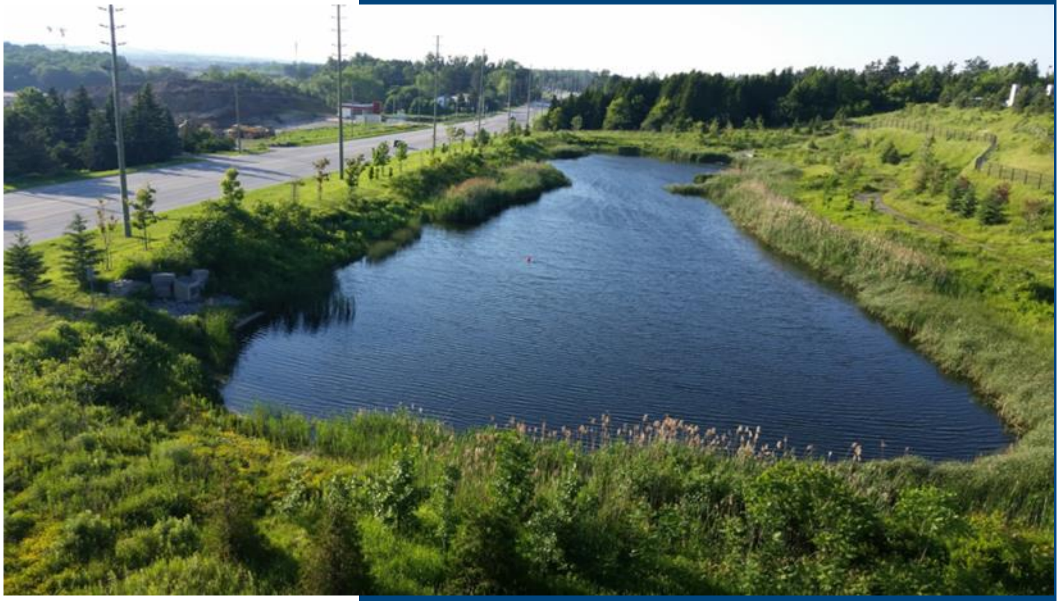
2020

Assessing Stormwater Management Pond Performance



Lake Simcoe Region conservation authority

Assessing Stormwater Management Pond Performance

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The data contained in the maps in this report have been compiled form various sources. While every effort has been made to accurately depict the information, data mapping errors may exist.

The views expressed in this report do not necessarily reflect the Province or any of the funding organizations.

Contents

1.0 Executive Summary

This report documents a project that investigated the impact of external and in-pond environmental drivers on the performance, efficiency, and functioning of urban stormwater management ponds in the southern portion of the Lake Simcoe Watershed. These ponds were originally designed to intercept and reduce the sediment load in surface run-off before it reaches receiving waters such as rivers and streams. As nutrients are typically attached to suspended particles, when these ponds are functioning efficiently there is also a reduction in nutrient loading to receiving waters. Based on our initial (2010) stormwater pond surveys in the Lake Simcoe Watershed, the significant environmental drivers of conditions in these ponds are, unsurprisingly, related to the urban landscape where the ponds were built: nutrient runoff and winter de-icing salt. In a more targeted survey of 11 ponds along a gradient of urbanization (from catchments with primarily residential homes to catchments of commercial / institutional uses with expansive paved parking lots), we found that all the study ponds were functioning less efficiently than predicted by environmental models, and this reduced performance is related to several factors. First, inputs of surface run-off laden with winter de-icing salt forms a chemical stratification of the pond water column, which impairs nutrient retention by the pond as low dissolved oxygen concentrations oxygen near the bottom facilitate release of phosphorus stored in sediments. Second, the higher the salinity of the bottom waters of the pond, the more resistant the pond is to mixing of the water column, likely leading to many precipitation events flowing through the pond, on top of the saline bottom water layer, and directly into the aftbay and receiving waters. Third, the release of sediment-bound inorganic phosphorus is readily used by organisms (aquatic plants and algae), or flushed from the pond if a large rainfall event occurs and mixes the water column, leading to higher loading of biologically-available phosphorus in the receiving waters designed to be protected by the pond. Finally, some ponds in residential areas have persistent elevated turbidity in the pond water. Although ponds may have increased turbidity and suspended sediment levels following precipitation and run-off events, the long, rectangular, shape of the ponds were designed to promote sediment settling. In ponds with persistent turbidity, we recorded the presence of goldfish, likely released by residents living near the pond, which keep pond sediments suspended and further impair stormwater pond functioning.

2.0 Overview of Stormwater Management Ponds

Stormwater management ponds (SWMPs, also known as stormwater retention ponds, or stormwater detention ponds) are primarily constructed for retaining surface stormwater run-off and reduce flooding in urban areas. Flooding can be a particular problem in urban settings as removal of natural cover, changing landscape topography, and the increased area of impermeable surfaces (e.g. roads and parking lots) change surface water flow patterns, limit precipitation infiltration and groundwater recharge, and increase surface runoff during storm events (Hogan and Walbridge 2007). In the mid to late-1980's, new developments in stormwater management occurred as concern arose from the federal Department of Fisheries and Oceans (DFO) which viewed the release of untreated stormwater as a "discharge of a deleterious substance" under the terms of the federal Fisheries Act. Based on the experiences of jurisdictions outside of Ontario, different methods were introduced to help mitigate the effects of stormwater runoff (SWAMP, 2005). These methods were formalized during the development of the Ontario Ministry of the Environment and Energy (MOEE)'s 1994 Stormwater Management Practices Planning and Design Manual. Some ponds were constructed as a simple "dry" basin that held back surface run-off after rain events to prevent flooding (also called "quantity" ponds). However, it was noticed that when water was retained in SWMPs (i.e. "wet" ponds), suspended particles tended to settle out, thus providing some passive water treatment and improving the water quality before discharge to receiving waters - usually urban streams and rivers (Hogan and Walbridge 2007).

The addition of wetland vegetation to ponds further improved the removal of suspended particles and reduced nutrients in discharged water. As these suspended particles are attachment sites for phosphorus, removing the particles results in some reduction in nutrient loading to receiving waters. Thereafter, these newer SWMPs (termed "quality" ponds) were then designed with additional engineered features (e.g. two basins, bottom draw outlets, vegetation, floating plant islands, etc.) to enhance phosphorus capture and storage (Hogan and Walbridge 2007, Marsalek 2002). These ponds were designed to be shallow with a large surface area to prevent water column stratification and prevent release of sediment nutrients by keeping the bottom waters oxygenated (Frost et al. 2019). Through their design, SWMPs are intended to be effective at capturing suspended solids to reduce turbidity in receiving waters, but they are not efficient at capturing biologically active compounds such as phosphorus and other nutrients. Estimates of SWMP efficiency showed a removal of up to 85% of suspended particles from incoming run-off. Although this pond design is effective for pollutants such as suspended solids; biologically active compounds, such as phosphorus and other nutrients, do not follow a sinking-capture-burial model and are cycled within the pond by physical, chemical, and biological processes. Studies have shown that overall removal of phosphorus in SWMPs is 30-40% due to in-pond nutrient cycling, use by pond organisms (e.g. algae and plants), and chemical transformation to more mobile forms of phosphorus (Frost et al. 2019). As such, concerns have been raised about how in-pond physical conditions and chemical cycling have impacted the efficiency and performance of SWMPs, features that have been relied on as one of the most common types of urban surface water treatment, and are required in newly built urban areas in Ontario (MOE 2003). In the Lake Simcoe watershed, there are approximately (as of 2018) 350 SWMPs.

Although the original SWMPs were designed as a simple, single basin to retain a volume of stormwater in flood-prone areas, newer SWMPs have been redesigned to enhance suspended sediment and nutrient capture. These quality control ponds often have an elongated appearance in order to maximize settling time for suspended solids, with two (or more) chambers (also called basins, bays, or pools). A “forebay” near the inlet serves to slow the velocity of incoming water, and ends at a barrier or berm (typically submerged) to separate the chambers. A relatively larger “aftbay” (or permanent pool) holds back stormwater while allowing suspended particles to settle out before release to streams, rivers, or urban stormwater infrastructure (Figure 1). Most of the original SWMPs had surface outlets to release water, but some ponds have been built with deeper water depths and a “bottom draw” outlet that is intended to cool pond water temperatures to minimize thermal shock to receiving waters, particularly those that are habitat for sensitive cold-water fish species.

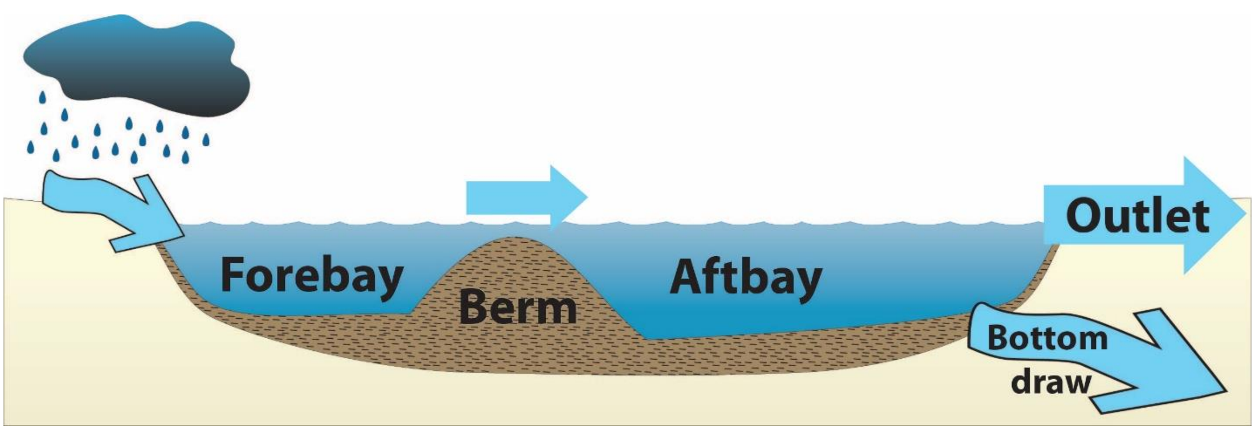

Figure 1. Conceptual diagram of a typical stormwater management pond showing precipitation and surface run-off entering the forebay, flowing across the berm to an aftbay, then release to receiving water by either, or both, a surface outlet or bottom draw outlet.

3.0 Rationale for this study

Based on surveys of SWMPs in the Lake Simcoe Watershed, Lake Simcoe Region Conservation Authority (LSRCA) found that many ponds were not functioning as designed, a problem found to be widespread across Ontario (SWAMP 2002, Williams et al. 2013) and elsewhere (Taguchi et al. 2020, Troitsky et al. 2019, Sonderup et al. 2016) . Many ponds are exporting higher phosphorus concentrations than predicted using ecological models, particularly in bioavailable and dissolved forms; other ponds had excessive growth of aquatic plants and algae; and some bottom draw ponds may not have the predicted reduction in bottom water temperatures that would protect cold water habitats (Drake et al. 2016). As SWMPs are widely used to retain and treat stormwater in urban areas of the Lake Simcoe Watershed, we undertook an investigation to assess SWMP performance from two approaches. First, we investigated the physical and engineering aspects (pond filling rates, maintenance schedules, resilience of pond infrastructure, pond longevity, etc.) to determine appropriate routine maintenance and clean-out schedules (see LSRCA, 2011). Second, we investigated in-pond environmental aspects (e.g. water and chemistry, physical variables, and biological issues) that may impair pond performance, which is detailed in this report.

To assess the performance and functioning of SWMPs, key environmental variables were recorded, used to assess the main drivers of changes in these ponds, and then monitored to determine environmental trends. Initially, physical water quality data was collected, during the summer and fall of 2010, from 92 urban SWMPs in the Lake Simcoe Watershed. To determine significant environmental drivers impacting these ponds, a Principal Components Analysis (PCA) was carried out. Variables included in this analysis were: maximum depth of the pond, pond volume, surface area of the pond catchment, dissolved oxygen (DO) at top and bottom of the water column, minimum recorded DO, water temperature at top and bottom of the water column, pH at top and bottom of the water column, specific conductance at top and bottom of the water column, and sediment total phosphorus concentration. In order to estimate the strength of any water column stratification / layering, we included the difference in top and bottom values recorded for DO, water temperature, and specific conductance. From this analysis, specific conductance and the difference in specific conductance between top and bottom layers of the water column (our indicator of strength of salt stratification) explained 85% of the variance, suggesting that inputs of winter de-icing salts are the main driver of variation in SWMPs.

Of the 92 SWMPs analyzed above, 34 ponds also had water samples taken for chemical analysis of nutrients (total phosphorus (TP), and nitrogen species). Rather than use median values as estimated data on the full suite of 92 ponds, this subset of 34 ponds was analyzed by a separate PCA to determine if nutrients in pond water were a significant environmental driver. In addition to the 16 drivers used above, this second PCA also included TP, total Kjeldahl nitrogen (TKN), and ammonia (NH_3) data from samples taken at the top and bottom of the water column. Results of this analysis (Figure 2) were consistent with the full suite of 92 ponds in that specific conductance and strength of salt stratification were main environmental drivers in these ponds (Axis 1 explaining 59.3% of variance), but nutrients also played a supporting role (Axis 2, TKN and TP, explained 13.9 % of variance). Combined, these four variables explained 73.2% of the variance in the dataset, suggesting that variables associated with urban runoff, and not environmental drivers such as ambient temperature, are the main drivers of variation in these ponds. These results were not unexpected as SWMPs are mostly located in urbanized areas, which have higher concentrations of salt (from winter de-icing practices) and nutrients (less infiltration due to impervious surfaces) in stormwater runoff.

Based on these results, the impacts of winter de-icing salt application and urban surface runoff were the main drivers of environmental conditions of these ponds, and possibly have a large influence on SWMP functioning and performance. Even in SWMPs designed with a relatively shallow water depth to avoid issues associated with thermal stratification of the pond water column (and thus limit the anoxic release / internal loading of sediment-bound nutrients), salt-caused stratification of the water column was a significant driver of pond environmental conditions.

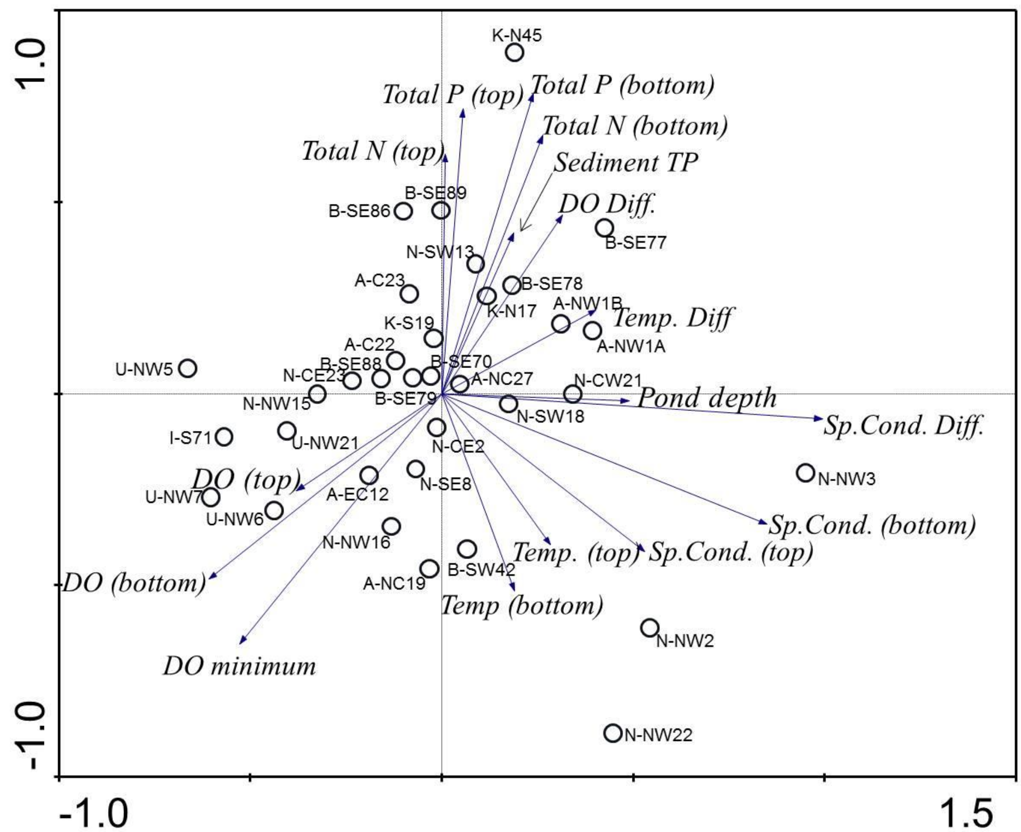

Figure 2. Principal components analysis (PCA) plot of 34 urban stormwater management ponds in the Lake Simcoe Watershed in relation to significant environmental variables. The environmental variables tested are shown by the blue vectors radiating from the origin, pointing toward the higher end of the gradient, with more closely related variables grouped together. The individual ponds (shown by the open circles with the pond code) are arranged in three dimensional space relative to the environmental variables. Perpendicular lines, relative to an individual vector, can be drawn from each pond to show placement along the gradient of interest. For example, pond K-N35 would have the highest Total P recorded among the ponds, pond N-NW22 would have lowest Total P. Strongest salt stratification (Sp. Cond. Diff.) would have been recorded at pond N-NW3, and the lowest would be at U-NW5. Environmental variables, related to specific conductance, along the horizontal axis were the primary drivers, while variables along the vertical axis, related to nutrients, were the secondary drivers.

3.1 Goals and Objectives

The purpose of the project was to investigate environmental influences that may be impacting the performance efficiency and functioning of SWMPs in the Lake Simcoe Watershed. As many of the performance issues associated with SWMPs are similar across southern Ontario, the results of this study are likely applicable to other areas.

The first objective of this current investigation was to further explore the impacts of urban land uses on SWMPs. For this, we selected 11 ponds in the vicinity of Aurora and Newmarket (Ontario) (Figure 3) with catchment land uses covering a gradient of urbanization (residential versus commercial catchments) and thus salinity / specific conductance differences. Four ponds had predominantly residential land use in their catchments, made up of homes with lawns and relatively low traffic streets and had relatively lower salinities. Six ponds had commercial / institutional use in the catchments with retail or office buildings and expansive parking lots, generating relatively higher salinities due to winter maintenance practices. One pond (Mulock) had a mixture of residential and commercial / institutional use in its catchment (Table 1). The ponds selected also covered a range of ages, the oldest being built in 1987 and newest in 2011.

The second objective was to investigate trends in SWMP functional and performance assessments regarding water column stratification. Three ponds were selected for continuous monitoring with water quality sensors (measuring water temperature, DO, and specific conductance) deployed near the surface and near the bottom in both the forebay and aftbay of each pond. In addition to the trends recorded in these variables, these data were used to develop a modified water column stratification model to determine critical impairments to pond performance such as resistance to water column mixing and the impact of highly saline bottom waters in a pond to the efficient functioning of SWMPs.

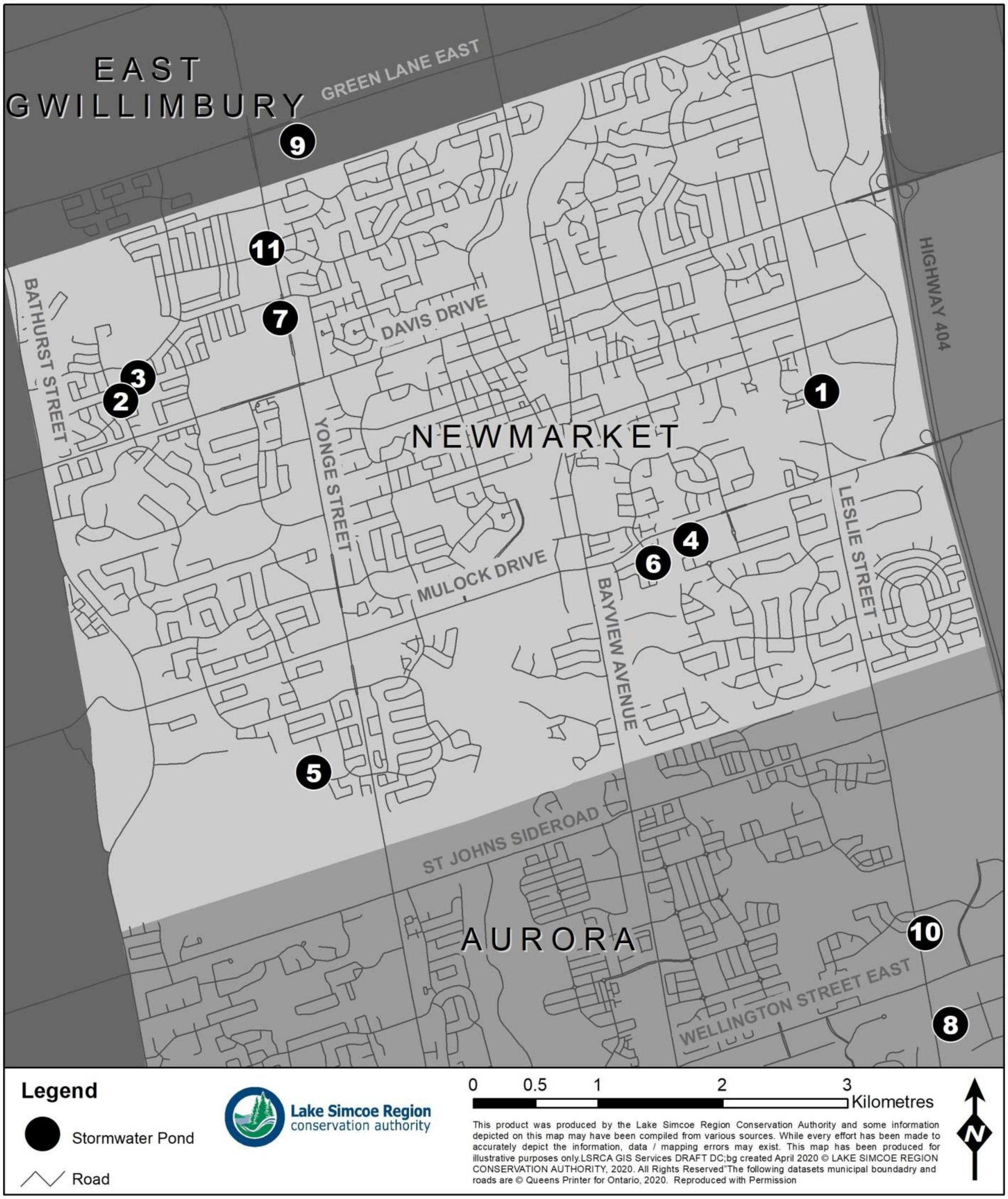

Figure 3. Map showing locations of 11 stormwater management ponds selected for detailed study. Numbers correspond to pond names in Table 1. Three ponds selected for continuous monitoring were numbers 5, 8, and 10.

Table 1. Physical and chemical data from the 11 ponds investigated by this study. Numbers reference locations on map in Figure 3. Pond code is identifier used by the municipality. Bolded font indicates three ponds selected for continuous monitoring.

	Pond name (Code)	Year built[1]	Max. depth (m)	Pond volume (m^3)	Minimum recorded DO(mg/L)	Bottom sp. cond. (uS/cm)	Sediment TP (ug/g)	Water TP (ug/L)
	Residential:							
1	Crowder (N-CE2)	1987	2.6	7088.05	0.6	1062	860.79	23.0
2	Ford Wilson	2011	1.6	3961.09	1.5	3600	770.67	73.0
3	Labyrinth (N-NW15)	2007	3.0	6675.41	2.8	481	939.67	31.0
4	Mulock	2007	1.3	2877.27	0.5	5702	703.70	32.0
5	**Oaktree (N-SW18)**	**2005**	**2.54**	**1764.76**	**0.2**	**633**	**903.19**	**33.0**
6	Walpole (N-SE8)	1994	1.53	4172.06	7.8	722	762.74	18.0
	Commercial:							
7	Dawson Manor (N-NW3)	1995	1.92	610.86	0.2	46620	758.25	36.0
8	**Hillock (A-EC1)**	**2011**	**1.7**	**4079.38**	**0.2**	**35690**	**645.21**	**40.0**
9	Green Lane (N-NW21)	2002	1.8	8570.67	1.2	26140	835.41	-
10	**Site C (A-NE10)**	**2008**	**2.4**	**8069.47**	**0.2**	**59130**	**848.55**	**42.0**
11	Bonshaw (N-NW2)	2000	3.3	11888.33	6.65	8747	945.47	13.0

1- Field denotes age of the pond from construction or date of last clean out maintenance.

4.0 Monitoring Results on 11 SWMPs

As stated above, from the suite of 92 ponds, 11 SWMPs were selected for on-going, routine, monitoring in 2017 and 2018, with some key variables (turbidity, TSS, and water clarity) selected for additional monitoring in 2019. For five of these 11 ponds, water samples were analyzed at a contracted laboratory (Caduceon Laboratories, Richmond Hill ON) for TSS, turbidity, TP, with pond sediments being analyzed for sediment TP, chloride, and sodium absorption ratio (SAR). As with the 11 pond study set, these five ponds were divided based on predominant land use in the catchment. Three ponds were classified as residential (Labyrinth, Ford Wilson, and Oaktree ponds), and two ponds being classed as commercial / institutional (Hillock and Dawson Manor ponds). Sampling was undertaken from April until November in each year, with water quality samples being taken at least 48 hours after a rainfall event in order to allow settling of suspended solids and comparison of more representative pond conditions. Sediment samples were collected from the center of forebay and aftbay (where present) of each pond. Water quality samples were collected 30 cm below the surface and 30 cm off the bottom, in the center of the forebay and aftbay (where present).

4.1 Total Suspended Solids

Reduction of TSS was the primary objective for re-envisioning the design and implementation of SWMPs from simple, quantity control basins that might dry out to permanent wet features that could provide more water quality control and thus mitigate the effects of the urban environment on receiving waters. By capturing the suspended material from surface run-off, a relatively low-cost water treatment could be gained before this run-off entered receiving waters, usually urban creeks and tributaries. In the five ponds selected for water chemistry, samples were collected at least 48-72 hours after a rainfall, to allow particle settling, and discern seasonal trends in the TSS data. There were no statistically significant differences in TSS between residential or commercial ponds, although the individual ponds did show either no trend, or a slight decreasing trend in TSS during the monitoring year (Figure 4). TSS was typically higher in spring, compared to summer or fall samples, likely due to spring surface run-off from snowmelt. Additionally, the presence of aquatic plants in some ponds, during summer and fall, likely contributed to slowing water flow and improving the sedimentation of suspended solids. Although these plants also had a role in the uptake of biologically available phosphorus, this form of the nutrient was released during plant die-back and decomposition in fall.

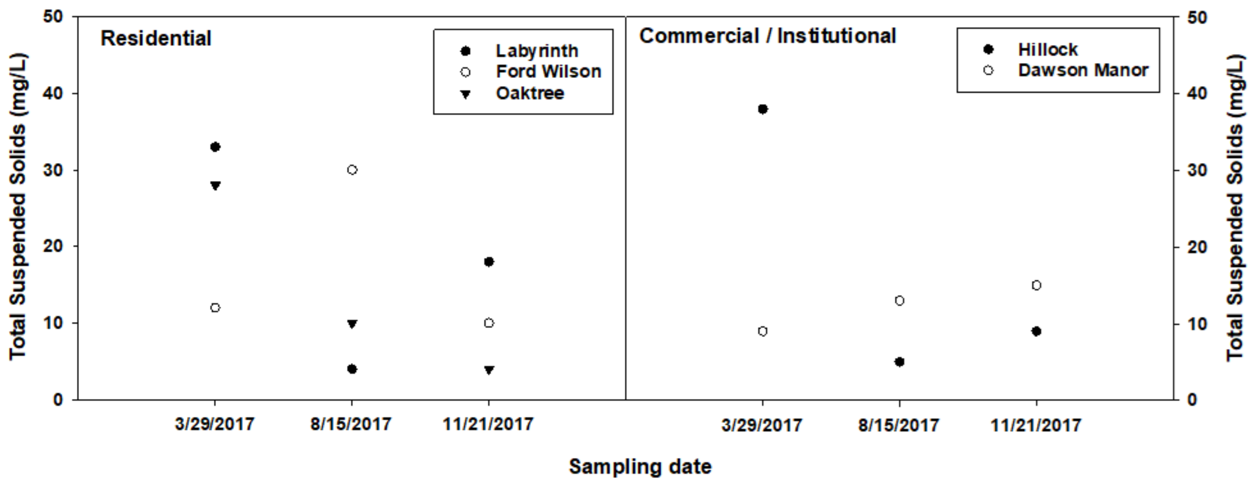

Figure 4. Total suspended solids (TSS) concentrations recorded during spring, summer, and late fall 2017 at three ponds with residential catchments and two ponds with commercial / institutional catchments.

4.1.1 Turbidity and the presence of released goldfish

Closely related to TSS is the turbidity of the water column. In the eleven ponds monitored with a water quality sonde for turbidity (EXO 2 water quality sonde by YSI, Yellow Springs, OH), there were no statistically significant differences in turbidity between ponds with residential or commercial catchments (Figure 5). Two ponds however, Ford Wilson and Walpole, did have turbidity values significantly greater than average. Observations of satellite imagery (Google Earth) also showed these ponds had persistent higher turbidity over several years, compared to their nearest neighbor ponds, with similar catchment use, a short distance away (i.e. Ford Wilson vs Labyrinth and Walpole vs Mulock ponds). Further investigation of this occurrence showed that the two turbid ponds had goldfish (*Carassius auratus*) present, whereas the other ponds did not. Both persistently turbid ponds were located in residential locations with easy public access (e.g. walking trails). The source of the goldfish is likely deliberate release from home aquaria, or for religious / cultural practices.

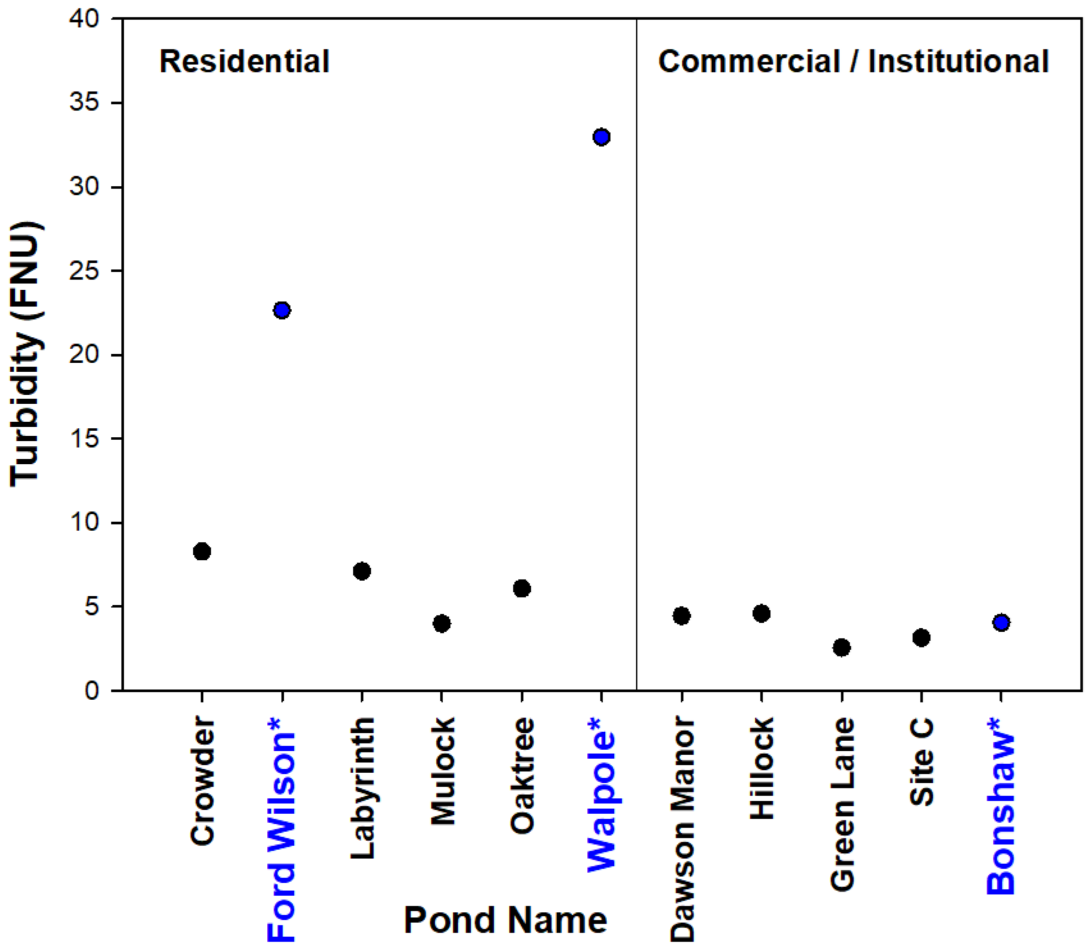

Figure 5. Mean (2017-19) turbidity values recorded at stormwater pond study sites. Three ponds noted with an asterix (*) and bolded blue font (Ford Wilson, Walpole, Bonshaw) have recorded presence of goldfish.

Further investigations of data collected during 2017-19 of the three goldfish ponds (Ford Wilson, Walpole, and Bonshaw), and their respective nearest neighbor pond without goldfish (Labyrinth and Mulock) indicated an impairment in pond functioning due to goldfish behavior. Ponds containing goldfish had significantly decreased ($p<0.001$) Secchi disk transparency (i.e. water clarity; Figure 6), compared to most ponds without goldfish, Bonshaw Pond being an exception. Other key environmental variables (total phosphorus, $p<0.05$; total suspended solids, $p<0.001$; and turbidity, $p<0.001$) were also significantly higher in ponds containing goldfish, except Bonshaw Pond (Figure 7). In order to adjust water clarity for the varying water depth in these SWMPs, Secchi disk transparency was converted to a proportion of pond depth. For example, a Secchi disk transparency of 0.6 m indicates turbid conditions in Ford Wilson Pond (water depth ~1.8 m) but would equal the maximum depth in Mulock Pond. Using this indicator (Figure 7c), ponds containing goldfish had a significantly higher ($p<0.001$) proportion of the water column with turbid conditions, regardless of pond depth.

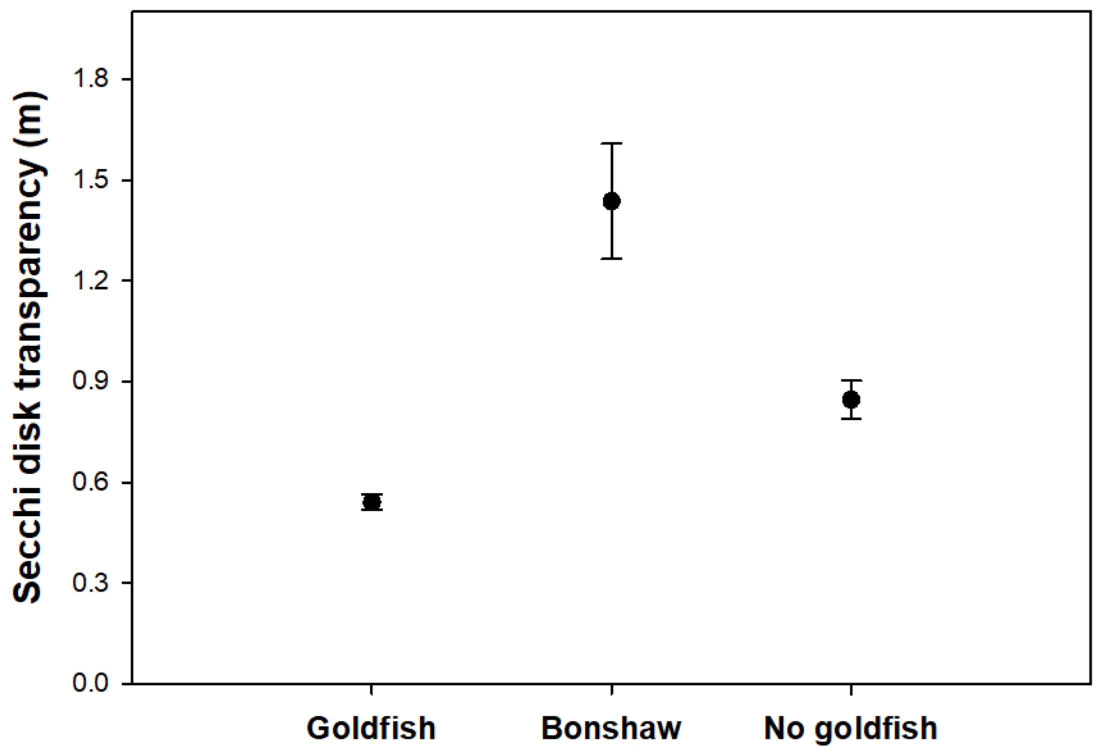

Figure 6. Graph of mean (2017-8) Secchi disk transparency recorded at stormwater pond study sites with and without presence of goldfish. Note: Bonshaw Pond had goldfish present, but also relatively higher water clarity.

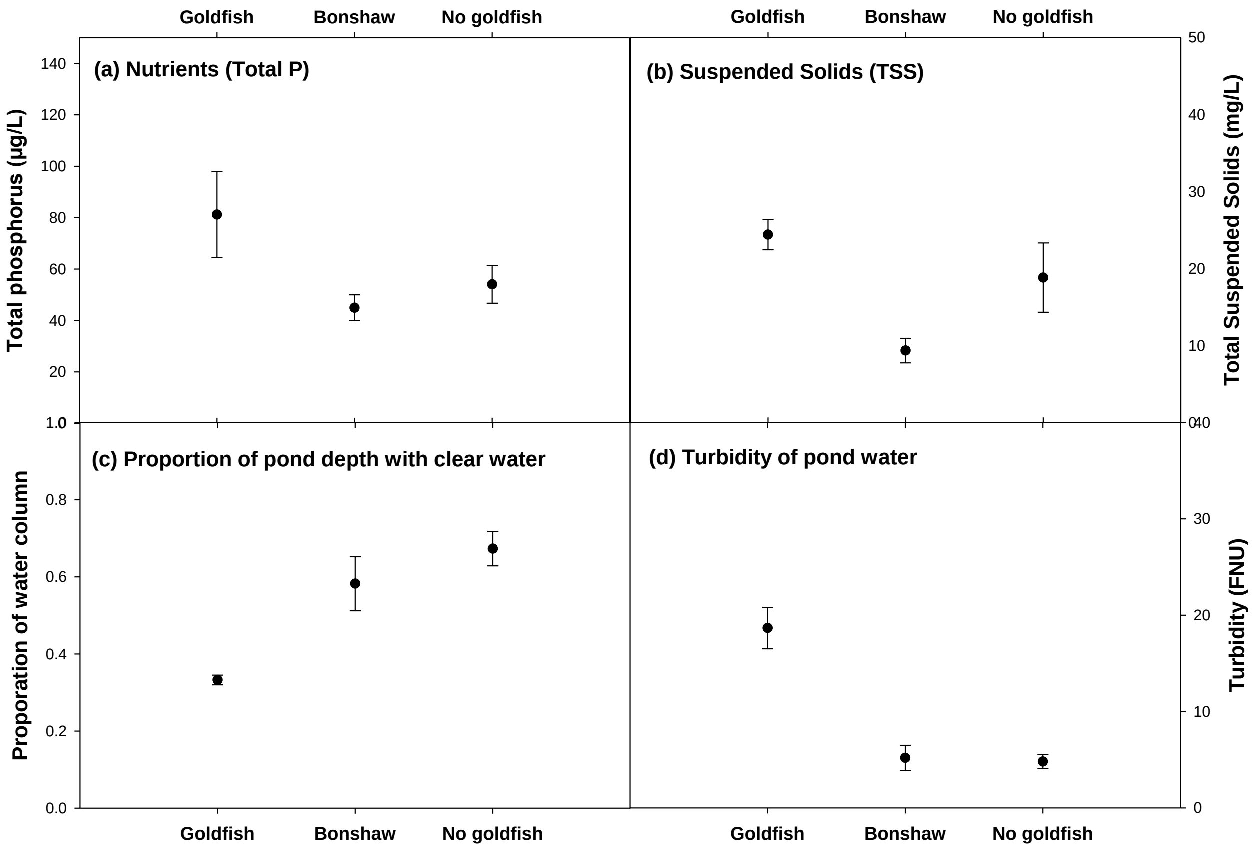

Figure 7. Graph comparing key environmental variables for ponds with and without goldfish present: (a) total phosphorus, (b) total suspended solids, (c) proportion of the pond depth with transparent water, and (d) water turbidity. Note: Bonshaw pond had goldfish present, but key pond variables more closely resembled the non-goldfish ponds.

The results obtained from our suite of SWMPs are similar to other studies that investigated the impact of goldfish, and common carp (*Cyprinus carpio*), on waterbodies. As predominantly benthivores that also graze on aquatic plants, these fish stir up bottom sediments with feeding and spawning behaviours (Lynch and Norland, 2001). As such, goldfish, and common carp have been found to be a cause of increased turbidity, TSS, and phosphorus concentrations, as well as decreased water clarity and vegetation density, in ponds where they are present (Bajer and Sorensen 2015, Richardson et al. 1995). In a study from a coastal marsh in Hamilton, Ontario, Thomsen and Chow-Fraser (2011) reported that carp account for 35-40% of water turbidity. When considering the primary function of quality control

SWMPs is to remove suspended solids by sedimentation, goldfish are impairing the performance of these ponds.

As mentioned above, there was one outlier pond in our study (Bonshaw Pond, Figure 5) that contained goldfish, but also consistently had very low turbidity and TSS. In comparison with the other study ponds, Bonshaw Pond was more similar to ponds without goldfish than with goldfish. Further investigations are required to determine a cause of this dissimilarity, but our records show a significantly lower number of goldfish in Bonshaw Pond, compared to Ford Wilson or Walpole ponds, but the Bonshaw Pond fish were a larger body size. In addition, Bonshaw Pond had less silt / sediment on the bottom, compared to other ponds in our study, which may also contribute to lower turbidity / TSS recordings. This lower amount of bottom sediment may be attributable to a lower sediment load in runoff from the primarily commercial use catchment for Bonshaw Pond, compared to the residential catchments of Ford Wilson and Walpole ponds. Future planned studies on the effects of goldfish on SWMPs will include monitoring Walpole Pond over the next few years when the water and goldfish are removed for planned maintenance, and differences in environmental quality, as well as pond functioning, after this construction is completed.

4.2 Phosphorus

In-pond water samples were collected for TP analysis at three ponds with residential catchments (Ford Wilson, Labyrinth, and Oaktree ponds) and two ponds with commercial / institutional catchments (Dawson Manor and Hillock ponds). As with samples for TSS and turbidity, TP samples were collected at least 48 hours after a rain event in order to minimize the influence of pond mixing on water quality data. No statistically significant differences were recorded between commercial and residential ponds, and no significant annual trends were recorded within each pond (Figure 8). However, Labyrinth Pond did show a decrease in TP concentration in fall samples, relative to spring samples, that mirror the trend in TSS, reported above, for this pond. Concentrations of TSS are correlated with TP as suspended solid particles can be an important source of phosphorus.

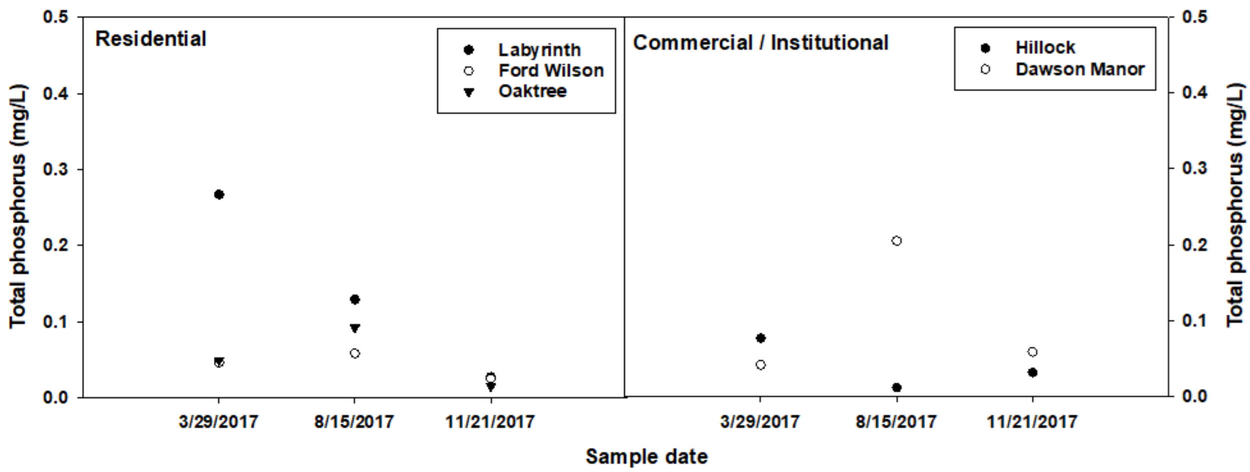

Figure 8. Total phosphorus concentrations recorded during 2017 at three ponds with residential catchments and two ponds with commercial / institutional catchments.

Recorded reactive PO_4 values were low with most sample results being below the 0.01 mg/L detection limit of the analysis lab. Only three samples were above detection, all recorded on March 29, 2017: 0.04 mg/L at Labyrinth, 0.01 mg/L at Ford Wilson, and 0.02 mg/L at Hillock ponds. These results are likely a seasonal effect as PO_4 is readily used by plants and algae, neither of which have a high biomass in March. Lower PO_4 values were recorded in later samples, as would be expected due to uptake and use by plants and algae in the pond (Troitsky et al. 2019).

4.2.1. Sediment phosphorus and fractionation

Sediment samples were taken from 11 ponds in 2016 and 2017 for phosphorus fractionation analysis. This study investigated the proportion of different sediment-bound phosphorus fractions, in order to study changes and trends in these fractions, particularly with respect to the release of sediment phosphorus during periods of low DO in bottom waters. No statistically significant differences were recorded within each phosphorus fraction between years (2016 vs 2017 for each fraction and TP), or between samples from residential and commercial ponds (Figure 9). Sediment TP concentration, and the concentration of each fraction from these SWMPs were similar to samples taken from other sites (other SWMPs, Holland River, Lake Simcoe, and Oak Ridges Moraine kettle lakes) throughout the Lake Simcoe watershed, all having sediment TP concentrations in the range of 800-1000 ppm (Ginn 2011, Moos and

Ginn 2016, LSRCA unpublished data). The calcium-bound fraction of the sediment TP was the largest and values were, again, consistent with other studies (500-600 ppm; LSRCA unpublished data), likely reflecting the predominance of calcium-based limestone bedrock in the Lake Simcoe Watershed.

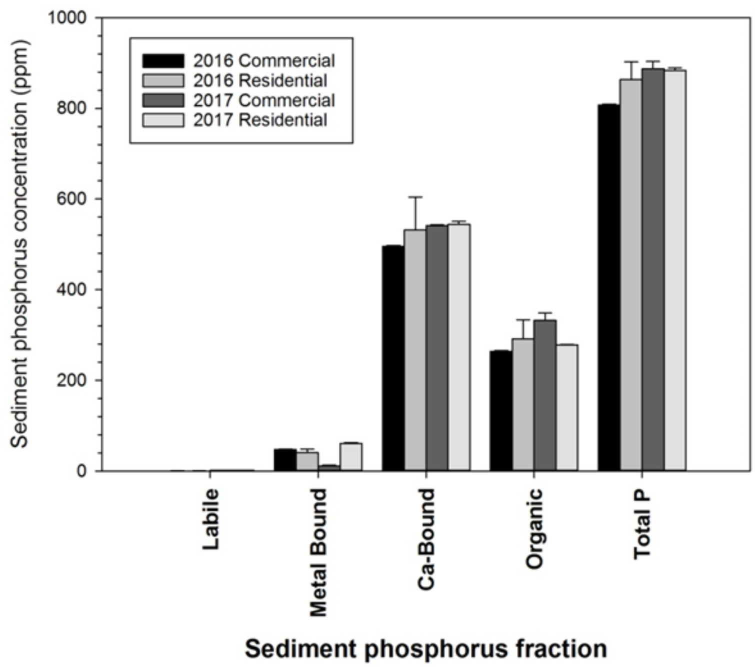

Figure 9. Graph of results from sediment phosphorus fractionation to determine difference between residential and commercial ponds, 2016 vs 2017, and proportion of sediment-bound phosphorus fractions.

Releasable fractions of sediment phosphorus constitute the remainder of the phosphorus in pond sediments, suggesting that ~40-50% of P in sediments is temporarily stored and available for release under certain environmental conditions (see Frost et al. 2019). In our ponds, the labile fraction was smallest (~1.1 ppm) and is the portion of sediment P in flux with overlying bottom water. The organic P fraction (~290 ppm) is cycling with aquatic plants or algae, or the result of the breakdown of biological material and is a form readily available for use by organisms. According to Frost et al. (2019), dissolved organic P is prevalent in pond water and sediments and is relatively mobile, being released from sediment to the water column under both low and high oxygen conditions. The metal-bound fraction (~40 ppm; phosphorus sorbed on to iron, magnesium, aluminum, or other metal particles) is available for release into the overlying water column by microbes and redox chemical reactions when bottom water dissolved oxygen concentrations fall below ~2-3 mg/L. With the temporary nature of phosphorus storage in pond sediments, it is possible that a significant portion of nutrients in a SWMP can be in the water column at any one time, available to be released into receiving waters with precipitation events that may cause pond flushing. The cycling of phosphorus within SWMPs likely reduces the efficiency of these systems for phosphorus retention (Frost et al. 2019, Troitsky et al. 2019), and can make SWMPS a net source of phosphorus to receiving waters (Erickson et al. 2018).

4.3 Chloride

As a consequence of their locations in urban catchments, with the winter use of de-icing salts on paved surfaces, SWMPs have elevated concentrations of chloride, and higher specific conductance values, than more naturalized or reference locations. As expected, commercial / institutional ponds have higher chloride concentrations and specific conductance values than ponds with residential use catchments (Figures 10, 11). This is likely a result of the heavier use of de-icing salts on large areas of commercial paved surfaces (e.g. parking lots) compared to residential catchments where paved surfaces are limited to roads, walkways, and driveways, and there is a greater surface area of grass and vegetation. Additionally, ponds that have catchments with larger parking lots (e.g. Site C: 14.2 ha paved surface area) have higher chloride and specific conductance than ponds in catchments with less impervious area (e.g. Hillock: ~1.9 ha paved surface) or undeveloped portions of the commercial lot (e.g. Green Lane: ~4.7 ha). An exception to this is Dawson Manor Pond (paved catchment area: 2.2 ha) that has higher than expected chloride and specific conductance. This may be due to different application practices, such as using more de-icing salt on this sloped parking lot, compared to other, relatively flat parking areas (Lam et al. submitted).

Inputs of chloride and de-icing salts to SWMPs have the potential to set up a chemical stratification similar to that seen in marine estuaries or meromictic lakes (Sibert et al. 2015). Saline water has a higher density than freshwater and forms a water column in the pond that is stratified not only by temperature, but also by salinity. The amount of salt inputs, related to application rate of winter de-icing salt and the area of impervious surface treated, determines the strength of this stratification, with the density difference between freshwater (on top) and salt water (on bottom) layers, making the pond more resistant to mixing by wind or inputs from precipitation. Using data recorded at three continuously monitored SWMPs, this salt stratification phenomenon was explored and is further discussed below (Section 5.4-5.5). This stratification is not unique to SWMPs, as larger lakes located in urban areas have recently been reported to have persistent chemical stratification that may inhibit water column mixing due to road salt use and result in anoxic bottom waters; a condition that has been termed "cultural meromixis" (Dupuis et al. 2019, Hintz and Relyea 2019, Sibert et al. 2015, Wiltse et al. in press).

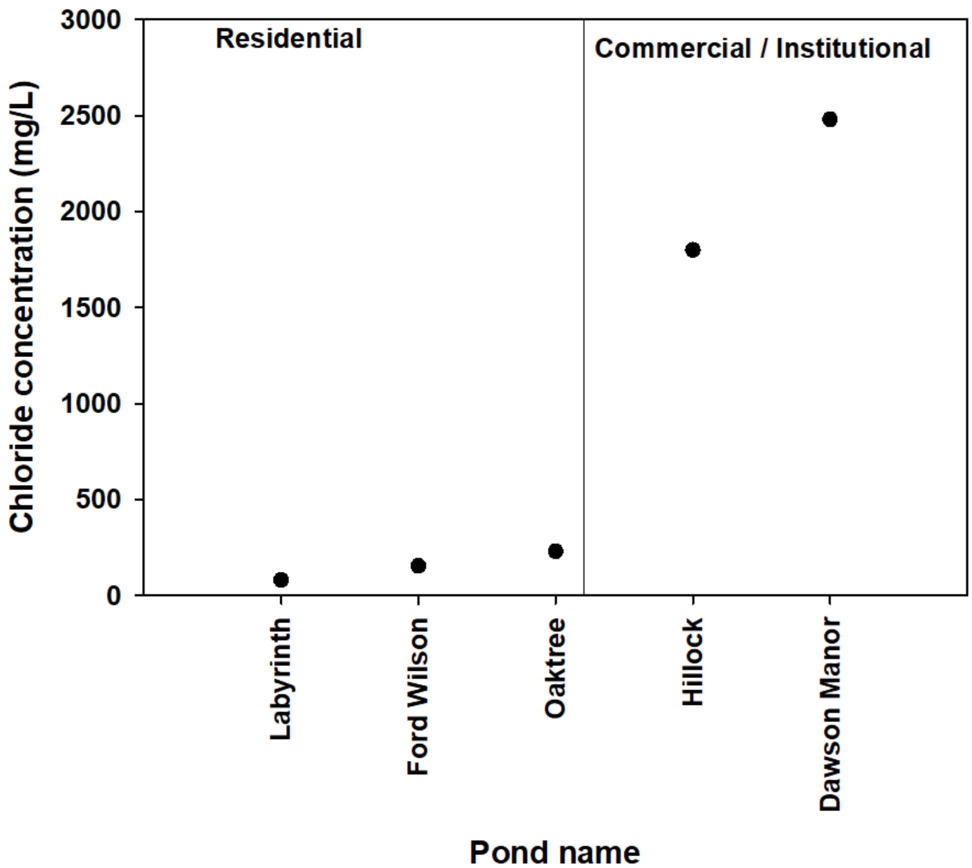

Figure 10. Chloride concentrations recorded from water at study ponds on 21 November 2017.

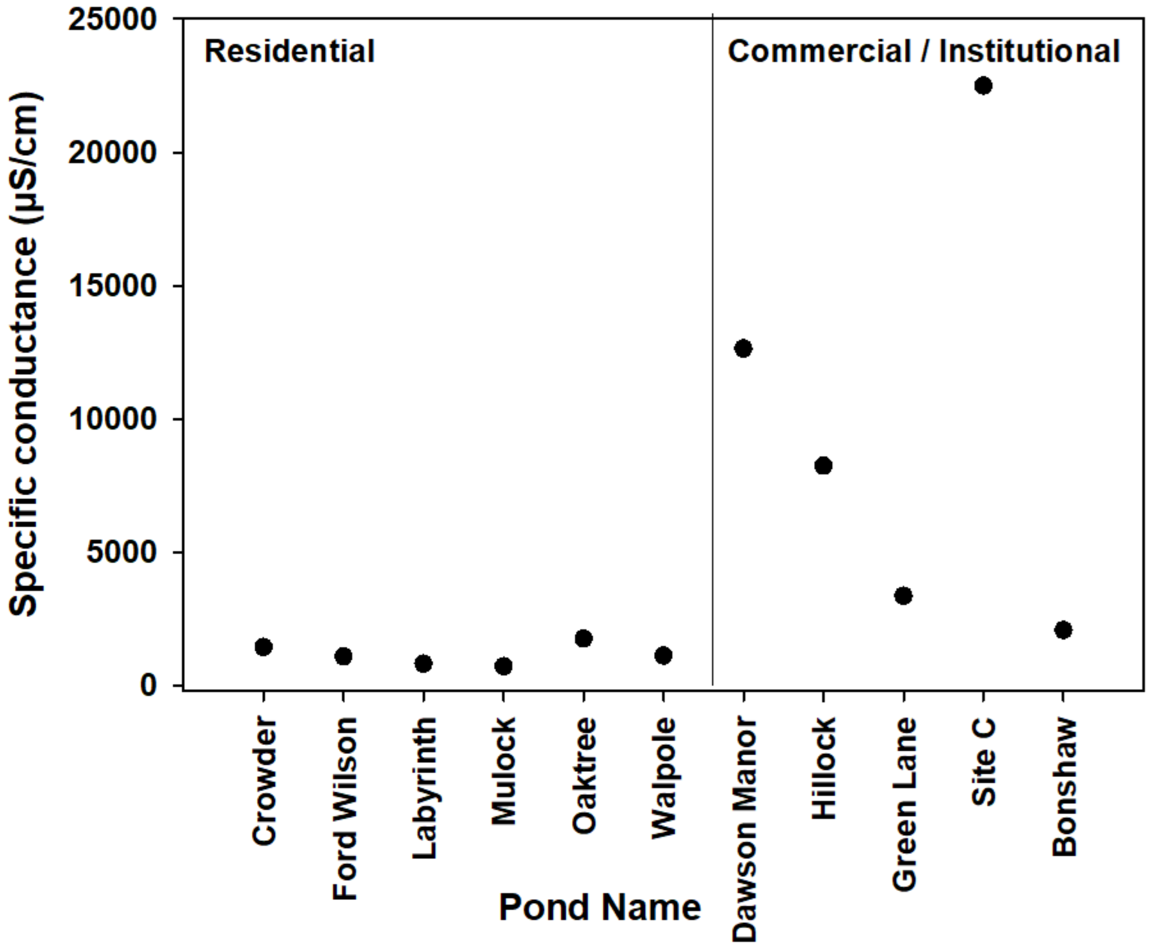

Figure 11. Mean (2018) specific conductance values recorded at stormwater pond study sites.

4.3.1 SAR results

Sodium Adsorption Ratio (SAR) is used as a guideline of the suitability of sediment for reuse on agricultural, residential or commercial / industrial lands. SAR was used in this study as an indicator of salt impact on SWMPs. In the graph below (Figure 12), ponds with commercial catchments had higher sediment SAR values then residential ponds. The red line indicates SAR = 12, the guideline for commercial / industrial lands application (CCME 1999). The SAR guideline for application on agricultural and residential soils, or parklands, is even lower at 5 (CCME 1999). In our study, five (of 11) ponds had annual mean SAR values that were above the commercial/industrial land application guideline. The ponds located in catchments with large paved areas (in particular Dawson Manor and Site C) had SAR values that were ~15X in exceedance of the CCME guideline value.

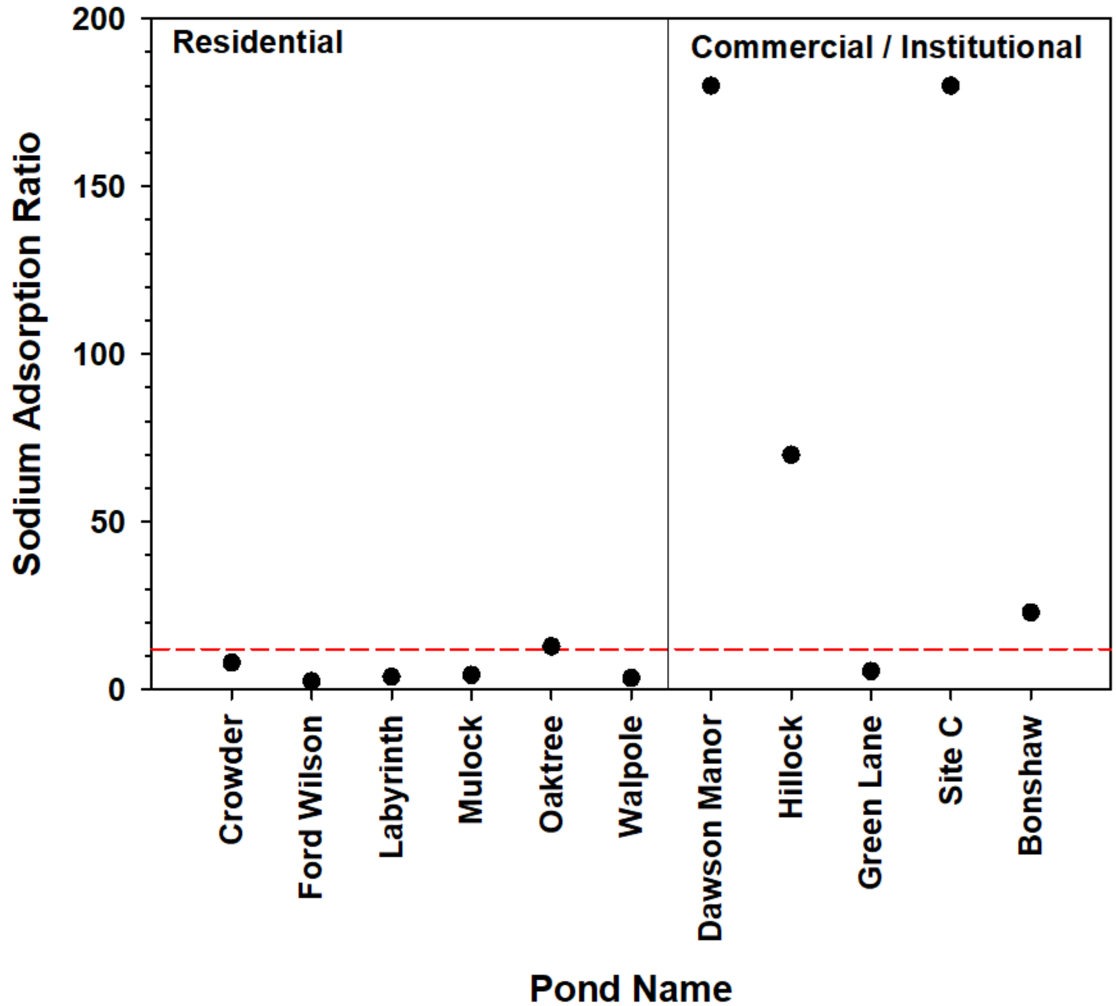

Figure 12. Mean Sodium Adsorption Ratio (SAR) records from stormwater pond sites. Values above 12 (horizontal red dashed line) are considered unsuitable for re-vegetation. This value (12) is the CCME guideline for commercial / industrial soil quality, SAR = 5 is the value for residential and agricultural soils (CCME 1999).

5.0 Detailed monitoring of three urban SWMP

To further investigate the functioning of SWMPs, loggers were deployed in three SWMPs: two commercial / institutional ponds (Site C and Hillock) and one residential pond (Oaktree). Loggers were deployed at the top and bottom of the water column in both the forebay and aftbay to record specific conductance, temperature, and dissolved oxygen during the ice-free season (April to November, set to record at 15 minute intervals) for 2015 to 2018. From these data, we hoped to gain a better understanding of seasonal trends in SWMPs, the impacts of salt inputs, the effects of precipitation inputs, and insight into environmental drivers that may impact pond functioning. Site C Pond was selected as the primary study pond as it had the largest area of impervious surfaces among the study ponds, was easy to access, and was of a size similar to larger commercial ponds. As a comparison, Hillock Pond, with a smaller commercial / institutional catchment, and Oaktree Pond, with a mostly residential catchment, were included. In addition to the study period described above, loggers were left in over winter (November to April) at Hillock Pond to capture trends during winter and early spring. Site C pond has a top draw outlet while both Hillock Pond and Oaktree Pond have bottom draw outlets.

5.1 Climate and weather considerations during the study period

The period of study (2015-2018) for this monitoring project captured a reasonable gradient of weather / climate conditions currently impacting southern Ontario, with a relatively warmer and drier year that included near-drought conditions in summer (2016, a year with strong El Niño) followed by one of the coolest, wettest years in recent memory (2017, a year with a La Niña). In the graphs below (Figures 14, 15), temperature and precipitation is presented for the periods June 9 – Dec 16, 2015 (days of year 160 – 350), May 3 – Nov. 14, 2016 (days of year 124 – 319), April 11 – Nov. 23, 2017 (days of year 101 - 327), April 24 – Nov. 14, 2018 (days of year 114 – 318). For comparative statistical analysis between the years, the period of common overlap of the four years was used.

In 2015, a relatively cool and wet spring became hot and dry in mid-July as an El Niño pattern developed. This El Niño persisted throughout 2016, giving the warmest and driest year of the study period (Table 2). In 2017, the presence of a La Niña pattern gave a very wet and relatively cool year that was a marked contrast to 2016. 2018 was categorized as a “neutral” year (neither an El Niño or La Niña pattern was present) but was still wetter and cooler than 2015 or 2016.

Despite the differences in climate patterns between the years, trends were relatively similar throughout the study period. As such, environmental drivers such as air temperature and annual rainfall amount were not found to greatly influence the patterns observed in pond functioning. However, the intensity and frequency of some summer precipitation events did influence water column stratification in the ponds, and the frequency of mixing. In addition, early snowfalls and the period of application of winter de-icing salt did determine the strength of salt-related chemical stratification in the following spring.

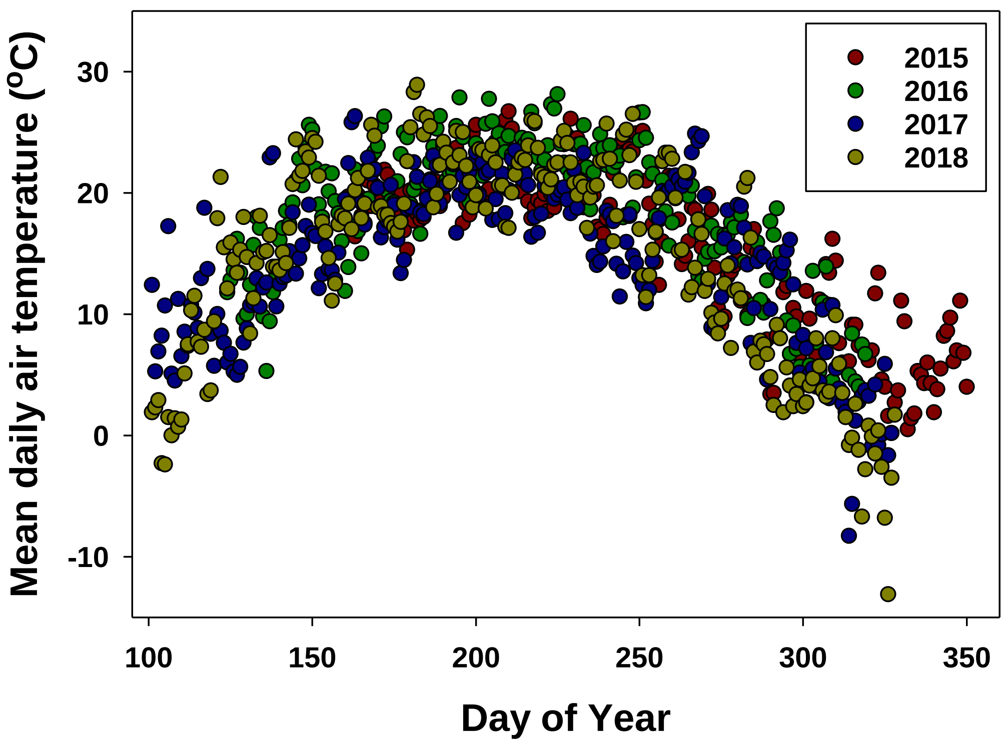

Figure 14. Mean daily air temperatures recorded at the LSRCA monitoring station in Newmarket, Ontario for the study period: 9 June – 16 Dec 2015, 3 May – 14 Nov 2016, 11 April – 23 Nov 2017, 24 April - 14 Nov 2018.

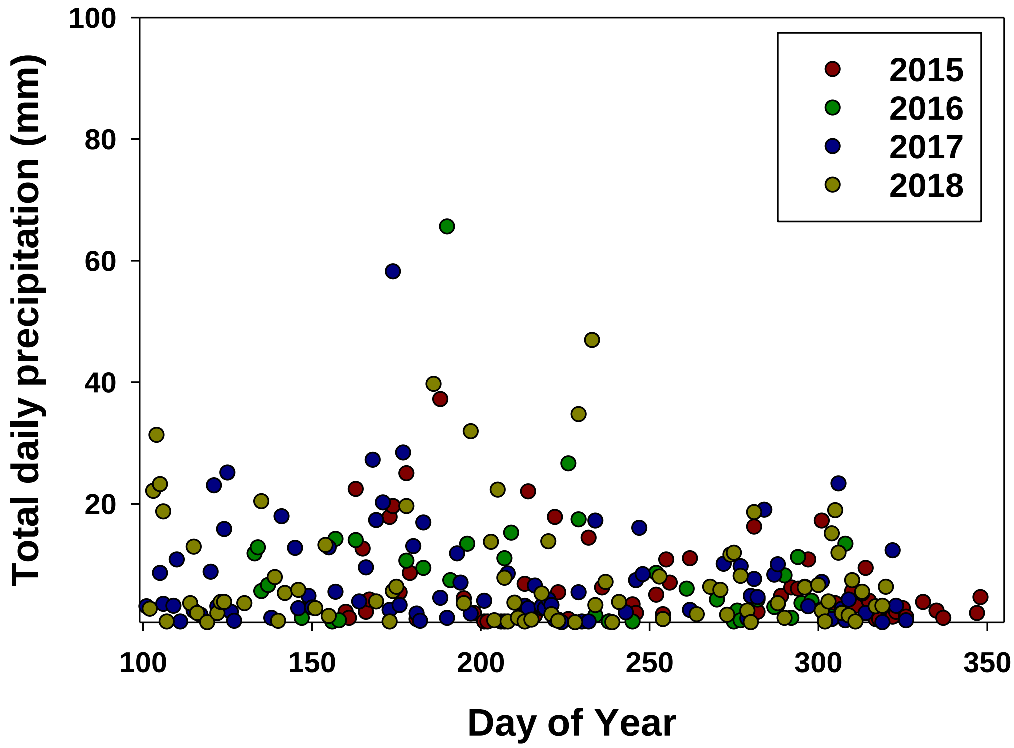

Figure 15. Total daily precipitation accumulated at the LSRCA monitoring station in Newmarket, Ontario for the study period: 9 June – 16 Dec 2015, 3 May – 14 Nov 2016, 11 April – 23 Nov 2017, 24 April – 14 Nov 2018.

Table 2. Comparison of precipitation and air temperatures recorded at the LSRCA monitoring station in Newmarket, Ontario for the study dates common between all years (day of year 101 to 318): 11 April – 14 November 2015, 2017, 2018; 10 April – 13 November 2016.

	2015	2016	2017	2018
Total precipitation (mm)	497.4	467.1	639.7	642.4
Maximum daily precipitation (mm)	37.2	65.6	58.2	46.9
Mean daily air temperature (°C)	16.7	17.5	15.3	15.5
Maximum daily air temperature (°C)	26.7	28.1	26.3	28.9
Minimum daily air temperature (°C)	2.1	2.7	-8.3	-6.7

5.2 Temperature stratification

All three SWMPs (Site C, Hillock, and Oaktree) showed thermal patterns typical with seasonality in the Lake Simcoe Watershed (Figs. 16-21). Cooler temperatures were prevalent in early spring and late fall, with both surface and bottom waters in the forebays and aftbays warming from late spring to early autumn. Although the intent of SWMP design is to maintain relatively cooler bottom temperatures, particularly in bottom-draw designs such as Hillock and Oaktree ponds that discharge to cold- / cool-water habitats in tributaries, all three ponds with continual data logging had bottom waters that exceeded 19^{o}C during the summer. With intense rain events, this relatively warm bottom water would be discharged into cold- and coolwater habitats, possibly causing thermal shock for any sensitive organisms present (e.g. brook trout, mottled sculpin, endangered redside dace, etc.). Additionally, as discussed in detail below, these bottom waters may also have low dissolved oxygen (DO) concentrations and high specific conductance (used as an inference of salinity, or chloride concentrations) that could amplify the effect of temperature and decrease habitat quality due to multiple, synergistic stressors.

Temperature trends were analyzed using a one-way analysis of variance (ANOVA) to compare forebay vs. aftbay, across the three detailed study ponds, and between the four monitored years. The analysis showed statistically significant differences ($p<0.05$ to $p<0.001$) between forebay and aftbay surface waters and bottom waters. There were no significant differences in temperature data between the pond with a predominantly residential use catchment (Oaktree) and those with commercial uses (Site C and Hillock). Between years, Site C had no significant differences between data recorded in 2015, 2017, and 2018 (the cooler, wetter, La Niña years), but temperatures in 2016 were significantly different ($p<0.001$) from the other three years. Temperature data from Hillock and Oaktree ponds had significant differences ($p<0.05$) between all years.

In a study of ponds from central Ontario, Song et al. (2013) found that despite SWMPs being designed with shallow water depths to inhibit thermal stratification of the water column, prolonged periods of stratification still occur and enables dissolved P to be released from pond sediments. This stratification and periodic mixing alters the timing and quantity of nutrient loading and can potentially affect water quality and biological communities in receiving waters.

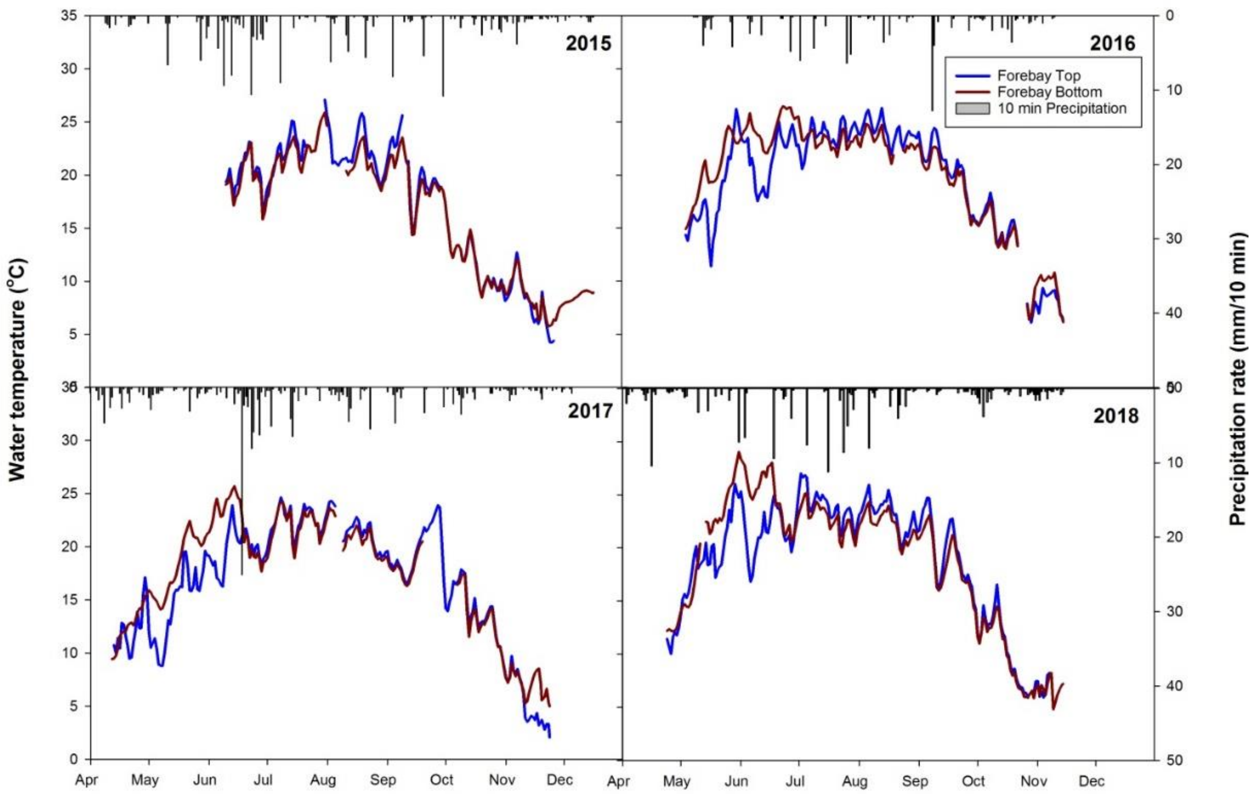

Figure 16. Mean daily water temperature recorded at the forebay of Site C Pond, for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

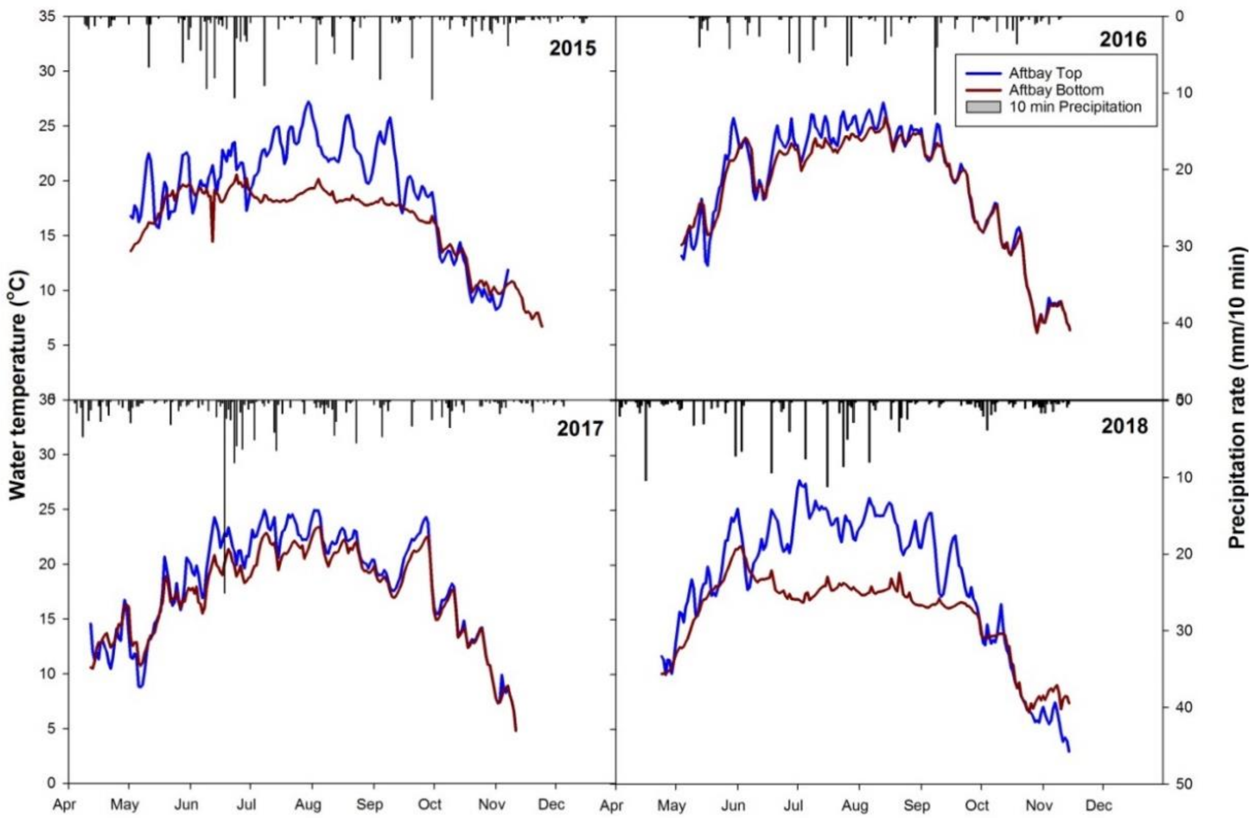

Figure 17. Mean daily water temperature recorded at the aftbay of Site C Pond, for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

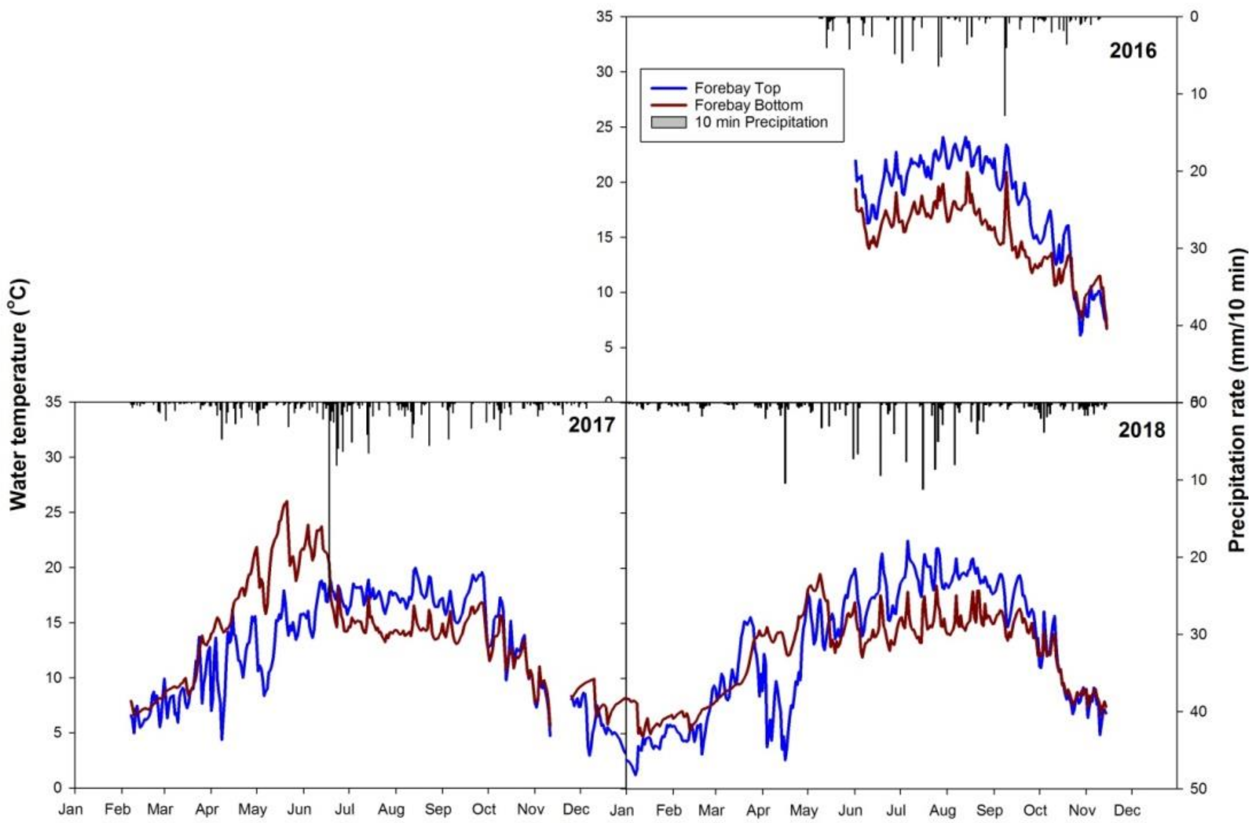

Figure 18. Mean daily water temperature recorded at the forebay of Hillock Pond, for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

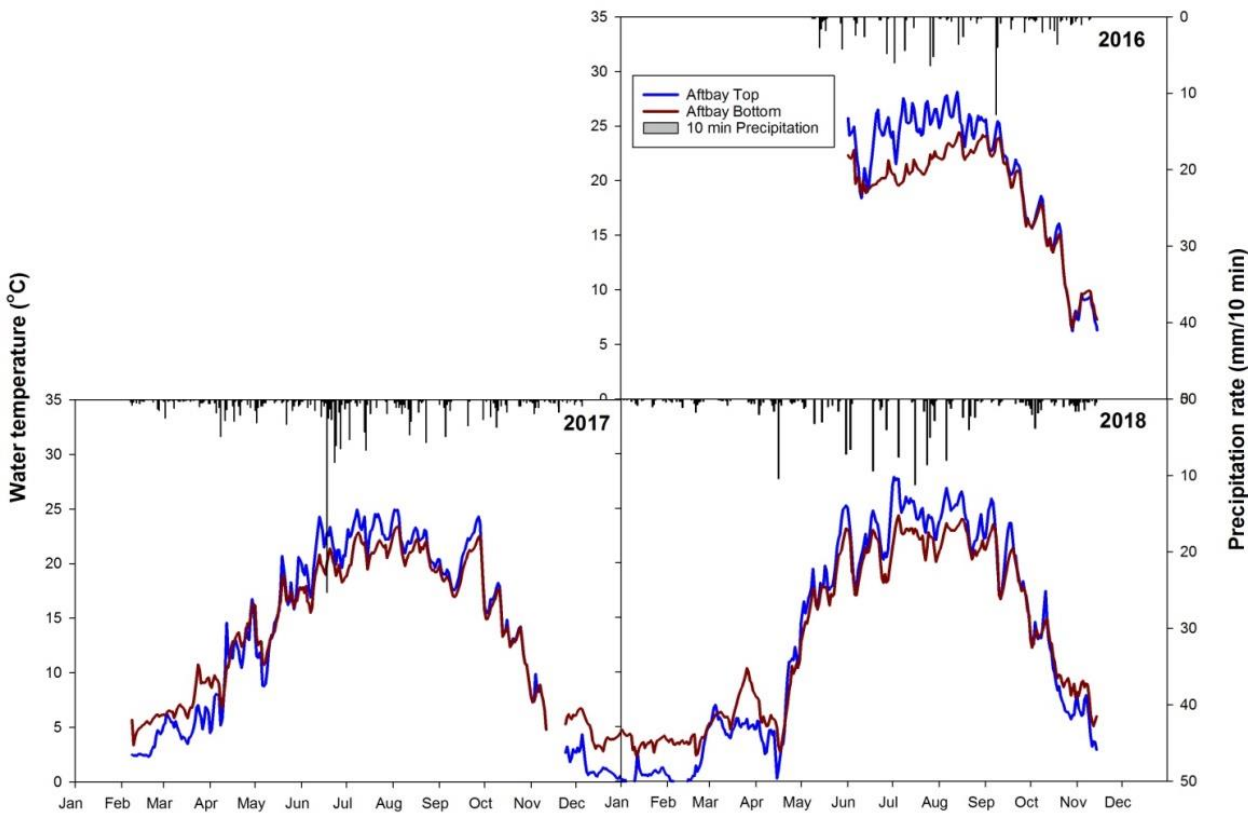

Figure 19. Mean daily water temperature recorded at the aftbay of Hillock Pond, for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

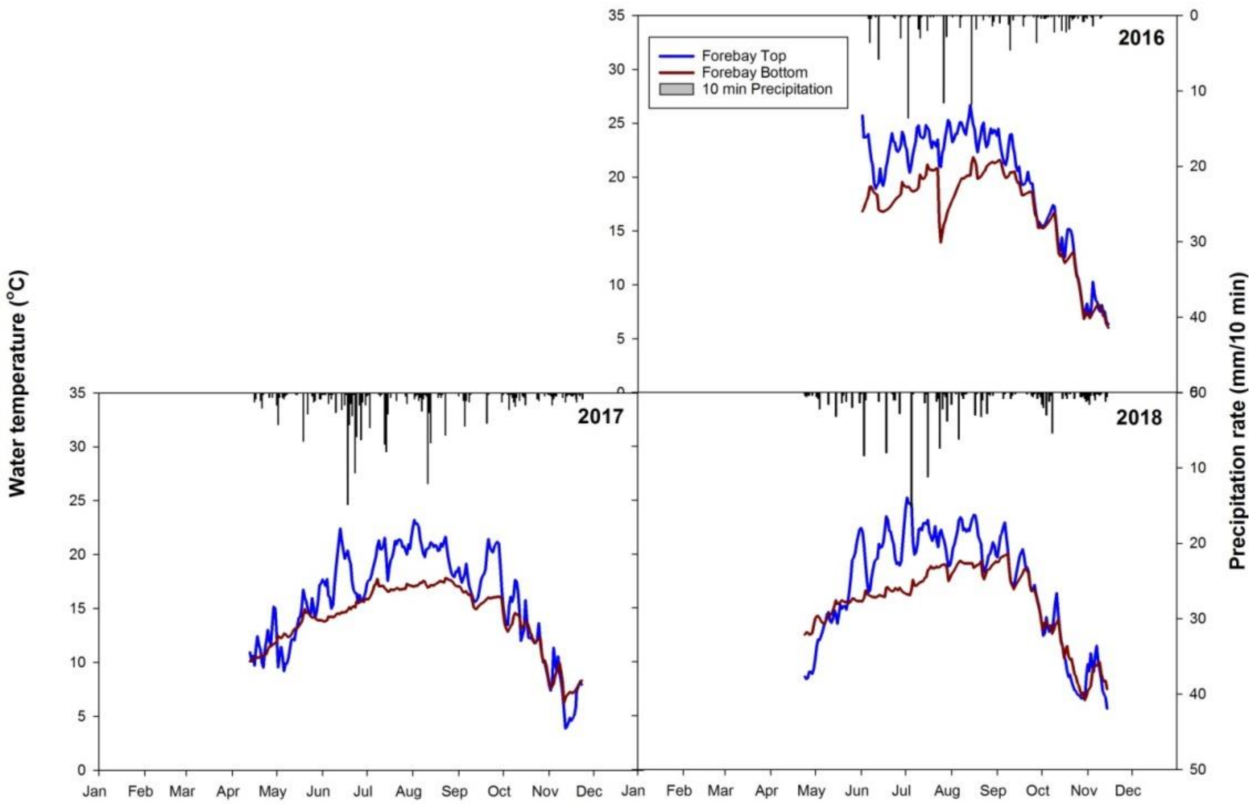

Figure 20. Mean daily water temperature recorded at the forebay of Oaktree Pond, for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

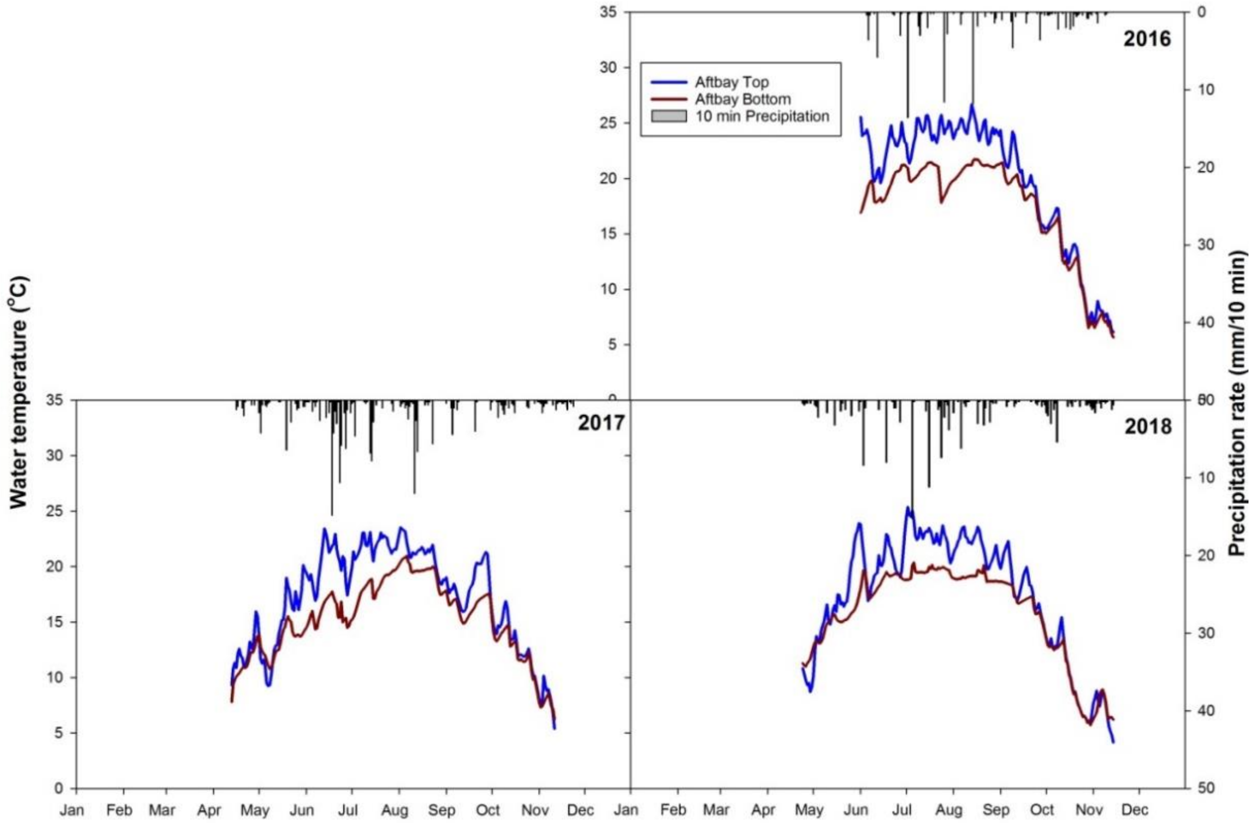

Figure 21. Mean daily water temperature recorded at the aftbay of Oaktree Pond, for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

5.3 Dissolved oxygen stratification

Trends in dissolved oxygen were recorded in the three ponds for 2015-2018 (Site C Pond, Figures 22-23) and 2016-2018 (Hillock and Oaktree ponds, Figures 24-27). In general, all three ponds had periods of low dissolved oxygen in bottom waters, likely due to prolonged thermal and chemical stratification of the water column. In the two commercial ponds, these low oxygen periods were mostly limited to early spring, due to aquatic plant growth in later spring and summer that supplied oxygen to the bottom waters. Statistical analyses showed significant ($p<0.05$) differences between forebay and aftbay top samples and between bottom samples in all three ponds, likely due to flushing by incoming water. As discussed in detail below, the desired environmental conditions in SWMPs (i.e. conditions assumed in pond design specifications) seem to be related to how frequently the water column of the pond mixes and limiting the length of time the water column is stratified. All three ponds had statistically significant ($p<0.01$) differences in DO between years, with the exception of the Site C aftbay where 2015 was not different from 2016 or 2017 (but 2016 and 2017 were significantly different from each other ($p<0.001$)).

At Site C Pond, purple sulphur bacteria, an extremophile typically found in low oxygen saline environments, was recorded in the early spring of 2016, 2017, and 2018 during low DO conditions. With seasonal succession of the flora to submersed aquatic plants, DO conditions improved and showed diurnal fluctuations typical of natural ponds and lakes (Figures 22-23). Aquatic plants were recorded mostly in the forebay and were typically pondweeds (*Potamogeton* spp.) and pond lilies (*Nuphar* sp., *Nymphaea* sp.) with a high percentage of invasive species (curly-leaf pondweed, *Potamogeton crispus*; starry stonewort *Nitellopsis obtusa*) likely transported to the ponds by waterfowl (Pullman and Crawford 2010) as boats and trailers, the most common method of spreading invasive species, are not used in these small SWMPs. The lower amount of plants in the aftbay is likely the cause of longer periods of low DO here, compared to the forebay.

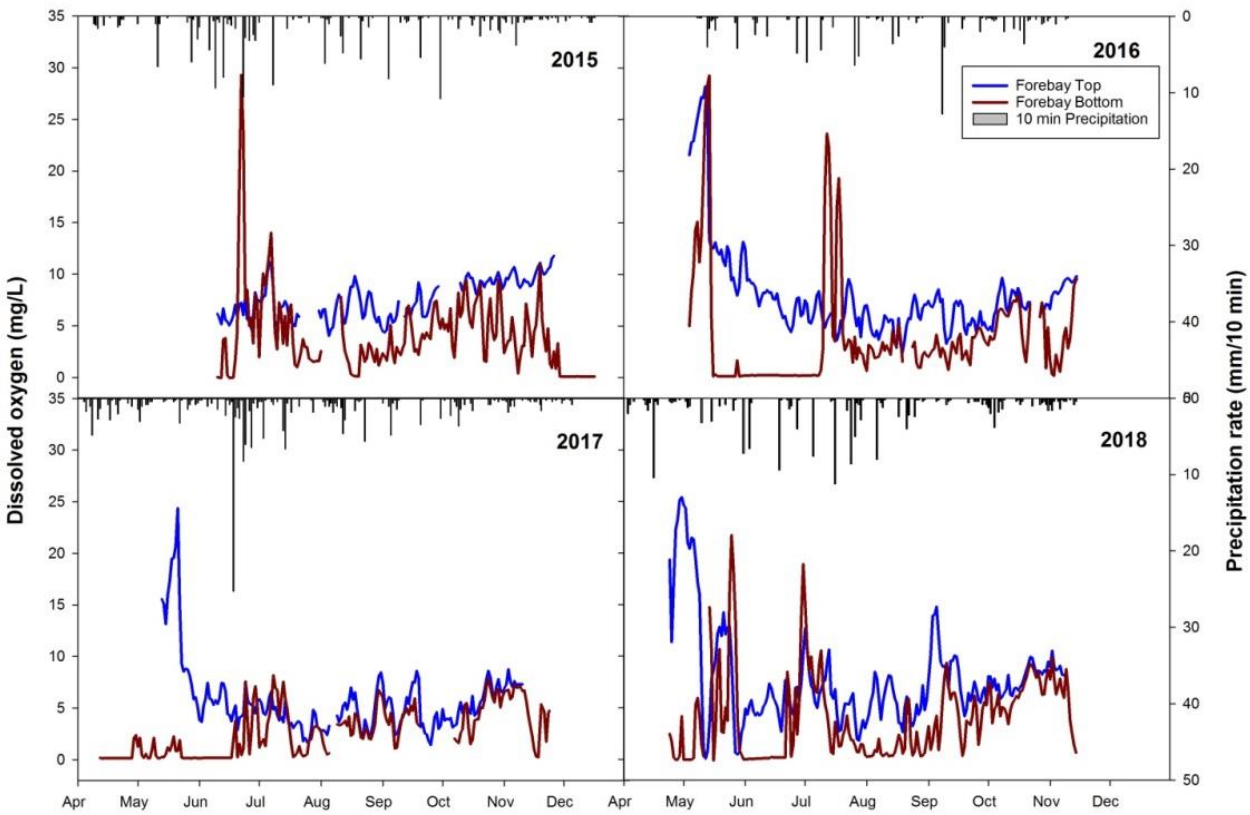

Figure 22. Mean daily dissolved oxygen recorded at the forebay of Site C Pond, for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

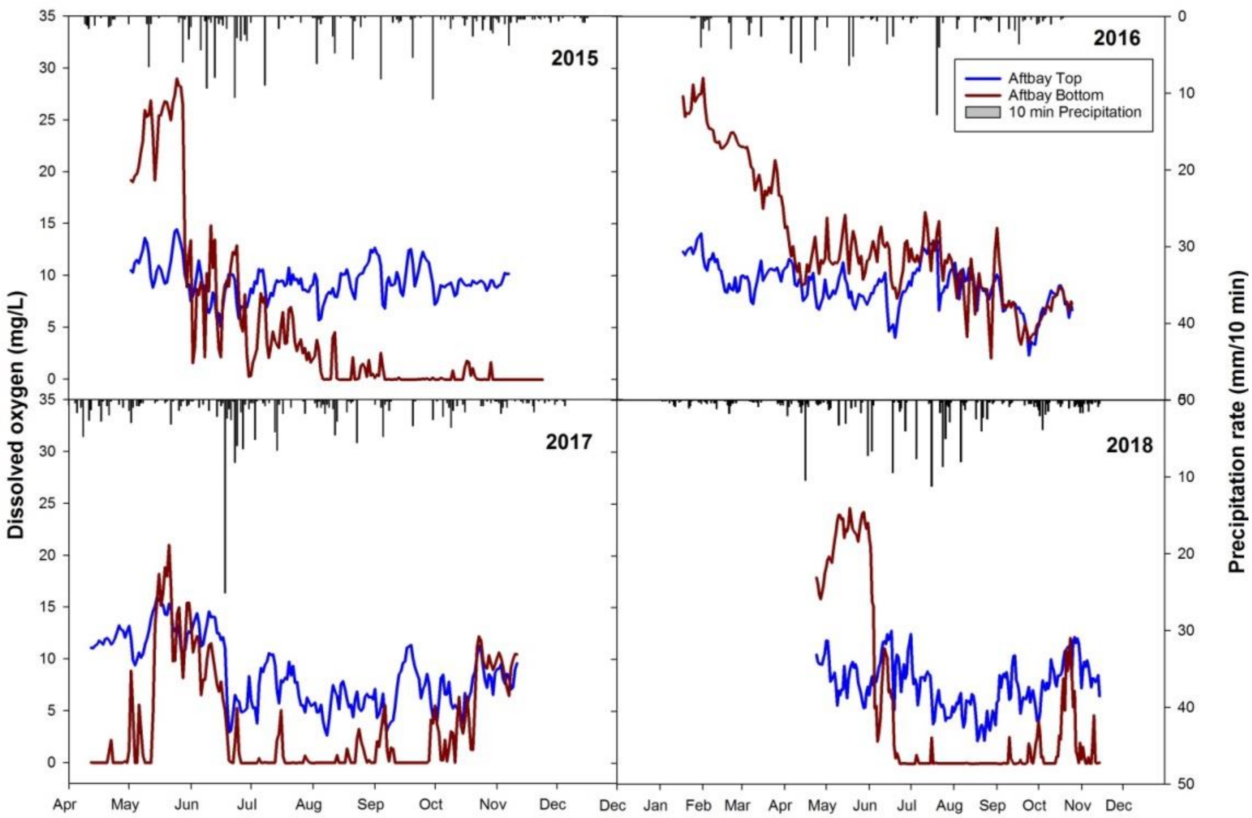

Figure 23. Mean daily dissolved oxygen recorded at the aftbay of Site C Pond, for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

5.3.1. Dissolved oxygen in a smaller commercial pond

Dissolved oxygen conditions in Hillock Pond were similar to Site C Pond, with low DO conditions in spring, the presence of purple sulphur bacteria, and aquatic plants supplying oxygen to the bottom waters later in spring and throughout the period of water column stratification (Figures 24-25). As in Site C Pond, there were diurnal fluctuations in DO related to plant photosynthesis during the day, and respiration at night. Hillock Pond also contained invasive plants (curly-leaf pondweed and starry stonewort), with the highest densities found in the aft bay. Again, the aftbay here showed longer periods of low DO in the bottom water in 2016 and 2017, compared to the forebay.

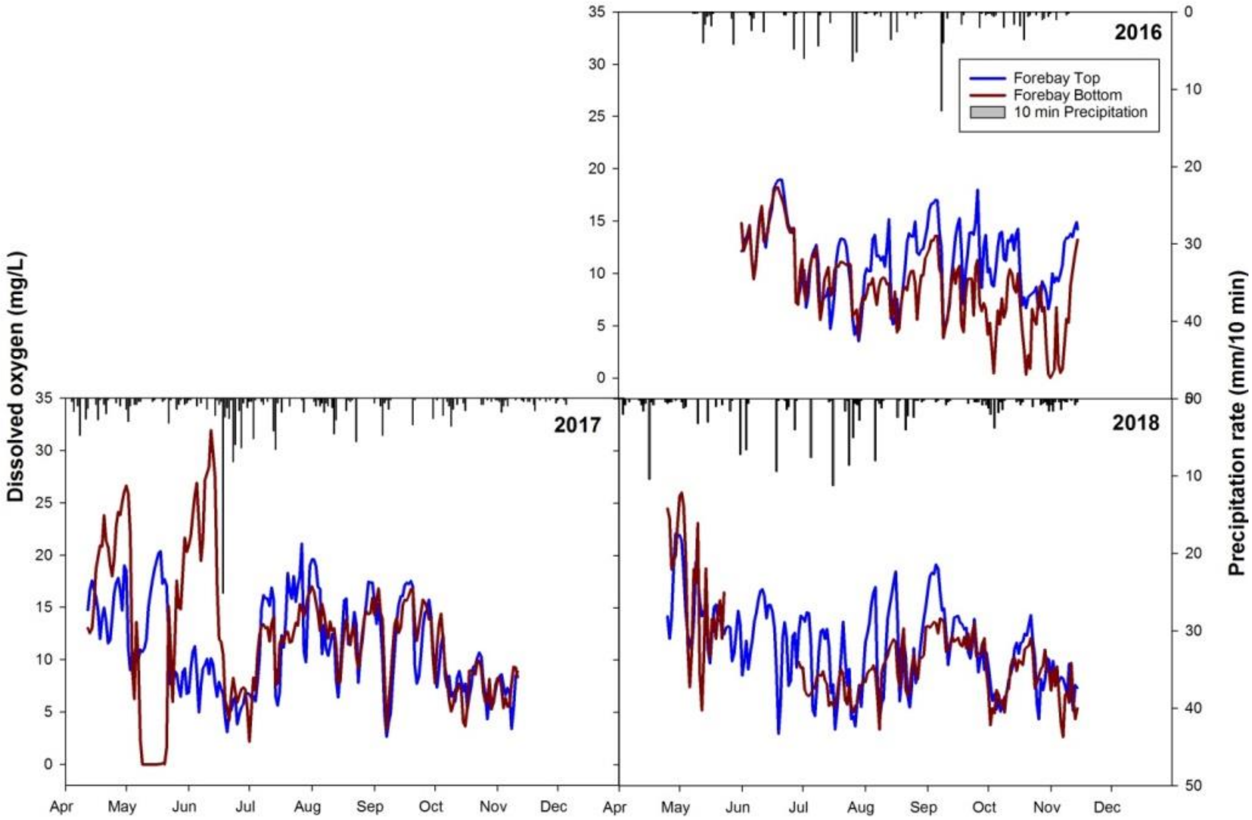

Figure 24. Mean daily dissolved oxygen recorded at the forebay of Hillock Pond (smaller commercial catchment), for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

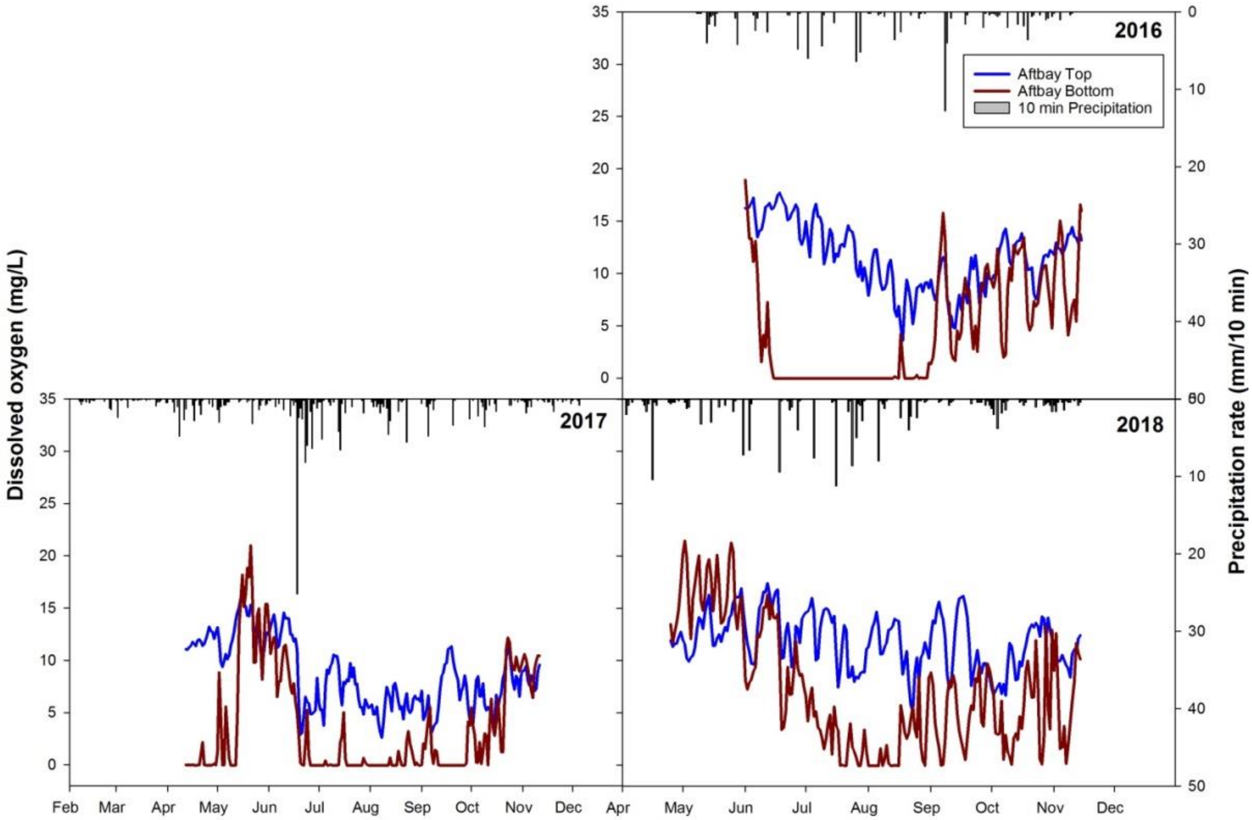

Figure 25. Mean daily dissolved oxygen recorded at the aftbay of Hillock Pond (smaller commercial catchment), for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

5.3.2. Dissolved oxygen in a residential pond

Oaktree Pond, with a predominantly residential catchment, showed a different pattern in dissolved oxygen compared to the two commercial ponds. After high DO conditions in spring, likely due to the pond mixing from water inputs, both the forebay and aftbays had extended periods of low bottom DO conditions (Figures 26-27). Unlike the two commercial ponds above, Oaktree Pond has a higher turbidity, which limited light penetration to the bottom and prevented plant growth, leading to dominance by floating plants (e.g. pond lilies and duckweed) and algae. Without the bottom water oxygen supplied by submerged aquatic plants, as in the two commercial ponds, water column stratification and infrequent mixing resulted in low DO conditions. Inputs of precipitation that mix the pond do result in an increase in bottom DO, particularly in the forebay, however this oxygen is rapidly used up by biogeochemical processes.

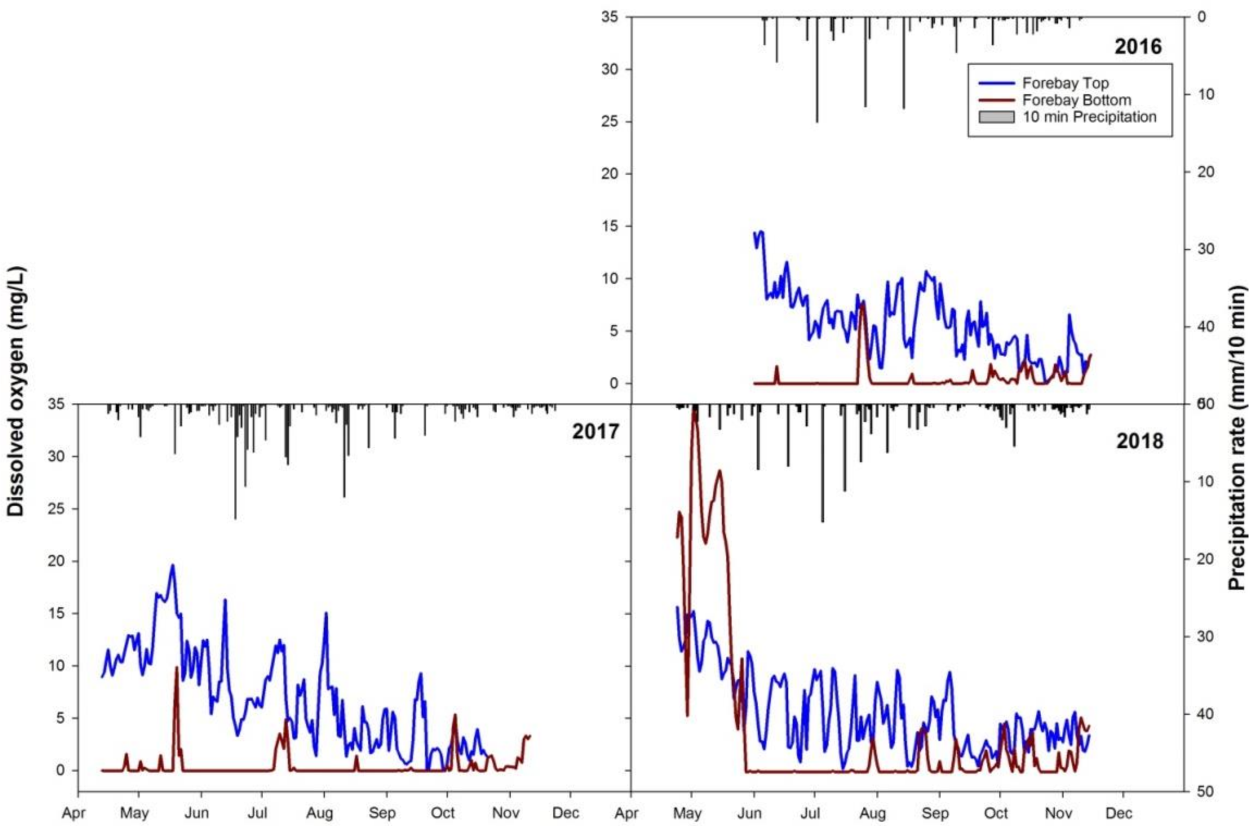

Figure 26. Mean daily dissolved oxygen recorded at the forebay of Oaktree Pond (residential catchment), for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

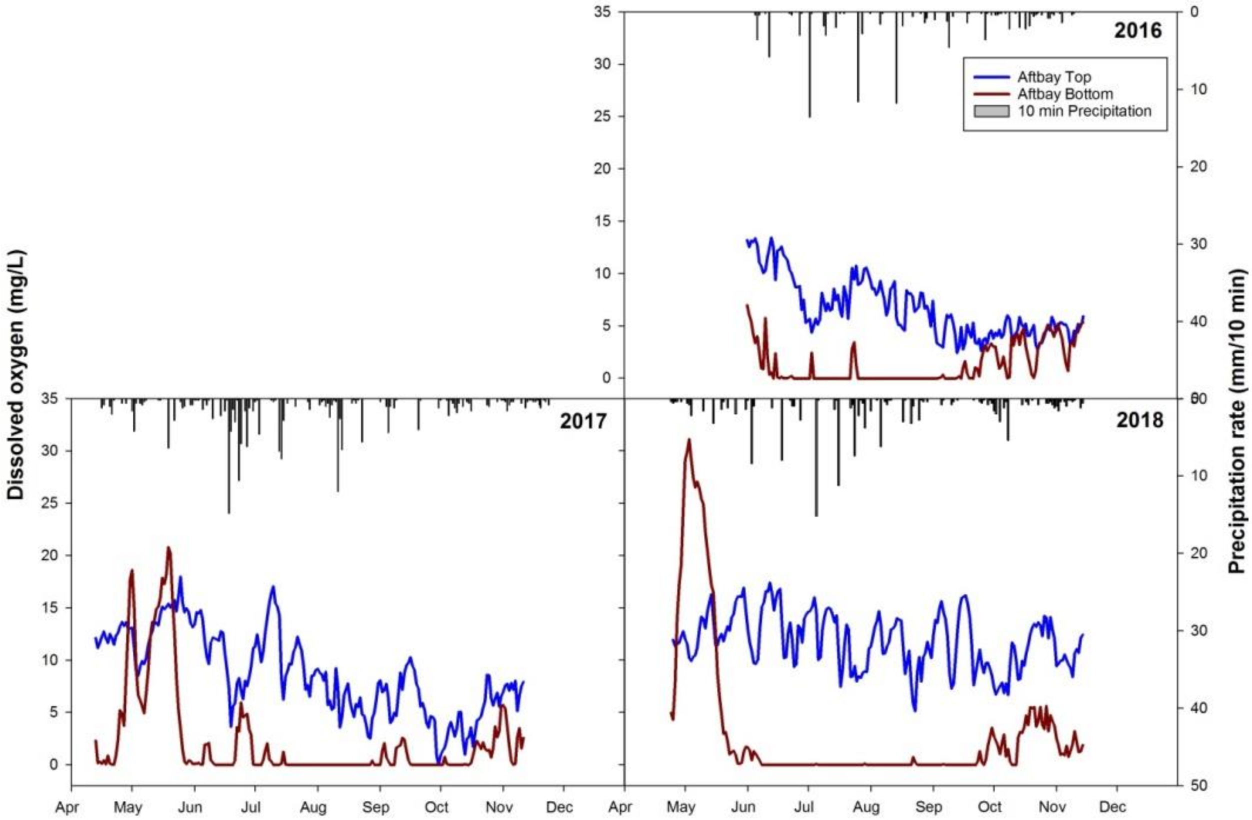

Figure 27. Mean daily dissolved oxygen recorded at the aftbay of Oaktree Pond (residential catchment), for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

5.4 Specific Conductance / Salinity stratification

Specific conductance was used as a proxy indicator of salinity / chloride concentrations. As stated above, measures of salinity (specific conductance, salinity, and chloride concentrations) are the main environmental drivers of conditions within the SWMPs studied across the Lake Simcoe Watershed. This circumstance was partly expected as the majority of ponds are located in urban areas, which receive run-off from impervious (paved) surfaces, and de-icing salts are widely used during the winter months. As the highly saline surface run-off entering the ponds has a higher density than typical freshwater, it sinks to the bottom of the ponds, and produces a layer of highly saline water. We have recorded chloride concentrations up to 39,345 mg Cl^-/L (or over 2X that of seawater; ~19,000 mg Cl^-/L) in commercial SWMPs (Hillock Pond in spring 2017); well above the Canadian chloride guideline values of 120 mg Cl^-/L for chronic toxicity and 640 mg Cl^-/L for acute toxicity to aquatic organisms (CCME, 1999). The density difference between a freshwater upper layer and the saline bottom water limits the ability of the SWMP to fully mix (as was assumed in the design) and impairs the functioning and efficiency of the pond. Large inputs of freshwater from intense rain events can, however, provide enough water volume to mix the water column of the SWMP, as discussed in detail below.

Some trends in specific conductance (hence salinity) were as expected. Ponds (Site C and Hillock) with commercial catchments (and large areas of impervious surfaces, such as parking lots, where winter de-icing salts are typically applied more heavily than on roads) have higher specific conductance values than ponds with residential catchments (Oaktree Pond), where there are less impervious surfaces and lower de-icing salt application rates. Although all three ponds had elevated specific conductance (compared to more natural conditions in the watershed ~300-400 µS/cm), particularly in the forebay in spring when de-icing salts get washed off pavement and into SWMPs by rainfall and snowmelt. For the rest of the year, rainfall inputs reduce salinity until late fall when winter salt is again applied and specific conductance values begin to increase. Surface waters of the ponds typically have reduced salinity earlier in the year due to spring runoff and regular rainfall inputs. Bottom waters, however, have more persistent elevated chloride concentrations and the water column is resistant to mixing due to the density difference between fresh and salt-laden water. This resistance to mixing is particularly pronounced in the aftbay where the velocity of stormwater inputs have been slowed, as designed to occur in the pond, to promote settling of suspended solids. High salinities in the aftbay can persist as late as August or September, especially in years with reduced precipitation amounts (e.g. 2016). As described in more detail below, the force of inflowing water is likely responsible for mixing the water column and eventually diluting the salinity in the forebay, whereas overall precipitation volume likely dilutes salinities in the aftbay.

5.4.1 Specific conductance trends in a large commercial pond

Trends in specific conductance varied between the three ponds. Forebay and aftbay, top and bottom, respectively, water samples were significantly different from each other ($p<0.001$) with the exception of top water samples from the fore- and aftbays of Site C (Figures 28-29). There were also significant differences ($p<0.001$) in each monitored section (forebay top and bottom, aftbay top and bottom) between years, with 2016 (the low precipitation, relatively warm, year) being set apart from 2015 and 2017. At Site C, the forebay bottom waters are highly saline (specific conductance = 40,000-65,000 µS/cm) in spring due to applications of winter-deicing salt on a large parking lot, and very saline surface run-off after rainfall and snowmelt (Figure 28), which forms a layer of very dense, salty, bottom water.

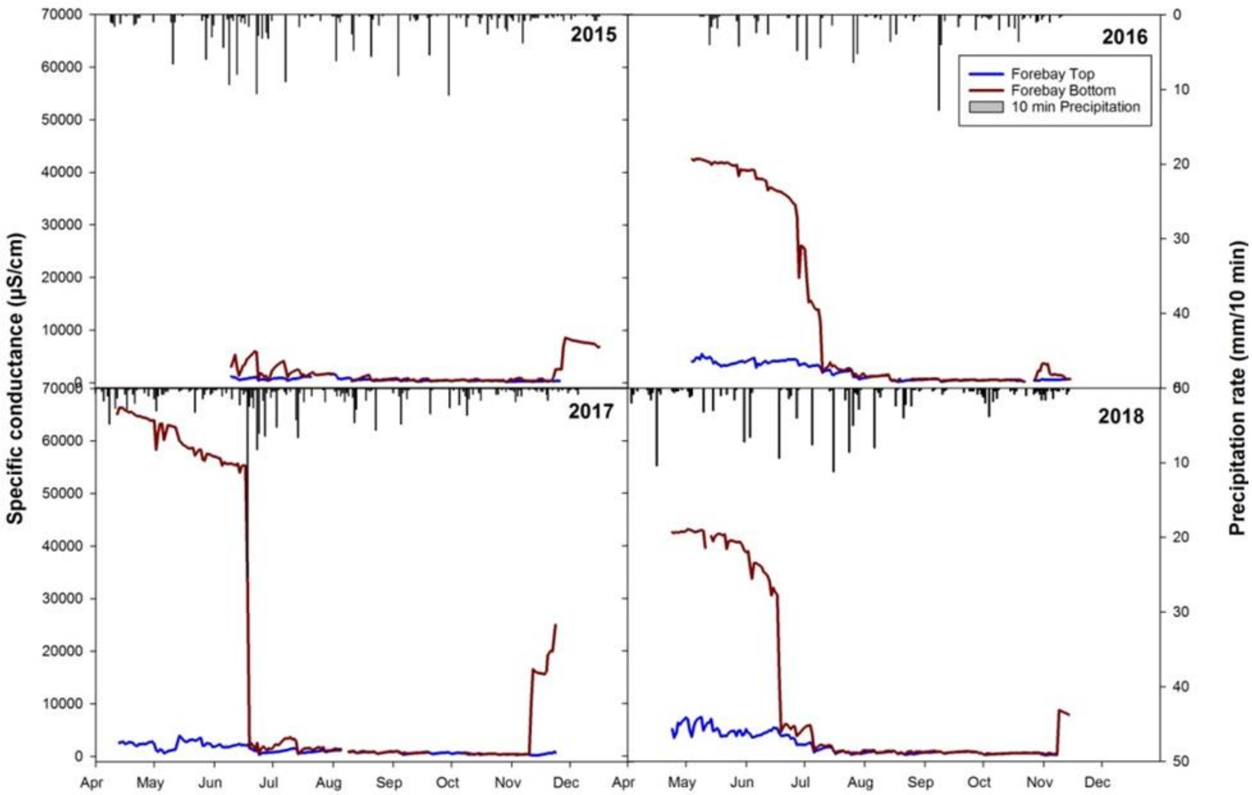

Figure 28. Mean daily specific conductance recorded at the forebay of Site C Pond, for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

Although the specific conductance of the bottom water slowly decreases during spring, each year shows a very sudden drop in specific conductance caused by a rainfall event in mid-June. The exception to this trend was 2015, when the sudden decrease in specific conductance likely occurred before loggers were deployed. Analysis of precipitation data indicate that rainfall intensity, rather than total volume, being the cause of this sudden decrease in specific conductance. It is estimated, based on our monitoring, that a rainfall intensity of ~5 mm accumulation in a 10-minute period is needed to mix the Site C forebay water column, and flush the saline layer. Further analysis of this data is described in Section 5.5-5.5.1, below. After this decrease in specific conductance, values recorded in the forebay bottom waters are similar in value to those in the top layer, until the application of winter salt begins in November.

At the Site C aftbay, there is a similar pattern to the forebay with bottom waters being more saline, but a more gradual decrease (except in 2017) than the sudden change recorded in the forebay

(Figure 29). This trend indicates that rather than the intensity of rainfall driving change (as seen in the forebay), it is likely that a certain volume of freshwater inputs needs to be achieved, after forebay mixing, to flush out the aftbay salt layer. In 2017, the sudden decrease in the aftbay was likely caused by a large rainfall event (42.0 mm, 4-5 May 2017) that was enough to mix and flush the aftbay, even though the intensity (0.8 mm in 10 min) was low and did not impact the forebay. As discussed below (Section 5.5), this situation would suggest that the water column of the forebay was so strongly salt-stratified that inputs from this event slid over top of the forebay bottom waters and entered the aftbay, where the volume mixed the water column and flushed the aftbay water. In order to more fully investigate this phenomenon, a water column thermal stability model was adapted for use with salt, and applied to SWMPs, as discussed in detail in Section 5.5-5.5.1.

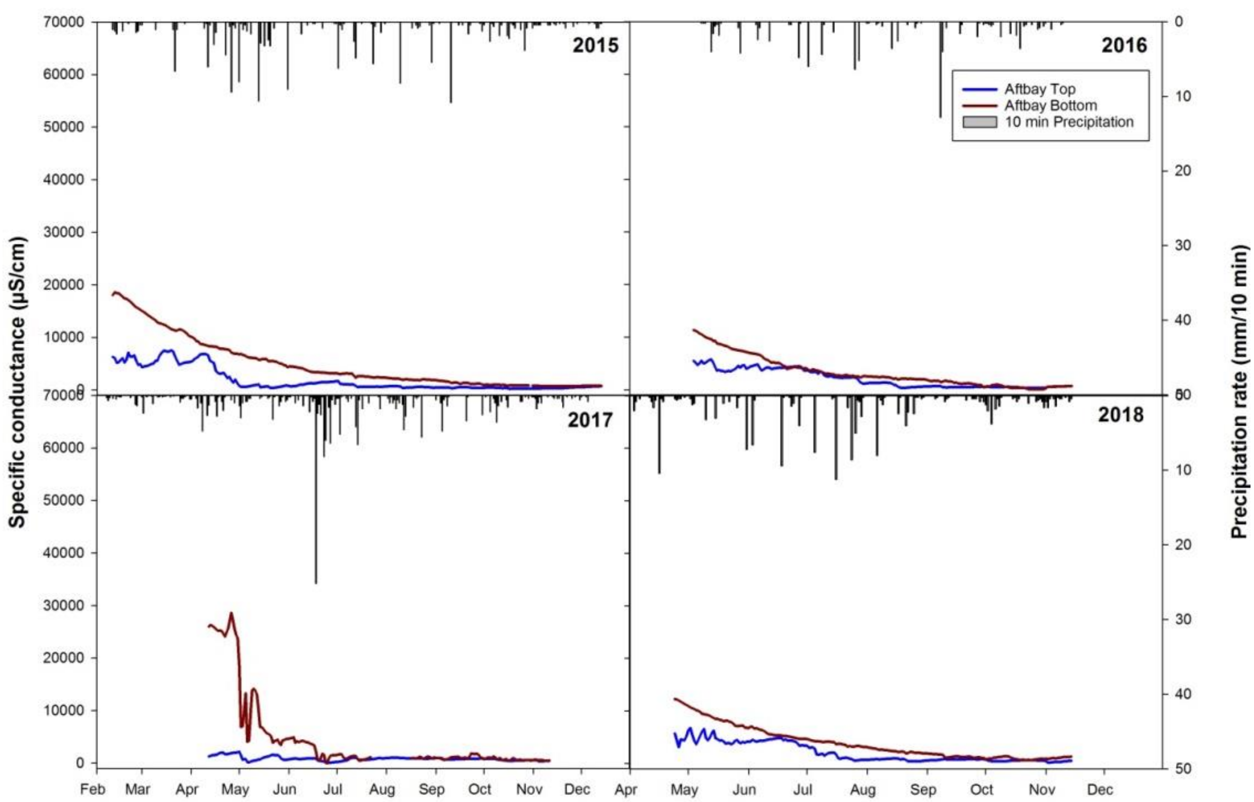

Figure 29. Mean daily specific conductance recorded at the aftbay of Site C Pond, for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

5.4.2 Specific conductance trends in a smaller commercial pond

Trends in Hillock Pond were similar to those observed at Site C, although the delayed start to monitoring in 2016 likely missed high specific conductances recorded that spring (Figures 30-31). Again, the forebay had higher specific conductance than the aft bay with precipitation eventually mixing the forebay water column in spring and flushing out higher salt content water. As recorded at Site C Pond, specific conductances of the top and bottom waters in both the forebay and the aftbay remain similar until increasing in late fall after the first applications of winter salt. As the loggers were left in the pond over the winter of 2017-18, some interesting trends were observed. In the forebay (Figure 30), the highest specific conductance values were recorded in spring 2017, but did not occur until mid-April, likely coinciding with snowmelt that transported salt to the pond. A similar pattern was recorded for the winter 2017-18, with specific conductance increasing with the application of de-icing salt over the winter and a short-lived peak in late March, likely coinciding with snowmelt. The spring 2017 peak had the highest specific conductance recorded at any pond during the multi-year study. Assuming deicing rates and practices at Hillock Pond are similar to those at Site C, the winter applications before the 2017 peak (November 2016 to March 2017) had more salt applied (895 tonnes at Site C) compared to 2015-16 (477 tonnes at Site C). However, the 2017-18 application period at Site C had an even higher application amount (1,012 tonnes). Based on the sharp peak in spring 2017, compared to 2018, it appears spring 2017 had a delayed (late-April to mid-May) but rapid snowmelt period, compared to 2018 (late March). The 2017 increase and peak in specific conductance occurred independently of precipitation inputs. In the forebay of Hillock Pond, mixing of the water column in 2017 and 2018 coincided with rainfall events with an intensity of ~3 mm in 10 minutes. Less intense precipitation was required to mix this water column, even though mean spring specific conductance values are similar to Site C, because the pond is of smaller volume.

In the aftbay, a seasonal pattern in specific conductance similar to the forebay was recorded (Figure 31) although, like Site C, the specific conductance values are lower in comparison with the forebay. Likely related to the smaller size of Hillock Pond, water column mixing and flushing of the saline bottom waters occurred at approximately the same time as the forebay, unlike Site C where a large difference in timing was recorded.

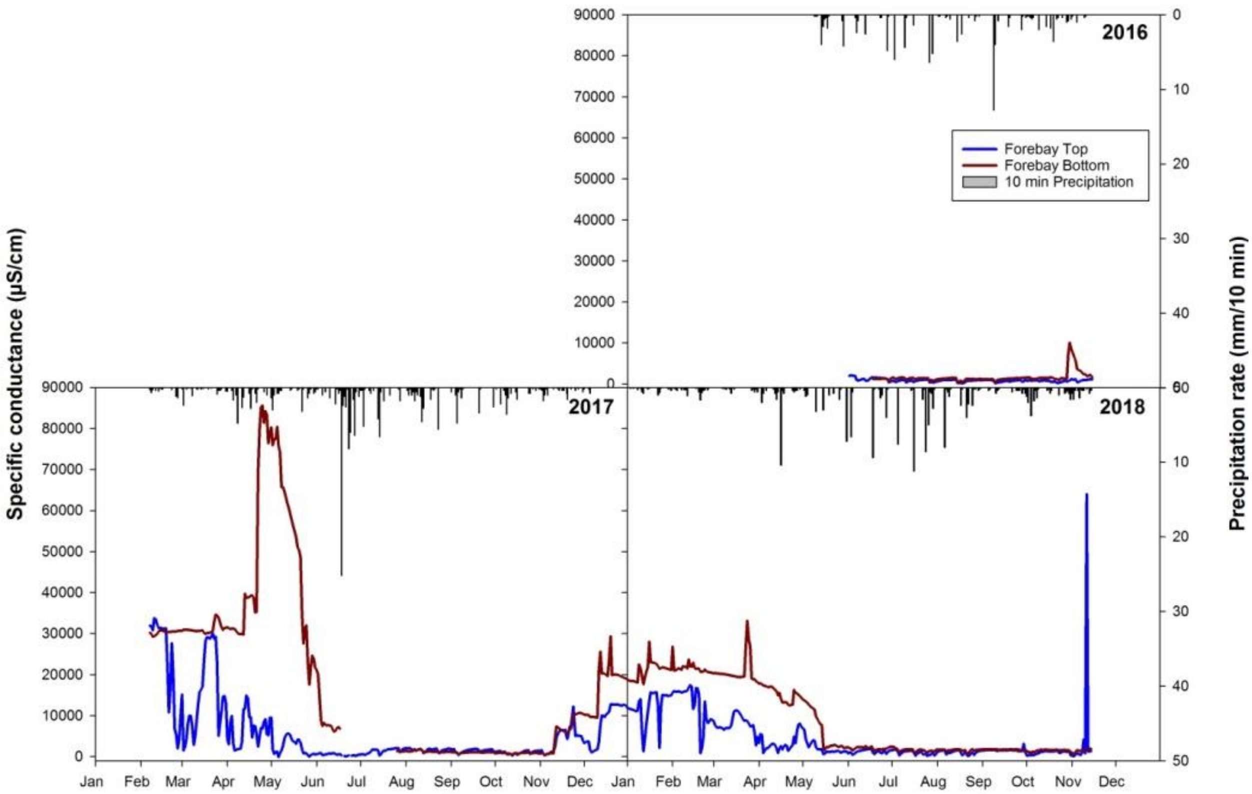

Figure 30. Mean daily specific conductance recorded at the forebay of Hillock Pond (smaller commercial catchment), for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).. Note change in y-axis scale to 90,000 µS/cm.

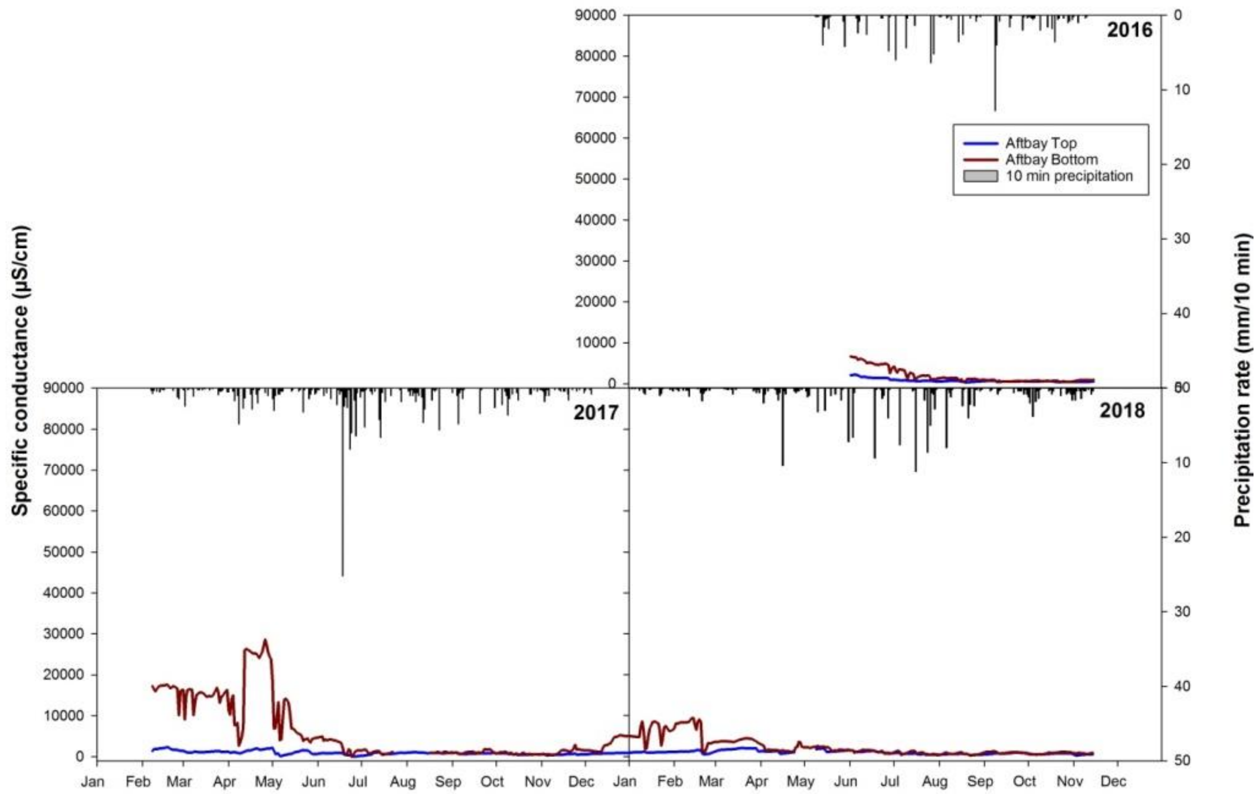

Figure 31. Mean daily specific conductance recorded at the aftbay of Hillock Pond (smaller commercial catchment), for the top (top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

5.4.3 Specific conductance trends in a residential pond

A residential catchment requires less winter salt application, having a lower area of impervious surfaces and lower traffic volume. As a result, Oaktree Pond had much lower specific conductance recordings than the two commercial ponds. Even with a relatively lower salt concentration in this residential pond, the patterns observed were similar to the commercial ponds. Higher specific conductances were recorded in the forebay, compared to the aftbay, with a similar pattern of declining salt concentrations due to precipitation inputs through the summer month, then increases following winter salt applications with late fall snow. In contrast to the commercial ponds, which had a sudden mixing of the water column that correlated to an intense rainfall event, Oaktree Pond seemed more resistant to mixing, likely due to less intense inflows and more of each precipitation event being intercepted by mature vegetation, and infiltrated into lawns and park areas. Trends in specific conductance showed a much slower decline, with the lowest records in the forebay being reached in August or September, compared to June in the commercial ponds.

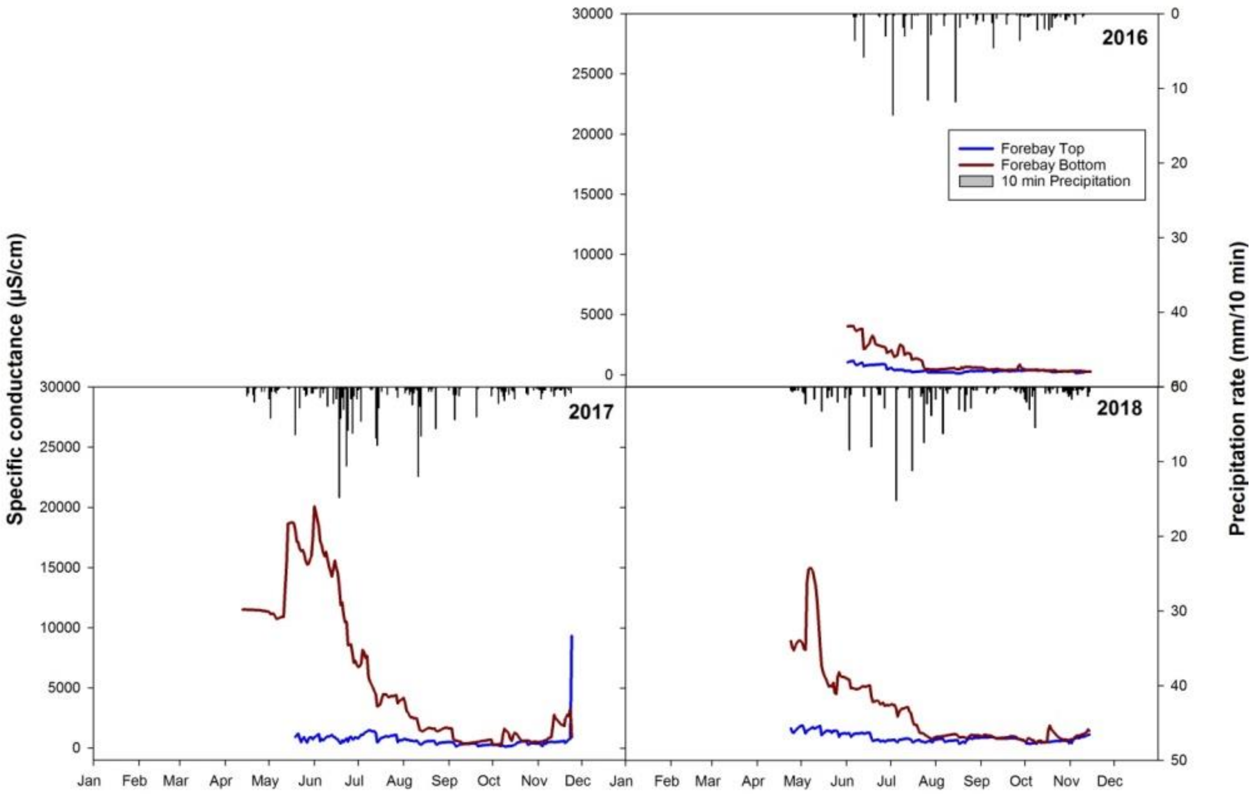

Figure 32. Mean daily specific conductance recorded at the forebay of Oaktree Pond (residential catchment), for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

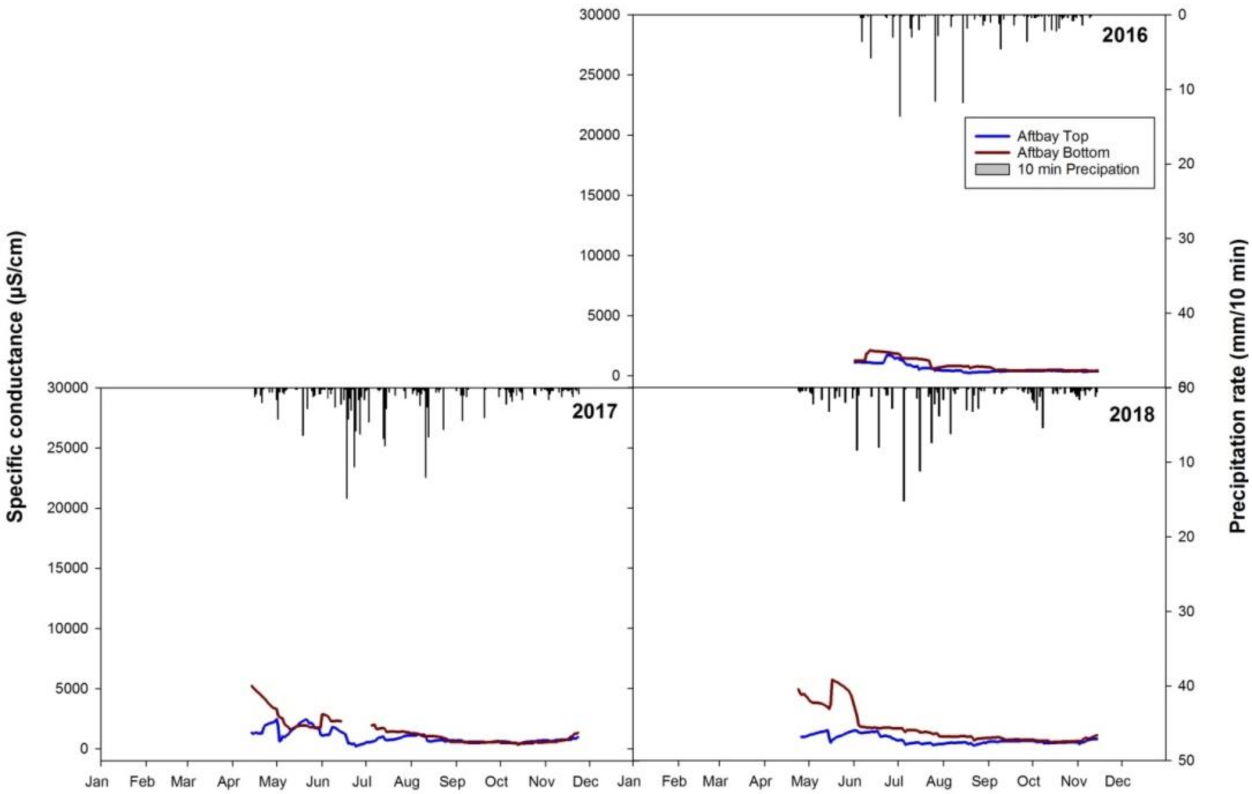

Figure 33. Mean daily specific conductance recorded at the aftbay of Oaktree Pond (residential catchment), for the top (0.5 m below the water surface) and bottom (0.3 m above the bottom).

5.5 Environmental drivers of pond stability and mixing

In order to further understand salinity stratification in the three SWMPs, Schmidt Stability (also known as relative thermal resistance to mixing) was calculated to determine the amount of energy required to fully mix the water column, disrupt the salinity stratification, and more completely flush water from the pond. Schmidt Stability is typically calculated to determine the period of water column thermal stability in lakes, and the amount of wind energy required to mix a thermally stratified water column, usually associated with fall overturn (see Stainsby et al. 2011 for applying this model to Lake Simcoe). At its core, thermal stratification is caused by differences in water densities due to temperature differences in layers of water (i.e. fresh water is less dense as its temperature increases or decreases from 3.8°C). Density differences in water can also be due to salt concentrations and, as such, water of different salinities also has different densities (an example of this are estuaries where freshwater from rivers overlays denser marine saltwater). Thus, the Schmidt function can be adapted for use in our

SWMPs to track changes in the strength of stratification and the resistance to mixing in the water column. In lakes, water column mixing is usually related to wind energy across the water surface; however, our study ponds have a relatively small surface area available for wind-driven mixing, and research from Minnesota suggests that as SWMPs are built at lower elevations, surrounded by tall vegetation, and thus they are very sheltered from wind energy / mixing (Erickson et al. 2018; Taguchi et al. 2019, 2020). As such, a lack of wind energy likely also impairs the ability of the water column in SWMPs to mix fully as was designed. In our study, the predominant force for disrupting stratification in the three ponds is likely water inputs, usually caused by precipitation and surface run-off, and can be used as a proxy driving force for pond mixing in our modified Schmidt model. Thus, our adapted Schmidt function can be used to assess the changing strength of salinity-driven pond water column stratification, and the amount of water input needed to breakdown this stratification.

At Site C Pond, the stability patterns (Figure 34) are quite similar to the specific conductance graphs above (Figures 28-29), as would be expected in a case where the water column stratification is due to salt-caused density differences. Stability is relatively high in spring, when accumulations of winter salt-laden water form a densely stratified water column of highly saline bottom water and fresher water at the pond surface. At a point during the late spring / early summer, a precipitation event occurs that presents the critical amount of energy to mix the water column of the forebay. This precipitation event appears to be related to the intensity of the rainfall, rather than overall amount of rainfall received. For the forebay of Site C Pond, this critical precipitation event is an accumulation of ~5 mm of rain in a 10-minute period. Less intense precipitation events appear to influence the stability of the water column by slowly diluting the saline bottom layer or moving on top of the density boundary and staying in the surface water, but these “sub-critical” or “sub-threshold” events are unlikely to thoroughly break the density stratification and mix the forebay water column.

Stratification of the aftbay is more persistent than that of the forebay, likely due to the forebay receiving the main force of incoming precipitation and water velocities being much reduced by the berm between forebay and aftbay. Of course, this feature is part of SWMP design where the forebay serves to buffer the shock of incoming surface run-off and the aftbay is the settling chamber for suspended particles. Rather than intensity of rainfall disrupting the aftbay stratification, it seems more likely that the volume of precipitation received is a driver of change. At Site C Pond, this critical volume of precipitation inputs is not typically reached until late fall, often just prior to the first applications of winter salt with the first snowfall. Either the incoming volume reaches an amount great enough to mix the aftbay, or a continuous influx of freshwater throughout the spring and summer gradually lower the salinity of the bottom water until a lower energy is needed for water column mixing.

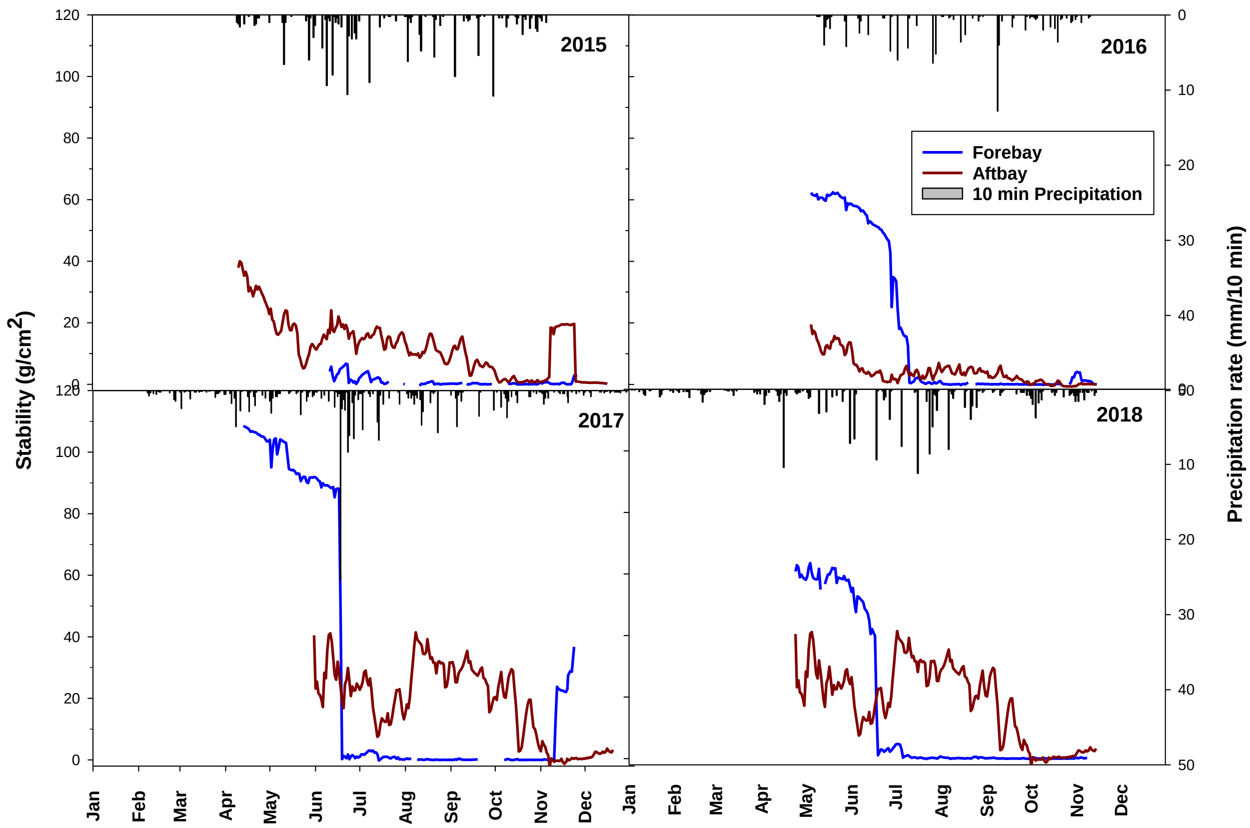

Figure 34. Schmidt stability for the forebay and aftbay of Site C Pond, 2015-2018. Vertical bar graphs show maximum recorded ten minute precipitation accumulation

From the detailed monitoring of Site C Pond, it is evident that salt stratification has a large impact on pond function, in particular the efficiency of the pond at protecting receiving waters from polluted urban surface run-off. Low dissolved oxygen in bottom waters can threaten receiving waters if the pond has a bottom draw outlet, when low DO water is released into critical tributary habitats, potentially causing a “low-oxygen shock” to organisms. In addition, low DO caused by intense salt stratification of the water column can impair phosphorus cycling by releasing sediment-bound nutrients that then get flushed into receiving waters (McEnroe et al. 2013). In fall, after the die-off of aquatic plants and algae, this phosphorus may be a bioavailable form (Troitsky et al. 2019) that would be readily available in downstream waters. Finally, salt stratification that results in a very stable, hard to mix, water column can result in precipitation events passing through the pond (i.e. “skipping” across the top of the deeper salt layer, as described in detail below) and discharging stormwater into receiving waters relatively untreated.

With a very dense bottom saline layer, inputs from precipitation that do not exceed the critical threshold for water column mixing (e.g. ~5 mm rainfall in 10 minutes) may travel on top of the saline layer to the aftbay, which has more persistent stratification, and discharge to receiving waters without mixing in the pond reservoir (Figure 35). For older catchments with more mature vegetation and / or naturalized areas (particularly residential catchments) greater capture of precipitation events may delay the flushing of ponds leading to stagnation. Another important consideration is the potential that widespread implementation of Low Impact Development (LID) technologies in a catchment with a poorly mixing SWMP may result in unintended consequences, exacerbating stagnation in ponds by reducing the energy transferred to the more conventional SWMP further downstream during larger precipitation events. It is imperative that a decision making tool be designed for maximizing the efficiency of stormwater treatment and that the type of technology used is the most effective method (see review by Taguchi et al. 2020).

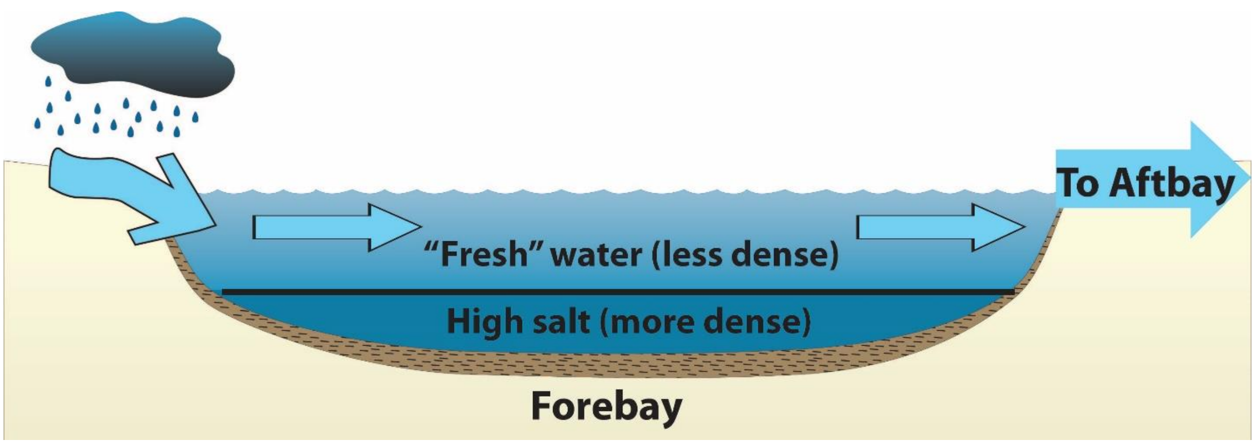

Figure 35. Conceptual diagram of a stormwater management pond forebay showing precipitation events, at an intensity below the critical force to mix the forebay, travelling on top of the denser salt layer to the aftbay and eventual discharge.

In 2017, this "skipping" of stormwater inputs across the surface of the water column density barrier was recorded in our monitoring of Site C Pond. Two relatively large rainfall events were enough to mix the water column of the aft bay and flush out some of the saline bottom water (Figure 36), even though the intensities of the event were below the threshold needed to disrupt the salinity stratification of the forebay (Figure 37). It is likely that these precipitation events did not have the volume required to mix the aftbay water column, and the incoming stormwater may have skipped across the density boundary layer of both basins and entered receiving waters untreated. Further research on these high-volume, low-intensity, events is required in order to more fully understand how winter deicing salt run-off, density stratification of the water column, and precipitation impact the performance and efficiency of SWMPs. In addition, future research should consider the impact to the functional volume of SWMPs, particularly those with commercial catchments, large paved parking lots, and heavy winter salt applications. With a dense salt layer in the bottom water of SWMPs, what volume of water can be effectively treated by the pond before the incoming stormwater is forced out by overflow outlets.

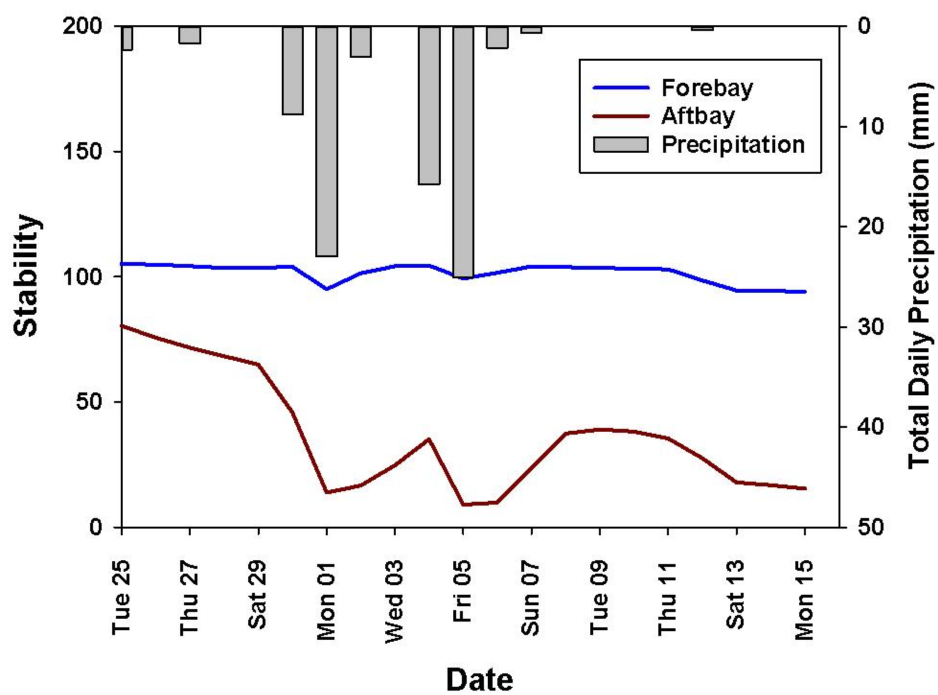

Figure 36. Graph of Schmidt Stability in the forebay and aftbay of Site C Pond, April 25 – May 15, 2017 showing rainfall events that "skipped" over the stratified water column of the forebay and mixing the aftbay. Vertical bars show total daily precipitation of two events that had sufficient volume inputs to reduce stability (i.e. mix) the aftbay water column while leaving the forebay water column stability unaffected.

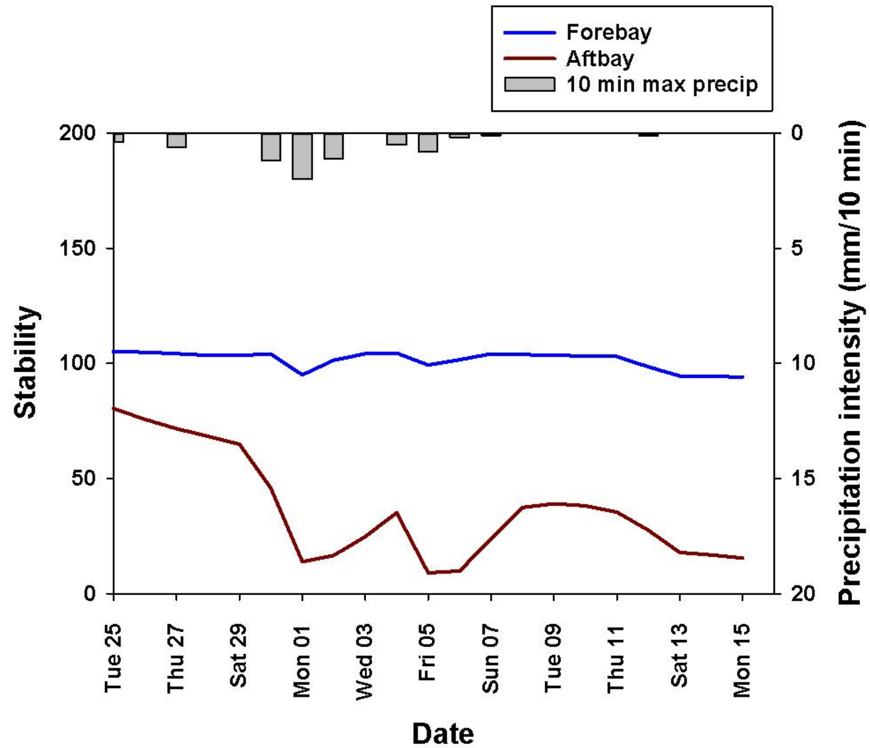

Figure 37. Graph of Schmidt Stability in the forebay and aftbay of Site C Pond, April 25 – May 15, 2017 showing rainfall events that "skipped" over the stratified water column of the forebay and mixing the aftbay. Vertical bars show maximum precipitation intensity recorded for two events that, while having sufficient volume inputs to reduce stability (i.e. mix) the aftbay water column, the intensity was below the threshold required (~5 mm rainfall in a 10 minute period) to disrupt forebay water column stratification.

5.5.1 Comparison of ponds with commercial and residential catchments

Hillock and Oaktree ponds, a smaller commercial and a residential catchment, behave similar to Site C Pond in terms of water column stability. All three ponds have similar patterns, however the physical factors of the ponds, and differences in the catchments allow some differences. Site C Pond has a large catchment, and is a large pond, whereas Hillock Pond, is smaller in both pond size and catchment area. Although higher specific conductances were recorded at Hillock, a shallower pond depth and steeper run-off from the catchment allow less intense rainfalls (~2-3 mm rainfall in 10 minutes) to mix the forebay water column (Figure 38) more readily than at Site C. As Hillock had sensors deployed over winter, this pond enabled the opportunity to learn that although salinity begins to increase following winter salt application in late fall, it is the spring run-off that causes much of the increase in pond salinity, and water column density stratification. Oaktree Pond (Figure 39) has lower water column stability (and lower specific conductance) values than the commercial ponds, due in part to a smaller proportion of impervious surface in the catchment, less salt application, but also the smaller size of the pond compared to Site C Pond. In the case of Oaktree Pond, a rainfall intensity of ~2-3 mm in 10 minutes appears to be required to disrupt salt stratification of the water column in the forebay. However, Oaktree Pond does not seem to undergo the rapid, complete flushing of the forebay recorded in the commercial ponds, likely due to lower incoming water volume, and less velocity from a gentler slope from the upstream drainage catchment area into the pond.

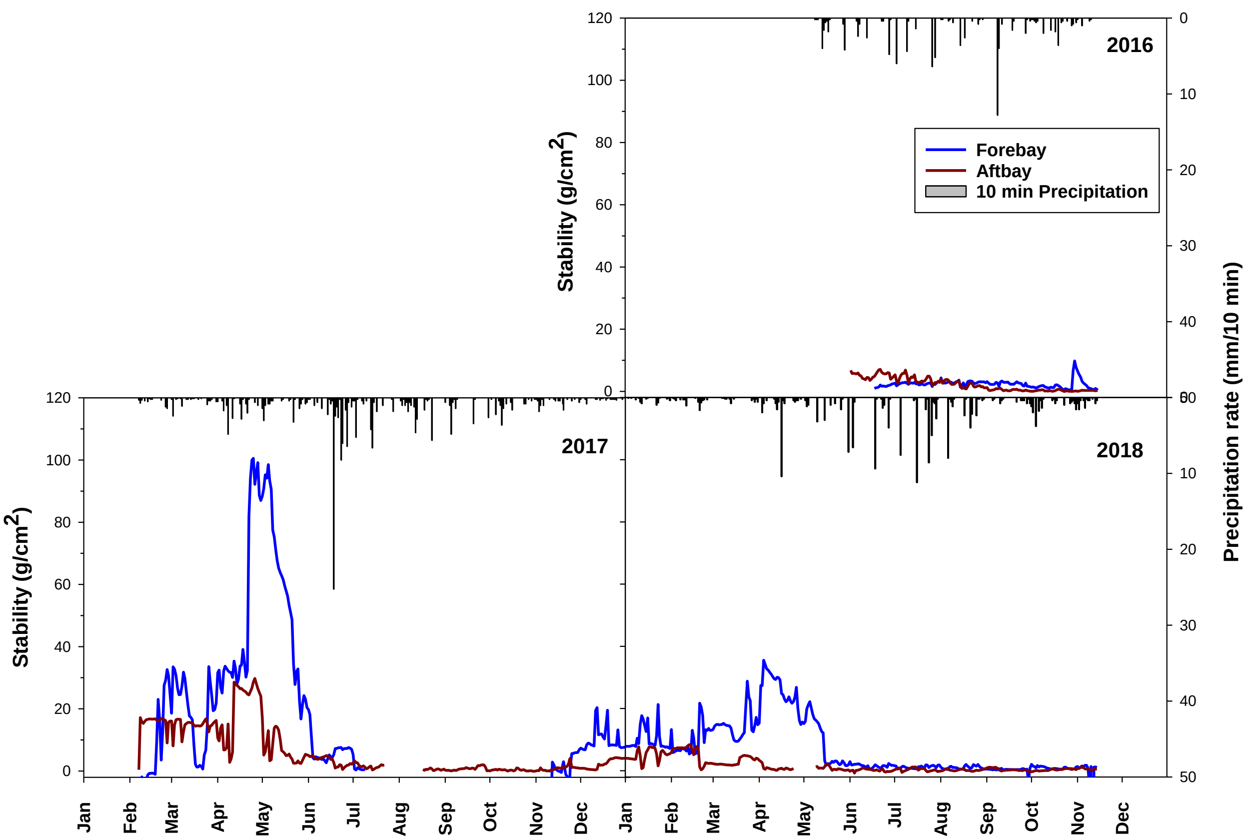

Figure 38. Schmidt stability for the forebay and aftbay of Hillock Pond, 2016-2018. Vertical bar graphs show maximum recorded ten minute precipitation accumulation

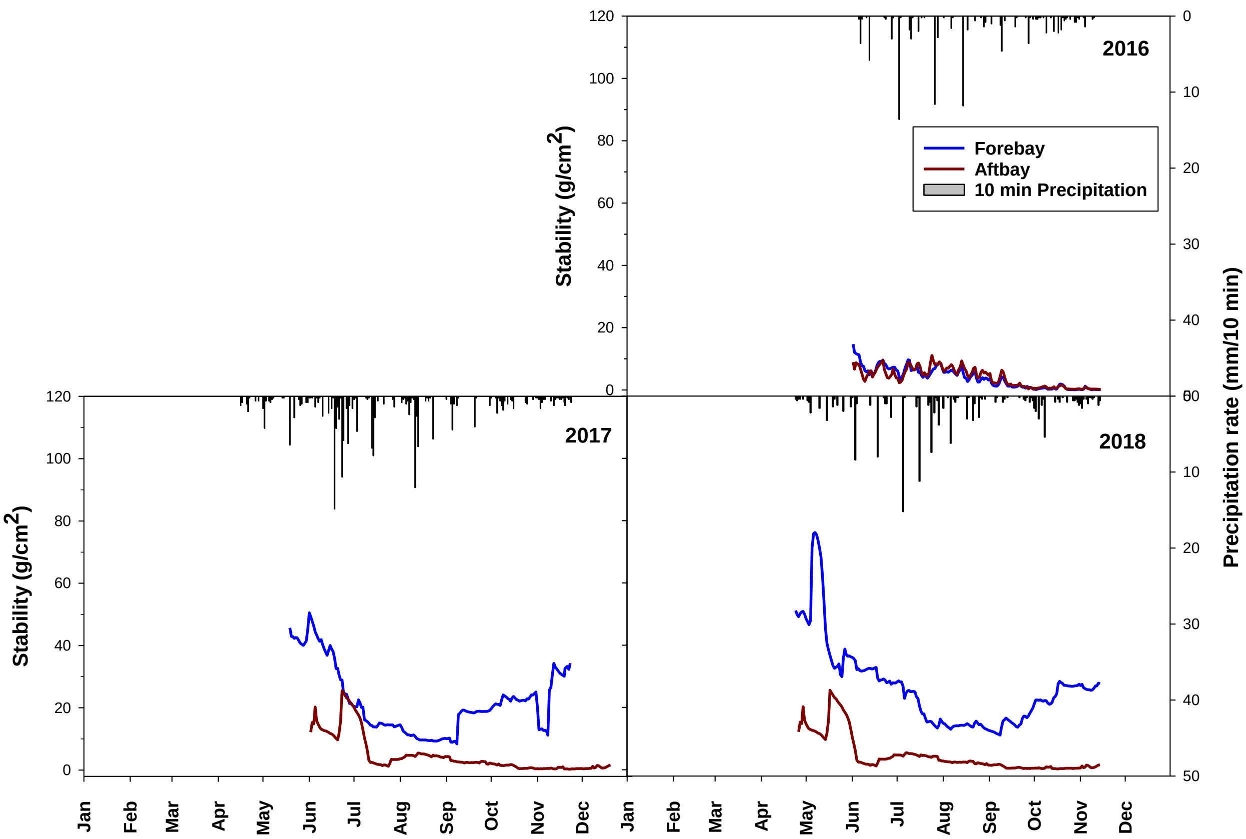

Figure 39. Schmidt stability for the forebay and aftbay of Oaktree Pond, 2016-2018. Vertical bar graphs show maximum recorded ten minute precipitation accumulation

6. Environmental considerations for phosphorus cycling in SWMPs

In general, although SWMPs appear to be good at their original purpose of retaining the suspended solids from surface run-off reaching receiving waters, a variety of environmental drivers are impairing their ability to retain nutrients. In theory, sediment and nutrients enter a pond and settle to the bottom, being locked by chemical processes (to calcium and sediment metals), enabling water with relatively improved quality to exit the aftbay (Figure 40a). In practice, however, in-pond chemistry (such as nutrient cycling) and physical conditions (i.e. stratification of the water column by temperature, dissolved oxygen concentration, and salinity) impair the retention of nutrients that are likely cycled between water and sediments, and converted to more bioavailable forms (Frost et al. 2019). If these nutrients are in the water column during large rainfall events, it is likely they are flushed into the receiving waters the SWMP was designed to protect. Research on ponds in southern and central Ontario indicate that although up to 80% of suspended solids may be retained in the pond, the overall retention of phosphorus is much lower at 40-50% (Frost et al. 2019). Phosphorus attached to sediment particles (particulate phosphorus) is typically removed from the water column in SWMPs but can be transformed to dissolved P forms by microbial breakdown of organic matter in the sediments. This dissolved phosphorus (in both organic and inorganic forms) is then released back into the water column during low-oxygen periods (metal-bound phosphorus) or under both high and low oxygen conditions (organic phosphorus) (Frost et al. 2019).

With the inputs of saline surface run-off due to winter salt use, water column stratification in SWMPs may exacerbate problems with pond performance, particularly in ponds with commercial use catchments. The dense salt layer in the bottom water is resistant to mixing, thus increasing the likelihood of dissolved oxygen depletion and creating environmental conditions amenable to release of metal-bound sediment phosphorus (Figure 40b) and the presence of extremophile organisms such as purple-sulphur bacteria (Figure 41). In commercial ponds that have aquatic plant growth (such as Site C and Hillock ponds), these low-oxygen conditions are prevalent in spring, before plants appear and start to supply oxygen to the bottom water. In residential ponds, particularly those with high turbidity, aquatic plant growth can be inhibited, due to lack of sunlight reaching the pond bottom, and persistent low oxygen conditions can occur, even after the saline water has been flushed from the pond and the water column stratifies by water temperature. Additionally, the amount of salt inputs causes high SAR values in pond sediments that exceeds the guideline values for both residential / agricultural and commercial / industrial uses.

In addition to impacting in-pond phosphorus cycling and the quality of water leaving the pond, salinity stratification may further impair SWMP efficiency by reducing the pond volume. SWMPs work based on the permanent pool volume of the pond (McEnroe et al. 2013). As such, they are expected to become less efficient at removing suspended particles as the pond slowly fills with sediment. SWMP maintenance and clean-out (i.e. removing the sediment accumulated from surface run-off) is intended to restore pond functioning to design criteria. However, the dense lower salt layer is resistant to mixing (until a rainfall input of sufficient energy to mix the pond) and likely limits the permanent pool volume of the SWMP. Rainfall events that lack the energy threshold to mix the pond water column, and being less dense than the saline bottom water, likely travel along the density boundary layer, across the aftbay and into receiving waters. Thus, reduced permanent pool volume of the pond may further impair SWMP functioning and performance.

Behavioral practices of residents within the catchments of SWMPs may also have an impact on pond functioning. In several of our study ponds, particularly those in residential catchments with easy access points, the presence of released goldfish was related to higher levels of turbidity and suspended solids in the pond. The primary mechanism for SWMPs to improve water quality in receiving waters is to remove suspended particles. Goldfish increase the turbidity of the water and limit the settling required to reduce suspended solids in water discharged from the pond to receiving waters, thus limiting pond performance and efficiency.

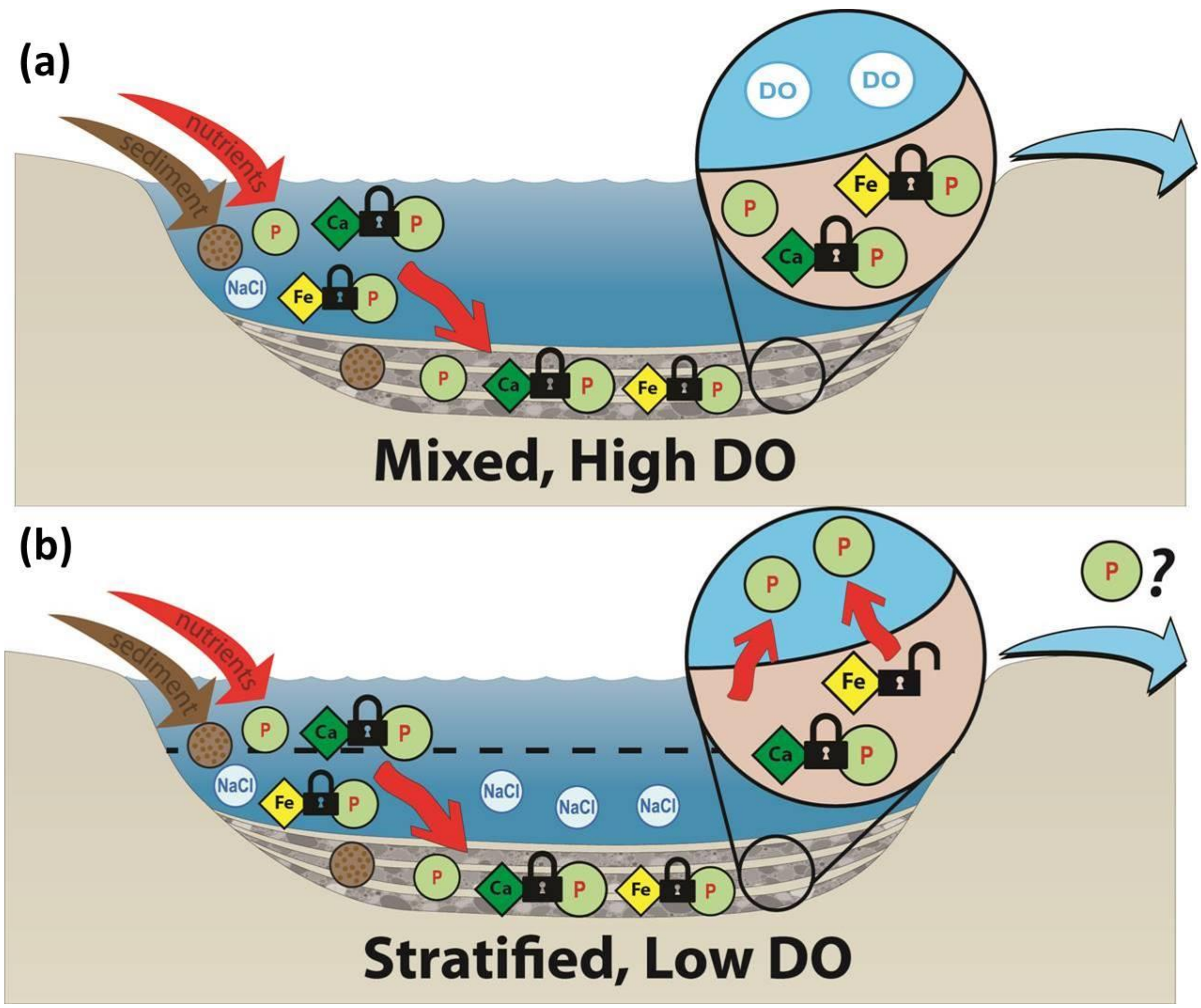

Figure 40. Conceptual diagram of a stormwater management pond showing (a) ideal functioning with suspended particles (and sorbed phosphorus) being deposited to pond bottom and captured in sediment; and (b) more typical functioning in urban catchments with inputs of winter de-icing salt

stratifying the pond by density and dissolved oxygen, enabling release of sediment nutrients under low oxygen conditions.

Figure 41. Purple-sulphur bacteria, and extremophile, colonizing a water quality sonde at Site C Pond, June 20, 2016.

7.0 Conclusions

This project investigated the impact of external and within-pond environmental drivers on the performance, efficiency, and functioning of urban stormwater management ponds in the southern portion of the Lake Simcoe Watershed. Based on earlier (2010) pond surveys, the significant environmental drivers of conditions in these ponds are, unsurprisingly, related to the urban landscape where the ponds were built: nutrient runoff and winter de-icing salt. In a survey of 11 ponds along a gradient of urbanization (from catchments with primarily residential homes to catchments with commercial use and expansive paved parking lots), we found that all ponds are functioning less efficiently than predicted by environmental models. The more detailed study of three ponds underscored the basic trends recorded in the larger study set, but provided much greater insight into how pond performance and efficiency is being impacted by several environmental factors. First, inputs of surface stormwater run-off, particularly in spring when it is laden with winter de-icing salt, forms a chemical stratification of the pond water column. Second, highly saline bottom waters of the pond set up a very strong density barrier between the top and bottom waters of the pond, making the pond very resistant to mixing, likely leading to many precipitation events (that are below a threshold for either intensity or volume) flowing through the pond, on top of the salt layer, and into the receiving waters. Third, low dissolved oxygen concentrations in the bottom water of the pond, primarily caused by infrequent mixing of the water column, leads to the release of sediment-bound inorganic phosphorus that is readily used by organisms (aquatic plants and algae), or flushed from the pond if a large rainfall events occurs and mixes the water column. Finally, some ponds in residential areas have persistent elevated turbidity in the pond water, which is related to the presence of goldfish, likely released by residents living near the pond.

7.1 Recommendations

- Encourage winter salt application best management practices (BMPs) to reduce application rates. These BMPs would serve to reduce the salinity of surface run-off, minimize the strong stratification of the water column that is resistant to mixing, and enhance protection of "end-of-pipe" receiving waters.
- Maintaining dissolved oxygen concentrations in the bottom water above ~2-3 mg/L would prevent the release of sediment-bound inorganic phosphorus. Although aerators or bubblers are effective in this regard, they do counteract the intent of SWMPs, which are to settle out suspended solids from surface run-off. A reduction in the bottom water salinity would allow the SWMP to mix more frequently by allowing smaller, less intense, rainfall events to mix the water column.
- Wind action can assist in mixing ponds. However, ponds are typically situated in low lying areas and surrounded by vegetation, thus sheltering them from wind that could enhance water column mixing and improve pond functioning. To assist in promoting pond mixing through wind action, the coverage and extent of bank vegetation should be limited on the side of prevailing winds.
- As bioavailable phosphorus is taken up by aquatic plants and algae, and then released when these decay in the fall, removal of plants at the end of the growing season would remove any phosphorus that they have accumulated in their tissues. This management strategy would limit the release of tissue phosphorus to the pond water column, and subsequent flushing of phosphorus into receiving waters.
- In addition to receiving waters being impacted by phosphorus, particularly biologically available forms of P, they are also affected by suspended particles and chloride that are flushed from the SWMP. Improvements in limiting water column stratification, harvesting plants to remove nutrients, and preventing the release of goldfish would serve to protect receiving waters.
- Develop a management strategy to prevent the release of alien species (e.g. goldfish) to SWMPs through education and signage.
- For SWMP facilities with existing goldfish populations, a strategy for eradication should be developed. Such strategies could be more easily implemented during routine sediment clean out but care must also be given to prevent the spread of this invasive species into receiving waters.
- Implementation of LID or other stormwater management solutions in catchments already serviced by a SWMP should consider the cumulative impact of these solutions, such as how slowing the flow of water, or reducing the volume of run-off, will impact the ability of the SWMP to fully mix. In addition,

the impact of SWMPs with currently impaired functioning may be a polluting influence on receiving waters needs to be considered.

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