SWAN LAKE WATER QUALITY MANAGEMENT

Prepared by **Gertrud Nürnberg, Ph.D.**

Bruce LaZerte, Ph.D.

Freshwater Research

gkn@fwr.ca

Baysville, Ontario, P0B 1A0

Prepared for **The City of Markham, Ontario**

Date July 17, 2020

Respectfully submitted: 17, July, 2020

Dr. Gertrud Nürnberg

Disclaimer

Acknowledgement

Zahra Parhizgari and Kate Rothwell, Environmental Engineers, and Rob Muir and Rob Grech, Managers of the Environmental Services Department, City of Markham, provided steady support during the preparation of this report and their enthusiasm is gratefully acknowledged. Many previous staff of the City of Markham provided insights, knowledge, and skills to the long-term monitoring effort in Swan Lake.

Title photo: Apr 10, 2017 (Photo, Gertrud Nürnberg)

Executive Summary

Swan Lake in the City of Markham is a highly eutrophic (hyper-eutrophic) lake with cyanobacterial blooms (“bluegreen blooms”). These blooms can contain species that produce toxins with health effects on pets, livestock or humans. A detailed monitoring study in 2011-2012 and historic data (1993-2008) revealed that most of the water quality problems originated primarily from internal phosphorus (P) sources, such as the bottom sediments. After the determination of available treatment methods, a chemical treatment with Phoslock was conducted in the spring of 2013. Several post-treatment monitoring studies evaluated the effects and efficiency of the treatment in several reports and published peer-reviewed papers.

Although the Phoslock treatment improved water quality from hyper-eutrophy to eutrophy for two years after application, by 2016 water quality was as low as or lower than in the pre-treatment year 2011, and trophic state variables indicated hyper-eutrophic conditions again (Summary-Table 1). Phytoplankton biomass (as determined by chlorophyll concentration and Secchi disk transparency) were above pre-treatment measurements. Occasional cyanobacteria surface blooms were observed at specific sites of at least 7 genera of potentially toxic cyanobacteria. Microcystins were measured well above (3.7 times, 73 µg/L) the provisional federal guideline for recreational activities (20 µg/L).

Summary-Table 1. Trophic state of Swan Lake based on water quality of the monitored pre- (2011) and post-treatment growing periods (2013-2018) at Deep Site 3 or Dock Site 1.

	Swan Lake						Definition	
	2011	2013	2014	2016	2017	2018	Eutrophic	Hyper-eutrophic
Secchi Disk Transparency (m)	0.47	0.43	1.4	0.29	0.40	0.53	1 – 2.1	< 1
Total phosphorus (mg/L)	0.247	0.099	0.060	0.272	0.191	0.221	0.031 – 0.100	> 0.100
Total nitrogen (mg/L)	2.71	1.64	1.14	4.50	2.26	2.55	0.65 – 1.20	> 1.20
Chlorophyll *a* (µg/L)	32	52	12.6	111	61**	41**	9.1 – 25	> 25
Anoxia in polymictic lakes	---- severe ----		some	--------- severe ------------			occasional during summer stratification	
Anoxia (AF, d/summer)***	50	31	33	48	40	51	40 – 60	> 60
Modeled anoxia (AA, d/summer)****	90	70	59	92	84	87	Not applicable	

* For total nitrogen the sum of total Kjeldahl-N and nitrate concentration (mostly below the detection limit) was used when available, otherwise, just total Kjeldahl-N.
**Predicted from Secchi average according to a significant regression model (Appendix B).
***AF values are computed from dissolved oxygen profiles (Appendix A). They present the number of days an area equal to the total surface area of Swan Lake is overlain by anoxic water per growing period. In shallow, wind-mixed lakes, such as Swan Lake, the (observed) AF presents an underestimate with respect to the trophic state classification that is based on thermally stratified, deeper lakes.
****AA values are based on morphometry and long-term TP concentrations according to a significant regression model (Appendix A). AA values more likely present the anoxic sediment area involved in sediment release than AF values, which describe anoxia throughout the lake water.

Because of the deteriorated conditions closer to shore, 2017 monitoring added two shore line sites (Site 1 and 2), in addition to (2017) or instead of (2018) the previously monitored open water site (Site 3) accessible by boat only. 2017 water quality characteristics of the three monitoring sites were quite similar and we concluded that Site 1 (main dock) water was representative of the nutrient and trophic state of Swan Lake and could be used in comparison with previous lake data.

A total phosphorus (TP) mass balance analysis using hydrological budgets provided by City of Markham staff was conducted for 10 years, 2009-2018, to determine the TP contributions from specific sources and to estimate internal load. The TP inputs from external sources were estimated on an annual basis, using monitored data for phosphorus and water volumes provided by the City. Waterfowl contributed about 55% and the immediate lake shore contributed 38% to the external load long-term average of 29 kg/yr (range: 20-50 kg/yr). The remaining 7% of external load was contributed by the stormwater management ponds and precipitation unto the lake.

In comparison, internal load averaged 41 kg/yr (range 0-60 kg/yr). Internal load was a low 25 kg in 2013 when treatment was applied in early May and assumed to be 0 kg in 2014 (because of low water TP concentrations). The mass balance model estimated internal load to be above 35 kg in all other years including pre- and post-treatment periods indicating a diminishing treatment benefit. A mass balance model converted the loads to water TP concentrations (by considering P settling according to annual hydrological conditions), so that the contribution of individual sources to the lake concentration could be assessed (Summary-Table 2).

Summary-Table 2. Modelled TP contribution from external and internal P sources (presented as concentration and percent of total TP input).

P Source	2009-18		2016-18	
	mg/L	%	mg/L	%
Atmospheric deposition	0.003	1%	0.003	1%
SWM ponds	0.003	1%	0.003	1%
Immediate lake shore	0.036	16%	0.033	13%
Geese	0.054	24%	0.078	31%
Internal (sediment)	0.132	58%	0.136	54%
Total input	0.228	100%	0.254	100%

Internal P loading usually stems from former external inputs which are stored in bottom sediments, and it may originate in nutrient-rich previous land fill and recent inputs from water fowl feces in Swan Lake. P released from the bottom sediments has a high biological availability, and its release during elevated water temperature in the summer increases its effect on summer-fall lake water quality. For these reasons, the management of internal load is more promising for immediate results than that of external load, although long-term management must also attempt to reduce external sources to prevent further increase in legacy sediment P.

Even by just considering the relative contribution from the different TP sources (Summary-Table 2) without acknowledging the different bioavailability, the importance of various treatments can be assessed. For example, with a treatment that intercepts P release as in 2014 (after the 2013 Phoslock application), the resulting TP concentration would be below 0.100 mg/L, indicating eutrophic, rather than hyper-eutrophic conditions (Summary-Table 1). Consequently, phytoplankton would be diminished and Secchi disk transparency improved, possibly to the 2014 growing period of 1.4 m also indicative of eutrophy. In contrast, a 50% success in geese and runoff management that decreases TP from these main external sources by half would only decrease TP concentration by 0.056 mg/L (half of the concentration attributed to geese and shoreline in 2016-18) yielding still a TP concentration of almost 0.200 mg/L.

Based on this understanding, long-term management goals are proposed as a growing period TP average of below 0.100 mg/L and an average Secchi transparency of above 1.0 m. In the short-term, we determined a more appropriate (interim) goal from the relationship between TP and Secchi observed in Swan Lake. These include the growing period average of 0.150 mg/L or less of TP and a Secchi depth of 0.45 m or better.

We suggest a series of triggers that would lead to an initiation of appropriate management actions. Our proposed triggers are:

1. The surface bloom of a potentially or proven toxic strain of cyanobacteria, confirmed by a licenced or Provincial (MECP) lab or by Abraxis strips to trigger direct attention.
2. The occurrence of two blooms within a period of four years that cover at least 25% of Swan lake area.
3. Water quality not compliant with the interim goal of growing period average 0.15 mg/L total phosphorus concentration in the surface mixed layer.
4. Water quality not compliant with the interim goal of growing period average 0.45 m Secchi disk transparency.

While a single occurrence of cyanotoxin requires action (posting signage), only the average conditions concerning water quality measured as nutrients and Secchi disk transparency over a growing period (May/Jun - Sep) should trigger more comprehensive actions. The proposed goals are based on monitored data, consequently we present a detailed monitoring plan.

The proposed triggers 3 and 4 were tripped every year since 2016, and we present a review of applicable management approaches to deal with (a) external and internal TP loading, which can be judged the cause for the water quality deterioration, and (b) the overabundance of phytoplankton and the potential toxicity of cyanobacteria, which likely are the symptoms and consequences of the high TP levels. Triggers 1 and 2 cannot be evaluated for lack of information on the spread of cyanobacteria.

Detailed recommendations include (Summary-Table 3):

- A treatment to address internal loading as sediment P release. Sediment analysis will assist with proper dosing.
- Continued water fowl management.
- Continued water quality monitoring at the two shore sites.
- Determination and potential management of the abundance of bottom dwelling fish.
- Application of best management practices to decrease the nutrient contribution from the shoreline.
- Investigation of P load from historic dumpsites.

Two treatments are the most important and promise to be most effective: continued water fowl management and repeat internal load abatement (Summary-Table 3).

As preferred treatment to abate the internal P loading, we chose another Phoslock application. This treatment provided at least two growing periods of improved water quality and successfully treated sediment P release according to the mass balance analysis. There was no obvious negative effect from the previous treatment. Estimated cost for such a treatment is a total of $171,000. Treatment may need to be repeated in several years, depending on the successful management of the external P sources. A five-year repetition is suggested, with adjusted frequency according to the occurrence of the triggers.

An alternate treatment that includes aluminum compounds (poly aluminum chloride) may also help to reduce internal loading in the short run. Concerns of public perception and non-committal statements by Ontario Ministry of the Environment, Conservation and Parks (MECP) render it less recommendable, despite possibly lower costs.

To ensure effectiveness, we suggest that the dosage be based on the mobile P sediment fractions that can be determined by a specialized commercial laboratory (at a cost of about $20,000). Fish management involves a fish survey and fish salvage at a cost of $5,000 and $20,000. The annual costs for the long-term continuous waterfowl management and water quality monitoring are $13,500 and $12,000. Treatment costs of shoreline runoff through naturalization are estimated at $100,000. Costs of investigation of phosphorus load from the historic dumpsite are around $20,000. Detailed costs for 2021, including 25% contingency are presented in Summary-Table 4.

Following the initial treatment and other short-term measures, a long-term strategy will need to be developed to maintain and enhance Swan Lake's water quality in a sustainable manner. This strategy may include repeat treatments, continuous management of geese and fish, as well as additional measures.

Summary-Table 3. Evaluation, implementation, and estimated cost of recommended actions

Recommended Tasks	Importance*	Comment	Approximate Cost
Internal Load treatment	1	One application Incl. monitoring, evaluation	$171,000 (Table 27)
Fish management	1	Once (but $5,000 annually)	$25,000 (Section 6.3)
Sediment quality determination	2	Once	$20,000 (Table 26)
Water fowl management	1	Continuous	$13,500 (Table 28)
Water quality monitoring	2	Growing season	$12,000 (Section 6.6)
Shoreline BMPs	3	Naturalization	$100,000 (Table 28)
Historic dumpsites investigation	3	Five boreholes, testing for phosphorus	$20,000 (Table 28)

*Valued 1-3 with "1" most important

Summary-Table 4. Costs of short-term strategy for 2021 *(long-term management not included)*

Item	Total Cost
Implementation Plan	$10,000
Material (Phoslock)	$105,000
Application	$31,000
Enhanced Monitoring and Evaluation	$25,000
Fish Management	$25,000
Contingency (25%)	$49,000
Total cost	**$245,000**

Table of Contents

Tables

Figures

Glossary

Acronyms

HC, Health Canada

MECP, Ontario Ministry of the Environment, Conservation and Parks

MDDELCC, Ministère du Développement durable, de l'Environnement de la Faune et des Parcs

MNRF, Ontario Ministry of Natural Resources and Forestry

Definitions

Anoxic area factor, AA (days/summer or days/year): active period and area that releases P and contributes to internal load

Chlorophyll a: A measure of algae biomass, the green algal pigment. This measure of algal biomass in lake water is prone to analytical errors and its standardization is difficult, so that accuracy and precision are often low.

Cyanobacteria: Often called *bluegreens* or *bluegreen algae*, although they belong to bacteria. They can produce toxins that can create health effects if ingested in quantity (life stock, pets).

External load, L_{ext}: The sum of annual TP inputs from all external sources, i.e. stream, non-point and point sources, precipitation and groundwater. Units are in kg/yr or in mg per square meter of lake surface area per year ($mg/m^2/yr$). External load is a gross estimate. Much of its phosphorus is in a chemical form that is not immediately available to algae.

Internal load, L_{int}: Annual TP inputs from internal sources, i.e. the sediments. Units are in kg/yr. Gross estimates are usually used, but net estimates, based on mass budgets, can also be calculated. Most of the TP in L_{int} is in a chemical form (phosphate) that is highly available to phytoplankton and bacteria.

Limiting nutrient: Algae, bacteria and phytoplankton in lakes are usually nutrient limited, so that any addition of the bioavailable form would increase such biomass. The nutrient that elicits the largest response is called the limiting nutrient (usually phosphorus or nitrogen).

Polymixis: The mixing regime in lakes and reservoirs that describes frequent (daily to weekly in the summer) mixing of the whole water column. Swan Lake is polymictic because of its relatively shallow depth.

Secchi disk transparency: The depth (m) at which the round black and white disk disappears is an integrated measure of algal biomass. Because its use is wide-spread many relationships with nutrients and chlorophyll concentration from other lakes are available (as regression equations).

Sediment oxygen demand (SOD): organically enriched bottom sediment takes up oxygen from the overlaying water which creates anoxic conditions

Soluble reactive P, SRP: soluble fraction of TP that consists mostly of the biologically available phosphate

Total phosphorus, TP: All phosphorus (P) that can be analyzed in a water or sediment sample. It includes phosphate (highly available for algae), particulate forms (includes algae and non-living suspended particles), and forms not easily available to algae.

Total nitrogen, TN: TN is the sum of Kjeldahl-N (TKN), which includes ammonia (NH_3) and organic N compounds, and of Nitrite & Nitrate (NO_2&NO_3).

1 Introduction

Swan Lake in the City of Markham used to be a highly eutrophic (hyper-eutrophic) lake with cyanobacterial blooms ("bluegreen blooms"). These blooms often contained species that can produce toxins with health effects upon ingestion in pets and livestock or humans (Chorus and Bartram, 1999). A detailed monitoring study in 2011-2012 (Nürnberg and LaZerte, 2012) and inspection of historic data (1993-2008) revealed that most of the water quality problems, including algal blooms and low Secchi disk transparency, originated primarily from internal P sources, especially bottom sediments. After the determination of available treatment methods, a chemical treatment with Phoslock was proposed as it was acceptable to regulatory agencies because of its proven lack of toxicity. Consequently, Phoslock treatment was conducted in the spring of 2013. A total of 25.2 tonnes of Phoslock at a rate of 4.6 metric tonnes/ha was applied during several days from 29 April – 1 May 2013. A detailed 2-year post-treatment monitoring study (2013 and 2014) evaluated the effects and efficiency of the treatment and was reported in two reports and two published peer-reviewed papers (Nürnberg, 2017; Nürnberg and LaZerte, 2016, 2015, 2014).

Although the Phoslock treatment improved water quality from hyper-eutrophy to eutrophy for two years after application, by 2016 water quality was as low as or lower than in the pre-treatment year 2011, and trophic state variables indicated hyper-eutrophic conditions again (Nürnberg and LaZerte 2017). Especially nitrogen concentrations and phytoplankton biomass (as determined by chlorophyll concentration and Secchi disk transparency) were above pre-treatment measurements. While there were no cyanobacteria surface blooms observed during routine visits at the open water deep site, a sample collected at the eastern shore on Aug 31, 2016 included a high biovolume of at least 7 genera of potentially toxic cyanobacteria. The tested microcystin concentration of 73 µg/L was almost 3.7 times the provisional federal guideline for recreational activities is 20 µg/L (Health Canada, 2009).

The main objective of this study is to define a short term and a long-term strategy for water quality management in Swan Lake. This strategy is to be based on the understanding of the current state of Swan Lake, factors affecting water quality, and management goals and objectives. To accomplish this, we describe the development of Swan Lake's water quality based on the review of historic monitoring data in Task 1 (Section 2). The contribution of various sources to the phosphorus mass balance is determined in Task 2 (Section 3). Task 2 helps separate external from internal loads. In particular, influences from manageable P sources (internal load, waterfowl, shoreline) are separated from those that cannot be changed (climate-related loads, i.e., precipitation) in support of the following tasks: Task 3, *Development of Management Goals and Targets* (Section 4), and Task 4, *Development and Evaluation of Management Approaches* (Section 5). Finally, a detailed implementation plan for the preferred approaches is presented as Task 5 (Section 6).

2 Task 1 Review of Historic Monitoring Data - Swan Lake characteristics and water quality

Swan Lake in the City of Markham (Table 1) is a former gravel pit and dump site, located at 43°54' 79°15' and 208.35 m above sea level. It was operated as sand and gravel pit until 1970 and filled with soils from construction sites around 1980. Development started in the mid-nineties and included a low-density resort-like senior citizen community serviced by two stormwater management ponds.

Table 1. Average morphometry

Characteristics	Value
Watershed area, A_d (ha)	38.65
Shore line area involved in runoff (ha)	9.6
Surface area, A_o (ha)	5.48
Volume (10^3 m^3)	102
Mean depth, z (m)	1.86
Maximum depth (m)	4.4
Morphometric ratio, $z/A_o^{0.5}$ (m/km)	7.9

2.1 Monitoring sites and effort

In lake studies (limnology), usually the open water at the deepest location is monitored because its water quality conditions are deemed representative of the entire lake. Empirical relationships and predictive models are typically based on such sites. Accordingly, a station at the deepest point in the lake was set as the main monitoring site when a detailed monitoring program was introduced in 2011 (Nürnberg and LaZerte, 2012) and was monitored until 2017 (Site 3 in Figure 1). But the deepest site in Swan Lake can only be reached by boat and cannot be monitored by City of Markham staff. Using a location accessible from the shore is cost effective as it can be monitored by City personnel. In addition, the water quality close to shore is potentially more important because of its closeness to lake user and their pets. As well, Swan Lake is rather shallow and only 4.4 m deep at maximum volume at the deepest site.

Therefore, two shore sites were installed in 2017 for comparison with the deep site monitored previously (Figure 1). The 2017 study concluded that the water quality characteristics (nutrients, temperature and oxygen profiles, and Secchi transparency) at the three monitoring sites were quite similar and the water at Site 1 (at the southern dock) was representative of the nutrient and trophic state of Swan Lake (Nürnberg and LaZerte, 2018). (Similarity of the surface water and the small influence of the deepest layer at Site 3 can be explained by the small water volume of only 2.5% that is deeper than 3 m.) Consequently, the data of Site 1 are comparable with previous lake data of Site 3 and the sampling effort since 2017 concentrated on the shore line sites. A third site at the northern shoreline, “Bridge” (Site 2, Figure 1), was monitored since 2017 as it reflects relatively deteriorated conditions at the embayment of relatively stagnant water.

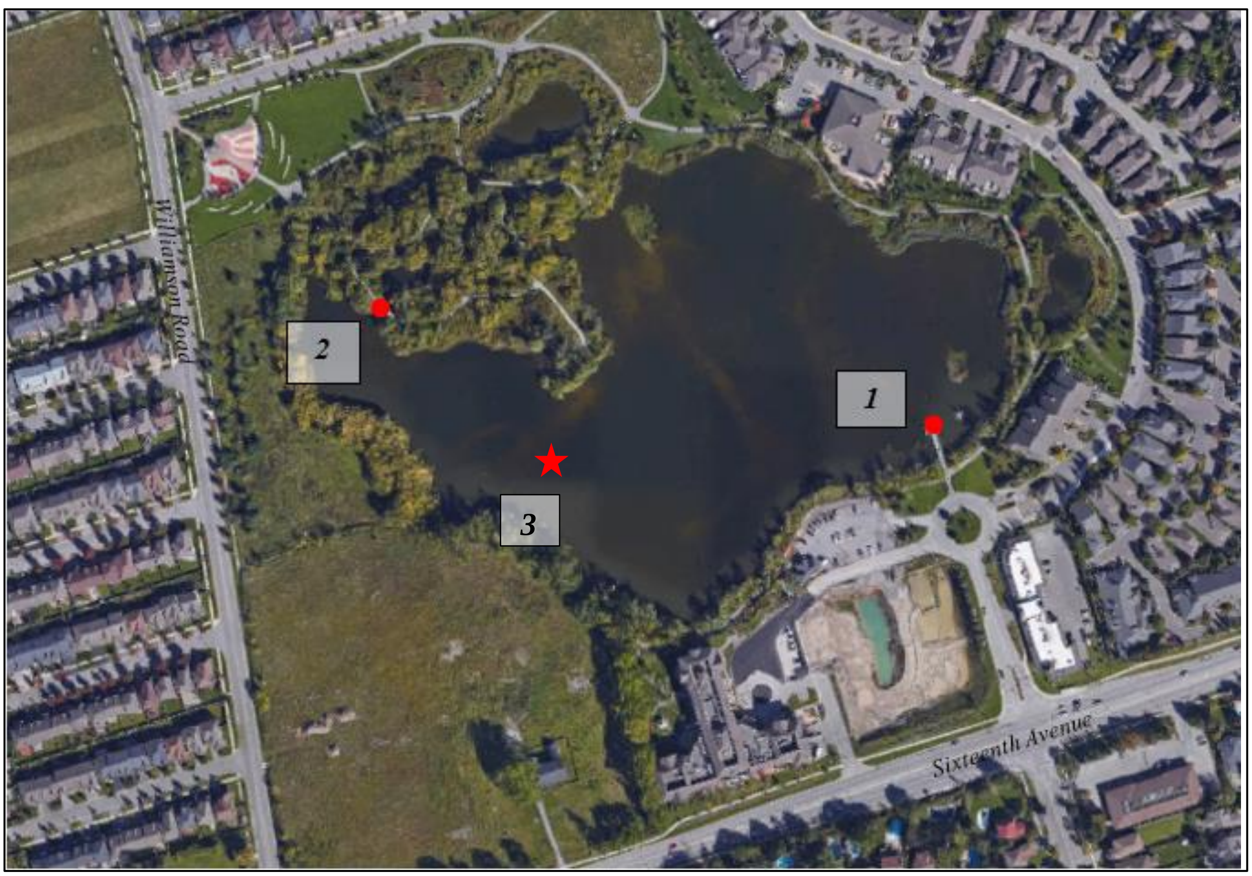
The monitoring site “Dock” (Site 1, Figure 1) and “Deep” (Site 3, Figure 1) were about 3 and 4 m deep respectively. Temperature and dissolved oxygen (DO) profiles were taken every 0.5 m in 2011- 2017 at Site 3 and since 2017 at Site 1. The nutrients total phosphorus (TP), occasionally, soluble reactive P (SRP, also called ortho-phosphate), ammonia (NH_3), nitrate (NO_3) or nitrite and nitrate (NO_2&NO_3), and total Kjeldahl N (TKN) were analyzed at 3-5 separate depths. Total N

(TN) was computed from TKN and NO_2&NO_3 or NO_3. (Because NO_2 is usually extremely small and below the detection limit, it was not always monitored.) When analyzed, dissolved organic carbon (DOC), chloride (Cl), and water colour were determined for surface (between 0 and 0.5 m) water. Phytoplankton biomass and identification, the green pigment chlorophyll *a* and the cyanotoxin microcystin were monitored occasionally throughout the years. (Specific methods are presented in the Freshwater Research reports 2012-2018 by Nürnberg and LaZerte).

The monitoring site, "Bridge" (Site 2), was only 0.45-0.65 m deep and probably mixed at most times. Water samples were taken at mid-depth at Site 2. Shore sites were monitored from April to November in 2017 and 2018 every 2 to 4 weeks by City staff. The deep Site 3 was monitored in 2011, 2013, 2014, 2016 and twice in 2017 by consultants or the Toronto and Region Conservation Authority (TRCA). Chemical and phytoplankton analyses were conducted in the laboratories of the Ontario Ministry of the Environment, and Parks (MECP) and accredited commercial laboratories.

Secchi transparency was determined when and where other monitoring occurred. Waterfowl number, especially that of Canada geese (*Branta canadensis*) on and around Swan Lake, were determined frequently (at least twice per week) from April to November by a goose deterrent consultant and City of Markham staff in 2016, 2017, and 2018 and less frequently in 2014 and 2015.

Figure 1. Monitoring sites: 1, Dock; 2, Bridge; 3, open water deep site *(Revised from City of Markham, 2017).*



2.2 Water quality

Section 2.2 summarizes the water characteristics and trends through the 2011-2018 period of intensive monitoring. In particular, the recent water quality are explored. Year-specific data analysis and interpretation were provided in the previous reports and memos by Nürnberg and LaZerte to the City (2012, 2014, 2015, 2017, 2018). A scientific paper, summarizing the evaluation of the Phoslock treatment for trends in 2013 and 2014, was published in a peer-reviewed journal (Nürnberg and LaZerte, 2016).

2.2.1 Trophic state classification and water quality since 2011

Based on several water quality variables a lake can be classified with respect to its trophic state. Clean pristine and clear lakes and ponds 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 and ponds that have a high nutrient load from the watershed and from the sediments are hyper-eutrophic, showing extremely high nutrient and algae concentrations including toxin-producing cyanobacteria, high turbidity and an oxygen deficit in their bottom waters (hypolimnion) when conditions are stagnant.

Swan Lake’s trophic state variables indicated hyper-eutrophic conditions in the pre-treatment monitoring period of 2011. Phytoplankton indicators (Secchi transparency and chlorophyll concentration) were not appreciably changed in the first post-treatment growing season 2013, even though nutrient concentrations had declined. Trophic state changed drastically to the better classification of eutrophy in 2014 for all related variables, but regained hyper-eutrophy in 2016, 2017, and 2018 (Table 2). Spatial and temporal variation of the individual trophic state variables is described in detail in the following sections.

Table 2. Trophic state categories (Nürnberg, 1996) based on water quality of the monitored pre- (2011) and post-treatment growing periods (2013-2018) at Deep Site 3 or Dock Site 1.

	Swan Lake						Definition	
	2011	2013	2014	2016	2017	2018	Eutrophic	Hyper-eutrophic
Secchi Disk Transparency (m)	0.47	0.43	1.4	0.29	0.40	0.53	1 – 2.1	< 1
Total phosphorus (mg/L)	0.247	0.099	0.060	0.272	0.191	0.221	0.031 – 0.100	> 0.100
Total nitrogen (mg/L)*	2.71	1.64	1.14	4.50	2.26	2.55	0.65 – 1.20	> 1.20
Chlorophyll *a* (µg/L)	32	52	12.6	111	61**	41**	9.1 – 25	> 25
Anoxia in polymictic lakes	---- severe ----		some	--------- severe ------------			occasional during summer stratification	
Anoxia (AF, d/summer)***	50	31	33	48	40	51	40 – 60	> 60
Modeled anoxia (AA, d/summer)****	90	70	59	92	84	87	Not applicable	

* For total nitrogen the sum of total Kjeldahl-N and nitrate concentration (mostly below the detection limit) was used when available, otherwise, just total Kjeldahl-N.
**Predicted from Secchi average according to a significant regression model (Appendix B).
***AF values are computed from dissolved oxygen profiles (Appendix A). They present the number of days an area equal to the total surface area of Swan Lake is overlain by anoxic water per growing period. In shallow, wind-mixed lakes, such as Swan Lake, the (observed) AF presents an underestimate with respect to the trophic state classification that is based on thermally stratified, deeper lakes.
****AA values are based on morphometry and long-term TP concentrations according to a significant regression model (Appendix A). AA values more likely present the anoxic sediment area involved in sediment release than AF values, which describe anoxia throughout the lake water

2.2.2 Mixing regime, temperature, and dissolved oxygen

Profiles of temperature and dissolved oxygen (DO) indicate that Swan Lake thermally stratifies during the summer despite its shallow depth. This pattern of stratification and anoxia remained similar after the treatment. For example, Deep Site 3 was stratified and anoxic below the thermoclines (depth at largest temperature difference) at 2.5 m on 25-Aug, at 3 m on 30-Aug, and at 4 m on 15-Sep in 2011 before the treatment (Nürnberg and LaZerte, 2012). Similarly, hypoxia was extreme in 2016, when DO below 3 mg/L occurred in layers below 1-1.5 m in July and August 2016 at Site 3 (Figure 2). Hypoxia also occurred below 2 m at Dock Site 1 from June to September in 2017 and 2018 (Table 3).

Figure 2. Temperature in degree C (A) and dissolved oxygen concentration in mg/L (B) contours at Deep Site 3 in 2016, note the different time axis.

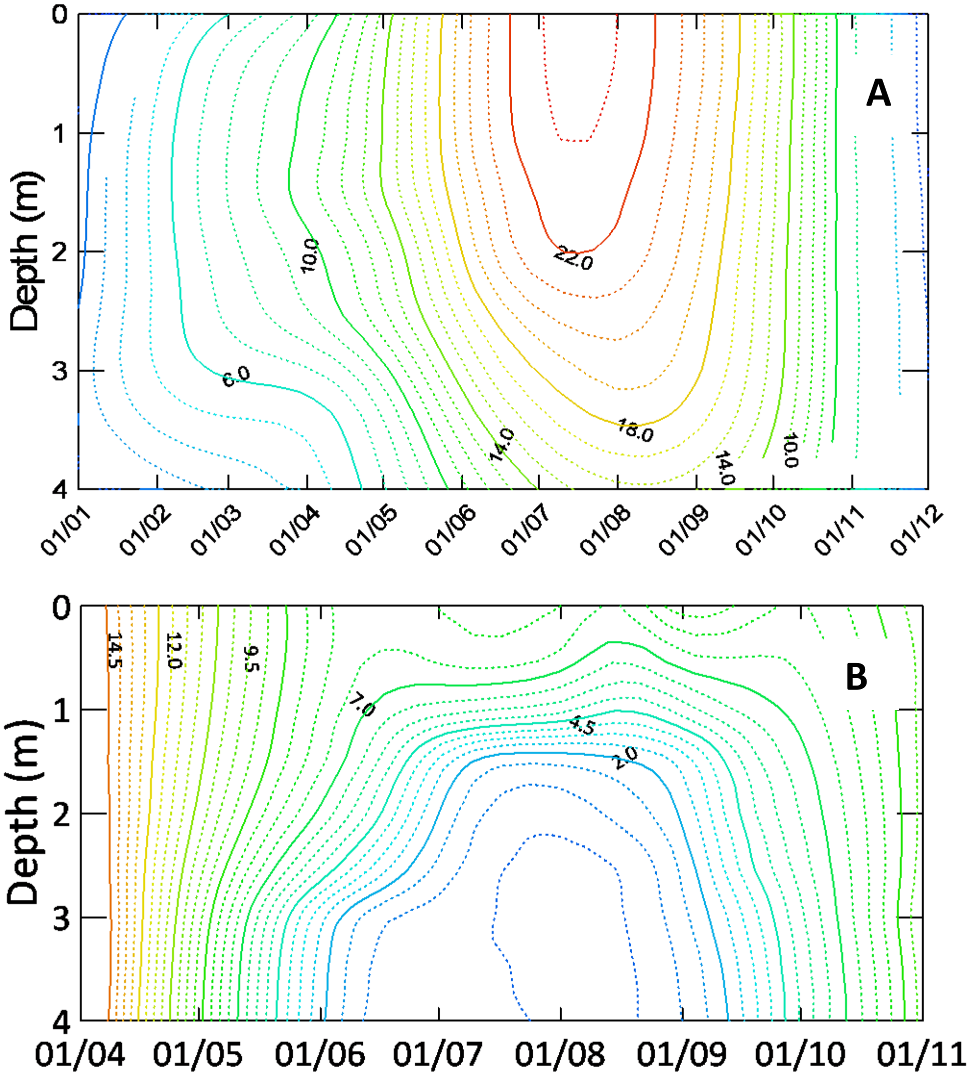


Table 3. Depth below which the DO concentration is less than 3 mg/L at Dock Site 1 and Deep Site 3. (Dates when DO was above 3 mg/L are indicated as ">3 DO")

Date	Upper depth for DO < 3 mg/L		*Date*	Upper depth for DO <3 mg/L
	Dock Site 1	Deep Site 3		Dock Site 1
10-Apr-17	>3 DO	Na	*19-Apr-18*	>3 DO
08-May-17	>3 DO	Na	*16-May-18*	>3 DO
05-Jun-17	2.0 m	Na	*01-Jun-18*	0.5 m
12-Jun-17	1.5 m	Na	*15-Jun-18*	>3 DO
28-Jun-17	2.5 m	2.0 m	*29-Jun-18*	2.0 m
10-Jul-17	2.5 m	Na	*17-Jul-18*	3.0 m
27-Jul-17	2.0 m	Na	*31-Jul-18*	2.0 m
09-Aug-17	2.0 m	Na	*15-Aug-18*	2.0 m
24-Aug-17	>3 DO	Na	*30-Aug-18*	3.0 m
31-Aug-17	>3 DO	2.5 m	*14-Sep-18*	2.0 m
18-Sep-17	1.5 m	Na	*28-Sep-18*	>3 DO
12-Oct-17	>3 DO	Na	*23-Oct-18*	>3 DO
24-Nov-17	>3 DO	Na	*28-Nov-18*	>3 DO

Na, data not available

Low DO concentrations (hypoxia) are not confined to the bottom water but often spread throughout the water column during all recorded growing periods. Typically, temperature changed gradually with depth without any evidence of a distinct change (thermocline). Such pattern is consistent with upwelling of anoxic water with nutrients and reduced substances released from the bottom sediments in a hyper-eutrophic system.

When measured, anoxia was also detected in the winter under ice throughout the water column (Jan 8, 2014) and in early spring in the open water (March 20, 2012), when DO was less than 1.5 mg/L below 2 m. Such wide-spread hypoxia at low temperature only occurs under severely eutrophic conditions, when the sediment has a high oxygen demand (i.e., takes up oxygen which creates anoxic conditions) and is extremely organically enriched.

Wide-spread anoxia despite the relative shallowness is obvious by the anoxic factor (AF) that describes the days an area equal to the surface area is anoxic (Appendix A). The values represent eutrophic conditions, except for the two post-treatment values in 2013 and 2014, that present mesotrophy (Table 2). It appears that the Phoslock treatment may also have helped decrease the spread of anoxia in Swan Lake.

Long-lasting anoxia is also predicted by a model that quantifies the days a sediment area is anoxic from growing period TP concentration and morphometric characteristics (Appendix A). The high values present hyper-eutrophic conditions, except for a border line value for eutrophy in 2014 (Table 2). The high value of the morphometric ratio (7.9 m/km, Table 1) indicates the high likelihood of stratification despite the shallow depth.

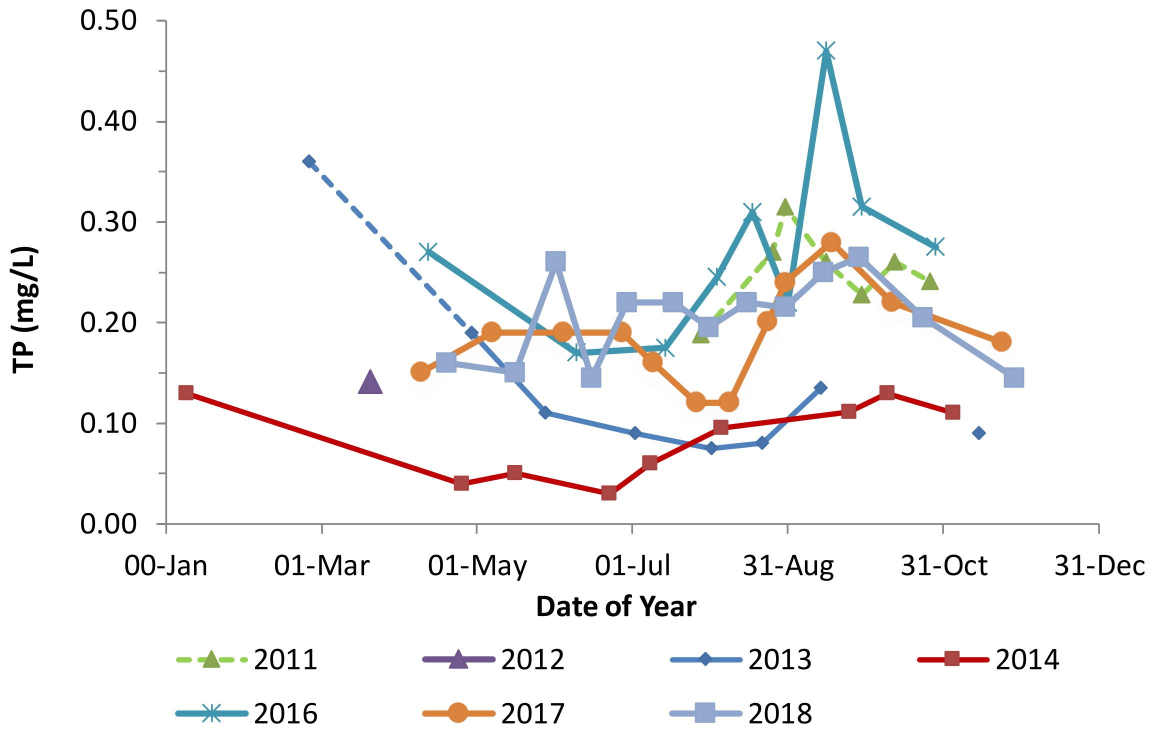
The pattern of low DO in the bottom water and occasionally throughout the summer and winter water column indicates severe eutrophication and the potential of sediment P release, unless release is interrupted by a treatment.

2.2.3 Nutrients

2.2.3.1 Phosphorus

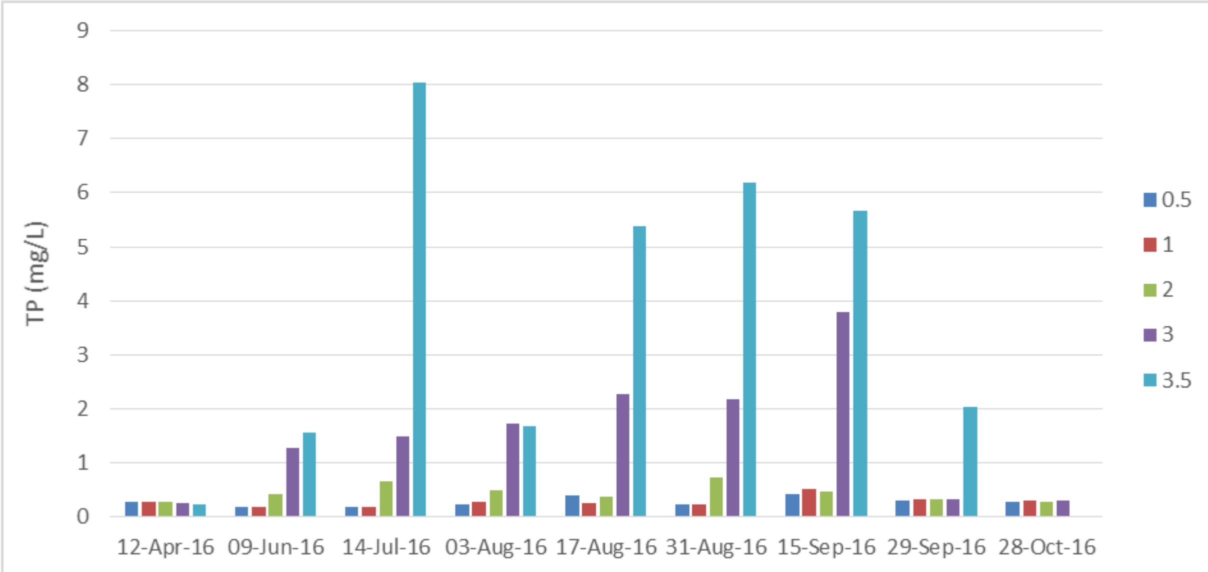
Total phosphorus is the most important trophic state indicator in P-limited Swan Lake. This is also the nutrient that was targeted with the Phoslock treatment. Average growing period (May - October) TP concentrations indicate hyper-eutrophic conditions in all monitored years except for the two post-treatment years 2013 and 2014 (Figure 3, Table 2). TP concentration typically decreases after spring and increases in late summer and fall, partially caused by sediment P release during summer anoxia and the arrival of migratory geese. TP tends to decrease in the fall and winter, but can still remain high at 0.15 mg/L and above.

Figure 3. Comparison of total phosphorus (TP) in the surface layer (0-2 m) for 2011-2018 at deep Site 3 (2011-2017) and Dock Site 1 (2017 and 2018). The Phoslock treatment occurred 29 April – 1 May 2013. (Pre-treatment data are indicated as broken lines.)



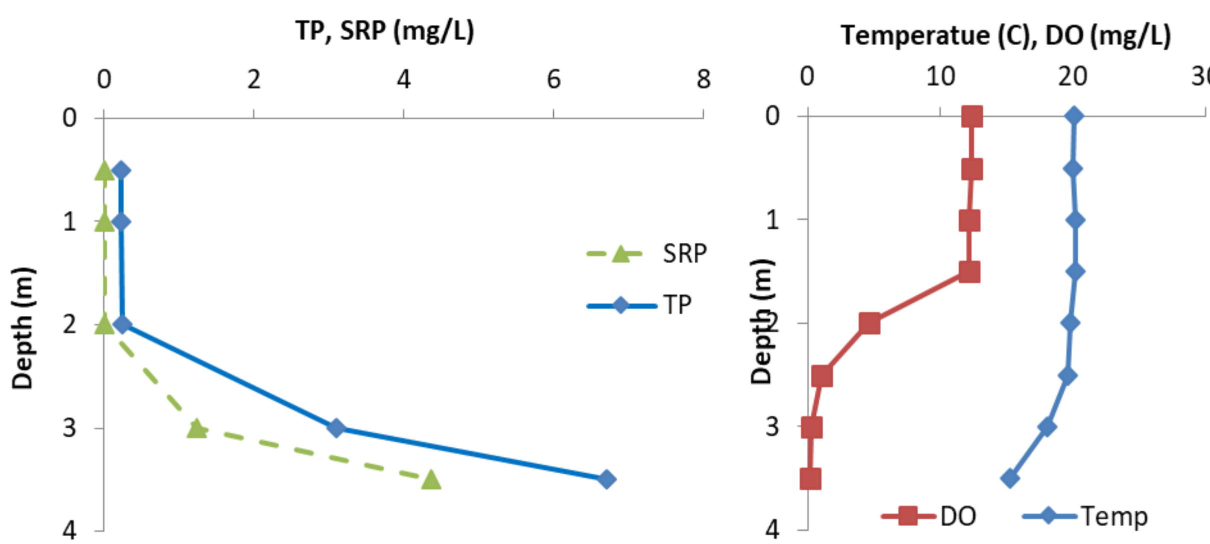
When there is sediment P release from internal P loading, the surface TP concentration increase throughout the growing season is accompanied by TP increase with depth. For example, deep TP concentrations were especially high in 2016 (measured at Deep Site 3, Figure 4).

Figure 4. Total phosphorus (TP) concentration measured at 5 depths in 2016.



The coincidence of stratification and bottom water anoxia with the large proportion of SRP (likely bioavailable orthophosphate chemically released from reduced sediments, Figure 5) support the idea that elevated TP in the summer and fall is caused by sediment released P (Section 2.5.1). This source may be supplemented by the enhanced presence of migratory geese in recent years (Section 3.2).

Figure 5. Total phosphorus (TP), soluble reactive P (SRP), temperature, and dissolved oxygen (DO) profiles measured on Aug 31, 2017, Deep Site 3.



During the early summer surface (0-1m) SRP (i.e., the immediately bioavailable form of phosphorus resembling orthophosphate) was usually below the detection limit of 0.010 mg/L at

Deep Site 3 and Dock Site 1. The low SRP values indicate P limitation, probably as a consequence of high nitrogen concentrations.

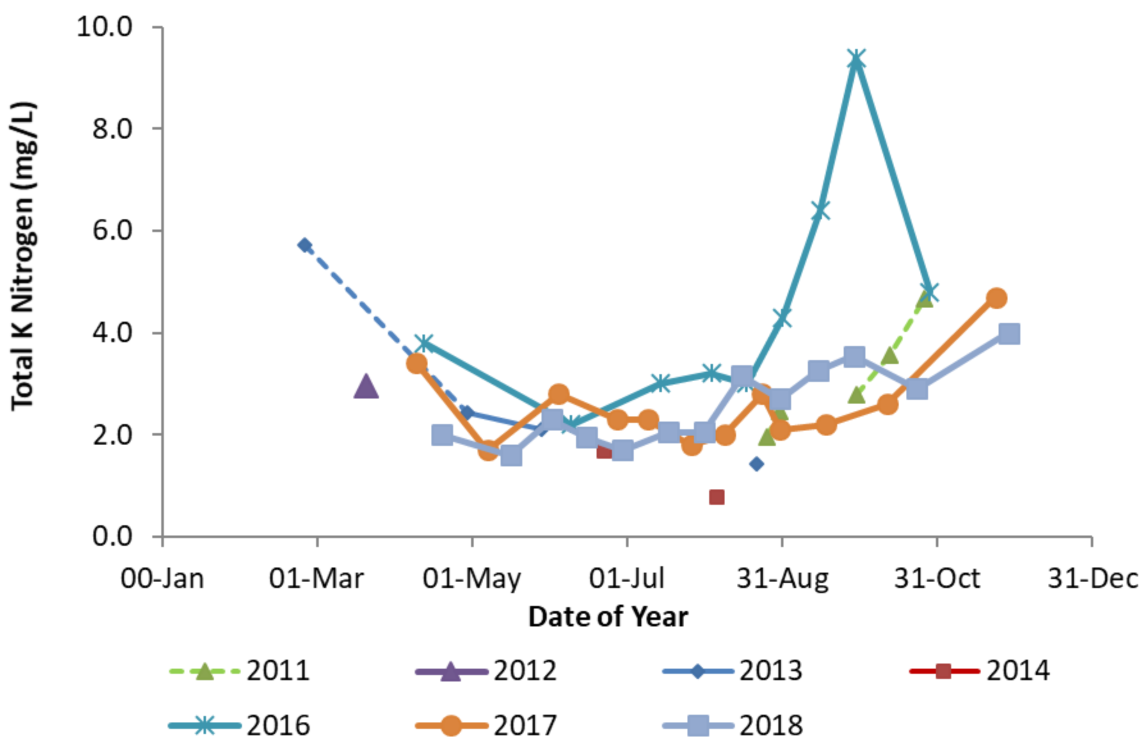
2.2.3.2 Nitrogen

Total nitrogen (TN) consists of total Kjeldahl-N (TKN), which includes ammonia (NH_3) and organic N compounds, and of Nitrite & Nitrate (NO_2&NO_3). Of those compounds, ammonia and Nitrite & Nitrate are the most directly bioavailable forms, where ammonia is the most usable form for cyanobacteria and algae.

TN growing period mixed layer average is another indicator of trophic state. This measure was always larger than 1.2 mg/L that is representative of hyper-eutrophy, except in post -treatment year 2014 (Table 2). Inorganic N-compounds (NO_2&NO_3 and NH_3) were often below detection limits of 0.05, 0.10 or 0.25 mg/L, meaning relatively low available N concentrations. Occasional N-limitation in the summer can occur because of the rather high P concentrations. (But is not clear whether these values indicate that N was limiting algal growth because the detection limits are relatively high.)

Because most of the NO_2&NO_3 measurements yielded values below detection, TKN is close to TN in Swan Lake. Therefore, TKN was examined (instead of TN) to increase the number of available data. TKN concentration decreases after spring, remains relatively constant throughout the summer, and increases in the fall (Figure 6).

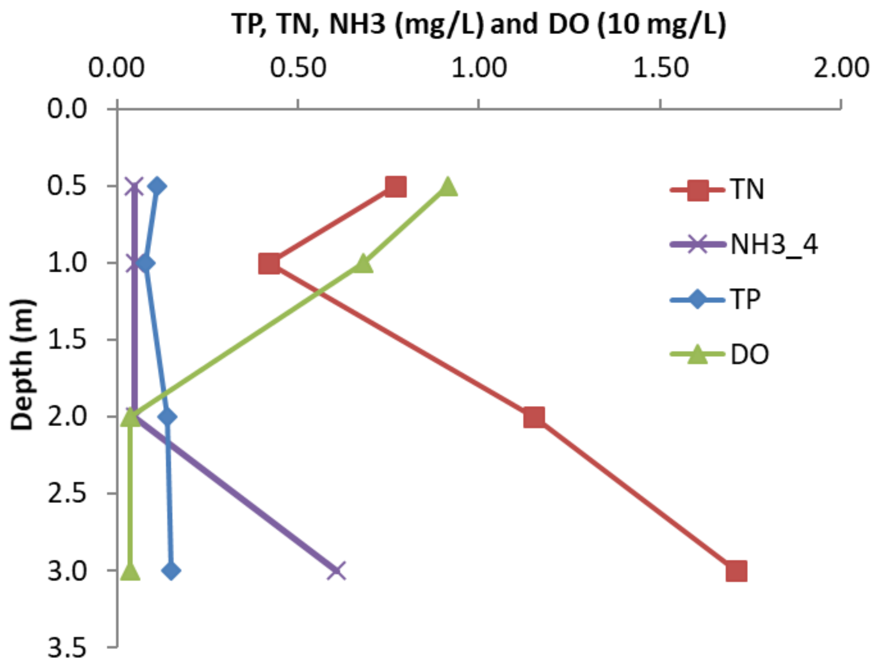
Figure 6. Comparison of total kjeldahl nitrogen in the surface layer (within 0-2 m) for 2011-2018 at deep Site 3 (2011-2017) and Dock Site 1 (2017 and 2018). The Phoslock treatment occurred 29 April – 1 May 2013. (Pre-treatment data are indicated as broken lines.)



Ammonia, TKN and TN concentration were typically elevated closer to the lake bed even after the Phoslock treatment (e.g., 5 Aug 2014, Figure 7). Ammonia increases because under hypoxic conditions nitrates and other N-compounds become converted to the reduced ammonia gas. TKN is increased probably because of the accumulation of organic matter. As N is not released from anoxic sediments like P, these fall increases are most likely caused by the arrival of migratory geese, perhaps shoreline contributions during wet conditions, and vegetative senescence in the fall.

Persistent N increases with depth even after a Phoslock application is expected because Phoslock does not treat nitrogen or anoxia directly. The lower TN growing period average in 2014 can be explained by a generally lower productivity due to decreased P and evident by phytoplankton decrease (Table 2 and Section 2.2.4).

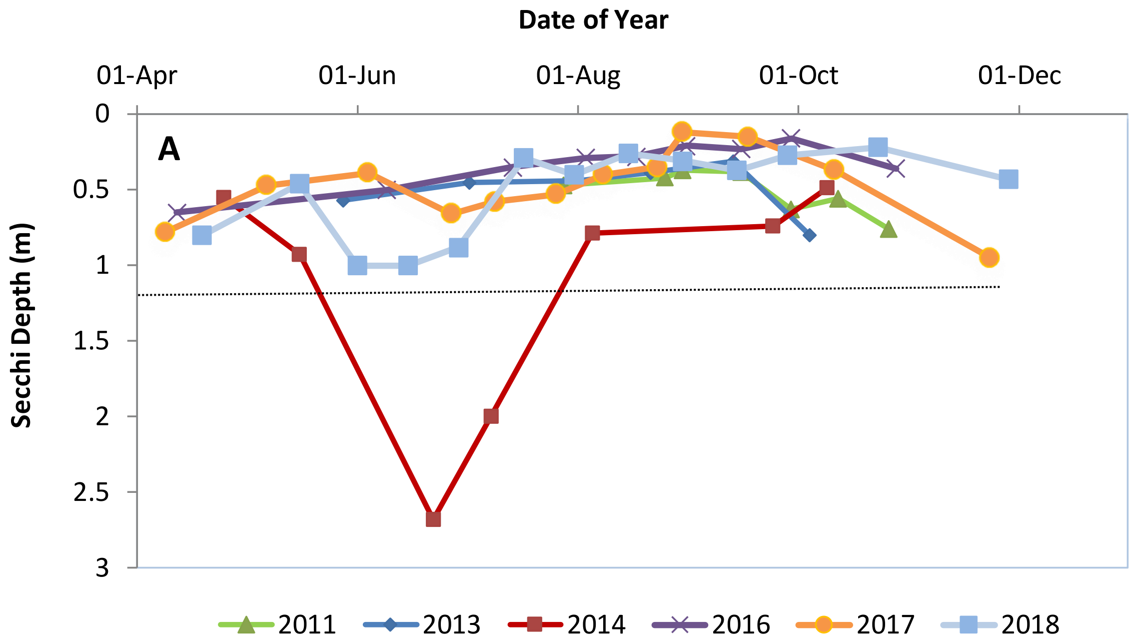
Figure 7. Nutrient and oxygen profiles at Deep Site 3 (5 Aug 2014). NO_3 was below the detection limit of 0.10 mg/L at all depths and is not presented.

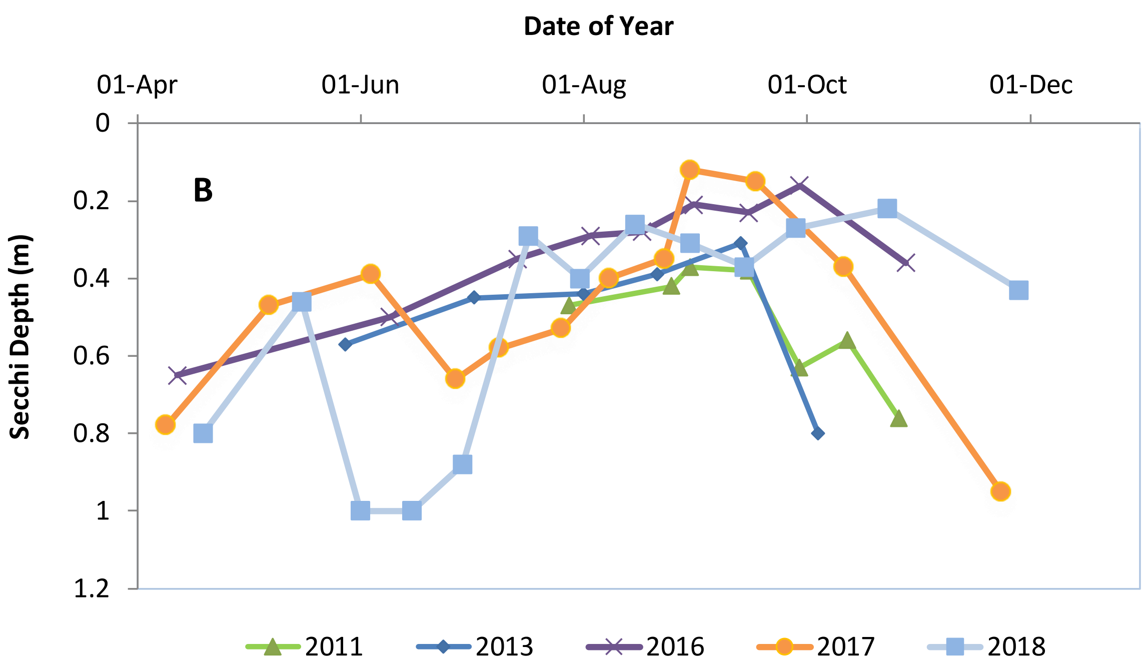


2.2.4 Phytoplankton biomass as chlorophyll concentration and Secchi transparency

Secchi disk transparency is an important water quality indicator and measure of the general appearance of the lake water. Secchi can reflect phytoplankton biomass, colour, and other turbidity in lake water. In Swan Lake, Secchi indicates mainly phytoplankton because its readings are highly and significantly correlated with the algae pigment chlorophyll *a* (see below and Appendix B). Secchi is typically quite low (shallow visibility) throughout the summer period but increases in November, reflecting the end of the growing period for phytoplankton (Figure 8). The Provincial guideline for recreational water of 1.2 m was exceeded (lower transparency than 1.2 m) for all monitored dates except in 2014. The Secchi growing period average indicated hyper-eutrophy in all monitored years except the second post-treatment year, 2014 (Table 2).

Figure 8. Secchi disk transparency readings 2011-2018. The 1.2 m Provincial guideline is indicated by a horizontal broken line (A). Large transparency values presented in the upper graph (A) have been removed to better show the more typical development in the lower graph (B). Treatment occurred 29 April – 1 May 2013.



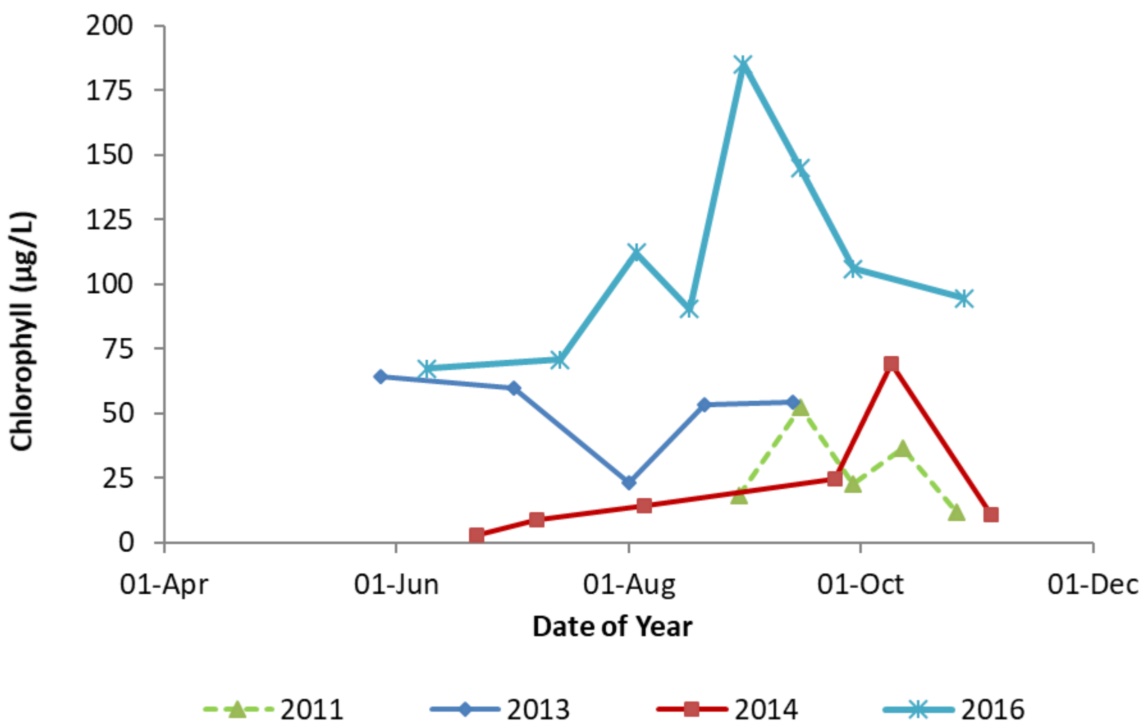


A measure of algae biomass besides the actual identification and counting of cells is the extraction and analysis of chlorophyll *a* (Chl), which is the green algal pigment used in photosynthesis.

Average growing period Chl concentration was above 25 mg/L, the threshold for hyper-eutrophy in all monitored years but the second post-treatment year, 2014 (Table 2, Figure 9). Chl was highest in 2016 at 111 µg/L, which is more than twice that of the next highest measured average in 2013 at 50 µg/L and indicates a severely degraded state.

Chl varied much throughout the growing periods (Figure 9) and this algae indicator is known to be spatially and temporally patchy. Chl is also a relatively unreliable measurement and its analysis is error-prone and expensive, therefore it was discontinued in 2017. Instead, Secchi disk transparency can be used as an indicator of phytoplankton biomass in Swan Lake, because a comparison between Secchi and Chl values revealed a useful significant relationship (Appendix B). Accordingly, in years with missing Chl data this relationship was used to predict Chl from Secchi average values for the trophic state evaluation (2017 and 2018, Table 2).

Figure 9. Chlorophyll a concentration in the surface layer (within 0-1 m) for 2011-2016 at deep Site 3. The Phoslock treatment occurred 29 April – 1 May 2013. (Pre-treatment data are indicated as broken lines.)



2.2.5 Cyanobacteria and cyanotoxins

Swan Lake has a history of blooms of cyanobacteria ("blue green algae" or "bluegreens") (Nürnberg, 2018). Such blooms are not only unsightly when they form scums covering whole sections of lake surfaces, but they can be toxic to humans, pets, and wild animals upon contact or ingestion. There are several different types of cyanotoxins (including microcystin, cylindrospermopsin, nodularin, anatoxin and saxitoxin), but the most easily analyzed is the hepatoxin (affecting the liver) microcystin-LR, for which guidelines are available.

Health Canada (HC) recreational guidelines and provincial drinking water standards exist for cyanotoxins. The maximum concentration for microcystin-LR under the Ontario Drinking Water Quality Standard is 1.5 µg/L (O. Reg. 169/03, Schedule 2) and the HC guideline for recreational activities is 20 µg/L (Health Canada, 2012).

Ontario Ministry of the Environment, Conservation and Parks (MECP) regards any cyanobacterial bloom as potentially toxic, whether or not toxins are detected in the water upon testing (Winter et al., 2011). This is because toxicity changes with the state of the bloom and is not necessarily correlated to cell number.

The Quebec MDDELCC assembles lists of vulnerable lakes with a concentration of cyanobacteria of 20,000 cells/ml or more. 20,000 cells/mL is also WHO's (World Health Organization) recommended threshold for avoiding irritative effects. 100,000 cells/mL is WHO's threshold for significantly increased risk for human health (Chorus and Bartram, 1999). This is also the recreational water quality guideline by Health Canada (2012).

There are not many studies on the toxicity in the air close to freshwater environments with cyanobacteria (Wiśniewska et al., 2019), and we do not know of any regulatory guidelines on airborne toxins in the vicinity of ponds and lakes.

Microcystin concentrations in Swan Lake were always under the detection limit when measured in the pre-treatment year of 2011 and in the post-treatment years until 2016. But several blooms with potentially toxic cyanobacteria were determined in years before 2011. Two species of (non-toxic) cyanobacteria were found on almost all 2011 sampling occasions, *Anacystis* spec. and *Gomphosphaeria* spec. (Table 7 of Nürnberg and LaZerte 2012). Further, *Dolicospermum* (former *Anabaena*), which can produce the cyanotoxin microcystin, was identified twice and a small amount of potentially toxic *Oscillatoria* (now "*Planktothrix"*) once in 2011.

There were no apparent cyanobacteria proliferation and blooms in the years since Swan Lake's treatment with Phoslock in 2013 until 2016 and the City did not receive any complaints related to the Lake's water quality during that period. In 2016, a bloom was detected at one location, none was observed in 2017, but extended blooms were observed at several sites in 2018.

A sample collected at the eastern shore in Swan Lake on Aug 31, 2016 included a high biovolume of at least 7 genera of potentially toxic cyanobacteria (Nürnberg and LaZerte, 2018). Most of the species contributing to the bloom were *Microcystis* (dispersed cells) at a high density of 3.5 million cells/ml and *Planktothrix* spec. at 17,226 cells/mL but *Limnothrix*, observed earlier, was not listed. The tested microcystin concentration of 73 µg/L was about 3.7 times the provisional federal guideline for recreational activities (20 µg/L).

The 2018 blooms (Table 4, Table 5) included a large amount of cyanobacteria that had a cell density that would have been considered to be listed in Quebec, considered irritative by WHO and were about half WHO's concentration of significant risk, as described above. It is unknown whether these blooms had toxicity associated because no samples were analysed for cyanotoxins.

Table 4. Phytoplankton classes collected at two sites in Swan Lake, Sep 28 2018.

Group	Density (cells/mL)		Biomass (mg/mL)	
	Dock	Bay	Dock	Bay
Chlorophyceae	20,676	10,976	1,543	751
Chrysophyceae	255	255	529	65
Cryptophyceae	0	1,787	0	423
Cyanobacteria	46,967	52,072	70,533	117,787
Diatoms	3,318	3,063	687	1,010
Dinophyceae	51	0	7,698	0

Table 5. Genera and species of cyanobacteria, collected at two locations (Site 1 and 2) in Swan Lake, Sep 28 2018, and their potential toxins.

Name	Toxins	Density (cells/mL)		Biomass (mg/mL)	
		Dock (1)	Bridge (2)	Dock (1)	Bridge (2)
Dolicospermum planctonicum (Brunnthaler) Wacklin et al. 2009 (former *Anabaena)*	m	255	1,021	144	982
Aphanocapsa delicatissima West & West	m	1,787	1,276	898	770
Aphanizomenon flos-aquae (Linne) Ralfs	m	766	255	6,837	2,947
Cuspidothrix issatschenkoi (Usacev) Rajaniemi et al.	a	8,168	5,871	6,239	4,825
Limnothrix sp	new	26,546	30,631	8,236	11,644
Merismopedia glauca (Ehrenberg) Naegeli	?	1,021	255	2,406	120
Merismopedia tenuissima Lemmermann	?	1,532	766	308	103
Microcystis wesenbergii (Komarek) Komarek	m	255	0	1,337	0
Planktothrix agardhii Gomont	m	5,360	11,742	44,105	96,389
Synechocystis sp (bi-cell)-spherical	?	0	255	0	9
Synechococcus sp (unicell)-rod-shaped	?	1,276	0	24	0

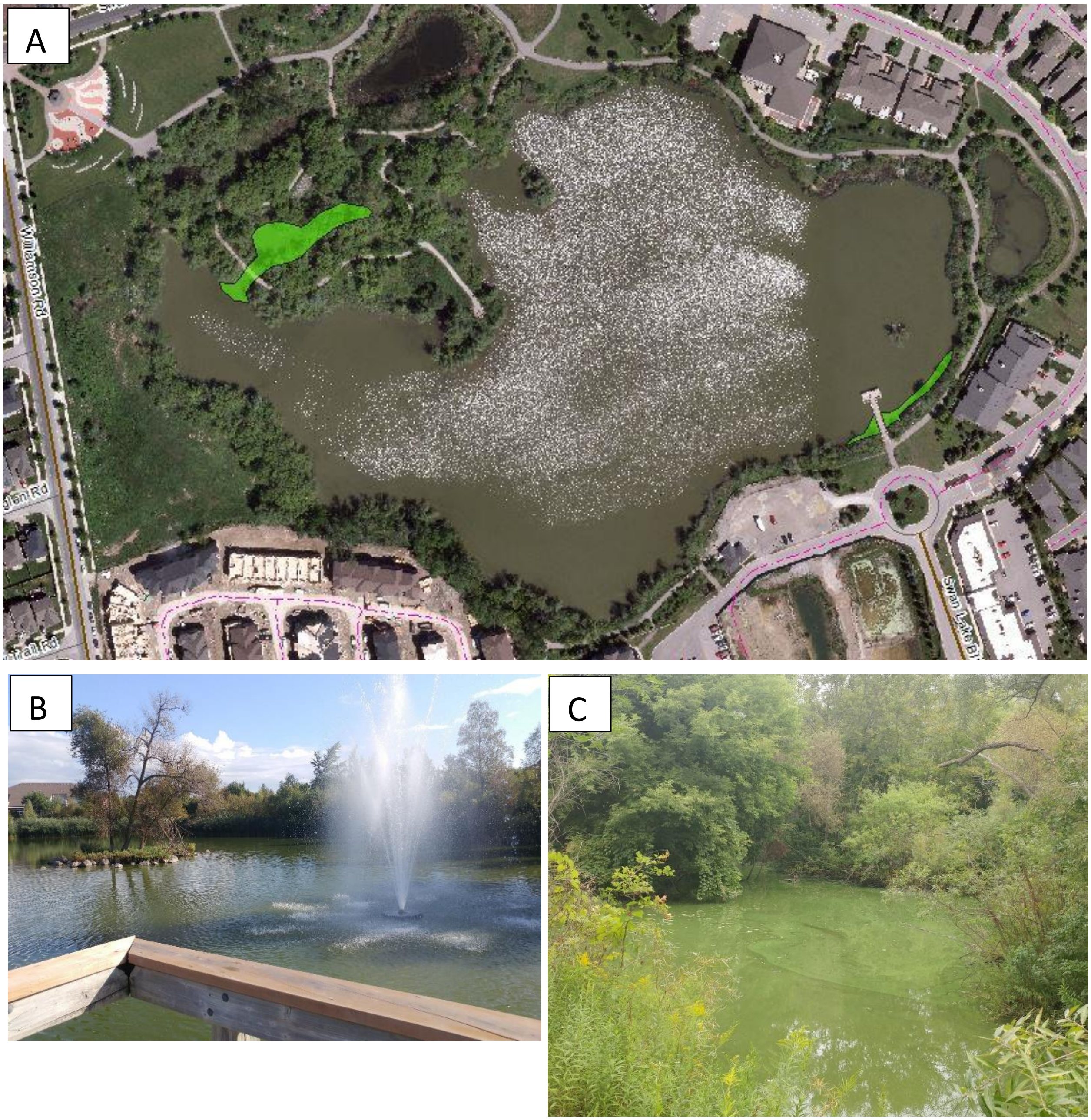
Toxins:

Cyanobacteria and known toxin production:
m, microcystin
a, anatoxin
new, toxins were recently described in tropical lakes.
?, most of these cyanobacteria are found together with toxin producing types. It is not always clear whether they are toxin producers themselves.

Surface scums were at high density along the shoreline around the southern dock (Site 1) and covered the whole north-western bay at the bridge (Site 2) in 2018 (Figure 10 A). Staff of the City of Markham observed deteriorating water clarity starting at Bridge Site 2 in late June with surface scums from end of July to early October 2018. At the same time, “green” water indicating a high density of phytoplankton was obvious at the dock with scum conditions on Sep 28, 2018, the date of sample collection. It is possible that dense algal mats and their collection on the surface as scums was delayed because of water movement by a fountain in the vicinity of the dock (Figure 10 B).

While there was no cyanotoxicity monitored in 2018, Abraxis test strips (analyzes the cyanotoxin microcystin in the water) for two surface samples from the bridge (Site 2) and three from the dock (Site 1) on July 31, August 15 and August 30, 2019 (Figure 10 B, C) revealed microcystin concentration between 2.5 and 5 ppb, clearly below the recreational limit of 20 µg/L.

Figure 10. Maximum extent of surface scums in 2018 (A, bright-green markings) and photos of Dock, Site 1 on Sep 14, 2018 (B) and Bridge, Site 2 on August 30, 2018 (C) (provided by City of Markham)**.**



2.2.6 Organic content and colour

Dissolved organic carbon (DOC) and colour indicate the organic content of lake water. DOC and colour tended to have lower values after the treatment (Table 6), but there is no consistent trend. It is unclear what caused the fluctuation in colour and DOC over the years.

Table 6. Colour, DOC, and chloride annual averages since 2011, average of all sampling dates, number in parenthesis. 2017 and 2018 data are from Dock Site 1 all others from Deep Site 3.

Year	Colour (true colour units)	DOC (mg/L)	Chloride (mg/L)
2011	36 (3)	10 (6)	n.a.
2013	23 (2)	18 (2)	399 (1)
2014	17 (1)	8.9 (1)	440 (1)
2016	18 (6)	2.8 (7)	n.a.
2017	24 (12)	8.6 (12)	484 (12)
2018	15 (13)	3.6 (13)	496 (12)*

n.a., Not available
*Outlier for Cl, 1160 mg/L of Aug 15, 2018 removed

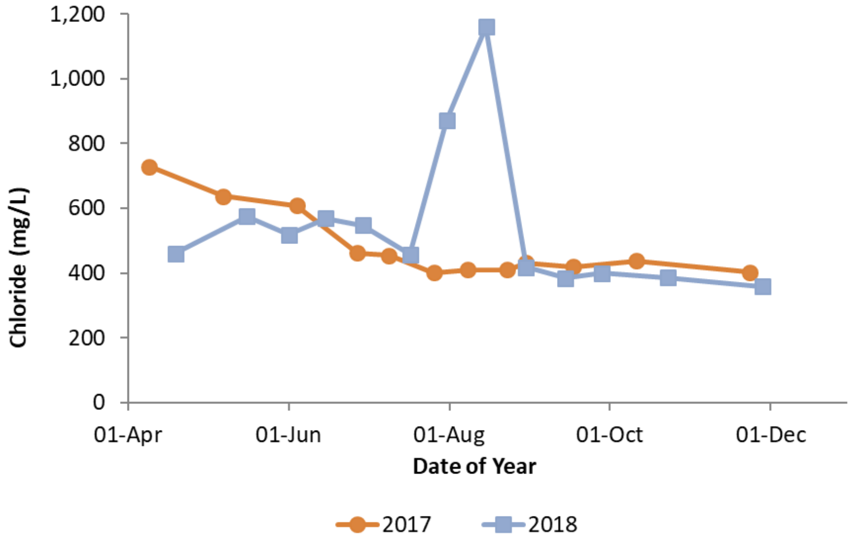
2.2.7 Chloride concentration

Chloride (Cl) concentration was investigated in more detail in Swan Lake starting 2017 (Table 6, Figure 11), because it has been increasing in neighboring lakes and is a tracer of anthropogenic development due to its prevalence in road de-icers. Incidences of elevated Cl concentration occur frequently in urban watersheds and can readily be seen in data from the Toronto and Region Conservation Authority (TRCA) report on Toronto area rivers (TRCA 2018), where 2017 median concentrations reached up to 700 mg/L Cl. 2017 median Cl concentration ranged from 80 to 280 mg/L in 2017 at various locations in the Rouge River (TRCA 2018), the watershed that includes Swan Lake.

While Swan Lake is not a natural lake, it is interesting to note the guidelines available for chloride to prevent detrimental conditions to biota. The Canadian Water Quality Guidelines for the Protection of Aquatic Life – Chloride (http://ceqg-rcqe.ccme.ca/download/en/337/) (Canadian Council of Ministers of the Environment, 2011) are 120 mg/L and 640 mg/L for long-term and short-term exposure respectively. There are no provincial guidelines for chloride set at time of writing.

In Swan Lake both thresholds were exceeded in the years with monitoring data, 2017 and 2018. Chloride concentration annual averages were almost 500 mg/L in 2017 and 2018 (Table 6, Figure 11), with spikes above 1000 mg/L. The general pattern of decreasing concentration from spring to fall can be explained by the winter de-icing operations.

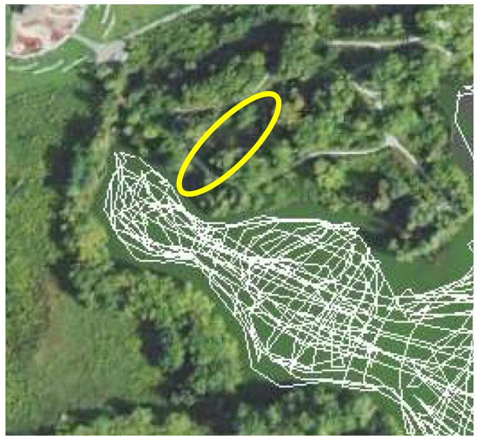
Figure 11. Chloride concentration at Dock Site 1 in 2017 and 2018 (0.5 m).



2.2.8 Water quality at Bridge Site 2 (Bay at north-western corner).

Bridge Site 2 (Figure 1) was chosen in 2017 as an additional monitoring site, because it may more closely present water quality in more confined areas of Swan Lake. While Site 2 revealed similar surface water quality as Site 1 in 2017 (Nürnberg and LaZerte, 2018), there was a conspicuous phytoplankton accumulation in 2018 (Figure 10, A). Several reasons can explain the increased productivity at Bridge Site 2 and in the bay. The confined and well protected bay of Site 2 encourages habitation by wildlife and water fowl while the water exchange with the open water is minimized. Further, the Phoslock application did not reach into the corners of the bay (Figure 12), so that any P release from the sediments has not been inhibited by the capping of the Phoslock compound. Consequently, it is possible that the undesirable conditions at Site 2 negatively affect the open water of Swan Lake by introducing sediment derived phosphate, organic substances, and cyanobacteria. Accordingly, this area should be considered in any treatment goals and options (Sections 4 and 0).

Figure 12. Phoslock application path in north western Swan Lake. The location of the bridge and bay close to Site 2 is indicated by yellow oval. (Based on a graph provided by Phoslock Water Solutio*ns*, *Australia*, *May 2013*.)



2.3 Sediment quality

The general characteristics of Swan Lake sediments are consistent with a hyper-eutrophic, hard water lake, as evident from sediment samples collected in 2013 and 2014. Calcium and indicators of organic content (ash weight, total organic carbon, and loss on ignition) were relatively high (Table 7). Lanthanum, the treatment component of Phoslock, was far higher than in 2011 sediment before treatment as expected.

Table 7. General sediment characteristics of a 10 cm surficial layer sample (August 2014). *(From Table 8 of Nürnberg and LaZerte 2015)*

Characteristics	Units	Results
Lanthanum (Total)	mg/g	0.30
TKN	mg/g	26.8
Calcium Carbonate equivalence as CaCO3)	%	21.6
Total organic C (TOC)	%	5.53
Moisture	%	77
Total solids	%	23
Ash	%	88.7
LOI	%	11.3
Crude bulk density	kg/m^3	260
Sediment grain size		fine

Since it is apparent that sediment P release is part of Swan Lake's water quality deterioration, sediment chemistry can be an important indicator of this process. Sediment total P can serve as an overall but crude indicator of potential fertilization of the overlaying lake water by bottom sediments, and its concentration of 1.1-2.1 mg/dry-weight (various analyses reported in the previous studies) indicates eutrophic to hyper-eutrophic conditions. However, only specific sediment fractionation techniques determine the proportion of TP that is actually involved in anoxic sediment P release.

The analysis of such P fractions conducted in 2013 and 2014 (post-treatment), indicated that the releasable P fraction is comparatively small. The mobile fraction, which is involved in P release, was 22% in May 2013, 24% in Aug 2014 but only 11% in Jan 2014. The P fractions considered to be “refractory” and not involved in P release (residual P, apatite P, Ca-P and the newly formed La-P due to Phoslock treatment) were high at 65% in May 2013 and 64 -72% in 2014. As published research after Phoslock applications showed (Meis et al., 2013), much of the sediment P that used to be available for release is bound by lanthanum and contributes to the residual P fraction that is inert. In 2013 and 2014, any phosphate dissolved within Swan Lake sediments would likely have been adsorbed by lanthanum.

Lanthanum in post application (Aug 17, 2016) sediment was 0.71 – 1.19 mg/g dry weight, compared to a maximum of 0.00691 mg/g before treatment (May 28, 2013). Despite this rather large amount, an analysis of the 2016 sediment also revealed that the lanthanum of the 2013 Phoslock application could only adsorb half of the releasable P in 2016 (Table 8. This calculation is based on a releasable P proportion of 23.1% according to P fractionation results of summer 2013 and 2014.)

Table 8. Sediment concentration of phosphorus compared to the P adsorbent, lanthanum, in sediment collected in 2016 at 3 sites throughout Swan Lake (Figure 13).

Location	Phosphorus			Lanthanum		**Ratio La:P***
	Total (mg/g dr wt)	Releasable (mg/g dr wt)	(mmol/kg dr wt)	mg/g dr wt	mmol/kg dr wt	
1	2.09	0.48	15.62	1.15	8.28	**0.53**
2	1.30	0.30	9.72	0.71	5.10	**0.53**
3	1.45	0.34	10.84	1.19	8.57	**0.79**

*Lanthanum (La) binds phosphate in a ratio of 1:1. When the ratio is below 1 insufficient La is available to adsorb all releasable P (Yasseri and Epe, 2016).

Figure 13. Sediment sampling sites in 2016

Considering the available information, it appears that not enough lanthanum is presently available to bind all the releasable phosphorus. It is likely that newly settled material, especially organic compounds from the plankton and feces from geese, have overwhelmed the added lanthanum concentration so that it is not preventing P release anymore. It has also been observed that bioturbation by bottom dwelling fish can severely compromise the adsorbing clay layer.

2.4 Vertebrates including fish and turtles

Fish species and abundance are useful water quality indicators and accordingly any information on fish and fish kills is assembled in Table 9. Based on a historic overview in MNA (2008) and a trap analysis by Gartner Lee (2006, on 2 Dec 2005) as summarized in Nürnberg and LaZerte (2012), no largemouth bass is left over from their stocking in 1992, and even pumpkinseed were rare. Instead, the community mainly consists of one species only, the fathead minnow (*Pimephales promelas*). This species endures low water quality with respect to low oxygen, low water clarity and muddy conditions, possibly because it is a surface feeder. The apparent decline of pumpkinseed indicates deteriorating water quality, because they are susceptible to silt and pollution.

There were very few fish kill events in the past. Two were observed in the past 14 years by a resident and a photo of a kill of small fish in the early spring of 2014 or 2015 is on file. The lack of any apparent fish kill in Sep-Oct 2011, when the whole water column was anoxic, also shows that there may be no fish present besides the resilient fathead minnow. However, local knowledge indicates that there are carp and possibly pike in the Swan Lake. A future fish survey is needed to determine the present fish population of Swan Lake. In particular, it is important to know the quantity and spread of bottom dwelling species, such as carp, if a treatment that involves the capping of sediments is to be applied (Section 6).

Table 9. Fish species in Swan Lake

Year	Fish species identified
Stocked before 1992	Largemouth bass
1996	Pumpkinseed Black crappie Largemouth bass
2005	Fathead minnow Pumpkinseed (few)
2012 and after	Carp, goldfish and "minnows" (small fish)

Non-native turtles have been sighted that are probably escaped from terrariums or introduced into the wetlands around Swan Lake (Staff of City of Markham, pers comm.). Should lake water quality improve, the stocking of native species can be envisioned.

2.5 Seasonal climatic conditions and their influences on water quality

To consider the influence of seasonal and long-term climatic conditions on the water quality of Swan Lake, precipitation patterns were examined in the previous studies. Rain records reveal a large variation between months and years (Figure 14).

Most striking was the rather dry year in 2016 with low rain in the spring (Figure 15 and Figure 16), followed by exceptionally low water quality (Table 2). In comparison, 2017 was wet until August while 2018 was dry until June with various signs of water quality deterioration.

As shoreline runoff is highly correlated with precipitation (based on the hydrological model, Appendix C), weather-related external inputs from shoreline runoff are high in wet years. It is counterintuitive that high external P input, would have a positive effect. This pattern could in part be explained by cool inflow water and relatively high water levels creating larger lake volumes. Markham Staff indicated that the wetland around Bridge Site 2 had visibly higher water and less algal mats in 2017 than in previous years.

In addition, internal P loading will be enhanced when the water becomes warm and stagnant (solidly stratified), during dry spells with low water levels in the summer (Section 2.5.1). Increased anoxia and sediment P release with warmer water temperature and smaller water volumes will increase phytoplankton biomass and favour cyanobacteria (Section 2.5.2).

Figure 14. Monthly precipitation volume onto Swan Lake for 2009 and 2018 (data based on Appendix C).

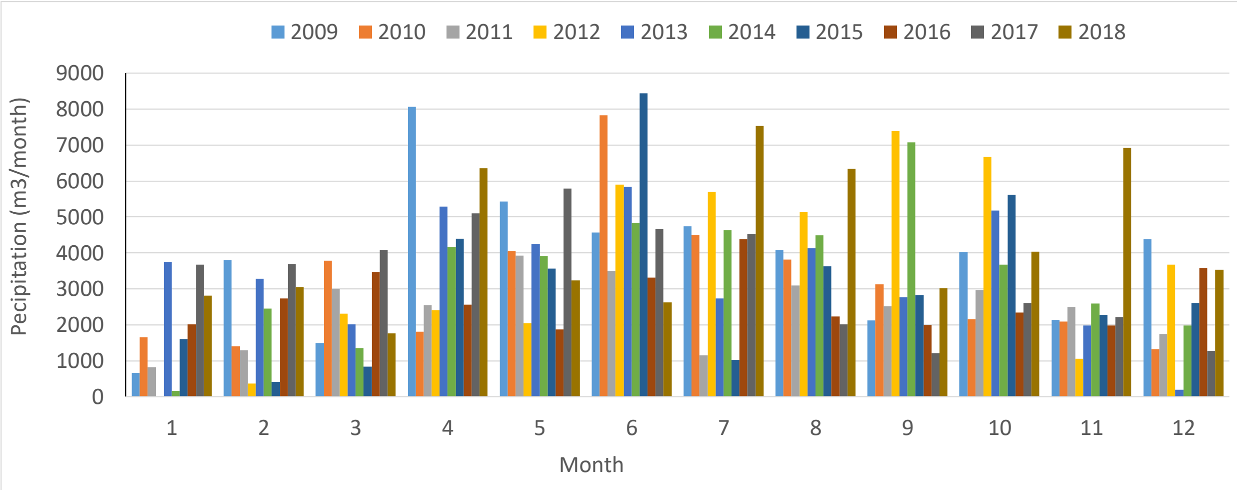
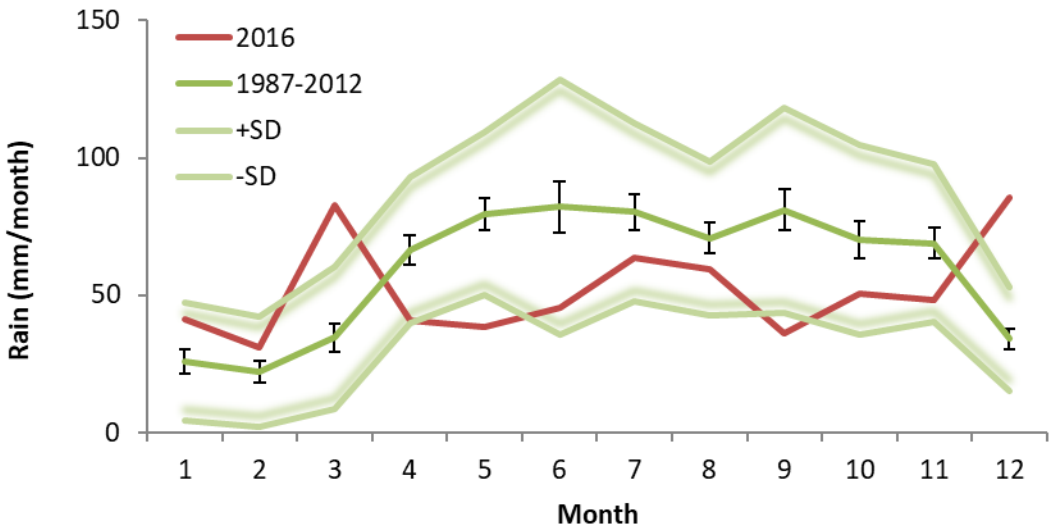
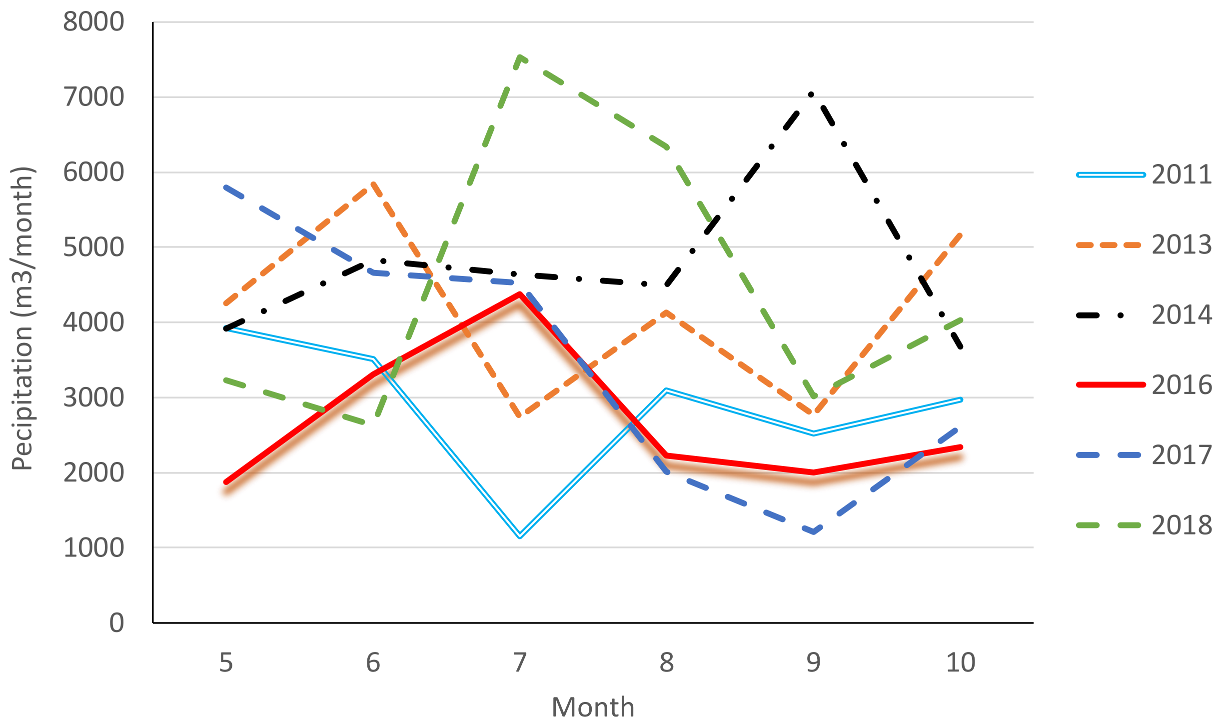


Figure 15. 2016 monthly rain height in comparison to long-term (1987-2012) monthly rain.



Averages, standard deviations (+SD, -SD) and standard errors (bars) from the closest federal rain gauge (Toronto Buttonville Airport, Latitude 43.86, Longitude -79.37, Elevation 198.1 m, http://climate.weather.gc.ca).

Figure 16. May to October precipitation volume for years with known water quality (data based on Appendix C).



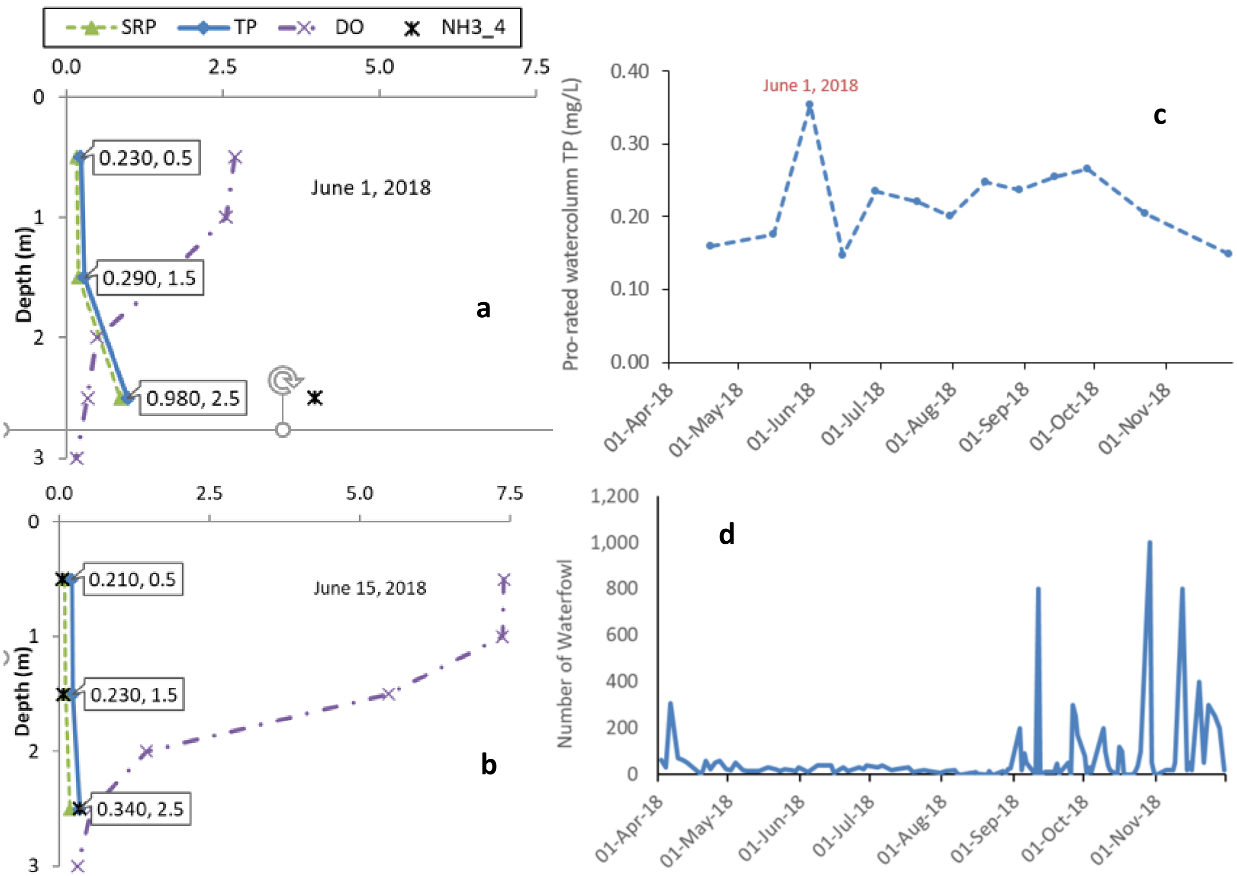
2.5.1 Climate conditions supporting sediment P release

Many observations throughout the monitoring effort show increased TP concentration during periods of thermal stratification and anoxia in Swan Lake, often related to dry spells. (Especially for 2011 and 2016, see previous reports.) In comparison, there were no such events in the immediate post-treatment years (2013 and 2014, Nürnberg and LaZerte 2016). We here discuss some examples that indicate the resurgence of sediment P release from legacy sediments and possibly from more recently settled nutrients of external input and water fowl.

During the dry season starting in August 2017 (Figure 16), profiles at Deep Site 3 indicated a water column average of 283 µg/L TP on Aug 31, 2017, strong stratification, and anoxia below 2.5 m. 21% of the TP mass was in the layer below 3 m, which represents only 2% of the total volume (Figure 5).

Stagnant conditions occurred earlier in 2018 during a dry spell (Figure 16), when severe anoxia throughout the water column on June 1, 2018 coincided with elevated bottom water TP and SRP concentration at Dock Site 1 (Figure 17 a). It is rare to actually be able to catch hypoxic conditions throughout the water column including the surface water, because shallow Swan Lake is easily mixed and aerated by wind and wave action. Stagnation had passed on June 15, 2018 (Figure 17 b) after increased precipitation and storm events (Figure 16) that also decreased the temperature from 25.5° C to 21.2° C in the top 2 m surface layer. The increased volumetrically averaged water column TP concentration (Figure 17 c) could not be explained by waterfowl abundance, which is low in the summer since the goose population was managed (Figure 17 d). The effect on phytoplankton was not immediately obvious and Secchi only started to decrease from 1 m on June 1 and 15 to 0.9 m on Jun 29 and 0.3 m on July 17, 2018.

Figure 17. 2018 Dock Site 1 indication of sediment P release. Profiles in the left panel (a, b) present nutrient and oxygen (DO) concentration (boxed numbers indicate TP and depth); development from April to Nov of water column average (c) and number of counted geese (d) are presented on the right.



2.5.2 Climate conditions supporting cyanobacteria

Water temperature supports phytoplankton productivity and can trigger cyanobacteria proliferation because their optimum temperature is higher than that of more benign phytoplankton. For example, monitoring profiles suggest that 2017 was cooler than all previously monitored years except for the pre-treatment year 2011 (Table 10). (As Dock Site 1 was slightly warmer than the open water, this occurrence was probably even more severe.)

Table 10. Comparison of 2017 summer water temperature (°C) within the mixed layer with previous years (usually 0.5-2.0 m depth).

Year	Temperature C	Comment
2017	**21.9**	**Site 1, Jul-Aug**
2011	21.7	Site 3, August only
2013	22.4	Site 3, Jul-Aug
2014	22.8	Site 3, Mid-Jun to Aug
2016	24.1	Site 3, Jul-Aug

Further indications that 2017 was much cooler than for example, 2016, a year with large phytoplankton biomass indicators, are July and August mean air temperature, and degree days below 18 °C or above 18 °C (Table 11, Environment and Climate Change Canada data from Buttonville Airport, http://climate.weather.gc.ca/prods_servs/cdn_climate_summary_e.html).

Table 11. 2016 and 2017 mean air temperature (°C) and degree days below and above 18 C at Buttonville Airport.

Year	Month	Mean Temperature (°C)	Degree Days below 18 °C	Degree Days above 18 °C
2016	Jul	22.8	1.5	150.9
2016	Aug	23.3	0.3	159.1
2017	Jul	21.2	1.8	99.7
2017	Aug	19.3	18.1	57.8

The relatively low water temperature in the summer of 2017 would also have curtailed internal P load, because release rates and sediment oxygen demand, which enhances release, are strongly related to temperature. Conversely, warmer temperatures increase sediment P release.

Climate change predictions for the City of Markham region include increased summer air temperature and potentially more extreme storm events. While historical trends in storm intensities have decreased in Markham and in Southern Ontario (Muir, 2018), and bias-corrected regional climate models for the region do not consistently predict increases (Ganguli and Coulibaly, 2019), potentially larger storms would increase the likelihood of external P loading and, if not managed, phytoplankton productivity including cyanobacteria blooms.

3 Task 2- Update of Water Balance and Load Estimates - Phosphorus sources and their contribution to lake TP concentration

To determine a chain of action when water quality is severely deteriorated in a lake, a study is needed that determines the origins of the most important agent, which is the nutrient phosphorus. The following main P sources were investigated:

1. Stormwater ponds
2. Immediate runoff from the shoreline and other 'untreated' runoff
3. Atmospheric deposition (rain, snow and associated dry fall out like dust)
4. Waterfowl defecation
5. Internal P loading

The P inputs from various external sources were estimated on an annual basis, using monitored data for phosphorus and water volumes provided by City of Markham staff (Appendix C). We estimated P from internal sources as explained in Section 3.4.

Based on a monthly water balance for 10 years (2009-2018) (Appendix C) we established a P mass balance for all years with available P data (2011, 2013, 2014, 2016, 2017, 2018). In addition, the 2011 P concentration average was extrapolated to three unmonitored pre-treatment years (2009, 2010, 2012) considered having similar water quality characteristics. Because 2015 TP concentration was not monitored and could not be interpolated, the model was not applied to the post-treatment year of 2015, positioned before the resurging eutrophication in 2016.

We estimated lake TP concentration averages of the growing period from annual P loads of the specified P sources and waterflows using the steady-state mass balance model.

3.1 Precipitation, stormwater treatment ponds, immediate runoff

TP loading from **atmospheric deposition** is usually measured on an areal basis (mg/m^2/yr) and includes precipitation and dry fallout directly to the lake surface. Areal deposition is quite variable and difficult to measure because of contamination issues. Based on intensive monitoring the ON-MECP proposed an annual deposition rate of 16.7 mg P per lake surface area of m^2 in their lake shore capacity model (Ministry of Environment, 2010). Accordingly, we used a constant TP input from precipitation of 0.92 kg for the Swan Lake area (5.5 ha).

Annual loads from the **stormwater treatment ponds** 104 and 105 were computed as the product of annual water volumes and P concentration average (0.116 mg/L) of 8 monitoring data for 2011 and 2012.

Annual loads from **shoreline runoff**, as well as runoff from the most recent phase of development (properties on Oasis Way), which can be considered 'untreated', were computed as the product of annual water volumes and an average P concentration (0.211 mg/L) prorated from perviousness and landuse of the area runoff (Table 12).

Table 12. Calculation of the pro-rated average TP concentration in the immediate shoreline runoff and other untreated runoff

Description		Area (ha)	% Area	TP (mg/L)*
Pervious		4.77	50%	0.121
Impervious:				
	Rooftop	2.01	21%	0.192
	Other impervious**	2.73	29%	0.383
Total		9.51	100%	
Pro-rated average concentration				0.211

*TP concentration based on the US Nationwide Urban Runoff Program (EPA, 1983)
**Includes all impervious landuse besides rooftop and lake: driveways, parking lots, roads, sidewalks pools, amenities (see Appendix C for assumptions).

The previously discussed potential nutrient contribution by groundwater and **historic contamination of the surrounding areas** (Nürnberg and LaZerte 2017) aught to be investigated (Figure 18). Especially the southern dump site may be influential, because the general groundwater flow is east to west (assessment by Staff of City of Markham, Sep 2019).

It is not clear whether these former dumpsites presently contribute to Swan Lake load, but as they are classified as "household dump", leaching of nutrients is likely. Therefore, it is possible that some external loading is still occurring and may have contributed to the higher TP concentration, turbidity, and phytoplankton growth. The potential of contributions from such dumpsites should be considered and investigated further. If the site was operational at least from 1983 until 1999 as the photos indicate, the nutrient load to the dump would have been large and residual leaking might still occasionally occur, perhaps during storm events and at certain conditions, such as frozen ground.

Figure 18. 1983 (left) and 1999 (right) photos indicating former dumpsites (light coloured areas in red circles) before development *(City of Markham)*.

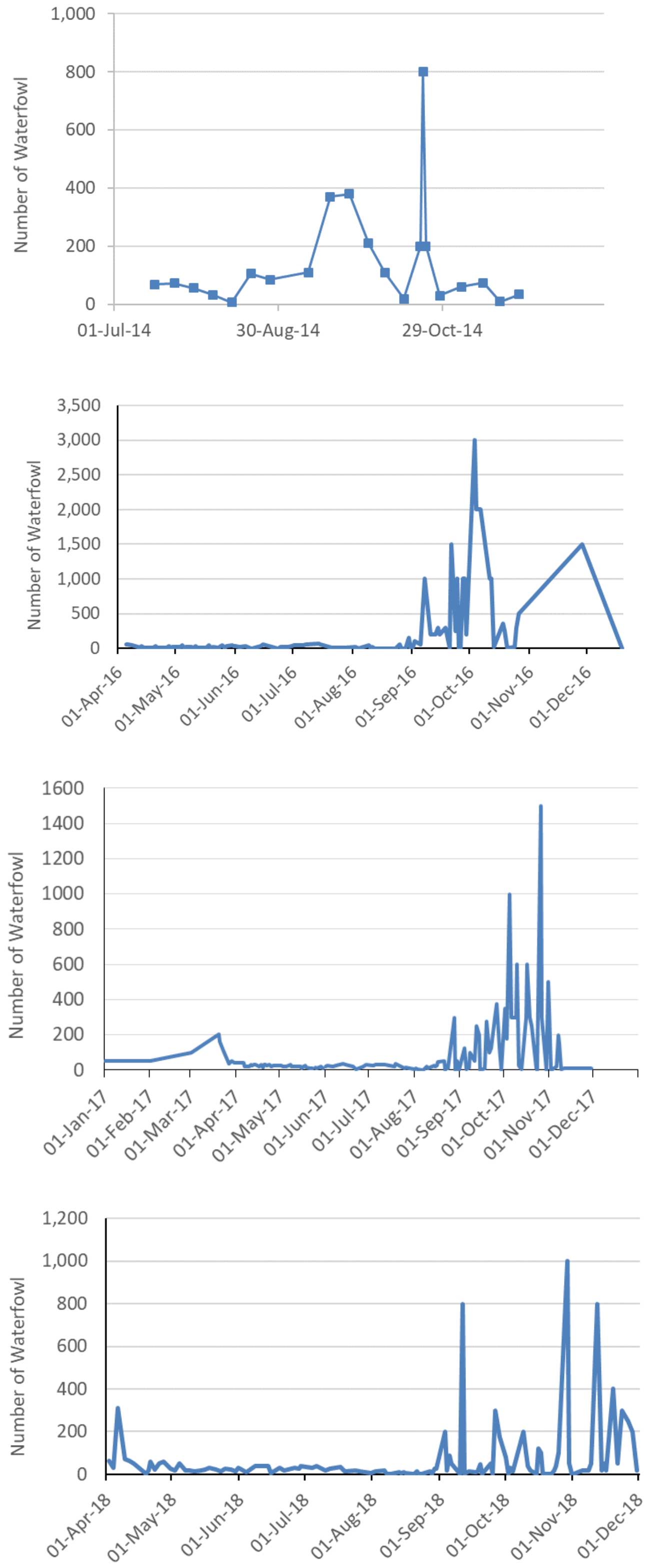
Potential variation of contaminated P input from groundwater between years has not been considered but should be small. (The long-term average inflow rate is negative, meaning more is leaving via groundwater than entering according to the water budget, Appendix C).

3.2 Canada geese and other waterfowl

Previous analysis determined that water fowl, especially the Canada goose (*Branta canadensis*) contribute to the nutrient loading in Swan Lake. Because of the realization that geese abundance is so important, the City of Markham started a goose counting program in 2014, when City of Markham staff obtained goose numbers for 21 days (Figure 19). Since 2016, the City hired a waterfowl deterrent expert (Border Control Bird Dogs) to hinder geese from thriving on Swan Lake and to count goose number several times per week from March to November each year. Supplementary counts were also done by City of Markham staff (Figure 19).

The hired goose control that included chasing of terrestrial geese by border collies and oiling of eggs proved successful throughout the summer and limited resident goose populations to below 25 per day on average during Apr-Aug 2017 and 2018 (Figure 19). At the end of August with the beginning of the migration period, goose number drastically increased but remained below 1,500 and 1,000 per visit in 2017 and 2018, as compared to 3,000 per visit in 2016. Therefore, the goose management program is working and should be continued.

Figure 19. Number of counted waterfowl in 2014, 2016, 2017, and 2018. Geese were not counted in 2015. Note the different scales.



Much of the geese feces settles out as determined by a mesocosm study (Unckless and Makarewicz, 2007). This means the feces would fertilize the water and enrich the sediments of Swan Lake. Liberated organic matter and nutrients would enter the water and lead to sediment anoxia and P release as internal load.

We quantified TP contribution from geese into Swan Lake using published values of TP contribution from geese and other waterfowl (when counted) according to Table 13 (Moore et al., 1998). For years with monitored water fowl number, annual estimates were arrived at by pro-rating water fowl number according to past reports (Nürnberg and LaZerte, 2018, 2016, 2015). For the earlier years without observations, half of the most recent counts (2018) were applied to 2009 and then an increment of the difference between 2009 and the first observed counts of 2014 added to each subsequent year until 2013. This estimate is based on news articles stating that waterfowl doubled within the last 10 years in Ontario and that observations increased during the fall hunting period in southern Ontario urban areas (<https://www.simcoe.com/opinion-story/8074290-goose-population-on-the-rise/). In this way, the TP contribution from water fowl was determined as input into the P mass balance model (Table 14).

Table 13. TP contribution from waterfowl from other studies

Species	TP g/day	Reference
Canada Goose, *Branta Canadensis*, only night	**0.61***	**Moore et al. 1998**
Canada Goose, conservative winter estimate	0.49	Manny et al. 1994
Canada Goose	0.37	Ayers et al. 2010
Mute swan (*Cygnus olor*)	0.57	Hahn et al. 2008
Cormorant (*Phalacrocorax carbo*)	2.6	Hahn et al. 2007
Small Gull (*Larus minutus*)	0.15	Hahn et al. 2007
Nordic geese (*Anser fabalis* and *Anser albifrons*)	0.22	Rutschke & Schiele (1978/1979)
Lesser Snow Geese (*Chen caerulescens caerulescens*) and Ross Goose (*Chen rossii*)	0.45	Post et al. 1998

* Mostly used in the P mass balance

Table 14. Monitored (2014, 2016-2018) and estimated (2009-2013, 2015) fowl number and P contribution.

Date	#/year	kg TP per year
2009*	12,217	7.5
2010*	14,404	8.8
2011*	16,591	10.1
2012*	18,778	11.5
2013*	20,965	12.8
2014	23,152	14.1
2015**	45,155	27.5
2016	67,158	41.0
2017	23,403	14.3
2018	24,433	14.9

*Estimated by assuming half of 2018 geese in 2009 and pro-rating for the subsequent years to reach the first count in 2014.
**Interpolated from previous and subsequent years.

3.3 External load (summary)

The 2009-2018 average annual external load was highest from the shoreline and almost as high from water fowl (Table 15). Flow-related assumptions of the new water budget (Appendix C) differ from those used in the previous P model, where the main water source was direct precipitation on its surface and runoff from the immediate shoreline area. According to the new budget some of the precipitation onto the surrounding watershed area reaches Swan Lake even when storms are below 25 mm. The increased volume and P load from the stormwater treatment ponds and the shoreline result in a smaller relative contribution from geese to the P budget, although absolute values remain the same. This means that water fowl now contribute less to Swan Lake TP concentration in comparison to all other external P sources, as discussed below.

Table 15. Phosphorus contributions from external sources (2009-2018 average)

Annual average TP fluxes	kg/yr	%	Comments
Atmospheric deposition	0.92	3.1%	Long-term average wet and dry fallout for South Central Ontario (Ministry of Environment, 2010)
SWM pond 104 SWM pond 105	0.08 0.88	0.3% 3.0%	Based on the average monitored outflow concentration of 0.116 mg/L TP (2011, 2012, n=8)
Immediate lake shore runoff and other untreated runoff	11.3	38.5%	Based on the pro-rated TP concentration in runoff of 0.211 mg/L TP (LaZerte and Nürnberg, 2000), see Table 12.
Geese	16.2	55.1%	Extrapolated from counts (Section 3.2)
Total external input	**29.5**	**100%**	Sum of above

Of the estimated external fluxes only those of the geese and precipitation do not depend on the flow estimated in the water budget. While atmospheric deposition is assumed constant among years because of difficulties in exact concentration estimates, the number of geese is quite variable and geese derived loads can be larger than the rest of the estimated TP fluxes, e.g., in 2016 (Table 16).

External loads were not specifically addressed by the Phoslock treatment in 2013 and there is no sign of decreased load in 2013 and 2014, the post-treatment years with improved water quality. Groups of years indicate an upward trend from smaller loads before the treatment and larger loads in the recent years (Table 16). This pattern is mainly caused by the increased geese population and some hydrological variability, since the model input related to TP was kept constant throughout the years (except for geese input).

Table 16. Annual calculated TP loads for separate flows of external sources (kg/yr).

Year	SWM pond 104	SWM pond 105	Shore	Atmospheric deposition	Geese	External (total)
2009	0.11	0.92	13.84	0.92	7.45	23.2
2010	0.10	0.72	10.01	0.92	8.79	20.5
2011	0.02	0.74	8.16	0.92	10.12	20.0
2012	0.12	0.93	12.13	0.92	11.45	25.5
2013	0.05	0.95	13.20	0.92	12.79	27.9
2014	0.11	0.90	12.40	0.92	14.12	28.5
2015	0.12	0.75	9.60	0.92	27.54	38.9
2016	0.01	0.79	7.25	0.92	40.97	49.9
2017	0.02	1.00	10.44	0.92	14.28	26.7
2018	0.14	1.12	16.43	0.92	14.90	33.5
Averages:						
2009-18	**0.08**	**0.88**	**11.35**	**0.92**	**16.24**	**29.5**
2009-12	**0.09**	**0.83**	**11.03**	**0.92**	**9.45**	**22.3**
2013-14	**0.08**	**0.93**	**12.80**	**0.92**	**13.46**	**28.2**
2016-18	**0.06**	**0.97**	**11.37**	**0.92**	**23.38**	**36.7**

3.4 Internal load

Internal P load is one of the most challenging P sources to quantify in lake and pond eutrophication and restoration. While inputs from external P sources can be estimated from land use, precipitation, and point sources, internal sources cannot be easily quantified. P that originates within a lake from anoxic bottom sediments, macrophytes, or sediment disturbances immediately mixes with the P already present in lake water so that any fluxes or inputs cannot be directly estimated in shallow lakes. But comparison and quantification of the different P sources is needed to establish successful lake management approaches. When the effects of internal sources on water quality outpace those from external sources or for more immediate remediation, an in-lake treatment is recommended.

Internal P loading usually stems from former external inputs which are stored in bottom sediments, and it may originate in nutrient-rich previous land fill and recent inputs from water fowl feces in Swan Lake. P released from the bottom sediments has a high biological availability, and its release during elevated water temperature in the summer increases its effect on summer-fall lake water quality.

Internal load is typically a result of redox changes at the sediment-water interface. Anoxia leads to the dissolution of iron hydroxides in the sediments and release of adsorbed P (i.e., P attached to the iron surfaces) to adjacent lake water. P may also be released from organic compounds, poly-phosphates, and recently added feces. Whatever the mechanism, P is released as the highly biologically available form of phosphate (approximately measured as SRP) which becomes quickly incorporated into phytoplankton and bacteria.

There are many potential problems associated with separating the contribution of internal from external P sources to a lake (Nürnberg 2009). Accordingly, we usually apply a variety of independent approaches to quantify sediment derived P. In Swan Lake several estimates were considered based on monitoring data obtained in 2011-2018.

1) In situ internal load from summer water-column TP increases (summer-fall)
2) Internal load determined from anoxic factor and release rates (summer-fall)
3) Net and gross internal load from mass balance (all year)

Approach 1 includes a potential settling of the released P and therefore is a partially net estimate (Appendix D). This approach was used in 2011, but since then can be considered compromised in Swan Lake because of the large and variable number of geese that contribute to the water column TP concentration in unknown ways. In other words, seasonal increases over summer and fall cannot be solely attributed to sediment P release but may also stem from geese feces. Nonetheless in situ water column TP increases are presented as indications of sediment released P and/or geese contributed loading in Section 3.5.

Approach 2 is based on the anoxic sediment release area (Appendix A) and release rates and thus presents a gross estimate. These variables can generally be modeled from water and sediment TP concentrations, except the release rate model does not apply to sediments after a capping treatment. It is not known how much of the lanthanum in the added Phoslock material is still present to inhibit sediment P release. Hence release rates cannot be predicted from sediment TP content after the treatment.

Approach 3 is based on TP and water budgets (Appendix E) and this approach can now be used for the first time (because of the new water budget) since internal load was studied in Swan Lake. Based on the hydrological budget and known or assumed P concentrations, as well as geese counts, a complete P budget was assembled and internal P load could be computed using Approach 3. In this approach, observed P retention [(in – out)/in] is compared to retention computed for a year without internal load (i.e., 2014, when internal load was most likely non existent, L_{int}=0). The difference between the two estimates of retention can be ascribed to internal loading, as explained in Appendix E. We assumed that annual TP concentration averages are similar to those of the growing period for which measured concentrations are available. Consequently, export (out) was estimated as the product of outflow and monitored average growing period TP concentration (available for 2011, 2013, 2014, 2016, 2017, 2018). The initial unmonitored years (2009 – 2012) assume the 2011 value (0.253 mg/L).

Internal load derived from Approach 3 has been variable throughout the study years, and was smallest right after the Phoslock treatment (Table 17). While internal P loading had been drastically reduced by the treatment in 2013 (from 67% of the combined loads down to 0%, Table 17), it appears to slowly regain its importance (57% within the last three study years, 2016-2018).

This trend is supported by the characteristics of bottom sediments sampled in August 2016. Lanthanum and mobile P in the sediments indicated that only half of the sediment P available for release was still bound by lanthanum after the 2013 Phoslock treatment (Section 2.3, Table 8). This means that the amount of lanthanum was no longer large enough to intercept all releasable P in 2016, and releasable sediment P seems to have increased since then, as also indicated by the mass balance-based internal load estimates discussed above.

Previous internal load estimates, available for pre- and immediate post-treatment years via the different approaches discussed above, support those of Approach 3 (Table 17).

Table 17. TP mass balance components used to calculate internal loads. (Assumes zero P flux from the sediments in 2014.)

Year	External Load (total) (kg/yr)	Export (kg/yr)	Measured R (In-out/in)	Internal Load (kg/yr)	Internal/(External + Internal load) (%)	Previous internal load estimates (kg/yr)
2009	23.2	12.1	0.48	59.8	72%	
2010	20.5	10.2	0.50	49.6	71%	
2011	20.0	9.1	0.54	43.0	68%	34 - 59*
2012	25.5	12.2	0.52	58.4	70%	
2013	27.9	7.7	0.72	25.3	48%	0**
2014	28.5	4.1	0.85	0.0	0%	0 - 5
2015	38.9	na	na	na	na	
2016	49.9	12.4	0.75	35.8	42%	"increasing"***
2017	26.7	10.6	0.60	46.1	63%	
2018	33.5	12.6	0.62	53.5	61%	
Averages:						
2009-18	**29.5**	**10.1**	**0.62**	**41.3**	**55%**	
2009-12	**22.3**	**10.9**	**0.51**	**52.7**	**70%**	
2013-14	**28.2**	**5.9**	**0.79**	**12.6**	**24%**	
2016-18	**36.7**	**11.9**	**0.66**	**45.1**	**56%**	

*Literature release rates based on other hyper-eutrophic lakes suggest a considerably higher load (Nürnberg and LaZerte, 2012)

**Internal load of the pre-treatment months (Jan - May, 2013) not included

***Based on the ratio of lanthanum to releasable P in the sediments (Section 2.3)

3.5 Relative contribution of sources to lake TP concentration (modeling)

We used P input from external sources as determined above, which include the stormwater management ponds, runoff from the immediate lake shore area, precipitation, and contributions from waterfowl and internal load (Table 16, Table 17) in a mass balance model (Appendix E) to estimate the contribution from each P source to the TP concentration in the lake (Table 18, Table 19). This conversion considers hydrological conditions by using the flows and thus even constant inputs (e.g. from the atmosphere) can result in slightly differing annual TP concentration contributions.

According to the model structure, the sum of the individual contributions from the considered P loads equals the “observed” annual average TP concentration (Appendix E).

Observed Jun-Sep water column average TP increases presented in Table 18 may indicate effects from geese, or sediment P release, or both. (See also discussion in Appendix D.) These concentration changes were exposed to settling (retention) only for 4 months, and therefore cannot be directly compared to (may be larger than) the (modelled) annual TP averages, which includes settling for 12 months. Nonetheless, it is likely that the large increases in 2011 and 2016-2018 are much influenced by sediment P release. In addition, there is a large contribution by geese in 2016 that would have contributed to the Jun-Sep TP increase. This contribution will then have settled to the sediment from where it can get released as internal loading in later years.

Table 18. Modeled annual average contribution to Swan Lake TP concentration from various sources and monitored (observed) TP (mg/L).

Year	Modeled TP concentration from specific loads						Observed TP	
	SWM Ponds	Shore	Atmos-phere	Geese	External (total)	Internal load	Annual average*	Jun-Sep Increases**
2009	0.003	0.042	0.003	0.023	0.071	0.182	0.253	na
2010	0.003	0.036	0.003	0.032	0.074	0.179	0.253	na
2011	0.003	0.033	0.004	0.041	0.080	0.173	0.253 (6)	0.115
2012	0.003	0.037	0.003	0.034	0.077	0.176	0.253	na
2013	0.003	0.038	0.003	0.037	0.081	0.073	0.154 (7)	0.010
2014	0.003	0.037	0.003	0.042	0.085	0.000	0.085 (7)	0.081
2015	0.003	0.033	0.003	0.096	0.135	na	na	na
2016	0.003	0.027	0.003	0.155	0.189	0.135	0.324 (9)	0.313
2017	0.003	0.031	0.003	0.043	0.080	0.138	0.219 (2)	0.130
2018	0.003	0.041	0.002	0.038	0.084	0.135	0.219 (13)	0.120
Averages:								
2009_18	**0.003**	**0.036**	**0.003**	**0.054**	**0.096**	**0.132**	**0.224**	0.128
2009_12	**0.003**	**0.037**	**0.003**	**0.032**	**0.075**	**0.177**	**0.253**	0.115
2013_14	**0.003**	**0.038**	**0.003**	**0.040**	**0.083**	**0.037**	**0.120**	0.046
2016_18	**0.003**	**0.033**	**0.003**	**0.078**	**0.118**	**0.136**	**0.254**	0.188

*Annual average value is based on all monitored dates for each year (n in brackets); when available, depth samples were pro-rated to represent water column averages. Values for 2009, 2010 and 2012 are monitored values of 2011. 2013 post-treatment (May-Sep) average was 0.106 mg/L, compared to pre-treatment (Feb and Apr) 0.275 mg/L.
Sampling sites are main Site 3, except for 2018, when it was Dock Site 1.
**TP water column average concentration increases (Equation 4) are likely caused by sediment P release (internal load) and water fowl in annually variable contributions.

Table 19. Modelled TP contribution from external and internal P sources for two long-term periods (presented as concentration and percent of total TP input; based on Table 18).

P Source	2009-18		2016-18	
	mg/L	%	mg/L	%
Atmospheric deposition	0.003	1%	0.003	1%
SWM ponds	0.003	1%	0.003	1%
Immediate lake shore	0.036	16%	0.033	13%
Geese	0.054	24%	0.078	31%
Internal (sediment)	0.132	58%	0.136	54%
Total input	0.228	100%	0.254	100%

Based on the relative P contribution from the different TP sources, the importance of various treatments can be assessed. For example, with a treatment that intercepts P release as in 2014 after the 2013 Phoslock application, the long-term TP concentration would be below 0.100 mg/L, indicating eutrophic, rather than hyper-eutrophic conditions. Consequently, phytoplankton would be diminished and Secchi disk transparency improved, possibly to the 2014 growing period of 1.4 m (Table 2), also indicative of eutrophy.

A 50% success in geese and runoff management that decreases TP from these sources by half would only decrease TP concentration by 0.056 mg/L (half of the concentration attributed to geese and shoreline in 2016-18, Table 18) yielding a TP concentration of 0.198 mg/L. These deliberations support the observation on USA lakes with high internal loading, that it is more effective to treat the internal load rather than external load from non-point sources (Osgood, 2017). In the long run, external load has to be managed to constrain the settling and accumulation in the sediments that provide for internal loading in the future.

We were asked to provide an estimate of the potential longevity of a potential chemical treatment to combat internal load. A very crude estimate of the longevity of a chemical treatment that successfully inhibits all current P release predicts that the 2016 sediment P concentration would be accumulated again in about 10 years, for an external P load resembling that of the 2017 and 2018 average (Table 20). This estimate would mean that after a treatment that successfully binds all sediment releasable P, the same deteriorated sediment state as that of 2016 would be achieved after 10 years of external load for the 2017 and 2018 average. Changes in external load and increases in internal load over time (from settled external load) would increase the sediment P pool. Further, this estimate only considers total P and therefore does not exactly predict the P fraction available for release. It also does not consider P migration within the sediment or sediment disturbances or resuspension. We therefore assume an uncertainty of at least 50% to assume the likely period of 10±5 years or 5-15 years.

Table 20. Calculation of the potential longevity of a chemical treatment

Characteristics		Explanation and values	
Settled TP (wet) of 2017-18:		Retention	0.85
kg/yr	25.7	External Load (kg/yr), Table 16	30.08
mg/m^2	469	Lake area (m^2)	54,799
mg/cm^2	0.047		
mg/cm^3	0.047	Assumes accumulation depth of 1 cm	
Years to obtain concentration of recent sediment	9.8	Because of the large uncertainty, 5-15 years	
Sediment TP (Site 1, 2016, Table 8, where the measured TP concentration was largest):			
TP (mg/g dr wt)	2.09		
TP (mg/g wet wt)	0.418	Moisture	80%
TP (mg/cm^2 wet sediment) x1.1	0.460		

4 Task 3 Development of Management Goals and Targets - Water quality goals and treatment triggers

This task includes the definition of a recommended level of service (LOS) for Swan Lake and of appropriate numeric or qualitative targets.

In 1993, an Environmental Management Study (EMS) was prepared by *Swan Lake Limited* (now Daniels Corporation) in support of the draft plan of subdivision for the Swan Lake Community (Cosburn Patterson, Cosburn Ginnerson and Michal Michalski, 1993). This EMS developed a plan to naturalize the former gravel pit as part of the area's growth plan. Swan Lake was deemed a focal point of the development serving as a biophysical resource, offering a diverse natural habitat for aquatic and terrestrial wildlife, and a recreational amenity for passive uses.

Currently, Swan Lake does not have any official management goals. The trails around the Lake are well used by the area residents. Several platforms, docks and bridges enhance the viewing experience. It appears that the water quality of Swan Lake is of importance to the region.

In recent years, signs have been posted at park entrances indicating that the water is not safe for swimming. The City may want to envision certain management options to ensure a specific level of water quality. In order to decide on their necessity, we propose minimum water quality thresholds and discuss other considerations that would trigger specific treatments in Swan Lake. In the long-term, Swan Lake has the potential to be restored to a lower and better trophic state.

Water quality variables determining the trophic state (Section 2.2.1 and Table 2) are used in many thresholds and guidelines. For example, repeat Phoslock treatments were based on a target TP concentration in a German lake with similar characteristics as Swan Lake (Epe et al., 2017). Other guidelines include chlorophyll concentration and Secchi transparency as used in USA Total Maximum Daily Load (TMDL) criteria.

With respect to Swan Lake, the most direct concern would be one for public safety. The Lake is not a source of drinking water, and neither is it used for irrigation or fisheries. Therefore, and in the short-term, if there is any health concern, potential risks could be mitigated by exposure avoidance, including exposure of pets.

Health Canada (HC) guidelines for recreational activities is 20 µg/L microcystin-LR (Health Canada, 2012). But the Ontario Ministry of the Environment, Conservation and Parks (MECP) regards any cyanobacterial bloom as potentially toxic, whether or not toxins are detected in the water upon testing as explained in Section 2.2.5. We are not aware of any guidelines for airborne particles.

Accordingly, we propose the bloom of a potentially toxic strain of cyanobacteria as immediate trigger. If a surface bloom of phytoplankton appears anywhere on Swan Lake, most likely in summer and fall, water samples should be taken to be analyzed by a licenced or Provincial (Ontario Ministry of Environment, Conservation and Parks, MECP) laboratory or by Abraxis strips. Any identification of potentially toxic cyanobacteria or the detection of cyanotoxins themselves could be used as a trigger for signage identifying the potential hazard, at any concentration.

A wide-spread bloom indicates a potential hazard. We propose, if there is a surface bloom covering about a quarter of Swan Lake in 2 out of 4 years treatment dealing with the nutrient sources is required. Such treatment includes internal loading and the largest external source, water fowl feces (Section 5.4).

Other triggers could be based on the long-term monitoring data that have provided information on water quality trends and extremes. The growing period average TP and Secchi transparency reached pre-treatment levels in 2016 (Table 2). It is most likely that such extremes were not reached in 2017 and 2018, probably because of favorable weather influences and better geese management. During average weather conditions or relaxed geese management, water quality may approach the deteriorated state of 2016 again in the future.

Although both, nitrogen and phosphorus are highly elevated, phosphorus is the most likely to limit algal growth and should be treated. The trophic state of hyper-eutrophy starts at a TP threshold of 0.100 mg/L growing period average in lakes (Nürnberg 1996). It would be "nice" to keep urban Swan Lake in the lower category of eutrophy, which would mean a growing period TP average below 0.100 mg/L and an average Secchi transparency above 1.0 m. Such values were only attained after the Phoslock treatment (for TP in 2013, for both variables in 2014) and are recommended as long-term management goals. If these values are achieved, more stringent goals can be set in the future. However, it is unlikely to obtain a mesotrophic state or water quality representative of natural, non-urban lakes on the Canadian Shield. For the short-term, we determined a more appropriate (interim) goal from the relationship between surface TP and Secchi observed in Swan Lake (Appendix F). We propose a combination of TP and Secchi growing period average of 0.150 mg/L TP or less and 0.45 m transparency or better as interim goal.

In summary, we recommend that any of the following triggers elicit a management response:

1. The surface bloom of a potentially or proven toxic strain of cyanobacteria, confirmed by a licenced or Provincial (MECP) lab or by Abraxis strips to trigger direct attention.
2. The occurrence of two blooms within a period of four years that cover at least 25% of Swan lake area.
3. Water quality not compliant with the interim goal of growing period average 0.15 mg/L total phosphorus concentration in the surface mixed layer.
4. Water quality not compliant with the interim goal of growing period average 0.45 m Secchi disk transparency.

The proposed goals are based on monitored data. While a single occurrence of cyanotoxin requires signage, only the average conditions concerning water quality measured as spread of surface bloom, nutrient (TP) concentration, and Secchi disk transparency over a growing period (May/Jun - Sep) should trigger more comprehensive actions as presented in Section 6.

If despite the proposed management the triggers are tripped, an evaluation of the monitoring data (Section 6.6) should determine, whether internal loading has not been successfully curtailed anymore or whether another TP source, e.g., water fowl, or other causes, e.g., climate-related conditions, likely are the reason for unsuccessful treatment.

The long-term management plan would involve more stringent goals through additional measures to achieve a lower trophic state (see Section 6).

5 Task 4 Development and Evaluation of Management Approaches - Management options

Based on the goals and triggers presented in Section 4, we investigated possible management options. In general, we recommend treating the cause rather than the symptom. As the cause of Swan Lake's deteriorated water quality is the high productivity based on elevated P concentration, we discuss the management of external (Section 5.1) and internal P sources (Section 5.2). The recurrence of large sediment P release events indicates the importance of treating internal P loading and is addressed in detail. More advanced treatment options are presented for long-term management.

5.1 Watershed runoff and goose management

Based on the external load determination (Table 15), the uncontrolled runoff from around Swan Lake's shore and waterfowl defecation are responsible for almost all external input into the Lake. Thus, the treatment and decrease of these nutrient sources is recommended. Management options are listed in Table 21.

Table 21. External load remedial techniques

Targets	Specific treatment	Evaluation
BMPs close to lake shore	Stabilize the shoreline; protect the riparian zone Maintain vigorously growing shrubs and trees next to water shoreline Stabilize eroding shoreline Route drainage away from lake Establish vegetation Create and maintain shoreline buffer zones	Some BMPs have been attempted already. To further extend and improve in the long-term.
Treatment of stormwater and other inflows	Settling ponds Wetlands Chemical precipitation and adsorption Oil/grit separation Historic dumpsites	Two stormwater treatment ponds are already in place, but could be monitored further for nutrient output. Historic dumpsites to be reviewed
Education and active management of waterfowl	Poop and scoop ordinance Set up a geese management plan including the monitoring of goose population Regular chasing by trained dogs Destruction of nests Sterilization or hormone treatment (may require special permits) Relocation of geese Limited rearing and special introduction of swans Education respective no feeding of water fowl	Monitoring and management of resident geese has resulted in a significant decline in population, but migratory geese still abundant. According to the goose deterrent professional, relocation is not an effective measure. Swan removal is not popular with residents but feeding should be limited. Educational signage can be enhanced.

5.1.1 BMPs close to lake shore

As 38% (Table 15) of the external P source is estimated to originate in the immediate lake shore, management of these sources is recommended. Many of the suggestions in Table 21 have already been applied in the Swan Lake watershed. Poop and scoop ordinance should be enforced and runoff infiltration augmented, where possible. To this end, recent plantings (Sep 2019) added between the pathway and the water will have beneficial effects by adsorbing nutrients before they reach the water and by discouraging geese grazing. Increasing the naturalization of the shoreline buffer zone with only controlled boardwalk access to the lake would also help.

5.1.2 Treatment of stormwater and other inflows

Two stormwater treatment ponds are already in place and could be monitored further for nutrient output. However, expected nutrient export is only 3%, similar to that of the atmospheric deposition (Table 15).

Any treatment of the runoff bypassing the two stormwater management ponds would not have a notable impact and is not practical, given the runoff volume and physical constraints of the system.

5.1.3 Investigation of historic dumpsites

The potential nutrient contribution by groundwater and historic contamination of the surrounding areas (Figure 18), especially the southern dump site, becomes important when comparing sediment P loads with loads from external sources and should be investigated if possible. While no large contribution is anticipated, a future in-lake treatment could possibly fail because of this potential P source. Background information should be reviewed to determine whether a more detailed study is needed. (There seem to be no nutrient data available for those sites.) A groundwater hydrologist familiar with the Swan Lake area could provide an estimate about water volume quantity and flow direction from those sites towards Swan Lake. Potential boreholes to evaluate nutrient content, especially of TP and SRP concentrations, would help determine the P loading from this potential source.

5.1.4 Education and active management of waterfowl

Water fowl defecation contributed more than 55% to the external P load in Swan Lake (Table 15) and thus is an important load.

Resident geese have recently been successfully managed by bird deterrent consultants through the use of Border Collies, egg oiling and nest removal so that nutrient input can be kept at an acceptable level. Extending the egg oiling, and nest removal program to the adjacent areas (Swan Lake Village) will increase program efficiency. Migrating flocs of Canada geese likely contribute a large amount to the fall nutrient load. The migrating geese just occupy the open water of Swan Lake and cannot be managed in a similar fashion as the resident geese. Additional methods currently used to control the migratory floc include remote control boats and laser deterrent. Geese counts in 2017/2018 were much lower than in 2014/2015, which indicates that the existing program has been effective.

Education of the public to forgo feeding of the geese, and perhaps the discontinuation of the rearing and feeding of swans would also help decrease the nutrient load from this source.

5.2 Internal load abatement (phosphorus release from sediments)

While reducing external load by further abatement efforts is certainly important and is discussed in Section 5.1, internal load appears to be an influential cause of nuisance algal blooms and has to be treated separately. Because of its high biological availability and the timing of its release during the growing season, internal P loading has an immense negative effect on late summer and fall water quality in Swan Lake and triggers algal blooms. This conclusion is supported by the results of the P mass balance (Section 3.5), that indicate that much of the water quality problem in Swan

Lake, including algal blooms and low Secchi disk transparency, stem from internal nutrient sources as partially supported by water fowl feces in recent years.

For these and the following reasons, internal load abatement becomes again necessary after the Phoslock treatment in 2013:

1. Several of the above-mentioned triggers (Section 4) were tripped since 2016. Water quality variables showed elevated TP and chlorophyll concentration, extremely low Secchi transparency and a local cyanobacteria bloom. These conditions indicate that future water quality may not improve without intervention.
2. Water fowl was abundant in several of the post-treatment years and P will have accumulated on the sediments above the added Phoslock layer, leading to dilution of the P adsorbing lanthanum-amended clay. (Potentially in 5-15 years of recent external load the P sediment pool is replenished, Table 20.)
3. The strong hypoxia (not specifically treated, but can decrease with improving trophic state) that still prevails in the summer, facilitates P release as internal loading from P-enriched bottom sediments as seen by increased bottom TP concentration (e.g., Figure 5). Such internal loading can trigger and support blooms of cyanobacteria and is best minimized by a tested in-lake treatment.
4. Climate predictions include warmer summers that would increase sediment P release and improve growth conditions for cyanobacteria (Section 2.5).

Repeat treatments may become necessary, when the triggers are tripped in the future because of newly accumulated sediment P or due to underdosage or disturbance of the chemical.

There are several established restoration techniques that address internal P loading from bottom sediments. Projects are more likely to perform as expected and attain the restoration goals when based on well-studied limnology, P cycles, and nutrient-algal relationships. The selection of restoration treatments is most critical. Not only do major monetary and human resources need to be committed for a number of years, but also a treatment has to be selected that does not do any harm and actually improves water quality.

For example, many costly treatments had been used by the City of Lethbridge, Alberta to treat their urban Henderson Lake (Nürnberg, 2017). "A variety of techniques have been applied, including the aquatic herbicides Reglone and Reward, Liquid Live Micro Organisms, annual additions of grass carp (*Ctenopharyngodon idella*), 5 Solar Bees (solar powered water column mixing devices), and a diffuse aeration system to control rooted plants or reduce algae. None of these treatments has reduced TP concentration, however, and cyanobacteria continued to negatively affect lake water quality." Only a Phoslock treatment successfully reduced TP concentration 10-fold by virtually eliminating sediment P release contributions and noticeably increasing Secchi disk transparency.

Restoration attempts are not always successful and not many can be recommended to treat Swan Lake as explained in the following section and summarized in Table 22. Of these listed treatments only two are limnologically feasible (an evaluation of "3" or above) and are evaluated further with respect to public and regulatory approval and cost (Table 23). The following section explains all listed treatment approaches in detail.

Table 22. Possible remediation techniques targeting internal P load in Swan Lake

Treatment	Evaluation Limno-logical	Comments
Chemical treatment		
Phoslock	5	Immediate improvement depending on season, decrease of sediment P release
Aluminum	4	Immediate improvement in the water column, decrease of sediment P release at pH 7-9
Iron	0	Inefficient in anoxic hypolimnia, may increase blooms
Calcium	1	Inefficient in a polymictic and hypereutrophic lake
Capping		
Clay	1	Unspecific to P, not much P retention
Physical treatment		
Hypolimnetic withdrawal	0	Inefficient in a polymictic lake, not possible without an outlet
Flow augmentation	0	No clean, high-volume water source available
Destratification, aeration, oxygenation	0	May fertilize upper layers & increase blooms
Sediment removal, dredging	2	High dredging and removal costs & potentially toxic sediment in *Block 9*
Sediment solidification		
Removal of bottom feeding fish	5	Increase in efficiency of chemical treatments, Decrease of sediment P release
Planting of rooting aquatic plants	2	Could decrease sediment P release

Table 23. Limnologically feasible (deemed effective) remediation techniques targeting internal P load in Swan Lake

Treatment	Evaluation				Comments
	Limnological evaluation	Public Acceptance	Regulatory Approval	Cost	
Alum (aluminum sulfate)	3	2	3	Low	Immediate improvement at pH 7-9, sulfate addition
Poly aluminum chloride (PAC)	4	2	3	Low-medium	Immediate improvement at pH 7-9, P desorption possible
Phoslock	5	4	5	Medium	Immediate improvement at a wide range of pH, permanent chemical binding

Evaluation: 0, not; 1, little to 5, highly recommended.

5.2.1 Chemical treatment

5.2.1.1 Lanthanum-amended clays (Phoslock)

There are several clay-based P-binding materials of which lanthanum-modified bentonite has been most researched and successfully applied in lakes. The only commercially available form of this material is called Phoslock and was developed by the Commonwealth Scientific and Industrial Research Organization (CSIRO) of Australia. Phoslock is made by combining bentonite, a naturally occurring clay, with lanthanum (La, from $LaCl_3$), a metallic element that combines with phosphate.

Phoslock has been used in more than 200 (as of 2018) applications (Copetti et al., 2016) throughout Australia (Robb et al., 2003), New Zealand (Burns et al., 2009; Hickey and Gibbs, 2009), China (Liu et al., 2012), Europe (Lürling and Oosterhout, 2013; Meis et al., 2013, 2012; Spears et al., 2013), and the USA (Bishop et al., 2014) to treat, lakes, rivers, stormwater ponds, and reservoirs. It was applied without any apparent detrimental consequences in several drinking water reservoirs, in Clatto reservoir in 2009, which is a Scottish Water's Emergency Supply reservoir (Meis et al., 2012), in a small (active) reservoir of Wessex Water in SW England in 2008, and in several Brazilian reservoirs (Nigel Trail, pers. comm.).

Phoslock treatment has been applied in Canada because of its general lack of toxicity (Lürling and Tolman, 2010; Ontario Ministry of Environment, 2009). (A literature review of peer-reviewed papers on the subject published before 2016 is included as Appendix J in this report.) Phoslock received the US and Canadian NSF/ANSI Standard 60 Certification for use in drinking water in 2011 (Finsterle, 2014), the ISO 9001 for Quality Control and ISO 1400 for Environmental Management standard certifications.

Phoslock was applied in several meso- to hyper-eutrophic Canadian lakes in Alberta, Ontario and Quebec as reviewed (Nürnberg, 2017). Phoslock is most often applied to highly deteriorated lakes with a large internal load compared to external P input, e.g., in urban hyper-eutrophic Swan Lake where average TP concentration decreased from 0.247 mg/L to 0.099 mg/L in the first and to 0.060 mg/L in the second post-treatment year, practically eliminating internal loading for two years (Nürnberg and LaZerte, 2016). Nonetheless, its working mechanism is the capturing of any P released from the sediment, even when it is small (Ross et al., 2008) over a wide range of pH values (Mucci et al., 2018).

Phoslock consists of bentonite clay with a high exchange capacity, in which naturally adsorbed cations have been replaced by the rare earth element, lanthanum (La). La is the 29th most abundant element in the Earth's crust present on average at 30-40 mg kg-1 and is almost three times as abundant as lead (Wedepohl, 1995). In the presence of orthophosphate (SRP) La forms the highly stable mineral called rhabdophane (LaPO4 nH2O) (Ross et al., 2008). The amount of Phoslock necessary to inactivate the existing P is based on phosphate (SRP) in the water and releasable (mobile) P in the sediment (pore water P, iron bound P, and labile organic P; Reitzel et al., 2005). Phoslock contains 5% La by weight, and P is adsorbed by La at a molar ratio of 1:1 (Ross et al., 2008). In dosage calculations only the freely available P components are considered, i.e., SRP in the water at time of application and mobile P, mainly iron-bound P, in several centimeters of surficial sediment (Meis et al., 2012).

Phoslock is effective without changing the oxygen content of the hypolimnion. But decreases in internal load lead to decreases in productivity and organic settling that would eventually decrease sediment oxygen demand and hypolimnetic oxygen concentration as long as external loads are also decreased or already low. The immediate effect of Phoslock on lake water quality is the decrease of TP and SRP that causes a decrease in phytoplankton and cyanobacteria. Consequently, cyanobacteria blooms are less frequent or spatially and temporally more confined. Such improvement has been reported repeatedly in Phoslock treated lakes (Copetti et al., 2016; Epe et al., 2017b; Nürnberg, 2017; Nürnberg and LaZerte, 2016; Spears et al., 2016).

Phoslock has been applied in Canadian waters since 2008 where it has been used in Ontario, Alberta, and Quebec (Nürnberg, 2017). This treatment has already been tested in Swan Lake and found successful (Nürnberg and LaZerte, 2016). An application in Swan Lake in spring 2013 decreased lake trophic state from hyper- to eutrophic for at least two growing periods. This was the best water quality recorded in recent years. The deterioration in three years after the treatment may have been triggered by overlooked external P inputs from shoreline runoff and an increase in geese population, providing nutrient-rich fecal pallets to enrich water and sediment.

5.2.1.2 Aluminum compounds

Aluminum (Al) compounds have been used for several decades and 114 lakes were identified as being treated with aluminum sulfate (alum) or poly aluminum chloride (PAC) to reduce internal phosphorus loading (Huser et al., 2016b). Treatment longevity varied for lakes of different morphometry, applied dose and other factors, and averaged at 11 years. Shallow, polymictic lakes had average longevity of about six years.

Alum applications are not permitted in many European countries, because of the increase of reducing compounds by applying sulfate (in alum) that is easily turned into hydrogen sulfide under anoxic conditions hence potentially increasing spread and intensity of hypoxia. Instead, poly aluminum chloride (PAC) is the only permitted Al-based treatment in most European countries. While alum is relatively inexpensive and used preferably in North America, PAC is more costly, probably approaching the cost of the Phoslock material.

Aluminum compounds can be expected to remove P in the water column and to diminish internal load in Swan Lake by inhibiting any P flux from the sediment. Applications of aluminum sulfate (alum) or aluminum poly chloride (PAC) in neutral to alkaline water (such as Swan Lake's pH of mainly 7 to 8 in the open water), can be expected to be non-toxic (Gensemer and Playle, 1999). Applications would produce a flocculent gelatinous precipitate of aluminium hydroxide, $Al(OH)_3$, which is chemically stable up to a pH of 8.8, even under the hypoxic conditions in the water and sediment surfaces encountered in Swan Lake.

However, P is easily desorbed from these Al precipitates, reducing treatment efficiency at higher pH values that occasionally occur in Swan Lake during high phytoplankton biomass in the summer and fall (City of Markham staff, personal communication, Jan 22, 2020) and after aging processes, (de Vicente et al., 2008; Reitzel et al., 2013). Aging of the chemical prevents P adsorption and is stated as one of the reasons for less efficient P removal by alum than expected from dosage calculations (James and Bischoff, 2020). A possible additional complication was demonstrated in a shallow hypereutrophic lake with characteristics similar to Swan Lake (Half Moon Lake, WI, 4 m maximum depth, 0.110 mg/L TP) where the alum compound remained floating above the sediment without creating the expected barrier against sediment P release (James, 2017).

The application of any aluminum compound has not been permitted to date in Ontario and other Provinces. The response from MECP to an enquiry by the City of Markham (Memo Jan 6, 2020) is non-committal, alternatively stressing the importance of external load abatement, monitoring and fish management.

5.2.1.3 Other compounds

Iron is not expected to remove P in Swan Lake permanently because of its severe hypoxia that would reduce iron hydroxides and therefore release any adsorbed P back into the water as phosphate (this process is similar to the sediment P release found in soft water lakes). Further, iron addition may enhance cyanobacteria blooms, because of potential iron limitation. For example, the addition of iron chloride created an algal bloom in the first year of application in a small lake in British Columbia; it was attributed to alleviated iron limitation (Hall et al., 1994).

Calcium compounds are not expected to remove P in Swan Lake sufficiently. Attempts of precipitating P with calcium in Alberta and German lakes (Koschel et al., 2000; Prepas et al., 2001) have not been consistently successful to be recommended.

Clay without amendment is unspecific to P and does not retain it in the sediments. A clay layer on the sediment will not provide a barrier to P release.

5.2.2 Physical treatment

5.2.2.1 Flow management (hypolimnetic withdrawal and flow augmentation)

A flow-related treatment to reduce internal P loading in stratified lakes is bottom water **(**hypolimnetic) withdrawal (Nürnberg, 2007). This treatment withdraws lake water preferentially from the nutrient-rich hypolimnion instead of the surface and hence increases P export. However, Swan Lake is shallow and mixes occasionally throughout the summer with only small water in- and outflows, so water withdrawal is not applicable and such treatment cannot be recommended.

Augmented inflow is another flow-related treatment. This treatment requires clean, high-volume inflow water. Such water source does not exist for Swan Lake and the lack of an outflow prevents increased flushing.

Circulating Swan Lake water after treatment is not feasible for several reasons. The water cannot be effectively treated in a stormwater pond because it is not rich in settling sedimentary particles. Treatment in a waste-water treatment plant is also ineffective because of Swan Lake's relatively low P concentration compared to sewage.

5.2.2.2 Destratification, aeration, and oxygenation

Destratification, aeration, and oxygenation are treatment options that are sometimes applied to reduce internal P loading from bottom sediment. However, it has been shown in various studies that these techniques do not always manage to aerate the sediments sufficiently to inhibit sediment P release. None of these physical treatments is advised in Swan Lake, as explained below.

Destratification and **artificial mixing**: Thermal stratification delays the nutrient intrusion into the surface water. Summer destratification would distribute the nutrients throughout the water column at a time when light and temperature are high and therefore would benefit cyanobacteria. Consequently, artificial destratification can be expected to enhance cyanobacteria blooms throughout the summer and fall. Destratification by aeration has decreased water quality by increasing P in a study on 212 Minnesota lakes (Beduhn, 1994) and produced a bloom of deep layer cyanobacteria (Nürnberg et al., 2003). However, artificial mixing can shift the phytoplankton genera from cyanobacteria to green algae, if certain conditions, including permanent and deep destratification, are met (Visser et al., 2015).

A decorative fountain (Figure 20) is installed annually from May 1 to November 6 and operates from 6 AM to Midnight in Swan Lake. While mechanical mixing may prevent the accumulation of cyanobacteria on the surface during bloom conditions, the fountain does not appear to actually decrease the biomass. Instead, the water still appears green and the Secchi disk depth and other water quality variables are similar at Dock Site 1 to that of Deep Site 3 (Section 2.2.5). The fountain action is probably not strong enough to affect the phytoplankton either positively (by preventing the accumulation of phytoplankton on the surface) or negatively (by mixing and distributing nutrients from lower depth to the surface). However, a negative effect is possible as long as there is internal loading, because Swan Lake's occasional stratification prevents deep nutrients reaching the surface (after discussion with a fountain provider at the NALMS conference November 2019).

Area residents have raised concern about poor air quality associated with airborne particles sprayed by the fountain, and although there is no evidence to support any health effect, it is suggested that the installation of the fountain be postponed until water quality is improved.

Figure 20. Fountain close to the south end Dock, Site 1 (June 18, 2019, City of Markham).

Aeration and oxygenation: These treatments could not be applied without further destratifying the water column and effectively mix Swan Lake with the consequences discussed above. Deep layer aeration is not possible because of the shallow depth of the Lake. Neither aeration nor oxygenation can be expected to effectively inhibit internal P loading from Swan Lake bottom sediment because the sediment oxygen demand in a hyper-eutrophic lake is so extremely high. The increase in hypolimnetic temperature associated with mixing would also increase sediment oxygen demand and P release (Bryant et al., 2011), potentially increasing internal P loading.

Oxygenation did not improve mixed layer water quality in 4 out of 5 Danish lakes (Liboriussen et al., 2009). Only hypolimnetic aeration or oxygenation coupled with a chemical to adsorb P, such as iron and aluminum, have consistently decreased internal load and delivered positive effects on trophic state (Cooke et al., 2005; Moore et al., 2012), indicating that chemical adsorption is the main influential agent. Such a treatment involves extreme technical and financial resources and a costly long-term commitment.

An extensive literature review of about 70 references (Nürnberg unpublished studies) did not reveal any positive effects except that hypolimnetic anoxia was generally prevented where the systems were sized appropriately and fish habitat was consequently improved. But other water quality indicators including TP concentration and cyanobacterial biomass did not noticeably decrease in lakes treated with hypolimnetic aeration or oxygenation.

In conclusion, the techniques mentioned here have several drawbacks in common:

a. They need to operate continuously to be effective, which requires constant energy input, management, and maintenance.
b. They require equipment on the lake surface that is open to vandalism and present navigational challenges.
c. They mainly address the symptoms, but not the cause of the problems.
d. If they are interrupted, water quality can become worse than before because oxygenation increases P retention and decreases export. This means that P accumulates on the sediment during operation and is released when aeration treatment is discontinued. This accumulated sediment P would lead to enhanced P release once the hypolimnion becomes oxygen depleted again.

5.2.3 Dredging

Sediment removal is very expensive and would require an extensive permitting process. Any removed material would have to be examined for toxicity and then removed away from the lake; any drainage would have to be captured to prevent fertilization of ground and surface water and probably trucked away at great expense. This is not feasible, especially in Swan Lake, where pockets of potentially toxic sediments from former fills may still exist in the southeast area (Block 9), so that previous studies have recommended to keep bottom sediments undisturbed (WESA's Risk Assessment 2006). This method will also be very costly (in the order of tens of millions of dollars) given the amount of sediment to be removed and disposed of.

5.2.4 Sediment solidification (auxiliary treatment)

5.2.4.1 Bottom dwelling fish

As pointed out in the previous report, bottom-feeding fish species including gold fish (*Carassius auratus*) and carp (*Cyprinus carpio*) can stir up the sediments and increase internal loading, even after a sediment capping treatment such as Phoslock. A study on a shallow USA lake determined that the sediment mixing depth increased more than twofold, down to 13 cm, in carp-exposed areas, which increased the amount of releasable P by up to 92% (Huser et al., 2016a). Sightings of gold fish (Nov 2, 2015, Rob Grech, City of Markham) raise the question whether there are enough gold fish or carp in Swan lake to present a problem. Because of the low oxygen concentration throughout the summer, these fish would be expected to be around the shoreline, where they could be spotted by routine monitoring. If detected, it may be useful to determine their exact biomass by electrofishing.

If carp abundance is determined to be high, its management is quite difficult and remediation sometimes strives for eradication. Nonetheless, since the carp is mostly localized and can be isolated in Swan Lake, several management options can be suggested.

Drawdown is the technique of lowering the water level to expose lake sediments. It is useful for both plant and fish management. It controls certain non-native plant species, like Eurasian water-milfoil and curly-leaf pondweed, especially if done over the winter. It has been successful in greatly reducing carp populations. This would require pumping to drawdown the water level below the normal winter level to expose as much of the bottom area of the ponds as possible throughout the late fall and winter, when temperatures are below freezing.

Fish poison, Piscicides - The most common piscicide is *Rotenone* that eradicates all fish, if applied levels are high enough. According to the *American Fishery Society (AFS) Guideline Manual*, goldfish (and therefore also carp) requires a comparably high dosage of 500 µg/L to kill 50% of this species. The efficiency of *Rotenone* is temperature dependent (but not pH dependent) and it is best to apply it in the fall, when its half-life time is about 10 days as compared to less than 1 day during the warm season.

Another piscicide is the antibiotic *Antimycin*, which is especially lethal to sunfish. Again, higher dosage is required for carp (15 µg/L). However, there are two disadvantages: (1) it is much more expensive than *Rotenone* and (2) its efficiency is pH dependent and in hard-water (pH >8.5) it would be lethal only for several hours. Because of Swan Lake's hard water, this piscicide is not an option.

As far as we know, both piscicides are discouraged by MECP and the Ontario Ministry of Natural Resources and Forestry and need special permits.

Stocking with game fish and planting of native water plants - After the eradication of carp and other fish, the fish population has to be initiated by stocking species that can survive the conditions of the lake and prey upon young carp. Besides controlling sediment disturbance by bottom dwelling fish, the anticipated changes in fish population should benefit water quality in general by a method called biomanipulation. This treatment is based on decreasing the population of fish that feed on the zooplankton, like carp, sunfish and other small warm water fish. Since zooplankton feeds on algae, maximizing their population will increase water clarity. A fish specialist has to evaluate which type of fish could be successfully introduced, once the carp has been drastically reduced.

In addition, insectivorous fish that prey on mosquito larvae would be a useful addition to the fish population of the ponds to reduce mosquito infestation. Possible species include the mosquito fish, *Gambusia affinis* (top-minnow) and guppies (*Precilia reticulata*), as described in Smayda and Packard (1994).

Carp barriers - To keep carp from returning to the Lake after breeding in adjacent areas like the Stormwater management ponds, barriers can be installed, if it should be determined that fish can swim into Swan Lake at times of overflowing stormwater management ponds.

Public Education - The prevention of goldfish (*Carassius auratus*) introduction through public education is suggested. Signage may be considered stating that fish introductions from other bodies of water or an aquarium is illegal, can spread diseases, and alter fish populations so that water quality could be degraded.

5.2.4.2 Planting of aquatic plants (macrophytes)

Currently, Secchi disk transparency is too shallow to encourage any growth of water plants (macrophytes) on Swan Lake sediments. If other treatments successfully increase visibility, plantings to consolidate the sediment could be envisioned. For example, plantings of the macro algae, *Chara*, successfully consolidated sediments in oxbows adjacent to the River Danube in Vienna, Austria (Karin Pall, systema GmbH, Vienna, Austria, personal communication). Macrophyte planting may be considered a long-term strategy that maintains water quality after a chemical sediment treatment and fish removal.

5.3 Phytoplankton and cyanobacteria control

Swan Lake has rarely shown elevated cyanotoxin levels and much of the apparent water quality problems are due to algae rather than cyanobacteria (Section 2.2.5). Consequently, the treatment of harmful algal blooms (HABs) seems less urgent. Nonetheless, we here include some approaches to manage such occurrences that may be enhanced in the future. Predicted climate changes for the Markham area include warming temperatures, which can benefit cyanobacteria, and more extreme storm events, which can increase non-point nutrient sources.

There are many approaches offered to diminish phytoplankton growth and cyanobacteria blooms. Such methods involve ultra sound sonar emitters, electrical magnets, plant extracts, and the inoculation with specific bacteria or algae. As reviewed and evaluated in detail for small hyper-eutrophic lakes (Lürling et al., 2016), there are no proven benefits for most of these techniques, and not many successes are documented in the peer-reviewed scientific literature.

Some approaches that are more aggressively marketed are further discussed (Table 24). Very few of these methods may be helpful in maintaining the water quality after initial treatment, as discussed below.

Table 24. Techniques targeting algae and cyanobacteria

Treatment	Evaluation*				Comments
	Limno-logical Effectivenss	Public acceptance	Regulatory Approval**	Cost	
No proven benefit					
Algaecides	2	3	permit	low	Usually short-lived
Ultra sound sonar	0	5	5	med	Not effective, has been applied in ON
Diatomix	0	n.a.	permit	low	Usually short-lived
Ambiguous benefits					
Plant extracts, bacteria	2	4	4	low	Not effective
Barley straw	2	4	4	low	Not reliably effective
Positive benefits, but short-lived					
Algae removal (harvesting, flocculation, coagulation)	2	n.a	permit	med	Repeat application within growing period
Hydrogen peroxide	2	n.a.	permit	low	Specific to cyanobacteria, does not perform well with algae present

**Evaluation: 0, not; 1, little to 5, highly recommended. N.a., not known*
***Permit: requires approval or a permit*
Limnological effectiveness has to be at least 3 to be recommendable.

5.3.1 Algaecides

In small lakes and ponds, chemicals are sometimes used to kill excessive algae, but we usually do not recommend such a treatment. While it is often immediately effective, it may need to be repeated several times per year as it only treats the symptom of nutrient enrichment. 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 die off, oxygen depletion can be aggravated and result in fish kills and internal P load. In Ontario, a permit from MECP to purchase and perform extermination is required, and the treatment must be performed by a licensed pesticide applicator. Algaecides remain active for days after application, depending on the pH and temperature of the water, and could spill into groundwater and creeks downstream.

5.3.2 Ultra sound sonar emitters

A literature review on the potential effects of various ultra sound emitters mainly on *Microcystis aeruginosa* concluded that ultra sound treatment of cyanobacteria is premature (Rajasekhar et al., 2012). The authors acknowledged effects of sonication on cyanobacterial growth inhibition by the collapse of gas vesicles/vacuoles, membrane and/or cell wall disruption, and the interruption of photosynthetic activity observed in laboratory studies. While these effects did inhibit *Microcystin* biomass, much of the results were laboratory-based and used higher intensities than feasible in field applications. Rajasekhar et al. (2012) therefore suggested that more studies are needed to prove the benefit of ultrasound application to inhibit cyanobacterial growth at the larger scale of lakes.

These conclusions have not changed in a more recent review by different authors (Park et al., 2017). There is still the lack of conclusive results in field applications, as only a few field and pilot tests in small reservoirs were reported. The authors conduced that "There is a lack of information on the upscaling of ultrasonication devices for HAB [harmful algal blooms] control on larger water bodies, considering field influencing factors such as rainfall, light intensity/duration, temperature, water flow, nutrients loading, and turbidity."

Experience in the Netherlands (Lürling and Tolman, 2014, and Lürling personal communication, 2019) lead to the conclusion that low energy, low frequency ultrasound does not hamper cyanobacterial growth and control blooms. Only high energy ultrasound is effective at killing cyanobacteria. However, it is non-selective and also kills beneficial algae, zooplankton, and other aquatic life (Holm et al., 2008).

A recent application in a Muskoka, Ontario lake, Three Mile Lake, did not prevent cyanobacteria blooms, as obvious from the advisories by the Muskoka District Health Unit of Aug 22, 2018 and Aug 27, 2019.

5.3.3 Diatomix

The Diatomix website (https://algaenviro.com.au) announces: "Diatomix is a nano-silica nutrient mixture that has all the micronutrients required for growth of diatom microalgae adsorbed into the amorphous nano-silica structure. As only diatoms have a requirement to take up silica, they are the only algae that benefit from the micro-nutrient boost. This means that the diatoms successfully out-compete the other algae for nutrients, and reduce blue-green algae growth in a natural way." A Google scholar search (Sep 2019) does not reveal any peer-reviewed article, nor any report on the application of Diatomix in lakes (search word "Diatomix", Sep 20, 2019). Given the lack of any scientific data, we do not recommend this product.

5.3.4 Liquid Live Micro-Organisms (LLMO)

The inoculation with specific bacteria or algae for three growing periods did not reduce nutrient content nor did it prevent the cyanobacteria blooms in a small urban lake, in Lethbridge, Alberta (Nürnberg, 2017).

5.3.5 Barley straw and other plant-derived chemicals

The results concerning barley straw or extracts and other plant-derived extracts to control cyanobacterial are ambiguous. While there are many claims based on theoretical and laboratory studies, other experiments do not show beneficial effects in controlling or decreasing studied cyanobacteria (Lürling et al., 2016). Instead, possibly problematic results, including increases of toxicity and nutrients, and decrease of oxygen were reported. Because of the reported uncertainty of success and the repeat dosage required, we do not recommend plant extracts for the treatment of Swan Lake.

5.3.6 Phytoplankton removal (harvesting, flocculation, coagulation)

The harvesting of algae or cyanobacteria is usually done for creating biofuel, livestock feed, or food supplement. We could not find any method considered a management against cyanobacteria. The high water content and comparably low concentration of TP would make such management futile.

Coagulation and flocculation are other ways of removing algae for the sake of restoration. An addition of chemicals, such as alum, poly-aluminum chloride (PAC), iron chloride, or a more environmentally friendly, biodegradable material, chitosan, have effectively been used to strip the water column of organic substances including phytoplankton. Such applications are usually short-lived and have to be applied at least every growing period to be effective.

These techniques work in the laboratory at small scale and positive results of field trials have occasionally been reported in the scientific literature (Jančula and Maršálek, 2011; Lürling et al., 2016). In some cases, natural soils and clays or Phoslock are modified with flocculants to effectively remove cyanobacteria from the water column. This technique entraps the cells in the flocks so that they stay intact, without any toxins or nutrient release. These techniques may be useful, if there is a need to abate increased blooms with elevated and toxic cyanobacteria in the future.

5.3.7 Hydrogen Peroxide

Hydrogen peroxide has been tested for several years. It selectively affects cyanobacteria by rupturing gas vacuoles and inhibiting photosynthesis and growth. Past studies determined that the necessary concentration to kill and rapture the cyanobacteria cells leads to an simultaneous release of cyanotoxins, therefore increasing potential adverse effects related to toxicity (Lürling et al., 2014). Hydrogen peroxide dosage determination is complicated because the smallest working dose has to be determined in each system to avoid toxicity to zooplankton (Spoof et al., 2020; Weenink et al., 2015).

A large presence of eukaryotic algae ("good algae", i.e., green algae and diatoms) can degrade hydrogen peroxide, so that they protect cyanobacteria in the phytoplankton against oxidative damage, rendering the treatment less effective (Matthijs et al., 2016). Swan Lake's phytoplankton consists mainly of such eukaryotic algae (Section 2.2.5) and therefore this treatment would not decrease cyanobacteria and the total phytoplankton biomass. Further, application of hydrogen peroxide has to be done relatively frequently, possibly within weeks or months throughout the summer and fall to be effective (Matthijs et al., 2016). We do not recommend the use of hydrogen peroxide as treatment in Swan Lake because it is likely less effective for Swan Lake's phytoplankton composition, needs repeated applications within the growing period, and is still a new and relatively little field-tested method with possible negative effects for "beneficial" phyto- and zooplankton.

5.4 Preferred approaches

The need to manage and curtail internal load has been discussed in the published literature. Studies on highly eutrophic lakes with continuous internal sources determined that catchment-based restoration techniques do not necessarily show noticeable effects on the water quality unless the internal sources are managed as well (Osgood, 2017).

Because 62% of the total P load was internal in 2017 and 2018 (Table 17), internal load abatement is clearly the most promising treatment to decrease the nutrients that fuel phytoplankton blooms. More than half of the external load stems from waterfowl (Table 16) and therefore goose management is the next important management activity. Most of the remaining external input stems from untreated shoreline runoff (Table 16) and should be managed as well. The management of these three nutrient sources together in a combined approach should decrease Swan Lake productivity and thus increase its water quality.

As immediate approach to prevent cyanobacteria blooms, we do not recommend any symptomatic treatment (Table 24). It is preferable to treat the causes rather than symptoms of eutrophication, nonetheless we here provide some advice to treat algae growth and cyanobacteria blooms directly. In the case of increased blooms with elevated and toxic cyanobacteria in the future, a treatment that targets such blooms directly may be needed, if the nutrient-related approaches are not deemed sufficient.

To summarise, a combined management approach is suggested. Because the most limnologically feasible treatment approach is a chemical treatment of the internal load with lanthanum or aluminum (Table 22, Table 23), any management option that increase the efficiency of such internal load treatment are recommended. They include the prevention of sediment disturbance by the management of bottom-dwelling fish, and generally the management of external inputs because they settle to the sediment and can render the chemical treatment less effective. Of the external loads, the diminishment of contaminated shoreline runoff and geese are the most important management goals. We also recommend to further investigate the potential runoff from the historic dumpsites.

To summarize, a long-term strategy needs to be developed to maintain and enhance the Lake's water quality in a sustainable manner. This strategy may include the following components:

1. Repeat treatment to diminish internal load from the bottom sediments
2. Continue with geese management (to reduce population further, especially of migratory geese)
3. Continue with fish management (to avoid reintroduction of carp)
4. Continue water quality and cyanobacteria monitoring

6 Task 5 Implementation Plan for Preferred Approaches

6.1 Internal load abatement

Of all possible internal load abatement techniques only P precipitation and capping with either aluminum poly chloride (PAC) or Phoslock are limnologically feasible in Swan Lake. Such treatment is especially applicable because there is only little permanent outflow where the chemical could be lost, and there is no large inflow. The external manageable nutrient input that could interfere (external load in recent years without geese and precipitation) is only 15% of the total load (12.4 of 81.8 kg/yr, Table 16, Table 17). Of these chemicals, a Phoslock treatment is acceptable to regulatory agencies and has already been applied in Swan Lake. An application in spring 2013 decreased lake trophic state from hyper- to eutrophic for at least 2 growing periods. This was the best water quality recorded in recent years (Table 2). Therefore, the applicability of a Phoslock treatment to Swan Lake is considered in more detail.

6.2 Phoslock application

6.2.1 Permits and approvals requirements specifically for treatment with Phoslock

No permit is required and only monitoring is recommended as prescribed in the Standard Operating Procedures for Phoslock applications (SOP, Lake Simcoe Region Conservation Authority 2010). The SOP suggests specific pre- and post-application monitoring (Table 25).

Table 25. Monitoring efforts recommended in the *Standard Operating Procedures for Phoslock* applications by the LRSCA Phoslock Pilot Steering Committee

Data requirements	Variables
Water chemistry	Phosphorus, alkalinity, nitrogen ("filtered and unfiltered"), total and dissolved lanthanum
Water physics	pH, dissolved oxygen, salinity, redox (or do), water temperature, turbidity, conductivity
Biology	Phytoplankton
Morphometry and hydrology	Lake volume, inflows, outflow, depth, hydrology, catchment information

Because of the detailed monitoring effort summarized and interpreted in several reports and peer-reviewed publications, the pre-monitoring obligations are fulfilled by the 2011-19 monitoring effort. Additional monitoring with respect to the sediment characteristics (Section 6.2.3) and post-treatment monitoring (Section 6.2.4) are suggested.

6.2.2 Timing and location

The previous Phoslock treatment worked best in the second year of application. While the P concentration decreased in the same growing period, Secchi transparency, the indicator of phytoplankton biomass, did not improve until the second year. It is most likely that phytoplankton was already thriving in the spring of 2013, because the originally planned treatment had to be delayed due to ice on the Lake. Consequently, we recommend a treatment as early in the spring as possible. Because ice cover is rare in late winter on Swan Lake, a March treatment could be envisioned.

Any treatment best reaches all areas in Swan Lake, especially the bay north-west of Bridge Site 3, which typically exhibits deteriorated water quality (Section 2.2.8) but was not treated in 2013 (Figure 12). Further guidance on the spatial extent of a chemical treatment would be available after the suggested sediment analysis.

6.2.3 Dosage and application frequency

Specific sediment characteristics are needed to determine a more informed dosage. The efficiency of Phoslock depends on the quantity of the material in relation to the releasable sediment P. It is quite possible that the characteristics differ at different locations in Swan Lake (e.g., at Site 3, where no Phoslock was applied), and that Phoslock from the previous application has moved to deeper areas. Therefore, a more informed dosage can only be obtained after interpretation of the results attained by the suggested sediment analysis.

To determine the feasibility of an in-lake treatment, targeted at the bottom sediments, and for dosage identification, certain sediment characteristics should be known. A detailed monitoring plan concerning bottom sediment is presented in Appendix J.

The determination of sediment fractions (described in Section 2.3) is not trivial and we propose the services of the *Nowak Institut*, that is specialized in the analysis of fractions needed for the calculations of the dosage for chemical applications (Appendix J). Further, obtaining depth specific samples with a sediment coring device is best accomplished by an experienced consultant. The total costs include the collection of the sediments and the laboratory analysis, shipping, and interpretation and dosage calculation are outlined in Table 26.

Table 26. Proposed monitoring before the Phoslock treatment, potential costs.

Task	Responsible	Costs
Sediment analysis (20 cores at 5 sites, 2 depth layers, or at 10 sites 1 layer)	Nowak Institut	~$10,000 (6,777 Euros)
Sediment collection	Consultant: Fieldwork specialist	$8,000
Interpretation	Consultant: Limnologist	$2,000
Total		~ $20,000

Nonetheless, we provide a cost estimate for a comparably high dosage because Swan Lake is hyper-eutrophic and carbonate-rich so that P and carbonate in the water column will compete and combine with Phoslock before it settles to the bottom sediments. Further, a higher dosage should achieve a relatively thick layer of Phoslock on the sediment which is necessary to prevent leakage from disturbed sediment areas in this shallow lake. Even at these application rates, we do not expect any toxic effects due to lanthanum, because Swan Lake is well-buffered. Since Swan Lake does not have an outlet the treatment would not have any downstream effects. Even though toxicity is likely not of concern, active periods for amphibians and turtles, especially during their egg laying stages, could be avoided by appropriate timing of the application.

We usually recommend a single large dosage application rather than several small dosage treatments but in Swan Lake this strategy is debatable. The variable geese impact and climate-related variation in shore line runoff may add settling material that provides further P from the bottom sediments. In addition, depending on the carp and gold fish abundance, disturbance of the Phoslock layer may have to be considered. Therefore, we present several alternative dosage scenarios for consideration, once further sediment and fish information becomes available.

After the second Phoslock treatment, a further treatment may not be necessary for a longer interim period because of the larger dosage and if water fowl treatment and other BMPs prevent the accumulation of new sediment P above the Phoslock layers. Water quality monitoring and vigilance for any cyanobacteria blooms would ensure that triggers are tripped when necessary and the goals are met. With further reductions in external loads and as water quality improves, the TP and Secchi triggers should be re-evaluated.

6.2.4 Operation and maintenance

The Phoslock application is expected to provide enough benefits for at least 2-3 growing periods, probably longer if dosage and application coverage are enhanced. Specific operation and maintenance suggestions include routine monitoring to determine treatment effect and the need for repeat or additional treatment.

In addition to the routine monitoring (Appendix G) that is needed for the application of targets, there are additional considerations for the monitoring before and after a lake treatment. With respect to the water, the monitoring of Site 3, would help confirm that the in-lake treatment indeed decreased sediment release of P. While the shallower stations are much effected by internal loading and are expected to have decreased TP concentration as well, the most direct proof is gleaned by monitoring TP and SRP profiles throughout the 0-4 m water column at Site 3.

In addition to the routine monitoring (Section 6.6), the monitoring of the chemical applied by the treatment is recommended.

6.2.5 Roles and responsibilities

Phoslock Environmental Technologies Ltd (Contact: Nigel Traill <ntraill@phoslock.com.au>) will provide the material and the application (including boat rental and long hoses for sheltered areas and bays).

The City of Markham would set up and conduct the water-related routine and treatment-specific monitoring program (Appendix G).

6.2.6 Phoslock treatment costs

Estimated costs for a potential Phoslock application are based on recent (Oct 6, 2019) communications by Nigel Traill, the former Head of International Business Development, Phoslock Environmental Technologies Ltd (Table 27). We present a slightly higher dosage rate of 35 tonnes than the 25 tonnes applied in 2013. Because Swan Lake is hyper-eutrophic and frequented by geese, a higher rate of 6.4 t/ha or even more may be necessary for most efficiency and longest prevailing success. A more exact rate can be confirmed by the sediment fractionation recommended in Section 6.2.3. The highest application rate was 6.8 t/ha in 18 lakes studied in Spears et al (2016). Interpretation of the sediment analysis results should provide a Swan Lake specific dosage recommendation.

Table 27. Estimated treatment and monitoring costs for a Phoslock application

Item	Total Cost	Comment	Performed by
Implementation Plan	$10,000	Scenario definition and dosage	Consultant
Material (Phoslock, 35 tonnes)	$105,000	6.4 tonnes/ha over 5.48 ha ($3,000/t)	Phoslock Water Solutions Ltd
Application	$31,000	Based on 2013 cost, including inflation plus the cost of application in sheltered areas	Contractors
Enhanced Monitoring and Evaluation	$25,000	Sampling from open water and evaluation of success	Consultants
Total cost	$171,000		

6.3 Fish management

Bottom-dwelling fish like carp can severely compromise sediment capping by P adsorbing material, and obtaining more exact information about the number of such fish is recommended before applying the material. A fish community survey can be completed through electrofishing to prepare a fish community inventory. If fish number is considered large, a specialist may be consulted to determine whether fish management is possible and feasible in Swan Lake. Fish salvage can also be accomplished through electrofishing. However, the impact of fish removal may be temporary due to carp reproduction or (illegal) fish stocking.

The estimated costs for fish community survey and a fish removal campaign are about $5,000 and $20,000.

6.4 Goose and catchment management

External inputs from water fowl and runoff should be addressed by continued geese management and BMPs in the catchment basin and park around Swan Lake as presented in Section 5.1 including shoreline plantings and education. Because of the largest external load contribution of TP from geese (Table 15), its management can be expected to be the most effective one of all external load abatement.

6.4.1 Goose management approach

A goose control and monitoring program is already in place for Swan Lake. The purpose of this program is to reduce the resident waterfowl population at the Swan Lake and adjacent areas as much as possible. Activities included egg oiling, nest removal, hazing, and patrolling using specially trained dogs, as well as the use of remote-control boats and laser deterrents. It is recommended that the existing program continue with the expansion of the egg oiling activity into the adjacent areas.

Goose management requires a Canadian Wildlife Services Migratory Bird Damage Permit.

6.4.2 Runoff treatment

Shoreline runoff and the stormwater runoff that bypasses the ponds are not treated before entering Swan Lake.

Increasing the naturalization of the shoreline buffer zone using only controlled boardwalk access to the Lake is recommended for the treatment of shoreline runoff.

Treatment of the bypass runoff is not feasible given runoff volume and the physical constraints of the system. This runoff is not believed to contribute significantly to nutrient loading.

6.4.3 Investigation of historic dump site

We recommend the determination of the potential elevated P load from the historic dumpsites. A hydro-geologist familiar with the Swan Lake area could provide an estimate about water volume quantity from those sites towards Swan Lake. Potential boreholes to evaluate nutrient content, especially of TP and SRP concentrations, would help determine the P loading from this potential source.

6.4.4 Costs

Table 28 provides estimated costs for geese management, runoff treatment, as well as studies to determine the interaction of the historic dump site with the Lake.

Geese management costs are based on previous contracts, covering a range of activities to deter the birds. Shoreline naturalization would cost around $100,000. Dumpsite investigation, including the installation of five boreholes with level loggers and collection and testing of groundwater for phosphorus is estimated to cost about $20,000.

Table 28. Estimated costs for geese and catchment management

Purpose	Activity	Estimated cost
Geese management	Hazing and patrolling, egg oiling, nest removal, remote-control boat and laser	$13,500
Shoreline runoff treatment	Naturalization	$100,000
Dump site study	Boreholes and testing	$20,000

6.5 Long-term strategy

Following the initial treatment and other short-term measures, a long-term strategy will need to be developed to maintain and enhance the Lake's water quality and to achieve the long-term targets of 1.00 m Secchi depth and 0.10 mg/L of phosphorus. Once such conditions are reached, more ambitious goals can be set.

This strategy may include repeat treatments (a five-year minimum repetition is suggested, with adjusted frequency according to the occurrence of the triggers) and the continuous management of geese and fish. The development of the long-term strategy will involve the determination and evaluation of additional components, including the potential introduction of sport fishing. These goals may be achieved through the cooperation with educational institutions involved in the Water Environment Association of Ontario's Student Design Competition.

6.6 Recommended routine monitoring activities

Monitoring of Swan Lake's water quality is proposed to provide the necessary information whether action is triggered by exceedances of set targets, at a total cost of $12,000.

Site 1 can provide sufficient data for the evaluation of Swan Lake water quality as concluded in the 2017 report (Nürnberg and LaZerte, 2018). The increased number of lake visits as in 2017 and 2018 and associated data obtained by frequent monitoring by City staff would present sufficient information and data to determine exceedances. It could be further useful to extend the monitoring period to monthly sampling throughout the winter and spring. Winter conditions can inform about the general productivity of a lake, even if they may not be useful as triggers.

We recommend continuing the monitoring of Site 2 with one water sample at 0.5 m or mid-depth of the water column and visual and photographic recording. It may be reassuring in the future to compare Site 1 late summer water quality once again with Site 3 open water characteristics, but we do not think it is necessary every year.

Especially important with respect to public health is the determination of cyanotoxins concentration wherever there are any accumulations of cyanobacteria.

We provide a detailed monitoring plan in Appendix G, where we also indicate the variables that are most important in preparation of any future lake treatment as discussed in Section 5.4.

7 Conclusions

Based on the evaluation of historic water quality and the P mass balance it is obvious that internal loading has been the most important cause of the high productivity in Swan Lake before the in-lake treatment in 2013 and again since 2016. In addition, waterfowl feces have been contributing to the P budget, although persistent geese management has decreased that nutrient source in recent years. Climate change predictions for the City of Markham region include increased summer air temperature and potentially more extreme storm events. Both predictions increase the likelihood of internal P loading and, if not managed, phytoplankton productivity including cyanobacteria blooms.

Consequently, we suggest a sequence of triggers that would lead to an initiation of appropriate management actions. Our proposed triggers are:

1. The surface bloom of a potentially or proven toxic strain of cyanobacteria, confirmed by a licenced or Provincial (MECP) lab or by Abraxis strips to trigger direct attention.
2. The occurrence of two blooms within a period of four years that cover at least 25% of Swan lake area.
3. Water quality not compliant with the interim goal of growing period average 0.15 mg/L total phosphorus concentration in the surface mixed layer.
4. Water quality not compliant with the interim goal of growing period average 0.45 m Secchi disk transparency.

The proposed triggers 3 and 4 were tripped every year since 2016, and we present a review of applicable management approaches to deal with (a) external and internal TP loading, which can be judged the cause for the water quality deterioration, and (b) the overabundance of phytoplankton and the potential toxicity of cyanobacteria, which likely are the symptoms and consequences of the high TP levels. Triggers 1 and 2 cannot be evaluated for lack of information on the spread of cyanobacteria.

Because of the points described below, internal load treatment is important and we recommend a chemical treatment that is lanthanum-based (Phoslock).

1. The above-mentioned triggers 3 and 4 (Section 4) were tripped since 2016. Water quality variables showed elevated TP and chlorophyll concentration, extremely low Secchi transparency and a local cyanobacteria bloom. These conditions indicate that future water quality may not improve without intervention.
2. Water fowl was abundant in several of the post-treatment years and P will have accumulated on the sediments above the added Phoslock layer, leading to dilution of the P adsorbing lanthanum-amended clay. (In 5-15 years of recent external load the P sediment pool can be replenished, Table 20.)
3. The strong hypoxia (not specifically treated, but can decrease with improving trophic state) that still prevails in the summer, facilitates P release from P-enriched bottom sediments as seen by increased bottom TP concentration (e.g., Figure 5). Such internal loading can trigger and support blooms of cyanobacteria and is best minimized by a tested in-lake treatment.
4. Climate predictions include warmer summers that would increase sediment P release and improve growth conditions for cyanobacteria (Section 2.5).

Repeat treatments may become necessary, when the triggers are tripped in the future because of newly accumulated sediment P or due to underdosage of the chemical.

Detailed recommendations include:

- A treatment to address internal loading as sediment P release. Sediment analysis will assist with proper dosing.
- Continued water fowl management.
- Continued water quality monitoring at the two shore sites.
- Determination and potential management of the abundance of bottom dwelling fish.
- Application of best management practices to decrease the nutrient contribution from the shoreline.
- Investigation of P load from historic dumpsites.

Two treatments are the most important and promise to be most effective: continued water fowl management and repeat internal load abatement. As preferred treatment to abate the internal P loading, we chose another Phoslock application. This treatment provided at least two growing periods of improved water quality and successfully treated sediment P release according to the mass balance analysis. There was no obvious negative effect from the previous treatment. Estimated cost for this treatment is a total of $171,000. Sediment sampling and analysis need to be completed to help determine the proper dosage (at a one-time cost of $20,000). Treatment may need to be repeated in several years, depending on the successful management of the external P sources. A five-year repetition is suggested, with adjusted frequency according to the occurrence of the triggers.

An alternate treatment that includes aluminum compounds (poly aluminum chloride) may also help to reduce internal loading in the short run. Concerns of public perception and non-committal statements by MECP render it less recommendable, despite possible lower costs.

Fish management will involve a fish survey and fish salvage at a cost of about $25,000. The annual cost for waterfowl management and water quality monitoring is about $13,500 and $12,000, respectively. Treatment costs of shoreline runoff through naturalization are estimated at $100,000. Costs of investigation of phosphorus load from the historic dumpsite are around $20,000.

Following the initial treatment and other short-term measures discussed above, a long-term strategy will need to be developed to maintain and enhance Swan Lake's water quality. This strategy may include repeat treatments, continuous management of geese and fish, as well as additional measures.

8 References

Internal reports

Cosburn Patterson, Cosburn Ginnerson and Michal Michalski. 1993. The Swan Lake Community Environmental Management Study. Prepared for Swan Lake (Markham) Limited.

Toronto and Region Conservation Authority (TRCA). 2018. 2017 Surface Water Quality Summary – Regional Watershed Monitoring Program. 34 pp + appendices

WESA 2006. Risk assessment for Block 9 at Swan Lake, Markham, Ontario. For Thomas and Pelman and 1521271 Ontario Inc.

General publications and Swan Lake water quality reports

Beduhn, R.J., 1994. The Effects of Destratification Aeration on Five Minnesota Lakes. Lake Reserv. Manag. 9, 105–110. https://doi.org/10.1080/07438149409354737

Bishop, W.M., McNabb, T., Cormican, I., Willis, B.E., Hyde, S., 2014. Operational evaluation of Phoslock phosphorus locking technology in Laguna Niguel Lake, California. Water. Air. Soil Pollut. 225, 2018–2029. https://doi.org/10.1007/s11270-014-2018-6

Bryant, L.D., Gantzer, P.A., Little, J.C., 2011. Increased sediment oxygen uptake caused by oxygenation-induced hypolimnetic mixing. Water Res. 45, 3692–3703. https://doi.org/10.1016/j.watres.2011.04.018

Canadian Council of Ministers of the Environment, 2011. Canadian Water Quality Guidelines for the Protection of Aquatic Life - Chloride.

Chorus, I., Bartram, J., 1999. Toxic cyanobacteria in water - A guide to their public health consequences, monitoring and management. E & FN Spon (Routledge), London and New York.

Cooke, G.D., Welch, E.B., Peterson, S.A., Nichols, S.A., 2005. Restoration and management of lakes and reservoirs, 3nd ed. CRC, Boca Raton, FL, USA.

Copetti, D., Finsterle, K., Marziali, L., Stefani, F., Tartari, G., Douglas, G., Reitzel, K., Spears, B.M., Winfield, I.J., Crosa, G., D'Haese, P., Yasseri, S., Lürling, M., 2016. Eutrophication management in surface waters using lanthanum modified bentonite: A review. Water Res. 97, 162–174. https://doi.org/10.1016/j.watres.2015.11.056

de Vicente, I., Huang, P., Andersen, F.O., Jensen, H.S., 2008. Phosphate adsorption by fresh and aged aluminum hydroxide. Consequences for lake restoration. Env. Sci Technol 42, 6650–6655.

D'Haese, P.C., Douglas, G., Verhulst, A., Neven, E., Behets, G.J., Vervaet, B.A., Finsterle, K., Lürling, M., Spears, B., 2019. Human health risk associated with the management of phosphorus in freshwaters using lanthanum and aluminium. Chemosphere 220, 286–299. https://doi.org/10.1016/j.chemosphere.2018.12.093

EPA, U.S., 1983. Results of the Nationwide Urban Runoff Program: Volume I - Final Report (No. NTIS# PB84-185552). U.S. Environmental Protection Agency, Washington, DC, USA.

Epe, T.S., Finsterle, K., Yasseri, S., 2017. Phosphorus management with lanthanum modified bentonite in a eutrophic, shallow swimming lake in Hessen, Germany since 2007. Lake Reserv Manage 33.

Ganguli, P., Coulibaly, P., 2019. Assessment of future changes in intensity-duration-frequency curves for Southern Ontario using North American (NA)-CORDEX models with nonstationary methods. J. Hydrol. Reg. Stud. 22, 100587. https://doi.org/10.1016/j.ejrh.2018.12.007

Gensemer, R.W., Playle, R.C., 1999. The bioavailability and toxicity of aluminum in aquatic environments. CRC Crit. Rev. Environ. Sci. Technol. 29, 315–450.

Hall, K.J., Murphy, T.P.D., Mawhinney, M., Ashley, K.I., 1994. Iron treatment for eutrophication control in Black Lake, British Columbia. NALMS Lake Reserv. Manag.

Health Canada, 2012. Guidelines for Canadian Recreational Water Quality, Third Edition. Water, Air and Climate Change Bureau, Healthy Environments and Consumer Safety Branch, Ottawa, Ontario.

Holm, E.R., Stamper, D.M., Brizzolara, R.A., Barnes, L., Deamer, N., Burkholder, J.M., 2008. Sonication of bacteria, phytoplankton and zooplankton: application to treatment of ballast water. Mar. Pollut. Bull. 56, 1201–1208.

Huser, B.J., Bajer, P.G., Chizinski, C.J., Sorensen, P.W., 2016a. Effects of common carp (*Cyprinus carpio*) on sediment mixing depth and mobile phosphorus mass in the active sediment layer of a shallow lake. Hydrobiologia 763, 23–33. https://doi.org/10.1007/s10750-015-2356-4

Huser, B.J., Egemose, S., Harper, H., Hupfer, M., Jensen, H., Pilgrim, K.M., Reitzel, K., Rydin, E., Futter, M., 2016b. Longevity and effectiveness of aluminum addition to reduce sediment phosphorus release and restore lake water quality. Water Res. 97, 122–132. https://doi.org/10.1016/j.watres.2015.06.051

James, W.F., 2017. Phosphorus binding dynamics in the aluminum floc layer of Half Moon Lake, Wisconsin: Lake and. Lake Reserv. Manag. 33, 130–142. https://doi.org/doi:10.1080/10402381.2017.1287789

James, W.F., Bischoff, J.M., 2020. Sediment aluminum:phosphorus binding ratios and internal phosphorus loading characteristics 12 years after aluminum sulfate application to Lake McCarrons, Minnesota. Lake Reserv. Manag. 36, 1–13. https://doi.org/10.1080/10402381.2019.1661554

Jančula, D., Maršálek, B., 2011. Critical review of actually available chemical compounds for prevention and management of cyanobacterial blooms. Chemosphere 85, 1415–1422. https://doi.org/10.1016/j.chemosphere.2011.08.036

Koschel, R.H., Dittrich, M., Casper, P., Heiser, A., Rossberg, R., 2000. Induced hypolimnetic calcite precipitation - ecotechnology for restoration of stratified eutrophic hardwater lakes. Verh Intern. Ver. Limnol 27.

LaZerte, B.D., Nürnberg, G.N., 2000. Suburban development: Review of pollutants in surface runoff and their remediation (For Michalski Nielsen Associates). Freshwater Research.

Liboriussen, L., Søndergaard, M., Jeppesen, E., Thorsgaard, I., Grünfeld, S., Jakobsen, T., Hansen, K., 2009. Effects of hypolimnetic oxygenation on water quality: results from five Danish lakes. Hydrobiologia 625, 157–172.

Lürling, M., Meng, D., Faassen, E.J., 2014. Effects of Hydrogen Peroxide and Ultrasound on Biomass Reduction and Toxin Release in the Cyanobacterium, Microcystis aeruginosa. Toxins 6, 3260–3280. https://doi.org/10.3390/toxins6123260

Lürling, M., Tolman, Y., 2014. Beating the blues: Is there any music in fighting cyanobacteria with ultrasound? Water Res. 66, 361–373. https://doi.org/10.1016/j.watres.2014.08.043

Lürling, M., Tolman, Y., 2010. Effects of lanthanum and lanthanum-modified clay on growth, survival and reproduction of Daphnia magna. Water Res 44, 309.

Lürling, M., Waajen, G., Domis, L.N. de S., 2016. Evaluation of several end-of-pipe measures proposed to control cyanobacteria. Aquat. Ecol. 50, 499–519. https://doi.org/10.1007/s10452-015-9563-y

Matthijs, H.C.P., Jančula, D., Visser, P.M., Maršálek, B., 2016. Existing and emerging cyanocidal compounds: new perspectives for cyanobacterial bloom mitigation. Aquat. Ecol. 50, 443–460. https://doi.org/10.1007/s10452-016-9577-0

Meis, S., Spears, B.M., Maberly, S.C., O'Malley, M.B., Perkins, R.G., 2012. Sediment amendment with Phoslock® in Clatto Reservoir (Dundee, UK): Investigating changes in sediment elemental composition and phosphorus fractionation. J Env. Manag 93, 185–193.

Meis, S., Spears, B.M., Maberly, S.C., Perkins, R.G., 2013. Assessing the mode of action of Phoslock® in the control of phosphorus release from the bed sediments in a shallow lake (Loch Flemington, UK). Water Res 47, 4460–4473.

Ministry of Environment, 2010. Lakeshore Capacity Assessment Handbook: Protecting Water Quality in Inland Lakes on Ontario's Precambrian Shield. www.ontario.ca/environment, Toronto, Ontario.

Moore, B.C., Cross, B.K., Beutel, M., Dent, S., Preece, E., Swanson, M., 2012. Newman Lake restoration: A case study Part III. Hypolimnetic oxygenation. Lake Reserv Manage 28, 311–327. https://doi.org/10.1080/07438141.2012.738463

Moore, M.V., Zakova, P., Shaeffer, K.A., Burton, R.P., 1998. Potential effects of Canada Geese and climate change on phosphorus inputs to suburban lakes of the Northeastern U.S.A. Lake Reserv Manage 14, 52–59.

Mucci, M., Maliaka, V., Noyma, N.P., Marinho, M.M., Lürling, M., 2018. Assessment of possible solid-phase phosphate sorbents to mitigate eutrophication: Influence of pH and anoxia. Sci. Total Environ. 619, 1431–1440.

Muir, R.J., 2018. Evidence Based Policy Gaps in Water Resources: Thinking Fast and Slow on Floods and Flow. J. Water Manag. Model. https://doi.org/10.14796/JWMM.C449

Nürnberg, G.K., 2018. Interpretation of phytoplankton samples taken in Swan Lake on Sep 28, 2018 with respect to harmful species., Memo for the The City of Markham, Ontario. Freshwater Research, Baysville, Ontario.

Nürnberg, G.K., 2017. Attempted management of cyanobacteria by Phoslock (lanthanum-modified clay) in Canadian lakes, water quality results and predictions. Lake Reserv. Manag. 33, 163–170. https://doi.org/10.1080/10402381.2016.1265618

Nürnberg, G.K., 2007. Lake responses to long-term hypolimnetic withdrawal treatments. Lake Reserv. Manag. 23, 388–409.

Nürnberg, G.K., 1996. Trophic state of clear and colored, soft- and hardwater lakes with special consideration of nutrients, anoxia, phytoplankton and fish. Lake Reserv. Manag. 12, 432–447.

Nürnberg, G.K., 1995. Quantifying anoxia in lakes. Limnol. Oceanogr. 40, 1100–1111.

Nürnberg, G.K., 1984. The prediction of internal phosphorus load in lakes with anoxic hypolimnia. Limnol. Oceanogr. 29, 111–124. https://doi.org/10.4319/lo.1984.29.1.0111

Nürnberg, G.K., LaZerte, B.D., 2018. Swan Lake water quality in 2017, Report for the The City of Markham, Ontario. Freshwater Research, Baysville, Ontario.

Nürnberg, G.K., LaZerte, B.D., 2016. Trophic state decrease after lanthanum-modified bentonite (Phoslock) application to a hyper-eutrophic polymictic urban lake frequented by Canada geese (*Branta canadensis*). Lake Reserv. Manag. 32, 74–88. https://doi.org/10.1080/10402381.2015.1133739

Nürnberg, G.K., LaZerte, B.D., 2015. Swan Lake water quality in the 2nd post-treatment year (2014) after the Phoslock treatment (29 April – 1 May 2013), Report for the The City of Markham, Ontario. Freshwater Research, Baysville, Ontario.

Nürnberg, G.K., LaZerte, B.D., 2014. 2013 water quality after the Phoslock treatment in Swan Lake on 29 April – 1 May 2013, Report for the The City of Markham, Ontario. Freshwater Research, Baysville, Ontario.

Nürnberg, G.K., LaZerte, B.D., 2012. Water Quality and Remediation Options for Swan Lake, Report for the The City of Markham, Ontario. Freshwater Research, Baysville, Ontario.

Nürnberg, G.K., LaZerte, B.D., Olding, D.D., 2003. An artificially induced *Planktothrix rubescens* surface bloom in a small kettle lake in southern Ontario compared to blooms world-wide. Lake Reserv. Manag. 19, 307–322.

Ontario Ministry of Environment, 2009. Phoslock toxicity testing with three sediment dwelling organisms (*Hyalella azteca*, *Hexagenia spp.* and *Chironomus dilutus*) and two water column dwelling organisms (Rainbow Trout and *Daphnia magna*), Technical Memorandum. Ontario Ministry of the Environment, Toronto.

Osgood, R.A., 2017. Inadequacy of best management practices for restoring eutrophic lakes in the United States: guidance for policy and practice. Inland Waters 7, 401–407.

Park, J., Church, J., Son, Y., Kim, K.-T., Lee, W.H., 2017. Recent advances in ultrasonic treatment: Challenges and field applications for controlling harmful algal blooms (HABs). Ultrason. Sonochem. 38, 326–334. https://doi.org/10.1016/j.ultsonch.2017.03.003

Prepas, E.E., Babin, J., Murphy, T.P., Chambers, P.A., Sandland, G., Ghadouani, A., Serediak, M., 2001. Long-term effects of successsive Ca(OH)2 and CaCO3 treatments on the water quality of two eutrophic hardwater lakes. Freshw Biol 46, 1089–1103.

Rajasekhar, P., Fan, L., Nguyen, T., Roddick, F.A., 2012. A review of the use of sonication to control cyanobacterial blooms. Water Res. 46, 4319–4329. https://doi.org/10.1016/j.watres.2012.05.054

Reitzel, K., Jensen, H.S., Egemose, S., 2013. pH dependent dissolution of sediment aluminum in six Danish lakes treated with aluminum. Water Res. 47, 1409–1420. https://doi.org/10.1016/j.watres.2012.12.004

Spears, B.M., Lürling, M., Yasseri, S., Castro-Castellon, A.T., Gibbs, M., Meis, S., McDonald, C., McIntosh, J., Sleep, D., Van Oosterhout, F., 2013. Lake responses following lanthanum-modified bentonite clay (Phoslock®) application: An analysis of water column lanthanum data from 16 case study lakes. Water Res 47, 5930–5942.

Spoof, L., Jaakkola, S., Važić, T., Häggqvist, K., Kirkkala, Terhi, Ventelä, A.-M., Kirkkala, Teija, Svirčev, Z., Meriluoto, J., 2020. Elimination of cyanobacteria and microcystins in irrigation water—effects of hydrogen peroxide treatment. Environ. Sci. Pollut. Res. https://doi.org/10.1007/s11356-019-07476-x

Unckless, R.L., Makarewicz, J.C., 2007. The impact of nutrient loading from Canada Geese (Branta canadensis) on water quality, a mesocosm approach. Hydrobiologia 586, 393–401.

Visser, P.M., Ibelings, B.W., Bormans, M., Huisman, J., 2015. Artificial mixing to control cyanobacterial blooms: a review. Aquat. Ecol. https://doi.org/10.1007/s10452-015-9537-0

Waajen, G., van Oosterhout, F., Douglas, G., Lürling, M., 2016. Management of eutrophication in Lake De Kuil (The Netherlands) using combined flocculant – Lanthanum modified bentonite treatment. Water Res. 97, 83–95. https://doi.org/10.1016/j.watres.2015.11.034

Weenink, E.F., Luimstra, V.M., Schuurmans, J.M., Van Herk, M.J., Visser, P.M., Matthijs, H.C.P., 2015. Combatting cyanobacteria with hydrogen peroxide: a laboratory study on the consequences for phytoplankton community and diversity. Front. Microbiol. 6. https://doi.org/10.3389/fmicb.2015.00714

Winter, J.G., DeSellas, A.M., Fletcher, R., Heintsch, L., Morley, A., Nakamoto, L., Utsumi, K., 2011. Algal blooms in Ontario, Canada: Increases in reports since 1994. Lake Reserv Manage 27, 107–114.

Wiśniewska, K., Lewandowska, A.U., Śliwińska-Wilczewska, S., 2019. The importance of cyanobacteria and microalgae present in aerosols to human health and the environment – Review study. Environ. Int. 131, 104964. https://doi.org/10.1016/j.envint.2019.104964

Yasseri, S., Epe, T.S., 2016. Analysis of the La:P ratio in lake sediments–vertical and spatial distribution assessed by a multiple-core survey. Water Res. 97, 96–100. https://doi.org/10.1016/j.watres.2015.07.037

Appendix A: Observed (AF) and modelled (AA) anoxia by the anoxic factor

Extent and duration of anoxia and hypoxia can be quantified by anoxic factors for specific DO thresholds which summarize information available by DO profiles and hypsographic information into one number per season. This factor represents the number of days in a season or year that a sediment area equal to the lake surface area is anoxic.

The anoxic factor was computed from DO profiles according to the Equation 1 (Nürnberg, 1995):

Equation 1

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

where t_i, the period of anoxia (days), a_i, the corresponding area (m^2) (provided by City staff), A_o, lake surface area (m^2), and n, numbers of periods with different oxycline depths. (A threshold of 2.9 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.)

In shallow Swan Lake observed DO profiles underestimates the anoxic sediment area, because of the intermittent mixing and aeration of the water layers. In such mixed conditions a large sediment surface area is often anoxic and actively releasing phosphorus, especially when sediment and water are nutrient-rich. Nürnberg (2009) found that the areal extent and time for such active sediment of a polymictic lake can be predicted from a model originally developed for stratified lakes as anoxic area factor (AA, Equation 2, in units of days per summer).

Equation 2 $$AA = -36.2+50.1 \log (TP_{summer}) + 0.762\ z/A_o^{0.5}$$

where, TP_{summer}, growing period ("summer") average composite TP concentration (µg/L)

$z/A_o^{0.5}$, morphometric ratio (m/km)

z, mean depth (m)

This value can be visualized as if the whole lake area releases TP for *AA* days per growing period.

Appendix B: 2018 Comparison of Secchi disk transparency with chlorophyll concentration in Swan Lake

Comparisons between Secchi measurements and chlorophyll *a* (Chl) concentrations revealed a significant relationship between the two variables expressed as Equation 3 developed on Swan Lake data (Figure 21 and Figure 22; Nürnberg and LaZerte, 2017). This relationship can be used to predict (model) Chl concentration. E.g., using Equation 3, the growing period average Secchi in 2017 of 0.40 m (Site 1) predicts a Chl average of 61 µg/L.

Equation 3 log Chlorophyll = 1.231 (±0.061) – 1.399 (±0.128) log Secchi, n=13, R^2= 0.91

Figure 21. Secchi disk depth versus chlorophyll concentration for 2013, 2014, 2016 and pre-treatment year 2011. Lines present power curve and logarithmic relationships.

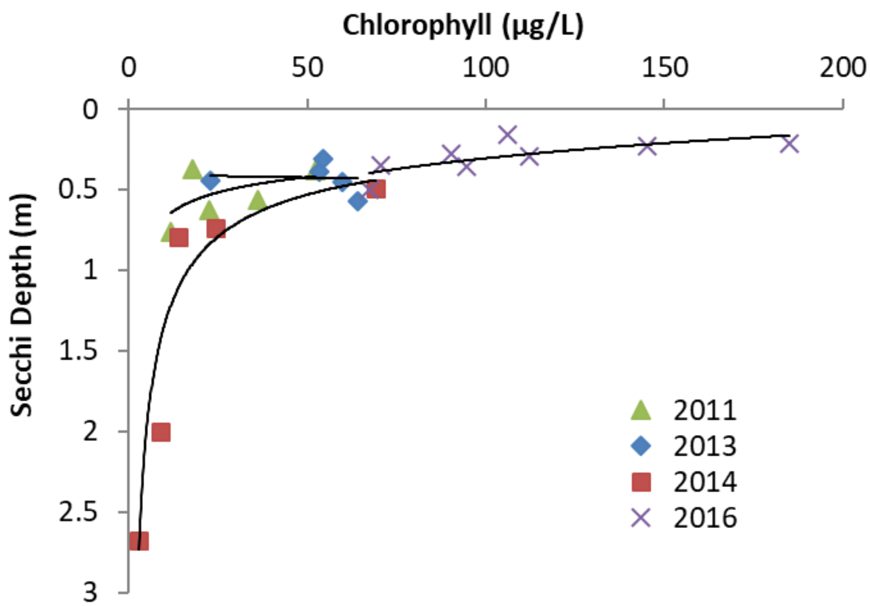
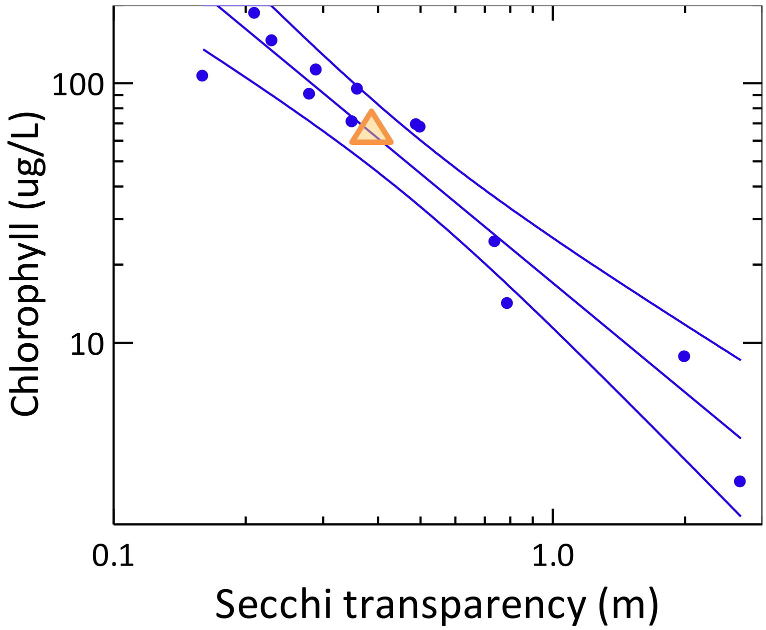


Figure 22. Secchi disk depth versus chlorophyll concentration for the post-treatment years 2014 and 2016. The regression line and 95% confidence limits are shown for, Equation 3 (n=13, R^2= 0.91). As example, the 2017 growing period predicted Chl average for Site 1 is indicated as triangle. (Redrawn from Nürnberg and LaZerte 2017).



Appendix C: Hydrological modelling, water budget *(City of Markham)*

------- *Next page* -----

M E M O R A N D U M

To: File
Prepared by: Zahra Parhizgari, Environmental Engineer
Kate Rothwell, Environmental Engineer
Date: July 2020

Re: Swan Lake Water Quality Management -

Introduction

This memorandum has been prepared by the City of Markham to describe the water balance model developed in support of the Swan Lake Water Quality Improvement project underway by Freshwater Research.

Swan Lake is a 5.5 ha waterbody located northeast of Sixteenth Avenue and Williamson Road in the City of Markham. Water enters Swan Lake through direct precipitation on the Lake, as well as controlled or uncontrolled stormwater runoff from the surrounding subdivisions. Outflows from the Lake include evaporation and flow through an outlet pipe which discharges to the southeast of the Lake towards Sixteenth Avenue. The difference between inflow and outflow manifests as change in Lake volume.

The water balance for Swan Lake was established for the period of 2009 to 2018. Different components of this water balance analysis are described below.

Meteorology

Meteorological parameters are the most frequently measured and affect several components of the water balance analysis, as described in this section.

Precipitation and Temperature

Precipitation data in 5-min intervals are available from the Markham Museum meteorological station, complemented with data from the Mount Joy Community Center station. Daily minimum, average, and maximum temperature are available from the Buttonville Airport station.

Figure 23 provides the total monthly precipitation data. During this period, annual precipitation ranged from 420 mm (in 2015) to 930 mm (in 2018). The 2015 data may be incomplete, and the next year with the lowest precipitation was 2011 at 530 mm.

Figure 24 provides average, minimum, and maximum daily temperature data for the Buttonville Airport station. During the study period, the daily air temperature has reached a high of 37 °C and a low of –27 °C. The annual averages are between 6.9 °C and 9.8 °C, whereas the long-term average is 7.7°C. Analysis of trends in temperature over the last 20 years does not show a decreasing or increasing trend.

Evapotranspiration

Evapotranspiration (ET) rates are not measured in any station close to the study area, and were therefore calculated using climate data from the nearest stations. Various empirical or process-based models (e.g., through energy budget, temperature equilibrium, or combination equations) exist for the calculation of evaporation. Based on data availability and the required resolution, ET was estimated using the Priestley-Taylor model (Priestley & Taylor, 1972).

Figure 23. Monthly Precipitation at Markham Museum or Mount Joy

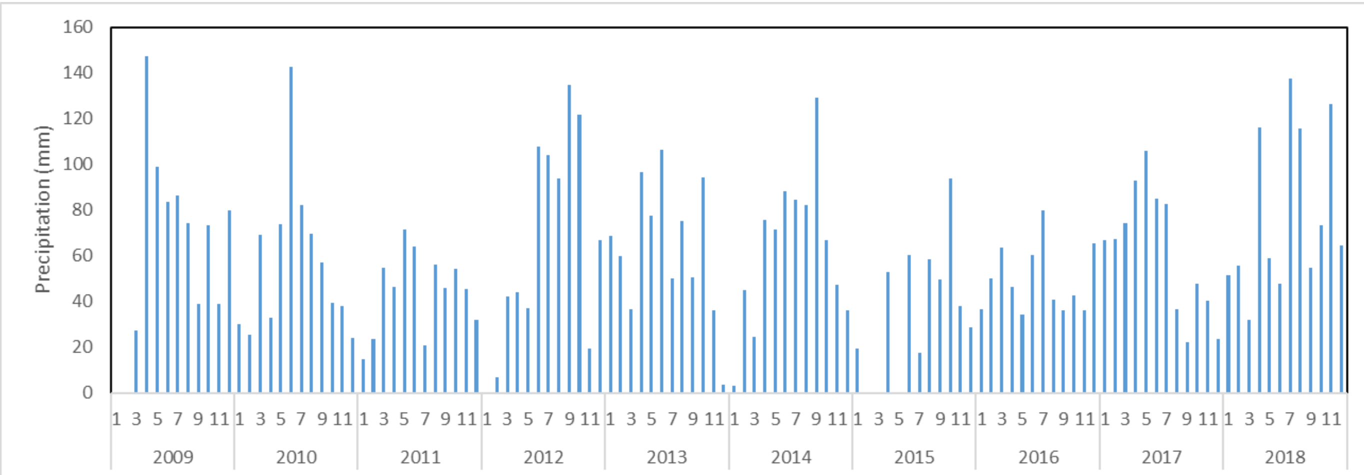
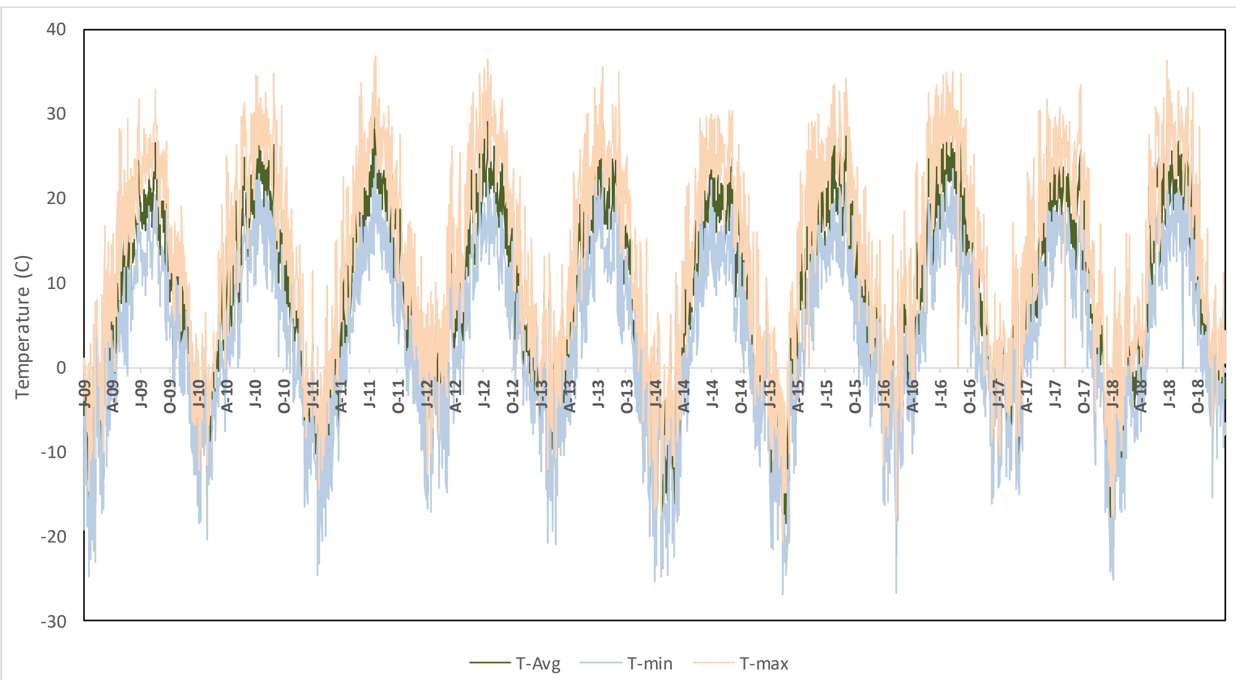


Figure 24. Daily Temperature at Buttonville Airport Station



Surface Runoff

Drainage Plan Design

The Environmental Master Drainage Plan for the Swan Lake Community (Cosburn Ltd. et al., 1995) analyzed the site outlet capacities, major/minor drainage systems, as well as quantity and quality control requirements. The study area encompassing the catchments draining to Swan Lake (as well as areas to the west, east, and north not related to Swan Lake) would drain predominantly to either Swan Lake, a storm sewer inlet on 16th Ave. and from the west drainage area to the Exhibition Creek (aka Mount Joy Creek).

As per the 1995 Master Plan prepared by Cosburn Ltd., the water captured by Swan Lake would either be infiltrated or evaporated. A restricted outfall was also recommended for the Lake discharging to the 16th Ave. storm sewer to maintain a constant water level and positive drainage of the Lake. The Lake release rate was limited to 100 L/s to accommodate the downstream drainage constraint at the 16th Ave. storm sewer (2-year peak flow of 1.166 m3/s).

Two (2) wet extended detention ponds (#104- North Pond and #105-East Pond) were proposed to accommodate the proposed site layout and required grading, and providing quality control. These ponds were designed to attenuate the stormwater runoff generated by a 2-hr duration, 25 mm runoff. Each pond would have a sediment forebay at each storm sewer inlet (to concentrate larger sediment particles), and a shallow area with emergent aquatic planting at the pond outfall (to provide natural habitat and nutrient uptake). The North and East Pond would have one and two inlets, respectively. A flow splitter immediately upstream of each pond outfall would direct the 25 mm storm runoff to the pond. Weirs within the flow splitters would direct the storm flows in addition to the pond volume to the Lake for flood control attenuation.

The normal water level elevation within the ponds and the Lake were to be coincident with the adjacent groundwater elevation. All stormwater inflows to the ponds would outlet to the foundation drain collection (FDC) system, which ultimately drained to the storm sewer south of 16th Ave.

Stormwater Ponds

The North Pond (ID #104) was constructed as an extended detention wet pond in xx to collect up to 25 mm storm event runoff from an area of 11.6 ha conveyed to the Pond by a 1050 mm diameter concrete storm sewer. The Pond has a permanent storage volume of 1558 m3 and a total extended detention storage volume of 810 m3. It has an inlet structure, a 28 m long sediment forebay, an access road, and a maintenance road. The outlet structure includes a reversed sloped pipe to a manhole with a 100 mm diameter orifice allowing discharge of 430 L/s (25 mm storm event) to a storm sewer on Williamson Road. Release is via a 200 mm diameter storm sewer, including a diversion structure located upstream of the Pond, a 2400 mm diameter manhole and a weir allowing a discharge of 1,053 L/s (5-year storm event) to Swan Lake and an emergency spillway located at the west end of the Pond discharging to Swan Lake (MECP 2001).

The East Pond (ID #105) was constructed as an extended detention wet pond in xx to service a total drainage area of 21 ha. The Pond has a permanent storage of 1750 m3 at an elevation of 208.2 m and an active storage volume of 1100 m3 at an elevation of 208.7 m with a minimum detention time of 24 hrs. The Pond outflow is controlled by a submerged perforated pipe and a precast twin catchbasin. The twin catchbasin allows for additional outflow once the active storage depth exceeds an elevation of 208.45 m. The pond release rate is controlled by a 66 mm diameter orifice plate located at the outlet. The south and north inlet structures divert flows to Swan Lake once the water surface elevation in the extended detention pond exceeds 208.7 m. Once the Swan Lake water level exceeds an elevation of 208.2 m, excess storage is released through the FDC system. A 190 mm orifice located at the outlet side of the Lake headwall adjacent to the East Pond controls the outflow from Swan Lake to the FDC system. Flows from the Pond as well as Swan Lake are conveyed to a storm sewer south of 16th Ave (MECP 1995.).

Hydrologic Modelling

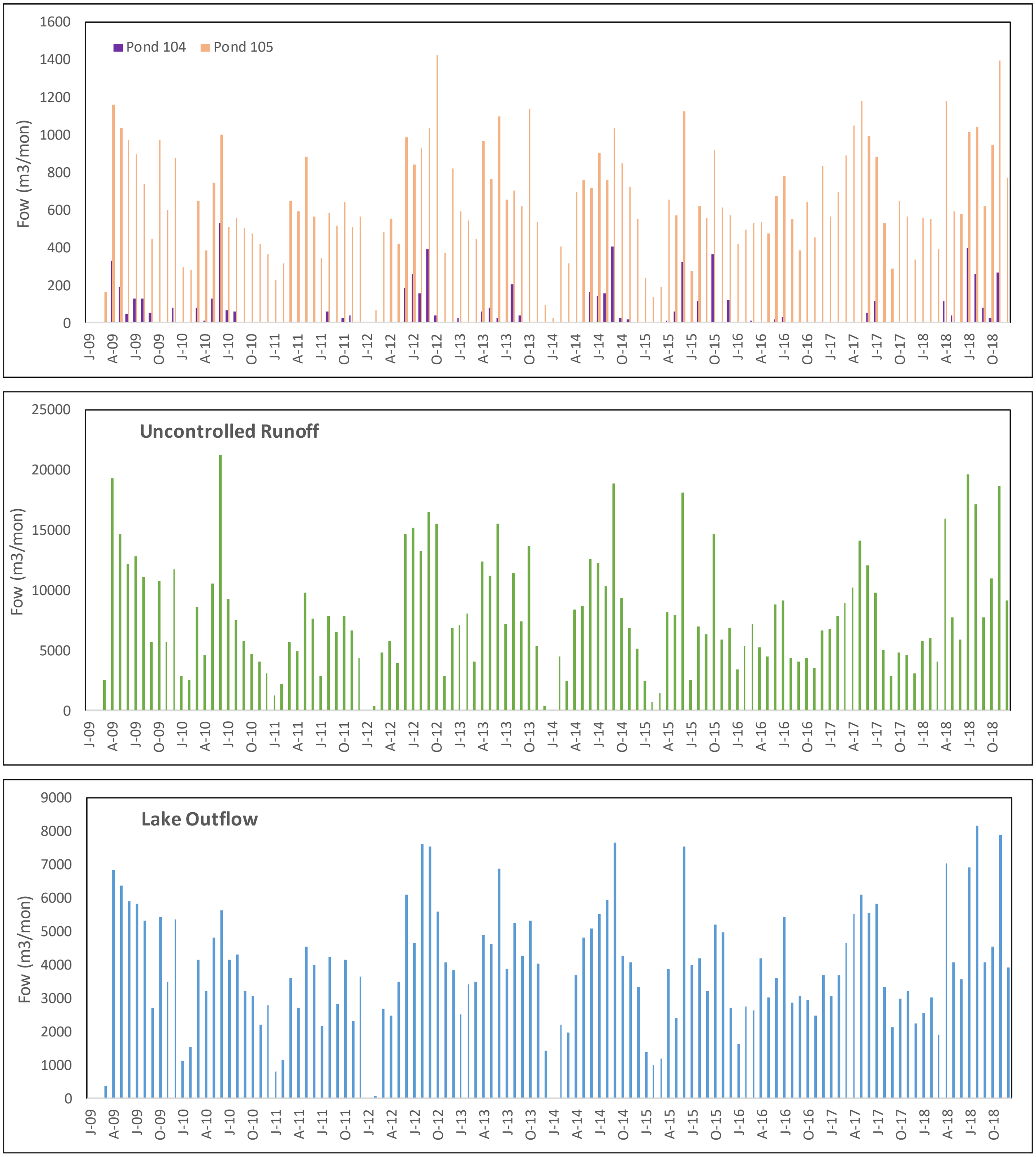
Recent LiDAR data and as-built information on the existing storm sewer network were used to delineate the catchments being conveyed to the Lake and adjacent stormwater pond facilities. As per the catchment delineation performed, the drainage area to the Lake is about 38.7 ha. Of this area, two (2) stormwater management ponds serve about 75% (29.1 ha) of the Lake's catchment area. Runoff from the remainder of the area (about 9.6 ha), which is comprised of overland flow from the immediate shoreline, was considered uncontrolled. In this modelling exercise, runoff from the last phase of development (5.6 ha around properties on Oasis Way) was also regarded as uncontrolled as the servicing layer for this area was not available at the time of model development. For this study, this assumption is considered conservative. The catchment area includes about 50% pervious land cover (landscape) and 50% impervious land cover (rooftop, driveways, parking lots and roads, and amenities).

Surface runoff from all outfalls was modelled for the 2009 to 2018 period using the PCSWMM model a. Both minor and major systems were modelled continuously on a 5-min time step.

Both SWM ponds which discharge to Swan Lake were initially designed with permanent pools for water quality control and extended detention volumes for the 25-mm event. As per the model results, these ponds provide sufficient storage for the 25-mm event and prevent the release of nutrients during the first-flush runoff to the Lake.

Monthly flows from major outfalls modelled from 2009 to 2018 are provided in Figure 25.

Figure 25. Modelled Monthly Flow from 2009 to 2018



Groundwater Exchange

Available information was reviewed to develop an estimate of groundwater exchange with the Lake and surrounding area.

As per the Environmental Master Plan, the regional groundwater flow in the study area is directed southwards or southwestwards, with an average hydraulic gradient of about 1%. The regional groundwater table is in hydraulic continuity with the static water level in Swan Lake with the static elevation ranging between 207 and 209 m.

Several geotechnical reports have been prepared over the years in support of development applications in the area. These studies have involved the installation of boreholes and the measurement of groundwater level. Another source of information is the work completed to remediate methane gas emanation from the former landfill site on the west.

The Phase II Conceptual Site Model prepared for Block 9 on the east side of the Lake includes measurements of groundwater level and flow direction. This report indicates groundwater flows towards the south at an estimated velocity of 2.9 m/year under the property. The groundwater level was an average depth of 3.2 m (exp. 2015a).

The Phase II Environmental Site Assessment (ESA)for 6330 16th Ave., located on the south of the Lake, concluded that the groundwater flows from to the southwest. In this study, the surface water elevations in San Lake were higher than the corresponding groundwater elevation, indicating there was a net flow of water from the Lake to the water table unit. Groundwater velocity was estimated at 74 m/yr (exp. 2015b).

Subsequent to reports of methane gas generation within the west part of the Lake, the City requested that the landowner remove all fill material from the site, and install a methane gas ventilation system to reduce the level below acceptable MECP levels. The system was installed in 1998 and operated for approximately five years. As part of this undertaking, the groundwater table was measured and found to range from 1.4 to 3.5 m below grade (AMEC 2014).

Groundwater flows in and out of Lake were estimated using the above information to range between 10 and 300 m^3/day each.

Change in Volume

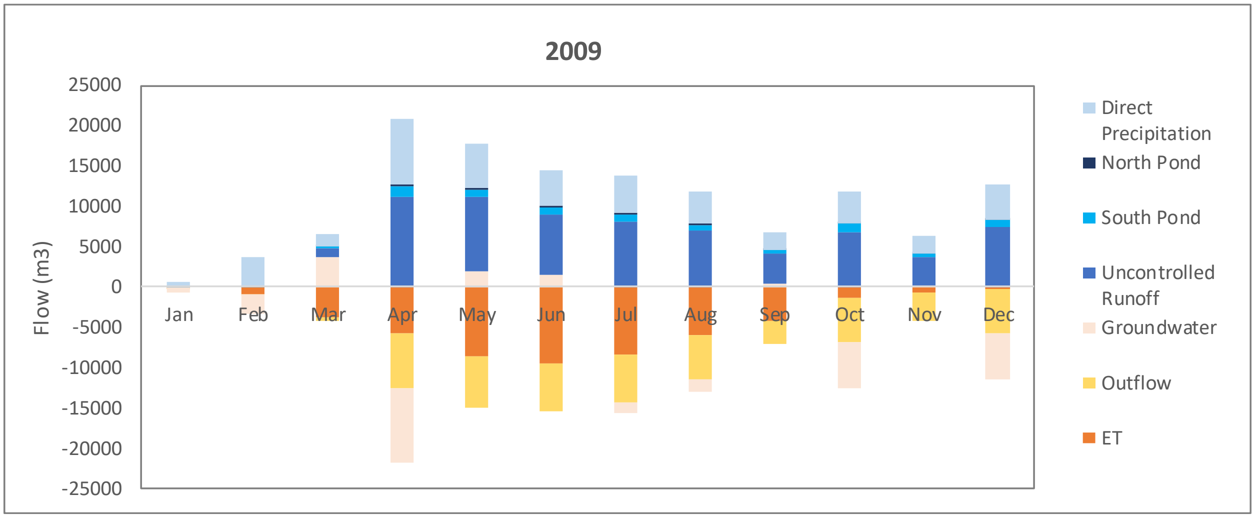
In 2017, a water level logger was installed next to the dock on the southern shore of the Lake. The logger has been recording the water level at that location continuously; however, the record is not long enough to be used in this water balance analysis.

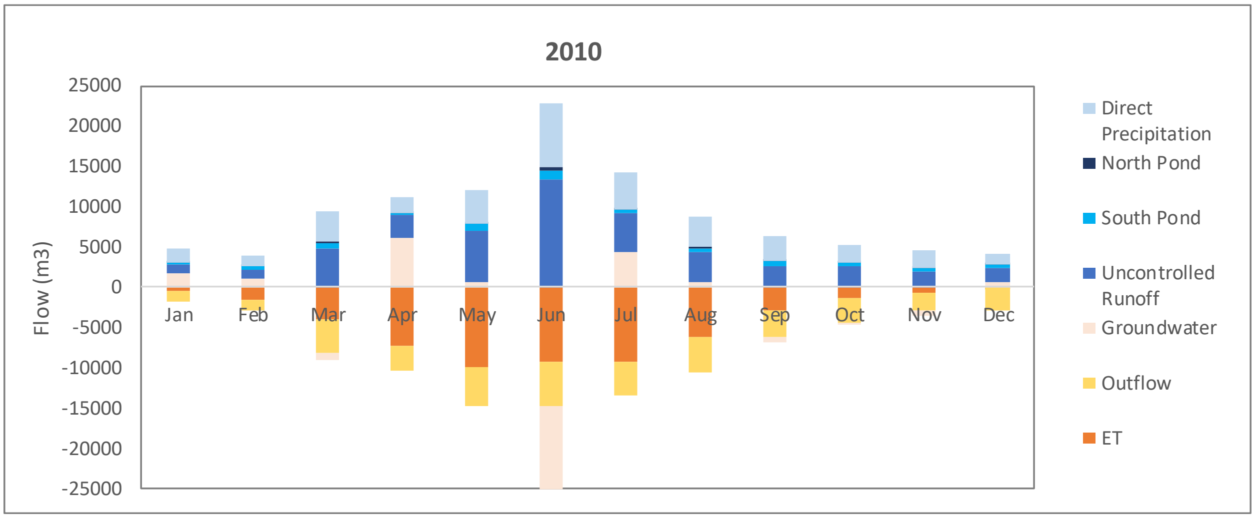
Change in the Lake volume was estimated considering precipitation and temperature (ET) records, given that all inflows are directly linked to precipitation and that the Lake does not have an outlet control device.

Water Balance Results

Monthly water balance results for the period of 2008 to 2018 are provided in Figure 26 below, showing direct precipitation on and ET from the Lake surface, uncontrolled runoff and SWM ponds and Lake outflows modelled using PCSWMM, and groundwater exchange estimated as described previously.

Figure 26. Monthly Water Balance Results





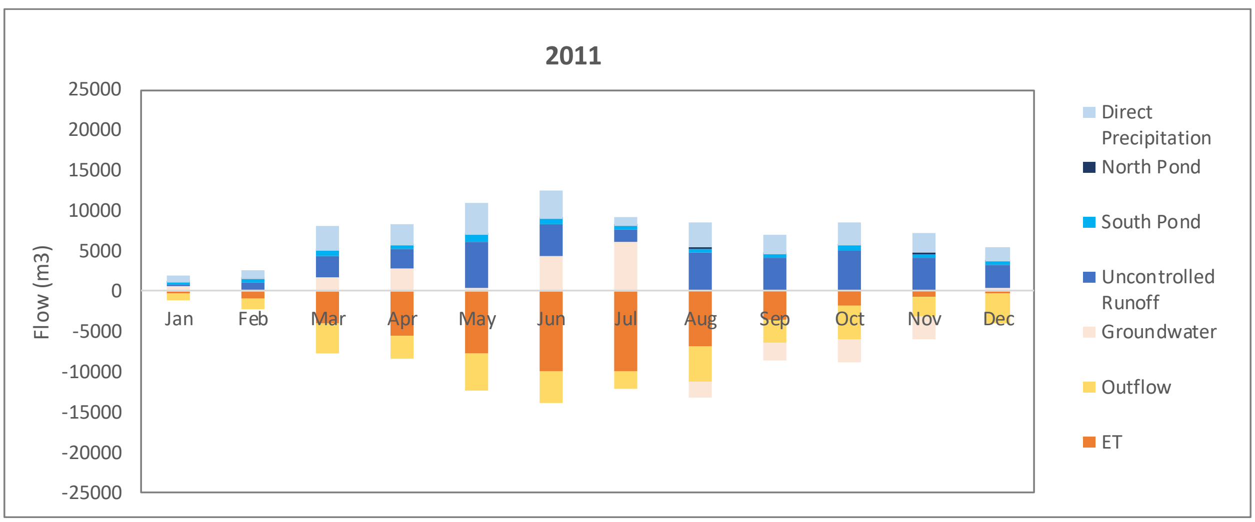
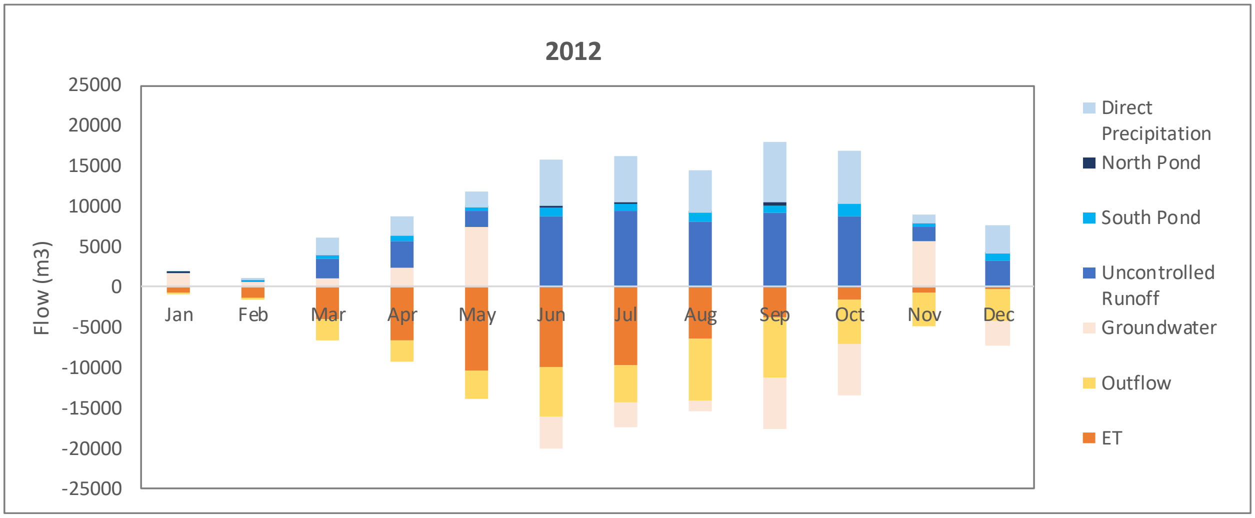
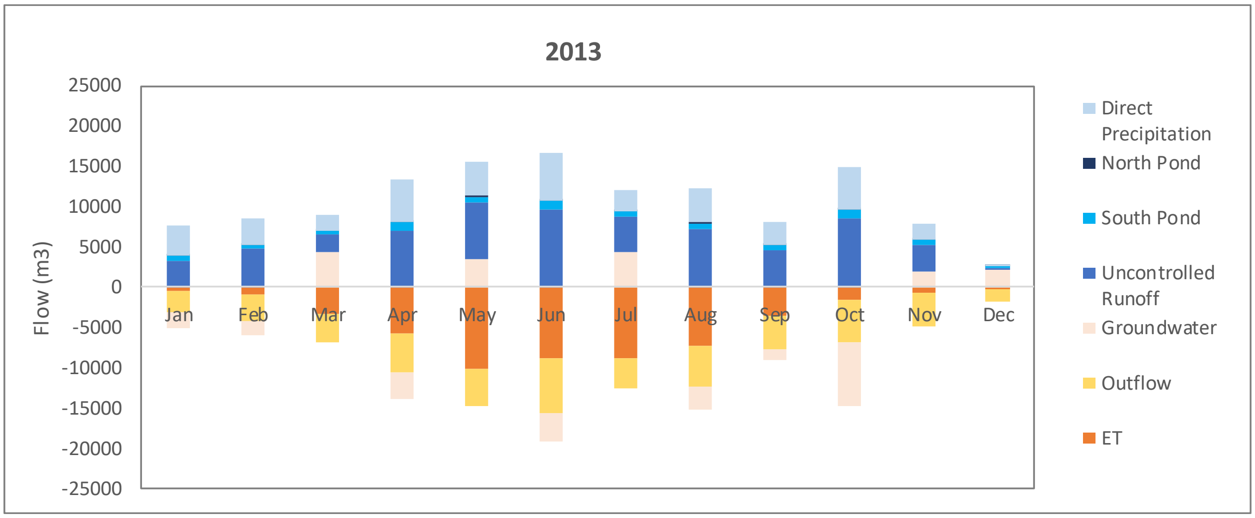


Figure 26. Monthly Water Balance Results, Cont'd





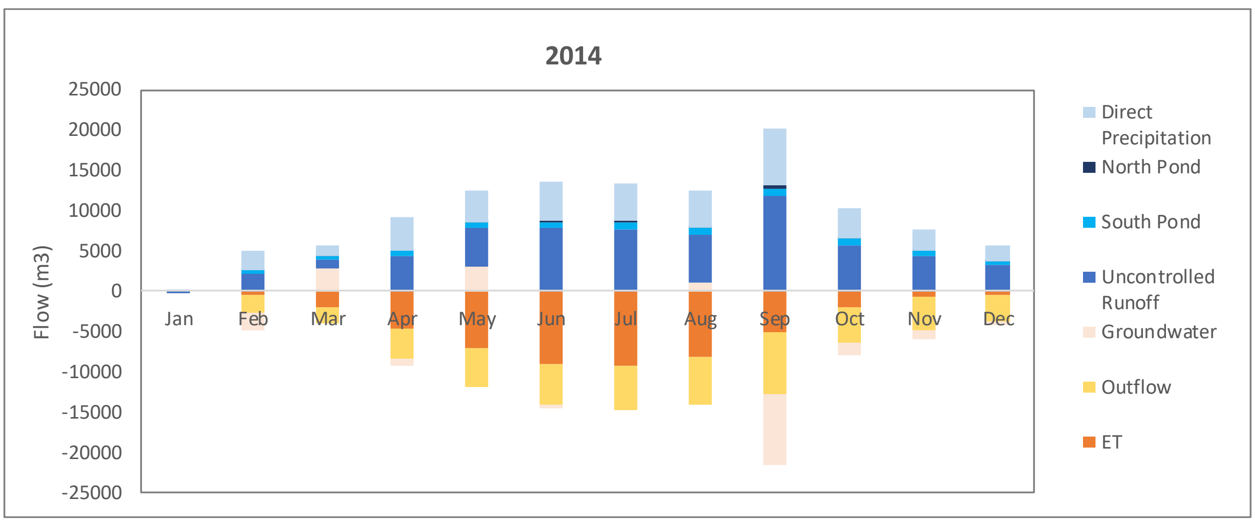
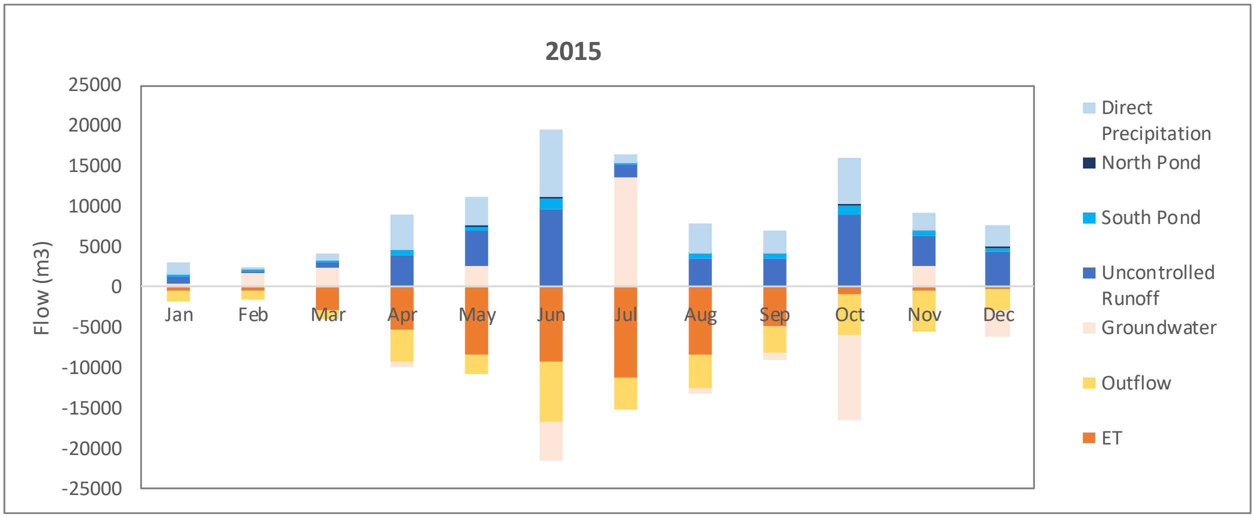
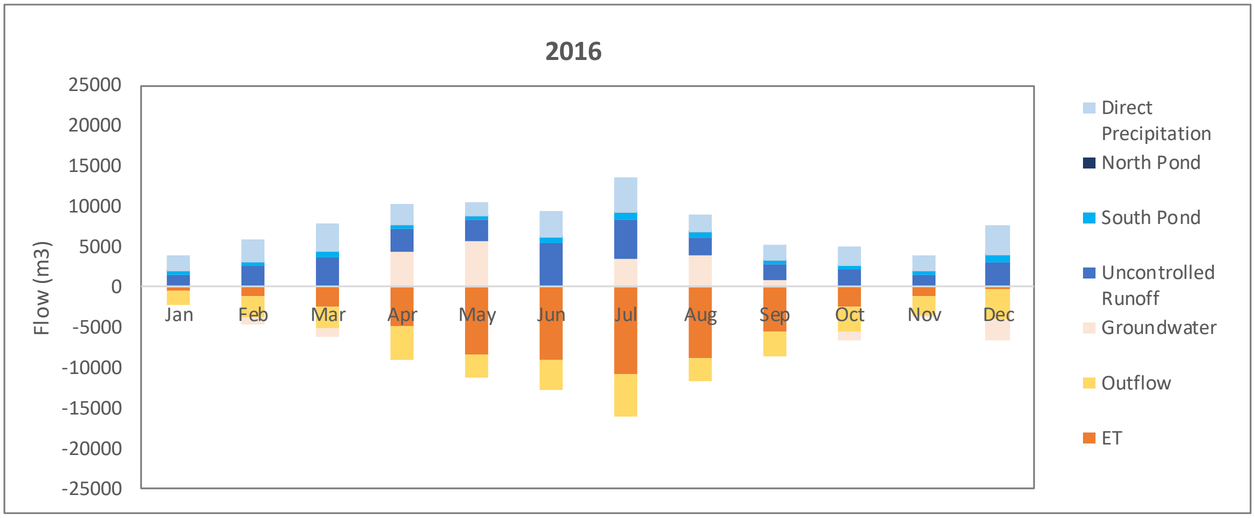


Figure 26. Monthly Water Balance Results, Cont'd





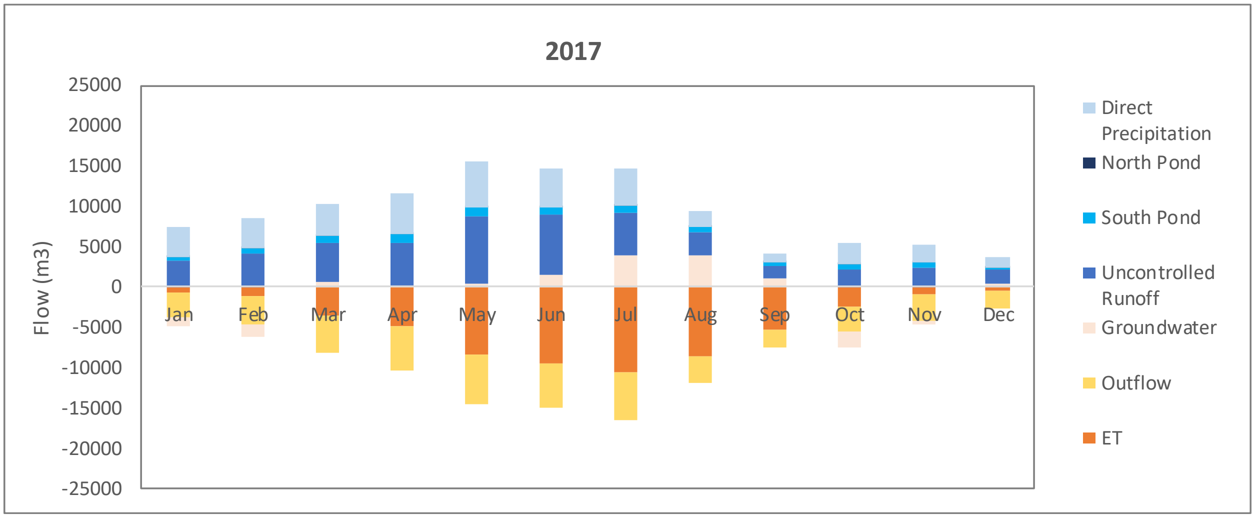
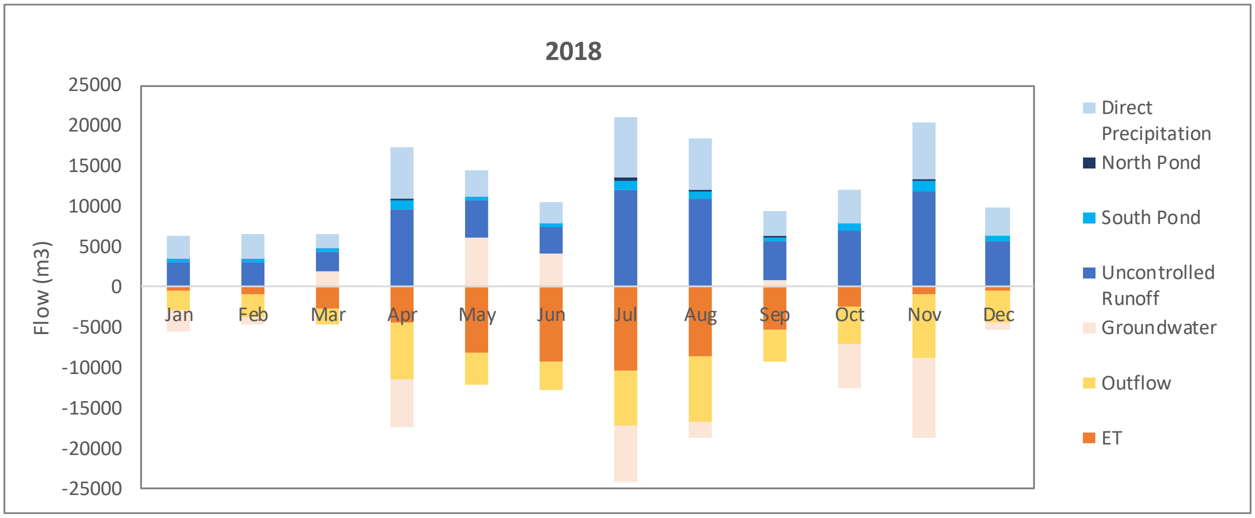


Figure 26. Monthly Water Balance Results, Cont'd



References

AMEC Environment & Infrastructure. 2014. 2012 - 2013 Landfill Gas Monitoring Report, Part 15, Swan Lake. City of Markham.

Cosburn Patterson Wardman Limited, Cosburn Giberson Landscape Architects and Michael Michalski Associates. 1995. Environmental Master Drainage Plan OPA 17 (Swan Lake Secondary Plan).

Exp. 2015a. Phase II Conceptual Site Model- Registered Plan 65R_22922, Part 17, Markham. Record of Site Condition Filing.

Exp. 2015b. Stargrande Custom Homes Corp Supplemental Phase Two Environmental Site Assessment and Remedial Excavation.

Ministry of Environment, Conservation and Parks (MECP, formerly MOE). 1995. Amended Certificate of Approval (CofA, now ECA) No. 3-1514-95-007.

Ministry of Environment, Conservation and Parks (MECP, formerly MOE). 2001. Amended Certificate of Approval (CofA, now ECA) No. 7027-53LRES.

Appendix D: Internal load estimation from *in situ* water-column TP increases

In situ internal loads (kg/yr) can sometimes be determined according to Equation 4 from the increases of water column TP concentration (mg/L) between spring and fall.

Equation 4 $$\textit{In situ}\ L_{int} = (P\_t_2 \times V\_t_2 - P\_t_1 \times V\_t_1)$$

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

This equation assumes steady state conditions with respect to external inputs and outputs. This estimate of internal load is approximate only, because of the uncertainty associated with (1) the loss of sediment derived P via settling during the calculation period, (2) external TP input, especially that potentially large inputs of geese feces, and (3) dilution by precipitation and inflows from the upstream SWMPs. In addition, it only considers the period in questions and not the entire year. – While usually summer sediment derived load is much larger than during the cold season, the high TP concentration and anoxia measured under ice before the Phoslock treatment and elevated spring TP concentrations in recent years (e.g., Apr 12, 2016, 0.270 mg/L) point to internal loading year-round.

A more comprehensive estimate is based on the P mass balance, as described in Appendix E.

Appendix E: Phosphorus mass balance model and the prediction of TP concentration and annual internal P load

Average annual TP concentration was modeled according to a simple mass balance equation (Equation 5), presented in detail in Nürnberg 2009).

Equation 5 $TP = L_{ext} / Q_s \times (1 - R_{meas})$,

where TP, annual TP concentration (mg/L); L_{ext}, external TP input (kg/yr); *Qs*, outflow volume (m^3), and $R_{meas,}$ measured retention, expressed as proportion according to Equation 6. This measured R term implicitly includes internal TP load, which contributes to TP export (L_{out}, kg/yr, outflow x annual average TP concentration).

Equation 6 $R_{meas} = L_{ext} / (L_{ext} - L_{out})$

Based on the mass balance for 2014, when the low TP concentration and generally good water quality indicated that internal load was at its minimum and possibly non-existent (“zero”), we used the retention computed for that year as a typical P retention (R_{sed}) that only includes downward fluxes (Equation 7). This R_{sed} term can be expected to be similar from year to year in Swan Lake, despite variable internal load.

Equation 7 $R_{sed} = 0.855$

This relatively high measured retention in 2014 makes sense, as it is typical of lakes with a high water residence time and low water load and no internal load (Brett and Benjamin 2008).

The prediction of the specific contribution (TP_i, mg/L) from the individual modelled sources ($Load_i$) to lake TP concentration follows the general mass balance equation (Equation 5), where R_{meas} is replaced by R_{sed} (Equation 7).:

Equation 8 $TP_i = Load_i / Q_s \times (1 - R_{sed})$

The prediction of the average annual lake TP concentration was accomplished by adding all individual loads specific to their sources, or total external and internal loads (L_{int} and L_{ext}, Equation 9), to the general mass balance equation.

Equation 9

$$TP = \frac{L_{ext} + L_{int}}{Qs} \times (1 - R_{sed})$$

The settled fraction of external load can be predicted by the term “R_{sed} x L_{ext}”. In lakes with internal load, predicted retention overestimates measured retention approximately by the net amount of P released from the sediments (net L_{int_2}), (Nürnberg, 1984) so that

Equation 10 $$\text{Net } L_{int} = L_{ext} \times (R_{sed} - R_{meas})$$

As it is based on an annual mass balance, L_{int} includes both summer and a potential winter internal load. Net and gross estimates of L_{int} are related by R_{sed} according to Nürnberg (1998) who showed that internal P released from the sediment settles back down at the same rate as external P on an annual basis. Consequently, a mass balance-based, gross internal load can be calculated from net internal load:

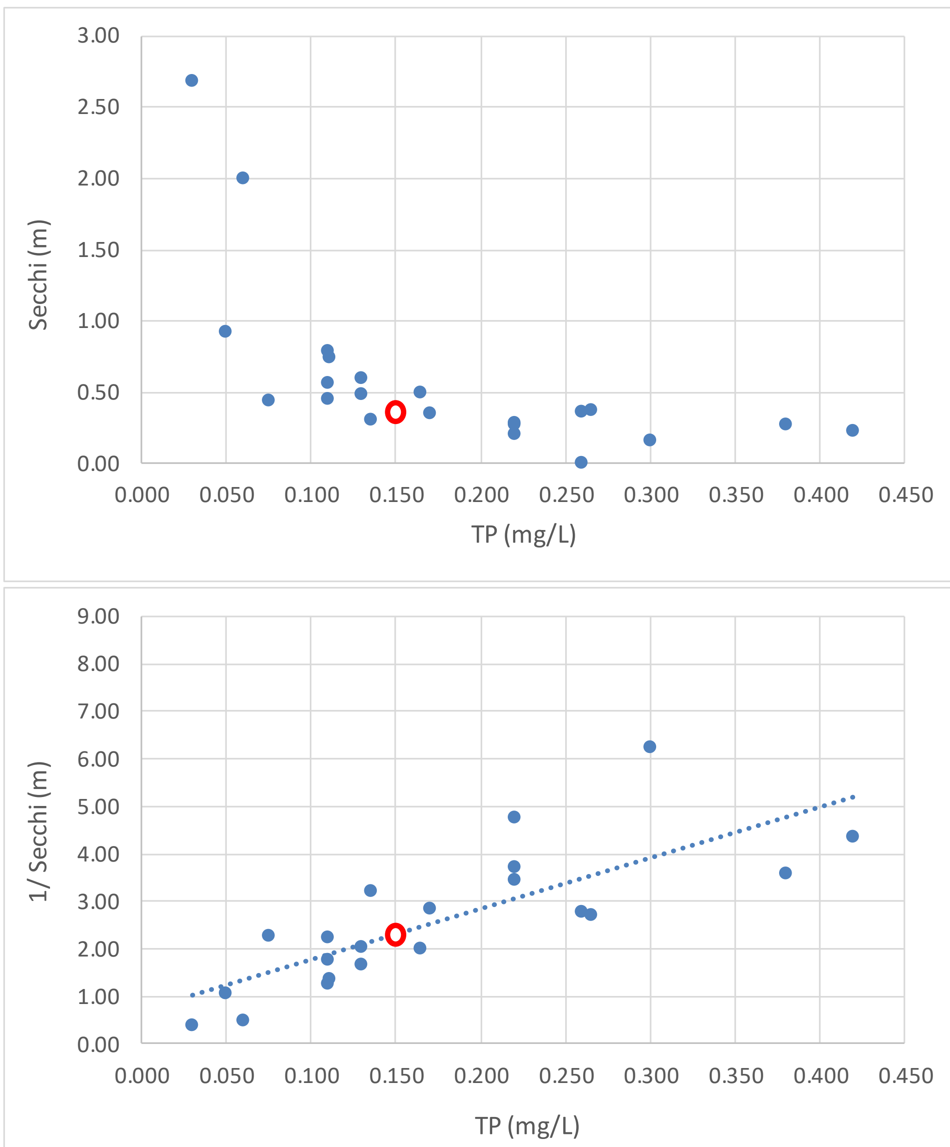
Equation 11 $$\text{Gross } L_{int} = \text{Net } L_{int} / (1 - R_{sed}).$$

(*More simply*: The internal load is computed from the difference between the observed R= (in-out)/in, that includes settling and sediment release, and the R of 2014, when sediment release can be assumed zero.)

Appendix F: Determination of potential water quality goals for Swan Lake

Relationships between Secchi disk transparency and surface (0 - 1 m) TP concentration in Swan Lake indicate that 0.15 mg/L TP approximately coincides with a Secchi depth of 0.45 m (or its inverse of 2.22, Figure 27). Above this TP concentration, Secchi becomes more or less constant; below this TP level, Secchi transparency rapidly increases. We propose to set this combination as an interim goal for water quality in Swan Lake.

Figure 27. Individual observations of TP concentration and Secchi transparency recorded at the deep station (Site #3) for the growing periods May-Oct for all available data in years 2011 through 2017. Top, untransformed Secchi versus TP concentration, bottom, inverse of Secchi versus TP with regression line. The proposed goal is indicated as red circle.



Appendix G. Monitoring plan for Swan Lake water quality

------- *Next page* -----

A. Sampling at "Dock" (Site 1, Figure 1), twice monthly in May through Nov, once monthly at other times:		
Parameter		**Directions**
Temperature	√	To be taken at 0.5 m intervals from lake surface to bottom.
Dissolved Oxygen	√	To be taken at 0.5 m intervals from lake surface to bottom.
Transparency, Secchi	√	Secchi Disk Reading
Colour		0.5 m
Phytoplankton Biomass, Cyanotoxins	√	Routine monthly sampling June - October. Take sample at the depth of highest apparent biomass to determine phytoplankton groups (cyanobacteria, diatoms, etc.). For apparent phytoplankton accumulations: Determine if toxic species are present and test for microcystin by Abraxis test strips.
Phosphorus: - Total P - SRP	√	At least 2 discrete samples at 0.5-1m intervals down to 1m above the bottom sediment using a sampling bottle (Van Dorn bottle). Given 3m depth, measure at 0.5m, 1m, 2m (if possible) from surface. The deepest sample should be collected about 1m above bottom sediment.
Nitrogen: - Ammonium - Nitrate - total Kjeldahl N		0.5m
Dissolved Organic Carbon		0.5m
Chloride		0.5m
General inspection		Waterfowl, carp or gold fish.
B. Sampling at "Bridge" (Site 2, Figure 1), same as Site 1, except for only 1 depth sample		
All chemicals as A		0.5 m if possible, otherwise take sample at half the water column depth.
Temperature, DO		Where possible
Secchi		When possible
C. Monitoring additions specific to an in-lake treatment: deep open water Site 3, 2- 3 times before and 5 times after		
Temperature	√	To be taken at 0.5 m intervals from lake surface to bottom.
Dissolved Oxygen	√	To be taken at 0.5 m intervals from lake surface to bottom.
Transparency, Secchi	√	Secchi Disk Reading
Phosphorus: - Total P - SRP	√	At least 2 discrete samples at 0.5-1m intervals down to 1m above the bottom sediment using a sampling bottle (Van Dorn bottle). Given 4m depth, measure at 0.5m, 1m, 2m, 3m (if possible) from surface. The deepest sample should be collected about 1m above bottom sediment.
Nitrogen: - Ammonium - Nitrate - total Kjeldahl N		0.5m
Dissolved Organic Carbon		0.5m

Total and dissolved lanthanum (TLa and DLa)		TLa can serve as tracer of the Phoslock material. DLa can serve as an approximate or maximal estimate of free lanthanum. It is possible that LaP (rhabdophane) particles evade separation by the standard 0.45 µ pore size filters so that DLa may overestimate “free” La (Nürnberg and LaZerte, 2012).

√ Indicates the most important variables to be monitored in an abbreviated program. Phosphorus could be monitored at one depth, 0.5 m, only. Sampling frequency could be shortened to monthly visits throughout June-October.

Appendix H. Literature review about Phoslock toxicity studies (as of 2016)

Since this review of Jan 2016, more and more literature with respect to possible toxicity of Phoslock has been published, including (Copetti et al., 2016; D'Haese et al., 2019). *None of those publications revealed any toxicity concerns that should apply to the study reservoirs.*

Freshwater Research has reviewed about 60 studies on Phoslock, most of them peer-reviewed and published since 2008 which shows that Phoslock is a relatively new technique in lake restoration. Many of the following references are referred to elsewhere in this report. We here include some verbatim citations from toxicological studies to provide more detailed information. (Abstracts and sometimes the whole text of all published papers are available via Google Scholar on the internet.)

Toxicity to **humans** can be considered negligible or not-existent, because Phoslock® received US and Canadian NSF/ANSI Standard 60 *Certification for use in drinking water* in 2011. This certification ensures that Phoslock® applications to drinking water supply sources, at the maximum use rate specified on the product label, do not contribute contaminants that could cause adverse human health effects. NSF/ANSI Standard 60 is the US nationally recognized health effects standard for products which are used to treat drinking water.

A large amount of scientific literature is available on lanthanum toxicity to human health as lanthanum carbonate (trade name Fosrenol®)[1] is used orally to treat hyperphosphatemia in patients with chronic kidney disease who are undergoing dialysis. The medical doses far exceed those recommended in lake restoration.

As part of the permitting process prior to the use of Phoslock in Ontario toxicity tests were undertaken by MECP on three types of sediment dwelling organisms (***Hyalella azteca*, *Hexagenia spp*.** and ***Chironomus dilutes***), **rainbow trout** and ***Daphnia magna*** (Ontario Ministry of Environment, 2009).

> *Cited from executive Summary of 55 p. report*:" Standard water only 96-hour toxicity tests were performed with rainbow trout and 48-hour toxicity tests were performed with *Daphnia magna*. Standard sediment toxicity tests were performed using *Chironomus dilutus* (10-day exposure), *Hexagenia spp.* (21-day exposure) and *Hyalella azteca* (14-day exposure) in sediment and water collected from the Lake Simcoe watershed. Two application rates were assessed in the sediment test; the filtered reactive phosphorous (FRP) rate dictates the amount of Phoslock required to remove the FRP from the water column and the capping rate (3.4 mg/L) delivers enough Phoslock to create a 1 mm thick layer on the sediment surface. Nutrient and metal concentrations were monitored in the sediment and water for all tests. The 48-hour LC50 for *Daphnia magna* was 4.9 g/L and > 6.8 g/L Phoslock. The rainbow trout 96-hour LC50 was>13.6 g/L. No significant survival or growth impacts were observed in any of the sediment toxicity test species for either of the dose rates. It should be noted that application rates used in 2008 field trials in the Lake Simcoe watershed were 0.02 and 0.05 g/L."

There are many peer-reviewed studies on Phoslock in general and its potential toxicity in particular. For example, a special issue in 2015-16 of the international journal "Water Research" is devoted to *Geo-Engineering in Lakes* by chemical treatments, with more than 50% of the

[1] Stewart J., 2002. Administration of a novel phosphate binder, Fosrenol®, with food is associated with good tolerability and low systemic absorption. J. Am. Soc. Nephrol. (2002) 13, 386A

contributions dealing with Phoslock. Some papers that specifically deal with toxicity are summarized here. They describe a general lack of toxicity of lanthanum based applications using analytical chemistry and biological tests.

It is important to differentiate between total and dissolved, i.e., bioavailable lanthanum (La) (the metal that could provide toxicity in Phoslock). Total La can be relatively large after a treatment, but presents no threat to the biota, only the free La^{3+} ion causes toxicity in Phoslock. A study on the persistence of La^{3+} concentration on 16 applications over up to 60 months found:

> "This modelling indicated that the **concentrations of La^{3+} ions** will be very low (<0.0004 mg L/) in lakes of moderately low to high alkalinity (>0.8 mEq/L)…" (Spears et al., 2013).

A laboratory study determined that the binding of phosphate in the water by Phoslock can lead to stunted growth of Daphnia (Lürling and Tolman, 2010).

> "A life-history experiment with the zooplankton grazer ***Daphnia magna*** revealed that lanthanum, up to the 1000 mg l_1 tested, had no toxic effect on the animals, but only in medium without phosphorous. In the presence of phosphorous, rhabdophane … formation resulted in significant precipitation of the food algae and consequently affected life-history traits. With increasing amounts of lanthanum, in the presence of phosphate, animals remained smaller, matured later, and reproduced less, resulting in lower population growth rates. Growth rates were not affected at 33 mg/L La, but were 6% and 7% lower at 100 and 330 mg/L, respectively, and 20% lower at 1000 mg/L. A juvenile growth assay with Phoslock tested in the range 0–5000 mg/L, yielded EC50 (NOEC) values of 871 (100) and 1557 (500) mg/L Phoslock for weight and length based growth rates, respectively. The results of this study show that no major detrimental effects on Daphnia are to be expected from Phoslock or its active ingredient lanthanum when applied in eutrophication control."

The impact of a Phoslock treatment on algae assemblage composition and **macrozoobenthos** composition was studied in Laguna Niguel, California (Bishop et al., 2014).

> "Invertebrate communities were not significantly impacted following addition of Phoslock based on richness, diversity, and functional feeding groups …. The habitat score was similar in preapplication and postapplication as measured by the following: in stream cover, sediment deposition, water chemistry, and channel alteration."

Changes in the biota of a Scottish lake after a Phoslock application were recorded for the first post-treatment year (Meis et al., 2012).

> "… a significant decrease in dominant benthic macroinvertebrate groups (Chironomidae, Oligochaeta and Sphaeriidae) in summer and autumn in the first year post-application, whereas abundance of zooplankton groups did not change significantly in the first year following the application of Phoslock®. Similarly, abundance of fish (three-spined stickleback, Gasterosteus aculeatus L.) did not change significantly postapplication."

The author further speculated that the decrease in macrobenthos may have been due to the reduction in the trophic state (lower TP concentration) or changes in the sediment habitat (Phoslock's bentonite material) rather than La toxicity.

> "… studies assessing the response of macroinvertebrate abundance after external load reductions find similar declines in abundance (…), so that any potential toxic effect cannot be easily separated from effects occurring during the reduction of nutrient concentrations.

> However, the application of Phoslock® may be comparable to loading scenarios of fine inorganic sediment which can detrimentally affect benthic macroinvertebrate abundance”

In a small Dutch application that included iron chloride in addition to Phoslock, **fish biomass** and therefore indirectly macrobenthos increased after treatment (Waajen et al., 2016). The biomass of most fish species remained constant with small increases in Pike. The main increase was due to increased weight in carp, a fish that was appreciated by local fishermen.

> “The fish stock increased from a low 50 kg/ha before to more than 130/ha after the treatment. This increase was mostly due to adult carp reflecting the greater appreciation of fishermen (and consequently uncontrolled stocking) of the improved water quality.”

In summary, toxicity concerns are low for humans and the biota. While there may be some decrease in the abundance of benthic macroinvertebrate within the first months after treatment (possibly due to smothering by the clay material), studies showed no long-term detrimental effect on fish, macrobenthos, and zooplankton. The apparent lack of toxicity in the field is supported by numerous laboratory toxicity tests. Fish kills during or after a proper (and pure) Phoslock application have not been observed in lakes with appropriate alkalinity.

Appendix I. Standards and policies by governmental agencies world-wide

Much of the information in this section is based on Finsterle (2014) and personal communication with Nigel Traill, Regional Manager - Europe, North and South America, Phoslock Environmental Technologies Ltd, Digital World Centre, 1 Lowry Plaza, Salford Quays, M50 3UB, Great Britain, ntraill@phoslock.com.au.

North America

Phoslock® received US and Canadian NSF/ANSI Standard 60 **Certification for use in drinking water** as discussed in Appendix H. There were several applications done in the **US** with permitting requirements differing by State. Phoslock conforms to ANSI Z400.1-2004 Standard (http://www.sepro.com/documents/Phoslock_MSDS.pdf).

In **Canada**, provinces have the jurisdiction and permitting agency except for lakes and water bodies on Federal Crown Land or Federal Crown Sea Bed. For these water bodies an approval from the federal Department of Fisheries and Oceans (DFO) is needed. Several treatments have been conducted as described by Nürnberg (2017).

British Columbia: not used up to now.

Ontario: The MECP, Ontario Ministry of Environment, Climate Change and Parks, allows Phoslock to be used without further approval, provided the "Standard Operating Procedures" (2010) are followed and provided the receiving water body is not located on "Crown Sea Bed". In that case, the provincial Ministry of Natural Resources and Forestry (MNRF) still needs to give a "work permit". Phoslock has been used in the Lake Simcoe watershed and in Swan Lake, Markham. Contact: Dan Orr, Manager, Technical Support, Central Region, MECP <Dan.Orr@ontario.ca>

Alberta: Alberta Environment and Sustainable Resource Development (AESRD) provided a permit to one or multiple applications on Henderson Lake in Lethbridge. The City of Lethbridge treated Henderson Lake in early 2016. Contact: Lethbridge Parks Manager Dave Ellis (tel: 403-320-3848), AESRD, Ron Zurawell <Ron.Zurawell@gov.ab.ca>. Besides a lake treatment, the Cities of Edmonton and Lethbridge have been using Phoslock for two years in their stormwater ponds, where no permit is required.

Manitoba: It is not certain what permit is required by Manitoba Conservation and Water Stewardship. MCWS has been interested in a potential application on Killarney Lake and contracted Phoslock for sediment fractionations. Contact: Elaine Page (Manager, Water Quality Management Section, Water Science and Management Branch, <Elaine.Page@gov.mb.ca>) and Cassie McLaine.

New Brunswick: The City of Moncton is planning a Phoslock treatment to prevent the establishment of cyanobacteria in one of their drinking water reservoirs. Contact: Nicole Taylor, Engineering and Environmental Sciences, City of Moncton <nicole.taylor@moncton.ca>

Quebec: A permit is required by the Ministère du Développement durable, de l'Environnement de la Faune et des Parcs (MDDEFP). The City of Bromont conducted a Phoslock treatment of Lac Bromont in 2017.
Contacts: Anne Joncas of the Lac Bromont Restoration Group, <anne_joncas@hotmail.com>; Louis Roy of MDDEFP <Louis.Roy@mddefp.gouv.qc.ca>

Applications to two more lakes to treat cyanobacteria blooms are being discussed in Quebec.

Other continents

In **Brazil**, Phoslock® has been certified by IBAMA (the Brazilian Ministry for the Environment) for import, sale and use in Brazil.

In **Australia**, it is certified by NICNAS (National Industrial Chemicals Notification and Assessment Scheme).

In **Europe**, the product can be legally imported and sold under REACH (Registration, Evaluation, Authorisation and Restriction of Chemicals) regulations. Several applications have been done throughout Europe.

In **Asia**, Phoslock has been used in pilot experimental studies in China, permitting information is not available to us.

This list is not comprehensive and updated only to 2017.

Appendix J. Monitoring plan for Swan Lake sediment

Sediment characteristics are to be analyzed by a lab specialized in Psenner-type sediment fractionation.

Locations of recommended sediment sampling sites (based on Figure 13). The previous sites 1 to 3 should be resampled for comparison purposes plus two additional sites as indicated.

Table 29. Proposed sediment fractionation analysis and quote by Institut Dr. Nowak GmbH & Co. KG.

Price list

No. of locations	5
No. of sediment segments/depths	2
No. of replicates/location	2 (**duplicate cores**)
No. of samples/location	4
total No. of samples	**20**

Analysis		n	Price/unit (€)	Total (€)
Dry weight		20	4,20	84,00
LOI 550°C		20	5,25	105,00
TP		20	9,75	195,00
La		20	8,15	163,00
S		20	8,15	163,00
Ca		20	8,15	163,00
Fe		20	8,15	163,00
Mn		20	8,15	163,00
Al		20	8,15	163,00
reduced Psenner (BD-P and NaOH-P fractions)		20	235,00	4.700,00
BD-fraction:	o-PO4, TP, Fe	20	15,25	305,00
NaOH-fraction:	o-PO4-P, TP, Fe, Al	20	20,50	410,00
total sediment work; duplicate cores		20	338,85	**6.777,00 €**

Prices are valid until 2020/03/31.