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ARTICLE

Models to predict total phosphorus concentrations in coastal embayments of eastern Georgian Bay, Lake Huron

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Abstract: Several coastal embayments of eastern Georgian Bay show signs of water-quality impairment thought to be caused by human activities. Here, we evaluate the ability of the Lakeshore Capacity Model (LCM), developed for Precambrian Shield lakes, to assess the impact of cottage development on the trophic status of ten Georgian Bay embayments. The LCM could only be applied to eight embayments due to the large size and complexity of two watersheds and produced unacceptably high estimates of mean seasonal total phosphorus concentrations ([TP]; i.e., exceeded 20% of measured values for five of eight embayments); accuracy of [TP] estimates could not be improved by accounting for internal phosphorus loading. We developed an additional model, the Anthro-Geomorphic Model (AGM), which uses building density and basin morphometry as variables. Estimates of [TP] for the AGM were within 20% of measured values for all sites. Compared with other aquatic systems, coastal embayments of Georgian Bay have significantly higher chlorophyll *a* concentrations per unit [TP]; we suggest that the TP–chlorophyll relationship presented in this study be used to estimate productivity in these systems.

Résumé : Plusieurs échancrures de la partie est de la baie Georgienne présentent des indices de dégradation de la qualité de l'eau qui serait causée par l'activité humaine. Nous examinons la capacité du modèle d'évaluation de la capacité d'aménagement des rives (« Lakeshore Capacity Model », LCM), mis au point pour les lacs du bouclier précambrien, d'évaluer l'impact de l'aménagement de chalets sur l'état trophique de dix échancrures de la baie Georgienne. Le LCM n'a pu être appliqué qu'à huit échancrures en raison de la grande taille et de la complexité de deux bassins versants, et il a produit des estimations trop élevées, voire inacceptables des concentrations saisonnières moyennes de phosphore total ([PT]) (de 20 % supérieures aux valeurs mesurées pour cinq échancrures sur huit), l'intégration de la charge interne de phosphore ne permettant pas d'accroître l'exactitude de ces estimations. Nous avons développé un autre modèle, le modèle « anthro-géomorphologique » (MAG), qui utilise la densité d'immeubles et la morphométrie du bassin comme variables. Les estimations du [PT] par le MAG ne diffèrent pas de plus de 20 % des valeurs mesurées pour tous les sites. Comparativement à d'autres systèmes aquatiques, les échancrures des côtes de la baie Georgienne présentent des concentrations de chlorophylle *a* par unité de [PT] significativement plus élevées. Nous proposons que la relation entre le PT et la chlorophylle présentée dans l'étude soit utilisée pour estimer la productivité dans ces systèmes. [Traduit par la Rédaction]

Introduction

Georgian Bay, the eastern arm of Lake Huron, is the largest bay in the Laurentian Great Lakes, with a maximum depth of 171 m (McCarthy and McAndrews 2012) and a surface area of 15 111 km^2 (Sly and Munawar 1988), which is almost 80% of the surface area of Lake Ontario (18 960 km^2; Chapra et al. 2012). Georgian Bay is rimmed by the Paleozoic limestone cliffs of the Niagara Escarpment to the west and the granitic rock of the Precambrian Shield to the east. Although both of these features influence water chemistry, Georgian Bay owes its oligotrophic status primarily to the lack of agricultural development in its eastern watersheds (Weiler 1988). As of 2012, the total phosphorus concentration ([TP]) was 4.23 $\mu g \cdot L^{-1}$ and the total nitrogen concentration ([TN]) was 0.327 $mg \cdot L^{-1}$, levels only slightly higher than those of Lake Superior (Dove and Chapra 2015). Maintenance of good water quality is essential for the thriving cottage and tourism industries in this region and benefits the productive recreational fishery that includes a world-class trophy muskellunge (*Esox masquinongy*) fishery (Weller et al. 2016).

Despite the overall oligotrophic status of Georgian Bay, some nearshore areas along the eastern shore are showing signs of water-quality impairment, especially in coastal embayments that are protected and have restricted mixing with the open waters of Georgian Bay. In these areas, there have been reports of recurring episodes of hypolimnetic oxygen depletion and occurrences of nuisance algal blooms (Schiefer et al. 2006; Chiandet and Sherman 2014), which appear to be associated with water-quality impairment normally attributed to cottage development. Given that the economic and ecological viability of this region depends on good water quality, there is an urgent need to develop management tools to evaluate the relative inputs of phosphorus (P) from cottage development versus those from natural sources in coastal embayments.

Even though there is no published model to predict P concentrations in coastal embayments in eastern Georgian Bay, a model has been developed for lakes located in south-central Ontario with similar bedrock. This model is the Lakeshore Capacity Model (LCM; Paterson et al. 2006), a modified version of the mass-balance model of Dillon and Rigler (1975), which has been calibrated specifically for application in inland lakes of the Precambrian Shield. The LCM calculates separate P loading from natural and anthropogenic sources and accounts for the amount of P retained to yield estimates of seasonal mean [TP] (Hutchinson et al. 1991; Paterson et al. 2006), which in turn can predict seasonal mean chlorophyll *a*

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concentrations ([CHL]), an indicator of the trophic state of lakes (Chow-Fraser et al. 1994).

There are good reasons for expecting the LCM developed for south-central Ontario lakes to apply to coastal embayments, since both types of ecosystems have recreational building development along the shoreline (e.g., cottages and marinas), as well as thin soil structures that allow aging septic infrastructure to leach out P (Dillon et al. 1994; Dillon and Molot 1996; Robertson et al. 1998). But there are also several reasons why the LCM may not be applicable. First, embayments are frequently shallower than lakes in south-central Ontario; the Lakeshore Capacity Assessment Handbook specifically warns against applying the LCM to shallow (Z_{mean} < 5 m; Ministries of Ontario 2010) water bodies. More importantly, water in coastal embayments can become mixed with the oligotrophic waters of Georgian Bay, whereas water in inland lakes does not mix with downstream waters. Hence, without a proper study, there is no scientific basis for assuming that the LCM can be used to accurately predict [TP] in coastal embayments.

An additional piece of information that is important for lake managers to effectively evaluate lake productivity is the unique TP–CHL relationship in the water body being evaluated. Without an understanding of this system-specific relationship, lake managers run the risk of incorrectly estimating the productivity (i.e., [CHL]) of a lake in response to changes in [TP]. An understanding of this relationship is therefore important to estimate changes in lake productivity accurately as a result of alterations in natural or anthropogenic P loading to a particular system. TP–CHL relationships for inland lakes are relatively well documented (Dillon and Rigler 1974; Molot and Dillon 1991; Havens and Nurnberg 2004); however, this relationship is not fully understood in coastal embayments of eastern Georgian Bay. This information is needed in order for lake managers to estimate changes in productivity and evaluate restoration efforts.

One of the primary goals of this study was to validate the general applicability of the LCM for coastal embayments of eastern Georgian Bay. To achieve this, we applied the LCM (with and without internal P load) to embayments that vary with respect to basin morphometry and residential development along the coastline. A second objective was to develop an alternative management model that accounts for the hydrological connection between embayments and Georgian Bay, since the degree to which water is diluted by oligotrophic Georgian Bay water would affect the nutrient status of the embayment in question. Lastly, we wanted to compare log-linear relationships between [TP] and [CHL] for a large number of bays and lakes in Canada to determine if there are significant differences between sites occurring inland and those along the coast of large lakes. Results of this study will identify the most appropriate management models to ensure that cottage development in eastern and northern Georgian Bay will proceed without jeopardizing the excellent quality of the water that is vital for maintaining the economy and ecology of the region.

Methods

Site descriptions

Georgian Bay is located within the Great Lakes – St. Lawrence Forest Section (Zone L.4d) in Canada that extends from the Paleozoic sedimentary rock of the Bruce Peninsula in the south to the Sudbury–North Bay Forest zone in the north (Rowe 1972). This region hosts a variety of mesic tree species, including sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*), yellow birch (*Betula alleghaniensis*), eastern hemlock (*Tsuga canadensis*), eastern white pine (*Pinus strobus*), northern red oak (*Quercus rubra*), and white ash (*Fraxinus americana*) (Rowe 1972). The forested landscape of inland Georgian Bay has remained relatively intact for the past century due to low population density, minimal industrial development, and thin acidic soil conditions that are not suitable for intensive agriculture (Weiler 1988).

The dominant geological feature in eastern Georgian Bay is the Precambrian Shield, which consists primarily of granitic bedrock that is less erodible than the limestone bedrock of the western shore. This granitic bedrock has been shaped through past glaciations (McCarthy and McAndrews 2012) and differential weathering processes to create a variable coastline with many finger-like indentations that terminate in wetlands and embayments. These embayments are unique coastal systems that are often only connected to Georgian Bay through narrow channels.

The ten coastal embayments included in this study (Fig. 1) vary on a gradient of basin morphometry (Table 1; Fig. 2), landscape characteristics, and human development along the shoreline (Table 2). Except for Woods Bay, which is too shallow to develop thermal stratification, all other sites are dimictic, becoming thermally stratified during the summer (i.e., Tadenac Bay, South Bay, North Bay, Coganshene Lake, Twelve Mile Bay, Deep Bay, Musquash Bay, Sturgeon Bay, and Longuissa Bay). Musquash Bay and Woods Bay are two riverine embayments positioned at separate outflows of the Muskoka River, which drains into Georgian Bay. The remaining seven embayments in our study are lacustrine, with small to moderate-sized drainage basins, and have only small streams discharging into them. Since the LCM was developed for inland lakes, we included a relatively deep (Z_{max} = 60.80 m; Table 1) inland lake that we could use as a reference. This lake is located in the Moon River watershed that eventually drains into Georgian Bay (P. Chow-Fraser, unpublished data, 2005; Figs. 1 and 2K) and is oligotrophic with a seasonal mean [TP] of 6.56 ($\mu g \cdot L^{-1}$). Blackstone Lake was sampled in the same way as we had sampled all ten embayments.

Water-quality sampling

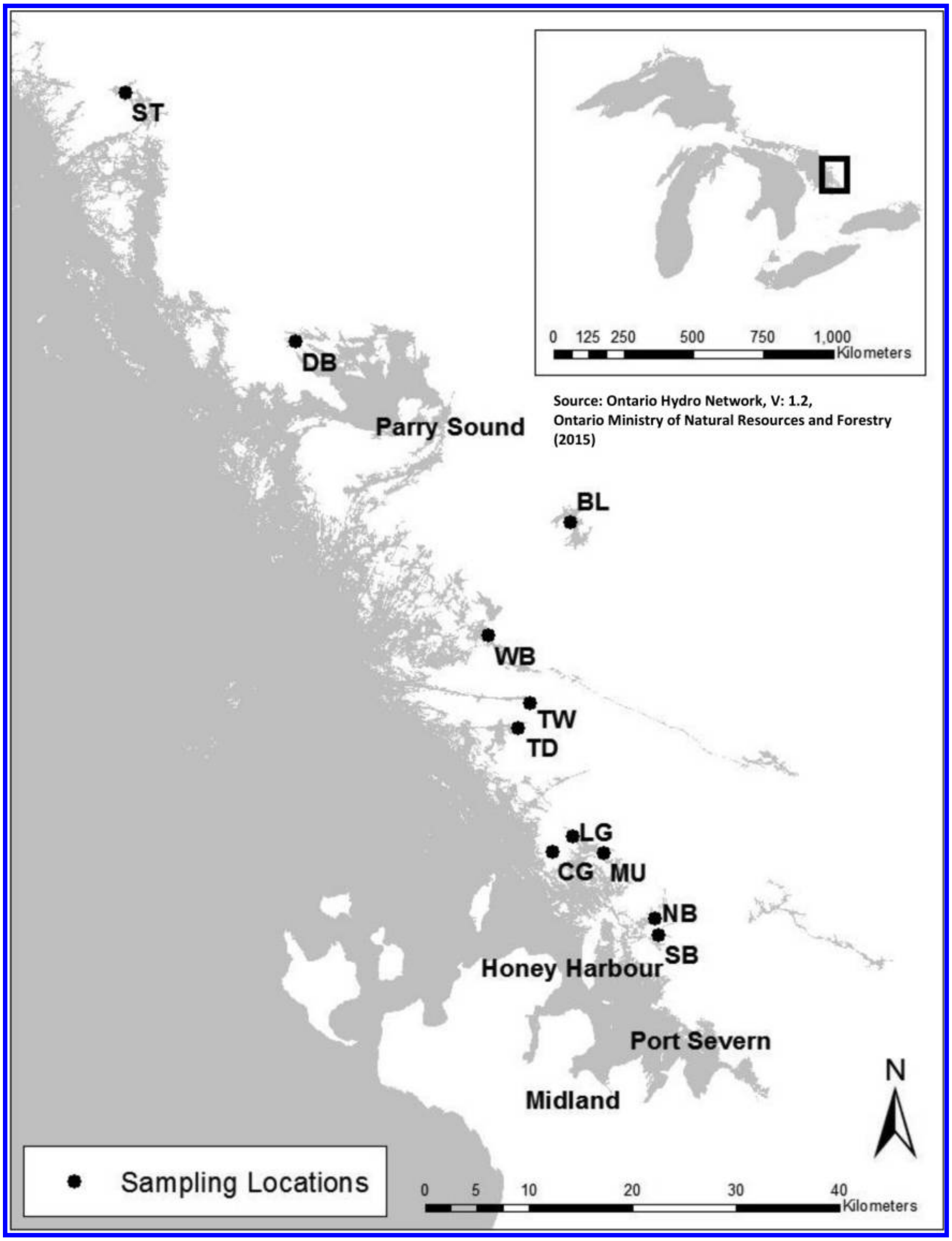
To capture seasonal variation in [TP] and [CHL], we sampled each embayment or lake on a monthly basis from May to September in a single year between 2012 and 2015. We collected all water samples and measured water temperature (TEMP; °C) and dissolved oxygen (DO; $mg \cdot L^{-1}$) at the deepest point at each site. On each sampling occasion, we obtained physicochemical information throughout the water column at 1 m intervals to a maximum depth of 25 m in deeper bays and to 1 m above the sediment surface in shallower bays. A YSI 6920 V2 sonde was calibrated prior to each sampling week and used to measure TEMP and DO in situ to determine the presence and extent of hypolimnetic anoxia during the stratification season. We used a Van Dorn sampler to collect water at the midpoint of the epilimnion, metalimnion, and hypolimnion to determine concentrations of [TP] ($\mu g \cdot L^{-1}$); however, only samples collected from the epilimnion were used to determine concentrations of [CHL] ($\mu g \cdot L^{-1}$).

Samples to be analyzed for [TP] were frozen and transported back to McMaster University, Hamilton, Ontario. Unless otherwise indicated, all analyses were performed in triplicate for each variable. [TP] was determined with the molybdenum blue method (Murphy and Riley 1962) following potassium persulfate digestion in an autoclave for 50 min (120 °C, 15 psi (1 psi = 6.894 kPa)). [CHL] filters were extracted in 90% reagent grade acetone in a freezer over a 24 h period. Following extraction, samples were acidified with hydrochloric acid (0.1 $mol \cdot L^{-1}$), and fluorescence was read with a Turners Design Trilogy Fluorometer. Note that grab samples for [CHL] determination were missing from North Bay, South Bay, and Tadenac Bay in 2012; therefore, we estimated [CHL] for these sites using a calibrated bbe Moldaenke FluoroProbe during 2015.

Trophic status determination

For calibration purposes, we used the trophic state classification system developed by Reckhow and Chapra (1983) to determine the trophic status for all ten embayments and the inland lake reference system. Seasonal mean values of [TP] collected from each stratum (epilimnion, metalimnion, and hypolimnion) were averaged to determine trophic status. Using this classification sys-

Fig. 1. Location of ten coastal embayments in eastern Georgian Bay (Cognashene Lake, CG; Deep Bay: DB; Longuissa Bay: LG; Musquash Bay: MU; North Bay: NB; South Bay: SB; Sturgeon Bay: ST; Tadenac Bay: TD; Twelve Mile Bay: TW; Woods Bay: WB) and the inland lake (Blackstone Lake: BL) within the Georgian Bay drainage basin.



tem, water bodies with [TP] < 10 μg·L^{-1} were classified as oligotrophic, and those with [TP] > 20 μg·L^{-1} were classified as eutrophic, and all intermediate values (i.e., >10 to <20 μg·L^{-1}) were classified as mesotrophic.

Lakeshore Capacity Model

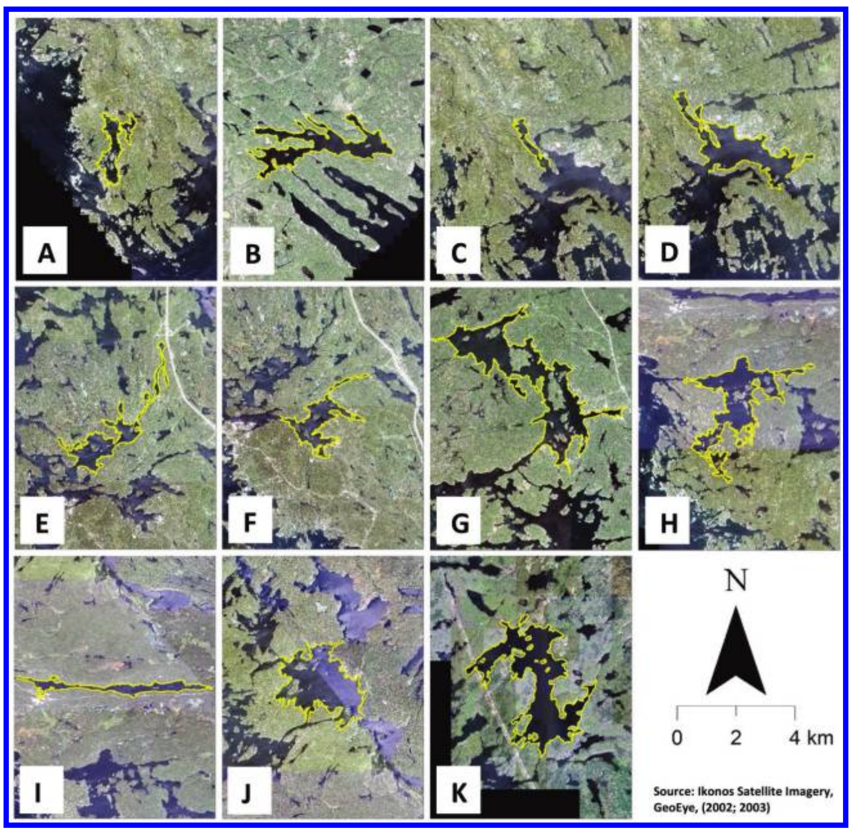
We applied the LCM to embayments and Blackstone Lake using the components, coefficients, and technical assumptions outlined by Paterson et al. (2006) and by following instructions provided in the Lakeshore Capacity Assessment Handbook (Ministries of Ontario 2010). These two resources are the recommended guidance documents for applying the LCM to lakes located in south-central Ontario. All calculations were tabulated in Microsoft Excel spreadsheets. As instructed by the Lakeshore Capacity Assessment Handbook, we ignored upstream lakes ≤25 ha in surface area unless they had substantial levels of human development

Table 1. Summary of basin morphometry for ten embayments and Blackstone Lake.

Site name	A_o (10^6 m²)	V (10^6 m³)	L (m)	B (m)	Z_{mean} (m)	Z_{max} (m)	P (10^3 m)	Z_r (m)	D_L	D_V
Blackstone Lake	5.083	101.534	3983	1587	19.97	60.80	37.86	2.39	4.75	0.99
Tadenac Bay	3.931	19.695	3684	1239	5.01	29.10	37.59	1.30	5.35	0.52
South Bay	1.494	4.389	1403	891	2.94	15.96	17.82	1.16	4.11	0.55
North Bay	1.886	9.180	2137	907	4.87	22.74	22.53	1.47	4.63	0.64
Cognashene Lake	0.965	2.590	1416	576	2.66	16.88	13.13	1.52	3.77	0.47
Twelve Mile Bay	1.189	6.516	3823	461	5.48	13.89	16.47	1.13	4.26	1.18
Deep Bay	2.664	15.880	4390	760	5.96	19.51	24.50	1.06	4.23	0.92
Musquash Bay	3.201	40.293	3326	1398	12.59	42.86	22.21	2.12	3.50	0.88
Sturgeon Bay	4.988	22.109	2646	1346	4.43	14.51	33.10	0.58	4.18	0.92
Woods Bay	3.630	13.003	2781	1966	3.58	13.42	20.30	0.62	3.00	0.80
Longuissa Bay	0.355	1.276	1477	347	3.59	11.89	5.30	1.77	2.53	0.91

Note: A_o = surface area (m²), V = volume (m³), L = maximum length (m), B = breadth (m), Z_{mean} = mean depth (m), Z_{max} = maximum depth (m), Z_r = relative depth (m), D_L = shoreline development, D_V = volume development.

Fig. 2. IKONOS satellite imagery of study sites acquired during 2002 and 2003: (A) Cognashene Lake, (B) Deep Bay, (C) Longuissa Bay, (D) Musquash Bay, (E) North Bay, (F) South Bay, (G) Sturgeon Bay, (H) Tadenac Bay, (I) Twelve Mile Bay, (J) Woods Bay, and (K) Blackstone Lake. [Colour online.]



along the shoreline. All statistical analyses were performed with JMP 12 software (SAS, Cary, North Carolina, 2015).

Internal loading of P

Loss of hypolimnetic oxygen can facilitate the release of nutrients from lake sediments through internal loading (Sondergaard et al. 2001; Nurnberg and LaZerte 2004). Because internal load was not explicitly calculated in the LCM, and we presumed this process had occurred in coastal embayments that had anoxic hypolimnia, we used Nurnberg and LaZerte's (2004) approach to estimate internal P loading using the anoxic factor (AF; Nurnberg 1995) for all embayments that experienced hypolimnetic anoxia during the stratification season (i.e., Cognashene Lake, Deep Bay, Longuissa Bay, North Bay, South Bay, Sturgeon Bay, and Twelve Mile Bay). To calculate the AF, we first determined the area of sediments overlain by anoxic waters for each sampling period (n = 5). These anoxic area calculations were then summed and divided by the lake surface area. This was done to measure how anoxia spread in time and space over the period of thermal stratification. We then calculated internal load (mg·year^{-1}) by multiplying the AF by the P release rate (mg·m^{-2}·day^{-1}) presented in Nurnberg and LaZerte (2004) that corresponded with the trophic status of each respective embayment. We used P release rates developed by Nurnberg and LaZerte (2004) specifically for lakes located on the Precambrian Shield (i.e., oligotrophic lakes ([TP] < 6 μg·L^{-1}): 0 mg·m^{-2}·day^{-1}; oligotrophic lakes ([TP] = 6–9 μg·L^{-1}): 0.7 mg·m^{-2}·day^{-1}; mesotrophic lakes ([TP] = 9–20 μg·L^{-1}): 1.4 mg·m^{-2}·day^{-1}; mesotrophic and eutrophic lakes ([TP] > 20 μg·L^{-1}): 3 mg·m^{-2}·day^{-1}).

The LCM also takes into consideration the amount of P that is retained in each lake modelled through settling of P to the lake sediments. Two different settling velocities can be applied depending on the extent of hypolimnetic oxygen depletion over the period of thermal stratification, with a settling velocity of

Table 2. Summary of building density (number of buildings·shoreline length (m)$^{-1}$), dock density (number of docks·shoreline length (m)$^{-1}$), and road density (length of road (m)·ha^{-1}) within a 2 km buffer of the shoreline, and the Index of Resistance to Mixing (IRM).

Site name	Building density	Dock density	Road density	IRM
Blackstone Lake	0.00419	0.00400	5.87	NA
Longuissa Bay	0.00112	0.00112	0.00	54.04
Twelve Mile Bay	0.00759	0.01591	7.54	57.27
North Bay	0.01096	0.00768	7.55	77.32
Musquash Bay	0.00288	0.00297	0.00	84.50
Woods Bay	0.00443	0.00907	3.78	124.96
Deep Bay	0.00948	0.00617	11.95	127.47
South Bay	0.01094	0.01886	15.81	131.25
Tadenac Bay	0.00008	0.00016	1.62	131.84
Cognashene Lake	0.00533	0.00571	0.00	133.51
Sturgeon Bay	0.01061	0.01777	7.44	261.08

12.40 m·year^{-1} for lakes with oxic hypolimnia and a velocity of 7.20 m·year^{-1} for lakes that develop anoxic hypolimnia (Paterson et al. 2006). To prevent overestimates of internal loading by using both the anoxic P settling rate and the calculation of internal load, we used the P settling rate for oxic lakes when adding our calculated internal P load to the model.

Degree of mixing with Georgian Bay

We developed an Index of Resistance to Mixing (IRM) to indicate the degree of mixing between embayment waters and the waters of Georgian Bay; parameters of this index included morphometric parameters that are assumed to influence water exchange between the embayment and open waters of Georgian Bay (eq. 1):

$$\text{IRM} = \sqrt{(P/L \cdot \text{Least Cost Pathway})} \tag{1}$$

where lake perimeter and maximum length are P (m) and L (m), respectively, and the Least Cost Pathway (m) is the line of least resistance that connects the intersecting point of maximum length and maximum breadth in the embayment with the narrowest point in the outflow channel to Georgian Bay. We calculated this using the Least Cost Pathway Tool in ArcGIS 10.3.1 (ESRI, Redlands, California, 2015).

Examination of additional factors

We considered a number of additional anthropogenic and morphometric factors to derive an alternative model that we call the Anthro-Geomorphic Model (AGM). We used Google Earth satellite images taken in 2015 to enumerate primary residential dwellings (i.e., cottages, trailers, etc.), excluding structures not typically connected to septic systems (e.g., garages, warehouses, sheds, and boathouses). The number of docks at each site were enumerated with the same methods used to enumerate primary residential dwellings. The total number of docks and buildings were then divided by the embayment perimeter (m) to calculate dock and building density, which represented the number of structures per unit length of the shoreline (Table 2). Additionally, we calculated road density by creating 2 km buffers around the embayment perimeter and then used the Provincial Road Network layer to calculate length of road per unit buffer area (m·ha^{-1}) (Table 2). To examine the relative impact of building density, dock density, road density, and the IRM, as well as other morphometric variables (e.g., maximum depth, mean depth, etc.; Table 1) on the nutrient status of embayments, we used forward stepwise multiple regression analysis in JMP 12 (SAS, Cary, North Caralina, 2015). We then used Akaike information criterion (AIC) analysis, based on our forward stepwise multiple regression, to select the most suitable model.

Assessment of model fit

The precision of each model to estimate seasonal mean [TP] was evaluated to determine which model performed best when applied to coastal embayments. All predicted values that were within ±20% deviation from measured values were considered acceptable. This criterion was established by Hutchinson et al. (1991) when they evaluated the performance of the LCM during the early stages of model development. The Ontario Ministry of the Environment and Climate Change continues to use this criterion as a threshold to determine whether an estimated value should be accepted or rejected when applying the LCM (Ministries of Ontario 2010).

Following Pineiro et al. (2008), we evaluated the performance of each model by regressing measured values against estimated values and assessing how closely they approximated a perfect fit between measured and estimated values (i.e., slope of 1 and intercept of 0). Additionally, Grace-Martin (2012) proposed three statistics to evaluate fit for ordinary least squares regression models: R^2, the overall F test, and the root mean square error (RMSE). All three are based on the sum of squares total (SST) and the sum of squares error (SSE), where the SST measures departure of the data from the mean and SSE measures departure of the data from the model's predicted values. In this paper, we will report all three statistics. We also calculated mean absolute error (μg·L^{-1}) and mean percent error between measured and estimated values. These measures of performance were used collectively to evaluate the accuracy of estimates produced by each model. We omitted Blackstone Lake in all cases, since we were only interested in how the model performed on coastal embayments.

Data for comparison of TP–CHL relationships

We compared our TP–CHL regression equation against those in published and unpublished studies of similar inland and coastal systems. This was done to determine if it is necessary to use a system-specific TP–CHL regression equation to estimate primary production in coastal embayments. The data were from several sources: (*i*) 134 locations in the nearshore (coastal) zone of eastern and northern Georgian Bay between 2003 and 2005 (Diep et al. 2007), (*ii*) 24 inland lakes of central Ontario (Zimmerman et al. 1983), (*iii*) eight lakes in northwestern Ontario, Experimental Lakes Area (ELA) (Chow-Fraser et al. 1994), and (*iv*) 15 inland lakes in central Ontario sampled between 1976 and 1987 (Molot and Dillon 1991).

Results

Embayment trophic status

As expected, Reckhow and Chapra's (1983) trophic state classification system classified the reference inland lake, Blackstone Lake, as oligotrophic (Table 3). Using the same trophic state classification system, we found that Longuissa Bay, Musquash Bay, and Tadenac Bay were also classified as oligotrophic ([TP] < 10 μg·L^{-1}), while Cognashene Lake, Woods Bay, Twelve Mile Bay, North Bay, Deep Bay, and South Bay were classified as mesotrophic ([TP] > 10 to 20 μg·L^{-1}; Table 3). Sturgeon Bay was the only embayment to be classified as borderline eutrophic, as the [TP] measured in this embayment were just above the threshold limit ([TP] > 20 μg·L^{-1}).

LCM

We applied the LCM to Blackstone Lake as a check on the validity of our calculations, knowing that this inland lake had been assessed as being oligotrophic by previous monitoring conducted by the Blackstone Lake Cottage Association as part of the Lake Partner program sponsored by the Ontario Ministry of Environment and Climate Change (Sale and Sale 2013) and by data collected in this study. The model produced an estimated [TP] of 6.96 μg·L^{-1}, which was in close agreement with the measured value of 6.56 μg·L^{-1} (Table 4). The absolute error resulting from this model was 0.04 μg·L^{-1}, with a percent error of 6.10, which is

Table 3. Mean (±SE) seasonal total phosphorus concentration ([TP]; μg·L^{-1}; samples collected from all strata) and mean (±SE) seasonal epilimnetic chlorophyll *a* concentration ([CHL]; μg·L^{-1}) for ten embayments and Blackstone Lake.

Site	Year sampled	[TP] (μg·L^{-1})	[CHL] (μg·L^{-1})	Trophic status
Blackstone Lake	2013	6.56 (±0.54)	1.90 (±0.18)	Oligotrophic
Longuissa Bay	2015	7.39 (±0.70)	4.48 (±0.34)	Oligotrophic
Musquash Bay	2015	7.85 (±0.94)	3.78 (±0.38)	Oligotrophic
Tadenac Bay	2012	8.24 (±0.57)	5.15 (±1.29)	Oligotrophic
Cognashene Lake	2014	10.60 (±0.89)	4.86 (±0.83)	Mesotrophic
Woods Bay	2015	11.11 (±2.72)	4.04 (±0.74)	Mesotrophic
Twelve Mile Bay	2014	13.00 (±1.11)	5.77 (±0.75)	Mesotrophic
North Bay	2012	13.41 (±0.66)	7.62 (±1.17)	Mesotrophic
Deep Bay	2014	18.41 (±2.32)	11.72 (±3.32)	Mesotrophic
South Bay	2012	18.85 (±2.13)	6.66 (±0.80)	Mesotrophic
Sturgeon Bay	2015	20.96 (±4.45)	6.59 (±0.97)	Eutrophic

Note: In all cases, using Reckhow and Chapra's (1983) trophic state classification system, sites with [TP] < 10 μg·L^{-1} were interpreted as being oligotrophic, those with [TP] >10 to <20 μg·L^{-1} were interpreted as being mesotrophic, and all sites with [TP] > 20 μg·L^{-1} were interpreted as being eutrophic. n = 5 for all sites.

well within the acceptable ±20% deviation from our measured value. This comparison confirmed that the manner in which we applied the LCM to Blackstone Lake and calculated the estimated [TP] was correct.

The LCM could only be applied to eight of the ten embayments sampled, because the watersheds of Woods Bay and Musquash Bay were too large and complex to be analysed during the course of this study (watershed sizes of 5145.80 and 5008.48 km^2, respectively). The seasonal mean [TP] predicted by the LCM for three of the embayments fell within ±20% deviations of the measured values (Table 4). The least squares linear regression between measured and estimated [TP] did not produce a significant fit (F = 4.42; P = 0.08) and resulted in a weak relationship, with a correlation coefficient of 0.42 and an RMSE of 4.16 (Table 5; Fig. 3A), while the slope of 0.75 and intercept of 3.62 compared poorly against the expected values of 1 and 0. The mean absolute error was 3.14 μg·L^{-1} and the mean absolute percent error was 22.40 for this model (Table 5).

Given that the process of internal P loading is thought to occur in embayments that developed hypolimnetic anoxia (<1 mg·L^{-1}) during the period of thermal stratification, we incorporated an estimate of internal P load into the LCM calculation for all sites that exhibited these symptoms (i.e., Cognashene Lake, Deep Bay, Longuissa Bay, North Bay, South Bay, Sturgeon Bay, and Twelve Mile Bay). Accounting for additional inputs of [TP] from internal loading yielded acceptable estimates for four of the eight embayments (within ±20% of observed values; Longuissa Bay, North Bay, Tadenac Bay, Twelve Mile Bay; Table 4). Least squares linear regression between measured and estimated [TP] did not produce a significant fit (F = 5.90; P = >0.05), while the correlation coefficient improved slightly to 0.49 and the RMSE was reduced to 3.90 (Table 5; Fig. 3B). Although this regression produced an improved intercept value of 0.09, which is closer to the expected intercept of zero, the slope value was 1.33 and did not alignment well with the expected slope of 1 (Table 5; Fig. 3B). The mean absolute error with the inclusion of internal load increased slightly to 3.47 μg·L^{-1} and had an mean absolute percent error of 20.81 (Table 5).

AGM

We wanted to develop a predictive model using additional landscape and morphometric variables (e.g., IRM, surface area (m^2), volume (m^3), breadth (m), mean depth (m), maximum depth (m), perimeter (m), catchment size (m^2), percent wetland in catchment area, road density, dock density, and building density). Preliminary explorations revealed that variables reflecting the degree of anthropogenic disturbance along the shoreline were significant predictors of seasonal mean [TP] in embayments. These variables included building density (number of residential structures per unit length of shoreline (m); n = 10; R^2 = 0.79; P = 0.0006), road density (road length (m) within 2 km buffer of shoreline (ha); n = 10; R^2 = 0.73; P = 0.0018) as well as dock density (number of docks per unit length of shoreline (m); n = 10; R^2 = 0.61; P = 0.0069). We also reasoned that increased mixing between the oligotrophic water of Georgian Bay and the water in embayments should decrease mean [TP] in embayments. Therefore, irrespective of the anthropogenic P load, we expected embayments with low IRM scores to have low [TP] due to greater water exchange and embayments with high IRM scores to have high [TP] as a result of a lesser degree of water exchange (Table 2). This expectation was upheld when we regressed seasonal mean [TP] against the IRM values and found a significant positive relationship (n = 10; R^2 = 0.40; P = 0.0489).

Both dock density and road density were removed during model development because of their high correlation with building density. We retained building density as a variable because it is a more direct indicator of nutrient input than the other two variables given that the greatest anthropogenic source of P input into the embayment is from septic systems associated with the buildings along the lakeshore (Dillon et al. 1994). In addition, when we regressed [TP] against building density and IRM, we found no significant interaction (eq. 2):

$$\text{(2)}\quad \begin{aligned}&\text{Mean [TP]} = 3.670\ (1.345) + 895.792\ (145.184)\cdot\text{Building Density} + 0.031\ (0.010)\cdot\text{IRM}\\ &(n = 10;\ R^2 = 0.91;\ P < 0.0002;\ \text{SE of regression coefficients given in parentheses})\end{aligned}$$

This regression model was significant and explained 91% of the total variation in [TP] values. The AGM also predicted seasonal mean [TP] within ±20% of measured [TP] for all ten embayments (Table 4) and produced the least error with a mean absolute error of 1.20 μg·L^{-1} and a mean absolute percent error of 9.97 (Table 5). Least squares linear regression between measured and estimated values was significant (F = 77.97, P < 0.0001), with a correlation coefficient of 0.91, RMSE of 1.59, a slope of 0.99, and an intercept of 0.0001, which were almost in perfect agreement with the expected slope of 1 and intercept of 0 (Table 5; Fig. 3C).

TP–CHL relationship

We obtained a significant log-linear relationship between mean seasonal epilimnetic [CHL] and [TP] for this study (eq. 3; Fig. 4A).

$$\text{(3)}\quad \begin{aligned}&\log\ [\text{CHL}] = -0.157\ (0.192) + 0.853\ (0.177)\cdot\log\ [\text{TP}]\\ &(n = 10;\ R^2 = 0.74;\ P = 0.0013;\ \text{SE of regression coefficients given in parentheses})\end{aligned}$$

Table 4. Summary of measured seasonal mean total phosphorus concentration ([TP], μg·L^{-1}) and estimates of [TP] calculated by the Lakeshore Capacity Model (LCM), the Lakeshore Capacity Model + Internal Loading (LCM+IL), and by the Anthro-Geomorphic Model (AGM).

		Estimated [TP]			% Deviation		
Site	Measured [TP]	LCM	LCM+IL	AGM	LCM	LCM+IL	AGM
Blackstone Lake	6.56	6.96	6.96	NA	6.10	6.10	NA
Cognashene Lake	10.60	9.72	7.66	12.58	−8.30	−27.74*	18.68
Deep Bay	18.41	13.54	11.70	16.11	−26.45*	−36.45*	−12.49
Longuissa Bay	7.39	9.77	6.94	6.35	32.21*	−6.09	−14.07
Musquash Bay	7.85	NA	NA	8.77	NA	NA	12.99
North Bay	13.41	17.15	12.55	15.89	27.89*	−6.41	18.49
South Bay	18.85	19.67	13.63	17.54	4.35	−27.69*	−6.95
Sturgeon Bay	20.96	14.42	10.46	21.27	−31.20*	−50.10*	1.48
Tadenac Bay	8.24	7.38	7.38	7.83	−10.44	−10.44	−4.98
Twelve Mile Bay	13.00	17.99	12.80	12.24	38.38*	−1.54	−5.85
Woods Bay	11.11	NA	NA	11.52	NA	NA	3.69

Note: Statistics to evaluate model performance are percent deviations (% Deviation) between estimated and measured values. NA = value unavailable because LCM and LCM+IL could not be applied to these embayments due to the extremely large size and complexity of their respective watersheds.

*Denotes that estimated value exceeds ±20% of measured value (Hutchinson et al. 1991).

Table 5. Summary of regression analysis relating measured total phosphorus concentration ([TP], μg·L^{-1}) against estimated [TP] by three management models.

Model	$Error_{mean}$	$Error_{mean\%}$	R^2 value	*F* ratio	RMSE	Slope	Intercept
LCM	3.14	22.40	0.42	4.42	4.16	0.75	3.62
LCM+IL	3.47	20.81	0.49	5.90	3.90	1.33	0.09
AGM	1.20	9.97	0.91	77.97*	1.59	0.99	0.0005

Note: LCM = Lakeshore Capacity Model, LCM+IL = Lakeshore Capacity Model + Internal Load, and AGM = Anthro-Geomorphic Model. $Error_{mean}$ = mean absolute error (μg·L^{-1}), $Error_{mean\%}$ = mean absolute percent error, RMSE = root mean square error. The expected slope and intercept in all cases are 1 and 0, respectively.

*Denotes that *F* test was significant (<0.05).

We assembled data from four other studies whose sites were underlain by Precambrian Shield (see Methods for sources) and plotted them together with those from this study (Fig. 4B). The data can be grouped into four categories: (1) inland lakes (both in central Ontario and in northwestern Ontario), (2) sites occurring in the coastal zone of eastern Georgian Bay, (3) sites occurring in the coastal zone of northern Georgian Bay, and (4) the 10 coastal embayments included in this study. We found no significant differences among these groups with respect to the slope of the regression equation (ANCOVA; $P = 0.843$; Fig. 4B), but we found significantly different *y* intercepts. Sites in northern Georgian Bay had the lowest [CHL], followed by eastern Georgian Bay, then inland lakes, while our embayments had the highest values of [CHL] per unit [TP] (Fig. 4B).

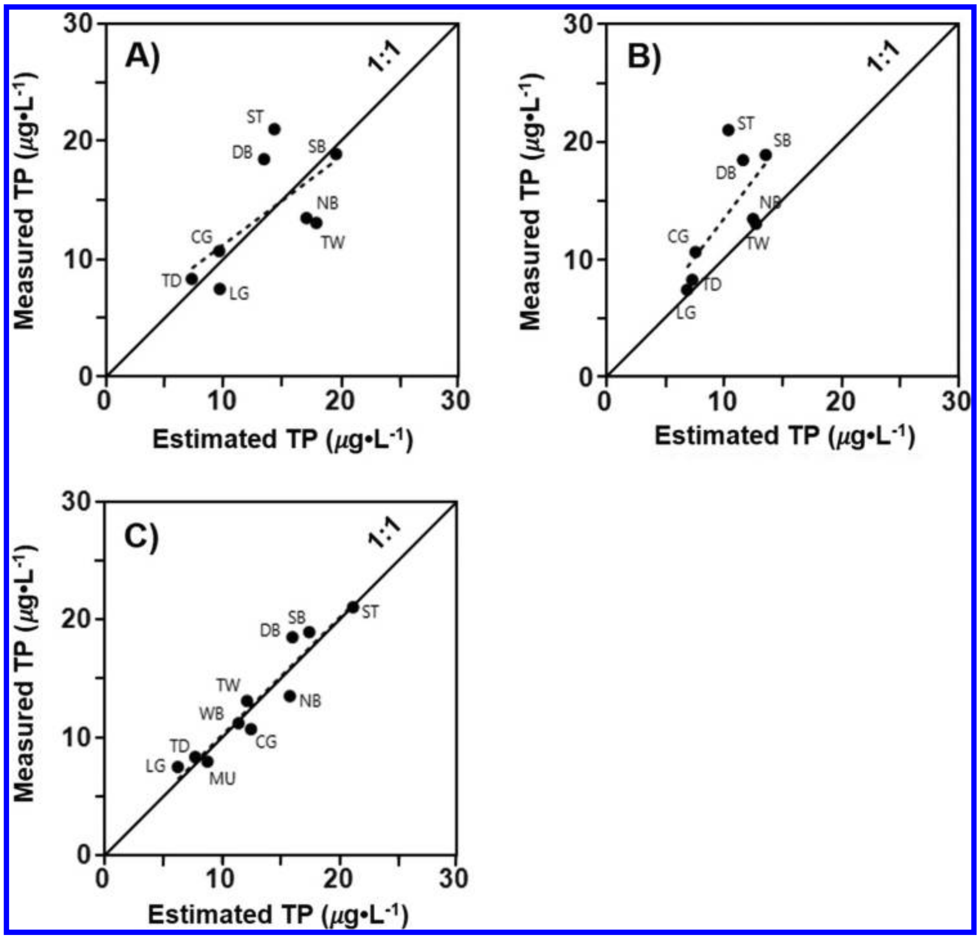
Discussion

One of our objectives was to assess the appropriateness of using the LCM to differentiate between natural and anthropogenic sources of P for coastal embayments of eastern Georgian Bay. The estimated [TP] for Blackstone Lake, calculated with the LCM and based on our assumptions outlined in the Methods section, was within 6.10% of measured values. When we applied the LCM to embayments, we found the model overestimated [TP] by ±20% for five of eight embayments. Incorporating internal P load did not result in a marked improvement of the model's predictive ability and exceeded acceptable limits for four of eight embayments, with no considerable reductions in model error. By comparison, the AGM we developed, based on the density of primary residential dwellings along the shoreline of embayments as well as basin morphometry, successfully estimated [TP] within ±20% deviations from measured values for all ten embayments. Least squares linear regression between measured and estimated values was significant, producing the lowest RMSE and highest correlation coefficient of the three models evaluated. The AGM also produced the least amount of error associated with slope and intercept estimates corresponding to the regression of measured against estimated values.

Accuracy of the AGM depended heavily on our ability to include sites that spanned the entire gradient of [TP], building density, and IRM values. Tadenac Bay and Musquash Bay both had extremely low building densities (<0.0001 and 0.0029, respectively) that were orders of magnitude lower than those in Sturgeon Bay and South Bay, which had the highest building densities (0.0106 and 0.0109, respectively; Table 2). The IRM scores varied by fivefold, with Longuissa Bay having the lowest score of 54 and Sturgeon Bay having the highest score of 261 (Table 2). The high [TP] in Sturgeon Bay reflects a combination of high cottage density as well as high IRM score, which presumably exacerbated the P enrichment in this embayment, since the extensive network of channels prevented the embayment water from being diluted by the oligotrophic water of Georgian Bay. Although the AGM produced accurate estimates of [TP] in coastal embayments, there are factors, such as variability in local climate patterns and fluctuations in Lake Huron water levels, that could impact the performance of the model over time. It is important that the impact of these variables be considered when applying the AGM to coastal embayments as conditions change.

Embayments are unique coastal systems that are connected to Georgian Bay through narrow channels. This configuration results in limited flushing and mixing patterns compared with those in more exposed bays; therefore, the amount of water that is exchanged between coastal embayments and the open water of Georgian Bay is variable as a result of the unique geomorphology specific to each site. Embayments that are elongated and connected to Georgian Bay through a narrow channel or network of channels experience low water exchange with Georgian Bay proper, whereas embayments that are more open and connected to Georgian Bay through wide channels experience high water exchange with Georgian Bay proper. In embayments that experience a low water exchange (e.g., Sturgeon Bay, Deep Bay), much of

Fig. 3. Measured versus estimated scatterplots for (A) Lakeshore Capacity Model (LCM), (B) Lakeshore Capacity Model with inclusion of internal loading (LCM+IL), and (C) Anthro-Geomorphic Model (AGM). Solid line is the line of unity (measured = estimated). Dotted line represents best-fit linear regression of the data.



the external nutrient load entering these systems is expected to stay within the embayment due to limited flushing. These additional nutrients could be sufficient to make the system develop symptoms of eutrophication that are not normally associated with more exposed bays of Georgian Bay and Lake Huron (Wells and Sealock 2009).

The LCM was developed and calibrated with six inland lakes (Dillon et al. 1986; Paterson et al. 2006) that had small surface areas (A_{mean} = 55 ha) with deep basins (mean Z_{max} = 32 m), whereas our eight embayments were generally large (A_{mean} = 178 ha) and relatively shallow (mean Z_{max} = 18 m). The difference in morphometry between the calibration lakes and embayments is likely the cause of some of the variation found in model results due to differing lake processes. Another reason for the difference in performance of the LCM may be related to how the P settling velocity had been empirically determined and the choice of using oxic versus anoxic rates. Inland lakes that had been used to measure settling rates were deep and well-stratified and had uniform basin morphometry (Kirchner and Dillon 1975; Dillon and Kirchner 1975). By contrast, the coastal embayments sampled in this study were shallow, though they typically had one deep basin that could develop thermal stratification. Even when anoxic conditions developed in the hypolimnia, the proportion of bottom sediments overlain by anoxic waters was small (a low of 1.09% seasonal mean in Longuissa Bay to high of 21.45% seasonal mean in Deep Bay; mean of 10.76% across all embayments that developed anoxic hypolimnia). Our attempts to correct for overestimates of internal loading by applying anoxic settling velocities to only the portion of the sediments that were overlain by anoxic waters did not improve the overall predictive power of the LCM.

Since these coastal embayments produce a larger amount of [CHL] per unit [TP] compared with other systems (Fig. 4B), lake managers run the risk of underestimating the amount of [CHL] produced from increases or reductions in [TP]. The system-specific TP–CHL relationship in this study should yield more accurate estimates of algal biomass in response to management actions. We do not know why the intercept for embayments in the comparison of TP–CHL relationships is significantly higher, but it may be related to inefficient grazing pressure (Hansson 1992; Sarnelle 1992; Mazumder 1994) or a higher proportion of biologically available P (Schindler et al. 1978; Hoyer and Jones 1983; Butkus et al. 1988).

Use of the AGM for management

The trophic state classification system developed by Reckhow and Chapra (1983) used 10 μg·L^{-1} to separate oligotrophy from mesotrophy and 20 μg·L^{-1} to separate mesotrophy from eutrophy. According to this classification system, Longuissa Bay, Musquash Bay, and Tadenac Bay should be classified as oligotrophic (7.39, 7.85, and 8.24 μg·L^{-1}, respectively), Sturgeon Bay as borderline eutrophic (20.96 μg·L^{-1}), and the remaining six embayments as mesotrophic (Table 3). Consistent with conventional understanding of oligotrophic systems, none of the oligotrophic coastal embayments experienced severe symptoms of cultural eutrophication (Longuissa Bay only developed oxygen depletion late in the stratification season). By comparison, five of the six mesotrophic

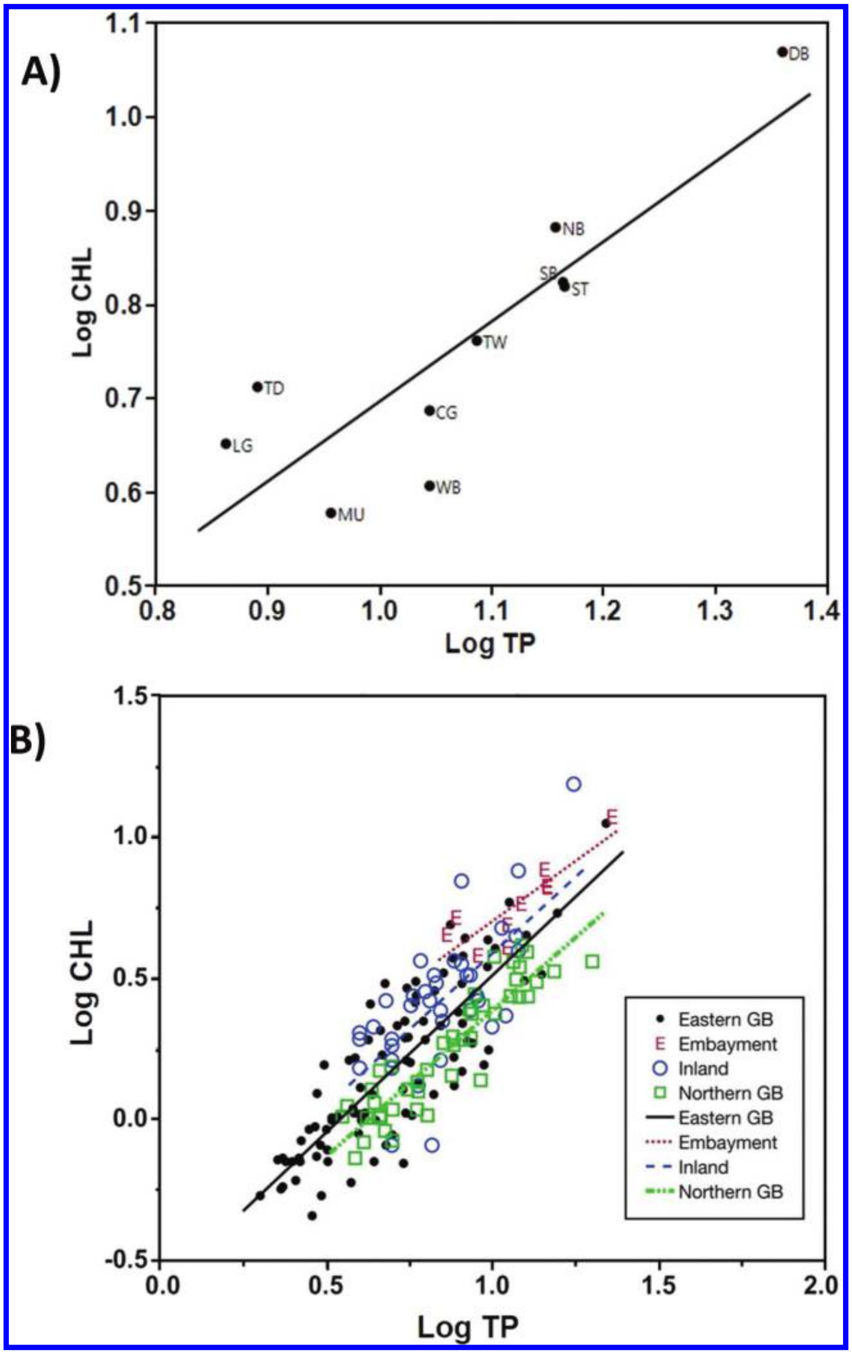


Fig. 4. Plot of mean seasonal epilimnetic $\log_{10}$ chlorophyll (CHL; μg·L^{-1}) versus mean seasonal epilimnetic $\log_{10}$ total phosphorus (TP; μg·L^{-1}) for (A) ten embayments in this study and (B) comparison of ten embayments in this study with data from other studies. [Colour online.]

embayments developed hypolimnetic oxygen depletion by midsummer, while Sturgeon Bay, the only embayment in the eutrophic category, exhibited severe symptoms of cultural eutrophication, including hypolimnetic oxygen depletion throughout the summer, reduced water clarity in the epilimnion, and several instances of harmful and (or) nuisance algal blooms over the past decade (Gartner Lee Limited 2008).

Since building density is the only variable in the AGM that can be effectively managed, we calculated the threshold corresponding to [TP] of 10, 15, and 20 μg·L^{-1} for each embayment (Table 6). To avoid development of hypoxic hypolimnia, [TP] in coastal embayments should remain below 10 μg·L^{-1}, and it is clear that the majority of the embayments in this study have already surpassed that threshold. In fact, the AGM estimated a [TP] of 11.46 μg·L^{-1} in Sturgeon Bay prior to the development of any shoreline dwellings, which would have exceeded this threshold. This confirms the paleolimnological studies conducted in Sturgeon Bay that indicate it was naturally mesotrophic even before European contact (Gartner Lee Limited 2008).

Based on this study, we do not recommend using the LCM to assess the trophic response of coastal embayments in eastern Georgian Bay to human development. Instead, we recommend using the AGM because it produced more accurate estimates of [TP] and could be applied to all ten embayments without exception. We do, however, recommend that further research be carried out to investigate the potential influence that changing climate patterns and Lake Huron water level fluctuations could have on the performance of the AGM. We found that embayments

Table 6. Building densities (number of buildings·shoreline (m)$^{-1}$) predicted by the Anthro-Geomorphic Model that correspond to three trophic states as determined by total phosphorus concentration ([TP], μg·L^{-1}) in the 10 coastal embayments.

Site name	10 μg·L^{-1}	15 μg·L^{-1}	20 μg·L^{-1}	Existing no. of dwellings
Cognashene Lake	34	107	179	70
Deep Bay	69	204	340	232
Longuissa Bay	28	57	87	6
Musquash Bay	94	217	340	64
North Bay	101	226	350	247
South Bay	48	146	245	195
Sturgeon Bay	—*	129	313	351
Tadenac Bay	102	326	552	3
Twelve Mile Bay	84	176	276	125
Woods Bay	59	171	283	90

*The AGM predicts that background concentration for Sturgeon Bay would be 11.46 μg·L^{-1} even when building density is zero.

produced significantly higher [CHL] per unit of [TP] when compared with other geographic regions in Georgian Bay and inland Precambrian Shield lakes. It is therefore important that environmental managers use the appropriate TP–CHL relationship for specific aquatic systems when evaluating changes in primary production in response to management efforts.

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References

Butkus, S.R., Welch, E.B., Horner, R.R., and Spyridakis, D.E. 1988. Lake response modeling using biologically available phosphorus. J. Water Pollut. Control Fed. **60**(9): 1663–1669. Retrieved from http://www.jstor.org/stable/25046796.

Chapra, S.C., Dove, A., and Warren, G.J. 2012. Long-term trends of Great Lakes major ion chemistry. J. Gt. lakes Res. **38**: 550–560. doi:10.1016/j.jglr.2012.06.010.

Chiandet, A., and Sherman, K. 2014. Report on water quality from 2010–2012 in the Honey Harbour area of Georgian Bay [online]. Severn Sound Environmental Association. Available from https://georgianbay.civicweb.net/document/108363/HH_2010-2012_WQ_Report_20140404FINAL.pdf?handle=409566E293FD44A0A63FFBA842ECE76C.

Chow-Fraser, P., Trew, D.O., Findlay, D., and Stainton, M. 1994. A test of hypotheses to explain the sigmoidal relationship between total phosphorus and chlorophyll *a* concentrations in Canadian lakes. Can. J. Fish. Aquat. Sci. **51**(9): 2052–2065. doi:10.1139/f94-208.

Diep, N., Howell, T., Benoit, N., and Boyd, D. 2007. Limnological conditions in eastern Georgian Bay: data summary of the 2003–2005 water quality survey. Water Monitoring and Reporting Section, Environmental Monitoring and Reporting Branch, Ontario Ministry of the Environment and Climate Change. [Unpublished data.]

Dillon, P.J., and Kirchner, W.B. 1975. The effects of geology and land use on the export of phosphorus from watersheds. Water Res. **9**(2): 135–148. doi:10.1016/0043-1354(75)90002-0.

Dillon, P.J., and Molot, L.A. 1996. Long-term phosphorus budgets and an examination of a steady-state mass balance model for central Ontario lakes. Water Res. **30**(10): 2273–2280. doi:10.1016/0043-1354(96)00110-8.

Dillon, P.J., and Rigler, F.H. 1974. The phosphorus–chlorophyll relationship in lakes. Limnol. Oceanogr. **19**(5): 767–773. doi:10.4319/lo.1974.19.5.0767.

Dillon, P.J., and Rigler, F.H. 1975. A simple method for predicting the capacity of a lake for development based on lake trophic status. J. Fish Res. Board Can. **32**(9): 1519–1531. doi:10.1139/f75-178.

Dillon, P.J., Nickholls, K.H., Scheider, W.A., Yan, N.D., and Jeffries, D.S. 1986. Lakeshore Capacity Study — Trophic status. Ontario Ministry of the Environment and Climate Change. Print.

Dillon, P.J., Scheider, W.A., Reid, R.A., and Jeffries, D.S. 1994. Lakeshore Capacity Study: Part I — Test of effects of shoreline development on the trophic status of lakes. Lake Res. Manage. **8**(2): 121–129. doi:10.1080/07438149409354464.

Dove, A., and Chapra, S. 2015. Long-term trends of nutrients and trophic response variables for the Great Lakes. Limnol. Oceanogr. **60**(2): 696–721. doi:10.1002/lno.10055.

Gartner Lee Limited. 2008. Evaluation of remedial strategies for the reduction of algal biomass in Sturgeon Bay: Phase 1 — System characterization [online]. Available from http://archipelago.municipalwebsites.ca/Editor/images/DOCUMENTS/ENVIRONMENT/Sturgeon%20Bay/Remedial_Strategies_Phase%201.pdf.

Grace-Martin, K. 2012. Assessing the fit of regression models. Cornell Statistical Consulting Unit.

Hansson, L. 1992. The role of food chain composition and nutrient availability in shaping algal biomass development. Ecology, **73**(1): 241–247. doi:10.2307/1938735.

Havens, E.H., and Nurnberg, K.N. 2004. The phosphorus–chlorophyll relationships in lakes: potential influence of color and mixing regime. Lake Res. Manage. **20**(3): 188–196. doi:10.1080/07438140409354243.

Hoyer, M.V., and Jones, J.R. 1983. Factors affecting the relation between phosphorus and chlorophyll *a* in midwestern reservoirs. Can. J. Fish Aquat. Sci. **40**(2): 192–199. doi:10.1139/f83-029.

Hutchinson, N.J., Neary, B.P., and Dillon, P.J. 1991. Validation and use of Ontario's trophic status model for establishing lake development guidelines. Lake Res. Manage. **7**(1): 13–23. doi:10.1080/07438149109354250.

Kirchner, W.B., and Dillon, P.J. 1975. An empirical method of estimating the retention of phosphorus in lakes. Am. Geophys. Union. **11**(1): 182–183. doi:10.1029/WR011i001p00182.

Mazumder, A. 1994. Phosphorus–chlorophyll relationships under contrasting herbivory and thermal stratification: predictions and patterns. Can. J. Fish. Aquat. Sci. **51**(2): 390–400. doi:10.1139/f94-040.

McCarthy, F., and McAndrews, J. 2012. Early Holocene drought in the Laurentian Great Lakes Basin caused hydrologic closure of Georgian Bay. J. Paleolimnol. **47**: 411–428. doi:10.1007/s10933-010-9410-z.

Ministries of Ontario (Ministry of the Environment and Climate Change, Ministry of Natural Resources and Forestry and Ministry of Municipal Affairs and Housing) Government of Ontario. 2010. Lakeshore Capacity Assessment Handbook [online]. Available from http://lakes.chebucto.org/TPMODELS/ONTARIO/OME_std01_079878.pdf.

Molot, L.A., and Dillon, P.J. 1991. Nitrogen/phosphorus ratios and the prediction of chlorophyll in phosphorus-limited lakes in central Ontario. Can. J. Fish. Aquat. Sci. **48**(1): 140–145. doi:10.1139/f91-019.

Murphy, J., and Riley, J.P. 1962. A modified single solution method for the determination of phosphate in natural waters. Anal. Chim. Acta, **27**: 31–36. doi:10.1016/S0003-2670(00)88444-5.

Nurnberg, G.K. 1995. Quantifying anoxia in lakes. Limnol. Oceanogr. **40**(6): 1100–1111. doi:10.4319/lo.1995.40.6.1100.

Nurnberg, G.K., and LaZerte, B.D. 2004. Modeling the effect of development on internal phosphorus load in nutrient-poor lakes. Water Resour. Res. **40**: 1–9. doi:10.1029/2003WR002410.

Paterson, A.M., Dillon, P.J., Hutchinson, N.J., Futter, M.N., Clark, B.J., Mills, R.B., Reid, R.A., and Scheider, W.A. 2006. A review of the components, coefficients and technical assumptions of Ontario's Lakeshore Capacity Model. Lake Res. Manage. **22**(1): 7–18. doi:10.1080/07438140609353880.

Pineiro, G., Perelman, S., Guerschman, J.P., and Paruelo, J.M. 2008. How to evaluate models: observed vs. predicted or predicted vs. observed? Ecol. Modell. **216**(3–4): 316–322. doi:10.1016/j.ecolmodel.2008.05.006.

Reckhow, K.H., and Chapra, S.C. 1983. Engineering approaches for lake management. Vol. 1. Data analysis and empirical modeling. Butterworth Publishers, Woburn, Mass. pp. 201–314.

Robertson, W.D., Schniff, S.L., and Pacek, C.J. 1998. Review of phosphate mobility and persistence in 10 septic system plums. Ground Water, **36**(6): 1000–1010. doi:10.1111/j.1745-6584.1998.tb02107.x.

Rowe, J.S. 1972. Forest regions of Canada [online]. Canadian Forestry Service Department of the Environment. Available from http://cfs.nrcan.gc.ca/pubwarehouse/pdfs/24040.pdf.

Sale, D., and Sale, V. 2013. Blackstone Lake water quality testing [online]. For the Blackstone Lake Cottagers Association. Available from http://www.blackstonelakeassn.ca/info/2014_Complete_2103_Water_Quality_Results.pdf.

Sarnelle, O. 1992. Nutrient enrichment and grazer effects on phytoplankton in lakes. Ecology, **73**(2): 551–560. doi:10.2307/1940761.

Schiefer, K., Schiefer, K., and Wiancko, P. 2006. Water quality monitoring report 2006, Township of Georgian Bay [online]. Bluewater Biosciences. Available from http://www.georgianbayassociation.com/Water%20Quality%20Monitoring%20Report-2006.pdf.

Schindler, D.W., Fee, E.J., and Ruszczynski, T. 1978. Phosphorus input and its consequences for phytoplankton standing crop and production in the Experimental Lakes Area and in similar lakes. J. Fish. Res. Board Can. **35**(2): 190–196. doi:10.1139/f78-031.

Sly, P.G., and Munawar, M. 1988. Great Lake Manitoulin: Georgian Bay and the North Channel. *In* Limnology and fisheries of Georgian Bay and the North Channel ecosystems. pp. 1–19.

Sondergaard, M., Jensen, P.J., and Jeppesen, E. 2001. Retention and internal loading of phosphorus in shallow, eutrophic lakes. Sci. World J. **1**: 427–442. doi:10.1100/tsw.2001.72.

Weiler, R.R. 1988. Chemical limnology of Georgian Bay and the North Channel between 1974 and 1980. Hydrobiologia, **163**(1): 77–83. doi:10.1007/BF00026921.

Weller, J.D., Leblanc, J.P., Liskauskas, A., and Chow-Fraser, P. 2016. Spawning season distribution in subpopulations of muskellunge in Georgian Bay. Trans. Am. Fish. Soc. **145**(4): 795–809. doi:10.1080/00028487.2016.1152300.

Wells, M.G., and Sealock, L. 2009. Summer water circulation in Frenchman's Bay, a shallow coastal embayment connected to Lake Ontario. J. Gt. Lakes Res. **35**(4): 548–559. doi:10.1016/j.jglr.2009.08.009.

Zimmerman, A.P., Noble, K.M., Gates, M.A., and Paloheimo, J.E. 1983. Physicochemical typologies of south-central Ontario lakes. Can. J. Fish. Aquat. Sci. **40**(10): 1788–1803. doi:10.1139/f83-208.

Appendix A

This appendix has been included as a part of this manuscript to provide a detailed outline of the intermediate steps used to calculate the Lakeshore Capacity Model (LCM; Paterson et al. 2006; Ministries of Ontario 2010).

Estimate of total phosphorus concentrations

The LCM estimates seasonal mean [TP] for lakes using empirical relationships based on phosphorus (P) budget, watershed hydrology, and basin morphometry (Paterson et al. 2006; see model inputs in Table A1). Equation A1 in Table A2 was used to determine seasonal mean [TP] in water bodies being modelled, where L_T is the total areal P load (mg·m^{-2}) from both natural and anthropogenic sources calculated by dividing the total P load (mg·year^{-1}) by the lake surface area (m^2), R_P represents the P retention coefficient, and q_s represents the areal water load.

Natural phosphorus inputs

The LCM estimates P export (kg·year^{-1}) (eq. A2 in Table A2) from the watershed based on catchment area (km^2) and percentage wetland area (Paterson et al. 2006). We used the Land Information Ontario Wetland Layer to calculate wetland area of each subwatershed and then summed all subwatersheds to determine total wetland area (km^2) for each embayment. In addition to P export from overland runoff, the LCM also takes into account atmospheric deposition of P onto the lake surface. We calculated atmospheric P load by multiplying lake surface area (m^2) by an atmospheric P deposition rate of 16.7 mg·m^{-2}·year^{-1}, calculated as a 17-year mean from three meteorological stations in central Ontario (Paterson et al. 2006).

Anthropogenic P inputs

The Lakeshore Capacity Assessment Handbook (Ministries of Ontario 2010) assumes that all buildings associated with septic tanks situated within 300 m of the shoreline would contribute P to adjoining water bodies if soils in the watershed are thin. Since the soil structure of coastal Georgian Bay is coarse (primarily of sandy loam with good drainage) and bedrock is Precambrian rock at 1 foot (1 foot = 0.3048 m) or less (Canada Department of Agriculture 1960), we have assumed that all septic systems will eventually contribute some P to adjacent waters. We used Google Earth satellite images taken in 2015 to enumerate dwellings (i.e., cottages, trailers, etc.), excluding structures not typically connected to septic systems (e.g., garages, warehouses, sheds, and boathouses). To estimate contribution of anthropogenic P properly from residents, we needed to determine the total number of

Table A1. Sources of information used to apply the Lakeshore Capacity Model (LCM) in this study.

Type	Model component	Value	Source and data type
Source of input	Atmospheric phosphorus deposition rate	16.7 mg·m^{-2}·year^{-1}	Paterson et al. 2006
	Land information Ontario wetland layer		ArcGIS shapefile from Ontario Ministry of Natural Resources and Forestry 2011
	Ontario Hydrology Network — small-scale waterbody		ArcGIS shapefile from Ontario Ministry of Natural Resources and Forestry 2012
	Extended seasonal unit	1.27 capita·year^{-1}	Paterson et al. 2006; satellite imagery
	Seasonal unit	0.69 capita·year^{-1}	Paterson et al. 2006; satellite imagery
	Resort unit	1.18 capita·year^{-1}	Paterson et al. 2006; satellite imagery
	Trailer park unit	0.69 capita·year^{-1}	Paterson et al. 2006; satellite imagery
	Campground–tent trail unit	0.37 capita·year^{-1}	Paterson et al. 2006; satellite imagery
	Youth camp guest	125 g·capita^{-1}·year^{-1}	Paterson et al. 2006; satellite imagery
	Per capita phosphorus contribution	0.66 kg·capita^{-1}·year^{-1}	Paterson et al. 2006; satellite imagery
Retention	Oxic phosphorus settling velocity	12.4 m·year^{-1}	Paterson et al. 2006
	Anoxic phosphorus settling velocity	7.2 m·year^{-1}	Paterson et al. 2006
Hydrology	Runoff estimate	0.400 m·m^{-2}·year^{-1}	Government of Canada, Department of Fisheries and the Environment, Atmospheric Environment Service 1975
	Provincial digital elevation model version 3.0		ArcGIS Raster File from Ontario Ministry of Natural Resources and Forestry 2013

Table A2. List of equations used in the Lakeshore Capacity Model (LCM).

Equation No. and formula		Source
(A1)	$TP_{Mean} = L_T \cdot (1 - R_P) \cdot (0.956 \cdot q_s)^{-1}$	Dillon et al. 1986
(A2)	$TP = \text{Catchment Area} \cdot (0.47 \cdot \%\ \text{Wetland Area} + 3.82)$	Paterson et al. 2006
(A3)	$Q = (A_d + A_o) \cdot \text{Mean Annual Runoff}$	Paterson et al. 2006
(A4)	$R_p = v \cdot (v + q_s)^{-1}$	Paterson et al. 2006
(A5)	$TP_{Outflow} = 0.956 \cdot TP_{Mean}$	Paterson et al. 2006
(A6)	$TP_D = TP_{Outflow} \cdot Q$	Paterson et al. 2006

Note: Eq. A1 estimates seasonal mean total phosphorus concentration (μg·L^{-1}); eq. A2 estimates terrestrial total phosphorus export (kg·year^{-1}); eq. A3 estimates lake discharge (m^3·year^{-1}); eq. A4 estimates phosphorus retention coefficient; eq. A5 estimates total phosphorus concentration (μg·L^{-1}) at lake outflow; and eq. A6 estimates total phosphorus export to downstream lakes (kg·year^{-1})

Table A3. Data used to calculate anthropogenic and natural terrestrial phosphorus loading for the Lakeshore Capacity Model (LCM) for eight embayments and Blackstone Lake.

Site name	Seasonal	Extended	Trailer	Campground	Wetland	Watershed
Blackstone Lake	43	116	0	0	4.68	107.04
Cognashene Lake	70	0	0	0	0.23	4.68
Deep Bay	66	166	0	0	4.28	90.66
Longuissa Bay	6	0	0	0	0.42	5.06
North Bay	130	28	89	0	1.58	10.21
South Bay	54	99	42	0	1.76	9.13
Sturgeon Bay	111	240	0	81	9.52	74.74
Tadenac Bay	3	0	0	0	5.67	50.13
Twelve Mile Bay	0	125	0	0	1.96	12.73

Note: Seasonal = number of primary seasonal residential dwellings; Extended = number of primary extended seasonal residential dwellings; Trailer = number of residential trailer units; Campground = number of campsites; Wetland = wetland area in watershed (km^2); and Watershed = watershed area (km^2).

Table A4. Summary of subcompartments of phosphorus load (kg·year^{-1}) for each water body calculated by the Lakeshore Capacity Model (LCM).

Site name	ATM	Runoff	Upstream	Anthropogenic	Internal	Total
Blackstone Lake	88.79	282.21	233.88	116.81	0.00	721.69
Cognashene Lake	13.39	25.81	0.00	31.88	15.47	85.92
Deep Bay	46.37	533.75	20.09	169.20	56.90	826.31
Longuissa Bay	7.00	38.95	0.00	2.73	0.34	49.02
North Bay	26.19	113.06	0.00	123.20	27.34	289.79
South Bay	23.25	117.61	0.00	126.70	12.25	279.81
Sturgeon Bay	94.75	503.62	82.33	271.50	52.68	1004.88
Tadenac Bay	55.31	246.71	74.41	2.34	0.00	378.77
Twelve Mile Bay	23.80	135.51	0.00	104.78	14.53	278.63

Note: Internal phosphorus loading was calculated independently with Nurnberg and LaZerte's (2004) formula and included as an additional source in the LCM. ATM = atmospheric phosphorus load, Runoff = terrestrial phosphorus load, Upstream = phosphorus load from upstream lakes, Anthropogenic = anthropogenic phosphorus load, Internal = phosphorus load from sediments, and Total = total phosphorus load).

seasonal, extended seasonal, and permanent residences in each embayment. We classified all dwellings with reliable year-round road access as "extended seasonal" and dwellings with no reliable year-round road access as "seasonal" (Table A3). To calculate total anthropogenic P export, the respective seasonal usage values (see Table A1) for each dwelling type were multiplied by the estimated per capita P contribution value of 0.66 kg·capita^{-1}·year^{-1} (Paterson et al. 2006) and then multiplied by the number of corresponding residential units.

Watershed characteristics

We delineated watersheds using ArcGIS 10.3.1 Hydrology Toolset (ESRI, Redlands, California, 2015) and a digital elevation model (DEM; 10 m accuracy) provided by the Ontario Ministry of Natural Resources and Forestry. We then used the Waterbody layer in the Ontario Hydro Network Small Scale geographic information system database to identify all lakes in the watersheds of each embayment with surface area ≥25 ha. We delineated the subwatershed of each lake so they could be modelled individually on a watershed basis. We could not obtain data pertaining to the discharge of each lake modelled, because permanent discharge stations do not exist for all watersheds, and we did not have the resources to collect in situ field measures. In the absence of this information, we used eq. A3 in Table A2 to calculate the discharge as per recommendations by Paterson et al. (2006), where Q represented discharge (m^3·year^{-1}), A_d represented drainage area (m^2), and A_o represented lake surface area (m^2). Average annual long-term runoff estimates for the eastern Georgian Bay region were obtained from a Department of Fisheries and Environment (1975) annual runoff database.

Downstream P input

To model each lake on a watershed basis, we calculated the amount of P flowing from one lake into another by using eq. A6 (Table A2), where TP_D was the P input to the downstream lake, $TP_{Outflow}$ was the estimated [TP] (μg·L^{-1}) at the outflow of a water body, and Q was the lake outflow discharge (m^3·year^{-1}) (eq. A6, Table A2).

Internal loading of P – sampling methodology

To characterize basin morphometry, bathymetric information for each site was collected in situ with a Lowrance Elite-7 HDI fish finder to produce DEMs for all embayments excluding North Bay, South Bay, and Twelve Mile Bay; corresponding information for the latter three embayments were provided by H. Biberhofer, Environment and Climate Change Canada (unpublished data, 2014). All data were imported into ArcGIS 10.3.1 (ESRI, Redlands, California, 2015) to create site-specific DEMs (~5 m accuracy). Vertical profiles of TEMP and DO were taken monthly (May to September) at the deep station to determine the duration and the extent of hypolimnetic anoxia (<1 mg·L^{-1}; DO) in each embayment.

LCM

Terrestrial runoff accounted for the largest estimated source of P input in the LCM and was influenced by the size of the catchment area and the proportion of wetlands in the drainage basin, with Deep Bay having the largest input (533.75 kg·year^{-1}) and Cognashene Lake having the smallest (25.81 kg·year^{-1}) (Table A4). By comparison, the anthropogenic P load was the second largest contributor of P to embayments, with Sturgeon Bay having the largest input (271.50 kg·year^{-1}) and Tadenac having the smallest input (2.34 kg·year^{-1}). Contribution from atmospheric P deposition was generally small and was dependent on the surface area of the water body; Sturgeon Bay had the largest input (94.75 kg·year^{-1}) while Longuissa Bay had the smallest (7 kg·year^{-1}) (Table A4). Calculations of P from sediment remineralization varied considerably among embayments, from a minimum of

0.34 kg·year^{-1} in Longuissa Bay to a maximum of 56.90 kg·year^{-1} in Deep Day (Table A4).

References

Canada Department of Agriculture. 1960. Soil associations of southern Ontario [online]. Available from http://sis.agr.gc.ca/cansis/publications/surveys/on/on30/index.html.

Dillon, P.J., Nickholls, K.H., Scheider, W.A., Yan, N.D., and Jeffries, D.S. 1986. Lakeshore capacity study — trophic status. Ontario Ministry of the Environment and Climate Change. Print.

Government of Canada, Department of Fisheries and the Environment, Atmospheric Environment Service. 1975. Hydrological Atlas of Canada — Annual Runoff. Print.

Ministries of Ontario (Ministry of the Environment and Climate Change, Ministry of Natural Resources and Forestry and Ministry of Municipal Affairs and Housing) Government of Ontario. 2010. Lakeshore capacity assessment handbook [online]. Available from http://lakes.chebucto.org/TPMODELS/ONTARIO/OME_std01_079878.pdf.

Nurnberg, G.K., and LaZerte, B.D. 2004. Modeling the effect of development on internal phosphorus load in nutrient-poor Lakes. Water Resources Res. **40**: 1–9. doi:10.1029/2003WR002410.

Ontario Ministry of Natural Resources and Forestry. 2011. Land Information Ontario File Geodatabase.

Ontario Ministry of Natural Resources and Forestry. 2012. Ontario Hydro Network Small Scale Waterbody File Geodatabase.

Ontario Ministry of Natural Resources and Forestry. 2013. Provincial Digital Elevation Model – Version 3.0.

Paterson, A.M., Dillon, P.J., Hutchinson, N.J., Futter, M.N., Clark, B.J., Mills, R.B., Reid, R.A., and Scheider, W.A. 2006. A review of the components, coefficients and technical assumptions of Ontario's Lakeshore Capacity Model. Lake Res. Manage. **22**(1): 7–18. doi:10.1080/07438140609353880.

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