QUANTIFYING CHLORIDE RETENTION IN URBAN STORMWATER MANAGEMENT PONDS USING A MASS BALANCE APPROACH

By

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Abstract

Quantifying chloride retention in urban stormwater management ponds using a mass balance approach

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Chloride (Cl^-) from runoff containing deicing salts is retained in watersheds after deicing ends, resulting in deleterious effects on aquatic biota. Stormwater management ponds (SWMPs) are known to impact pollutant transport. However, there is little information on what role SWMPs play in the timing and magnitude of Cl^- transport in urban watersheds. This study quantifies the mass of Cl^- retained in two urban SWMPs over varying timescales and the in-stream response to Cl^--laden pond outflows.

The findings suggest that SWMPs likely play a role in watershed-scale Cl^- retention. In the receiving creek, Cl^- pulses corresponded to Cl^- release from the pond. The results of this study suggest that SWMPs concentrate spatially distributed salt inputs and modify the timing and magnitude of their release to receiving streams. This study will help parameterize the role of SWMPs in watershed-scale Cl^- transport models and geospatial models of salt vulnerable areas.

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Contribution of Authors

The manuscript included herein has three authors: Wai Ying Lam (Ryerson University), David Lembcke (Lake Simcoe Region Conservation Authority), and Claire Oswald (Ryerson University).

W.Y. Lam, D. Lembcke, and C. Oswald all contributed to the initial design of the study. W.Y. Lam and C. Oswald made some adjustments to this design over the study period.

W.Y. Lam was responsible for collecting and filtering grab samples as well as calibrating and deploying inlet, outlet, and stream data loggers and consolidating the raw data. D. Lembcke was responsible for deploying in-pond data loggers as well as the inlet and outlet weir and flow loggers and consolidating the raw data.

Data analysis and interpretation were carried out by W.Y. Lam. Text, tables, and figures were drafted by W.Y. Lam and critically reviewed by C. Oswald and D. Lembcke.

W.Y. Lam, D. Lembcke, and C. Oswald all approved of the final version to be published.

1 Introduction

It is widely known that increasing urban development, accompanied by increased impervious area, has negative impacts on streams. In urban areas that experience cold winters, chloride (Cl^-) is a pollutant of note. Cl^- in urban waterways is largely derived from runoff containing deicing salts applied to maintain safe driving conditions (Novotny and Stefan, 2010; Perera *et al.*, 2010; Oswald *et al.*, 2019). It can have deleterious effects on aquatic biota and may pose a threat to drinking water (Walsh *et al.*, 2012; Kaushal, 2016; Wallace and Biastoch, 2016). Cl^- can accumulate in freshwater systems for extended periods; therefore, the negative impacts of elevated Cl^- concentrations can be observed beyond the winter deicing period (Chapra *et al.*, 2009; Winter *et al.*, 2011).

One common method employed by municipalities to mitigate changes to natural water quantity and quality is the installation of stormwater management ponds (SWMPs) (LSRCA, 2011; TRCA, 2012). However, there is a paucity of information on what role SWMPs play in the timing and magnitude of Cl^- export from urbanizing and urban watersheds. In this study, a mass balance approach is employed to quantify Cl^- retention in two SWMPs located in an urbanizing watershed in south-central Ontario. This research aims to address the role that urban SWMPs play in the timing and magnitude of Cl^- transport by (1) quantifying the mass of Cl^- entering and leaving a pair of SWMPs; (2) determining annual, seasonal, and event-scale Cl^- retention for the SWMPs using a mass balance approach; (3) comparing changes in Cl^- retention to in-pond Cl^- concentrations; and (4) quantifying the response of a receiving stream to Cl^- exported from a SWMP. It will inform the parameterization of the role of SWMPs in watershed-scale Cl^- models, as well as the identification of Salt Vulnerable Areas.

2 Literature Review

This review will briefly describe urban impacts on hydrology and explain how stormwater control measures impact water quantity and water quality. It will then delve into Cl^- in urban environments and provide a background on Cl^- transport through SWMPs. Lastly, it will discuss trends and effects of Cl^- in freshwater systems.

2.1 Overview of Urban Hydrology

Urban development has three main impacts on urban hydrology: increased total runoff, altered peak flow characteristics, and compromised water quality (Anderson, 1970). These impacts arise from current urban water management practices, which aim to move storm- and wastewater away from urban areas as efficiently as possible (Mitchell *et al.*, 2001).

Urban development results in increased imperviousness, which restricts the infiltration of water into the ground (Davis and McCuen, 2005). This produces greater surface runoff volumes. Often, stream channels are straightened and streambeds are altered to increase hydraulic capacity and prevent erosion (Marsalek *et al.*, 2006). Transportation corridors, such as roadways and accompanying storm sewers, ditches, and culverts, efficiently direct overland flow into these channelized streams, and the overall speed of runoff is increased (Meierdiercks *et al.*, 2010a). Overall, urban areas have reduced water storage capacity and generate more runoff, which travels more quickly, than natural areas.

Greater surface runoff and reduced natural water storage in urban environments also facilitate the transport of urban pollutants to freshwater systems (Walsh *et al.*, 2012). Common pollutants include but are not limited to oil and grease from vehicles, Cl^- from road salt, nutrients from fertilizers, and elevated water temperature due to the urban heat island effect (Walsh *et al.*, 2012; Ehrenfeld *et al.*, 2015; Booth *et al.*, 2016).

2.2 Impacts of Urbanization on Water Quantity

Urban drainage systems, which may include structures such as stormwater pipes, street gutters, roadside swales, other surface channels, and stormwater management ponds, play a significant role in establishing urban hydrologic response (Meierdiercks *et al.*, 2010b). Land clearing and paving and grading of surfaces for urban development contribute to major changes in catchment water quantity (GHD, 2017). Land clearing results in reduced vegetation, decreasing the available interception storage and protective covering of natural vegetation (Leopold, 1968). This, along with paving of surfaces, leads to reduced potential for infiltration, reduced surface roughness, and more surface runoff (Davis and McCuen, 2005). Grading of surfaces reduces the available depression storage (Douglas, 2015). This reduction in storage and roughness increases the volume and decreases the travel time of runoff, resulting in increased peak runoff rates (Sauer *et al.*, 1983). In heavy precipitation events, overbank flows may occur. Over time, large runoff volumes will also lead to channel erosion, flooding issues, and possible infrastructure damage (Douglas, 2015).

2.2.1 Stormwater Control Measures

In the attempt to compensate for lost natural storage and minimize flood risk, many municipalities require the installation of stormwater control measures (SCMs) (Environment Canada, 2018). While SCMs are unable to restore pre-development flow regimes, they can extend lag times relative to existing development conditions (James and Dymond, 2011). Implementation of SCMs has been found to reduce mean annual runoff by 35% or more at the watershed scale (Meierdiercks *et al.*, 2010a).

Since 2003, the province of Ontario has promoted a ‘treatment train’ approach to stormwater control. This prioritizes prevention (e.g. by effective street cleaning, reducing street widths and driveway lengths), then lot-level controls (e.g. rainwater harvesting, soakaway pits), then conveyance controls (e.g. pervious pipe systems, grass swales), and lastly, end-of-pipe controls (e.g. stormwater wetlands, wet ponds) (Ontario Ministry of the Environment, 2003; Bradford and Gharabaghi, 2004). Other examples of SCMs include infiltration pits, roadside swales, green roofs, and permeable pavements (Casey *et al.*, 2013). Among the most common structures for end-of-pipe stormwater management are stormwater detention basins, also known as stormwater management ponds.

2.2.2 Stormwater Management Ponds

SWMPs allow runoff collected in stormwater sewers to pool temporarily (LSRCA, 2011). The pond’s outlet releases water from the pond to a nearby stream system at a controlled rate, thus controlling the flow of water and reducing dramatic changes in streamflow volume during rain events (TRCA, 2012). These ponds are poor mimics of natural systems (Rooney *et* *al.*, 2015), but are effective in reducing peak discharges at the lot level (LSRCA, 2011). Results of stormwater control modelling in Baltimore County's Dead Run watershed suggest that the presence of SWMPs impacts peak discharges more significantly than land use type or percent impervious area (Meierdiercks *et al.*, 2010b).

Most municipalities use SWMPs to slow the movement of stormwater runoff, and the ponds can cover up to 4% of the municipality's area (Douglas, 2015). The ponds are often linked to green corridors with ecological value that double as public recreational areas (Brilly, 2007; Hassall and Anderson, 2015). These corridors are intended to preserve areas with significant biodiversity and wildlife habitat while connecting the public to trails and greenways (Moore and Hunt, 2012; Houck, 2015).

2.3 Impacts of Urbanization on Water Quality

Increasing urbanization has adverse impacts on the water quality of terrestrial groundwater and aquatic systems (Walsh *et al.*, 2005; Douglas, 2011). Pollution from both point and non-point sources can be transported to streams along a variety of pathways, resulting in elevated nutrient, sediment, and pathogen loads (Ehrenfeld *et al.*, 2015). In Ontario's urbanizing Lake Simcoe watershed, the primary pollutants are phosphorus, nitrogen, and Cl^-; phosphorus from urban runoff and septic systems, nitrogen from agriculture, and Cl^- from surface runoff (O'Connor *et al.*, 2012).

2.3.1 Pollution of Urban Waterways

For most of North America in the 20th century, point source discharges represented the majority of pollutant inputs—mostly organic wastes, toxic compounds, and nutrients—to freshwater systems (Davis and McCuen, 2005). Controlling regulations were put into practice and largely addressed the problem in the 1970s. However, non-point sources of pollution such as stormwater runoff were not regulated in the same way (Davis and McCuen, 2005).

Today, most pollutant inputs to aquatic systems are from non-point sources. Pollutants are derived from multiple sources such as industrial effluent, nutrient-rich runoff from upstream agricultural areas, and urban development (Walsh *et al*., 2012). Oils and grease, road salt, heavy metals, pesticides and fertilizers, and pathogenic microorganisms are common contaminants in urban and urbanizing areas (Gray and Becker, 2002; Davis and McCuen, 2005; Soller *et al.*, 2005; Arnone and Perdek Walling, 2007; Egemose *et al.*, 2015). Surface runoff transports these pollutants rapidly to receiving streams (Bazinet *et al.*, 2010; Cooper *et al.*, 2014). Because of this, the degradation of stream water quality is highly correlated to increasing watershed population and impervious surface area (Walsh *et al.*, 2005; Oswald *et al.*, 2019).

2.3.2 Water Quality in Stormwater Management Ponds

SWMPs are thought to improve stream water quality largely because the ponds offer opportunities for runoff to pool. As the water is detained, suspended solids and associated pollutants are allowed to settle out (Gallagher *et al.*, 2011; Schwartz *et al.*, 2017). Biogeochemical and physicochemical processes can remove pollutants. For example, SWMPs allow interaction of denitrifiers in sediment with nitrate-laden water, increasing potential for denitrification (Bettez and Groffman, 2012). Additionally, buffer areas around the ponds are usually landscaped with natural vegetation for nutrient uptake and filtering of sediment (Moore and Hunt, 2012). These mitigating processes work well at low flow rates but are less effective during high flow events when retention time is short and currents may scour the bottom of the pond, carrying out previously captured sediments and pollutants (Davis and McCuen, 2005).

Therefore, there is some concern around whether SWMPs are exacerbating downstream impacts of polluted stormwater runoff. For example, in Delaware, eight SWMPs failed to prevent the nearly complete downstream loss of sensitive benthic macroinvertebrate taxa once the catchment exceeded 20% impervious area (Maxted and Shaver, 1997). SWMP water and sediment frequently exceed Canadian Water Quality Guidelines (CWQGs) for contaminants such as chromium, lead, zinc, and PAHs, especially as they age (Bishop *et al*., 2000; LSRCA, 2011). There are also pollutants, such as Cl^-, that do not settle out once the water is pooled; it is unclear whether and how SWMPs treat such pollutants (Ehrenfeld *et al.*, 2015).

2.4 Cl^- in Urban Environments

Cl^- is abundant in the environment (Kelly *et al.*, 2008; Tyree *et al.*, 2016). Sources of Cl^- to ecosystems can be naturally occurring, such as natural salt deposits, rock weathering, and dry deposition (Oswald *et al.*, 2019). Anthropogenic sources of Cl^- include water softeners, wastewater treatment effluent, agriculture, and road deicing salt (Kelly *et al.*, 2008).

In rural watersheds, agricultural runoff accounts for a significant proportion of environmental Cl^-, due to the use of KCl fertilizers (Corsi *et al.*, 2015). However, in urban areas, road salt (97% sodium chloride (NaCl) in Canada) is the dominant contributor of Cl^- and increased salinity to aquatic systems (Kaushal *et al.*, 2005; Chapra *et al.*, 2009; Winter *et al.*, 2011; Oswald *et al.*, 2019). NaCl dissolves readily in meltwater and contributes to elevated Cl^- concentrations in ground and surface waters year-round (Judd, 1970; Todd and Kaltenecker, 2012).

Small ponds and streams draining large urban and urbanizing areas are the most sensitive to road salt inputs (Mayer *et al.*, 1999). In urbanizing watersheds in southern Ontario, stream Cl^- loads are highest in the spring (54% of annual total); the summer and fall months account for approximately 12% of the annual load (O'Connor *et al*., 2012).

2.4.1 Road Salt Usage in North America

Road salt application on paved surfaces in a typical North American city can average 2.2 kg m^{-2} per year (Meriano *et al.*, 2009). This number increases with increasing urbanization, as does stream Cl^- concentrations (Corsi *et al.*, 2015). It has been found that this increase is not linear. In a ten-year study conducted by Wallace and Biastoch (2016), mean Cl^- concentration increased by 48%, while road density only increased by 6%. This may be because, while government-level management of road salt application has generally improved, private applicators (e.g. homeowners, businesses, building operators) are not subject to regulation (Labashosky, 2015; Ontario Ministry of the Environment, Conservation, and Parks, 2017; Environment Canada, 2018).

Canada's current Code of Practice (Environment Canada, 2018) for the management of road salts came into effect in 1999 after a five-year assessment of Cl^- salts and brines used in road-deicing was carried out under the Canadian Environmental Protection Act. The Code recommends the development of Salt Management Plans and implementation of best management practices as outlined in the Syntheses of Best Practices developed by the Transportation Association of Canada. The Plans are meant to be focused on safety, efficiency and cost-effectiveness and should include details on spreading, Salt Vulnerable Areas, salt storage, training, and monitoring (TAC, 2013). The best management practices are focused on reducing negative effects of road salts by placing an emphasis on correct quantity, location, and timing of road salt application (TAC, 2013; Environment Canada, 2018).

The City of Toronto had a 26% reduction in the rate of road salt applied after implementing mitigations from the Code of Practice, and it was thought that this might provide ecological benefit to fourteen to twenty-eight species of fish, invertebrates, and plants (Kilgour *et al.*, 2014). However, these benefits are likely undermined in urbanizing watersheds where road networks, and thus areas of road salt application, continue to expand (O'Connor *et al.*, 2012).

2.4.2 Cl^- Transport Through Urban Watersheds

The Cl^- ion does not settle, is not removed biologically, is highly soluble and not subject to any considerable degree of degradation. These properties make Cl^- an ideal tracer for watershed urbanization over time (Thunqvist, 2004; Chapra *et al.*, 2009; Kaushal *et al.*, 2014).

Many natural and manmade streams and ponds receive runoff directly via overland flow and indirectly via subsurface flow from urban streets and highways (Oswald *et al.*, 2019). When deicing salt is transported into waterways, some is transported via overland flow, while some accumulates in the shallow subsurface (Meriano *et al.*, 2009). In these cases, salt concentrations in both nearby groundwater and surface water increase rapidly (Löfgren, 2001). Small amounts of Cl^- may be retained by plants, but more is retained in soils and groundwater (Kelly *et al.*, 2008). Up to 77% of Cl^- from road salt applied in a given deicing season may be retained in the watershed (Novotny and Stefan, 2010; Perera *et al.*, 2010).

These Cl^- transport pathways are illustrated through watershed-scale mass balance studies, which often use road area along with provincial and municipal road salt application data (Chapra *et al.*, 2009; Oswald *et al.*, 2019) and estimates for private applications (Meriano *et al*., 2009) to deduce inputs, and conductivity monitoring to deduce outputs (Kelly *et al*., 2008; Chapra *et al*., 2009; Meriano *et al*., 2009).

Baseline Cl^- concentrations increase over time as continued salt input overwhelms the system's ability to recover to background concentrations before the next deicing season begins (Corsi *et al.*, 2015). In the warm seasons, Cl^- can still be released into stream networks via baseflow. Such multi-season impacts suggest that extended-duration exposure to Cl^- in urban streams would be possible, placing some aquatic organisms at risk outside of the winter period during dry weather flow conditions (Perera *et al*., 2010).

2.4.2.1 Cl^- Transport Through Stormwater Management Ponds

SWMPs are an accepted best management practice for attenuating peak flows but have been criticized as inefficient with regards to mitigating negative water quality impacts (Gold *et al.*, 2017). Without frequent maintenance such as sediment removal, SWMPs may promote long-term storage and gradual release of Cl^- and may play a role in mobilization of other roadway-derived contaminants (Casey *et al.*, 2013).

Casey et al. (2013) assessed salt accumulation in surface waters of ponds, groundwater beneath ponds, and ground and surface water of adjacent floodplains. They noted that road salt application had year-round impacts on Cl^- concentrations in surface water of ponds and groundwater beneath ponds, and that specific conductance of groundwater in floodplains draining SWMPs was consistently elevated. They concluded that subsurface leakage from SWMPs is a source of groundwater salinity, and that Cl^--enriched groundwater can be discharged to surface water year-round.

Snodgrass et al. (2017) carried out a similar study in three different watersheds with differing degrees of stormwater control, including SWMPs. They found that Cl^- concentrations in pond outflows were at their maximum in February, declining by June, and at their minimum in November. This pattern occurred in all watersheds, but watersheds with SWMPs experienced lesser declines. These watersheds also experienced higher baseflow conductivity and larger conductivity spikes following storm events. In agreement with Casey et al. (2013), they concluded that SWMPs were not preventing the release of Cl^- in high concentrations to streams and were releasing highly contaminated flows year-round.

Incoming stormwater, with elevated Cl^- concentrations, can enter a pond as a buoyant jet (tending to rise), interflow (tending to mix), or a sinking jet (tending to sink), depending on incoming flow rate and existing in-pond stratification (Marsalek, 2003). Marsalek (2003) observed densimetric stratification in an urban SWMP, with the Cl^- concentration at the surface being 20% to 50% that of the concentration at the bottom. Strong stratification can inhibit vertical mixing and thus, aeration of bottom layers (Duan *et al.*, 2016). The low levels of oxygen at the bottom of the pond, combined with high concentrations of Cl^-, results in toxic environments for certain biota, and may stimulate contaminant release from the sediment (Schwartz *et al.*, 2017). Depending on the depth at which pond water is released, the impact on downstream Cl^- concentrations may change. For example, ponds that are designed to release stormwater through bottom outlets throughout the winter may prevent strong stratification (Marsalek, 2003).

The role that SWMPs play in the timing and magnitude of Cl^- transport through urban and urbanizing watersheds is not well understood. While the effects of road salting practices on soil, surface water, groundwater, and aquatic communities were recognized by the 1970s (Judd, 1970; Bubeck *et al.*, 1971; Huling and Hollocher, 1972; Kunkle, 1972; Crowther and Hynes, 1977), Cl^- in stormwater ponds had not been studied until the early 2000s. Prominent research to date detailing Cl^- in stormwater management ponds and their environs includes the study by Casey et al. (2013) describing the year-round impacts of road salt application, and the study by Snodgrass et al. (2017) describing pond Cl^- concentrations in different watersheds. The former did not carry out a mass balance, and the latter did not consider differing temporal scales.

2.4.3 Long-Term Cl^- Concentration Trends in Freshwater Systems

Elevated Cl^- first became a concern in Ontario in the mid-1960s, when Alfred Beeton (1965) reported that Cl^- concentrations had been rising in the Great Lakes starting as early as the late 19th century. O'Connor and Mueller (1970) developed Cl^- models that corroborated Beeton's findings and suggested that the increases were primarily due to industrial discharges and runoff of deicing salts. Industrial controls around the Great Lakes introduced in the 1960s started to slow the long-term build-up of Cl^- (Chapra, 2009).

Kelly et al. (2008) conducted a long-term study of Cl^- concentrations and road salt use in rural New York from 1986 to 2005; while road salt use did not increase, stream Cl^- concentrations increased by an average of 1.5 mg L^{-1} per year. This lag effect likely arose from subsurface build up from long-term road salt application (Kelly *et al.*, 2008; Casey *et al.*, 2013).

Winter et al. (2011) examined Cl^- concentrations over time in and around Ontario's Lake Simcoe and found that, between 1993 and 2007, lake Cl^- concentrations increased by up to 10 mg L^{-1} per year. Cl^- fluxes were positively correlated with the proportion of urban land and number of roads drained (Winter *et al*., 2011). Slightly south of Lake Simcoe, in Toronto, Wallace and Biastoch (2016) found that in 2002, 33% of winter stream samples exceeded CWQG chronic Cl^- limits of 120mg L^{-1}; in 2012, 58% of samples exceeded CWQG chronic Cl^- limits (CCME, 2011) . Clearly, Cl^- is an increasing threat to freshwater systems.

2.4.4 Effects of Elevated Cl^- on Freshwater Systems

The application of deicing salt causes secondary effects for urban freshwater systems including but not limited to increased mobility of metals (Norrström and Jacks, 1998) and reduced acid neutralizing capacity in surface water by ion exchange (White, 2000; Löfgren, 2001). Other impacts can be physical, chemical, or biological. Density gradients and circulation of receiving water bodies may be altered (Bubeck *et al.*, 1971; Diment *et al.*, 1973). Dissolved oxygen may be depleted, as increased salinity reduces the ability to dissolve oxygen (Davis, 1975; LSRCA, 2011). Increasing Cl^- and other contaminants in urban runoff are strong drivers of species loss; diversity is reduced and smaller populations and shifts to pollutant-tolerant species are observed (Maltby *et al.*, 1995; Wallace and Biastoch, 2016; Walsh and Webb, 2016). Overall, road salt and other urban contaminants contribute to what has been coined as the Urban Stream Syndrome (USS). USS describes a trend towards elevated contaminant profiles and flashier hydrographs in streams of urban catchments (Walsh *et al.*, 2005; Booth *et al.*, 2016). Additional 'symptoms' of USS include altered channel morphology and distorted biotic assemblages, favouring pollutant-tolerant invertebrates, fish, and plants (Paul and Meyer, 2001).

2.4.5 Current Methods of Cl^- Mitigation

Four primary design features for reduction of salt use were identified by GHD, retained by the LSRCA to prepare guidelines on salt reduction (GHD, 2017). First was the reduction of paved surfaces by adding vegetated features such as landscaped islands. It is also critical to promote effective grading and stormwater collection to minimize freezing of wet surfaces and prevent meltwater from ponding and later refreezing, requiring further salt treatment. Snow storage piles should be placed strategically to reduce the likelihood of meltwater draining across high traffic areas and refreezing. Lastly, layout of pedestrian walkways should be carefully considered to reduce salting of unused walkways. Beyond these design features, Ontario conservation authorities are increasingly promoting public education initiatives to limit pollutants from entering SCMs (LSRCA, 2011; TRCA, 2012).

3 Quantifying Chloride Retention in Urban Stormwater Management Ponds Using a Mass Balance Approach

3.1 Introduction

In urban areas that experience cold and icy winters, chloride (Cl^-)-based road salts are applied to paved surfaces to maintain safe driving conditions (Novotny and Stefan, 2010; Perera *et al.*, 2010; Oswald *et al.*, 2019). Road salts dissolve readily in water and contribute to elevated Cl^- concentrations in ground and surface waters year-round (Judd, 1970; Howard and Haynes, 1993; Daley *et al.*, 2009; Kincaid and Findlay, 2009; Todd and Kaltenecker, 2012; Robinson *et al.*, 2017; Kelly *et al.*, 2019). While there are other anthropogenic sources of Cl^-, including wastewater treatment effluent (Vengosh and Pankratov, 1998; Panno *et al.*, 2006), water softeners (Kaushal, 2016; Lax *et al.*, 2017), and agricultural fertilizers (Sherwood, 1989; Hill and Sadowski, 2016), road salt is the dominant contributor of Cl^- and increased salinity to urban aquatic systems (Sherwood, 1989; Mayer *et al.*, 1999; Kaushal *et al.*, 2005; Chapra *et al.*, 2009; Winter *et al.*, 2011).

Even with improved management practices (Environment Canada, 2018), mean annual stream Cl^- concentrations have continued to increase with increasing urbanization (Kelly *et al.*, 2008; Corsi *et al.*, 2015; Oswald *et al.*, 2019; Kelly *et al.*, 2019). Increasing lake Cl^- concentrations have also been observed (Beeton, 1965; Chapra *et al.*, 2009; Dugan *et al.*, 2017) in lakes with greater than 1% impervious surface area within a 500m buffer (Dugan *et al.*, 2017). Typical background Cl^- concentrations in natural surface waters are around 10 mg L^{-1}, and Cl^- is relatively benign at low concentrations (Fellenberg, 2000). However, Cl^- has negative toxicological impacts on aquatic biota as chronic concentrations exceed 120 mg L^{-1} and/or acute concentrations exceed 640 mg L^{-1} (Fellenberg, 2000; Corsi *et al.*, 2010; CCME, 2011; Walsh *et al.*, 2012; Brown and Yan, 2015; Wallace and Biastoch, 2016). These exposure concentrations do not necessarily apply to all species; for example, some species of endangered and special concern freshwater mussels may not be protected at a long-term Cl^- concentration of 120 mg L^{-1} (CCME, 2011). Elevated Cl^- concentrations in surface waters can alter the composition of aquatic communities and cause species richness and abundance to decline (Karraker *et al.*, 2008; Hintz *et al.*, 2017; Jones *et al.*, 2017). Freshwater species at all trophic levels, as well as nutrient and energy flows at the ecosystem scale, may be negatively impacted (Van Meter and Swan, 2014; Hintz and Relyea, 2019). Elevated Cl^- concentrations can also result in increased transport and bioavailability of heavy metals (Schuler and Relyea, 2018) and increased lake stratification, resulting in anoxia and nutrient resuspension (Sibert *et al.*, 2015; Dupuis *et al.*, 2019). These changes drive species loss, reduced diversity and population densities, and population shifts to pollutant-tolerant species (Maltby *et al.*, 1995; Searle *et al.*, 2016; Wallace and Biastoch, 2016; Walsh and Webb, 2016).

The Cl^- ion does not settle, is not removed biologically, is highly soluble and not subject to any considerable degree of degradation (Cerendulo, 1988; Environment Canada, 2001; Thunqvist, 2004; Chapra, 2009). Precipitation and snowmelt events wash Cl^- off of paved surfaces and it is then transported via subsurface flow, piped flow, or overland flow to receiving waters (Oswald *et al.,* 2019). Given its conservative nature, Cl^- can accumulate over time in groundwater (Williams *et al.*, 2000; Panno *et al.*, 2006; Howard and Maier, 2007), and be transported into streams, rivers (Löfgren, 2001; Thunqvist, 2004; Kaushal *et al.*, 2005; Kelly *et al.*, 2008; Corsi *et al.*, 2010, 2015), and lakes (Novotny *et al.*, 2008; Chapra *et al.*, 2009; Meriano *et al.*, 2009; Likens and Buso, 2010; Müller and Gächter, 2012; Swinton *et al.*, 2015). Previous studies have shown that 28 to 77% of Cl^- from road salt applied in a given deicing season may be retained in the watershed in groundwater and stormwater control measures (SCMs) (Novotny and Stefan, 2010; Perera *et al.*, 2010; Casey *et al.*, 2013; Oswald *et al.*, 2019), creating a legacy effect that can maintain elevated Cl^- concentrations into the growing season, even if salt application is reduced (Gutchess *et al.*, 2016; Kelly *et al.*, 2019). The role that SCMs play in retaining Cl^- and influencing downstream Cl^- dynamics is not well studied.

SCMs were developed for flood protection and to mitigate water quality problems associated with suspended sediment (Fletcher *et al.*, 2013; Environment Canada, 2018; Pennino *et al.*, 2016). A common end-of-pipe SCM is the stormwater management pond (SWMP) (OME, 2003; Bradford and Gharabaghi, 2004). SWMPs are considered a best management practice for attenuating peak flows and allowing suspended sediments to settle out (Meierdiercks *et al.*, 2010b; Newcomer Johnson *et al.*, 2014). However, previous research suggests that SWMPs could promote long-term storage and release of Cl^- due to densimetric stratification (Marsalek, 2003) and potential leakage to groundwater (Casey *et al*., 2013). On the watershed scale, it is expected that SWMPs are not retaining high amounts of Cl^- relative to overall watershed loads (Snodgrass *et al*., 2017); however, in some watersheds, for example the East Holland River watershed in southern Ontario, Canada, over 40% of the urban watershed area is drained through SWMPs. Given the wide range in area drained through SWMPs in watersheds across this region and the continued use of SWMPs for stormwater management in urbanizing areas, a better understanding of Cl^- retention in SWMPs over different time scales is warranted. We addressed this research gap by quantifying Cl^- retention for two SWMPs located in the East Holland River watershed in southern Ontario, Canada and examining the influence of pond Cl^- export on downstream water quality for one of the SWMPs. Our four specific objectives were to: (1) quantify the mass of Cl^- entering and leaving the SWMPs; (2) determine annual, seasonal, and event-scale Cl^- retention for the SWMPs using a mass balance approach; (3) compare changes in Cl^- retention in the ponds estimated in (1) to in-pond surface and bottom Cl^- concentrations; and (4) quantify the response of a receiving stream to Cl^- exported from one of the SWMPs. The results of this study will be used to parameterize the role of SWMPs in watershed-scale Cl^- transport models and geospatial models of Salt Vulnerable Areas (SVAs).

3.2 Methodology

3.2.1 Study Watershed

The 247 km^2 East Holland River (EHR) watershed (Figure 1) drains into Lake Simcoe, which is located in south-central Ontario, Canada. Land use in EHR is 24% wetland and forest, 35% agriculture, and 40% urban (Oswald *et al*., 2019). Urban expansion is occurring in the southern part of the watershed while agricultural practices occur in the northern and southeastern parts of the watershed. The average annual precipitation in EHR is 858mm, of which 14.25% is snow. Snowfall occurs from November to April. The mean air temperature in EHR is 7°C (EC Weather Office, 2019). It is on glacial till and mostly composed of glaciolacustrine silt and clay. EHR has very deep bedrock and long flow paths (Oswald *et al*., 2019). 65% of the EHR drains through SWMPs. Approximately one-sixth of the Lake Simcoe watershed is predicted to exceed the Canadian Water Quality Guidelines (CWQG) for the protection of aquatic life for chronic exposure to Cl^- (LSRCA, 2015). EHR is one of two sub-watersheds in the Lake Simcoe basin to host the greatest number of aquatic taxa impacted by Cl^- (LSRCA, 2015).

3.2.2 Study Stormwater Management Ponds

This study examines two SWMPs located in the EHR watershed (Figure 1). The Don Hillock (DH) pond has one inlet, one outlet, one overflow route, a maximum area of 8920 m^2 and a maximum depth of 1.7m. DH was constructed in 2006, drains a business park area (Town of Aurora, 2010), and has a contributing catchment of 133,000 m^2, of which 93% is directly connected impervious area (DCIA). DCIA is defined as "the portion of impervious area with a direct hydraulic connection to the waterbody via continuous paved surfaces, gutters, drain pipes, or other conventional conveyance and detention structures that do not reduce runoff volume" (USEPA, 2011). The Oaktree (OAK) pond has one inlet, one outlet, two overflow routes, a maximum area of 3812 m^2 and a maximum depth of 2.2m. OAK was constructed in 2004, drains a stable residential area (Town of Newmarket, 2014), and has a contributing catchment of 48,000 m^2, of which 96% is DCIA. Both ponds contribute to first order headwater streams; DH contributes to the beginning of the stream and OAK connects along a stream reach. Both ponds fall under the jurisdiction of the Lake Simcoe Region Conservation Authority (LSRCA) and were selected because they were part of a broad-scale pond study examining water and sediment balance, hence water quality and flow monitoring data for these ponds were available from the LSRCA for this study. The ponds are clay-lined with bottom-draw outlets and release water into streams that join the EHR.

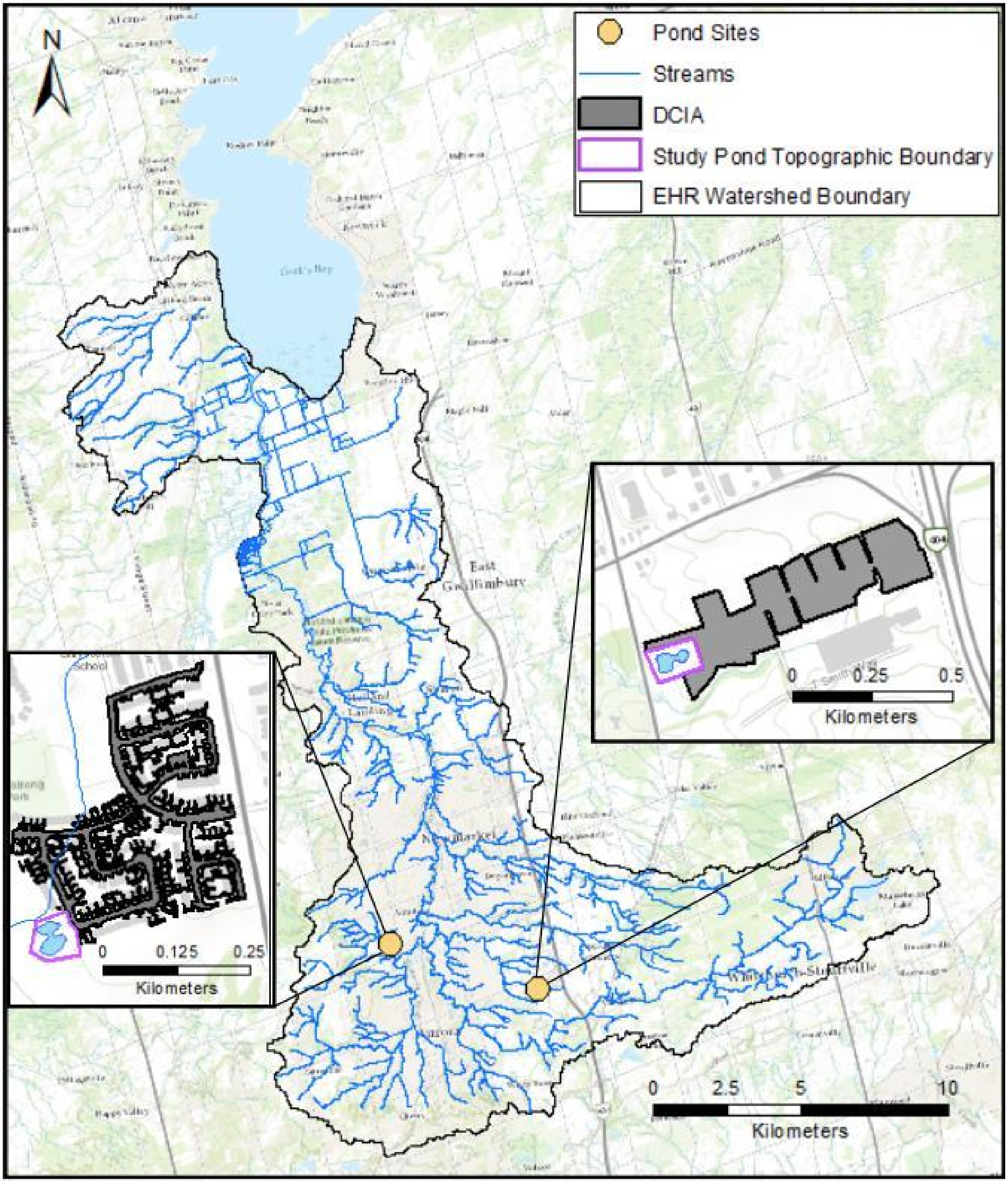

Figure 1. EHR watershed boundary relative to Lake Simcoe. Inset maps are OAK (upper) and DH (lower) and their contributing catchments, indicated by the directly connected impervious area (DCIA) and study pond topographic boundaries.

3.2.3 Chloride Concentrations

Solinst® Leveloggers recording water temperature, level, and conductivity measurements at 15-minute intervals were deployed for 24 months (November 2016 to November 2018) at both ponds' inlets and outlets, and an additional 6 months (up to April 2019) at the OAK inlet and outlet. Levelogger level measurements were corrected for barometric pressure using historical climate data from the Toronto Buttonville A weather station (Environment Canada, 2019). As we were unable to access the inside of the inlet and outlet pipes, the Leveloggers were deployed at the mouths of the pipes. LSRCA also deployed 2 conductivity probes each in the DH forebay and wet cell (June 2016 to May 2018); one was positioned 20cm above the bottom of the pond, and the second 20cm below the surface of the pond. Leveloggers recording water temperature, level, and conductivity were also deployed for 8 months (August 2018 to April 2019) upstream and downstream of the confluence of the OAK outlet and the receiving tributary of Tannery Creek to quantify the impact of pond Cl^- export on in-stream Cl^- concentration. DH and another pond contribute equally to the beginning of a headwater stream and therefore upstream and downstream impacts of Cl^- export from DH were not measurable.

As most urban stream chemistry shows a linear relation between Cl^- concentration and specific conductance, high-frequency monitoring for conductivity is accepted as a proxy method for collecting long-term Cl^- data (Sanford *et al*., 2012; Casey *et al*., 2013; Hubbart *et al*., 2017). The actual conductivity measured by the Solinst® Leveloggers used in this study was converted to specific conductance as follows:

$$Sc = \frac{Ac}{1+r(T-25)} \qquad (1)$$

where Sc is the specific conductance in μS cm^{-1}, Ac is the actual conductivity in μS cm^{-1}, T is the water temperature in °C and r is a temperature correction coefficient (Carlson, 1999).

Monthly grab samples were taken over the course of one year (March 2018 to February 2019) at pond inlets and outlets to calibrate the Cl^- concentration-specific conductance relationship (Cooper *et al.*, 2014). Samples were also collected during two rain events (summer and fall 2018) and two melt events (winter 2018 and winter 2019) to better define the relationship at high Cl^- concentrations, as rain-on-snow and snowmelt events contain the largest pollutant loading (Batroney *et al.*, 2009). Samples were collected into clean HDPE bottles and filtered through 0.5μm glass fiber filter discs within 3 days of collection. Filtered samples were kept in the dark at 4°C until processed via ion chromatography (USEPA, 2016) using the Thermo Scientific™ Dionex™ ICS-6000 HPIC™ system equipped with IonPac AS18 analytical column (Cl^- detection limit: 0.1 mg L^{-1}). Ion analyses were conducted by the Biogeochemistry Research Group of the Department of Geography and Environmental Management at the University of Waterloo. The Cl^- concentration-specific conductance relationships derived were:

$$[Cl^-] = 0.2851 \cdot Sc - 20.75 \tag{2}$$

and

$$[Cl^-] = 0.3769 \cdot Sc - 202.4 \tag{3}$$

for DH ($r^2 = 0.95$, n = 39), and

$$[Cl^-] = 0.2879 \cdot Sc - 34.70 \tag{4}$$

and

$$[Cl^{-}] = 0.4389 \cdot Sc - 443.8 \quad (5)$$

for OAK ($r^2 = 0.99$, n = 39), where [Cl^-] is the Cl^- concentration in mg L^{-1}. Equation (2) was used for specific conductances below 1979 µS cm^{-1}, and equation (3) for specific conductances above 1979 µS cm^{-1}. Equation (4) was used for specific conductances below 2709 µS cm^{-1}, and equation (5) for specific conductances above 2709 µS cm^{-1}. These breakpoints and associated relationships were determined using the 'segmented' package for regression models with unknown break-points (Muggeo, 2003; Muggeo, 2008) for the R language and environment for statistical computing (R Core Team, 2017).

3.2.4 Water Balance and Cl^- Mass Balance Calculations

The LSRCA collected in-pipe level and flow data at the inlets and outlets of both ponds from May 2017 to November 2017 using a V-notch weir at the DH outlet and ISCO 2150 area velocity flow modules at OAK and the DH inlet. The LSRCA in-pipe levels were compared to our corresponding mouth-of-pipe Levelogger level measurements for this study to estimate the relationship between LSRCA levels and our monitored levels ($0.95 < r^2 < 0.99$). The level-to-flow relationship from LSRCA data was then used to determine flow from our monitored levels for the remaining 21 months ($0.73 < r^2 < 0.99$). For each 15-minute interval, the flow (L s^{-1}) was multiplied by the time interval (900 s) and corresponding Cl^- concentration (mg L^{-1}) to determine the Cl^- load. Loads were summed over event, seasonal, and annual time periods. Evaporation from the ponds was estimated using the Thornthwaite formula (Thornthwaite and Holzman, 1939), which derives potential evapotranspiration as a function of air temperature, and Chow's adjustment method (Chow, 1965), which corrects for monthly sunshine duration. Precipitation inputs were derived from LSRCA rain gauge data at the Newmarket Office location (44°03'50.3"N 79°27'27.0"W). This rain gauge is 4.5km from OAK and 6.9km from DH.

Percent retention was calculated as follows:

$$Retention = \frac{Input - Output}{Input} \times 100\% \qquad (6)$$

For the water balance, the output includes outflows and evaporation. Precipitation falling directly onto the ponds' topographic catchments was also considered as an input in the water balance.

For seasonal analyses, the study period was divided into winter (November - April) and summer (May - October) seasons based on historical snowfall data. Years are therefore defined in this study as November - October. The pond outflow hydrographs were classified into event and non-event periods using the USGS HYSEP minimum flow routine (Sloto and Crouse, 1996) executed in R.

3.2.5 Statistical Analyses

All statistical analyses were completed using the R language and environment for statistical computing (R Core Team, 2017).

3.3 Results and Discussion

3.3.1 Annual and Seasonal Variability in SWMPs' Water Balances

OAK retained over 25% of the water received in both years, while DH released more water than it received in 2017 and retained 50% in 2018. This raises the possibility that there are other significant inflows and outflows that were not accounted for in this study. Potential inflows that were not accounted for include (i) a second stormwater pipe that did not appear in the DH pond design diagram draining a small parking lot area; and (ii) groundwater flows into the ponds. Potential outflows that were not accounted for include (i) loss under the outlet weir at DH; and (ii) underrepresented outlet flows during high flow events over the area-velocity flow module at OAK. It is also possible that inflow and outflows during winter seasons were under-reported due to freezing of the Leveloggers. Area-velocity sensors also work best at moderate flow and poorly at low and very high flows, so there may be some measurement error associated with our flow estimates.

To verify the accuracy of pond inflows, the DCIA for each pond was multiplied by precipitation amounts over the study period. Inflows at OAK were similar to what precipitation data and DCIA would suggest at approximately 99% of the expected volume for both winter seasons and approximately 84% of the expected volume for the summer seasons. Again, the difference may be due to the distance of the precipitation gauge used from the pond (4.6km). Inflows at DH were also similar to what precipitation amounts multiplied by DCIA would suggest (93% of expected volume for the winter seasons and 95% of the expected volume in S18), except during summer 2017 when inflow exceeded the expected volume by 50%. This is likely due to measurement error; there was one very large event in summer 2017 in which DH and the storm system upstream of DH were filled but flow was recorded as very low over the area-velocity sensor. It is likely that there were other similar errors at low or very high flows throughout the summer.

As the ponds are closed systems and in-pond monitoring indicates that water levels were comparable at the start and end of study years, the water balance was forced to close for the Cl^- balance calculations. Having verified the pond inflows, only outflows were adjusted. As ponds are designed to control the outflow of water, the flows tend to be lower and thus not as accurately captured by the area-velocity sensors. For example, if the pond water balance indicated that water was retained (storage > 0), it was assumed that the equivalent volume of water, with the average Cl^- concentration for the season, was released in addition to the recorded Cl^- output (Figure 2c, W18 and S18, and Figure 2d). If the pond water balance indicated that water was released (storage < 0), it was assumed that the equivalent volume of water, with the average Cl^- concentration for the season, was reduced from the recorded Cl^- output (Figure 2c, W17 and S17).

With this adjustment, the volume of water and mass of Cl^- received by DH are much higher than those received by OAK (Tables 2 and 3), which was expected as DH has a larger contributing catchment and larger DCIA. However, DH has 2.65 times more DCIA than OAK yet received over 6 times as much water and over 18 times as much Cl^- over the study period. This suggests that land use and land management practices (e.g. irrigation) must be considered along with % impervious area when modelling Cl^- loading across a watershed. For example, there are more parking lots and commercial properties in DH's catchment than in OAK's, where most of the DCIA is composed of residential streets. The Towns of Newmarket and Aurora have road salt application rates of 100 kg $lane^{-1}$ km^{-1} (unpublished data), which amounts to 33 g m^2 when an average lane width of 3m is assumed. Comparatively, LSRCA monitoring indicates that parking lot application rates are typically 90 – 100 g m^2.

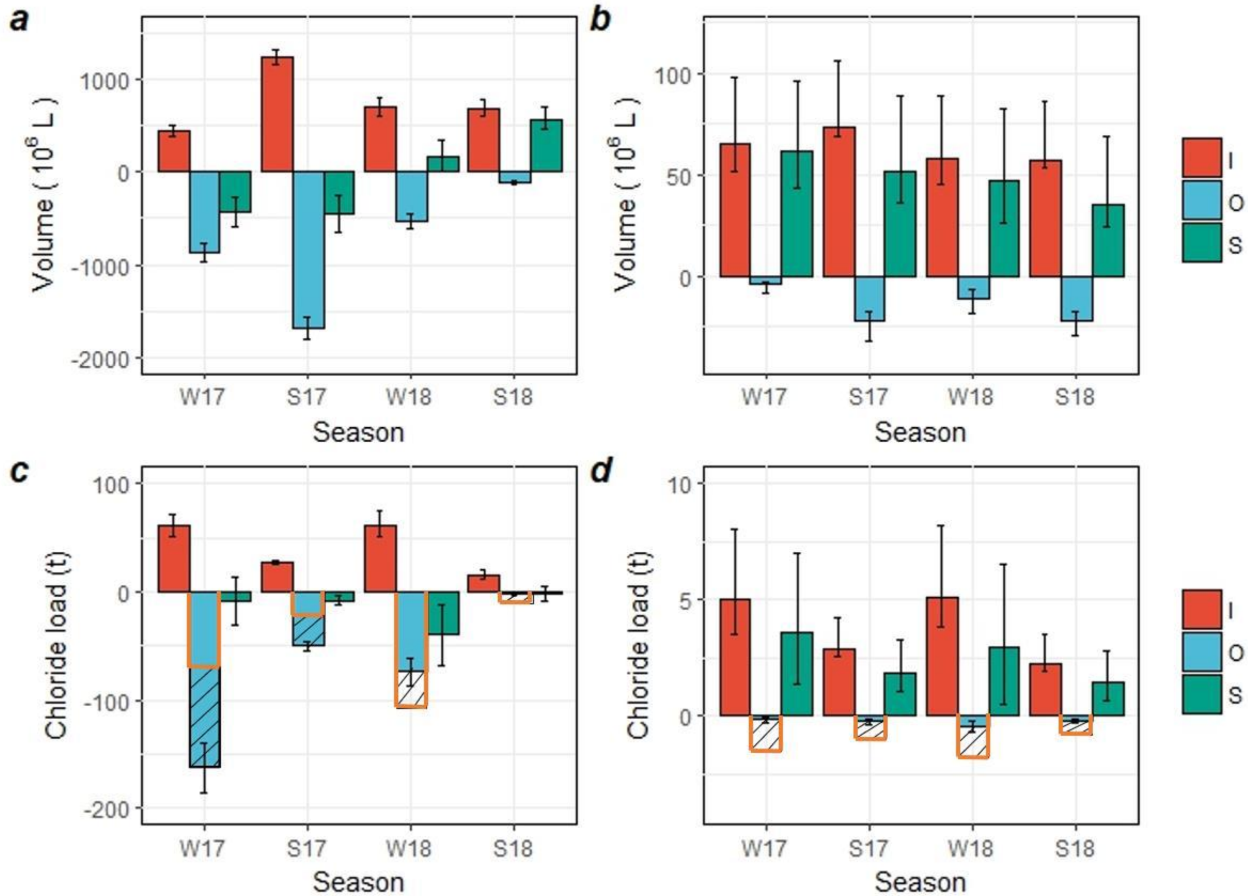

Figure 2. Water and Cl^- balances for DH ((a) and (c)) and OAK ((b) and (d)) for each season of the study period (W = winter, November – April; S = summer, May – October). For water balances ((a) and (b)), inputs are inflows and precipitation; outputs are outflows and evaporation. I = input, O = output, S = storage (retention). Hatched lines indicate the adjustment applied to the outflow; the bolded outline indicates the outflow used to determine Cl^- storage. Note that (i) negative retention indicates release, and (ii) y-axes differ between ponds.

3.3.2 Annual and Seasonal Variability in SWMPs' Cl^- Mass Balances

The Cl^- retention differed between the two ponds. DH consistently released Cl^- across seasonal and annual scales, whereas OAK consistently retained Cl^- (Figure 2). In OAK, Cl^- retention was greater than 50% on both annual and seasonal scales throughout the study (Figure 2).

For both ponds, we see that there are still Cl^- inputs in the summer seasons. This may be because the Cl^- stored in soils over the winter deicing period is being flushed out by infiltrating rainwater in the spring and early summer (Higashino *et al.*, 2017). There are Cl^- outputs in the summer seasons as well, indicating that the ponds are releasing Cl^--laden water throughout the year. This supports findings from Kelly et al. (2008) that deicing salts contribute to long-term increases in streamwater salt concentrations.

Both ponds had similar patterns of Cl^- inputs, with slightly more Cl^- received in W18 than W17, and less Cl^- received in S18 than S17. The deicing season was colder and started earlier in late 2017, likely leading to more road salt application in W18 (November 2017 – April 2018) than in W17 (November 2016 – April 2017). While the total rainfall was similar for S17 and S18, S17 featured larger events; this may have contributed to greater flushing of Cl^- stored in soils in S17 and thus greater Cl^- inputs to the ponds in S17 than in S18.

3.3.3 In-Pond Cl^- Monitoring

Water enters the forebay of both SWMPs via stormwater pipes. The forebay is separated from the wet cell (also known as the aft bay) by a berm, so water primarily enters the wet cell from the surface of the forebay. Due to densimetric stratification, water at the pond's surface has lower Cl^- concentrations than at depth, so Cl^- concentrations in the wet cells are lower throughout the winter and early spring than in the forebay (Figure 3). However, large rainfall events can mobilize Cl^- from the forebay into the wet cell, thereby decreasing the forebay Cl^- concentrations and increasing the wet cell Cl^- concentrations.

Predictably, Cl^- concentrations at both the surface and at depth in the forebay and wet cell are relatively lower through the summer, then rise in the winter (Figure 3). In the forebay, winter Cl^- concentrations at depth exceed that of saltwater (19,250 mg L^{-1}) at times. Fluctuations in concentrations are of greater magnitude at depth than at the surface. This is because the ponds are bottom-drawing; when water is released, Cl^- concentrations at depth in the wet cell are the most affected.

The relatively low Cl^- concentrations at both the surface and the bottom of the pond in the summer seasons indicates that Cl^- should be flushed out of the ponds in late spring (May - June). In 2017, we see that this process occurs over the course of several months (Figure 4), rather than all at once during the spring melt as was observed by Barbier et al. (2018). While the mass balance indicates that this is the case in S17, it is not the case for S18 (not included in Figure 4 as in-pond sensors were removed), when some Cl^- was retained. LSRCA monitoring shows that Cl^- persists year-round in some ponds and there is not a full flushing (LSRCA, *in press*). Possible mechanisms for Cl^- retention on an annual basis include uptake by plants and/or release by seepage to groundwater; however, these mechanisms would jointly account for less than 30% of the annual Cl^- retention.

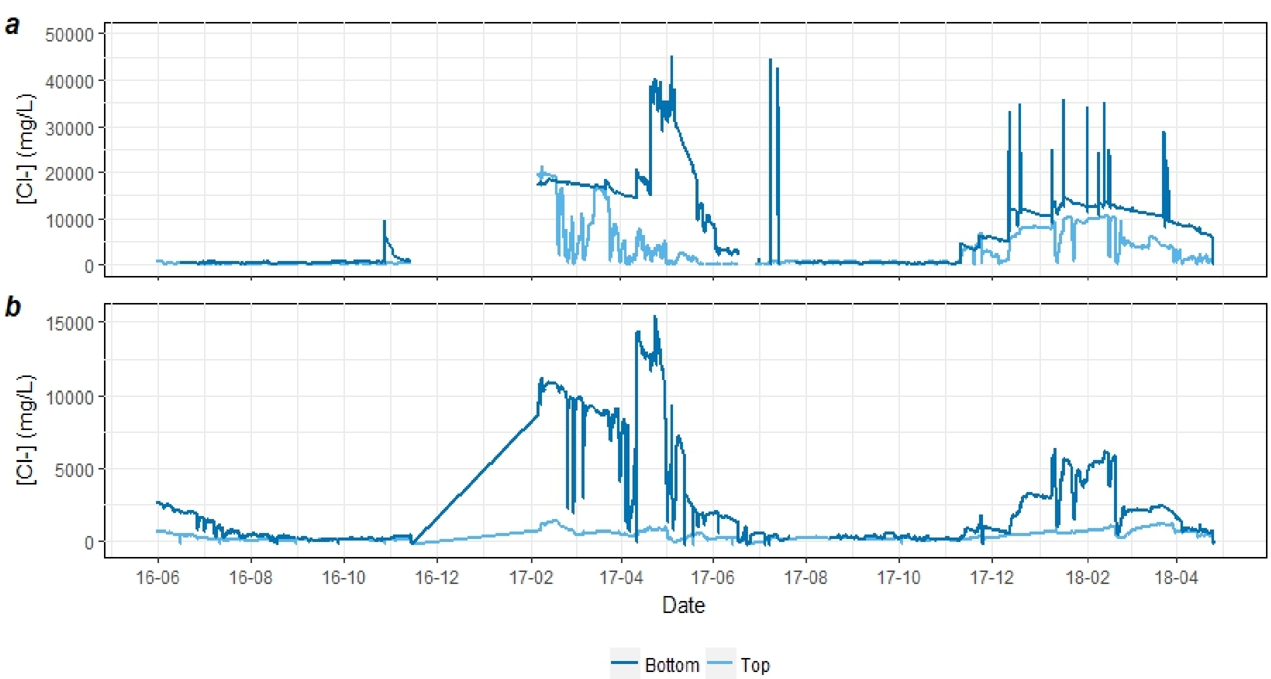

Figure 3. Cl^- concentrations in the (a) forebay and (b) wet cell of DH over a 2-year period. Data unavailable in wet cell between Nov ‘16 and Feb ’17. Note the different y-axis ranges.

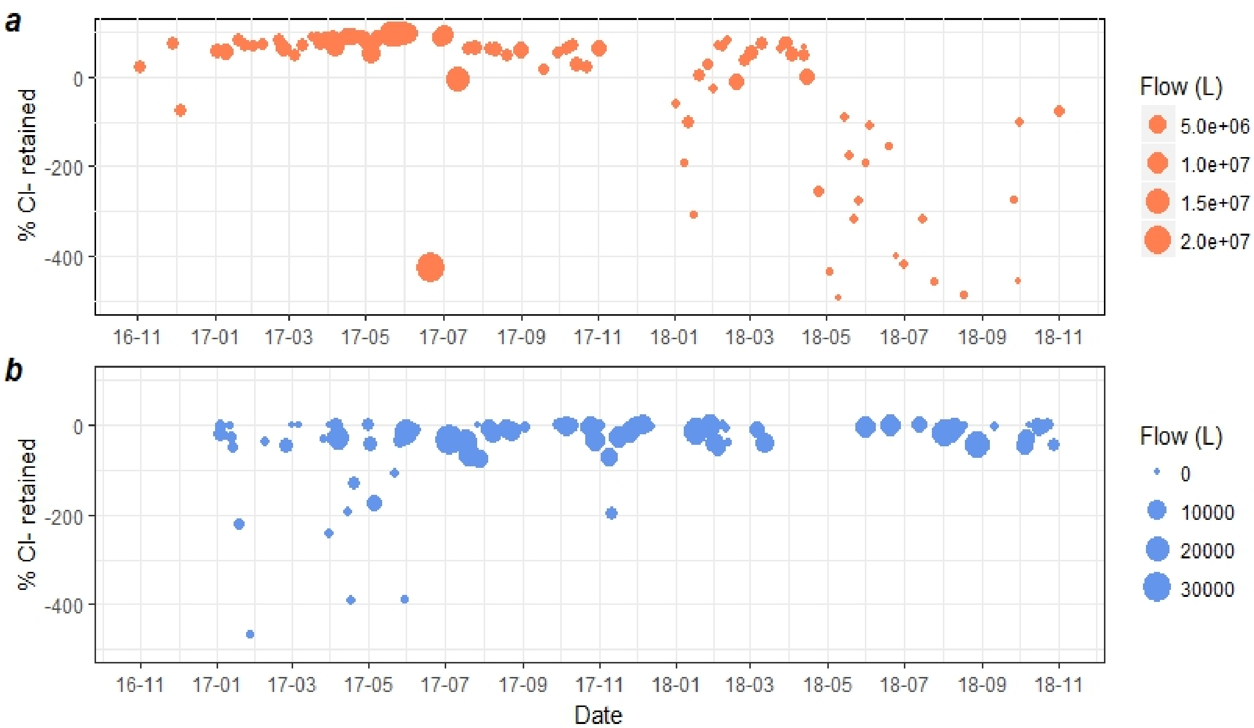

Figure 4. Percentage of Cl^- retained by (a) DH and (b) OAK during rain and melt events in the 2-year pond monitoring period (n = 201). Note that negative retention indicates Cl^- release.

3.3.4 Event-Scale Cl^- Retention

DH and OAK behaved very differently in terms of event-scale Cl^- retention (Figure 4). DH retained Cl^- for 60 of 103 events (58%) whereas OAK retained Cl^- for 14 of the 81 events (17%). OAK's Cl^--releasing events occurred mostly in the spring of 2017, whereas DH's Cl^--releasing events occurred mostly in the summer of 2018. Some minimums in Cl^- retention for both ponds exceed -100% (i.e., release), indicating that melt events and large spring and summer rains may be flushing out Cl^- retained in ponds over the winter season. This corresponds with watershed-level findings from Corsi et al. (2015) that streams receive heavily Cl^--laden water during low flow winter events, as well as high flow periods throughout the year.

Total flow of an event does not correspond to Cl^- retention; therefore, those melt and spring rain events must carry water with high Cl^- concentrations. For OAK and DH, 49% and 55% of the total Cl^- release occurred during events, respectively. This supports findings from Snodgrass et al. (2017) that SWMPs do not prevent high concentrations of Cl^- from reaching streams.

3.3.5 Stream Response to Pond Cl^- Export

The Cl^- concentration response of the headwater stream receiving flow from the OAK pond is shown for the subset of the study period when in-stream conductivity sensors were deployed (Figure 5a). During the late summer of 2018, when stream volume was higher, the difference between upstream and downstream Cl^- concentrations at OAK was relatively constant. Once the winter deicing season started in November 2018, the difference between upstream and downstream Cl^- concentrations began to fluctuate. The peaks in upstream-downstream Cl^- concentration differences correspond to pulses of high Cl^- export from the pond (Figure 5b).

Both upstream and downstream baseline Cl^- concentrations increase over the monitoring period, suggesting that sources upstream of the pond input are contributing to a slow increase of Cl^- in the stream, and OAK is contributing to Cl^- peaks (Figure 5a). Upstream of OAK is an area with 115 ha of residential land, 296 ha of agricultural or undifferentiated rural land, 5 stormwater management ponds, and a headwater creek that exceeds chronic CWQG for the protection of aquatic life (120 mg L^{-1}) almost year-round. Upstream Cl^- concentrations exceeded chronic CWQG for 72% (~21 weeks) of the stream monitoring period and exceeded acute guidelines (640 mg L^{-1}) for 3% (~ 1 week) of the stream monitoring period. Comparatively, downstream Cl^-

concentrations exceeded chronic guidelines for over 99% (~ 30 weeks) of the stream monitoring period and exceeded acute guidelines for 6% (~ 2 weeks) of the stream monitoring period.

Summer stream Cl^- concentrations above chronic guidelines may potentially be maintained by Cl^--laden groundwater discharge as well as stream outflows (Casey *et al*., 2013). It is likely that stream Cl^- concentrations remain above the chronic guidelines year-round, suggesting that stream reaches that receive water from SWMPs are only hospitable to salt-tolerant biota. This may reflect a flaw in the design of bottom-draw ponds. The ponds were designed this way to protect coldwater habitat but are exporting highly saline water into receiving waters.

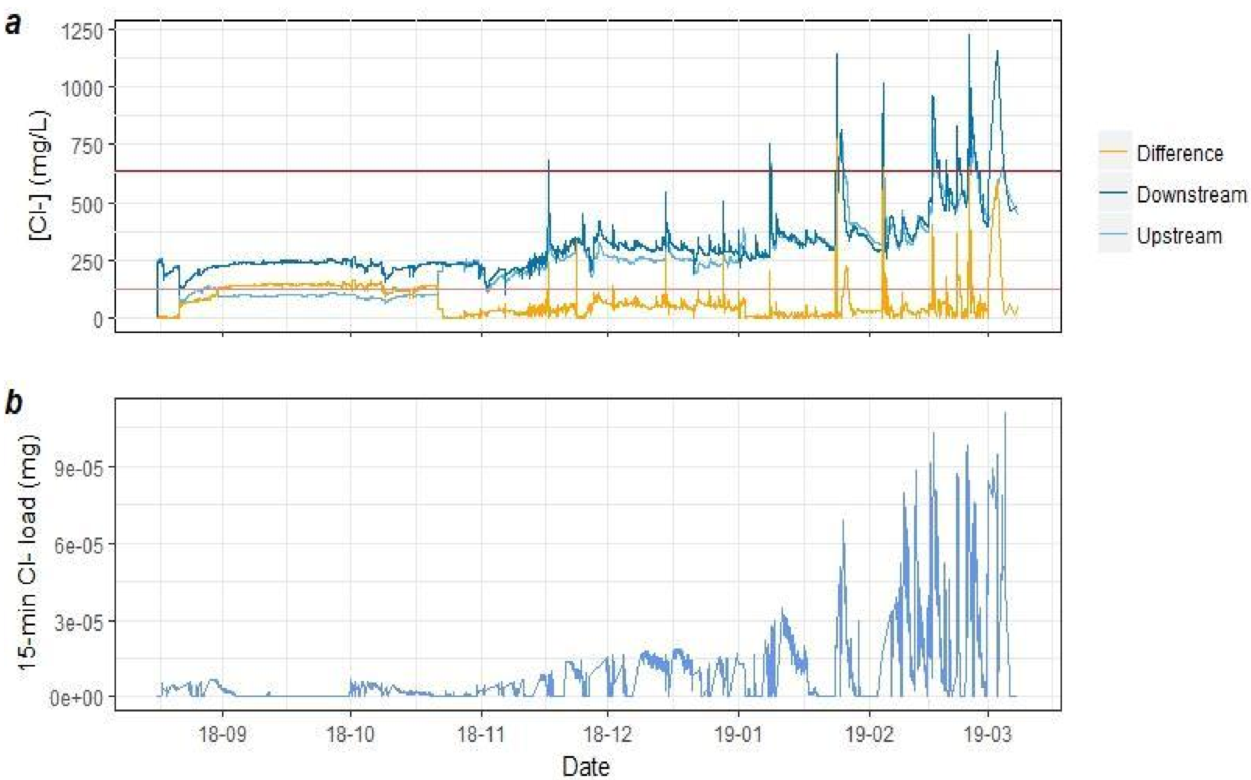

Figure 5. (a) Stream Cl^- response to the pond outflow at OAK. Lower and upper lines represent chronic and acute CWQG for the protection of aquatic life, respectively. (b) The recorded mass of Cl^- leaving OAK for each 15-minute interval in the stream monitoring period.

3.3.6 Implications for Understanding the Influence of SWMPs on Watershed-Scale Cl^- Retention and the Identification of SVAs

This research shows that on an annual basis, one pond consistently retained Cl^-, which suggests losses to groundwater. Future studies may incorporate several strategically chosen groundwater monitoring points to check for seepage. The other pond had variable annual retention; this may be in part due to limitations of our flow monitoring and study period, which may be biasing our findings regarding Cl^- storage. Small changes in the level-to-flow relationship can result in large changes in the overall water balance of the pond. It would be helpful to have flow data for a longer period, such as the entire ice-free season. A longer study period would also help to elucidate patterns in water and Cl^- retention; in this study, there were considerable fluctuations between Y17 and Y18, but we were unable to verify whether these were part of a longer-term pattern. It is likely that SWMPs play a role in watershed-scale Cl^- retention; however, further research is required to examine interannual variability, and to determine what factors influence the timing and magnitude of pond Cl^- flushing.

The in-pond data suggest major flushing in the late spring and early summer months. We also observed that the receiving stream experienced pulses of Cl^--laden pond outflows throughout the winter. These data suggest that SWMPs concentrate diffuse salt inputs onto the landscape. Streams receiving pond outflows may be more vulnerable than adjacent areas; this suggests that SWMP locations should be considered in the identification of SVAs. Future studies could consider including more ponds that span a land-use gradient. If these ponds were also in-line with streams, downstream impacts could be more accurately quantified.

3.4 Conclusions

The application of deicing salts in watersheds that experience icy winters leads to watershed-scale Cl^- retention and long-term elevated Cl^- concentrations in surface and ground waters. SWMPs, a commonly used SCM, are known to impact pollutant transport, but Cl^- retention in SWMPs over different time scales was previously unclear. To better understand Cl^- retention in SWMPs, this study: (1) quantified the mass of Cl^- entering and leaving two SWMPs through stormwater pipes; (2) determined annual, seasonal, and event-scale Cl^- retention for the SWMPs using a mass balance approach; (3) compared changes in Cl^- retention to in-pond Cl^- concentrations; and (4) quantified the response of a receiving stream to Cl^- exported from one of the SWMPs.

Stormwater management pond behaviour can be unpredictable; ponds may consistently retain Cl^- like OAK, or alternately retain and release Cl^- like DH. However, high- Cl^--concentration flushes dependably occur during spring melts and, to a lesser extent, during events throughout the year. The ponds do not prevent heavily Cl^--laden water from reaching streams and contribute to year-round inputs of Cl^- to streams. As such, SWMPs deserve particular attention in Cl^- transport models and geospatial models of SVAs in urbanizing watersheds, especially those with large areas drained by ponds. Further study is required to determine long-term patterns of Cl^- retention in SWMPs and to determine the effects of different land uses on the efficacy of SWMPs for Cl^- retention.

3.5 Acknowledgements

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4 Summary

Increasing urban development is accompanied by increasing impervious area and therefore, flashier stream hydrographs. This drives a need for stormwater control measures (SCM). Stormwater management ponds (SWMP) are a popular SCM in urbanizing watersheds and have some water quality benefits as well. However, in seasonally snowy and icy urban areas that receive deicing salts, they have been criticized as potentially promoting long-term storage and release of Cl^- to receiving waterways. Given the ubiquity of SWMPs and the rising concentrations of Cl^- in groundwater, rivers, and lakes that receive inputs from urban areas, a better understanding of Cl^- retention in SWMPs over different time scales is needed.

This study aimed to quantify Cl^- retention in 2 SWMPs in an urbanizing watershed using a mass balance approach and asses the impact of a SWMP on the receiving stream. We found that the SWMPs were unpredictable with respect to water and Cl^- retention; one consistently retained Cl^-, while the other retained Cl^- in one year and released in the other. The SWMPs may be losing water and Cl^- to groundwater and simultaneously releasing pulses of high-Cl^--concentration water to surface waters. Cl^- flushing from the SWMPs primarily occurred during the spring melt and large events throughout the year, indicating that the SWMPs are contributing to year-round inputs of Cl^- to streams. However, water released via stormwater pipes from the ponds has lower Cl^- concentrations that the water received. As such, SWMPs must be considered in Cl^- transport models and geospatial models of SVAs in urbanizing watersheds. Further study is required to determine long-term patterns of Cl^- retention in SWMPs and the drivers behind them.

Appendix

Table 1. Water balance for DH and OAK for each season and year of the study period (W = winter, November – April; S = summer, May – October; Y = year, November – October). All units in 100,000 L. I = input, P = precipitation, E = evaporation, O = output, S = storage (retention).

	DH_I	DH_P	DH_E	DH_O	DH_S	OAK_I	OAK_P	OAK_E	OAK_O	OAK_S
W17	430.2	13.3	2.5	867.8	-426.8	59.9	5.7	1.3	2.7	61.6
S17	1215.7	16.0	18.1	1663.2	-449.6	66.9	6.8	9.3	12.8	51.6
W18	681	11.0	0.8	524.3	166.9	53.5	4.7	0.4	10.6	47.2
S18	667.8	14.1	19.8	92.8	569.3	51.2	6.0	10.2	11.3	35.7
Y17	1646	29.3	20.6	2531	-876.3	126.8	12.5	10.6	15.5	113.2
Y18	1348.9	25.1	20.6	617.1	736.3	104.7	10.7	10.6	21.8	83.0

Table 2. Cl^- balance for DH and OAK for each season and year of the study period. All units in tonnes. I = input, O = output, A = adjustment, S = storage (retention).

	DH_I	DH_O	DH_A	DH_S	OAK_I	OAK_O	OAK_A	OAK_S
W17	60.7	162.3	93.2	-8.4	5.0	0.12	-1.3	3.6
S17	26.5	50.3	15.5	-8.3	2.9	0.23	-0.8	1.8
W18	61.2	73.3	-27.4	-39.6	5.1	0.45	-1.7	2.9
S18	15.3	2.4	-14.9	-2.0	2.2	0.2	-0.5	1.5
Y17	87.2	212.6	108.7	-16.7	7.8	0.35	-2.1	5.4
Y18	76.5	75.7	-42.3	-41.6	7.3	0.65	-2.3	4.4

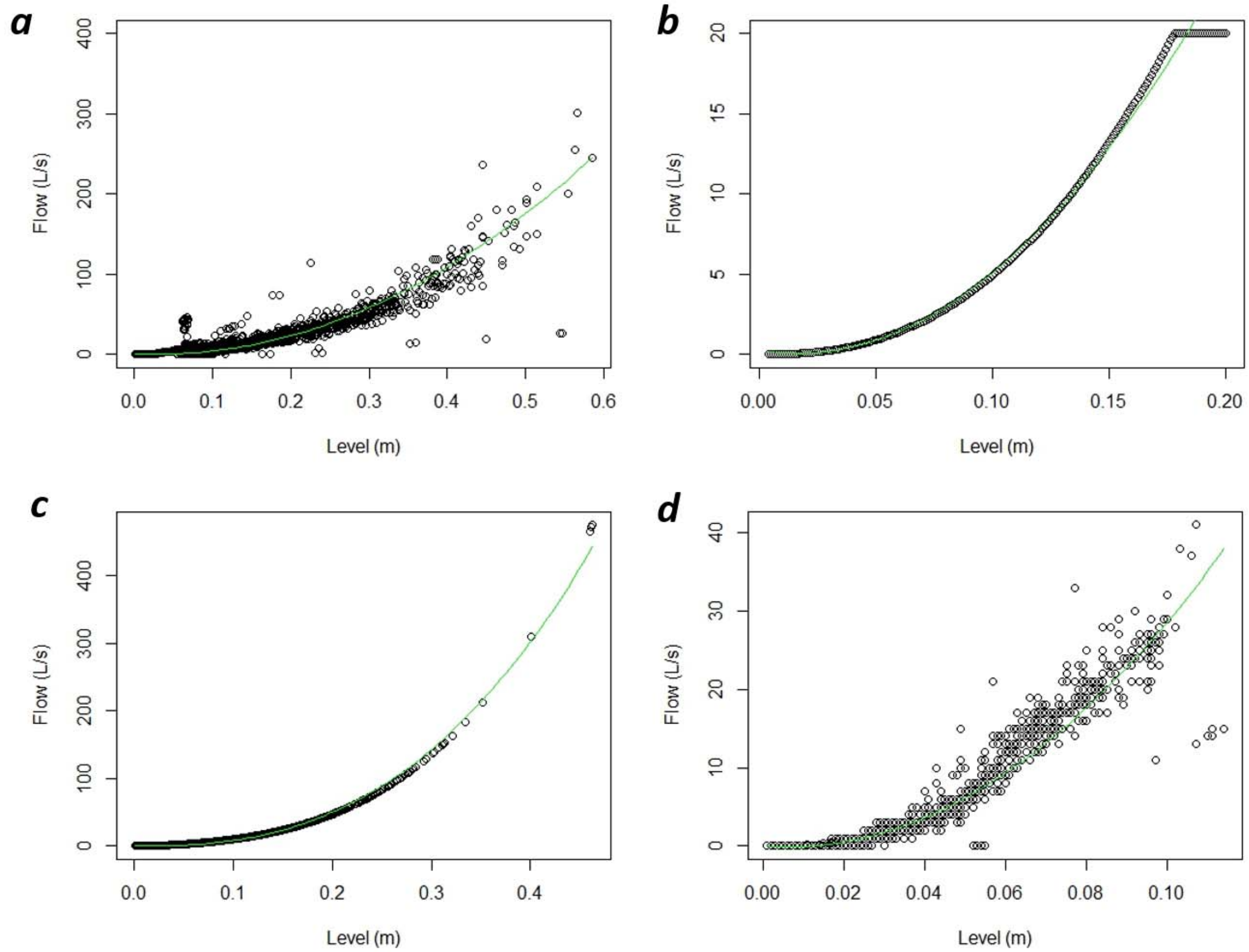

Figure 6. Level-to-flow plots for the (a) DH inflow ($R^2 = 0.73$), (b) DH outflow ($R^2 = 0.99$), (c) OAK inflow ($R^2 = 0.99$), and (d) OAK outflow ($R^2 = 0.98$).

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