Water Quality and Remediation Options for Bright Lake, *Pakawagamengan*

Prepared by

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Cover Photo is of 04 Oct 2009 taken by John Milito, showing a cyanobacterial bloom ("bluegreens") in Bright Lake.

Executive Summary

Bright Lake (46°12.9‘, 83°12.5‘, 190 m above surface level) is one of many remote northern Ontario lakes that often experience cyanobacteria blooms (Cover photo) in late summer and fall. In response to a toxic cyanobacterial bloom, half of the 107 residents formed the Bright Lake Association, Inc. (BLA) in 2009, a not-for-profit organization, and commenced collaboration with scientific partners to identify the major causes for these blooms and develop a community-based remedial action plan. BLA applied successfully to the Ontario Trillium Foundation for a Grant to support a thorough limnological evaluation of Bright Lake (Appendix A: Press Release). In particular, the study was to determine the causes of the blue-green algae blooms and recommend remedial options that would improve Bright Lake’s water quality and its results are described in this report. Remediation of Bright lakes appears to be particularly important since it drains via the Bolton River into Mississagi River which is a tributary to Lake Huron.

Bright Lake is relatively shallow (Table 1) compared to its large area which indicates that the exchange between surface and deep water is frequent and extensive. Such lakes mix frequently throughout the summer and are called polymictic. Because of the polymixis any nutrients released from the bottom sediments can directly fertilize the lake water and has to be evaluated in addition to external inputs as internal load.

An extensive monitoring plan has been conducted in late 2009 until Nov 2010 because there were no nutrient data and only limited other limnological observations available. Total phosphorus (TP), which is the most important nutrient to be determined, was so variable that analytical difficulties are suspected. Similarly, the measure of algal biomass, chlorophyll *a*, was patchy and variable, however that is often the case in large polymictic lakes. Consequently future monitoring is recommended to corroborate the results of this study.

Bright Lake’s trophic state was determined as mesotrophic for most of the investigated variables, but the algal biomass indicators were close to eutrophic. Therefore, Bright Lake is at a transitory state and is relatively enriched for a lake on the Canadian Shield, which indicates anthropogenic disturbances. There were no cyanobacterial blooms in summer and fall of 2010, as there were none in other Ontario lakes, probably due to an unusually dry and warm spring and summer.

	Bright Lake	Trophic State
Secchi Disk Transparency (m)	2.2	mesotrophic
Total phosphorus (µg/L)	14	mesotrophic
Total nitrogen (µg/L)	212	oligotrophic
Chlorophyll *a* (µg/L)	8.4	mesotrophic

Occasional anoxia during stratification periods was determined from bi-weekly temperature and oxygen profiles and also indicated enriched conditions and the potential of sediment P release. Nonetheless, Bright Lake entertains a thriving warm water fishery with 17 species measured in a survey in the summer 2010 by the Ontario Ministry of Natural Resources (MNR).

Total external load including precipitation and watershed sources is estimated as 1,630 kg/yr which is similar to an internal load estimate of 1,540 kg/yr. Internal loading, if it can be

corroborated by future measurement, expectedly has an tremendous effect on Bright Lake water quality for several reasons: (1) Timing and location: it is injected into the lake from the sediments in late summer and fall, a season when phytoplankton is often P starved because external input is low. (2) Chemical form: internal load is readily available to plankton because it is released from the sediments as phosphate. In comparison, external P inputs are often combined with particles and are not readily available, but sink to the bottom or are flushed from the lake.

With respect to remediation aspects, external load includes fluxes that cannot easily be controlled, like precipitation and runoff from natural areas; only 31% of the load is from anthropogenic sources (lake shore, agricultural and grass land) and may be manageable. Consequently, diminishing internal load is most desirable. Internal load is unexpected but important in relatively pristine lakes on the Canadian Shield, such as Bright Lake, because recent anthropogenic nutrient enrichment of the sediments can fertilize the lake in the late summer and fall inducing cyanobacterial blooms.

This study presents preliminary indications that lake levels, as influenced by flows and beaver dams and their specific locations and times, may contribute to the water quality variation in Bright Lake. At higher lake levels, longer periods of stratification prevailed and provoked cyanobacterial blooms observed in two years (2008, 2009), while there was no bloom at the lower level of dry 2010. This sequence of events indicates that possibly internal load is the driver of the water quality problems, as sediment release is enhanced under anoxic stagnant conditions that coincide with higher lake levels (1996, 2008, and 2009).

This study also shows that lake level is significantly correlated with inflows, for which long-term observations are available. Any water quality during this time can be compared to flows, which explains 35% of the lake level variation. It is hoped that readers of this report may remember past water quality conditions and provide additional observations for previous years of algal blooms so that the hypothesis expressed here can be tested.

Based on the available information, several recommendations are made in this report. Because the most apparent water quality issue conflicting with lake use and health is an overabundance of algae, the control of algal growth and especially cyanobacterial blooms should be attempted. The most common method is to reduce the nutrient inputs (phosphorus, in particular), as most excessive algal growth is the result of fertilization from external sources like agriculture, field and lawn runoff, septic and sewage outflows. Specifically in the Bright Lake watershed, septic system inspection and renovation, and agricultural and shoreline best management practices are recommended. In addition, a long list of direct treatment involving measures to decrease internal loading from the sediments are examined, but all are deemed unfeasible and costly because of Bright Lake's size and remoteness. It appears that the only feasible remediation option may be the management of water flow by optimally adjusting seasonal water levels to prevent prolonged stratification and internal loading. Such management would involve the monitoring and eliminating of beaver induced water retention.

In preparation of such management it is recommended to

- Continue future monitoring to corroborate the flow-water quality dependency
- Accomplish a detailed hydrological study
- Examine the potential effect on marginal zones such as wetlands and fish spawning areas
- Examine the feasibility of the restoration technique of *dilution and flushing* using Harris Creek water under particular consideration of Basswood Lake water quality.

A lake shore capacity assessment for Bright and Basswood Lakes commissioned by the Town of Huron Shores (Nürnberg and LaZerte 2011) contains relevant information and supplements this report.

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1 Introduction

Bright Lake (46°12.9', 83°12.5', 190 m above surface level) is one of many remote northern Ontario lakes that often experience cyanobacteria blooms (Cover photo) in late summer and fall. Local concerns about the implications for human and ecosystem health were reinforced when a surface bloom sample from fall 2008 contained a high abundance of the potentially toxic cyanobacterium *Aphanizomenon flos-aquae* that triggered prolonged beach closures.

In response, half of the 107 residents formed the Bright Lake Association, Inc. (BLA) in 2009, a not-for-profit organization, and commenced collaboration with scientific partners to identify the major causes for these blooms and develop a community-based remedial action plan. BLA applied successfully to the Ontario Trillium Foundation for a Grant to support a thorough limnological evaluation of Bright Lake (Appendix A: Press Release). In particular, the study was to determine the causes of the blue-green algae blooms and recommend remedial options that would improve Bright Lake's water quality. Remediation of Bright lakes appears to be particularly important since it drains via the Bolton River into Mississagi River which is a tributary to Lake Huron.

For this project, any available reports, surveys, documented communication, photos and local wisdom were inspected and the relevant information is presented here. The main part of this report presents the analysis of water quality data collected in the monitoring effort 2009-2011 and the evaluation based on previous and 2010 data.

Almost no historical data are available, but preliminary investigation identified that excessive nutrient input, in particular two main phosphorus (P) sources may facilitate these blooms: agricultural input from streams and internal load released from anoxic bottom sediment surfaces.

A lake shore capacity assessment for Bright and Basswood Lakes commissioned by the Town of Huron Shores (Nürnberg and LaZerte 2011) contains relevant information and supplements this report.

2 General Bright Lake characteristics

Bright Lake is located in Day and Bright Townships and belongs to the Municipality of Huron Shores, about 30 km south east of Sault Ste. Marie in the Algoma country. Its outlet, the Bolton River drains via the Mississagi River into Lake Huron (Figure 1).

Figure 1. Satellite view of Bright Lake, from Google Map

2.1 Morphometry

Bright Lake is relatively shallow (Table 1) compared to its large area. Its morphometric index (mean depth/square root of surface area) is a low 1.4 m/km and indicates that the exchange between surface and deep water is frequent and extensive. Such lakes mix frequently throughout the summer and are called polymictic.

More than half of the Bright Lake's water resides in the upper 3 m (Table 2,Figure 1). Only 1% is in the "deep hole" below 9 m, off to the south-western shore, where the main sampling station is located.

Table 1. Morphometric lake characteristics

Characteristics	Value
Surface Area, Ao (km^2):	12.32
Maximum Depth, zmax (m):	12.19
Mean Depth, z (m):	4.91
Volume (10^6 m^3):	60.45
Annual flushing rate (per yr):	1.43
Annual water load (m/yr):	6.99
Max Length (km):	8.80
Max Width (km):	2.60
Perimeter (km)	27.46
Morphometric Index ($z/A^{0.5}$):	1.40

Table 2. Layer-morphometry based on GIS mapping (Ray Lipinski, MNR, 6 Dec 2010)

Depth	Area at upper depth	Volume interval	
(m)	(m^2)	(m^3)	(%)
0 - 3.05	12,321,254	32,495,007	54%
3.05 - 6.1	9,083,051	21,676,151	36%
6.1 - 9.1	5,308,119	6,281,478	10%
9.1 - 12.2	110,069	335,491	1%
Total	**12,321,254**	**60,452,636**	**100%**

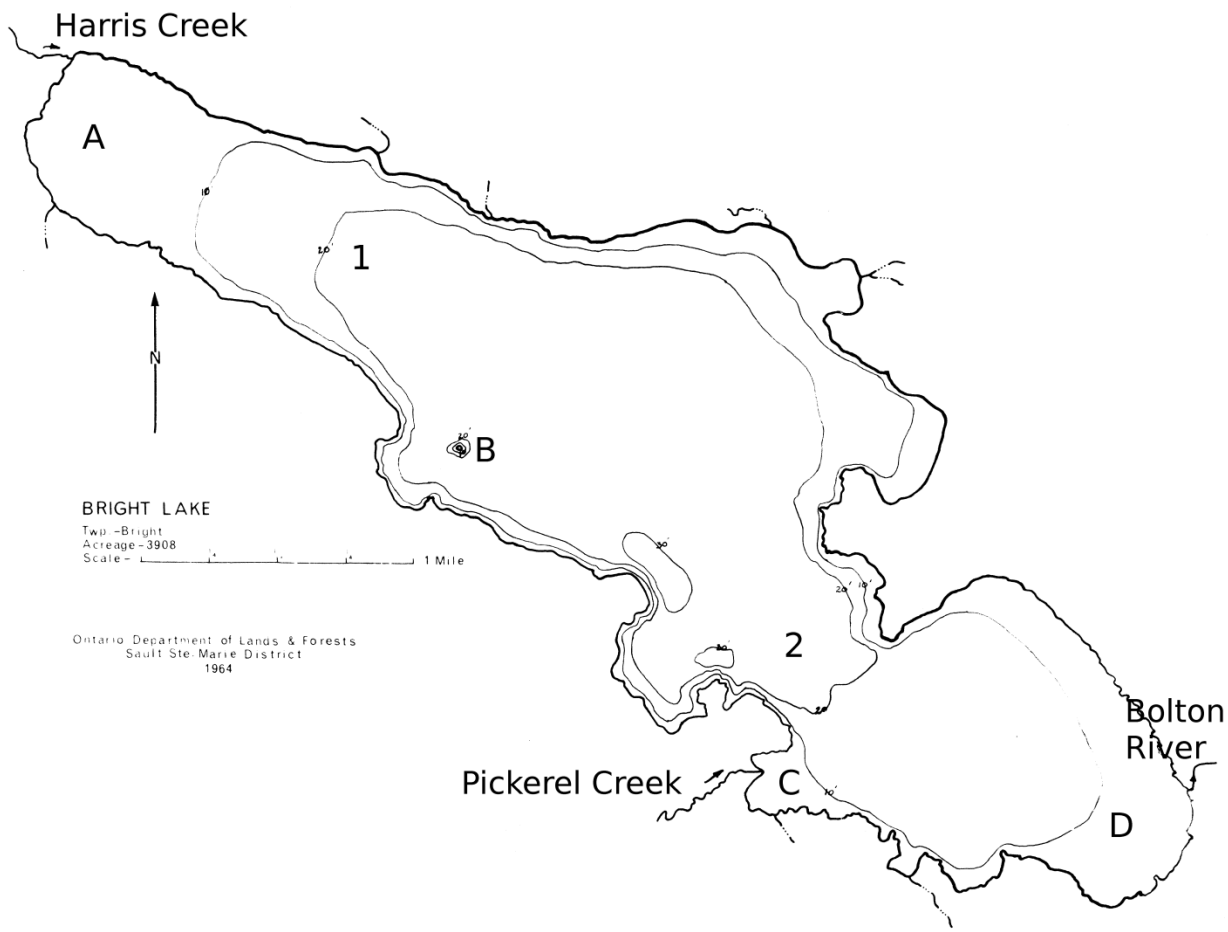

Figure 2. Bathymetric map of Bright Lake indicating sampling sites for the present and a previous study (Table 4)

2.2 Watershed

The watershed or catchment basin has a large influence on a lake's water quality, because most of the pollutants are flushed with the runoff to end up eventually in the lake. The area ratio between watershed and lake is 14 which is high and indicates that external load should have a large influence on Bright Lake (Table 3, shaded area).

Almost half of the Bright Lake watershed consists of the upstream Basswood Lake with its catchment basin (Table 3). Basswood's outflow is Harris Creek that flows into the north eastern tip of Bright Lake (Figure 2). Harris Creek conveys extremely clear and high quality water to Bright Lake, because Basswood's water is almost pristine with spring TP averages close to 3 µg/L (Nürnberg and LaZerte 2011). Consequently, the remainder of the watershed must contribute a much higher part of pollutants, in particular phosphorus, to make up the much larger concentration in Bright Lake (Section 5). Most of this remaining part consists of the Pickerel Creek watershed (30%, Table 3) and the immediate area around the lake (15%). More detailed watershed areas are presented in Appendix B.

Table 3. Bright Lake sub-watersheds *(MNR 26 April 2010, adjusted)*

Description	Area	
	km^2	%
Basswood Lake	27.07	16%
Basswood watershed to inflow of Bright	60.89	35%
Not accounted for	10.69	6%
Pickerel	52.67	30%
Southern shoreline	11.76	7%
Northern Shoreline	10.68	6%
Total Watershed w/o Bright Lake (A_d)	173.76	100%
Bright Lake (A_o)	12.32	
Ratio of A_o/A_d	14.1	

3 Water quality evaluation - Methods

3.1 Sampling locations and times

Seven sampling sites were established during the course of the study, four within the lake along its axes that were routinely sampled for water quality (Figure 2, A-D, Table 4) and three in the creeks that were sampled once on 2 May 2010. (The Harris Creek sample was taken at the Bridge on Dayton Rd., the Pickerel Creek sample at the most easterly bridge on Dayton Rd. and the Bolton River sample at the bridge on Bolton River Rd.). Sampling for water was either done as grab samples from the surface or by Van Dorn for discrete depths (Figure 3).

Table 4. Sampling location key and approximate depth

Location	Site	Water Depth (m)	Sampling depth (m)	# of Chemical Analysis	Year of sampling
Harris Creek Inflow Bay	A	2	1	6	2009-11
Main Station	B	11	1,3,5,7,8,9,10	9	2009-11
Pickerel Creek Inflow Bay	C	2	1	6	2009-11
Bolton River Outflow Bay	D	2	1	6	2009-11
North-west Basin Edge	1	6-7			1996
South-east Basin Edge	2	9			1996

The monitoring effort started on 24 Aug 2009 taking temperature and dissolved oxygen profiles. Such readings were obtained on 4 occasions in 2009 and on a further 11 occasions in 2010. Water was once analyzed on 17 Oct 2009 and routine sampling started on 2 May 2010 on a schedule of biweekly to monthly sampling until 7 Nov 2010. All sampling was done by volunteers, namely John and Peggy Milito with occasional support, especially for the more difficult task of sediment coring, by other members of the Bright Lake Association. The exact sampling dates can be seen in the tables presenting the various results.

To determine the contribution of the bottom sediments to water quality issues, sediment cores were collected by John Milito and helpers at the main station (B) on 4 Oct 2009 and 6 Sep 2010. At the latter date sediment was also collected from the open basin station (Table 5). Core samples from the same location and depth were combined and mixed to obtain more representative samples for each location. Sediment was analyzed by *Spectrum Analytical* (http://www.spectrum-analytical.com) for sediment TP, reductant soluble P (Fe-P) and general chemical composition.

Table 5. Sediment sampling location and depths

Location	Depth (m)	# of Cores	
		2010	2009
Main Station, Site B	11-12	3	1
Central Basin	4-5	2	-

Earlier sampling included two stations at the edges of the main basin, where temperature and DO profiles were measured (Blewett and Goold 1996).

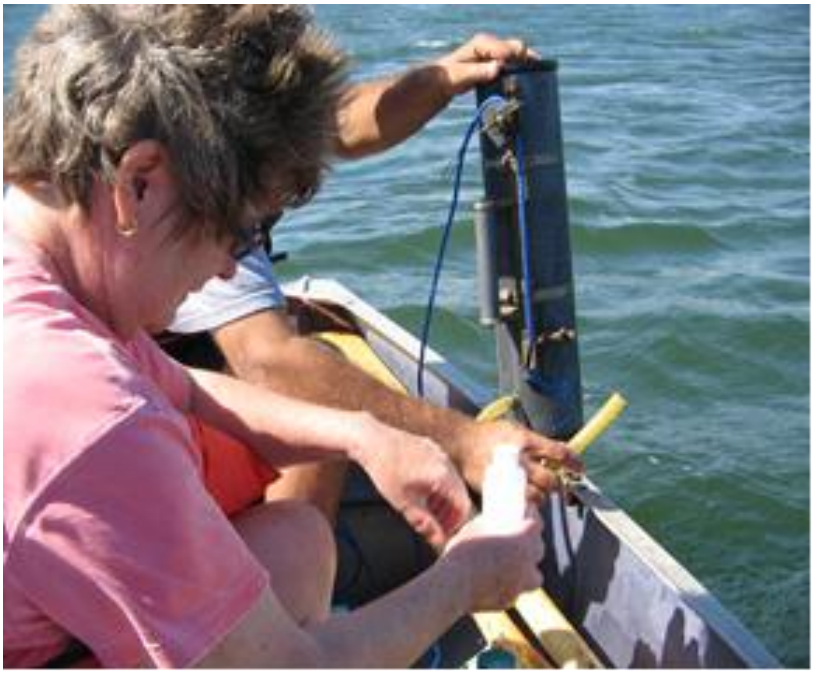

Figure 3. Obtaining water from different depths with the Van Dorn sampler

(Current and Past BLA presidents: Sharla Shine and John Milito; Photo: Peggy Milito)

3.2 Field and analytical methods

Temperature and dissolved oxygen (DO) profiles at 1 m depth intervals were taken with Hanna Instruments, HI 9146, Portable Waterproof Microprocessor DO Meter borrowed from the Ironbridge MNR (2009 to May 2010) and starting June 2010 with a purchased YSI 550A DO meter.

Water samples were collected by a discrete depth sampler (Van Dorn, Figure 3) at 0.5 m depth to determine the nutrient concentration of phosphorus (P) and nitrogen (N).

Until Nov 2010 all samples were analyzed with standard methods (mostly those of the American Public Health Association, APHA) by Testmark Laboratories Ltd. (www.testmark.ca). Starting Nov 2010, all analysis was done in the Trent University Laboratory at the Dorset Environmental Science Centre with Ministry of the Environment (MOE) certified methods (including: MOE Methods 3036_2007 E3374_2007, E3424_2008). Detection limits for the different labs are compiled in Appendix C.

Occasionally, total phosphorus (TP) was analyzed in duplicates because of the high possibility of contamination. However, it appeared that underestimation was a larger challenge than contamination. Analytical detection limits were 2 µg/L TP for Testmark and 0.2 µg/L for the Trent University Laboratory. On 15 Aug and 10 Oct 2010 several samples are below detection limit, which appears unrealistic. Subsequent re-analysis was unsuccessful to determine whether those very low values are real or due to an analytical interference problem.

A Secchi disk reading was taken as a measure of transparency on sampling trips 12 May to 7 Nov 2010 at several locations. Chlorophyll *a* was analyzed occasionally as well. It is the green pigment of phytoplankton and serves as an estimate of algal biomass. Often chlorophyll

concentrations are quite patchy and variable in space and time and frequent Secchi transparency readings may be superior in determining algal blooms.

General chemical composition of the water was determined twice, once in the spring (2 May 2010) and once in the fall, when the water was well mixed (7 Nov 2010).

Kits by Abraxis (http://www.abraxiskits.com) that can determine the presence of the cyanobacteria toxin, microcystin, were available but not used because there were no apparent blooms.

3.3 Trophic state classification

Based on several water quality variables a lake can be classified with respect to its trophic state (Table 6). Clean pristine and clear lakes are called oligotrophic and have high Secchi disk transparency, and low nutrient and algae concentrations, while lakes with more nutrients and algae are intermediate and called mesotrophic or eutrophic. Only lakes that have a high nutrient load from the watershed and from the sediments are hyper-eutrophic, showing extremely high nutrient and algae concentrations, high turbidity and exhibit an oxygen deficit (below saturation) in their bottom waters when conditions are stagnant. In shallow and relatively fast flushing lakes (polymictic lakes), such as Bright Lake, stagnant conditions are rare, prohibiting extended oxygen deficits even though the trophic state may be high.

As shown in the following section, Bright Lake's trophic state indicates mesotrophic conditions for most of the investigated variables. Mesotrophy is an intermediate state that is relatively enriched for a lake on the Canadian Shield and indicates anthropogenic disturbances.

Table 6. Trophic state categories based on summer water quality (Nürnberg 1996)

	Bright Lake*	Oligotrophic	Mesotrophic	Eutrophic	Hyper-eutrophic
Secchi Disk Transparency (m)	2.2	> 4	2 – 4	1 – 2	< 1
Total phosphorus (µg/L)	14	10	10 – 30	31 – 100	> 100
Total nitrogen (µg/L)	212**	< 350	350 – 650	650 – 1 200	> 1 200
Chlorophyll *a* (µg/L)	8.4	< 3.5	3.5 – 9	9.1 – 25	> 25
Anoxia in polymictic lakes	occasional	none	none	occasional during summer stratification	

Note: *For explanation see Section 4 Limnology and water quality of Bright Lake
**TN was analyzed only once, on 7 Nov 2010

Algal growth in lakes is usually limited by the supply of phosphorus. Even if other nutrients such as nitrogen, or light become limiting, algae biomass and blooms usually increase with increasing phosphorus concentrations in the water. Increasing the mass of phosphorus entering a lake or pond (loading) will increase the average concentration of phosphorus and consequently of algae, increasing eutrophication as well.

Nitrogen is the second most important nutrient in lakes and reservoirs, after phosphorus. In fact, it often co-limits algal growth, so that any addition of available nitrogen compounds enhances algal growth and eutrophication. Total nitrogen (TN) and total phosphorus (TP) concentrations are often closely correlated, but generally algae biomass (expressed as the green pigment,

chlorophyll *a* concentration) is better correlated with TP rather than TN. For this reason and because phosphorus can more easily be controlled than nitrogen, management and restoration efforts typically concentrate on the reduction of phosphorus.

4 Limnology and water quality of Bright Lake

Ice-out was unusually early in Central Ontario and was 3 April 2010 on Bright Lake. Therefore the summer period was defined as the period with the sampling events between 2 May and 6 Sep.

4.1 Temperature and dissolved oxygen concentration

Temperature profiles indicate that Bright Lake is usually well-mixed to about 5-6 m at the deep station (B). Even though temperature differences are not large (typically less than 4 C between surface and bottom) and therefore stratification periods probably short, incidences of hypoxia developed throughout the summers of 2009 and 2010 (Figure 4, Appendix D).

DO profiles especially in summer 2010 indicate sub-saturation concentrations throughout the water column (< 6 mg/L). These results indicate the relative difference between upper and lower layers and a decrease of DO concentration with depth. Such a pattern is usually created by a severe sediment oxygen demand which is consistent with the overall low DO concentration and likely increased because of relatively high temperatures also in the deeper water (19-20° C during hypoxia in 2009 and 20-22° C in 2010). Occasional stratification is indicative of polymictic lakes and is typical for shallow lakes such as Bright, as supported by the morphometric index (Table 1). Occasional hypoxia is representative of mesotrophic conditions in polymictic Bright Lake (Table 6).

Bright Lake is categorized as entertaining warm water fisheries so that the PWQO of 5 mg/L DO applies. DO concentration was below 5 mg/L at several occasions in 2009 and most of the summer 2010 at and below 8 m. Severe hypoxia starting at 6 m was measured on 18 July 2010. This means that 11% of Bright Lake (Table 2) was not acceptable to warm water biota during that period. In several previous springs, fish kill was observed in the south eastern portion of the lake (Section 4.9, John Milito, pers. comm.) and may indicate hypoxic conditions under ice.

At four occasions DO was also measured at sites A, C, and D. As expected, no sign of hypoxia was found because of the shallow depths of 1 and 2 m and concentrations between 7 and 10 mg/l were measured.

From a study in 1996 (Blewett and Goold 1996) five profiles for two stations at the basin edges (Table 4) are available between 10 Jun and 22 Jul (Appendix D). Hypoxia was intense especially at the deeper station (Station 2, Figure 2) and started at 2 m on 10 Jul, until the end of the observation period on 22 Jul (Figure 5). This means that more than 65% of the lake's volume was problematic for the warm water fishery. It is interesting to note that such low DO concentration occurred despite relatively cold summer water temperatures (16-18 C). One reason for the high anoxia in 1996 may be higher lake levels at the onset of the warm season due to extreme high inflow in May (maximum recorded since 1978 and 2.5 times the long-term average, Section 6) and later in July and August.

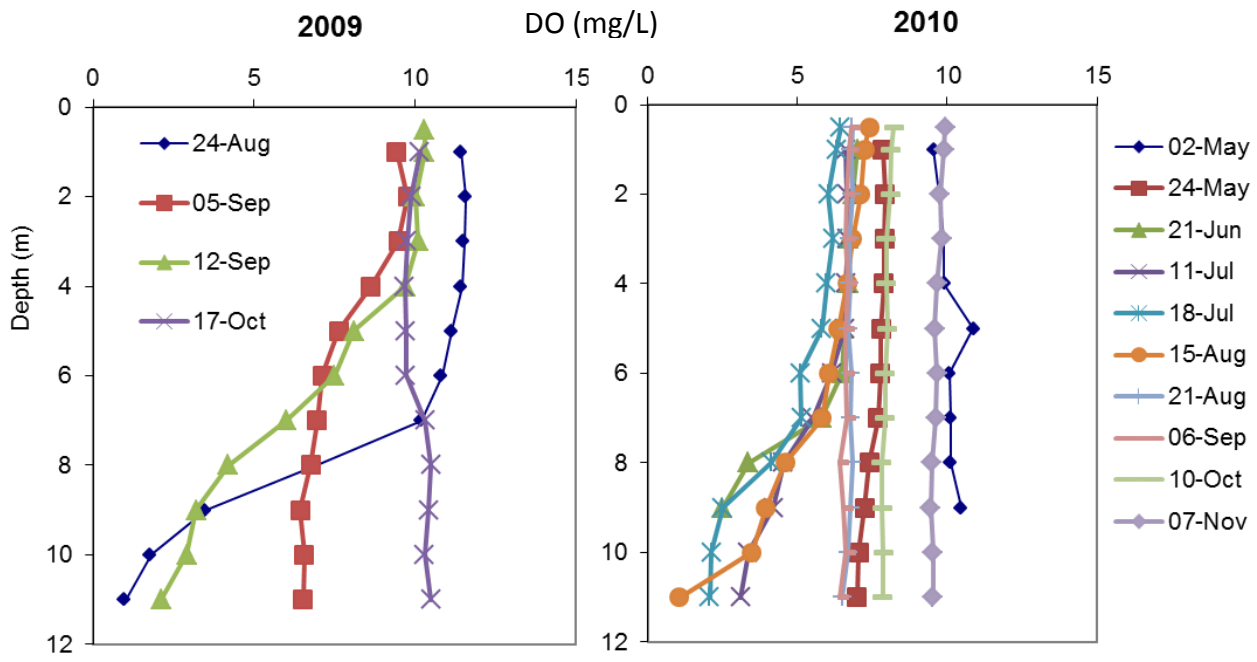

Figure 4. Dissolved oxygen profiles for the study period

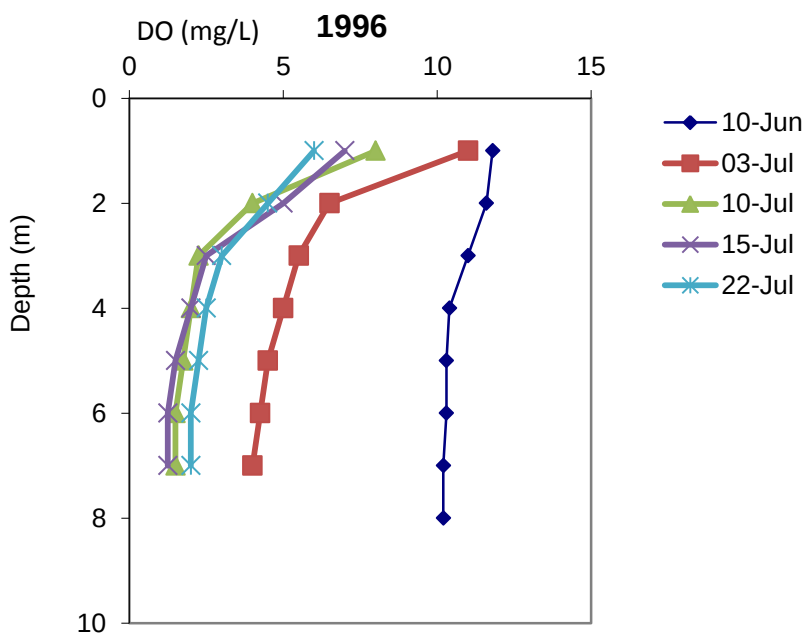

Figure 5. Dissolved oxygen profiles for 1996

Using the DO profiles for computing anoxic factor (AF), Bright Lake has a typically low AF because it is polymictic and mixes occasionally throughout the summer. AF in 2010 was estimated as 1 d/summer, but AF was probably much higher in 1996 if the anoxia that was measured in July persisted through the summer (Appendix E).

4.2 Apparent water quality, water clarity measured as Secchi disk transparency

Water quality in Bright Lake was comparably good in the growing season of 2010 as there were no cyanobacterial (bluegreen) blooms. Summer average Secchi transparency at the main station was 2.2 m and decreased in the fall (Table 7, Figure 6). Secchi disk readings were never below 1m, which is the Ontario guideline for contact sport. However, Bright Lake is relatively turbid considering its location on the Canadian Shield and its trophic state has to be classified as almost eutrophic (at 2 m and below) with respect to Secchi transparency (Table 6).

Table 7. Secchi disk transparency in 2010

Station	A	B	C	D
Secchi (m):				
24-May-10		2.5		
21-Jun-10		4.0		
11-Jul-10		2.5		
18-Jul-10		1.5		
25-Jul-10		2.5		
15-Aug-10		1.5		
21-Aug-10		2.0		
06-Sep-10		1.0		
10-Oct-10	2.0	1.25	1.5	1.75
07-Nov-10	1.5	1.5	1.25	1.5
Average		2.0		
Summer average		2.2		

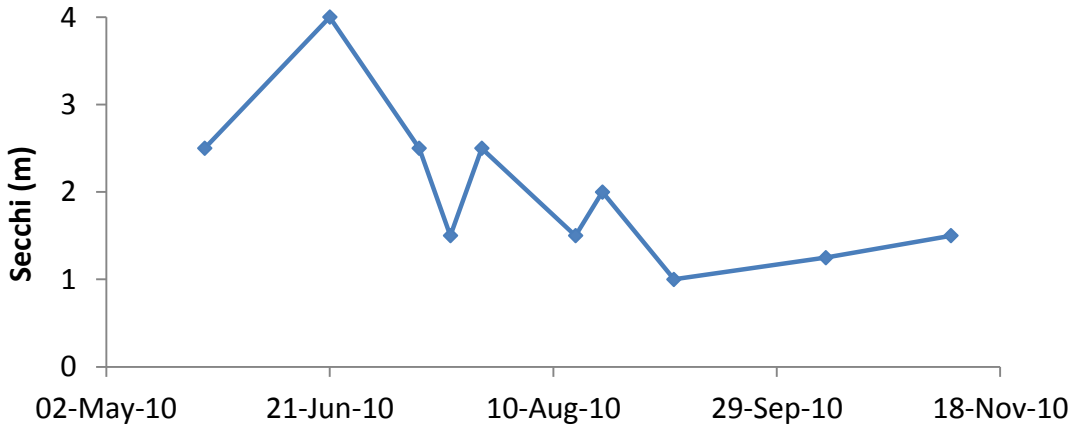

Figure 6. Secchi disk transparency at the main station in 2010

In comparison, one Secchi reading taken during the MOE Lake Partner Program on 29 May 2009 had a value of 2.9 m. In 1996 the average of the 6 week period 10 Jun-22 Jul, measured at two stations, was 2.6 m (Appendix F), also indicating mesotrophic conditions. In polymictic lakes water quality and especially transparency due to phytoplankton often decreases throughout the growing period as it did in Bright Lake 2010 (Figure 6), so that the 1996 records may not be representative for conditions throughout this period.

4.3 Phytoplankton and cyanobacteria blooms

Chlorophyll *a* was quite variable as often found in polymictic lakes, but levels were usually below bloom concentrations (below 20 µg/L). At the main station (B) the average would indicate almost eutrophic conditions because of one high value in July. This bloom was identified as mainly composed of a filamentous green alga, *Spirogyra* (Appendix G). A small amount of cyanobacteria was detected as well, but no toxicity.

Table 8. Chlorophyll concentration in summer 2010

Station	A	B	C	D
02-May-10		3.3		
11-Jul-10	2.6	18.5		
15-Aug-10	1.7	3.0	7.8	7.9
06-Sep-10		3.7		
Summer Average	**2.2**	**8.4**	**7.8**	**7.9**

While there are no earlier chlorophyll data available, there is evidence of at least two years with cyanobacterial blooms. October blooms were experienced in 2008, when the filamentous cyanobacterium *Aphanizomenon flos-aquae* was identified by the MOE (Appendix G) and in 2009, when blooms were photographically documented (Title Photo).

4.4 Nutrient concentration

Different compounds of nutrients were analysed to determine (a) their possible sources and (b) their impact on nuisance algal blooms. The nutrients that increase algal growth and biomass are typically phosphorus and nitrogen. Most temperate lakes are P limited and it can be expected to be the case for Bright Lake. Consequently most sampling effort was placed on P. Nonetheless some N compounds were occasionally analyzed as well, because of its inverse relationship with the occurrence of cyanobacterial (bluegreens) blooms. In particular, inorganic nitrogen concentration (nitrate-nitrite and ammonia concentrations) can decrease to below 50 µg/L levels while bluegreens proliferate. Because several cyanobacteria species have the ability of fixing nitrogen that they need for their growth they can outcompete other phytoplankton at these conditions.

The most important trends are discussed and presented in this section, while individual results are provided by electronic spreadsheets.

4.4.1 Total Phosphorus

There were only two values of total phosphorus concentration available before the initialization of this study (Table 9). Together with the monitoring data of this study, they indicate an increasing trend of TP throughout the growing season in 2009 from 11 to 25 µg/L. Such an increase was not as pronounced in 2010, because there were occasional low values. It is unlikely that values below 3 µg/L (with an analytical detection limit of 2 µg/L) are real; they may indicate an analytical interference problem. If the low values of 15 Aug were not considered, there would be an increase from 8 to 19 µg/L TP between 2 May and 6 Sep.

Table 9. TP concentration in the lake stations of surface water layer (1m)

Sampling Date	TP (µg/L) for Station A	B	C	D
2009				
29-May-09*		**11**		
19-Jul-09*		**15**		
17-Oct-09	19	**25**	19	19
Total average		**17**		
2010				
02-May-10	12	**8**	11	8
24-May-10		**11**		
21-Jun-10		**11**		
11-Jul-10	13	**14**	26	14
15-Aug-10	<2	**3**	<2	10
06-Sep-10		**19**		
10-Oct-10	<2	**<2**	<2	<2
07-Nov-10**	2	**16**	14	<1
Averages for TP >3 µg/L				
Total average	**12**	**13**	**17**	**11**
Summer average	**13**	**14**	**26**	**12**
Averages for all results				
Total average	6	10	11	7
Summer average	7	12	14	12

Note: *Collected within the Ontario MOE Lake Partner Program
** analyzed in the Trent University Lab

With respect to phosphorus, Bright Lake can be classified as mesotrophic in 2010, because the average TP summer concentration of the main station (B) was 14 µg/L if the abnormally low concentrations are excluded (Table 6), otherwise it would be classified as border-line oligotrophic. The TP average was higher in 2009 at 17 µg/L indicating mesotrophic conditions. In these two years the provincial water quality objective for lake water of 20 µg/L summer average TP (Ministry of Environment 1994) has not been exceeded.

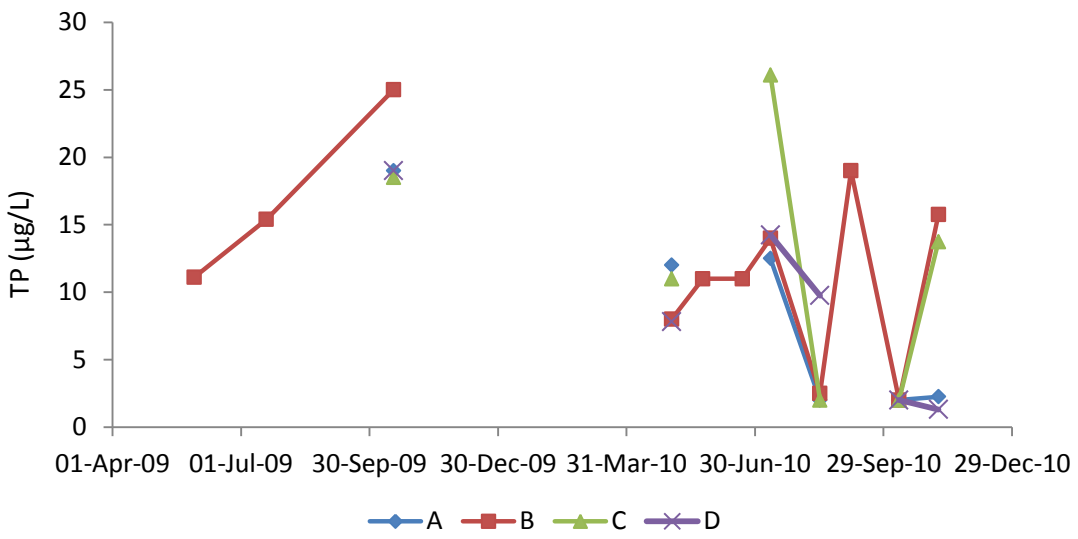

Figure 7. All available lake TP results in the mixed surface layer (with B, main deep station).

TP concentration at the other stations fluctuated, but was usually similar to the concentration at the main station (B), except for an occasional high P input at Pickerel Creek inflow like that on 11 July (Station C). A higher frequency and precision (repeat samples) of monitoring would be needed to reliably determine the importance of the various inflows with respect to P loading.

TP profiles and deep water samples were taken at 9 dates to look for any indication of P release from the bottom sediments. In a polymictic lake like Bright Lake, the possibility of sampling during a stratified period is slim, but at least at two occasions the lower depth samples were higher than the surface samples indicating the possibility of sediment P release on 11 Jul and 15 Aug. A slight period of stratification 21 Jun-15 Aug was also indicated by temperature and DO profiles (Appendix D). Starting 21-Aug the profiles indicate complete mixing, which may explain the high TP values throughout the water column on 6 Sep. However the extreme TP value at 5m (75 µg/L) is hard to take for real and may be due to contamination. Similarly, the values of below detection limit measured at five depths on 10 Oct are unrealistic. Further monitoring in duplicates is advised to determine whether such low values are caused by an adsorption mechanism, an analytical interference or an artefact of the specific lab.

Table 10. TP concentration at the main station (B) for different depths

	TP Concentration (µg/L)							
Depth (m):	1	3	5	7	8	9	10	0-10*
17-Oct-09	25	24				17	15	
02-May-10	8	31				6		
24-May-10	11	10			9		13	
21-Jun-10	11	9	8			11		9.2
11-Jul-10	14	8	15	12		25		11.4
15-Aug-10	3	8	3	12		20		
06-Sep-10	19	30	75	21		19		34.0
10-Oct-10	2	2	2	2		2		
07-Nov-10	16		16					16.0
Average > 3 µg/L	15	17	28	15	9	16	14	
Average	12	15	20	12	9	14	14	

*Volumetric average based on morphometry of Table 2 computed for profiles with TP>3 µg/L

4.4.2 Nitrogen compounds: nitrate and nitrite, total N

Nitrate concentration turned out to be below the detection limit of the Testmark Laboratories (100 µg/L) at all dates when it was determined (17-Oct-09, 02-May-10, 21-Jun-10) except on 21 June in the main station (B), when it was determined as 100 µg/L. The Nov 7 samples analyzed in the Trent University laboratory at a lower detection limit of 2 µg/L all were below 50 µg/L.

Nitrite that was analysed accidentally instead of nitrate was expectedly below the detection limit of 30 µg/L for all stations and dates (17-Oct-09, 02-May-10, and 10-Oct-10).

Ammonia concentration was expectedly low for these stations, between 7 and 14 µg/L.

Total nitrogen, computed as the sum of total Kjeldahl and nitrate/nitrite fractions was measured once, on Nov.7. It was lowest close to the Harris Creek inflow at 167 µg/L and averaged 212 µg/L at the main station for 1 and 5 m depths. If such low concentration prevailed throughout the summer they represent oligotrophic conditions (Table 6).

4.5 General background chemistry

The ions bromide, fluoride, phosphate and iron were measured on 17 Oct 2009 at the main station (B) and, not surprisingly, were under the detection limit, same as total suspended solids (TSS). Consequently they were not determined again.

Further samples taken on three occasions of thermal mixis (17 Oct 2009, 10 Oct 2010 and 7 Nov 2010) at four stations indicate that Bright Lake is a well-buffered lake that is not acid stressed as other lakes on the Canadian Shield. Its pH is slightly above neutral, alkalinity is high at above 30 mg/L as $CaCO_3$ and calcium and sulfate concentrations slightly higher than typical for the Canadian Shield (Table 11). Chloride, sodium, magnesium and potassium are at normal levels and do not display any indication of road salt effects. Dissolved organic carbon (DOC) and colour values indicate that Bright Lake is a generally unstained lake which is clearest at its inflow from Harris Creek (Table 11). The above neutral pH value also confirms that the lake is

probably not humic or fulvic acid stained despite some organic-rich, tea-stained inlets. The colour value of the main station is just above what is usually recognized as clear (10 TCU) and may be influenced by some of the stained inlets.

Table 11. Chemistry of the routine stations in Bright Lake

	Calcium	Chloride	DOC	Sulfate	K	Mg	Na	Alkalinity*	True Colour	pH
				(mg/L)					(TCU)	
Harris Creek Bay	5.8	6.1	1.8	6.7	0.6	2.5	3.9	17.2	nd	7.0
Main Station (B)	7.9	5.1	5.8	5.5	0.6	2.4	3.8	32.3	10.3	7.4
Pickerel Creek Bay	7.9	6.1	4.2	6.7	0.6	2.4	3.8	33.7	7.9	7.6
Bolton River Bay	7.9	6.4	5.4	6.8	0.6	2.5	3.9	31.6	5.0	7.5

*Alkalinity as CaCO3 (pH 4.5)

4.6 Water quality of Bright Lake – Summary and Conclusions

Bright Lake's water quality does not fit the general and expected trophic state relationships with respect to the five tested variables (Table 6). Variables of algal biomass (Secchi transparency and chlorophyll) indicate mesotrophic to almost eutrophic conditions; phosphorus indicates mesotrophic conditions, and nitrogen, if the fall value is representative of summer conditions, oligotrophic conditions. The occurrence of cyanobacteria blooms in the past indicates eutrophic conditions, although there were no toxic cyanobacterial blooms in Bright Lake in 2010. Dissolved oxygen profiles are difficult to interpret in polymictic lakes, but occasional stratification with simultaneous hypoxia was found in summer 2010 at the deep station (B), indicating meso- to eutrophic conditions.

This inconsistent display of trophic state variables may mean that Bright Lake is in transition. Based on its location on the Canadian Shield and its low-nutrient input from the outflow of oligotrophic Basswood Lake, it may have been at one time oligotrophic with clear transparency and low algal biomass. The low observed nitrogen concentration supports this theory. Anthropogenic nutrient input from its watershed may have induced an increase in phosphorus, associated algal biomass and cyanobacterial blooms. During this process nutrient-rich sediments accumulated and started to release P, when organic-rich runoff increased the oxygen demand in the sediment.

To address the potential watershed impact the Township of Huron Shores commissioned a lake shore capacity study for Bright and Basswood Lake (Nürnberg and LaZerte 2011).

The water quality in the summer of 2010 was unusually good in many Ontario lakes. In particular, there were no large cyanobacterial blooms in lakes that often exhibit them, including such Algoma Lakes as Desbarats Lake in Johnston Township, and ponds in Southern Ontario (Gertrud Nürnberg, unpublished data). The reasons are not clear, but an extreme weather pattern may be responsible. Snow melt and ice-out were extremely early and a warm spring was followed by a relatively dry summer everywhere in Ontario. Precipitation was below average until mid-July and air temperature was above average most of the year. Such climatic conditions

would have influenced flows and lake level, so that lower lake depth may have prevented long periods of stratification and anoxia as discussed in detail in Section 6.

4.7 Water quality indication of the creeks

While the main goal of this project is the assessment of the water quality of Bright Lake, the main inflows reflect the watershed and contribute to Bright Lake's pollutant loading. To properly assess these fluxes, continuous or frequent discrete sampling of water and nutrient would be necessary, but that was beyond this project. In an attempt to obtain at least some information about their different contributions to the water quality of Bright Lake and the potential effect on downstream waters, the outflow, the Bolton River, and the main inflows Harris and Pickerel Creek were sampled once on 2 May 2010. Harris Creek drains pristine Basswood Lake and its surrounding catchment basin, which is about half of the total Bright Lake watershed, while Pickerel Creek drains a large agricultural area, which is about one third of the total watershed (Table 3).

TP concentration was a high 39 µg/L in Pickerel Creek but below detection limits at the other stations (Table 12). DOC was also highest in Pickerel Creek, possibly reflecting organic inputs from the fields, as well as iron (associated with soils) and chloride. As expected from land use information, the water analysis indicates that Pickerel River is the most polluted of the investigated streams.

Table 12. Nutrient and chemical data collected in the three main creeks, 2 May 2010.

Stations	TP	Nitrate	Nitrite	DOC	TSS	Chloride	Iron	Sulfate
	(µg/L)	(mg/L)						
Harris Creek	nd	0.170	nd	3.9	nd	1.6	nd	4.1
Pickerel Creek	39	nd	nd	17.5	4.4	11.3	0.422	2.1
Bolton River	nd	nd	nd	6.1	7.1	3.9	nd	3.8

Note: nd, not detectable

Harris Creek, which is the outflow of the extremely nutrient-poor Basswood Lake, has expectably low TP concentration although nitrate is detectable at 170 µg/L. DOC, TSS and chloride, which can indicate anthropogenic disturbances in the watershed, are the lowest of all measured creeks.

Bolton River is the outflow of Bright Lake and its characteristics are largely affected by the lake. Therefore, it is not clear whether the non-detectable TP concentration is accurate, as mentioned previously.

4.8 Bottom sediments

Sediments that accumulate on the bottom of lakes document the past, but can also affect the current water quality. In Bright Lake the immediate past of logging by settlers is obvious, which may also effect the chemical composition and its effect on the overlaying water.

4.8.1 Sawdust and logging debris

Bright Lake was used in the logging years for the floating and processing of logs before they were boomed to a "jack ladder" on the south east corner and transported 4 miles to the shore of Lake Huron by a private railway. There is evidence from those times and the present that sawdust was abundant. (John Milito (2009): In 1901 a fisherman had a complaint about the sawmill refuse stating "the fishing ... is being spoiled by sawdust and other mill refuse which is being dumped into it." And John Milito, pers. communication: "The lake seems to be littered with wood bark and logs from the decades of logging. We see these wood chips wash up every year. I suspect the bottom sediments to be littered with these.")

There is ample evidence of former logging operation in Ontario and other Canadian lakes. Some of these lakes appear to become more eutrophic than lakes without a history of logging operations. The sawdust accumulated on the lake bottom presents a large reservoir of organic substance and could contribute to sediment oxygen demand that produces oxygen depletion (as suggested for several lakes studied by GN, i.e., South Lake, Haliburton District, ON; Bouchie Lake, BC). However, there is no analytical evidence of elevated organic substances from the sediment analysis of the Main station and the central basin (Section 4.8.2).

4.8.2 Chemical composition

Bottom sediment is best collected by a device that keeps the layers intact. In this way distinct sediment depths can be analyzed. The results of averages of two cores from the main station and the central basin (Table 5) for two depth intervals are presented in Table 13. The individual results are shown in Appendix H.

The sediment was dark, almost black and muddy without smell. The sediment was very loose requiring the corer to be lowered gently so as not to disturb it. No woodchips were observed. Chemical characteristics at both collection sites were similar. Organic content (loss on ignition, LOI), which is an indicator for eutrophic conditions and nutrient pollution in unstained waters, is below 10%. Sediment TP is less than 1.00 mg/g dry weight. Reductant-soluble P (Fe-P, which is involved in the redox-dependent P release from sedimentary iron hydroxides) is around 0.2 mg/g dry weight in the surface sample, but lower in the deeper samples at the main station as often found in recently eutrophied lakes. Results from both sampling occasions (2009 and 2010) were similar and support the Fe-P value for sediments at the main station (Table 13). For comparison, a stratified hardwater lake in southern Ontario (Lake St. George, Oakridges Moraine) with high internal loading had more than 30% organic content, a higher TP content (more than 1.2 mg/g dry weight, TP) but comparable releasable Fe-P (0.19 mg/g dry weight, Nürnberg 1988).

The size of the reductant-soluble P pool in Bright Lake sediments is large enough for sediment P release to occur (Section 5.3 "Internal P sources").

Table 13. Sediment characteristics of cores*

Station	Moisture	LOI	TP	Fe-P	Fe-P*	Calcium	Iron
	(%)		(mg/g dry weight)				
Main, 11 m depth							
0-5 cm	73.95	8.97	0.87	0.186	0.195	4.24	33.60
5-10 cm	73.15	8.15	0.77	0.118	0.094	4.55	31.50
Basin, 4-5 m depth							
0-5 cm	73.15	8.82	1.00	0.228		3.91	32.00
5-10 cm	71.50	8.81	0.99	0.274		3.92	29.00

*All cores were taken on 6 Sep, 2010, except one for Fe-P analysis on 4 Oct 2009

4.9 Fisheries and aquatic amphibians

There were several fish surveys conducted by the MNR. The first records are from summer 1964 (MNR. 1964) and include: Walleye, yellow perch, pike, smallmouth bass, rock bass, common sucker, northern sucker, bullhead, channel cat fish, pumpkinseed, ling, gar pike and bowfin.

Another survey by MNR was conducted on 5-9 July 2010 and 17 fish species were captured including: Longnose Gar, Bowfin, Rainbow Trout, Cisco (Lake Herring), Rainbow Smelt, Northern Pike, White Sucker, Shorthead Redhorse, Spottail Shiner, Brown Bullhead, Trout-Perch, Rock Bass, Pumpkinseed, Small- and Largemouth Bass, Yellow Perch and Walleye. Large 12 yr-old Walleye and 5 year-old Small- and Large- mouth bass were among the captured fish indicating a flourishing warm water fishery. The MNR report is attached in Appendix I.

The number of fish species or species richness of 17 compares well to that predicted from a model that takes into consideration the lake surface area and observed anoxia. Using Bright Lakes's area of 1,232 ha and an observed anoxic factor of 1 d/yr a species richness of 17.1 is predicted according to equation (1), ($n=52$, $R^2= 0.51$, $p<0.001$, Nürnberg 1995).

$$\text{Number of fish species} = -1.53*\log(AF+1) + 5.38*\log(A_o*100)+0.97 \quad (1)$$

The similarity of observed and predicted species number means that Bright Lake supports fisheries as is typical for its size and oxygen concentration.

In several previous springs fish kill was observed in the south eastern portion of the lake (John Milito, pers. comm.). Species involved were mainly bottom feeders like carp, ling and aquatic salamanders (mudpuppies, *Necturus maculosus*) that rely on external gills as their primary means of gas exchange, just like fish. As the kill happened during spring melt and ice-out conditions, it can be speculated that low oxygen concentration provoked this kill. Most likely, other, more sensitive species would have been affected as well. (Such as a 2.5 lb Northern Pike that washed up later in the summer, probably after having sunk to the bottom first.)

5 Phosphorus mass balance

5.1 Mass balance model

To determine the relative importance of different P sources, a simple mass balance model was applied (chapter *Predicting lake water quality* in The NALMS publication on managing lakes and reservoirs Holdren et al. 2001). In particular it was necessary to quantify the effect of loading (expressed as mass in kg/yr or by lake area in $mg/m^2/yr$ as in Equation 1) on the average annual P concentration (TP in Equation 1) in Bright Lake. To accomplish this, sedimentation within the lake was predicted from a retention (R) model based on annual average water load, q_s in m/yr (Nürnberg 1998).

$$TP = (\text{External Load} + \text{Internal Load})/q_s \times (1-R) \quad (1)$$

Where,

$$R = 15/(18+q_s) \quad (2)$$

5.2 External P sources

Based on land use information provided by the MNR (2010) and applicable TP export coefficients external load was evaluated in the lake shore capacity assessment for Bright and Basswood Lakes (Nürnberg and LaZerte 2011).

In this assessment, source areas are multiplied by P export for the entire watershed. Partitioning external load according to the land use information in the Bright Lake watershed (Figure 8) identifies the relative importance of the various P sources and their potential effects on lake water quality (Figure 9, Table 14). Loads from precipitation unto the lakes and creeks in the watershed are the highest single source of P (35%) followed by the extended forested area (24%), as is typical for relatively remote lakes on the Precambrian Shield. The next largest source is agriculture, which together with grass & meadows represent the extended farming activities in the watershed that together bring 25% of total external load. Wetlands contribute 8% and shoreline development of Bright Lake 6% of the P load. It is noteworthy that a pristine lake like Basswood retains a large proportion of its external load (predicted from annual water load as 0.76 or 76%) that does not reach Bright Lake as computed in the lower section of Table 14.

While the export coefficients used in this assessment represent an overall average of conditions for the specific land use that may be different in the Bright Lake watershed (for example, newly incorporated farm management practices may decrease the export from those areas), the results are different enough to support the following conclusions.

To decrease the P load to Bright Lake with the goal of achieving acceptable water quality, mainly anthropogenic loads must be considered, which is 31% of the total external load. Natural sources like precipitation or wetland, even though beaver ponds were found to contribute a large amount of P to receiving waters (Devito and Dillon 1993; Paterson et al. 2006), can or should only be marginally managed. Of anthropogenic sources, those of development and agriculture are potentially manageable and can be decreased. For example, septic system inspection and renewal, education to minimize fertilizer applications and agricultural BMPs can help minimize such sources.

The Pickerel Creek watershed and the immediate area around the lake convey the most of the P load to Bright Lake (Section 2.2, Table 3). Much of the land use of this section is agriculture and lake shore residence (Figure 8, Table 14 *and* Nürnberg and LaZerte 2011).

Table 14. Bright Lake watershed land use areas, specific TP-export coefficients and computed loads

Land Type	Area (m^2)	TP-Export ($mg/m^2/yr$)	Load* (kg/yr)	Source
Water (load from Precipitation)	49,931,682	16.7	834	Precipitation, LSC
Productive Forest	103,114,224	5.5	567	Forest <15 cleared, LSC
Treed Muskeg (Wetland)	223,362	50.0	11	Wetland, LSC
Open Muskeg (Wetland)	2,920,898	50.0	146	Wetland, LSC
Brush & Alder	2,170,846	5.5	12	Forest <15% cleared, LSC
Rock	3,842,603	5.5	21	Forest <15% cleared, LSC
Developed Agricultural Land	15,749,605	30.0	472	Intensive agriculture, LSC
Grass & Meadow	4,356,863	30.0	131	Intensive agriculture, LSC
Unclassified Land	3,770,640	9.8	37	Forest >15% cleared, LSC
Shoreline Development within 300 m (8,239,500m^2, included in above)			*115*	*LSC Model*
Bright Lake	12,321,254			
Watershed w/o Bright Lake (Ad)	173,759,468			
Total	186,080,722		2,347	
Load retained in Basswood (0.76 modeled retention)			-738	
Based on Load for Basswood 972 kg/yr from LSC				
External load to Bright from: *Total of above:*			1,609	

*Determined as: Load= area x export /10^6

Note: See Lake Capacity report for details (Nürnberg and LaZerte 2011).

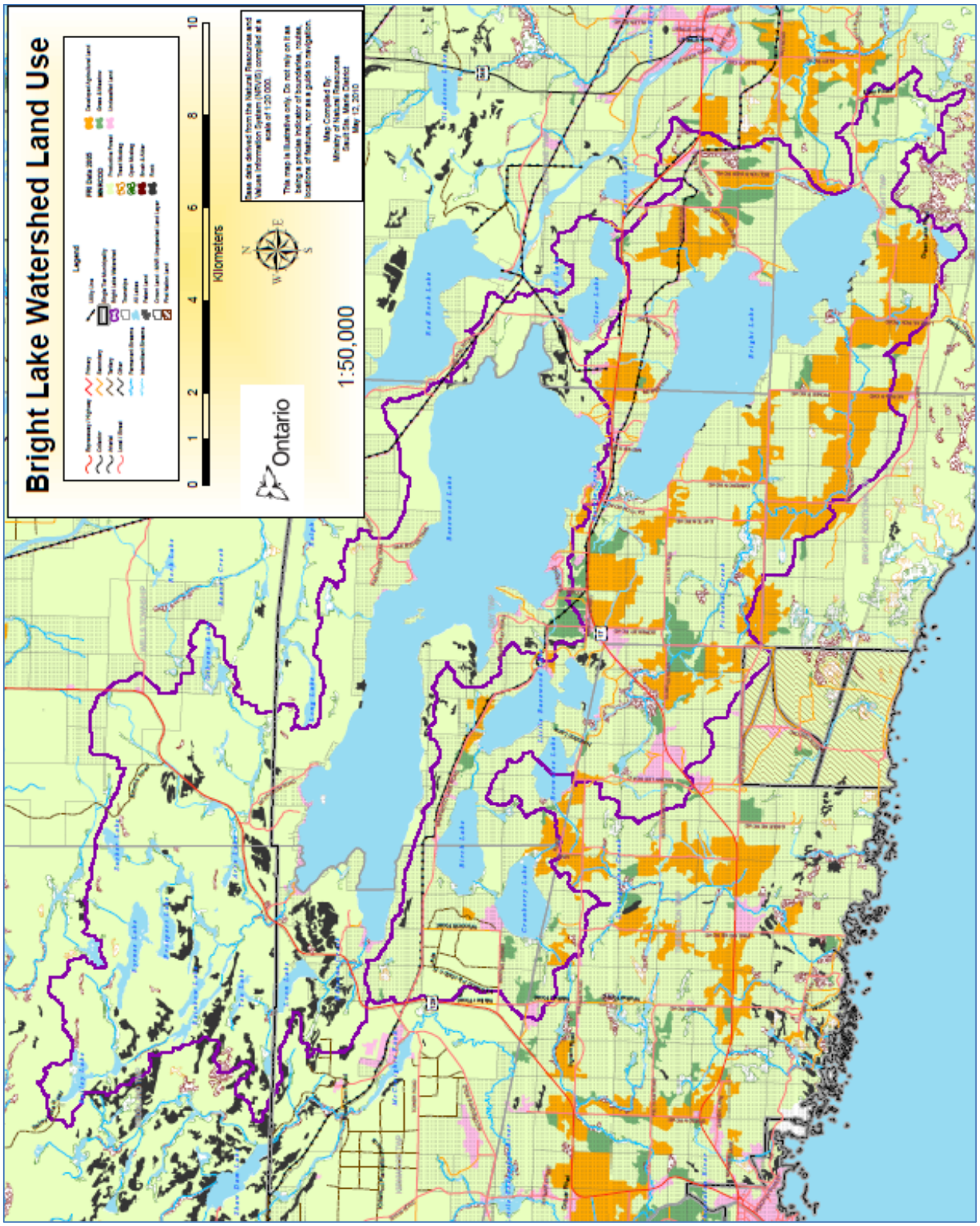

Figure 8. Bright Lake watershed land use map *(MNR 12 May 2010, an electronic map is included)*

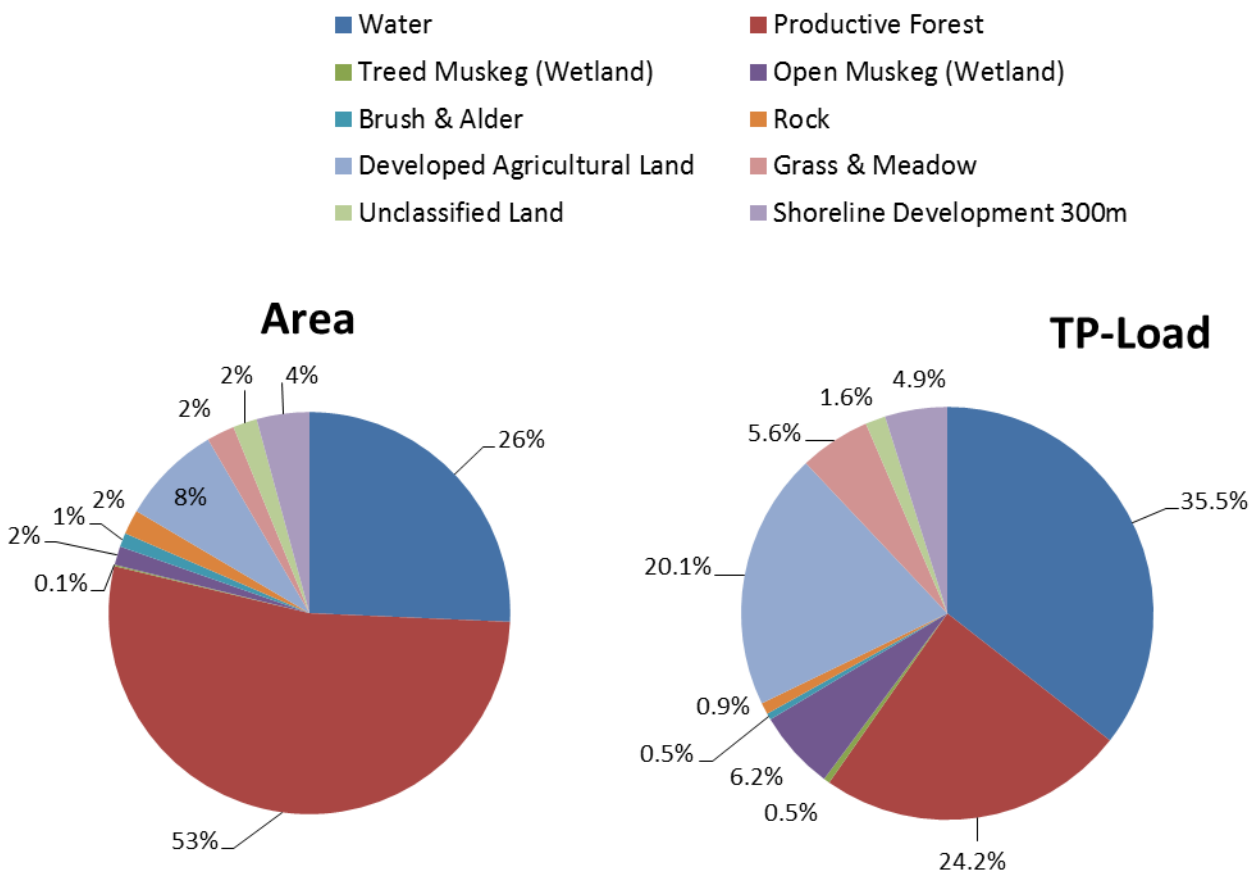

Figure 9. Watershed areal and loading proportions for Bright Lake (Table 14)

5.3 Internal P sources

Internal load as P released from bottom sediments is difficult to estimate especially in mesotrophic shallow and polymictic lakes. As described in detail in Nürnberg 2007a (Appendix J), it is especially important to consider this P source in relatively pristine lakes on the Canadian Shield, because recent anthropogenic nutrient enrichment of the sediments can fertilize the lake in the late summer and fall supporting cyanobacterial blooms.

The underlying mechanism is that anoxia leads to dissolution of iron hydroxides in sediments with concomitant release of adsorbed P (i.e., P attached to the iron surfaces) to adjacent lake water. Although internal loading stems from former external inputs which are stored in sediments, it is often ignored in P mass balance studies because of difficulties in obtaining estimates. Because of its high biological availability and the timing of its release during summer stratification, internal P loading can have an immense negative effect on summer water quality of a lake.

However, it is not always easy to determine the quantity of the internal load and there are many potential problems associated with separating the contribution of internal from external P sources

to a lake (Nürnberg, 2009). Accordingly, we usually propose a variety of independent approaches to quantify sediment derived P for as many years as possible. As expected, the lack of long-term TP and other records limits internal load analysis in Bright Lake. However, two approaches can be used based on newly collected Bright Lake data: the first uses *in situ* TP changes in the water of 2010 and the second involves the analysis of P fractions of the sediment.

5.3.1 In situ internal load from water TP increases

In situ internal loads were determined according to Equation 3 from the increases of water column TP concentration between spring and fall. This equation assumes steady state conditions with respect to external inputs and outputs, and any deviation has not been considered because such information was not available in sufficient detail. The effect should be small, because the flow is consistently low in the summer (Figure 10).

$$\textit{In situ}\ L_{int} = (P\_t_2 \times V\_t_2 - P\_t_1 \times V\_t_1) / A_o \qquad (3)$$

where, t_i with i=1 for initial date and i=2 for date at end of period
P_t_i, the corresponding P concentration
V_t, the corresponding lake volume
A_o, the lake surface area

In situ summer internal load estimates were calculated from the difference of the volumetric TP concentration between June and 6 September, a period that corresponds to hypolimnetic anoxia (Lake mixing may have occurred already on 21 Aug after a heavy thunderstorm according to temperature and DO profiles, Appendix D, but TP records are not available for 15 Aug - 6. Sep.)

The volumetric water-column TP concentration increased from 9.16 µg/L on 21 June to 34.05 µg/L on 6 Sep (Table 10) or 3.7 fold. If this increase can be solely attributed to sediment P release it would represent an internal load of 122 mg/m^2/summer (computed from the difference of 24.9 µg/L multiplied by mean depth, 4.9 m, Table 1).

The uncertainty of this estimate is large, for the following reasons: (1) data are available for one summer only, (2) TP values are highly variable potentially indicating analytical problems, and (3) external input has not been directly monitored, although it can be assumed to be negligible in the summer when inflow is low (Section 6). It is recommended that this type of monitoring and analysis be repeated in several future summers. Nonetheless, this value can be used as one approximate estimate of internal loading in the summer 2010.

5.3.2 Internal load determined from release rates and active area

In this approach the actively releasing bottom area is modeled from lake TP concentration and morphometry and the likely P release rate are estimated from several sediment fractions according to various models described in Appendix K. This approach considers the 0-5 or 0-10 cm layer of the bottom sediments and therefore presents an average for the last decade or so. Results are variable because of the different release rate estimates from different sediment fractions and different models, ranging from 0 to 127 mg/m^2/summer of internal load estimates (Table 15). Because the higher values agree well with that from the *in situ* method, an internal

load estimate of 125 mg/m^2/yr, which is 1,540 kg/yr is proposed until more data become available.

Table 15. Internal load estimates

Internal Load estimates (mg/m^2/summer)	
1. Sediment TP (unlikely)	0
2. Sediment Fe-P fraction	44
3. Sediment TP (literature)	127
4. Sediment TP & LOI	124
In situ Method	129

Winter internal load was not determined because there were no TP or oxygen profiles available that convey conditions under ice. Considering the relatively low organic content of the sediments it can be speculated that that sediment oxygen demand may be low, especially in the winter under ice at about 2-3 C temperature, so that winter internal load would be marginal.

5.4 Relative importance of phosphorus sources

Total external load is estimated as 1,630 kg/yr compared to a similar estimate of internal load of 1,540 kg/yr. Internal loading, if it can be corroborated by future measurement, expectedly has an tremendous effect on Bright Lake water quality for several reasons: (1) Timing and location: it is injected into the lake from the sediments in late summer and fall, a season when phytoplankton is often P starved because external input is low. (2) Chemical form: internal load is readily available to plankton because it is released from the sediments as phosphate. In comparison, external P inputs are often combined with particles and are not readily available, but sink to the bottom or are flushed from the lake.

With respect to remediation aspects, external load includes fluxes that cannot easily be controlled, like precipitation and runoff from natural area; only 31% of the load is from anthropogenic sources (lake shore, agricultural and grass land) and may be manageable. Consequently, diminishing internal load is most desirable.

6 Climate, hydrology and water level

Patterns of precipitation and flows affect the water quality of lakes in various ways and have to be investigated in a limnological assessment. Furthermore, the shoreline resident John Milito pointed to connections between lake level, beavers and Bright Lake water quality based on circumstantial evidence (Milito and Shine 2010), even though there were not many long-term observations for Bright Lake water quality available. Consequently such relationships are investigated in this project as well.

6.1 Long-term flows and lake levels

Seasonal patterns of climate related variables such as precipitation and flows exist. In a temperate location like that of Bright Lake, typically flows are highest in the spring after

snowmelt and lowest in the summer growing season. This is apparent for Harris Creek at Basswood Lake Dam about 1.6 km upstream of the inflow to Bright Lake. Flows were approximately estimated from two gauging stations with continuous records of 1978-2010 (Figure 10, Root River at Sault Ste Marie, Hydat 02CA002, weighted =0.6, and the Serpent River above Quirke Lake at Hydat 02CD006, weighted =0.4. Note that the flows for 2010 of Hydat 02CA002 are preliminary.) These stations and their weighting were suggested by MNR hydrologists as a representation of the flow of Harris Creek at the Basswood Dam (Regional MNR Hydrologist, Rich Pyrce, 6 Nov 2009, Appendix L).

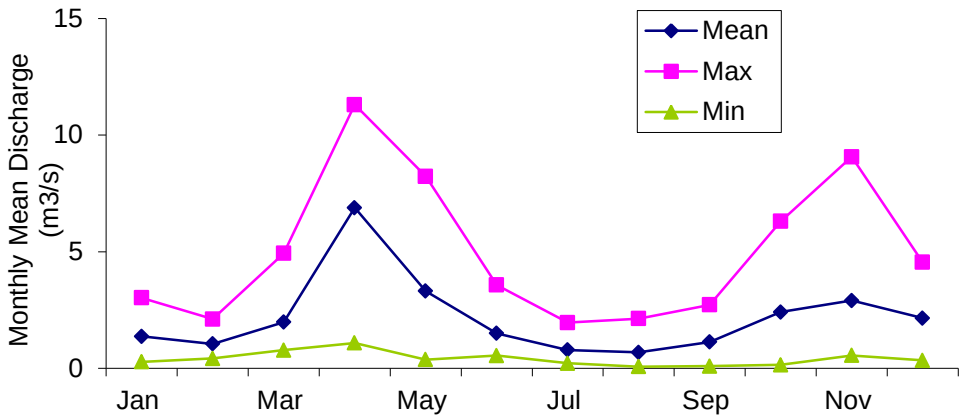

Figure 10. Long-term monthly hydrology for Harris Creek at Basswood Lake Dam (1978-2010)

Similar to flow, the lake level (recorded by John Milito at his property downstream of Pickerel Bay, Figure 11 Figure 12) peaks in the spring and drops in the summer by more than a meter (maximum annual observed elevation change was to 1.3 m in 2008).

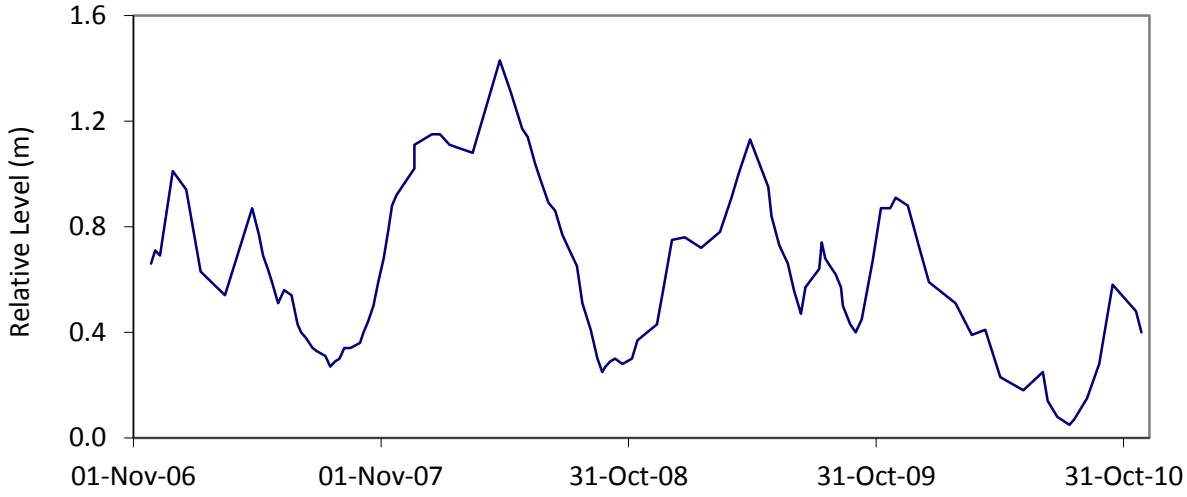

Figure 11. Relative lake level measured downstream of Pickerel inflow.

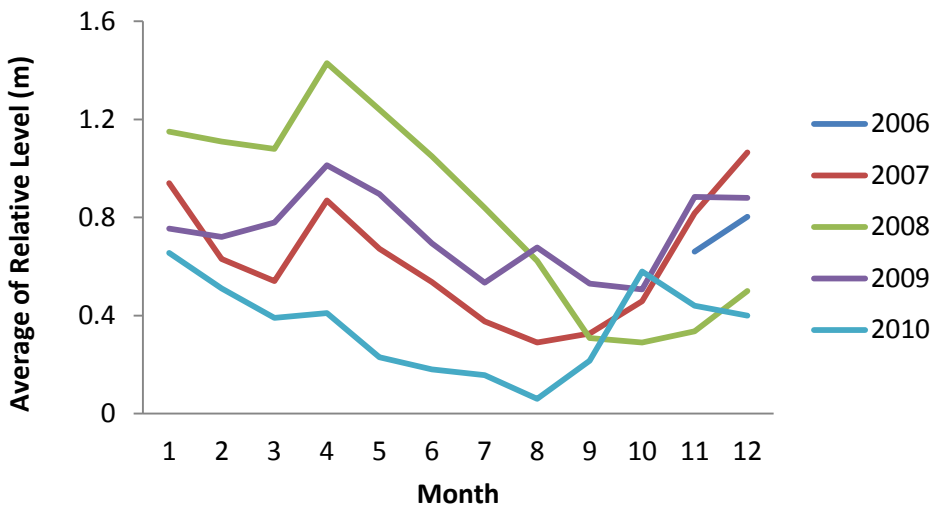

Figure 12. Annual differences in monthly average lake level

Monthly averages of lake levels are significantly correlated with flows in Harris Creek at Basswood Lake Dam, computed from two upstream stations as explained previously (Figure 13). One third of the variation in lake level is explained by this relationship (R^2= 0.35). The remaining variance may be influenced by beaver activities in the in- and outflow. Beaver dams in Harris Creek were detected in the summer of 2010; they would have a lowering effect on the level, while those in the Bolton River (outflow) would have an increasing effect. For example, there were several beaver dams on the Bolton River in summer and fall of 2009, that were then removed by the municipality of Huron Shores, because the associated high lake level threatened infrastructure.

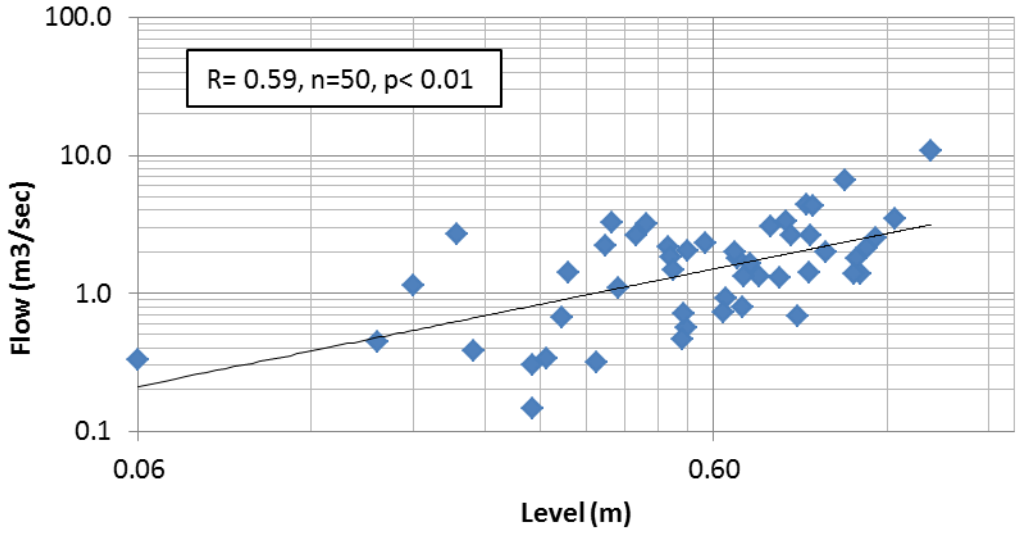

Figure 13. Comparison of monthly averages of computed flows at the Harris Dam with lake level measurements (Nov 2006 – Dec 2010).

6.2 Effects on water quality

In a pattern typically for temperate lakes, Bright Lake's flows and levels are highest in the spring and lowest during the summer. This means that most of external pollutant input occurs in the spring and fall and least in the summer, when any direct runoff into the lake is minimized. This is the time period when potential internal nutrient sources are most important, because they would be used by phytoplankton immediately. Variations of patterns between years may contribute to annual variability in water quality.

While not enough data are available to perform a statistical analysis, tendencies can be investigated by simple comparison of water quality with physical variables in specific years. Such a comparison reveals that high flows and water levels in the spring (Figure 14) are often followed by summer and falls with low water quality, while low spring flows as observed in 2010, coincide with better water quality in the summer (Table 16). An exception is 2009 with average flows and yet algal blooms in the fall. There is some evidence that the beaver dams in Bolton River outflow (Figure 15) inhibited the flow enough to create higher than average levels.

Therefore, it can be concluded that lake levels as determined by flows and beaver dams and their specific locations and times may contribute to the water quality variation in Bright Lake. At higher lake levels, longer periods of stratification prevail and provoke cyanobacterial blooms. This sequence of events indicates that possibly internal load is the driver of the water quality problems, as sediment release is enhanced under anoxic stagnant conditions.

Table 16. Comparison of water quality with physical variables

Year	Water Quality						Physical Variables		Beaver
		Phytoplankton		Anoxia	Strati-fication	TP	Flow	Level	Dam
		Bloom	Secchi				Apr/May		Location
1996	-	?	turbid	high	stratified	n.a.	max	n.a.	?
2008	-	severe	n.a.	n.a.	n.a.	n.a.	max	max	outflow?
2009	-	severe	n.a.	high	stratified	high	average	second highest	outflow
2010	+	none	clear	medium	polymictic	medium	min	low	inflow

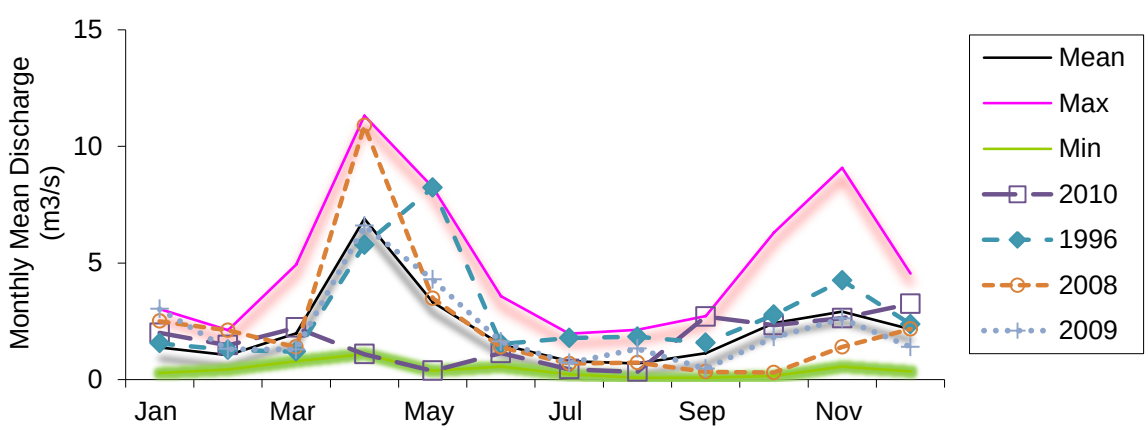

Figure 14. Flows for specific years compared to longterm averages.

Figure 15. Beaver dam in the Bolton River (27 Nov 2009, John Milito)

7 Recommendations

If the data assembled so far are representative it appears that bottom sediment derived internal TP load is as large as external input (from precipitation, the inflows and the surrounding area) and both loads contribute to any water quality problems in Bright Lake. Except for the fall of 2009 the monitoring in this study (2009-2010) demonstrated relatively good water quality conditions and cyanobacterial blooms were absent even in the warm summer and fall of 2010. Because such blooms have been observed in previous years, it is recommended that the basic monitoring continue in future growing seasons.

7.1 Future monitoring

To determine potential pollutant sources, and their effect on Bright Lake, water transparency as Secchi depth, nutrient concentration (TP, Nitrate, total Kjeldahl-N) should be monitored from May to October at least at the main Stations B, but also for the Pickerel Bay (Station C).

Pickerel Creek had elevated TP concentration when measured and the Pickerel inflow Bay (Station C) had elevated TP once (2 May 2010) as well. We recommend that Station C be moved closer to the inflow into Bright Lake so that dilution with Bright Lake water can be minimized, however, this may not be possible because of the extended shallow wetland area at the Pickerel Creek delta. In general, a higher frequency and precision (repeat samples) of monitoring would be needed to reliably determine the importance of the various inflows with respect to P loading.

The investigation of internal loading processes is still important as it can be expected to vary annually with flow, water level and other conditions. Consequently, profiles measurements of

DO, temperature and TP should be conducted routinely at the deep station throughout the growing season.

One winter sampling under ice, as late as possible before ice-out would indicate any winter internal load. Anoxia and elevated TP concentration above the bottom in late March would not only indicate that internal load may provide nutrients for a spring bloom, but it also would indicate that sediment oxygen demand is high enough to provide low DO concentration in bottom sediments and P release throughout the summer.

7.2 Potential management options

Although Bright Lake did not experience any water quality problems in 2010, descriptions of possible remediation techniques are included here that may be applicable to Bright Lake`s situation, and could prevent future cyanobacteria blooms.

In general, the most apparent water quality issue often conflicting with lake use and health is an overabundance of algae. The control of algal growth and especially cyanobacterial blooms in lakes can be accomplished in several ways. The most common is to reduce the nutrient inputs (phosphorus, in particular), as most excessive algal growth is the result of fertilization from external sources like agriculture, field and lawn runoff, septic and sewage outflows. In addition to external loading, treatment must often involve measures to decrease internal loading from the sediments, however usually the external sources have to be addressed first.

Bright Lake has a valuable warm water fishery and any treatment has to strive to support this fishery as well.

7.2.1 External load abatement

Typical remedial options are displayed in Table 17. Many of the suggestions are documented and addressed by the Eastern Algoma Stewardship Council and the Freshwater Quality Public Outreach of the Central Algoma Freshwater Coalition. A Guide to Stewardship of Ontario's Waters (Federation of Ontario Cottagers' Associations (FOCA) 2009) and the *Watershed* chapter in the North American Lake Management Society's (NALMS) publication on managing lakes and reservoirs (Holdren et al. 2001) provide further information.

Table 17. External load remedial options and techniques

- Source control
  - Identify and renovate leaky septic systems
  - Minimize erosion
  - Minimize impervious area & Maximize infiltration
  - Diminish runoff from agricultural non-point sources (BMPs)

- Manage beavers
  - Monitor beaver activity in the watershed
  - Introduce pipes or culverts through existing dams

- Have licenced trappers remove beavers and dams

- Reduce agricultural impact
  - Prevent livestock wading in creeks
  - Optimize timing of tilling, fertilizer application
  - Improve agricultural operation by further BMPs
  - Educate farmers
- Reduce loading from lake shore
  - Stabilize the shoreline; protect the riparian zone
  - Maintain vigorously growing shrubs and trees next to water surface
  - Stabilize eroding shoreline
  - Route drainage away
  - Establish vegetation
  - Educate lake shore residents and lake users

7.2.1.1 Lake shore residents and lake users

Fertilizer runoff from extended grassed areas can contribute a significant amount of nutrients, especially if combined with irrigation, which flushes nutrients and pollutants into the lake. The easiest way to avoid such nutrient-rich runoff is the keeping or extending of natural edgings, instead of mowing and fertilizing lawns up to the lake shore.

Other pollutant input was observed from ice fishing and increased horse traffic on the ice. An educational program involving a “poop and scoop” ordinance for horses and encouraging anglers to use the toilet facilities at the boat launch may minimized this nutrient source.

Compliance of septic systems and industries in the watershed like the Waste Disposal Site on Cullies R, West of Bright, W2, Concession 1&2 is of utmost importance. All waste water systems should be inspected and decommissioned if not adequate. Compliant waste disposal has to be proven also for about 20 trailers on the lake.

7.2.1.2 Agriculture

There is a lot of information available about agricultural BMPs (Thornton and Creager 2001). Besides the economic constraints, social patterns can ultimately determine whether BMPs will be applied in agricultural operations (Welch and Marc-Aurele 2001). Information exchange and mutual understanding are the preconditions to a successful management plan and have been initiated already. This is probably the reason that some small steps have already been accomplished, such as providing pumps to select farmers to draw water from Pickerel Creek for the cattle and providing electric fencing to keep livestock out of the water (Figure 16). It is expected that now native plants can grow in the treated areas and improve bank stabilization.

While some improvement has been made on the Pickerel Creek there are still farms at the Bolton River Rd along small creeks and inlets (like the “Ditch“ Creek) and on Watson Road that need remediation.

Figure 16. Livestock close to the water before electric fencing was installed (John Milito, 17 Oct 2009)

7.2.1.3 Natural sources: wetlands and beavers

Beavers increase nutrient load downstream (Klotz 1998) and could be managed by discouraging their dam construction (Section 7.2.2.2). In this way P export from wetlands may be decreased. Although retention of water by dams increases sedimentation and hence may help prevent P from reaching the lake, the beaver dams are not permanent structures. Instead, breakage during extreme flow events have been documented that flushes polluted sediments into downstream lakes. There is circumstantial evidence in at least two different lakes where an upstream breach probably induced water quality problems involving cyanobacterial blooms (Gertrud Nürnberg, unpublished data). There may be more evidence from future studies that prove the importance of such beaver dam removal in the restoration of remote lakes.

However, removal of beavers is controversial and care has to be exercised not to destroy any wetland areas. Target streams have to be carefully selected so they are most beneficial in reducing nutrient load to the lake and assisting flow management (Section 7.2.4).

7.2.1.4 Treatment of major polluted inflow(s)

Preliminary information suggests that Pickerel Creek is a major P source to Bright Lake. Instead or in addition to treating the watershed to decrease runoff from the source it is theoretically possible to treat the inflow before it enters the lake. Such treatment has been done in highly eutrophic lakes, where chemical precipitation of P by aluminum or iron is conducted (Harper 2005). Such treatment requires space for the treatment plant and there are continuous operating costs so that they are usually applied only in highly used ponds and lakes.

7.2.2 Abatement of internal load and mitigation of cyanobacteria blooms

Although internal loading stems from former external inputs which are stored in sediments, it has to be treated separately. Because of its high biological availability and the timing of its release during summer stratification, internal P loading can have an immense negative effect on summer water quality of a lake and trigger cyanobacterial blooms.

7.2.2.1 In-lake technologies

There are several restoration techniques that address internal phosphorus loading from sediment surfaces (Cooke et al. 2005 and the chapter *Management techniques within the lake or reservoir* in The NALMS publication on managing lakes and reservoirs Holdren et al. 2001), but considering the size of Bright Lake, its relative remoteness and its polymixis they are not feasible. Costs would probably be prohibitive in many lake restoration techniques to combat internal load directly in a large lake such as Bright. The most used remediation techniques (dredging, chemical treatment and hypolimnetic withdrawal) are discussed in Appendix M to indicate their lack of applicability to Bright Lake. In general, possible abatement of internal P sources should only be attempted after efforts of reducing external load, including watershed BMPs, septic system renovation, and others have been implemented for several years and if unacceptable water quality still prevails.

There are treatments that address nuisance algae and plants (macrophytes) in the lake directly (chapter *Management techniques within the lake or reservoir* in Holdren et al. 2001). However, they often only treat the symptoms and results are short-lived. Besides, they are more applicable to smaller lakes and ponds, as costs would be prohibitive in a large lake such as Bright. Further reasons why such techniques are not recommended are presented in Appendix M.

7.2.2.2 Flow management

Bright Lake is so large that no direct application and treatment of internal load seems feasible. Instead, operation of flows may be possible and advisable, because it could decrease internal P load and associated cyanobacterial blooms.

If the influence of water flow and lake level on water quality as described in Section 6.2 can be substantiated by future monitoring, managing flow for low summer lake level may be a possible management option. In such an attempt it would be important to manage impeding structures such as beaver dams in the upstream Harris Creek and in the outflow, the Bolton River. It would be advisable to monitor beaver activity in these streams, remove dams, and trap or “baffle” beavers that impede the flow and effect lake level.

Baffling beavers is promoted by MNR as a beaver management method that includes a long pipe installed through or under the beaver dam so that the pond is drained while the beaver does not hear any flushing noise (Ministry of Natural Resources 1995). In this way, wetlands are not destroyed but beavers are discouraged to remain at that particular location and move on. Such management would also diminish nutrient load from the animals and prevent accidental dam breakage during high flushing events.

Flow management could be combined with the restoration method of dilution and flushing, which is a recognized lake restoration procedure (Cooke et al. 2005). This technique is often not very costly and has been applied in lakes with a clean water source. Such a source is available in Harris Creek. Careful flow management via Basswood Lake Dam to increase flushing during an adequate season for Bright Lake without compromising Basswood Lake's water quality could be envisioned. Because Bright Lake's summer level appears to influence the occurrence of cyanobacterial blooms, careful timing and the maintenance of an operational outflow, the Bolton River, are necessary.

Any of these potential treatments would need a further and more detailed hydrological assessment, in combination with a limnological evaluation including a corresponding nutrient budget analysis. Other considerations that have to be investigated and perhaps approved by appropriate agencies are potential negative influences on the marginal wetland and fish recruitment areas that require certain seasonal water level fluctuations.

7.2.3 Non-native species introduction

As described in the previous section any introduction of non-native species is potentially harmful, and should be avoided and monitored, if possible. Public outreach and education by the Bright Lake Association and the Central Algoma Freshwater Coalition should be introduced or continued and information posted at the boat launch and marinas. In particular the following species may be relevant to the Bright Lake environment.

- Plants:
  - Eurasian Milfoil (*myriophyllum spicatum*),
  - Purple loose strive (*lythrum salicaria*),
  - Water Soldier (*stratiotes aloides*),
- Animals:
  - Rusty crayfish (*Orconectes rusticus*)
  - Molluscs
    - Zebra mussel (*dreissena polymorpha)*
    - Quagga mussel (*dreissena rostriformis bugensis*)
  - Zooplankton:
    - Spiny Waterflea (*bythotrephes longimanus*),
    - Fishhook Water Flea (Cercopagis pengoi)

This list is not complete and more information is available at the websites of several institutions. The Ontario Federation of hunters and Anglers: http://www.invadingspecies.com; the Invasive Species Research Institute (ISRI) on the Algoma University campus: http://www.isri.ca/Sections/Index and the NSERC Canadian Aquatic Invasive Species Network II (CAISN): http://www.caisn.ca/

8 Conclusions

The remediation of remote lakes is especially important when draining into the Great Lakes. Bright Lake drains via its outlet, the Bolton River, via the Mississagi River into Lake Huron. The Mississagi River has a TP concentration of 8 µg/L (median for the period 2001-2006, Ministry of Environment 2009). This value is much lower than Bright Lake's average of about 20 µg/L and it can be concluded that Bright Lake adversely affects and fertilizes downstream waters.

Based on the available information, several recommendations are made in this report. Because the most apparent water quality issue conflicting with lake use and health is an overabundance of algae, the control of algal growth and especially cyanobacterial blooms should be attempted. The most common method is to reduce the nutrient inputs (in particular of P), as most excessive algal growth is the result of fertilization from external sources like agriculture, field and lawn runoff, septic and sewage outflows. Specifically in the Bright Lake watershed, septic system inspection and renovation, and agricultural and shoreline best management practices are recommended. A long list of direct treatment involving measures to decrease internal loading from the sediments were examined, but all are deemed unfeasible and costly because of Bright Lake's size and remoteness. It appears that the only feasible remediation option may be the management of water flow by optimally adjusting seasonal water levels to prevent prolonged stratification and internal loading. Such management would involve the monitoring and eliminating of beaver induced water retention.

In preparation of such management it is recommended to

- Continue future monitoring to corroborate the flow-water quality dependency
- Accomplish a detailed hydrological study
- Examine the potential effect on marginal zones such as wetlands and fish spawning areas
- Examine the feasibility of the restoration technique of *dilution and flushing* using Harris Creek water under particular consideration of Basswood Lake water quality.

A lake shore capacity assessment for Bright and Basswood Lakes commissioned by the Town of Huron Shores (Nürnberg and LaZerte 2011) contains relevant information and supplements this report.

9 References

For education and general information there is a lot of material available from governmental and non-governmental organizations. For example, suggestions are documented and addressed by the Freshwater Quality Public Outreach of the Central Algoma Freshwater Coalition, the Kensington Conservancy, and the Federation of Ontario Cottagers' Associations (FOCA), which also published A Guide to Stewardship of Ontario's Waters (2009).

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Appendix A: Press Release

For Immediate Release July 31, 2010

MPP Mike Brown to Present Ontario Trillium Foundation Plaque to Bright Lake Association

MPP Mike Brown will be "knee deep" in Bright Lake when he presents an Ontario Trillium Grant plaque to the Bright Lake Association on August 14, 2010. "We want MPP Brown to 'wade deep' into this serious situation on our Algoma lakes," says Association Vice President John Milito.

After Algoma Public Health announced that a number of lakes east of Sault Ste Marie, including Bright, Desbarats, and Caribou, tested positive for blue-green algae, citizens groups have joined forces to deal with issues that have negatively impacted everything from outfitter stores to municipal tax bases. Area tourism has been hurt by news of the toxic algae and the posting of warning signs on many of the regions' lakes.

On Bright Lake, the largest of the affected lakes, property owners formed the Bright Lake Association to fight the noxious algae blooms which can kill wildlife, pets and humans. "The Bright Lake Association has been on the leading edge of these efforts," says Association Vice President John Milito.

While Bright Lake has strength in numbers and strong support from the Blind River MNR, the DFO and the Municipality of Huron Shores, some of the smaller lakes in the area have had to merge, forming the Central Algoma Freshwater Coalition. And another citizen's group led by the MNR, the Eastern Algoma Stewardship Council, bridges the gap between government and volunteers concerned with lake stewardship.

"We want everyone to know about the tremendous volunteer work that is being done for our lakes," adds Milito. The volunteers are supported by an array of experts, including Dr. Sue Watson from the Canadian Center for Inland Waters (Environment Canada), and Dr. Gertrud Nurnberg of the Freshwater Research Company. One of Dr. Watson's areas of expertise is the causes of noxious algae blooms. Dr. Nurnberg, an expert in lake geochemistry, is conducting a lake study on Bright and providing recommendations for treating the lake. The Trillium Grant money will be used to continue these studies and pay for expensive monitoring equipment.

"We share the monitoring equipment with our partner lakes ---- as well as our growing knowledge base," emphasizes Milito.

MPP Mike Brown will present the Trillium Grant Plaque to the Bright Lake Association at the Sunset Beach boat launch on August 14, 2010 at 10:00 a.m. On hand to accept the award will be association leaders and the acting mayor of Huron Shores, Jody Fullerton.

John Milito, Vice President
Bright Lake Association
705-542-1213

Tracey Cooke, Coordinator
Eastern Algoma
Stewardship Council
705-941-5117

Lindsay Verdone,
Coordinator
Central Algoma Freshwater
Coalition
705-971-5356

Appendix B: Watershed areas and land use from MNR maps

MNR 26 Apr 2010	Original Values Area (km^2)	Adjusted values to match MNR Land Use Map (km^2)
Basswood Lake area, A_o	27.07	27.07
Basswood A_d to inflow	58.90	60.89
	Southern shoreline	
	4.78	4.94
Pickerel Creek	50.95	52.67
	2.05	2.12
	1.60	1.65
	2.95	3.05
	Northern Shoreline	
	4.05	4.19
	3.47	3.59
	1.22	1.26
	1.59	1.64
Not accounted for in Map	*10.34*	10.69
Total to Bright Lake (A_d)	168.97	173.76
Bright Lake area, A_o	12.32	12.32
Out via Bolton	181.29	186.08

Note that all areas except the lakes areas for Basswood and Bright were proportionally adjusted to match the values of the land use area below for the Map displayed in Figure 8.

Land Use *(MNR 12 May 2010)*	Area (m^2)
Water	49,931,682
Productive Forest	103,114,224
Treed Muskeg (Wetland)	223,362
Open Muskeg (Wetland)	2,920,898
Brush & Alder	2,170,846
Rock	3,842,603
Developed Agricultural Land	15,749,605
Grass & Meadow	4,356,863
Unclassified Land	3,770,640
Total	186,080,722
Bright Lake	12,321,254
Watershed w/o Bright Lake (A_d)	173,759,468

Appendix C: Analytical detection limits

Chemical Group	Parameter	units	Testmark MDL	Trent MDL
Cations	Calcium	mg/L	0.05	tbd
	Magnesium	mg/L	n.a.	tbd
	Potassium	mg/L	n.a.	tbd
	Sodium	mg/L	n.a.	tbd
Anions	Sulphate	mg/L	1	tbd
	Chloride	mg/L	0.2	tbd
Nutrients	Total Phosphorus	µg/L	2	0.2
	Ammonium	µg/L	n.a.	2
	Nitrate/Nitrite	µg/L	100/30	2
	TKN	µg/L	n.a.	2
Carbon	DIC	mg/L	n.a.	n.a.
	DOC	mg/L	0.4	n.a.
Titrations	Alkalinity	mg/L	1	n.a.
	Turbidity	mg/L	2	n.a.
True Colour		TCU	1.5	tbd
Total suspended solids (TSS)		mg/L	n.a.	2
Metals	Fe	mg/L	0.02	n.a.
	Bromide	mg/L	0.1	n.a.
	Fluoride	mg/L	0.1	n.a.
Chlorophyll a		µg/L	0.5	n.a.

n.a., not applicable
tbd, to be determined

Appendix D: Temperature and dissolved oxygen data

Dissolved oxygen and temperature profiles for 1996 measured in a previous study
Five profiles for two stations at the basin edges (Figure 2 ,Table 4, Blewett and Goold 1996)

Dissolved Oxygen (mg/L)					
Depth (m)	10-Jun-96	03-Jul-96	10-Jul-96	15-Jul-96	22-Jul-96
	Station 1				
1	12.5	12.0	9.0	6.0	8.0
2	11.0	10.0	8.0	5.0	6.0
3	11.0	9.0	7.0	3.5	4.5
4	11.3	7.3	4.0	3.0	4.0
5	12.3	5.5	2.5	2.5	3.5
6		5.3			
	Station 2				
1	11.8	11.0	8.0	7.0	6.0
2	11.6	6.5	4.0	5.0	4.5
3	11.0	5.5	2.3	2.5	3.0
4	10.4	5.0	2.0	2.0	2.5
5	10.3	4.5	1.8	1.5	2.3
6	10.3	4.3	1.5	1.3	2.0
7	10.2	4.0	1.5	1.3	2.0
8	10.2				

Temperature (C)					
Depth (m)	10-Jun-96	03-Jul-96	10-Jul-96	15-Jul-96	22-Jul-96
	Station 1				
2	19	17	16.5	17	18.5
3	18.5	17	16.5	17	18.5
4	17	17	16.5	17	18.5
5	15	17	16.5	17	18
6		16.5			
	Station 2				
1	18	17	16.5	17	18
2	17	17	16.5	16.5	18
3	16.5	17	16.5	16.5	18
4	16	17	16.5	16	17.5
5	16	17	16.5	16	17.5
6	15	17	16.5	16	17
7	15	17	16	16	17
8	14				

Note: DO values around 5 mg/L and below are shaded.

Dissolved oxygen and temperature profiles for 2009 measured in this study at the deep (main) station.

Depth (m)	Dissolved Oxygen (mg/L)			
2009	24-Aug	05-Sep	12-Sep	17-Oct
0.5			10.3	
1	11.4	9.4	10.3	10.1
2	11.6	9.8	10.0	9.9
3	11.5	9.5	10.1	9.7
4	11.5	8.6	9.7	9.7
5	11.2	7.6	8.1	9.7
6	10.8	7.1	7.5	9.7
7	10.2	7.0	6.0	10.3
8	6.9	6.8	4.2	10.5
9	3.5	6.4	3.2	10.4
10	1.8	6.6	2.9	10.3
11	1.0	6.5	2.1	10.5
Depth (m)	Temperature (C)			
0.5			22.9	
1	23.0	22.2	23.1	12.2
2	22.0	21.2	23.0	12.5
3	22.0	20.5	23.0	12.7
4	21.8	19.8	22.6	12.9
5	21.7	19.4	20.7	12.9
6	21.6	19.0	20.4	12.9
7	21.4	18.8	19.8	13.0
8	20.9	18.7	19.5	13.0
9	20.4	18.6	19.4	12.9
10	20.0	18.4	19.2	12.9
11	19.6	18.3	19.2	12.9

Note: DO values below 5 mg/L are shaded.

Dissolved oxygen and temperature profiles for 2010 measured in this study at the deep (main) station.

Depth (m)	Dissolved Oxygen (mg/L)										
2010	02-May	24-May	21-Jun	11-Jul	18-Jul	25-Jul	15-Aug	21-Aug	06-Sep	10-Oct	07-Nov
0.5					6.4	6.5	7.4	6.8	6.8	8.2	9.9
1	9.6	7.9	7.0	6.6	6.3	6.5	7.2	6.8	6.7	8.1	9.9
2	9.7	7.9	6.8	6.7	6.0	6.4	7.1	6.8	6.6	8.1	9.8
3	9.9	7.9	6.7	6.7	6.2	6.4	6.8	6.7	6.6	8.0	9.8
4	9.9	7.9	6.7	6.6	6.0	6.2	6.7	6.8	6.7	8.0	9.7
5	10.9	7.8	6.6	6.6	5.8	6.1	6.4	6.7	6.6	8.0	9.6
6	10.1	7.8	6.5	6.1	5.1	6.2	6.0	6.8	6.6	7.9	9.7
7	10.1	7.7	5.8	5.5	5.1	6.1	5.8	6.8	6.6	7.9	9.6
8	10.1	7.4	3.4	4.5	4.1	6.1	4.6	6.8	6.4	7.8	9.5
9	10.5	7.2	2.5	4.2	2.5	2.6	4.0	6.8	6.5	7.8	9.4
10		7.0		3.4	2.1	1.6	3.5	6.7	6.6	7.9	9.5
11		7.0		3.1	2.1	1.1	1.1	6.5	6.4	7.8	9.5

Depth (m)	Temperature (C)										
0.5				23.3	23.6	24.1	25.0	21.8	19.9	14.1	7.2
1	16.3	18.8	21.3	23.7	23.7	23.9	24.9	21.8	19.9	14.0	7.1
2	15.5	18.0	20.7	23.7	23.7	23.6	24.6	21.8	19.9	13.8	6.9
3	15.1	17.5	20.7	23.1	23.6	23.4	24.0	21.8	19.9	13.7	6.8
4	14.8	17.4	20.5	23.0	23.6	23.4	23.7	21.8	19.9	13.6	6.7
5	14.4	16.5	20.5	22.7	23.4	23.3	23.4	21.8	19.8	13.6	6.7
6	14.1	14.7	20.0	21.8	23.1	23.2	23.4	21.8	19.8	13.6	6.7
7	14.0	12.9	19.7	21.3	23.0	23.2	23.1	21.8	19.8	13.6	6.6
8	13.8	12.6	18.1	20.6	22.2	23.1	22.6	21.8	19.8	13.5	6.6
9	13.5	12.5	16.9	20.3	21.1	22.2	22.3	21.8	19.8	13.5	6.6
10		12.4		20.0	20.9	21.6	22.0	21.8	19.8	13.5	6.6
11		12.4		19.9	20.8	21.1	21.7	21.8	19.7	13.5	6.7

Note: DO values below 5 mg/L are shaded.

Appendix E: Anoxic Factor

The anoxic factor was computed from the following equation (Nürnberg 1995):

$$AF = \sum_{i=1}^{n} \frac{t_i \times a_i}{A_o}$$

where t_i, the period of anoxia (days), a_i, the corresponding area (m^2), A_o, lake surface area (m^2), and n, numbers of periods with different oxycline depths. (A threshold of 2.5 mg/L was used for the DO profiles determined by *in situ* probe measurements, considering that the sediment surface likely is anoxic when the oxygen probe measures a small amount of DO in the overlaying water.)

Anoxic factor determined from DO profiles in Bright Lakes

Year	Date-beg	Date-end	Depth (m)	Days	Area (km2)	A-Factor (d/period)	AF (d/summer)
1996	10-Jul-96	15-Aug-96	3	36	9.08	26.54	31.92
	16-Aug-96	02-Sep-96	7.1	17	3.90	5.38	
2009	21-Jun-09	09-Aug-09	9.1	49	0.11	0.44	0.90
	10-Aug-09	01-Oct-09	9.1	52	0.11	0.46	
2010	21-Jun-10	18-Aug-10	9.1	58	0.11	0.52	0.52

Note: Shaded entries are estimates

Appendix F: Secchi disk data in 1996

Date	Secchi Transparency (m)		
	Station		Average
	1	2	
10-Jun-96	1.50	2.50	2.00
17-Jun-96	3.00	3.75	3.38
24-Jun-96	3.50	3.25	3.38
03-Jul-96	2.00	2.00	2.00
10-Jul-96	2.00	2.00	2.00
15-Jul-96	2.75	2.25	2.50
22-Jul-96	2.75	2.75	2.75
Average	2.50	2.64	2.57

Appendix G: MOE algal identification 2010 & 2008

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Water Monitoring & Reporting Section
Sport Fish & Biomonitoring Unit

August 20, 2010

MEMORANDUM

TO: Walter Shields
Sault Ste. Marie Area Office
Northern Region

FROM: Kaoru Utsumi
Technologist - phytoplankton

RE: Algal identification of 2 grab samples (10 to 20 feet apart) from Sellers Beach (east end) at Bright Lake where Pickle Lake enters, taken on July 20, 2010

Both samples (Sample #1 and Sample #2) contained very similar organisms in them although the Sample #2 showed more varieties of green algae.

In both samples, there was a filamentous green alga, *Spirogyra* quite commonly. *Spirogyra* is generally abundant in standing waters of neutral or slightly acidic pH. The genus usually exists as loose, floating mats and often forms widespread but non-toxic springtime blooms in freshwater ponds. In addition, the samples contained colonial green algae, *Sphaerocystis* and *Scenedesmus*. Green algae do not produce cyanotoxins. Moreover, there was a filamentous blue-green alga, *Trichodesmium* observed commonly. *Trichodesmium* is not known to produce cyanotoxins. In addition, potentially toxin-producing filamentous blue-green alga, *Aphanizomenon* was frequently seen in the sample. Some coiled fragments of another potentially toxin-producing filamentous blue-green alga, *Anabaena* was also noticed. *Anabaena* is known to produce hepatotoxins (microcystins). However, the ELISA result was negative for the presence of microcystins in both samples. *Aphanizomenon* and *Anabaena* are also known to produce neurotoxins (anatoxins) which ELISA does not test for. These can only be determined by mass spectrometry analysis. Finally, the samples contained diatoms (*Fragilaria*, *Synedra*, *Navicula*, etc.), bacteria and debris as well.

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Ontario

Water Monitoring & Reporting Section
Sport Fish & Biomonitoring Unit

October 3, 2008

MEMORANDUM

TO: Ron Dorscht
Sault Ste. Marie Area Office
Northern Region

FROM: Lynda Nakamoto
technologist - phytoplankton

RE: algae identification of grab sample from shore taken October 1, 2008 from Bright Lake, 482C Pioneer Rd., Huron Shores Tp., Algoma District

---

The sample indicated a bloom of the filamentous cyanobacterium *Aphanizomenon flos-aquae*. The filaments occur as bundles which resemble small grass clippings to the naked eye and often form a scum floating at the surface. It is a potential neurotoxin (anatoxin) producer. There was also a very small amount of the filamentous cyanobacterium *Anabaena* (also a potential anatoxin producer). Only a couple of colonies of the potentially hepatotoxin producing *Microcystis* were observed in the 2 mL of sample I examined. Some small colonial cyanobacteria were present including *Synechococcus* and *Chroococcus*. Small colonies of *Woronichinia* and *Snowella* were common.

In addition, small flagellate cryptophyte (*Cryptomonas*, *Rhodomonas*) and chrysophyte (e.g. *Chrysochromulina parva*) algae, colonial or single celled green algae (*Monoraphidium*, *Dictyosphaerium*, *Gloeocystis*, *Botryococcus*, *Coelastrum*), the desmid *Cosmarium* and diatoms (especially the filamentous diatom *Melosira*) were present. Ciliate protozoa were also observed.

Appendix H: Sediment data of individual cores

Cores were collected on 6 Sep 2010 unless stated otherwise. Original numeration is used.

Station	Moisture	LOI	TP	Fe-P	Calcium	Iron
	(%)		(mg/g dry weight)			
Main						
0-5 cm						
Oct 4, 2009				0.195		
1-Main Station 1-5 cm	76.70	9.32	1.00	0.155	4.24	33.60
5-Main Station 1-5 cm	71.20	8.61	0.75	0.217		
5-10 cm						
Oct 4, 2009				0.094		
2-Main Station 6-10 cm	76.00	8.59	0.91	0.189	4.55	31.50
6-Main Station 6-10 cm	70.30	7.70	0.64	0.046		
Basin						
0-5 cm						
3-Basin 1-5 cm	71.60	8.92	1.10	0.276	3.91	32.00
7-Basin 1-5 cm	74.70	8.72	0.89	0.180		
5-10 cm						
4-Basin 6-10 cm	69.60	8.55	0.72	0.117	3.92	29.00
8-Basin 6-10 cm	73.40	9.06	1.25	0.430		

Note: suspect extreme values are shaded

Appendix I: Fisheries Report MNR July 2010

Ministry of Natural Resources

From: Jeff Amos
Regional Fisheries Population Specialist
Ministry of Natural Resources
South Porcupine, Ontario.

RE: Broad-scale Monitoring, FMZ 10, (Bright Lake)

Dear Mr. Milito:

As you are aware, during the summer of 2010 crews from the Ministry of Natural Resources (MNR) conducted netting operations on Bright Lake; along with several other lakes across FMZ's 8 and 10. The netting and other monitoring activities undertaken were part of the broad-scale monitoring program which MNR began implementing across Ontario in 2008.

On Bright Lake we collected a variety of valuable information such as: fish species present, sex, age, length, weight, general fish health, fish contaminant levels, water quality information, and invasive species information. All of this information will be summarized in a State of the Resource Report which will be available no later than 2013, following completion of sampling and data analysis of the remaining lakes. In the meantime, we would like to provide you with the following preliminary results of the netting operations.

We appreciate your help and interest in this important program that will help to manage Ontario's fisheries. We encourage you to share the attached information with other interested parties.

Please feel free to contact Jeff Amos at (705) 235-1214 if you have further questions or concerns.

Sincerely,

Jeff Amos
Fisheries Population Specialist
Ontario Ministry of Natural Resources
Northeast Science and Information

Fisheries Report MNR July 2010, continued:

Bright Lake Summary

Sampled July 5th – 9th, 2010
25 nets set in depths ranging from 1m to 12m
No Spiny Water-flea were detected
17 species of fish were captured:

Longnose Gar	Bowfin
Rainbow Trout	Cisco (Lake Herring)
Rainbow Smelt	Northern Pike
White Sucker	Shorthead Redhorse
Spottail Shiner	Brown Bullhead
Trout-Perch	Rock Bass
Pumpkinseed	Smallmouth Bass
Largemouth Bass	Yellow Perch
Walleye	

Largest Walleye captured was:
1780 grams (4 lbs)
569mm fork length (22.4 inches)
Oldest Walleye captured was 12 years old.

Bright Lake Walleye length at age:

Age (years)	1	2	4	8	10
Average length (mm)	218.3	324.4	430.5	536.6	570.8
(inches)	8.6	12.8	16.9	21.1	22.5

Largest Northern Pike captured was:
1650 grams (3.7 lbs)
620mm fork length (24.4 inches)
Oldest Northern Pike captured was 6 years old.

Bright Lake Northern Pike length at age:

Age (years)	1	2	3	4	5	6
Average length (mm)	284.5	399.5	466.8	514.5	551.5	581.8
(inches)	11.2	15.7	18.4	20.3	21.7	22.9

Largest Smallmouth Bass was 386mm fork length, and 5 years old
Largest Largemouth Bass was 394mm fork length, and 5 years old

Appendix J: Internal load in Cottage Country

NEWSLETTER Canadian Society of Environmental Biologists

Internal Phosphorus Loading in Ontario Cottage Country or The Devil is in the Sediments

Revised from an article published in the Federation of Ontario Cottages Association's (FOCA) Lake Stewardship Newsletter

Gertrud Nürnberg, Ph.D., Freshwater Research, 3421 Hwy 117, Baysville, Ontario P0B 1A0 gkn@fwr.on.ca ***www.fwr.on.ca***

By now, everyone in Cottage Country (starting about 150 km north of Toronto on the Canadian Shield) has heard about phosphorus (P), the nutrient that makes the water green because it makes algae grow. Eutrophication, or the overabundance of nutrients in waters, is the single most important cause for the deterioration of the water quality in our lakes and rivers, unless they are acid-stressed. "Acid" lakes, which are very clear and have a pH below 6 or so, are not in danger of turning green, because they have other problems, like toxicity caused by heavy metals and acidity.

To keep eutrophication at bay, shoreline residents have been striving to reduce phosphorus inputs into their lakes. They have been instructed to use phosphate-free soaps and detergents, to not wash hair in the shallows or cars at the beach, and to keep the shoreline as natural as possible minimizing the need for fertilization. (Shoreline buffer zones are better than grass at adsorbing phosphorus in the runoff water after rain or snow melt and don't need to be fertilized.) Thus, ideally, the external input of phosphorus to a lake is kept to a minimum.

Of course, it was not always so. The early settlers of the cottage country did not know about eutrophication. Their outhouses and sinks drained, "conveniently," right into the stream. The potato and tomato fields needed a lot of manure on this poor soil, livestock drank right from the creeks (defecating at the same time), and the towns discharged any collected wastes right into the bay of the next lake. Much of these early inputs into the waterways were flushed downstream, but a proportion was retained at slow flowing and shallow locations and remains there now, a time bomb ready to be released.

What is the trigger? The trigger is anoxia, which means complete oxygen depletion. As long as the water directly over the sediments still contains oxygen (at least 1 to 2 ppb), phosphorus stays bound in the sediments. However, when oxygen is used up completely, the chemistry of the sediments changes, phosphorus is no longer bound to the sediments, and large amounts of phosphorus may be released into the overlaying water. This water eventually mixes with surface water, so that algae up in the sunlit water can thrive. The water becomes green. Phosphorus released from the sediments is called "internal phosphorus loading."

Internal P loading is a complicated process. While fertilization of bottom sediments in lakes and rivers is the prerequisite, chemical changes within the sediments and oxygen-free conditions above them all work together to release P in a form that is highly biologically available as phosphate (just like in a fertilizer).

On the Canadian Shield, where most of Ontario's Cottage Country is located, fertilized bottom sediments are still few. In other regions, for example, where former seas were situated (e.g., in the Great Lake/St. Lawrence basin), the soils were naturally P enriched even before European settlement. But the trigger, bottom anoxia, occurs naturally in many lakes in Cottage Country. Many of these lakes do not encourage mixing because of their shape, deep and small, or because their tea-like color traps sunlight in the warm surface water so that the bottom water remains cold. In addition, this brown stain enhances bottom water oxygen depletion as it is produced by organic material. When the organic material decomposes, it consumes oxygen. For example, in half of the lakes in the District of Muskoka, anoxia is so frequent in the bottom water it is as if the whole lake surface area was completely anoxic for 10 days per year. In more eutrophic lakes, bottom anoxia occurs more because of algae and other plankton that settle to the bottom and are consumed by bacteria that use up the oxygen in the process.

Important phosphorus forms

Phosphorus (P): Usually means total phosphorus, which is all phosphorus that can be analysed in a water sample. It includes phosphate, particulate forms, and other forms not easily available to be used by algae. Much external loading is comprised of all these forms.

Phosphate: A proportion of phosphorus that is directly available to plankton (algae, bacteria) in the water; it is usually below analytical detection limits in lakes on the Canadian Shield, except where internal loading occurs.

It is difficult to generalize the importance of internal load in lakes. The interplay between external and internal P loading is depicted as stages (Figure 1). Internal load was first described in highly eutrophic lakes in Europe and the USA (Stage 3), where, despite a major reduction of external load (usually by collecting and treating all waste water as point source reduction), in some lakes the P concentration did not decrease and water quality continued to deteriorate. More recently, it has been described in many other lakes even if it is not as obvious (Stage 2). Its quantification includes methods based on P budgets, P mass balance models, sediment incubation and analysis, and determination of anoxia. In general, it's been the consensus that internal loading may occur in more places than previously thought. Traditionally, it was only described in eutrophic lakes, as it usually takes a long time for sediments to become enriched and oxygen depleted enough to release P. But recent analyses has shown that oligotrophic systems on the Canadian Shield, like small deep lakes or those stained with organic acids, are vulnerable because of the natural occurrence

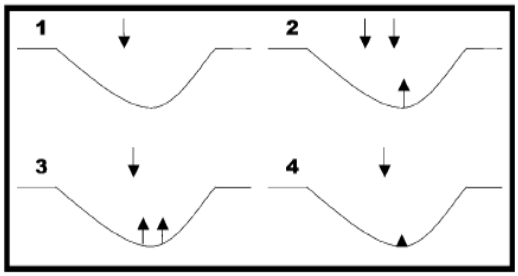

Figure 1. Presumed stages during the eutrophication process in lakes with respect to internal P load from the lake bottom (upwards arrow) in response to external load (downwards arrow). During Stage 1, external load happens, but no internal load. Even if the hypolimnia may be anoxic, there is not enough releasable P in the sediment surfaces to be released. In Stage 2 the external load increases, due to anthropogenic sources from development, and sediment P release will eventually commence, depending on the oxygen state of the sediment surfaces. Even when management efforts reduce the P load from the watershed as in Stage 3 internal load will still occur until the reductant-soluble sediment P has been flushed out (Stage 4).

of oxygen depletion; here, any P additions can potentially be released instantly and fertilize the water, perhaps creating cyanobacterial blooms.

Further Reading: Nürnberg, G.K. 2001. Eutrophication and Trophic State - Why does lake water (quality) differ from lake to lake? LakeLine (North American Lake Management Society) 21(1), 29-33.

Nürnberg, G.K., and LaZerte, B.D. 2004. Modeling the effect of development on internal phosphorus load in nutrient-poor lakes. Water Resources Research. 40, (1), W01105, DOI:01110.01029/02003WR002410.

Nürnberg, G.K. 2007. Internal phosphorus loading in Ontario Cottage Country or "the devil is in the sediments". Canadian Society of Environmental Biologists, Newsletter 64 (4), 11-12.

Appendix K: Calculations for internal loads (RR x AA)

determined from sediment concentration and active area

In this approach, summer internal load was determined as the product of release rate (RR) and active area (AA) according to (Nürnberg 2004, Equation 1)

$$L_{int} = RR \times AA \quad (1)$$

The actively releasing bottom area (possibly representing the sediment surface anoxia) was estimated from a model that considers summer surface TP concentration and morphometry to model active area (AA, Equation 2, Nürnberg 2009).

$$AA = -36.2 + 50.2 \log (TP_{summer}) + 0.762\ z/A_o^{0.5} \quad (2)$$

where TP_{summer} is average summer TP, and $z/A_o^{0.5}$ is a morphometric factor (with z, mean depth in m and A_o, lake surface area in km^2).

An average areal release rate (RR) was predicted from sediment TP concentration and reductant-soluble P fractions (Fe-P) to yield a long-term average internal load. In particular, RR was predicted from 0-5 cm sediment TP concentration (TP_{sed}) of the main station (B) according to Equation 3 (Nürnberg 1988, log, logarithm to base of 10).

$$\text{Log}\ (RR) = 0.8 + 0.76 \log (TP_{sed}) \quad (3)$$

Release rates were also tentatively computed from TP and Fe-P concentration ($Fe\text{-}P_{sed}$) of the main station (B) for the 0-5 cm sediment layer according to Equations 4 and 5 (Nürnberg 1988).

$$RR = -4.18 + 3.77\ \ TPsed \quad (4)$$

$$RR = -0.58 + 13.72\ Fe\text{-}P \quad (5)$$

Further RR was estimated from 0-10 cm TP and LOI content of the sediment (B) according to Equation 5 (Nürnberg 1988).

$$RR = 4.78 + 2.75\ TP - 0.177\ LOI \quad (5)$$

Above models were developed for the deepest sites of lakes and these results are reported here. Sediments from the shallower central basin that were collected as well are quite similar to those from the deeper location of Site B. This indicates that the sediments are not very heterogeneous and that the analysis is representative.

Results of AA, RR and internal load estimation are presented in Table 18 below.

Table 18. Release rate and internal load estimation results

Supportive variables	
Summer TP (µg/L)	14
$z/A^{0.5}$ (m/km^2)	1.4
Active Area (d/summer)	22
Predicted Release Rates (mg/m^2/summer)	
1. Sediment TP	-0.89
2. Fe-P fraction (Eq 4)	1.97
3. Sediment TP (Eq 3)	5.69
4. Sediment TP & LOI (Eq 5)	5.53
Internal Load estimates (mg/m^2/summer)	
1. Sediment TP	0
2. Fe-P fraction	44
3. Sediment TP (literature)	127
4. Sediment TP & LOI	124

Appendix L: Hydrology report from MNR Nov 6, 2009, first page

DRAFT

HARRIS CREEK AT BASSWOOD LAKE DAM NATURAL FLOW METRICS DATA SHEET

Station Information	
Site ID	2CC11
River Name	Harris Creek
Site Name	Basswood L. Dam
Region	Northeast
District	Sault Ste. Marie
Drainage Area	85.45 km²
Owner	-
Plant Capacity	-
Spill Capacity	-

Flow metrics are provided for the waterpower facility based on simulated natural flows as described in the draft *Waterpower Science Transfer Report 1.0* (MNR 2003). The target metrics provided are described in the *Aquatic Ecosystem Guidelines* (MNR 2002) and the *Waterpower Science Strategy* (MNR 2002). Metrics are based on simulated natural daily flow from 1978-2001 (24 yrs). The gauges used for the natural flow simulation were: i) 02CA002, Root River at Sault Ste. Marie (weight=0.6), ii) 02CD006, Serpent River above Quirke Lake (weight=0.4).

Annual:

I. Streamflow Time Series

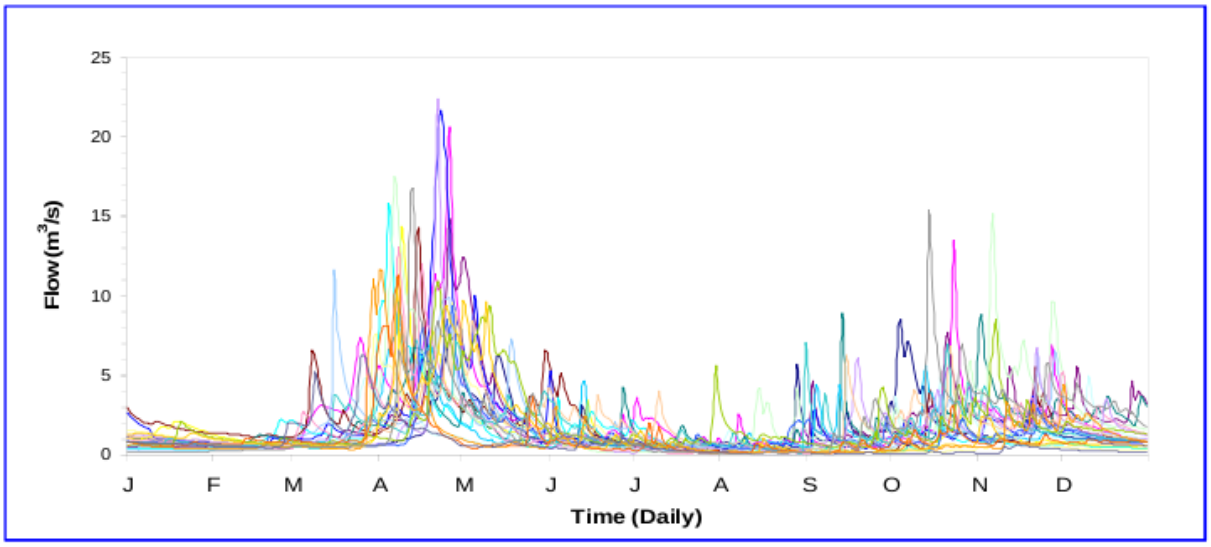

Figure 1: Annual daily flow hydrographs from 1978-2001.

Descriptive Metric	Value
Mean Annual Flow	1.51 m³/s
20% Time Exceeded Flow	2.13 m³/s
Median Flow	0.86 m³/s
80% Time Exceeded Flow	0.48 m³/s
Month of Max. Median Flow	April
Month of Min. Median Flow	August
Mean Rising Rate of Change of Flow	0.68 m³/s/day
Mean Falling Rate of Change of Flow	-0.28 m³/s/day
Extreme Low Flow Conditions:	
7-day-average low flow in 2-year return period, $7Q_2$	0.21 m³/s
7-day-average low flow in 10-year return period, $7Q_{10}$	0.09 m³/s
7-day-average low flow in 20-year return period, $7Q_{20}$	0.07 m³/s
Target Metrics	**Value**
Riparian Flows (Q_2 - Q_{20})	11.9 - 21.1 m³/s
Bankfull Flows ($Q_{1.5}$ - $Q_{1.7}$)	10.1 - 10.9 m³/s

Table 1: Annual flow metrics based on 24 years of data.

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Appendix M: In-lake restoration techniques

Sediment Dredging

Dredging of sediments is widely and routinely used in stormwater detention ponds and has also been occasionally recommended in the treatment of natural ponds and lakes (McComas 2003; Cooke et al. 2005). It would increase depth and could prevent future problems due to any substances kept in the sediments. However, Bright Lake is far too large to even attempt any partial dredging for water quality control and costs of the required toxicological assessment, dredging and disposal would be prohibitive. Besides, dredging is only successful when technologically difficult methods are employed that prevent the disturbance and re-distribution of the bottom sediments. Otherwise, water quality is reduced rather than enhanced by such a treatment.

Debris from milling operations

Even though some areas of Bright Lake appear to be littered with logs, bark, and saw dust from the logging era (Section 4.8.1), it does not appear feasible to remove such debris in the near future. Also, logs and other structure in the deeper water may improve fish habitat. Nonetheless, if all other recommendation and treatments fail, the removal of sawdust or debris should be re-evaluated for certain sections and bays of the lake after appropriate sediment analysis. There may be more evidence from future studies that prove the importance of such removal in the prevention of oxygen depletion and internal load.

Phosphorus precipitation and fixation

There are several restoration techniques involving chemical precipitation and adsorption of P at the sediment surface to prevent sediment P release and internal loading. Such techniques include well established applications of aluminum or iron and a novel treatment based on lanthanum laced clay (Phoslock). However, considering the size of Bright Lake and its relative remoteness, such treatment is not realistic.

Hypolimnetic withdrawal

In this treatment the P-rich hypolimnetic bottom water is withdrawn instead of the cleaner surface water after damming the outflow. This technique is useful in deeper lakes that stratify most of the growing season and where the deep basin is closely located to the natural outflow. However, compared to about 50 successful treatments in stratified lakes, it did not improve water quality in the two shallower polymictic lakes (Nürnberg 2007b), because of a lack of P accumulation in the deeper water. Because this is the case in polymictic Bright Lake hypolimnetic withdrawal would not be beneficial; in addition, the distance of the deep location of 4 km away from the outflow would make its cost prohibitive.

Application of algaecides

In small lakes and ponds, chemicals are sometimes used to kill excessive algae, but I usually do not recommend such a treatment. Costs would probably prohibitive in an application to a large lake such as Bright. While it may be immediately effective, it may need to be repeated several times per year. In general, it is preferable to treat the underlying cause of algal proliferation, i.e. overabundance of nutrients, rather than the symptom. Because all common algaecides are non-specific they can be quite toxic to other plants and organisms such as the macrobenthos, zooplankton and fish. In addition, when algae dye off, oxygen depletion occurs and can result in fish kills and internal P load. In Ontario, a permit from the MOE to purchase and perform

extermination is required, and the treatment must be performed by a licensed pesticide applicator.

Aquatic weed control

Bright Lake is a natural, fish supporting lake and a fair amount of macrophytes or water plants (sometimes referred to as "weeds") are necessary for the recruitment of fish and their food, the zooplankton. For examples, some studies suggest that a large area, the size of a quarter of the lake surface area, is ideal to support a healthy recruitment. Consequently, no macrophyte removal is proposed.

Some non-native, introduced weeds can imbalance nutrient and biological cycles and should be managed against (Section 7.2.3).