Final Report: Assessing cumulative effects of stormwater management pond outflows on aquatic ecosystems

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Summary

To mitigate the potential impacts of increasing urbanization and impervious cover within watersheds, Storm Water Management Ponds (SWMPs) are an infrastructure tool that have been widely adopted to reduce the impact of 'flashy' flow events within a watershed. In doing so, they can decrease erosion and associated economic costs of maintaining infrastructure associated with river valley lands. While SWMPs reduce the hydrological flashiness of a watershed to rain events, this impacts the amount and frequency of the delivery of water, natural/anthropogenic nutrients, and contaminants. The goal of this study is to determine the cumulative influence of SWMPs, considering the design and number of SWMPs on a watercourse, with the aim to quantify current and future risk of thermal and chloride stress to aquatic ecosystems.

Within an approximately 1 km stretch of the West Humber tributary of Campbell's Cross Creek, the results from two years of monitoring suggest that SWMPs account for most of the differences in thermal load observed instream in 2019 and 2020 (92% and 100%, respectively) over the period of study (June-September). On average, ponds generally tended to cool more than warm instream temperature in both years, whereas event based thermal loads appear to have the largest impact, by increasing instream temperatures. Moreover, sharp increases in chloride concentration are seen instream and at the outlets of SWMPs during precipitation events. Understanding the behaviour of SWMPs and the influence of other contributing factors on chloride concentration, will be crucial to ensuring chloride levels are kept below thresholds designated for the protection of aquatic life.

A commonly used indicator of water quality is the benthic macroinvertebrate community instream. The proportion of macroinvertebrate taxa that are known to be tolerant versus intolerant to poor water quality conditions (e.g., salinity, contaminants) can serve as a proxy for the quality of habitat that exists at different locations along the study reach. Analysis of benthic surveys from 2019 and 2020 demonstrate that more sensitive species are found upstream while more pollutant-tolerant species are found downstream. This implies that the condition of water leaving SWMPs is influencing the quality of habitat for aquatic benthic species. Notably, the most sensitive species, *Paracapnia sp.* (winter stoneflies), were found to be a major driver of diversity and compositional differences within the stream reach in 2019, while *Cheumatopsyche sp.* were a larger driver in 2020.

Lastly, in the second year of this study, the thermal and chloride conditions measured at the outlets of each bottom-draw pond along the study reach were compared to a reference top-draw pond. Top-draw ponds are designed to draw water from the surface of the pond rather than from the bottom, which may cause differences in the temperature and chloride levels in the water leaving the pond. We find that the top-draw pond has a unique thermal (e.g., higher thermal load overall and during events) and chloride signature (e.g., higher event-based chloride loading) compared to our bottom draw ponds. Thus, pond design can play a key role in mitigating thermal and/or chloride risks to aquatic health in-stream.

Table of Contents

Background 1

Objectives 3

Methods 4

Study Area 4

Environmental Pond Surveys 6

Cumulative Analysis 8

Study Results 11

Temperature 11

Chloride 16

Comparison of 2019 and 2020 Thermal and Chloride Results 20

Benthic Communities 20

Comparison of Bottom-draw Study Ponds to Top-draw Pond 23

Climate Conditions 28

Limitations and Suggestions for Future Research 32

Conclusions 32

References 35

Background

The watersheds of the Toronto and Region Conservation Authority (TRCA) jurisdiction are one of the most densely populated watersheds in Canada and the population is expected to continue to grow over the coming decades (Ontario, 2017). This has contributed to increasing urbanization of the TRCA jurisdiction and the City of Brampton, where natural pervious cover is being converted into impervious cover. Currently, the urban area (a proxy for impervious cover) now accounts for over 50 percent of the land cover in the TRCA jurisdiction (TRCA, 2018). This contributes to changes in multiple water quantity (e.g., base flow, flashiness) and quality (total suspended solids, nitrogen, phosphorous, chloride) conditions (TRCA, 2011). Among other consequences, these changes can have various adverse impacts on aquatic ecosystems and species (Steedman, 1988; Islam *et al.*, 2011).

To mitigate potential impacts of impervious cover, Storm Water Management Ponds (SWMPs) have been adopted as a tool. They are expected to reduce potential water quantity and quality stressors to aquatic life (TRCA & CVC, 2010; Prudencio & Null, 2018). However, these reductions may not be below suitable thresholds to sustain healthy aquatic ecosystems (Tuccillo, 2006; Islam *et al.*, 2011). Moreover, several SWMPs in sequence may have a cumulative impact on aquatic ecosystems. Of note, peak summer temperatures (July-August) produce high water temperatures in SWMPs, where outflows from ponds have the potential to raise the temperature of nearby fluvial systems (Hester & Bauman, 2013; Sabouri *et al.*, 2013). While the impact of a single SWMP may be minimal, the potential cumulative impact of warming and other stressors is unclear. Specifically, there is ongoing concern that many ponds in sequence may raise instream temperature beyond lethal temperature thresholds for many cool water aquatic species, such as the Endangered Redside Dace (*Clinostomus elongates*) (Jobling, 1981; COSEWIC, 2007). For Redside Dace stream reaches, the Ontario Ministry of Natural Resources provides best management practices that aim to have SWMP discharge below a temperature of 24˚C (MNRF, 2016). Given that a large proportion of new development is now occurring alongside many cool and cold-water streams in the City of Brampton (habitat that supports Redside Dace) and that future climate change may exacerbate increases of instream temperatures, there is a pressing need to quantify the potential risks that SWMPs present to maintaining healthy and functioning aquatic ecosystems.

Additionally, freshwater ecosystems and species have evolved for millions of years in the absence of high chloride loads that occur today. Both the acute and chronic levels of chloride have increased within the TRCA jurisdiction with an increasing need to expand road salting practices with new development during the colder months of the year (Wallace & Biastoch, 2016). SWMPs impact the process with which chloride is delivered instream by generally storing and releasing chloride over a longer period of time throughout the year, thus reducing acute stress, but increasing the chronic level of chloride (Fanelli, Prestegaard & Palmer, 2019; Haake & Knouft, 2019). In the same way that temperature impacts may be cumulative, chloride release from a series of ponds also has the potential to behave in the same manner. Canadian water quality guidelines for the protection of aquatic life provide chronic (120 mg Cl^-/L) and acute (640 mg Cl^-/L) exposure thresholds that should not be exceeded to ensure minimal impact to ecosystems (CCME, 2011). Understanding how SWMPs function with respect to chloride by considering the potential influence on instream chloride conditions can help to direct whether further design considerations are needed to mitigate this potential risk.

To further investigate the thermal and chloride effects that SWMPs have on aquatic ecosystems it is beneficial to compare the similarities and differences in these effects from SWMPs with different designs (i.e., bottom-draw vs. top-draw ponds). In addition, a common method of assessing water quality and suitability of habitat for aquatic communities is to assess the response of benthic macroinvertebrate communities and whether they may reflect species that are tolerant versus intolerant to poor water quality conditions. Both additional pieces of analysis will provide a more wholistic picture of the risks or benefits that SWMPs pose to aquatic ecosystems.

Altogether this analysis will provide evidence-based information and guidance to the City of Brampton on the design and configuration of SWMPs and their potential impacts on aquatic ecosystems. The overall goal is to further reduce the impact of development and SWMP outflows to aquatic ecosystems through watershed-based planning.

Objectives

The main objective of this work is to determine the cumulative influence of SWMPs, where the design and number of SWMPs can be explored with respect to reducing current and future risk of thermal and chloride stress to aquatic ecosystems. This project will consider both acute and chronic stress of thermal and chloride conditions within the system. We also aim to estimate the potential risk of future climate change and an assess whether design and configuration of SWMPs may offer further potential risks and/or benefits.

Specifically, the objectives of this project are to:

1. Assess cumulative impacts of SWMPs, which will consider thermal load and chloride concentration entering the receiving reach. This will consider the average chloride and thermal condition of the system over the study period and large precipitation events.
2. Explore the response of the aquatic ecosystem to the cumulative chloride concentration and thermal load inputs from SWMPs by conducting aquatic community surveys.
3. Assess the potential risk to aquatic communities given future land use change (more SWMPs) and climate change.

Methods

Study Area

The study area is a specific reach within Campbell's Cross Creek Tributary (located SE of Airport Road and Countryside Drive) in the Humber watershed within the City of Brampton (Figure 1). This is a cool water stream reach that provides habitat to Redside Dace and other cool water aquatic species (Hasnain, Minns & Shuter, 2010; COSEWIC, 2007). The catchment of the study reach has experienced ongoing urban development, predominantly residential/commercial land uses from the early 2000s, in an area that was historically dominated by semi-rural agriculture and estate residential areas. This has produced an increase in the number of online SWMPs found on the reach and provides an ideal location for the study.

The study reach is approximately a 1 km stream segment with 3 online SWMPs (named Carberry, Farad, and Maggie for the purposes of this study), where the reach catchment for this area is dominated by outflows from the SWMPs (see Figure 1 for the location of ponds). The three SWMPs all have bottom draw structures installed with defined watercourses for outflows. This ensures that it is easier to identify the point source from each pond. The ponds vary in catchment size, volume, and surface area, making each pond potentially unique in its contribution to instream conditions (Table 1). Initial surveys of ponds included referencing design drawings and conducting site inspections were completed to evaluate function. During the site inspection of ponds during spring 2019, Maggie Pond was identified as not functioning, since the outflow was completely routed through the below ground diversion and never entered the pond itself during low flow events. Thus, the Maggie pond largely functions as if there is no SWMP in place. It was also noted that given the design of the diversion for the Farad pond, it is possible that a large portion of surface flow from the catchment moves through this diversion during low flow events as well. Lastly, Carberry Pond appears to function as designed with no notable issues. In the second year of this study, a nearby top-draw SWMP was also monitored to provide further insight into the performance of different SWMP designs. The top-draw pond is located less than five kilometers from the three bottom-draw ponds within the City of Brampton (Southeast of Cottrelle Boulevard and McVean Drive).

Table 1. Summary characteristics of the three bottom-draw stormwater ponds included within the study. Shown is the pond name, the estimated surface area of the permanent pool, estimated volume of the permanent pool, estimated catchment size, and whether the pond appears to be functioning as designed. Note: catchment and surface area sizes are estimated from design drawings and flow path information provided by the City of Brampton.

Name	Estimated Surface Area (ha)	Estimated Volume (m^3)	Estimated Catchment (ha)	Functioning	Notes
Carberry	0.42	8775	27.74	Y	Appears to be operating as intended
Farad	0.31	3800	24.15	Y/N	Flow may divert around pond
Maggie	0.42	5250	41.22	N	Inflow is limited; almost all flow directed through diversion

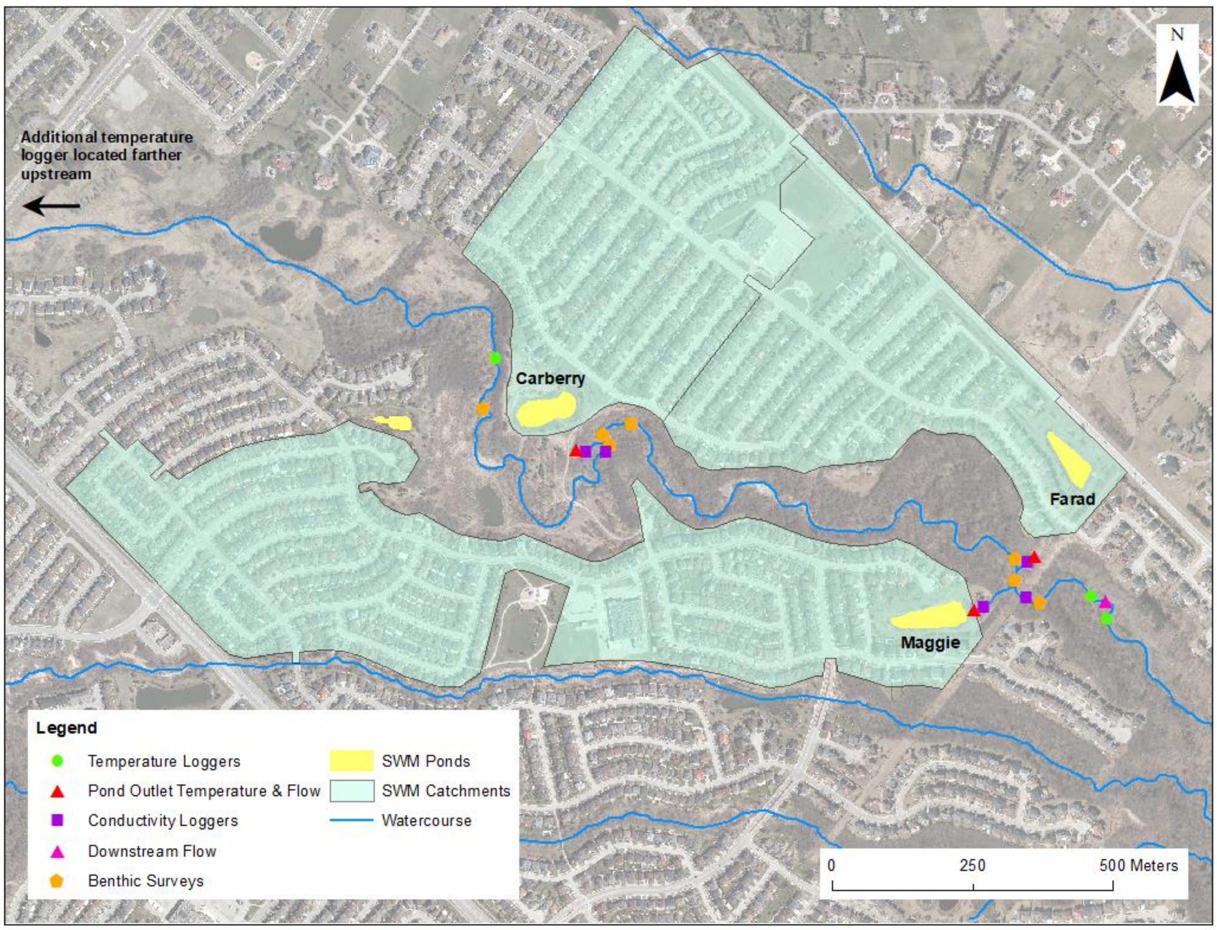

Figure 1. The study reach located at Campbell's Cross Creek Tributary in the West Humber River located in the City of Brampton. Shown are the study ponds, catchments of the ponds alongside the location of benthic surveys, and flow, temperature, and conductivity loggers within the reach.

Environmental and Pond Surveys

This project is a 2-year undertaking, where the first year of data provided context for how the system operates and provided insight into how the ponds may be influencing instream conditions. The second year of data provides a longer period of record for both temperature and conductivity data, which gives a better understanding of the annual function and influence of SWMPs on aquatic habitat. This additional monitoring also helps to capture differences caused by variation in weather between years.

To assess the thermal loading and chloride concentrations within the stream, temperature and conductivity data were collected in 2019 and 2020 within the outflows of the three SWMPs and immediately upstream and downstream of the SWMPs (Figure 1). Based on seasonal weather conditions and the availability of temperature, conductivity, and flow data, different periods of time for the analysis of temperature and chloride were chosen (Figure 2). Thermal loading effects of the ponds were analyzed between June 1 and September 27 for both years, while chloride effects were analyzed between October 5 and November 19 in 2019 and between March 18 and December 3 in 2020.

In addition to the three bottom-draw SWMPs, temperature, conductivity, and flow were measured at the outlet of a reference top-draw SWMP in 2020 to further investigate the performance of different pond designs, with regards to the thermal and chloride conditions. Temperature and conductivity data were collected from June 1 to December 3, while flow was recorded between March 31 and December 22 (Figure 2).

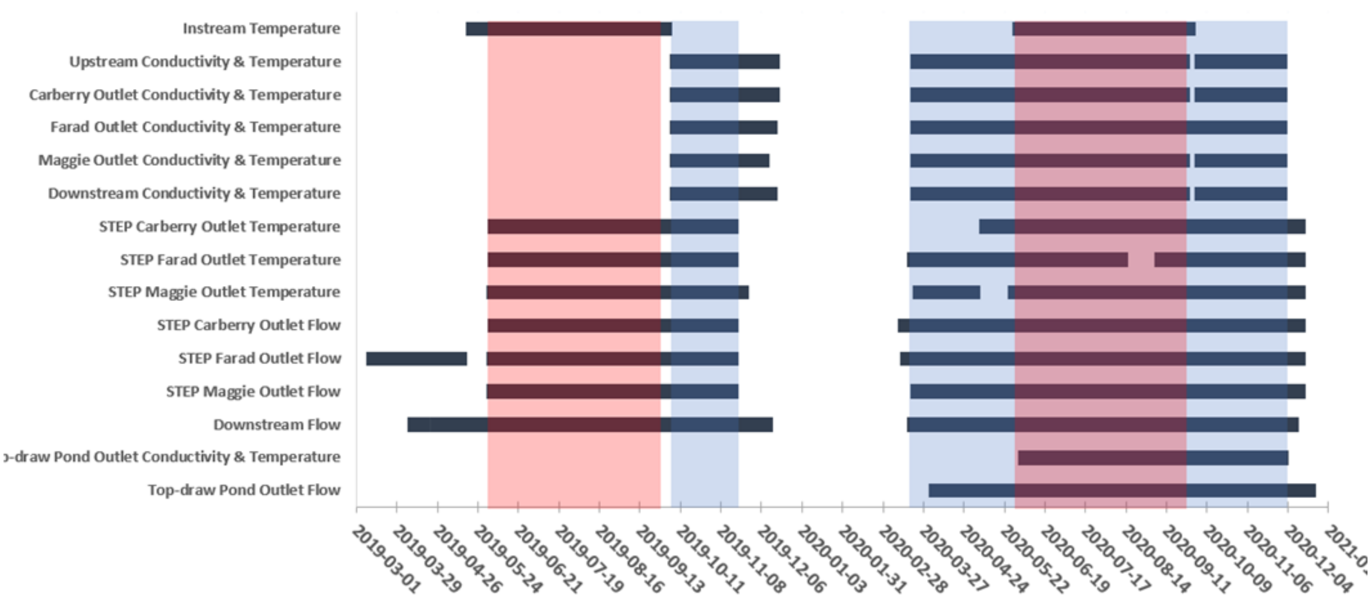

Figure 2. The timeline of temperature, conductivity, and flow sampling within the study reach in 2019 and 2020 and at the reference top-draw pond in 2020. The temporal period of analysis for temperature (red) and conductivity (blue) is shown.

Specifically, to collect flow and temperature data from the outflow of SWMPs we used ISCO 2150 Area Velocity Module flow meters and a variety of temperature loggers, including: HOBO MX2201 Pendant data loggers, HOBO Water temp pro V2 loggers, HOBO Water level loggers, and Van Essen mini-diver loggers (temperature and barometric pressure). Both the flow and temperature loggers were installed within the outlet structures of the ponds. For conductivity, ONSET HOBO freshwater loggers were deployed at all 5 locations and were paired with ONSET HOBO saltwater loggers in SWMP outflows.

Data from instream loggers were screened for out of water records by reviewing barometric pressure records as well as changes in stream temperature records at 15-minute and daily timesteps. First, if the temperature change between each 15-minute timestep was greater than 1˚C, these instances were flagged as potentially out of water and counted. Secondly, differences between the daily maximum and minimum temperature records of each logger were determined. If these differences were greater than 8˚C in a day, these instances were flagged as potentially out of water. The reason for this is that a change in temperature greater than 8˚C would be more indicative of an air temperature change than a stream temperature change. Lastly, the 99^{th} percentile values of these differences were determined and then ranked to observe which loggers had the largest differences between daily maximum and minimum temperature. Through these approaches, loggers that were suspected to have been out of the water during part of the period of analysis were excluded from use in analyses.

The hydrometric downstream flow data was collected using an In-Situ LevelTroll 400 non-vented level logger. Site visits were made shortly after various storm events to record the stream water level. Through the accumulation of visits a rating curve was developed to estimate flow from instream water level by correlating the observed stream level to the flow data recorded by the LevelTroll 400. There is a separate rating curve for each year of the project. Additionally, a shift in the curves was applied for each year in the fall. Based on the measurements recorded in the fall compared to earlier in the year it was determined that there was likely a build-up of leaves or other material in the stream that was enough to push the measurements off the original rating curve. This resulted in slight changes in the downstream flow data previously recorded in 2019.

Lastly, benthic invertebrate surveys were completed at six sites along the study reach in 2019 and seven sites in 2020 (Figure 1), to provide data on the potential impacts to biotic life in the stream. One was survey site was upstream, and one was downstream of all SWMPs. Surveys were completed using Hess samplers (Hess, 1941) in both years.

Cumulative Analysis

With both years of data collection, we conducted quality control/quality assurance (see above), compiled all information into a database and ran analyses of data from 2019 and 2020 for both temperature and conductivity (see Figure 2 for periods of analysis). For both temperature and chloride, we investigated the full duration of the analysis periods and highlighted large thermal and chloride events that were recorded. The contribution of SWMPs to instream thermal and chloride concentrations for each time period is summarized.

Thermal Loading

To determine the thermal load and the contribution of each pond, we calculated the thermal load (TL) using the flow and temperature of outlets and both upstream and downstream locations of the ponds. The thermal load (*TL*) of these five locations was calculated by:

$$TL = Q_t * p * T_t * C * t \qquad (Eq.\ 1)$$

Where,

TL is the thermal load (mJ)

Q_t is the flow over the period of time (t; m^3/s)

p is the density of water (kg/m^3)

T_t is the temperature of water over the time period (t; °C)

C is the specific heat capacity of water (4187J/kg/°C)

t is the period of time (seconds)

Within the study system the thermal loading of downstream was expected to be a function of upstream loading and the loading from the three SWMPs along to study reach (Figure 1). To investigate the contribution of all three SWMPs to the instream thermal load downstream, we would expect the downstream load to be a function of the sum of the thermal load from the upstream and three SWMPs, which can be represented by the equation:

$$TL_{DS} = TL_{P1} + TL_{P2} + TL_{P3} + TL_{US} \qquad (Eq.\ 2)$$

Where,

TL_{DS} is the thermal load of the downstream,

$TL_{P1,\ P2,\ P3}$ is the thermal load of each pond, and

TL_{US} is the thermal load of the upstream.

Chloride

Chloride concentrations instream and at SMWP outlets were not measured directly but rather were estimated from measured conductivity. After reviewing the results from both years of this study, we chose not to analyze chloride by quantifying loads as this did not effectively communicate the amount of chloride entering the stream due to the assumptions associated with estimating chloride from conductivity and using flow when calculating loads.

To estimate chloride concentration using conductivity measurements, we previously used the relationship from Wallace and Biastoch (2016), which used paired conductivity-chloride water quality samples from 2002 to 2012 within the TRCA jurisdiction. However, after reviewing results from year one of this study it was found that calculating chloride concentration using this relationship appeared to underestimate concentrations when high conductivity occurred in the reach monitored in this study. A similar pattern was addressed by Lam et al. (2020) such that at lower conductivity, the proportion of chloride is lower and other ions such as calcium and bicarbonate become greater drivers of conductivity. The Wallace and Biastoch (2016) relationship was also based on relatively older data collected over a broad scale and may not be representative of our study area or current conditions. Thus, an updated equation was needed to estimate chloride concentrations.

To achieve this, a breakpoint analysis was conducted to determine the change in the linear regression relationship between conductivity and chloride concentrations, specifically within the Humber River watershed. Here the breakpoint analysis estimates the change as the relationship between chloride concentration and conductivity changes depending on whether there is low or high salt inputs (Lam et al. 2020). Similarly, to Wallace and Biastoch (2016), the breakpoint analysis used paired conductivity-chloride water quality samples from within the TRCA jurisdiction, but over a longer period of time and including more recent samples (1965 to 2020). Based on samples from within the Humber River watershed, the breakpoint of the relationship of conductivity and chloride concentration occurred at 680 µS/cm. The two resulting relationships used to calculate estimated chloride concentration are:

$$[Cl^-]_{Sc<680\mu S/cm} = 0.1467 \cdot Sc \ - 36.90 \qquad \textit{(Eq. 3)}$$

and

$$[Cl^-]_{Sc\geq 680\mu S/cm} = 0.3226 \cdot Sc \ - 157.50 \qquad \textit{(Eq. 4)}$$

Where $[Cl^-]$ is total chloride in mg/L and Sc is specific conductance in µS/cm.

Benthic Community Analysis

Based on benthic macroinvertebrate samples in six sites along the study reach in 2019 and seven sites in 2020, collected using a Hess sampler (Hess 1941), the community assemblage was assessed from 133 taxa and a total of 25,721 organisms.

Species turnover was assessed across all sites through local contribution to beta diversity (LCBD; Legendre and De Cáceres 2013). LCBD indicates the relative uniqueness of a site in terms of its composition. Relative influence in each taxon to species turnover can also be measured as species contribution to beta diversity (SCBD; Legendre and De Cáceres 2013). We focused on taxa that contribute greater than the mean SCBD for species turnover, and further identified the classification of these taxa that were applied to sensitivity metrics.

The four-sensitivity metrics that were applied on the full community assemblage were the Hilsenhoff biotic index (HBI; Hilsenhoff 1988), EPT (Ephemeroptera, Plecoptera, Trichoptera) community taxa richness (based on the highest taxonomic resolution identified), and the relative abundance (%) of Chironomidae-Oligichaeta. Each of these metrics represent a gradient of sensitivity or tolerance to pollution as a proxy to water quality, such that the presence of a greater proportion of pollution-tolerant species indicates poorer water quality conditions, generally. The spatial relationship of the sites upstream (site 7) to downstream (site 1) was considered by determining if there was any spatial correlation of these metrics in addition to species turnover through a Principal Component Analysis (PCA).

Comparison to Top-draw Pond

To further investigate pond performance related to the thermal and chloride conditions a given stream is receiving from a pond outlet, the three bottom-draw study ponds were compared to a nearby top-draw pond. This type of SWMP is designed to draw water from the surface of the pond rather than from the bottom, which may cause differences in the temperature and chloride levels in the water leaving the pond. The same thermal load and chloride concentration analyses completed for the bottom-draw ponds were repeated with data from the top-draw pond. By comparing the conditions at the outlets of both SWMP design types, some conclusions can be made about which design produces more desirable outcomes in temperature and chloride levels entering aquatic ecosystems.

Study Results

Temperature

Overall, results show that SWMPs are contributing to the thermal load of instream conditions (Figure 3). We find that the SWMPs contributed to about 92% and 100% of the thermal load in the study reach over the study period in 2019 and 2020, respectively (Table 2). This represents an increase of 0.16˚C in 2019 and a decrease of 0.01˚C in 2020 for instream temperature for the entire study period (June-September). September is the only month in both years that a positive difference is seen in average instream temperature from the upstream to downstream locations of the reach (Table 3). The contribution of the ponds is approximately 82% and 100% of the thermal load in September, equating to a temperature increase of 0.5˚C and 0.4˚C in 2019 and 2020, respectively (Table 2).

Reviewing the largest thermal event in 2019 (September 10; 1 day), it is found that the SWMPs within the study reach on average contribute to 74% of the thermal load (Figure 4). This represents an increase in temperature of about 1.18˚C for instream temperature. The largest thermal event in 2020 occurred on August 2 and the ponds contributed to 95% of the thermal load, equating to an increase in temperature of 0.47 ˚C (Figure 4).

Notably, all ponds exceeded the 24˚C criteria within outflows at some point in time during the study, except Maggie pond in 2020 (Figure 6, Table 3)(MNRF, 2016). However, instream temperature upstream and downstream also exceeded this thermal threshold and generally the temperature difference between upstream and downstream was negative, implying that cooling occurred over the study reach (Figure 5, Table 3).

Table 2. Summary of thermal loads summed across all three ponds and the difference in thermal loads upstream and downstream of the study reach, in megajoules and degrees C. The approximate percent contribution of the ponds to the difference in thermal loads is also shown.

Time Period	Thermal Loading (mJ)		Thermal Loading (°C)		Approx. % Contribution of Ponds (+/- 10%)
	Ponds	Difference (Downstream – Upstream)	Ponds	Difference (Downstream – Upstream)	
June 2019	946097.5	1061745.6	17.2	0.1	89
July 2019	566634.7	577900.5	19.8	-0.2	98
August 2019	378155.8	351337.9	19.1	-0.2	100
September 2019	513451.9	629831.7	18.0	0.5	82
Entire (2019)	2404339.9	2620815.6	18.2	0.2	**92**
June 2020	2046799.5	2061571.9	18.6	-0.1	99
July 2020	1539489.2	1447472.0	20.7	-1.1	100
August 2020	2985034.3	3065214.2	21.4	0.2	97
September 2020	975960.7	897087.2	18.4	0.4	100
Entire (2020)	7547283.7	7471345.4	20.0	-0.01	**100**

Table 3. Summary of average temperature for upstream (US), downstream (DS), the DS-US difference, each pond, and the average of all ponds for each month, and the entire 2019 and 2020 study periods (June 1 – September 27, both years). Values in brackets represent the range of temperatures observed.

Time Period	Average Temperature (°C)						
	Carberry	**Farad**	**Maggie**	**All Ponds**	**Upstream**	**Downstream**	**Difference (Downstream – Upstream)**
June 2019	18.9 (15.3-23.1)	16.4 (11.8-22.6)	12.8 (10.1-20.9)	16.0 (12.7-21.9)	17.8 (11.7-23.8)	17.8 (12.1-23.1)	~0
July 2019	23.6 (19.7-26.6)	18.1 (11.4-25.8)	14.7 (12.1-25.0)	18.8 (15.4-25.4)	22.2 (17.1-28.0)	21.5 (16.5-27.4)	-0.7
August 2019	22.6 (14.6-24.5)	16.9 (11.7-23.1)	15.3 (13.5-23.1)	18.3 (15.0-23.4)	19.6 (14.8-24.5)	19.1 (14.7-23.5)	-0.5
September 2019	19.3 (17.9-21.2)	16.4 (11.6-20.0)	15.8 (13.5-20.5)	17.2 (14.4-20.5)	16.2 (11.9-21.0)	16.5 (12.1-20.9)	0.3
Entire (2019)	21.2 (14.6-26.6)	17.0 (11.4-25.8)	14.6 (10.1-25.0)	17.6 (12.7-25.4)	19.1 (11.7-28.0)	18.8 (12.1-27.4)	-0.3
June 2020	21.4 (17.8-24.8)	15.9 (11.3-23.7)	12.9 (10.2-22.2)	16.8 (13.1-23.1)	19.8 (13.0-25.4)	19.3 (12.7-24.2)	-0.5
July 2020	24.2 (21.8-27.3)	18.5 (12.9-26.4)	15.1 (12.2-23.9)	19.3 (16.3-25.2)	22.7 (18.8-27.3)	21.0 (17.1-24.5)	-1.7
August 2020	22.5 (21.0-24.7)	18.4 (16.1-24.8)	16.2 (14.2-23.4)	19.0 (17.1-24.2)	20.5 (14.6-26.1)	20.1 (14.8-24.8)	-0.4
September 2020	18.5 (15.2-23.1)	15.9 (8.4-22.3)	15.6 (11.5-22.0)	16.6 (11.7-22.3)	15.2 (8.1-22.1)	15.4 (8.6-21.8)	0.2
Entire (2020)	21.8 (15.2-27.3)	17.3 (8.4-26.4)	14.9 (10.2-23.9)	18.0 (11.7–25.2)	19.7 (8.1–27.3)	19.1 (8.6–24.8)	-0.6

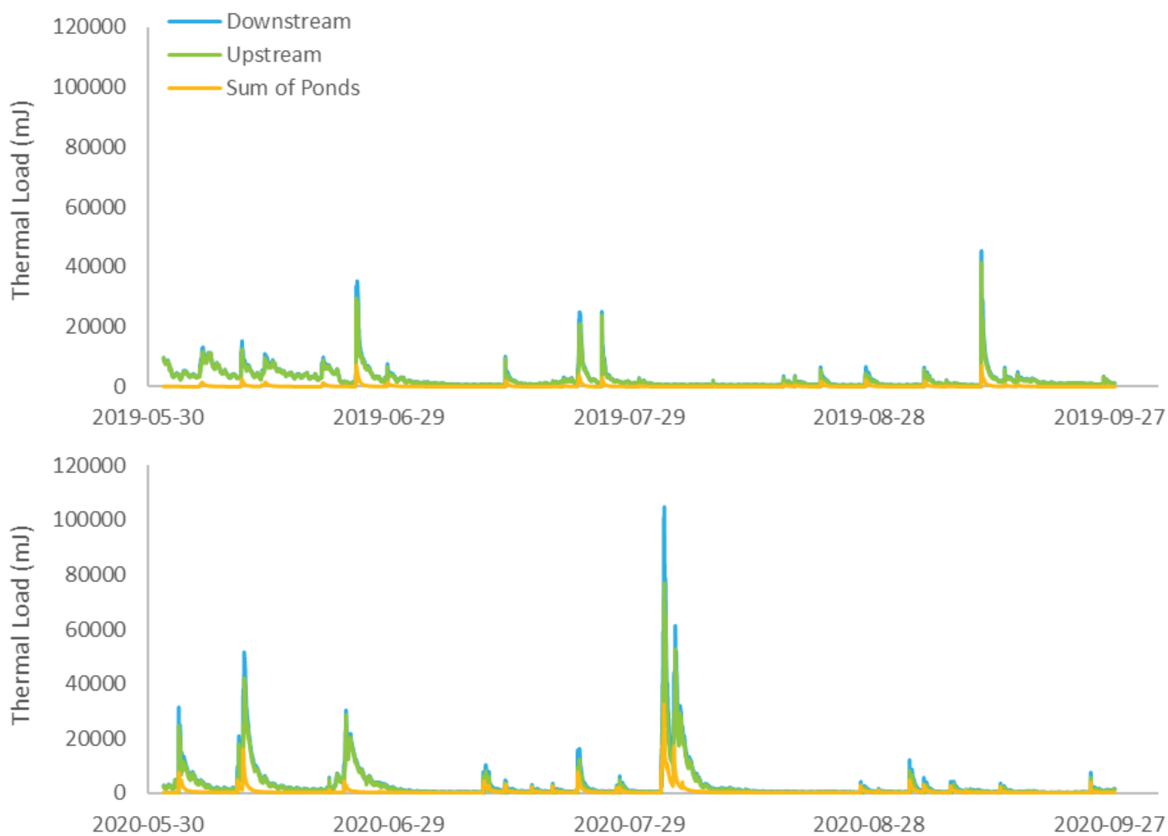

Figure 3. The thermal loading for the upstream, downstream, and the sum of all three ponds combined in the study reach over the entire study period, June 1 – September 27 in 2019 and 2020.

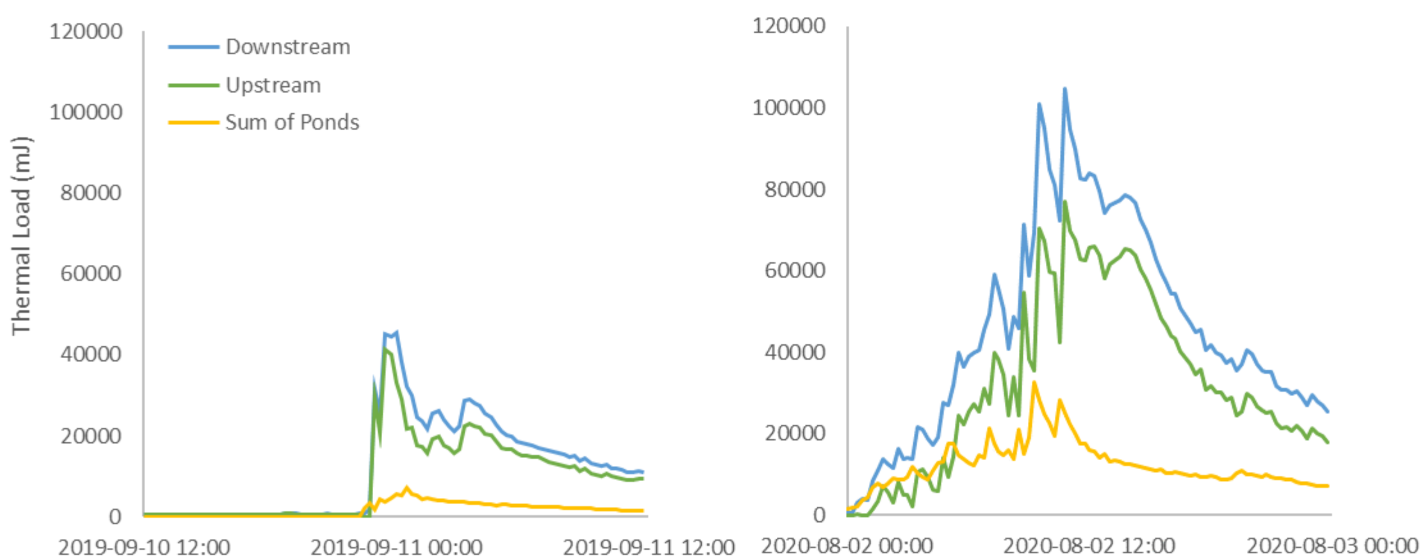

Figure 4. The thermal loading for the upstream, downstream and all three ponds combined for the study reach during large thermal events on September 10-11, 2019 and August 2, 2020.

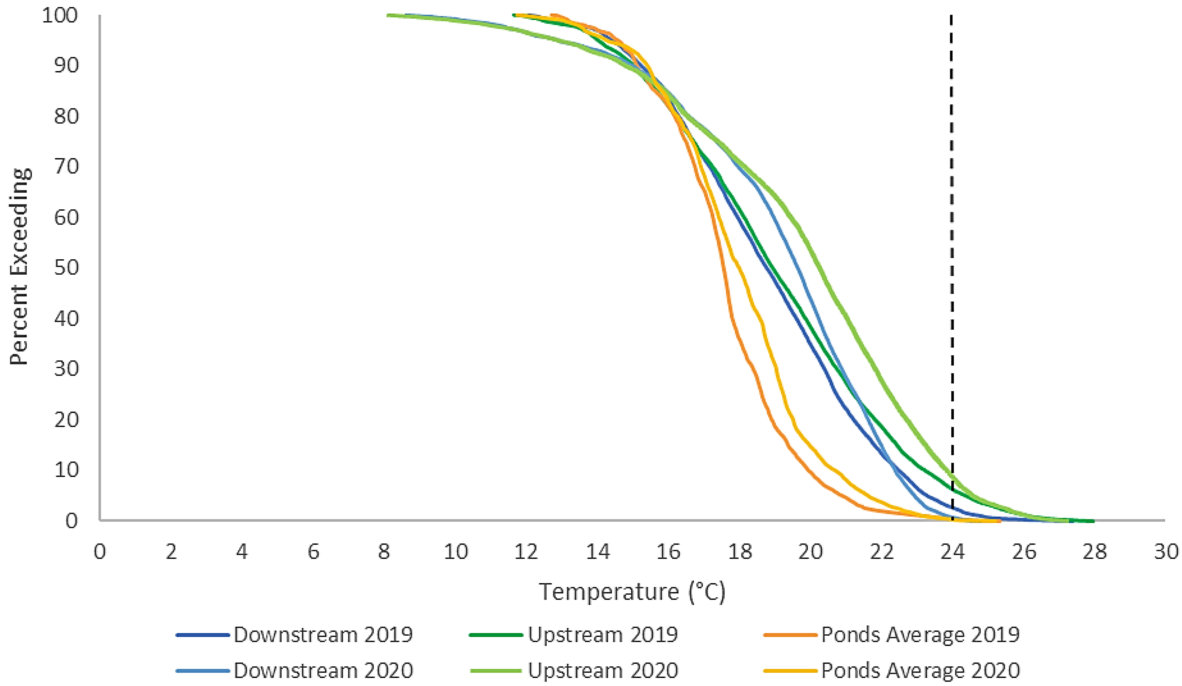

Figure 5. Exceedance plot of temperatures upstream, downstream and the average across all three ponds for the 2019 and 2020 study periods. The dashed line references the 24˚C threshold.

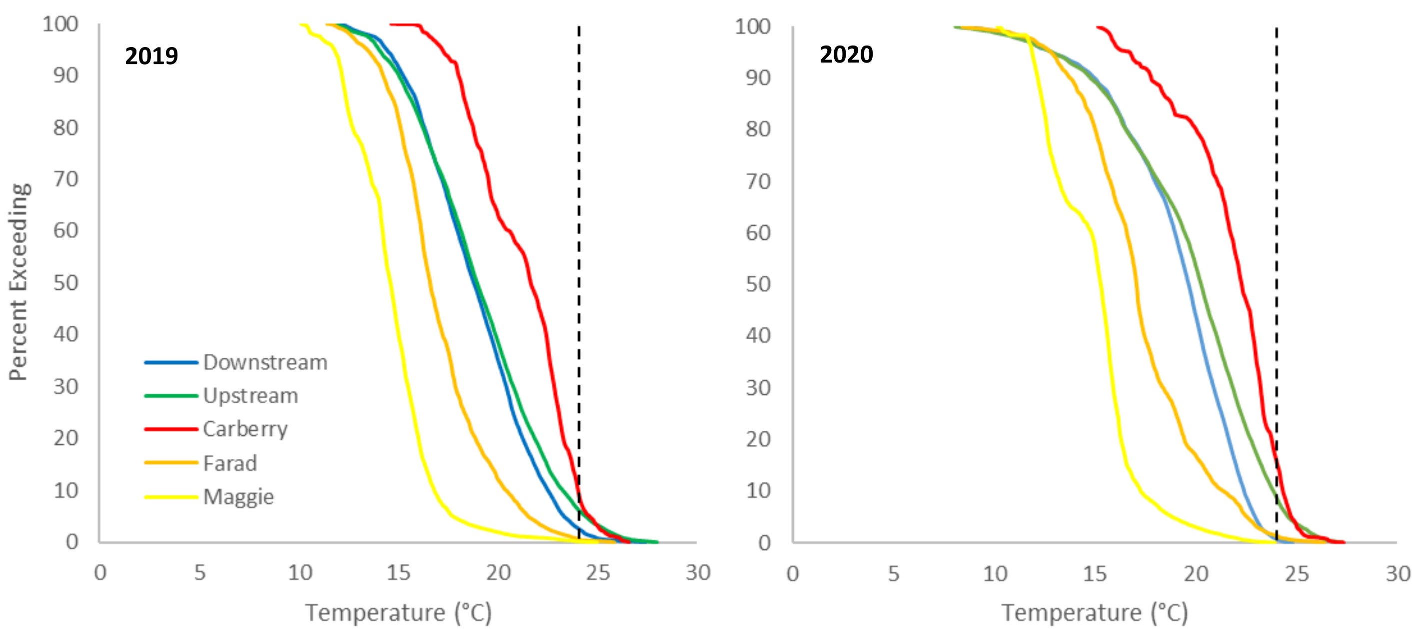

Figure 6. Exceedance plot for temperatures upstream, downstream and each study pond for the 2019 (left) and 2020 (right) study periods. The dashed line references the 24˚C threshold.

Chloride

For chloride, it is found that the SWMPs are large contributors to chloride concentrations observed within the stream (Figure 7-10). We find that the instream chloride condition exceeded both short-term acute (640 mg/L; 24-96 hours) and long-term chronic (120 mg/L; ≥7 days fish/invertebrates; ≥24 hours for algae/plants) thresholds for the protection of aquatic life at some point in the two years of monitoring (Figure 7-10, Table 4)(CCME, 2011).

The short-term acute chloride threshold was exceeded at the Maggie pond on one occasion for 46.25 hours (1.9 days) between November 14 and 16, 2019. This threshold was exceeded at the Carberry pond for 291.8 hours (12.2 days) and 28.8 hours at the Maggie pond in March 2020 (Figure 7 & 10).

In 2019, the long-term chronic chloride threshold was exceeded upstream, downstream, and at the Farad and Maggie ponds on two occasions, once in October and again in November (Figure 7 & 10). The duration of these events varied between 7.2 and 22.1 days. Notably, the Farad pond, upstream and downstream locations exceeded 120 mg/L of chloride for 94%, 88%, and 80% of the period of analysis, respectively. In 2020, the chronic threshold was exceeded for more than 7 days at some point between March 18 – December 3 at all SWMP outlets and both upstream and downstream (Figure 7 & 10). The Farad pond, upstream and downstream locations again showed the most concerning results, exceeding the 120 mg/L threshold 100%, 94%, and 93%, respectively.

Further, during events where relatively higher chloride concentrations were observed, the SWMPs appear to contribute to the increase in these concentrations downstream (Figure 8). During large events on November 18, 2019 and December 1, 2020, the approximate contribution of the ponds to the difference in chloride loads between upstream and downstream locations is 11% and 24%, respectively. It is important to note that the majority of this chloride contribution during events clearly comes from the Maggie pond (e.g. Figure 17). However, on other occasions the upstream location had higher chloride concentrations than the downstream, the ponds showed little effect, or even lagged behind timing of chloride concentration increases downstream.

Table 4. Summary of average estimated chloride concentration for upstream (US), downstream (DS), the DS-US difference, each pond, and average across all ponds for each month, and the entire 2019 and 2020 study periods (October 5 – November 19, 2019 and March 18 – December 3, 2020). Values in brackets represent the range of concentrations observed.

Time Period	Average Estimated Chloride (mg/L)						
	Carberry	Farad	Maggie	Ponds	Upstream	Downstream	Difference (Downstream – Upstream)
October 2019	56.7 (26.3-78.4)	308.8 (31.8-539.6)	138.0 (0-494.2)	167.8 (21.4-252)	136.2 (53.7-293)	127.3 (3.1-174)	-8.9
November 2019	40.8 (25.2-75.1)	495.3 (47.9-611.8)	465.2 (200.4-3915.8)	333.8 (39.5-1485)	167.7 (58.6-352)	206.4 (58.5-1147)	38.7
Entire (2019)	50.2 (25.2-78.4)	384.7 (31.8-611.8)	271.1 (0-3915.8)	235.3 (21.4-1485)	149.0 (53.7-352)	159.5 (3.1-1147)	10.5
March 2020	785.6 (601.2-878.3)	512.3 (369.8-608.2)	505.0 (14.0-760.9)	601.0 (423.6-725.6)	135.8 (37.4-426.6)	50.2 (20.4-602.4)	-85.6
April 2020	418.8 (358.3-600.8)	335.2 (284.4-394.3)	386.4 (6.6-535.3)	380.2 (245.5-455.1)	143.5 (102.4-245.1)	154.7 (75.1-253.8)	11.2
May 2020	373.9 (238.5-611.1)	317.1 (239.7-368.5)	416.6 (0-595.4)	369.2 (193.8-455.3)	163.7 (128.7-271.7)	180.4 (40.5-274.8)	16.7
June 2020	228.2 (66.5-331.7)	249.1 (147.1-309.5)	383.2 (0-553.1)	286.8 (109.3-381.5)	167.7 (60.5-227.7)	187.5 (10.8-239.5)	19.8
July 2020	163.4 (40.5-202.3)	248.9 (157.5-298.9)	438.5 (0-586.8)	283.6 (106.3-359.2)	204.4 (133.2-241.6)	217.0 (0-327.4)	12.6
August 2020	55.8 (27.8-129.5)	173.7 (96.2-242.7)	281.2 (0-501.9)	170.2 (42.9-284.6)	173.1 (43.3-243.3)	168.9 (0-279.9)	-4.2
September 2020	79.3 (54.8-96.7)	172.8 (154.1-211.9)	174.8 (0-287.1)	142.3 (80.1-178.7)	173.3 (136.4-201.6)	154.9 (0-229.9)	-18.4
October 2020	88.0 (49.9-228.2)	151.8 (122.9-188.1)	105.3 (0-209.4)	115.0 (67.3-160)	147.2 (126-171)	136.9 (10.7-164)	-10.3
November 2020	111.3 (73.8-156.6)	181.1 (140.6-235.4)	94.6 (0-634.7)	129.0 (96.8-310)	173.8 (133-434)	182.1 (54.8-593)	8.3
Entire (2020)	222.5 (27.8-878.3)	244.5 (96.2-608.2)	300.5 (0-760.9)	252.0 (16.5-774)	167.7 (37.4-527)	173.6 (0-593)	5.9

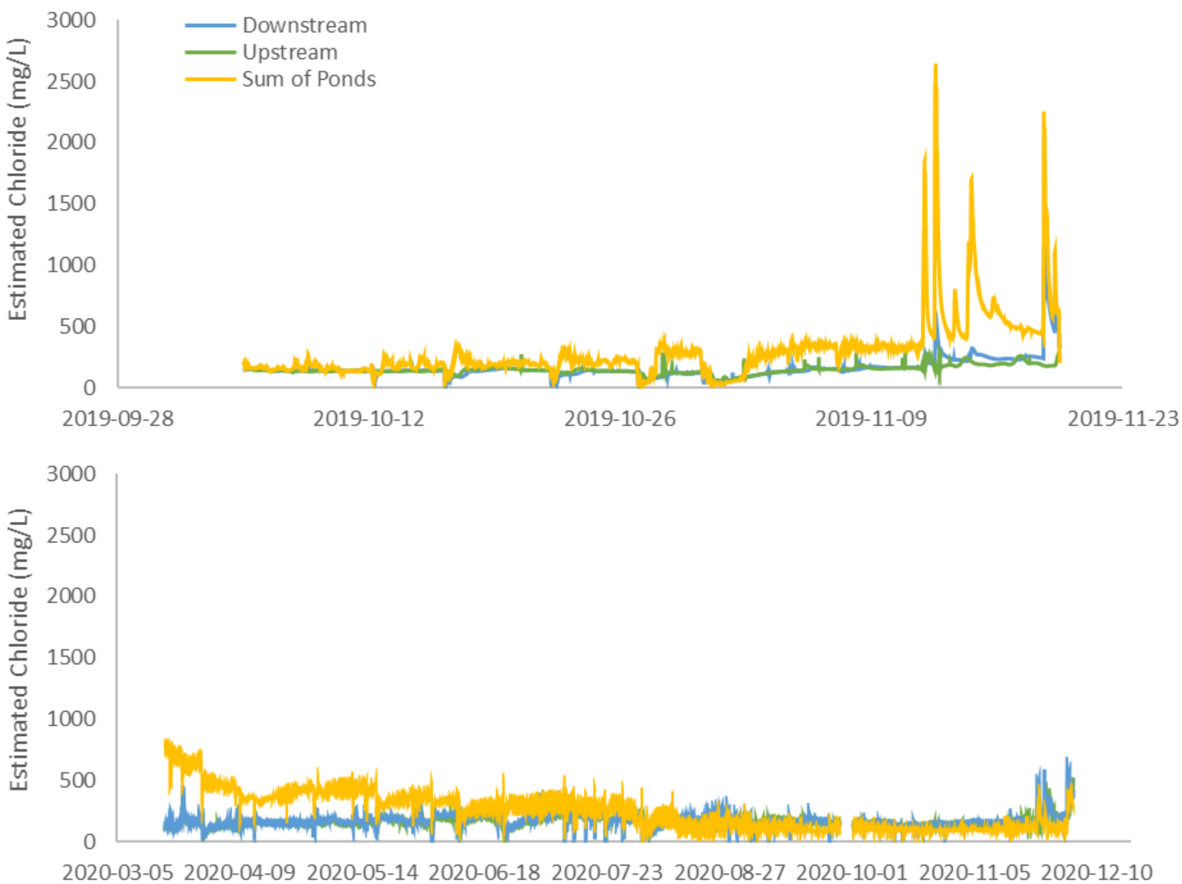

Figure 7. The estimated chloride concentrations in mg/L for the upstream, downstream and the sum of all three ponds combined in the study reach over the entire study period, October 5 – November 19, 2019 and March 18 – December 3, 2020.

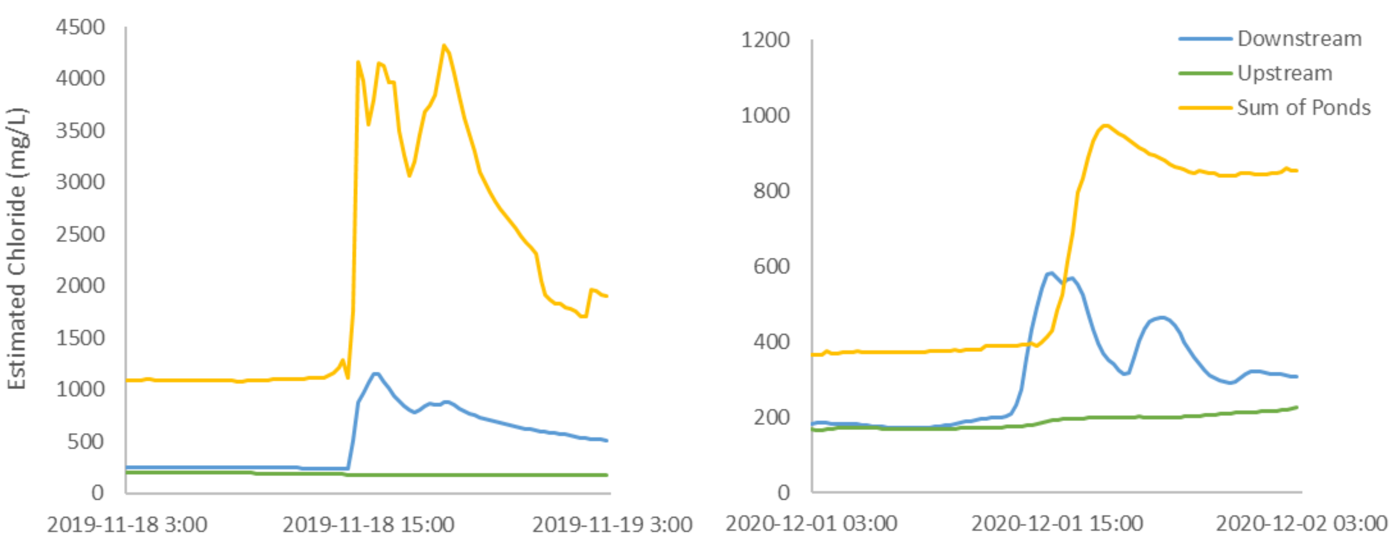

Figure 8. The estimated chloride concentration or the upstream, downstream and all three ponds combined over the period of 03:00-03:00 on November 18-19, 2019 and December 1-2, 2020.

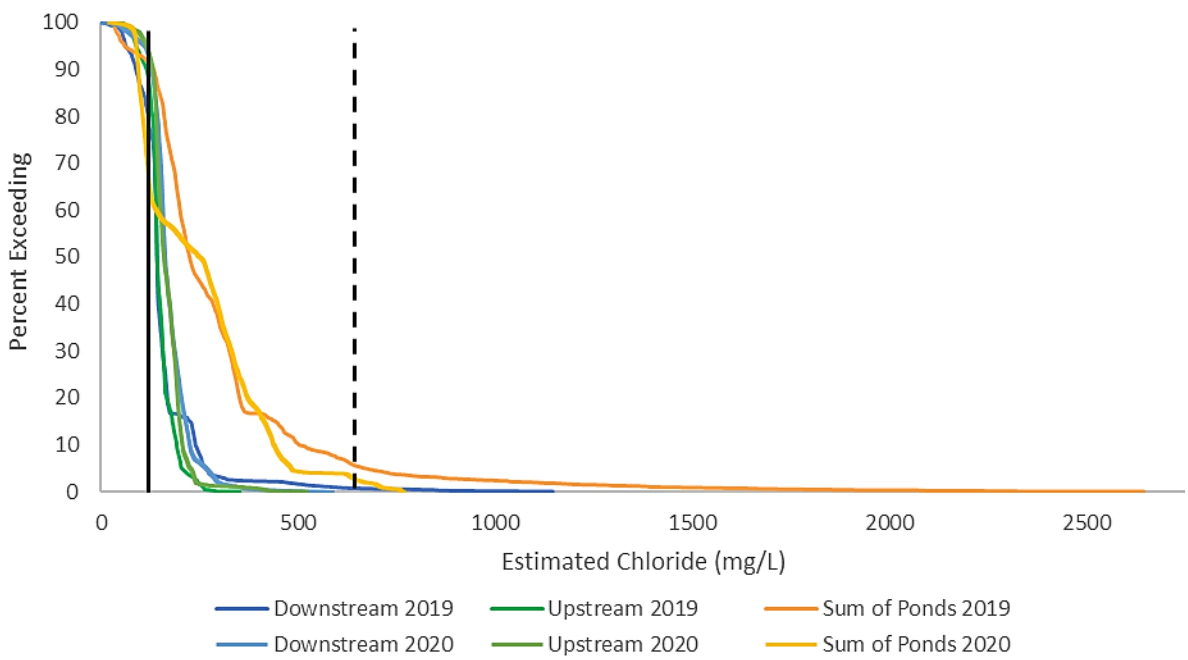

Figure 9. Exceedance plot of chloride concentrations upstream, downstream and the sum load across all three ponds for the study period of October 5 – November 19 in 2019 and March 18 – December 3 in 2020. The dashed line references acute (short-term) and solid line chronic (long-term) thresholds for the protection of aquatic life.

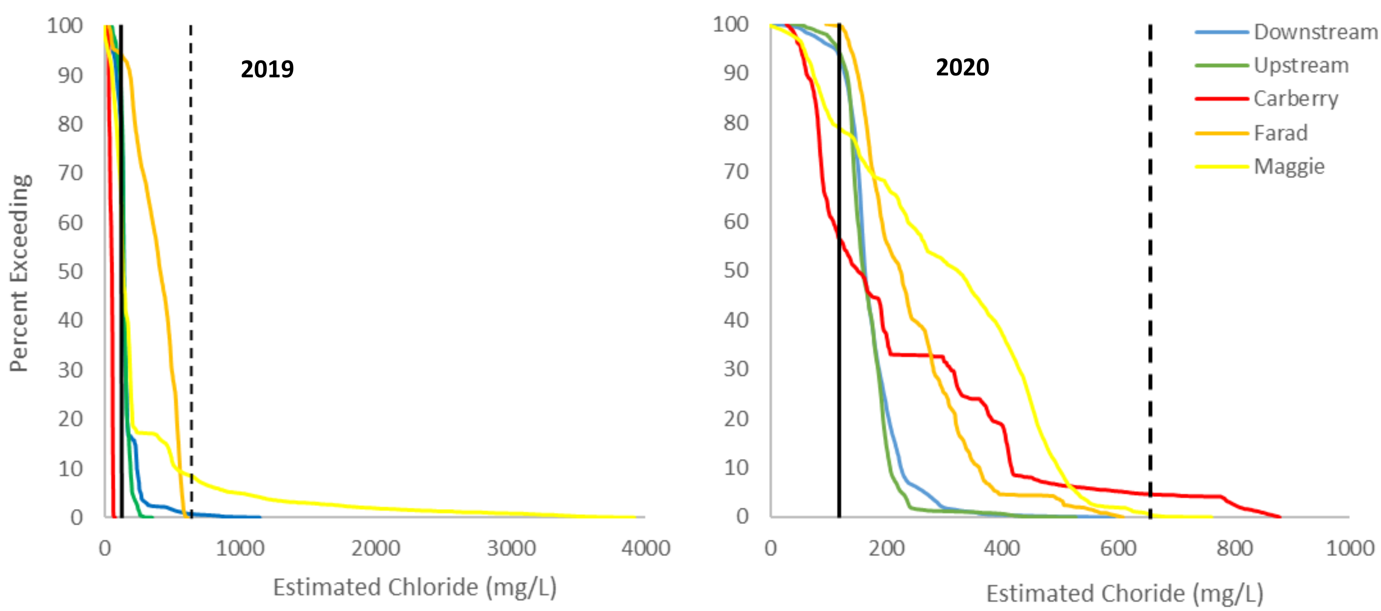

Figure 10. Exceedance plot for chloride concentrations upstream, downstream and the sum load across all three ponds for the study period of October 5 – November 19 in 2019 (left) and March 18 – December 3 in2020 (right). The dashed line references acute (short-term) and solid line chronic (long-term) chloride thresholds for the protection of aquatic life.

Comparison of 2019 and 2020 Thermal and Chloride Results

Warmer temperatures were observed instream and at the SMWP outlets in 2020 compared to 2019. In 2020, the average temperature upstream, downstream, and pond outlet temperature was 0.6, 0.3, and 0.4°C warmer, respectively, compared to 2019 (Table 3).

Although temperatures were relatively warmer in 2020, the SWMPs appear to have a cooling effect in both years. In fact, the difference in average temperature between the upstream and downstream locations of the stream reach showed increased cooling in 2020. The cooling between the upstream and downstream locations increased by 0.3°C from a -0.3°C difference in 2019 to a -0.6°C difference in 2020, over the duration of the study period from June 1 to September 27 (Table 3).

Similarly, the thermal loading increased in 2020 from 2019, and the approximate contribution of the three stormwater ponds to this loading increased by 8% (Table 2). The largest thermal loading event in 2020 was also greater than the largest event in 2019 (Figure 4).

When comparing chloride results in October-November between 2019 and 2020, the average chloride concentrations observed at the pond outlets decreased in 2020 from 2019 (Figure 7, Table 4). Further, unlike 2019, the instream chloride concentrations were higher on average than at the pond outlets in 2020, over the same time period (Figure 7, Table 4). Lastly, in both years, the Farad pond had the highest average chloride concentrations at the outlet, while the Maggie pond appears to have the greatest influence on chloride levels during stochastic events, with a maximum chloride concentration far greater than the other two SWMPs (Table 4).

Benthic Communities

For species turnover, there was a downstream-upstream relationship based on local contribution to beta diversity (LCBD; Figure 11). Downstream sites 1 and 2 were considered significantly different due to species turnover ($p < 0.05$, Table 5). This spatial difference in species turnover was mainly driven by taxa sensitive to pollution found mainly upstream and more tolerant taxa found downstream as indicated by the HBI values (Figure 11, Table 5).

Of the 133 taxa, seven taxa cumulatively contributed greater than the mean species contribution to beta diversity (SCBD) in 2019, and 11 taxa in 2020. These significant taxa were mainly EPT followed by Chironomidae-Oligochaeta (Table 3). In 2019, the most influential taxon for species turnover was the winter stonefly genus (*Paracapnia* sp.), which is also the most sensitive to pollution based on the HBI value (Table 6). By contrast, net-spinning caddisflies (*Cheumatopsyche* sp.) became the most influential taxon for species turnover in 2020 (Table 6).

Table 5. Sensitivity metrics for the six sites from upstream (site 7) to downstream (site 1) based on the Hilsenhoff Biotic Index (HBI), EPT richness proportion in the community, Chironomidae-Oligochaeta relative abundance (%), and local contribution to beta diversity (LCBD).

Site	Total Organisms	Taxa Richness	HBI	EPT (Richness Proportion)	Chironomidae-Oligochaeta (Relative Abundance %)	LCBD (* p < 0.05)
2019						
7	1667	43	5.073	0.233	22.1	0.129
6	2967	49	4.178	0.245	26.6	0.130
5	-	-	-	-	-	-
4	901	48	3.627	0.250	10.4	0.141
3	2278	48	4.655	0.292	16.1	0.125
2	1792	64	5.097	0.219	31.3	0.190
1	6325	49	5.378	0.163	16.8	0.284*
2020						
7	1465	42	5.838	0.214	56.5	0.175
6	1784	44	5.207	0.205	35.1	0.068
5	870	42	4.403	0.214	28.0	0.127
4	919	42	5.266	0.238	36.6	0.094
3	1199	40	4.997	0.325	14.3	0.120
2	1741	46	6.414	0.196	56.3	0.321*
1	1813	40	5.185	0.175	21.5	0.095

Table 6. List of taxa classification that cumulatively contribute greater than half of the species contribution to beta diversity (SCBD) representing seven taxa in 2019 and ten taxa in 2020.

Classification	SCBD	EPT	Chironomidae-Oligochaeta	HBI
2019				
Paracapnia sp.	0.1122	•		1
Hydropsyche betteni	0.0835	•		7
Chimarra sp.	0.0770	•		4
Cheumatopsyche sp.	0.0686	•		5
Ceratopsyche slossonae	0.0662	•		4
Microtendipes sp.	0.0425		•	5
Caenis sp.	0.0392	•		6
2020				
Cheumatopsyche sp.	0.1173	•		5
Paracapnia sp.	0.0543	•		1
Rheotanytarsus sp.	0.0521		•	6
Tanytarsus sp.	0.0503		•	6
Chimarra sp.	0.0473	•		4
Microtendipes sp.	0.0425		•	5
Caenis sp.	0.0366	•		6
Dicranota sp.	0.0300			3
Paratanytarsus sp.	0.0274		•	6
Naididae (Family)	0.0262	•		8

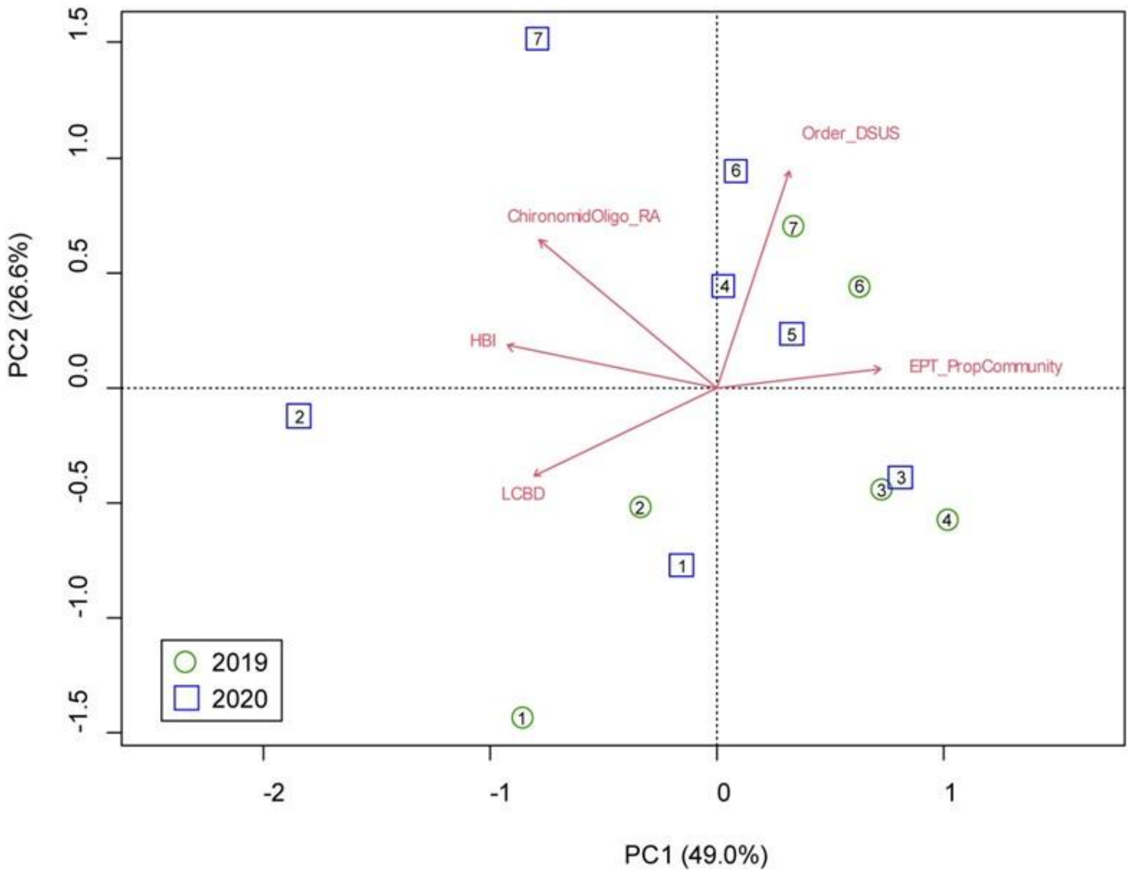

Figure 11: PCA biplot of sensitivity metrics (Hilsenhoff biotic index: HBI, EPT proportion of the community richness: EPT_PropCommunity, relative abundance of Chironomidae-Oligochaeta: ChironomidOligo_RA), species turnover (LCBD), and the spatial relationship of sites (downstream to upstream order: Order_DSUS) in relation to the six sampling sites (green circles) in 2019 and seven sampling sites (blue squares) in 2020.

Comparison of Bottom-draw Study Ponds to Top-draw Pond

When comparing the conditions measured at the outlets of the three bottom-draw study ponds to a nearby top-draw pond, the top-draw pond has a higher average temperature and larger thermal load than all three study ponds in every month and thus, over the entire period of analysis (Figure 12 & 13, Table 7). The top-draw pond also produces a wider range of temperatures at the outlet than the bottom-draw ponds (Figure 13, Table 7). These results are to be expected as water leaving the top-draw pond is drawn from the surface where water is exposed to solar radiation, wind, and ambient air temperature.

Details of the average chloride concentration at the reference top-draw pond and each bottom-draw pond are summarized in Table 8. Over the entire monitoring period the top-draw pond had lower chloride concentrations at the outlet on average, compared to the three bottom-draw ponds (Figure 15 & 16). This is seen across most months between June and November, except for the Farad pond in August, and the Farad and Maggie ponds in November.

Interestingly, the Carberry pond appears to function like a top-draw pond such that it produces higher temperatures and lower chloride concentrations relative to the other two bottom-draw ponds (Figure 12 & 15, Table 7 & 8). One explanation for this observation could be that most water leaving the Carberry pond exits through the pond outlet rather than through an underground diversion, as was seen at the Farad and Maggie ponds.

A comparison of the conditions during a large precipitation event on August 2 shows higher thermal loading at the top-draw pond outlet compared to the bottom draw ponds (Figure 12 &14). The chloride response to the large event on December 1-3 shows a smaller and delayed release of chloride compared to the Maggie pond, while the other two bottom-draw ponds show no response (Figure 17).

Table 7. The average temperature and summed thermal loading recorded at the outlet of each bottom-draw study pond compared to the outlet of a nearby top-draw pond for each month and the entire of June 1 – September 30, 2020. Values in brackets represent the range of temperatures observed.

Time Period	Average Temperature (°C)				Thermal Loading (mJ)			
	Carberry	Farad	Maggie	Top-draw	Carberry	Farad	Maggie	Top-draw
June 2020	21.4 (17.8-24.8)	15.9 (11.3-23.7)	12.9 (10.2-22.2)	24.2 (19.8-29.7)	956499.5	370686.0	719614.0	1080702.9
July 2020	24.2 (21.8-27.3)	18.5 (12.9-26.4)	15.1 (12.2-23.9)	27.1 (23.9-32.0)	725438.6	276521.4	537529.2	969401.9
August 2020	22.5 (21.0-24.7)	18.4 (16.1-24.8)	16.2 (14.2-23.4)	25.2 (22.0-29.1)	1091738.1	743110.1	1150186.1	1159774.5
September 2020	18.5 (15.2-23.1)	15.9 (8.4-22.3)	15.6 (11.5-22.0)	19.6 (15.0-26.2)	507478.4	150108.4	318373.9	522948.3
Entire 2020	21.8 (15.2-27.3)	17.3 (8.4-26.4)	14.9 (10.2-23.9)	24.0 (15.0-32.0)	3281154.5	1540426.0	2725703.2	3732827.5

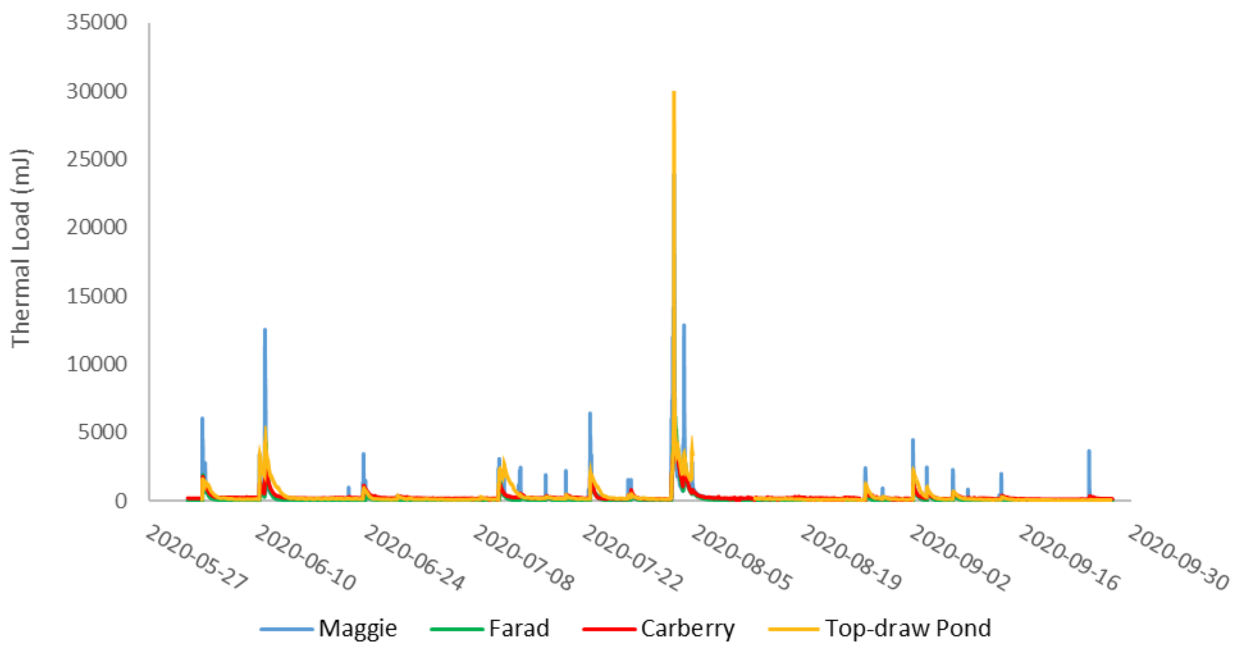

Figure 12. The thermal loading for the three bottom-draw ponds and the reference top-draw pond over the entire study period, June 1 – Sept. 27 in 2020.

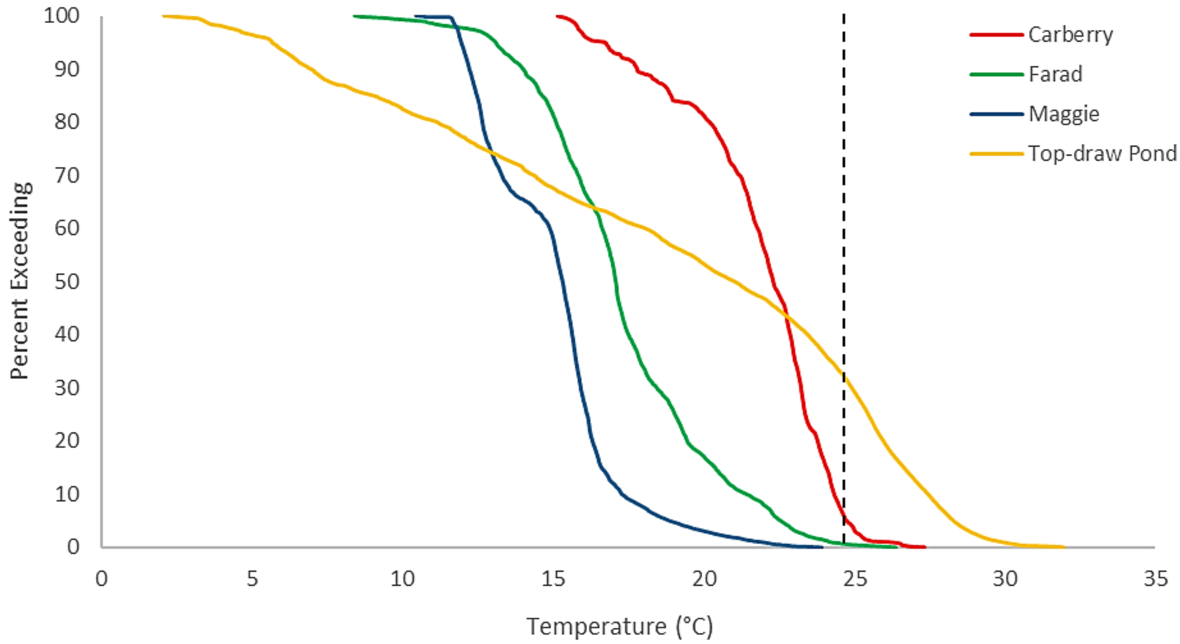

Figure 13. Exceedance plot for average temperature at the outlet of each bottom-draw pond and the reference top-draw pond for the study period in 2020. The dashed line references the 24˚C threshold.

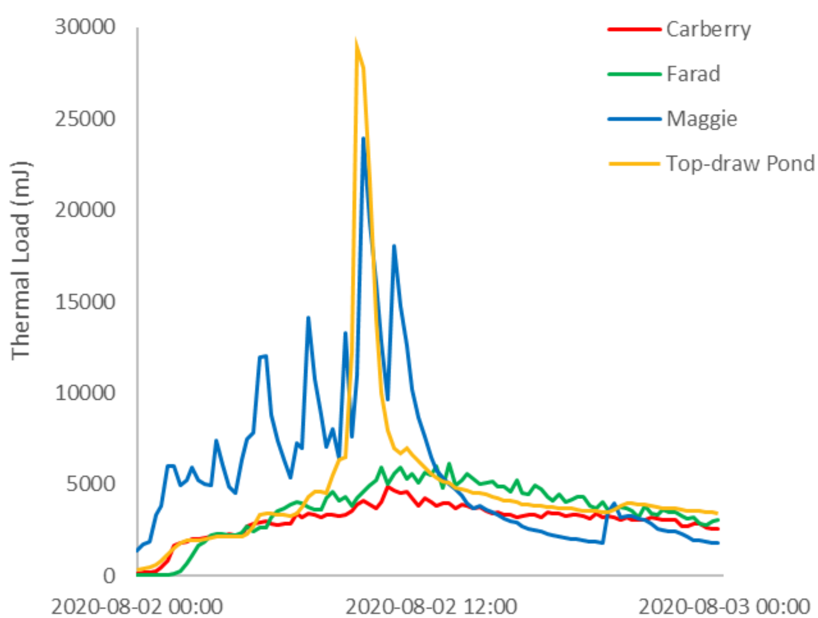

Figure 14. Comparison of a large thermal loading event at the three bottom-draw ponds along the study reach and at the reference top-draw pond on August 2, 2020.

Table 8. The average estimated chloride concentration at the outlet of each bottom-draw study pond compared to that of a nearby top-draw pond for each month and the entire period of June 1 – December 3, 2020. Values in brackets show the range of concentrations observed.

Time Period	Average Estimated Chloride (mg/L)			
	Carberry	Farad	Maggie	Top-draw
June 2020	228.2 (66.5-331.7)	249.1 (147.1-309.5)	383.2 (0-553.1)	191.8 (48.1-425.6)
July 2020	163.4 (40.5-202.3)	248.9 (157.5-298.9)	438.5 (0-586.8)	125.8 (62.1-190.1)
August 2020	55.8 (27.8-129.5)	173.7 (96.2-242.7)	281.2 (0-501.9)	71.4 (19.4-106.1)
September 2020	79.3 (54.8-96.7)	172.8 (154.1-211.9)	174.8 (0-287.1)	51.3 (0-134.2)
October 2020	88.0 (49.9-228.2)	151.8 (122.9-188.1)	105.3 (0-209.4)	78.1 (38.7-96.3)
November 2020	111.3 (73.8-156.6)	181.1 (140.6-235.4)	94.6 (0-634.7)	118.7 (80.1-234.0)
Entire 2020	**121.1** (27.8-331.7)	**196.6** (96.2-309.5)	**249.3** (0-671.7)	**108.0** (0-425.6)

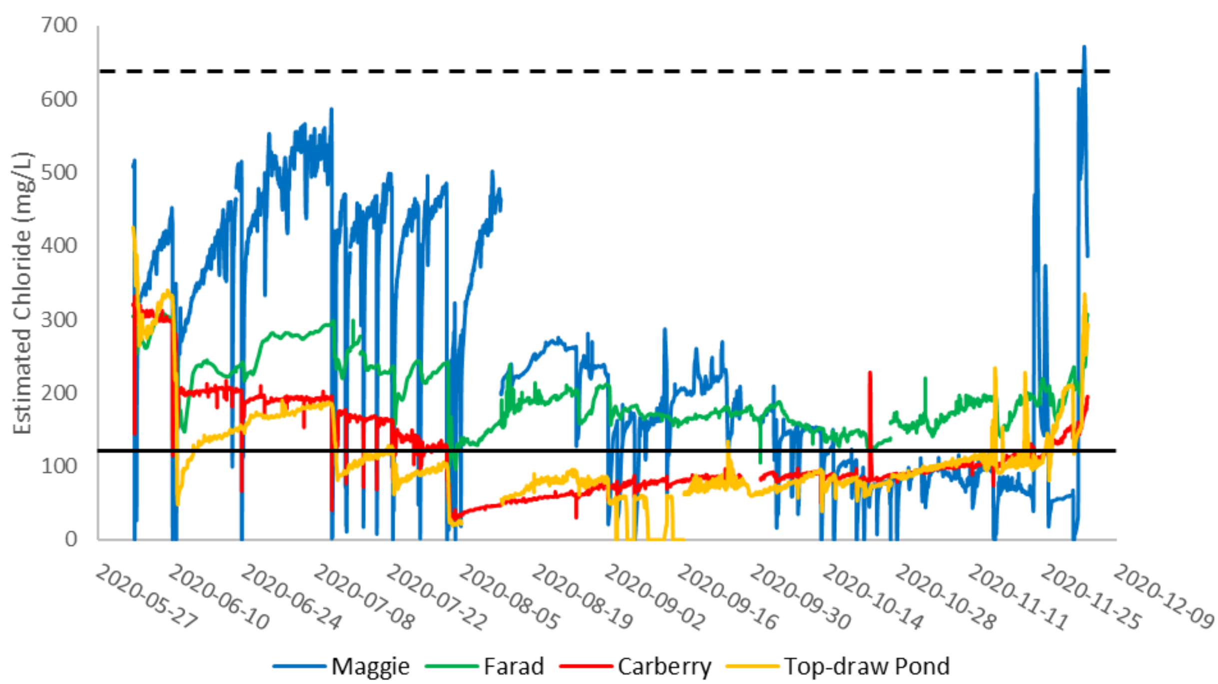

Figure 15. The average estimated chloride concentrations for the three bottom-draw ponds and the reference top-draw pond over the entire study period, June 2 – December 3 in 2020.

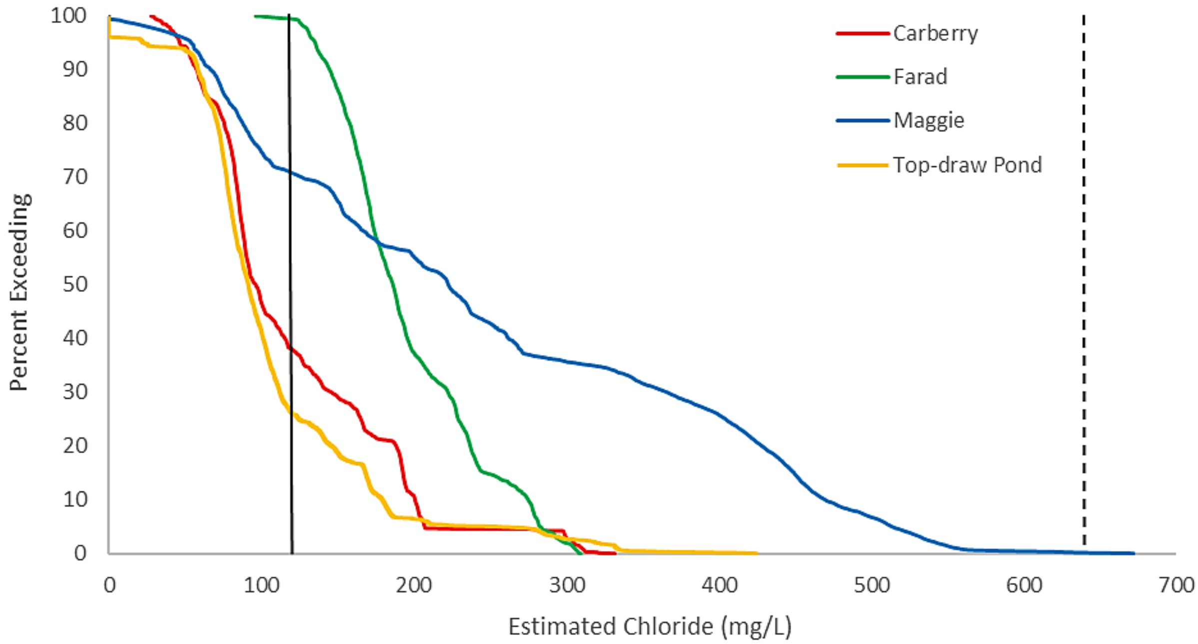

Figure 16. Exceedance plot for average chloride concentrations at the outlet of each bottom-draw pond and the reference top-draw pond for the study period in 2020. The dashed line references acute (short-term) and solid line chronic (long-term) thresholds for the protection of aquatic life.

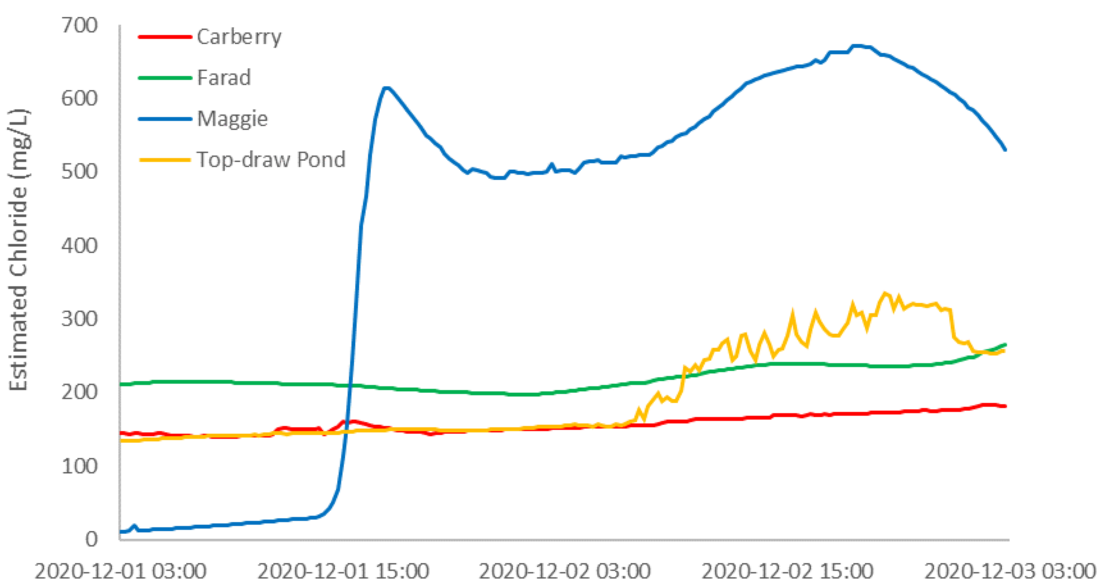

Figure 17. Comparison of a chloride concentrations during a large precipitation event at the three bottom-draw ponds along the study reach and at the reference top-draw pond between 03:00-03:00 on December 1-3, 2020.

Climate Conditions

Given the natural variation in weather conditions between years, it is important to acknowledge the effect this can have on the differences in observed thermal loading and chloride concentrations between 2019 and 2020 in our study reach.

Comparison of Climate Relative to 1961-1990 Baseline

Hourly air temperature and daily precipitation data collected at a station located at Pearson International Airport were obtained from the Environment and Climate Change Canada website. This data was used to calculated monthly, seasonal, and yearly averages to be used to compare to a 1961-1990 baseline condition.

When comparing 2019 and 2020 weather data to the baseline, 2019 was a relatively warmer and wetter year and 2020 was a record warm year, with average precipitation. Over the summer period, which is of interest for the analysis of temperature loading, both 2019 and 2020 had record warm conditions compared to a 1961-1990 baseline. In November, where event-based surges in chloride tend to occur, precipitation levels were average in both years.

Comparison of 2019 to 2020 Weather

At the annual scale, weather in 2020 was warmer and drier compared to 2019. This pattern is also seen across most months and seasons. In the summer months of June-August 2020 had a warmer summer than 2019 with record warm conditions across all three months in 2020. Further, the average air temperature for summer 2020 was 1.53°C higher than in 2019. The warmer summer conditions in 2020 clearly contributed to the higher average temperatures observed instream and at all SWMP outlets (Table 3). This likely also influenced the larger magnitude of thermal loading events observed in 2020 compared to 2019 (Figure 3 & 4).

As for precipitation events, there was average precipitation in the summer and fall for both years, however precipitation was 49.0mm greater in the summer and 56.4mm greater in the fall in 2019 compared to 2020. The greater precipitation levels during both seasons in 2019 likely explains the larger chloride concentrations observed during events and overall (Figure 8 & 9).

Future Climate

With future climate change, the frequency of stochastic weather events and average air temperatures are predicted to increase. As seen in our study area between 2019 and 2020, relatively warmer temperatures and increased precipitation appear to coincide with larger thermal loading and chloride concentrations, respectively. Over the two years, thermal issues were observed at the Carberry pond, which produced temperatures closer to that of a top-draw pond. Chloride issues were seen at the Maggie pond, such that surges in chloride occurred during large precipitation events. Current benthic macroinvertebrate survey results point to degraded habitat conditions for aquatic benthic species downstream of the SWMPs on the study reach. Predicted future climate conditions will likely further exacerbate the large thermal and chloride stress SWMPs pose to aquatic ecosystems and the negative effects this has on habitat quality. The effects of future climate should be considered when considering improvements to pond design. Modelling of future climate for the area and the average condition seen at each pond can help to explore the relationships between SWMP performance, climate conditions, and the impact to aquatic ecosystems.

Table 9a: 1961-1990 baseline precipitation conditions.

Precipitation Statistic 1961-1990	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Winter (Dec-Feb)	Spring (Mar-May)	Summer (Jun-Aug)	Fall (Sep-Nov)	Year (Dec-Nov)
30-Year Minimum (mm)	23.8	10.9	9.1	11.9	24.9	11.7	25	15	14.5	14	9.1	11.7	68.7	58.1	131.3	93.0	565.4
30-Year 20th Percentile (mm)	43.4	26.02	20.5	33.9	44.4	40.0	38.3	52.7	47.0	44.2	38.3	40.6	118.6	153.5	174.3	145.5	684.2
30-Year 80th Percentile (mm)	84.2	67.44	70.1	78.3	87.5	96.9	93.6	108.0	124.8	107.1	88.5	93.2	195.3	238.6	281.0	270.2	853.6
30-Year Maximum (mm)	112.8	82	107.2	121.2	111.9	140	150.9	182.3	163.6	212.3	134.9	161.8	241.1	303	335.9	326.6	971.3

Table 9b: 1961-1990 baseline air temperature conditions.

Air Temperature Statistic 1961-1990	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Winter (Dec-Feb)	Spring (Mar-May)	Summer (Jun-Aug)	Fall (Sep-Nov)	Year (Dec-Nov)
30-Year coldest monthly/season record (°C)	-10.1	-11.7	-10.8	-4.6	2.8	8.5	15.2	17.3	16.8	13.1	6.1	0.2	-8.5	3.8	17.3	6.9	5.9
30-Year 20th Percentile (°C)	-6.1	-9.6	-8.0	-2.5	4.6	10.4	16.4	19.6	18.4	14.2	7.1	2.2	-6.8	4.5	18.3	8.1	6.6
30-Year 80th Percentile (°C)	-1.1	-4.1	-3.6	0.7	7.6	14.2	18.6	21.4	20.8	15.7	10.3	4.0	-4.5	6.9	19.9	9.9	7.8
30-Year warmest monthly/season record (°C)	0.7	-0.8	-1.5	3.2	8.8	15.9	19.6	22.9	21.9	19.3	13.6	6.4	-1.8	9.7	20.7	11.5	8.7

Table 10a: Precipitation (mm) in 2019 & 2020 compared to a 1961-1990 baseline. December values are from the previous year (December 2018 and 2019).

Year of Study	Dec Sum	Jan Sum	Feb Sum	Mar Sum	Apr Sum	May Sum	Jun Sum	Jul Sum	Aug Sum	Sep Sum	Oct Sum	Nov Sum	Winter (Dec-Feb)	Spring (Mar-May)	Summer (Jun-Aug)	Fall (Sep-Nov)	Year (Dec-Nov)
2019	52	67.6	76	63.8	93.4	97.6	108.6	104.2	44.6	46.4	136.2	40.8	195.6	254.8	257.4	223.4	931.2
	avg.	wet	wet	avg.	wet	wet	wet	avg.	dry	avg.	record wet	avg.	wet	wet	avg.	avg.	wet
2020	71.2	131.4	52	53.8	41	43.2	49.8	67.6	91	40.8	60.4	65.8	254.6	138	208.4	167	768
	avg.	record wet	avg.	avg.	dry	avg.	avg.	avg.	avg.	dry	avg.	avg.	record wet	dry	avg.	avg.	avg.

Table 10b: Air temperature (°C) in 2019 & 2020 compared to a 1961-1990 baseline. December values are from the previous year (December 2018 and 2019).

Year of Study	Dec Avg.	Jan Avg.	Feb Avg.	Mar Avg.	Apr Avg.	May Avg.	Jun Avg.	Jul Avg.	Aug Avg.	Sep Avg.	Oct Avg.	Nov Avg.	Winter (Dec-Feb)	Spring (Mar-May)	Summer (Jun-Aug)	Fall (Sep-Nov)	Year (Dec-Nov)
2019	0.09	-6.49	-4.04	-0.96	6.29	11.93	18.49	23.52	21.34	17.83	10.94	1.04	-3.48	5.75	21.12	9.94	8.33
	warm	avg.	avg.	avg.	avg.	avg.	avg.	record warm	warm	warm	warm	cool	warm	avg.	record warm	warm	warm
2020	-0.68	-1.22	-2.68	3.17	5.95	12.38	20.76	24.97	22.21	16.87	9.51	7.01	-1.53	7.17	22.65	11.13	9.85
	warm	warm	warm	warm	avg.	avg.	record warm	record warm	record warm	warm	avg.	record warm	record warm	warm	record warm	warm	record warm

Limitations and Suggestions for Future Research

As stated, estimated chloride concentrations in this study are estimated from measured conductivity. Based on the observed relationship between chloride levels instream and at SWMP outlets, and the higher chloride levels in the spring and summer months compared to the fall in 2020 (Table 4), this suggests that factors other than road salt are driving conductivity levels at the pond outlets. That is, chloride coming from ponds is likely not the main cause of higher conductivity instream. Based on this observation, better ways to measure chloride more directly should be investigated if continuing to monitor the effects of SWMPs on chloride inputs to streams, to allow for more accurate results.

Moreover, since stochastic events have been pinpointed as a concern, further research into the relationship between different SWMP designs or levels of functioning and the size of rain events that SWMPs can handle may be important to protecting aquatic life from exposure to sudden large changes in temperature or chloride. For example, looking at the performance as well as the longevity of the functioning of different SMWP designs could be crucial to ensuring the outflows from these ponds are kept below the temperature and chloride thresholds critical to protecting aquatic life. Zeroing in on which designs produce the most favourable outcomes and require less frequent repair would ultimately save time, money, and effort.

Lastly, the results of the benthic invertebrate analysis indicate that future work looking directly at the effects that thermal and chloride stress from SWMPs have on aquatic life would be useful to informing effective pond design and performance criteria.

Conclusions

Results of this two-year study find that the SWMPs in the study reach contribute to the thermal loads observed instream over the study reach. In fact, the SWMPs account for most of the differences in thermal load observed within the 1 km study reach in 2019 and 2020 (92% and 100%, respectively; Table 2). In both years, the Carberry pond was notably warmer than the other two ponds, in fact its conditions were much similar to that of the reference top-draw pond (Table 7). The Farad and Maggie ponds produced temperatures cooler than the instream for most of the study period (Figure 6, Table 3). This pattern shows that the downstream Farad and Maggie ponds counteract the upstream Carberry pond, rather than creating cumulative warming as was originally predicted. Thus, despite a warmer summer in 2020 for the study area, the SWMPs generally appear to cool instream temperatures more than warm them during both years of this study, unlike findings of similar studies on the thermal effects of SWMPs (Hester & Bauman, 2013; Sabouri *et al.*, 2013). Altogether, this suggests that bottom-draw ponds may be effective at reducing thermal loads instream and appear to lower the risk to Redside Dace populations.

The SWMPs do appear to contribute to increases in chloride levels downstream, especially during events, where comparatively high chloride concentrations were observed (Figure 8). The chloride results of this study concur with previous research on SWMPs in that the ponds appear to reduce acute chloride stress on the ecosystem by increasing chronic levels (Fanelli, Prestegaard & Palmer, 2019; Haake & Knouft, 2019). Particularly, the Farad pond exceeded 120mg/L of chloride for the majority of the study period in both years. However, this pattern was observed for the upstream and downstream locations as well. It is also interesting to note that the Maggie pond appears to be flashier in its chloride levels, with no storage effect. Thus, the Maggie pond proves to be the most problematic during stochastic rain events, where the maximum chloride concentrations were far greater than at the other bottom draw ponds and the reference top-draw pond (Figure 16, Table 4 & 8). These results together speak to individual SWMP performance in that the Farad pond appears produce consistently high chloride inputs to the stream, while the Maggie pond poses an issue with creating excessive chloride inputs during large precipitation events. This suggests that there is likely a trade-off between the thermal and chloride performance of SWMPs as it relates to in-stream conditions for aquatic ecosystems.

In general, results suggest that event-based thermal and chloride releases from SWMPs appear to have the largest impact on the instream conditions during the two-year study period. The fact that stochastic events produce spikes in temperatures and chloride levels at pond outlets and downstream (Figure 7 & 9) speaks to SWMP design criteria. This can have adverse effects on aquatic life as fish tend to congregate in outflow channels in the winter because it is warmer. However, this puts fish at risk of exposure to high acute chloride concentrations during events when surges of chloride levels occur at SWMP outlets. In addition, with continuing climate change, the increased frequency of stochastic weather events as well as warmer average air temperatures may further exacerbate the thermal and chloride impacts that SWMPs pose to aquatic ecosystems. In all, the thermal issues observed at the Carberry pond and the chloride issues observed at the Farad and Maggie ponds should be taken into account when considering improvements to pond designs in the future.

The patterns in benthic macroinvertebrate species diversity along the study reach further support the inference that the SWMPs in this study reach are having a cumulative negative effect on the quality of habitat downstream. More sensitive macroinvertebrate species were found upstream compared to downstream (Figure 11, Table 5), which indicates that water quality and habitat suitability for aquatic life may be degraded downstream of the SWMPs.

Lastly, when looking at the results comparing bottom-draw ponds to a reference top-draw pond, it can clearly be seen that the top-draw pond produces outflows with higher temperature but lower chloride concentration, on average. This speaks to the need to further investigate the trade-off between thermal and chloride loading and the link to pond design. Isolating the biggest cause for concern as it relates to aquatic ecosystem health (and pertinent thresholds) will help determine what pond designs may produce more desirable thermal and chloride conditions at the outlet.

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