in P application to land may be required to bring soils into P balance, but that legacy of excess P application may not be removed on short timescales.

Finally, our simulations suggest that summertime high P levels may be related to P release from sediments or groundwater inputs, making this a high priority area for further research.

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