Pocket Wetland Impacts on Stormwater Runoff and Water Quality

by

Jason Krompart

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Impervious surface areas within an urban catchment can generate higher runoff volumes and degrade water quality in local streams. Pocket wetlands are an end-of-pipe stormwater management system that detain runoff and improve stormwater quality before entering the channel. These wetlands are used as a ‘polishing’ feature in stormwater management systems. The working hypothesis is that a pocket wetland will attenuate flow and improve water quality, which can be modeled with an areal decay model. Stormwater runoff and water quality was monitored upstream, downstream and at the inlet of a wetland in the Churchville subwatershed in Brampton, ON, April – October 2014. Results indicate the wetland provided short-term storage for the remediation of water quality through rainfall events, which were then estimated using an areal decay model. This study provides evidence on the performance of pocket wetlands for stormwater management and design.
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Chapter 1.0 Introduction

Land use changes associated with urban development alter watershed hydrology through an increase in impervious surface areas (ISA) (i.e., roads, sidewalks, houses). ISAs lead to changes in infiltration and create efficient pathways for surface runoff flow. The change in surface characteristics increase runoff volumes and negatively impact the physical, chemical, and ecological properties of local streams (Booth and Bledsoe, 2009; Brabec, 2009; Brown et al., 2009; Konrad and Booth, 2005; Paul and Meyer, 2001). Stormwater management (SWM) strategies have proven effective at detaining runoff volumes but more recent best management practices (BMP) focus on improving runoff water quality for the protection of local species at risk (Bradford and Gharabaghi, 2004; Hunt et al., 2011; Pitt, 1999; Pyke et al., 2011; Scholes et al., 2008). Pocket wetlands (PW) are an end-of-pipe SWM system which attenuate runoff and remediate water quality (Jones and Wadzuk, 2013; Malaviya and Singh, 2012; Shutes et al., 2005, 1999). The installation of BMPs are prioritized based on predictive models which assess the performance of stormwater infrastructure (Elliott and Trowsdale, 2007; Huber et al., 2006; Krebs et al., 2013; Wong and Geiger, 1997; Wong et al., 1999). Southern Ontario has undergone multiple restoration projects that include PWs but further study on the performance (including modelling outcomes) are needed to justify widespread inclusion into natural channel designs. This study provides an opportunity to support PWs within the academic community although these features have been regularly used in SWM and natural channel designs.

Streams are variable systems which react directly to flow characteristics and serve as habitats to aquatic species (Konrad and Booth, 2005; Sear et al., 2010). Stream health, defined by habitat quality, species diversity and abundance, is negatively impacted by an increase of the average rainfall runoff (Arnold, Jr. and Gibbons, 1996; Booth and Bledsoe, 2009; Brabec, 2009;
Pitt, 1999). As discharge increases, so does the potential for erosion and damage to fish habitats (i.e., vegetation, woody debris) (Arnold, Jr. and Gibbons, 1996; Booth and Bledsoe, 2009; Novinger and Coon, 2000). Erosion and deposition of sediment within the stream channel create bars and spill pools which limit passage for fish and lead to shallow pools that are uninhabitable to certain species (Booth and Bledsoe, 2009; Novinger and Coon, 2000; Paul and Meyer, 2001; Sear et al., 2010).

Urban land use results in a hydrologically efficient system with impervious surfaces reducing infiltration and evapotranspiration, promoting an increase in runoff (Fig. 1.1) (Booth and Bledsoe, 2009; Brabec, 2009; Brown et al., 2009; Konrad and Booth, 2005; Paul and Meyer, 2001). Arnold, Jr. and Gibbons (1996) suggest watersheds with impervious surface area proportions as small as 10% will cause runoff volumes to double. Impervious surfaces collect sediment and debris between rainfall events, degrading runoff quality entering local streams (Pitt, 1999; Sansalone and Cristina, 2004). Fertilizers and detergents used in residential areas are flushed into water systems from lawns, driveways, and sidewalks (Brabec, 2009; Pitt, 1999). Impervious surfaces exposed to solar radiation increases the water temperature as runoff flows over these surfaces (Arnold, Jr. and Gibbons, 1996; Booth and Bledsoe, 2009; Poole and Berman, 2001). Water quality issues are most apparent in areas with a large impervious surface area, without riparian buffers.

SWM systems aim to reduce the stress urban development has on local streams. In the past, urban streams were modified to convey the surface runoff in the most efficient way and move it through the urban catchment with minimal flooding (Brabec, 2009; Brown et al., 2009; Burns et al., 2012). Management strategies have changed as approaches to watersheds become more holistic and urban streams are considered as a resource and habitat for species at risk (Booth and...
Bledsoe, 2009; Brown et al., 2009). The primary goals of SWM are to control flooding, address water quality and maintain base flow conditions (Huntington, 2006; Loperfido et al., 2014; Scholes et al., 2008; Smith et al., 1993). This is achieved through BMPs, which effectively address management criteria and protect aquatic life and their habitats (Bradford and Gharabaghi, 2004; Huber et al., 2006; Scholes et al., 2008).

Low impact development (LID) practices achieve SWM goals of maintaining the pre-developed conditions of runoff quantity and quality by increasing retention and infiltration of stormwater runoff at the subwatershed scale (Burns et al., 2012; Pyke et al., 2011). Key factors of LID are depicted as follows: provide runoff prevention through structural features, manage stormwater close to the source, and design landscapes that consider various functions such as filtration and site aesthetics (Burns et al., 2012; Credit Valley Conservation, 2010; Huber et al., 2006; Pyke et al., 2011; Walsh et al., 2005). Southern Ontario LID design strategies attempt to preserve important hydrologic features such as wetlands and floodplains, and avoid development on permeable soils (Credit Valley Conservation, 2010).

Structural LID practices such as green roofs, soakaways, bioretention, vegetated filter strips, permeable pavement, perforated pipe systems, and treatment wetlands (Table 1.1), are integral to understanding the treatment approach of SWM (Malaviya and Singh, 2012; Scholes et al., 2005a). The design process involves the necessary components of evaluating the site conditions and defining the environmental criteria, such as flood protection, water quality, and erosion control through BMPs (Dietz and Clausen, 2008; Loperfido et al., 2014; Pyke et al., 2011). LID has proven to achieve SWM goals compared to traditional SWM facilities and development designs. Dietz and Clausen (2008) reported consistent pre and post development pollutant loads with the use of bioretention systems, swales and pervious pavement techniques. Their results
indicate that traditional development techniques resulted in an increase in total phosphorus (TP) and total nitrogen (TN) of 200 and 1000 g/km² y (Dietz and Clausen, 2008). Li and Davis (2009) found mass removal efficiencies for bioretention systems with a sandy loam media, ~ 0.8 deep, and weir PVC outflows, of 100 % and 97 % for TP and TN, respectively.

The objective of this thesis was to evaluate the performance of PWs as an end-of-pipe LID feature. The PW is the final stage of a stormwater treatment train, intercepting the effluent flow from a stormwater pond through rainfall events before it enters the adjacent stream. PWs function similarly to bioretention systems, designed without underlying drainage and structural basin constraints. A PW is a treatment wetland located within in the stream flood plain corridor, incorporated into the natural channel design using the available space. The PW is the final stage of a SWM treatment train, in efforts to remediate water quality and provided additional detention of stormwater runoff at the source. Through monitoring water quantity and quality around a PW, this study provides quantitative evidence to the benefits PWs contribute to restoration projects in Southern Ontario. The dataset created from monitoring runoff through the PW provide water quality data to validate the Unified Stormwater Treatment Model (USTM) for effluent water quality. A precise model benefits consultants and ministry researchers in the design and planning of restoration projects. The working hypotheses for this study are:

- the PW will attenuate discharge and provide adequate residence time to mitigate the impacts of water temperature, total suspended sediment (TSS) and total dissolved load (TDS) on local streams;

- the temporal application of the USTM will represent the response of effluent TDS from a PW on an event basis.
Both hypotheses contribute to the overall objective of demonstrating that PWs are effective at improving stormwater quantity and quality when included in SWM systems and stream restoration projects.
Figure 1.1: Influences to water budget from impervious surface area (Arnold, Jr. and Gibbons, 1996)
Figure 1.2: Hydrograph reaction to post-developed land use changes (modified from Leopold, 1968)
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<th>LID Structural Practice</th>
<th>Description</th>
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<td><strong>Green roofs</strong></td>
<td>A vegetated rooftop that provides runoff retention, filtration and promotes evapotranspiration. It is important to consider the drawbacks such as water damage to roofs, and the maintenance in the event of extreme weather. Green roofs have the capacity to remove water pollutants and achieve water balance.</td>
</tr>
<tr>
<td><strong>Soakaways</strong></td>
<td>An underground porous chamber that promote infiltration and provide onsite runoff storage. There is also a risk of soil contamination, seepage, standing water, and the awareness that if on private property, building owners will need to be educated. Pre-treatment is important to prevent sediment and debris from entering, by using leaf screens or in ground filters.</td>
</tr>
<tr>
<td><strong>Bioretention</strong></td>
<td>A porous soil media, underlying a layer of mulch, which allows the percolation and filtration of runoff and promotes groundwater infiltration. The main component is the filter bed media for filtration, while mulch ground cover is efficient at heavy metal removal and plants provide a level of pollutant remediation and transpiration. There is a risk of groundwater and soil contamination, minimized by using infiltration from source areas that are less contaminated and applying sedimentation pre-treatment practices.</td>
</tr>
<tr>
<td><strong>Vegetated filter strips</strong></td>
<td>A vegetated strip for sedimentation and filtration of flowing stormwater runoff. Filter strips could provide pollutant removal with proper maintenance, and offer areas for snow storage and treatment. There is a low risk for groundwater and soil contamination.</td>
</tr>
<tr>
<td><strong>Permeable pavement</strong></td>
<td>A porous asphalt or the use of paving stones that promote stormwater infiltration. Permeable pavements are effective in various circumstances such as low traffic roads, parking lots, driveways, pedestrian plazas, etc. There is a risk of groundwater contamination as most pollutants move through infiltration practices. These systems susceptibility to clogging and require maintenance.</td>
</tr>
<tr>
<td><strong>Perforated pipe systems</strong></td>
<td>The use of a porous pipe in place of conventional storm sewer pipes to allow for infiltration of low flow. They are suitable for treating runoff from roofs, walkways, parking lots and low to medium traffic roads. Infiltration of de-icing salt materials into a perforated pipe system cause concern for potential contamination in groundwater. To reduce this risk, avoid using these systems in high traffic areas or pollution hot spots, or use oil and grit separators. Pre-treatment to prevent sediment from entering the infiltration system is imperative as it could cause clogging and failure.</td>
</tr>
<tr>
<td><strong>Treatment Wetlands</strong></td>
<td>A vegetated system with plants tolerant to wet conditions, that offers extended detention of stormwater runoff and remediate stormwater pollutants. Designed to mimic natural wetlands, these systems exploit the use of vegetation to remove low levels of stormwater pollutants. Underlying porous media provides a storage capacity and promotes filtration for subsurface flow. Treatment wetlands are tolerant hydraulic changes and pollutants loads. These systems require little maintenance, as they tend to create feedback loops that repair and increase treatment over time.</td>
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Chapter 2.0 Pocket wetlands as a means to improve stormwater management control of flow and water quality: a case study from the Credit River Watershed in Brampton, ON, Canada

2.1 Abstract

Stormwater management (SWM) is important for maintaining stormwater runoff volumes and mitigating water quality from rainfall events. Stormwater management has evolved from simple stormwater ponds to multiple SWM systems applied in series, as treatment trains. In the case study presented, a stormwater treatment train (STT) conveys stormwater runoff from a residential area, through conduits into a settling pond, which travel through a pocket wetland (PW) before entering a stream. It is assumed that this treatment train approach mitigates urban impacts on adjacent streams by retaining and detaining flow and improving water quality. The addition of PWs to the treatment train is relatively new; it is believed to provide a final ‘polishing’ step and potentially serve as increased capacity during dry periods. To understand the performance of the PW, discharge and water quality parameters were monitored at a stormwater PW in the Churchville subwatershed in Brampton, Ontario.

The working hypothesis for this study was that a PW, as part of a stormwater treatment train, would attenuate peak discharge and mitigate runoff water quality entering the stream. Monitoring took place from May – Oct. 2014, with event analysis of 21 rainfall events. Discharge, temperature and total dissolved solids (TDS) were recorded upstream and downstream of the PW confluence, thus any changes in attributes were due to flow contributions from the PW. Similar measurements at the stormwater pond outlet (point of entrance into the PW) provided water quality variables of the PW influent discharge. Total suspended solids (TSS) were evaluated through discrete sampling, while two rainfall events were captured with ISCO pump-samplers to assess sediment flux through events. The PW attenuated flow as the maximum
change in discharge between the upstream and downstream monitoring sites was 0.55 mm over 
~5 days in response to a total rainfall event of 30.75 mm, with 9.12 mm influent discharge. 
Storage capacity within the wetland decreased after an intense rainfall (17.3 mm/hr) on Aug. 1. 
The average event residence time was 2.2 ± 1.0 hr and did not fully eliminate the ‘first flush’ but 
the overall change in water quality between upstream and downstream monitoring sites was 
negligible. In stream water temperature changes were minor (less than 1°C), however changes 
were observed. Small increases in water temperature (less than 0.5°C) were observed during base 
flow conditions as warm surface water drained into the stream while event flow caused a net 
cooling (less than 1°C). The average change in stream total TDS was small (base flow, -5 ± 60 
mg/L; event flow, 0.3 ± 7 mg/L) although there was a net increase in TDS from the upstream to 
downstream monitoring locations. Rainfall inputs generally diluted TDS within in the stream and 
the stormwater management system. The average change in TSS from upstream to downstream 
ranged from -21.96 – 31.34 mg/L, while influent TSS ranged from 1.81 – 30.29 mg/L. High 
standard deviation of effluent TSS during base flow conditions suggest a small flux of TSS as the 
PW drained. TSS input from rainfall events was greatest during rainfall events after Sept. 8, and 
coincides with larger rainfall events. Both the upstream and downstream pump-sampler TSS 
peaked roughly two hours after a rainfall event when effluent TSS was also highest. There was 
little difference in TSS after the initial pulse in runoff. This study demonstrates that PWs used 
within the stormwater treatment train is an effective means of mitigating water quality issues and 
providing additional water retention time. Although PWs are not formally included in current 
policy requirements, the evidence from this study suggests the inclusion of PWs in future 
projects would be valuable.
2.2 Introduction

Urban land use changes alter watershed hydrological properties with the potential to affect surface runoff response to rainfall events. Impervious surface area (ISA) in a watershed increases due to urban development (Booth and Bledsoe, 2009; Brabec, 2009; Brown et al., 2009; Konrad and Booth, 2005; Paul and Meyer, 2001). ISA limits infiltration, thus increases overland flow and surface runoff response to rainstorms and can affect runoff water quality in streams, ponds and lakes, affecting aquatic species and habitat (Arnold, Jr. and Gibbons, 1996; Booth and Bledsoe, 2009; Brabec, 2009; Konrad and Booth, 2005; Sansalone and Cristina, 2004). Local species at risk have specific preferences for water temperature, turbidity and chemical balance that are altered by urban runoff. ISAs accumulate particulates and due to the hydraulic efficiency of these surfaces, debris is easily washed into local streams during rainfall runoff events (Arnold, Jr. and Gibbons, 1996; Paul and Meyer, 2001; Sansalone and Cristina, 2004). The exposure of ISAs to solar radiation warms these surfaces resulting in an increase in runoff temperatures, degrading stormwater quality (Bradford and Gharabaghi, 2004; Poole and Berman, 2001). Studies suggest ISA in watersheds as low as 10% will result in the onset of observable water quality degradation while 20% ISA will result in hydrological changes (Brabec, 2009; Bradford and Gharabaghi, 2004; Eimers and McDonald, 2014; Paul and Meyer, 2001). Recently, Eimers and McDonald (2014) reported a total ISA threshold as low as 10% ISA will result in hydrological impairment, with an increase of extreme flow events and high flow variability.

Stormwater management (SWM) systems are used to mitigate increases in ISA and reduce the effects of urbanization on local water systems by improving water quality (Bradford & Gharabaghi, 2004; Hunt et al., 2011; Malaviya & Singh, 2012; Pitt, 1999; Pyke et al., 2011; Scholes et al., 2008; Shutes et al., 2005). Low impact development (LID) attenuate peak
discharge and remediate stormwater quality onsite, while maintaining aesthetics. Wetlands, bioretention systems, permeable pavements and vegetated swales are commonly used LID features (Arnold, Jr. and Gibbons, 1996; Booth and Bledsoe, 2009). Stormwater ponds (SWP) are a SWM facility used to detain stormwater, regulating flow from large storm events and provide space for sediment and pollutants to settle (Booth and Bledsoe, 2009). SWP are most effective at detaining water and are designed to provide settling for particles greater than 40 μm (Ministry of the Environment, 2003; Toronto and Region Conservation Authority, 2012). As smaller sized sediments and pollutants can affect the ecology downstream, other LIDs use filtration methods in unison with traditional SWM. Bioretention systems are areas preserved for the infiltration of surface runoff though an engineered media responsible for the removal of metals, fine-grained sediment, organics and harmful chemicals (phosphorus, nitrogen, etc.) (Endreny and Collins, 2009; Li and Davis, 2009; Roy-Poirier et al., 2010). These systems are also effective at controlling small magnitude storm events that occur more frequently, detaining most of the water which would otherwise contribute to discharge (Burns et al., 2012; Pitt, 1999).

A pocket wetland (PW) is an end-of-pipe LID practice that provides additional remediation of water quality parameters, with further water detention. The PW is an infiltration system, responsible for dispersing flow over a densely vegetated surface allowing water to infiltrate into the media (Davis et al., 2012; Hunt et al., 2011; Li and Davis, 2009; Malaviya and Singh, 2012; Roy-Poirier et al., 2010; Wadzuk et al., 2010). A PW will detain light to intermediate rainfall events providing infiltration into subsurface media before entering the stream, while heavy events may pass as surface flow as the PW becomes saturated (Hunt et al., 2011; Sun and Yang, 2012). The longer it takes stormwater runoff to move through a PW, the greater the potential residence time and water quality benefits. The extended flow path through the PW allows for
additional sedimentation, and filtration of stormwater, while the vegetation provides shade to prevent an increase in water temperatures (Davis et al., 2012; Hunt et al., 2011; Li and Davis, 2009). Subsurface flow moves through the underlying media much like a bioretention system, while media pore space stores water, which is then released slowly into the stream and maintains base flow conditions.

It is common for SWM systems to include multiple BMPs in series, creating a treatment train of infrastructure (Bradford and Gharabaghi, 2004; Scholes et al., 2008; Shutes et al., 2005). Stormwater treatment trains (STT) increase the flow path for stormwater runoff attenuation and additional means to improve water quality through multiple SWM and LID systems. Recent management practices in Ontario include STTs with end-of-pipe treatment at a SWP outlet (Bradford and Gharabaghi, 2004; Hunt et al., 2011; Malaviya and Singh, 2012). The working hypothesis in this study is the PW as the final stage in the STT will attenuate discharge and provide adequate residence time to mitigate the impacts of water temperature, total suspended solids (TSS) and total dissolved solids (TDS) on local streams. Monitoring and analysis through 21 rainfall events will estimate residence times within the PW and predict the wetland efficiency at moderating water quality for flow leaving the SWP and entering the adjacent stream.

2.3 Methods

2.3.1 Study Site

A major stream re-alignment and SWM project started in 2010 to deal with development northwest of Chinguacousy Rd. and Queen St. W, in Brampton Ontario (Fig 2.1; 43°39’49” N, 79°47’07” W). One objective of the SWM and stream restoration project was to improve the potential species at risk habitat, in particular redside dace (Credit Valley Conservation Authority,
A novel aspect of this SWM and stream restoration project was to incorporate PWs into the flood plain as a step within the STT. Discharge and water quality were monitored around a PW located within the Churchville sub-watershed from Apr. – Oct. 2014 (Fig. 2.1). The upstream area of the Churchville sub-watershed drains 3.1 km² of urban, commercial, construction, and conservation land. The PW is part of a 640 m stream restoration project completed in 2010. The stream is ~ 3 m wide, with bank full depths ranging from 0.3 – 0.6 m. This area receives an average annual rainfall of 681 mm, as measured at Toronto Pearson International Airport from 1981 to 2010. The substrate is poorly drained Chinguacousy series clay loams and the underlying geology is glacial till from calcareous limestone and shale (Ontario Geological Survey, 2010). Upstream of the completed channel alignment and restoration, roughly 0.6 km² of construction and the remaining 1960 m channel restoration were ongoing during the 2014 study period. The catchment contains stormwater management systems conveying stormwater runoff through subsurface conduits, into a SWP before entering the stream. The areas connected directly to the stream without SWM, are the remaining forested areas and the restored stream corridor (Fig 2.1).

The PW evaluated in this study is located northeast of the intersection at James Potter Rd. and Queen St. W (Fig 2.1). It is the final stage within the treatment train for stormwater runoff, designed to manage the total drainage of 0.15 km² residential area with about 1.8 km of road network (Fig. 2.1). The runoff initially enters a conveyance system beneath roads where it collects and enters the SWP (Fig 2.1). The SWP has a 3 m permanent water depth, which covers ~0.028 km² while the 5 m maximum water level covers ~0.093 km² due to the ponds sloped sides. The SWP drains to the PW, located in the realigned stream floodplain, ~5m from the Churchville tributary bank. The PW is a vegetated area (Table 2.1) roughly 900 m² allowing
dispersion and infiltration of SWP water into the subsurface aggregate below (60% rip-rap, 40% soil). The PW has limited storage capacity, allowing water to enter the stream as subsurface flow, or engage overland flow paths when the basin is full. The PW only receives the SWP effluent flow during runoff events, initiating the overland flow to the Churchville tributary, while the subsurface flow was consistent between rainfalls. An incised channel (~0.20 m wide; ~0.15 m deep) connects the PW with the stream. This connection was not part of the original PW design and developed throughout the study period. Average standing surface water depth (~ 0.15 m) remained within the PW, suggesting the subsurface media remained saturated. An increase of overland flow occurred during rainfall events as the surface water breached the banks of the PW and drained directly to the Churchville tributary.

2.3.2 Data Collection

Rainfall data recorded at 5-minute intervals was provided by the Region of Peel from a rain gauge ~2 km northeast of the study site (Fig 2.1). Stream gauging cross-sections and monitoring points were located upstream, downstream and at the SWP outlet to monitor water level, temperature, total suspended solids (TSS) and electrical conductivity (EC). Onset HOBO water level loggers (range: 0 – 9 m; accuracy: ± 0.5 cm) and conductivity sensors (range: 0 - 1000 μS/cm; accuracy: 5 μS/cm) measured and recorded water level and EC, respectively, at 5-minutes intervals at all three locations. Both sensors include thermocouples that measure water temperature (range: -20 – 50 °C; accuracy: ± 0.44 °C). A fourth logger was used to measure air temperature and atmospheric pressure (accuracy: ± 0.05 kPa). Stilling wells housed the water level loggers within the stream (~ 0.02m from stream bed) to ensure currents and ripples did not interfere with the measurements. EC sensors were installed in a protective PVC housing, attached to the stilling wells 0.20 m from the bottom of the water column. The water level and
conductivity sensors at the SWP outlet were strapped to the bottom of the conduit. A Marsh-McBirney Flow-Mate 2000 (range: -0.15 – 6 m/s; accuracy: ± 2%) was used to manually measure velocity at cross sections in the channel (upstream and downstream locations) to calculate stream discharge. Discharge at the SWP outlet was estimated by measuring the time that lapsed while filling a 40 L container.

Site maintenance, stream gauging and TSS sample collection were conducted on a weekly basis. A DH48 depth-integrating suspended solids sampler was used to collect ~250 ml water samples for TSS at all three monitoring locations. TSS samples were filtered through pre-weighed polycarbonate 0.4 µm filters, dried at 50 °C overnight and re-weighed to determine TSS. Additional site visits occurred during rainfall events to conduct additional gauging and sample collection. It was difficult to catch all rainfall events due weather forecasting and event timing, as some rainfall events occurred overnight and events would often miss the study site (e.g., convective summer storms). ISCO Portable Pump-samplers were used to collect 500 ml water samples upstream and downstream of the PW every hour over a 24-hour period. Although the pump-samplers were sent out multiple times, only two storms were successfully sampled. In several instances, one sampler would fail and thus a comparison between upstream and downstream would not be possible and in one case, flow was sufficiently high that the pump-samplers were disturbed and samples were lost.

2.3.3 Data analysis

Water level, temperature and conductivity sensors monitoring upstream, downstream and the SWP outlet, were processed to determine influent and effluent flow from the PW. Effluent flow conditions were determined by the difference in discharge or water quality variables between upstream and downstream location from the PW. Total discharge was calculated using the 5-min
data from the water level loggers both upstream, downstream and at the SWP outlet. Water level loggers measured total pressure underneath the water column and converted to stage measurements by subtracting atmospheric pressure from total pressure. Continuous discharge was assessed using stage/discharge rating curve for each monitoring location. Discharge was normalized to millimeters to compare at each monitoring location. Event analysis defined 21 rainfall events and corresponding runoff ratio coefficients. A straight-line hydrograph separation technique was used to define event flow. Antecedence was calculated as the total rainfall that occurred 5 days before the rainfall event. EC was converted to total dissolved solids (TDS) with a coefficient of 0.67 (Atekwana et al., 2004; Marandi et al., 2013; Pellerin and Wollheim, 2008; Walton, 1989). A t-distribution (α = 0.05) difference of means test was used to analyse any significant change in temperature, TDS and TSS from upstream to downstream of the PW. EC concentrations traced the ‘first flush’ through rainfall events indicating the PW residence time.

During the field season there was active construction upstream of the study area June 27 – Sept. 8, and at various times during the season construction disrupted our water quality monitoring efforts. The PW system was also altered when a concrete orifice plate, located at the SWP outlet, was removed and thus changed the influent runoff conditions to the PW. Water quantity was determined with a straight-line delineation but the water quality results presented in this study from before June 27 reflect the period when the orifice plate was in place, and results after Sept. 8 has no orifice plate.
2.4 Results

2.4.1 Rainfall and Runoff Events

Rainfall totalled 494 mm during 2014 field season (Apr. 25-Oct. 30, 2014) as measured at Chris Gibson Park, Brampton ON (Fig. 2.2a). This rainfall occurred over 21 rainfall-runoff response events (Table 2.2). Total monthly rainfall in June, July and Sept., 85 mm, 83.5 mm and 140 mm respectively, exceeded monthly totals from the past 30 years prior to 2010 (Environment Canada, 2010). Rainfall events typically lasted 0.3-8.0 hrs, with amounts ranging between 2.5 - 61.5 mm (Table 2.2). There was an average runoff response time of 90.1 ± 37.8 hrs, producing average event discharges of 2.10 ± 1.75 mm, 2.27 ± 1.96 mm, and 6.80 ± 5.61 mm at upstream, downstream and SWP outlet, respectively. As a result of a higher proportion of ISA within the SWP catchment, the average runoff ratio was 0.31 ± 0.11 for the stormwater catchment, versus 0.11 ± 0.06 at both upstream and downstream monitoring points.

Regression analysis between the total rainfall and total discharge for each event indicate that the catchment efficiently manages runoff for a variety of rainfall event magnitudes and durations (Fig. 2.3). The linear regression (upstream, $R^2 = 0.84$, RMSE = 0.67 mm; downstream, $R^2 = 0.86$, RMSE = 0.71 mm; SWP outlet, $R^2 = 0.85$, RMSE = 2.06 mm) of runoff magnitudes strongly correlate with rainfall. The linear trends indicated there was no threshold or major change attributed to antecedence. Throughout most rainfall events, the average residence time of the PW was 2.2 ± 1.0 hrs (Table 2.2).

2.4.2 Water Quality During and Between Runoff Events

There was little difference (less than 1 °C) between water temperatures at the upstream and downstream monitoring locations during the season. During rainfall events, there was a
measured decrease in water temperature through the event and in general, there was a strong diurnal temperature fluctuation on all days (Fig. 2.2 b, d.). Average water temperatures at the SWP outlet were 16.6 ± 3.1 °C, while the instream water temperatures were 16.0 ± 5.0 °C and 14.6 ± 4.6 °C, at the upstream and downstream sites, respectively. Water temperature lagged (~4-5 hrs) the diurnal atmospheric temperature, peaking between 4:00 pm - 5:00 pm. There was no measureable difference in water temperature from upstream to downstream after considering the accuracy of the sensors used (Table 2.3). A t-test also indicated there were no differences between the upstream and downstream temperatures (Table 2.4).

TDS decreased as rainfall runoff response increased at the three monitoring locations (Fig. 2.2 c, e, g). A difference of means t-test (α = 0.05) shows a statistically significant difference in TDS between upstream and downstream before June 27 and no difference after Sept. 8 (Table 2.4). The increase in TDS before June 27 was small (~33 - 134 mg/L), peaking ~2 hrs after a rainfall event. TDS concentrations were larger while the orifice plate was in place (Table 2.3). Average effluent TDS was 32 ± 26 mg/L and 27 ± 30 mg/L, for base flow and event flow respectively, before June 27. It was much less for both base flow and event flow conditions Sept. 8 (-5 ± 60 mg/L and 0.3 ± 7 mg/L, respectively). Effluent TDS during base flow conditions had a large standard deviation suggesting surface water would dilute the system but also release large TDS concentrations. Effluent TDS during rainfall events was subtle, especially after the removal of the SWP outlet orifice plate (Table 2.3; Table 2.4).

TSS was greater downstream from the PW as shown by the average TSS before and after the removal of the SWP outlet orifice during base flow and event flow conditions (Table 2.3). Only two events were captured with the pump-samplers, while the majority of TSS samples were taken between events during weekly field visits. There was no seasonal trend in the TSS. Pump-
sampler concentrations during the two measured events indicate that TSS was greater
downstream of the PW during events (Fig. 2.5). On June 17 maximum TSS was 20.16 mg/L and
23.28 mg/L for upstream and downstream stations, despite the large runoff response for this
event (0.057 m³/s and 0.062 m³/s, respectively) (Fig. 2.6). On Oct. 14 maximum TSS was 72
mg/L and 56 mg/L for the upstream and downstream stations, although runoff was lower
compared to the June 17 event (Fig. 2.7). The Oct. 14 rainfall intensity was low, with 4.5 mm in
2.84 hrs and 0 mm of rain in the previous 5-days (Table 2.2). A t-test shows that there was a
statistically significant difference between TSS at the upstream and downstream locations for the
June 17 event but not the Oct. 14 event (Table 2.4).

2.5 Discussion

2.5.1 Pocket wetland flow attenuation

The PW is the final stage in the STT, responsible for additional attenuation of stormwater
runoff and mitigation of the water quality for runoff entering the stream. Through monitoring
upstream, downstream and the SWP outlet, we determined whether the PW effectively managed
water discharge, temperature, TDS and TSS. As water entered the PW through the SWP outlet,
the difference in water parameters at the relative upstream and downstream monitoring locations,
determined effluent discharge and water quality. If minimal changes occurred in the water
quantity and quality between the upstream and downstream monitoring locations (Fig. 2.1), it
would be reasonable to infer that the PW was effective.

There was little change in runoff between the monitoring stations upstream and downstream
of the PW (Table 2.3). The discharge, which entered the PW from the SWP outlet, was roughly
equivalent to four times the discharge within the stream. This resulted in an average change in stream
discharge of 0.17 ± 0.36 mm. It is clear the SWP attenuated the rainfall response runoff as influent water to the PW had limited measurable impact on runoff measured at the downstream monitoring location. Runoff ratios also support this as the stream experienced small ratios (0.11 ± 0.06) while the runoff ratios entering the wetland were 0.31 ± 0.11 (Table 2.2). Urban runoff coefficients are expected to range from 0.40 – 0.95 for urban areas due to an increase in ISA (Arnold, Jr. and Gibbons, 1996; Eimers and McDonald, 2014). As ratios within the channel were low and more comparable to natural land use runoff coefficients (0.05 – 0.40), this indicated an increase in detention provided by the SWM. Dietz and Clausen (2008) compared traditional development techniques to LID techniques and found traditional techniques resulted in a 49,000% increase of annual runoff compared to a 0% increase under LID and SWM practices. Results suggest that the PW will attenuate an average 0.17 ± 0.36 mm of the influent 6.80 ± 5.61 mm runoff from the SWP outlet, by providing active pool storage and pore space of the media below the wetland (Table 2.2).

While few studies quantitatively measure the PW performance, Malaviya and Singh (2012) advocate the use of PWs to further attenuate peak flows and improve the retention of stormwater runoff to further protect stream systems in developed watersheds. Loperfido et al. (2014) compared the runoff from four similar sized catchments in a temperate environment with different land use and stormwater management practices. They found about a 30% increase in runoff due to a 20% increase in ISA over catchments ranging from 1.11 – 7.02 km² (Loperfido et al., 2014). The Churchville PW and other stormwater BMPs reduced mean runoff ratios from 0.31 at SWP outlet to 0.11 for the entire upstream catchment (Table 2.2).

The minor increase in discharge downstream of the PW suggests that only a small volume of water entered the stream during rainfall events (Table 2.3). It is suggested that the ideal PW
should detain water for 10 - 15 hrs for a minimum of 30 minute rainfall event (Shutes et al., 1999). The average event residence time for the Churchville PW was 2.2 ± 1.0 hrs, while base flow residence time as it is reported in the literature, would be much longer (Wadzuk et al., 2010). The PW residence time has great benefits to water quality improvements. The longer the residence time within the PW the greater the hydraulic efficiency (Persson et al., 1999). These systems must detain the water as close to the nominal residence time to achieve the greatest water quality performance (Wong et al., 2006). During times of frequent rainfall events, mean residence time may decrease, as the volume of water within the PW has not had time to drain. This occurred on June 11 and Sept. 5, which had residence times roughly half that of the preceding rainfall event (Table 2.2).

The regression analysis of the change in discharge between upstream and downstream of the PW shows a positive trend as more stormwater runoff entered the PW system from SWP outlet (Fig. 2.4). The regression also illustrated a change in storage after the Aug. 1 rainfall event, which was one of the more intense storms of the season (17.3 mm/hr). An intense event quickly fills both the SWP and PW. More water is forced to overland flow than subsurface flow as the hydraulic conductivity of the underlying media is compromised by clogging (Knowles et al., 2011). The lack of reed establishment in the PW (Fig. 2.8) supports Rousseau and Horton (2005) findings that media clogging can hinder reed establishment due to the anaerobic sludge accumulation. The difference in discharge between the stream flow response and SWP outlet had two distinct responses triggered by the Aug. 1 event (Fig. 2.4). A weak linear regression ($R^2 = 0.19$) suggests the PW detained water volumes differently for each rainfall event. The intense rainfall on Aug. 1 explains the division of the two ranges (above and below the linear regression) (Fig. 2.4). There was an increase in the amount of stormwater released into the channel from the
PW after rainfall on Aug. 1. This increase in effluent PW discharge occurred even though average runoff ratios before Aug. 1 were greater than those after at 0.36 ± 0.12 and 0.26 ± 0.06, respectively.

2.5.2 Thermal remediation of urban runoff

Atmospheric temperatures are most influential to thermal regimes within a stream (Caissie, 2006). Although atmospheric temperatures peaked around 30°C, channel temperatures rarely exceeded 25°C (June 27 – July 2). This is important to fish species as 25°C is the upper limit for the ideal range for the endangered redside dace (Novinger and Coon, 2000) which is a key species in the Churchville watershed in southern Ontario (Cosewic, 2007). By Ontario Stream Assessment Protocol (Stanfield, 2013), which states water temperature must be assessed between 4:00 pm – 4:30 pm, there was no change in mean water temperature from upstream to downstream (Table 2.4). The difference of means t-test showed statistically insignificant differences. Effluent PW discharge increased downstream water temperatures between 0.0-0.5°C, while a cooling effect ranging between 0.0 – 0.75°C occurred more frequently during base flow conditions. This response was clear from the temporal analysis of temperature time series, but the measured difference in water temperature from upstream to downstream was within error of the sensors, suggesting any change in temperature insignificant. The in-depth time series analysis suggests surface flow from the PW would warm stream temperatures slightly, but this had no measurable influence to stream conditions as mentioned in the previous statement. Mitsch et al. (2005) attributed the warm water temperatures to a lack of established vegetation cover, which led to surface water warming by solar radiation. PWs need to provide dense vegetation to shade pooled water, thus dampening the warming effect. The Churchville PW is newly constructed feature and will take time for wetland vegetation to fully establish (Fig. 2.8).
Subsurface flow from the PW into the stream generated a cooling effect after effluent surface flows returned to normal, as seen by others (e.g., Poole & Berman, 2001). The overall temperature gradient within the stream was minor (within the sensor accuracy) suggesting the PW design mitigates urban stormwater runoff quality with respect to thermal properties.

2.5.3 TDS remediation within the PW

TDS responded inversely to stream discharge, as it was relatively high during base flow conditions, and decreased as runoff diluted the system at all three monitoring locations. A dilution in the system resulted in a lower TDS concentration throughout the field season. These findings are similar to those presented in a study by Wadzuk et al. (2010) where TDS during base flows were greater than those during rainfall events. Both the stream and the SWP outlet had higher average TDS concentrations before June 27 (orifice plate in place) than after Sept. 8 (orifice plate removed). The increased TDS at SWP outlet before June 27 was due to the orifice plate at the SWP outlet. The orifice plate held back water in the conduits leading to the PW, allowing more evaporation, which lead to higher inferred TDS concentrations (Maurya, et al. 2011). Hydrologic systems with ISAs are vulnerable to ‘first flush’ peaks in dissolved nutrients responsible for increasing the concentration of TDS into the stream (Lawler et al. 2006; Sansalone & Cristina, 2004). The results from the Churchville site suggest that the ‘first flush’ was more prevalent when the orifice plate was in place as TDS behind the orifice plate was at greater concentrations before June 27 than after Sept. 8 (Fig. 2.2; Table 2.3). Effluent PW runoff display little change to TDS within the stream after Sept. 8 (Table 2.4).

The PW was designed to act as a sink for particles and dissolved solids, providing an environment for sedimentation and sorption to occur. As long as sorption equilibrium has not been reached within the PW, an adequate residence time should allow bioremediation and the
physical or chemical adhesion to soil particles of dissolved nutrients in an aqueous phase to occur (Malaviya and Singh, 2012). Wadzuk et al. (2010) found no change in TDS as water flowed through 4000 m$^2$ treatment wetland, with a rainfall event residence of just a few hours. There was no change in effluent conductivity during a 3 year study by Mitsch et al. (2005) where they artificially flooded a 11600 m$^2$ wetland. There were small improvements in effluent TDS of 78 mg/L during normal flow conditions (Mitsch et al., 2005). Both the Wadzuk et al. (2010) and Mitsch et al. (2005) studies have similar results to the present study and illustrate the relationship between residence time within treatment wetlands and runoff water quality.

2.5.4 Mobilizing suspended solids

TSS at the SWP outlet varied by 6 mg/L throughout the field season with four cases where it increased to 10 – 30 mg/L (Fig. 2.4). Three of these cases occurred before June 27 (orifice plate in place) at discharges greater than 0.0076 mm. Results suggest that at times when there was a noticeable increase in velocity at the SWP outlet, sediment that settled behind the orifice plate in the conduit from the SWP mobilized and was not from the pond, as this pond drains from the bottom of the settling water column. Although TSS was high at these times, there were minimal changes in the TSS within the stream. The difference of means t-test suggests there was no major difference between the upstream and downstream monitoring points, but this test was statistically insignificant due to the small sample size (Table 2.4). Van de Moortel et al. (2009) found the removal of TSS was most efficient as water moved through sub-surface pathways. They noticed sedimentation would also take place but it required sufficient retention time. Although there was no significant change in retention time at the Churchville PW, effluent TSS showed a significant decrease from influent TSS (Table 2.3).
Discrete sampling during rainfall events following Sept. 8 showed larger effluent TSS at the Churchville PW (Table 2.3). Discrete samples showed an increase in TSS during rainfall events (influent, $3.28 \pm 1.59$ mg/L; effluent, $4.26 \pm 15.67$ mg/L). Other studies reported improvements of TSS through PW for base and event flows (Carleton et al., 2001; Wadzuk et al., 2010). Mitsch et al. (2005) reported an increase of soil development within the wetland over a ten-year span, but found this to lead to an increase in the export of TSS. They suggest that during rainfall events, anaerobic soil water along the edge of the PW is mixed into moving surface water, flushing the system (Mitsch et al., 2005). The average rainfall at the Churchville subwatershed was much larger after Sept. 8 (~17.40 mm), which would regularly flush the system, releasing higher TSS. Effluent TSS during base flow had a low average but high standard deviation ($1.95 \pm 6.92$ mg/L and $-0.45 \pm 15.39$ mg/L, before and after the orifice plate, respectively). Due to the small incised channel, a small flux of sediment entered the stream as the surface water drained from the PW. When considering Mitsch et al. (2005) suggestions about still water, as surface water drains from different areas of the PW the flux of TSS within these locations would explain the high standard deviation.

The pump-sampler TSS responded similarly to TDS during the rainfall events (Fig. 2.5 – 2.7). The greatest difference from upstream to downstream, occurred roughly two hours after it began to rain, represented as the ‘first flush’ of mobilized sediments. After the initial pulse of water, there was no clear trend with a change in TSS downstream of the PW even though there was a peak change in discharge of $9.7 \times 10^{-4}$ mm and $5.8 \times 10^{-4}$ mm during the June 17 and Oct. 14 events, respectively. Other than the ‘first flush’ from the initial rainfall, the differences in upstream and downstream TSS were in range of the TSS differences recorded from discrete
sampling. This contradicts Wadzuk et al. (2010) findings that indicated an average increase in TSS of ~ 20 kg for all rainfall events compared to base flow conditions.

2.6 Conclusion

PWs are an effective addition to the STT as demonstrated by the assessment at the Churchville tributary in Brampton, Ontario. SWM around the Churchville tributary produced low runoff ratios from this residential area. The PW attenuated flow from the SWP outlet with an average residence time of 2.3 hrs. Antecedence was the only factor that changed residence time, as the PW had a limited storage capacity for stormwater runoff. This system provided resilience under different magnitudes of rainfall, shown by the change in storage capacity after the Aug. 1 rainfall event. The downstream water quality represented no significant change due to effluent PW discharge. The PW resulted in no measurable change in water temperature from upstream to downstream. The PW effectively remediated TDS concentrations, while close analysis suggested surface water drainage during base flow resulted in a small flux in TDS. Rainfall events later in the season resulting in larger releases of TSS but rainfall events were larger during this period with dryer antecedent conditions.

The PW would benefit from a longer residence time, to improve water quality further. An incised channel from the PW continuously drained into the Churchville tributary and reduced residency within the wetland. PWs prove to benefit stormwater runoff but require adequate space to handle different stormwater runoff volumes. An understanding of the ideal residency for stormwater quality improvements and spatial required would present the ideal circumstances for the implementation of a PW. This would aid urban planning and municipal SWM standards for the protection of urban channels.
Figure 2.1: The Churchville PW is located north of the intersection of James Potter Rd. and Queen Street West (43°39'49" N, 79°47'07" W). Runoff from the stormwater catchment (~ 0.15 km²) drains through conduits into the stormwater pond, and then flows through the PW (~ 900 m²) before entering the Churchville tributary.
Table 2.1: Wet meadow seed mix installed in the Churchville PW.

<table>
<thead>
<tr>
<th>Vegetation Species</th>
<th>Volume of Total (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Swamp Milkweed (Asclepias incarnata)</td>
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</tr>
<tr>
<td>Swamp Aster (Aster puniceus)</td>
<td>5</td>
</tr>
<tr>
<td>Three Way Sedge (Dulichium arundinaceum)</td>
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</tr>
<tr>
<td>Virginia Wild Rye (Elymus virginicus)</td>
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</tr>
<tr>
<td>Blue Flag (Iris versicolor)</td>
<td>5</td>
</tr>
<tr>
<td>Soft Rush (Juncus effuses)</td>
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</tr>
<tr>
<td>Rice Cutgrass (Leersia oryzoides)</td>
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</tr>
<tr>
<td>Sensitive Fern (Onoclea sensibilis)</td>
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</tr>
<tr>
<td>Tear-Thumb (Polygonum arifolium)</td>
<td>5</td>
</tr>
<tr>
<td>Wool Grass (Scirpus cyperinus)</td>
<td>5</td>
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<tr>
<td>Lesser Bur Reed (Sparganium americanum)</td>
<td>10</td>
</tr>
</tbody>
</table>
Figure 2.2: Summary of data collection: rainfall (mm), and atmospheric temperature (°C) (a.); discharge (mm), and water temperature (°C); total dissolved solids (mg/L), and total suspended solids (mg/L) at upstream (b., c.), downstream (d., e.) and the SWP outlet (f., g.) from May 24 – Oct. 30, 2014. The black lines indicate two rainfall events captured with the pump-samplers (June 17 and Oct. 14).
Table 2.2: Antecedent rainfall (mm), rainfall and runoff duration (hr), total rainfall (mm), total runoff (mm), runoff ratios and wetland residence time (hr) for 21 rainfall events.

<table>
<thead>
<tr>
<th>Event Date</th>
<th>5 day previous rainfall (mm)</th>
<th>Rainfall Duration (hr)</th>
<th>Runoff Duration (hr)</th>
<th>Rainfall (mm)</th>
<th>Total Runoff (mm)</th>
<th>Runoff Ratio</th>
<th>Wetland Residence Time (hr)</th>
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<tr>
<td></td>
<td></td>
<td></td>
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<td></td>
<td>upstream</td>
<td>downstream</td>
<td>SWP outlet</td>
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<td>1.48</td>
<td>1.76</td>
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<td>0.59</td>
<td>0.60</td>
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<td>3.93</td>
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<td>Oct. 06</td>
<td>23.00</td>
<td>1.5</td>
<td>113.2</td>
<td>8.00</td>
<td>1.39</td>
<td>1.75</td>
<td>1.94</td>
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<tr>
<td>Oct. 14</td>
<td>0.00</td>
<td>1.4</td>
<td>42.6</td>
<td>4.50</td>
<td>0.59</td>
<td>0.71</td>
<td>1.01</td>
</tr>
<tr>
<td>Oct. 16</td>
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<td>8.75</td>
<td>0.57</td>
<td>0.68</td>
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<tr>
<td>Oct. 20</td>
<td>10.75</td>
<td>0.8</td>
<td>46.5</td>
<td>2.50</td>
<td>0.12</td>
<td>0.16</td>
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</tr>
<tr>
<td>Mean</td>
<td>8.29</td>
<td>3.1</td>
<td>90.1</td>
<td>19.93</td>
<td>2.13</td>
<td>2.31</td>
<td>6.80</td>
</tr>
<tr>
<td></td>
<td>± 1.63</td>
<td>± 2.4</td>
<td>± 37.8</td>
<td>± 14.65</td>
<td>± 1.75</td>
<td>± 1.96</td>
<td>± 5.61</td>
</tr>
</tbody>
</table>
Figure 2.3: Linear regression between the total discharge and the total rainfall at upstream, downstream and SWP outlet per event from Apr. 25, 2014 to Oct. 30, 2014.
Figure 2.4: Linear regression of the discharge difference from upstream to downstream and the total SWP outlet discharge per event from Apr. 25, 2014 to Oct. 30, 2014.

\[ y = 0.0288x - 0.0133; R^2 = 0.19 \]
Table 2.3: Average event and base flow water temperature, EC, and TSS, before and after the removal of the orifice plate at the SWP outlet. TSS values are from discrete samples, excluding the pump-sampling events.

<table>
<thead>
<tr>
<th></th>
<th>Influent Flow (SWP outlet)</th>
<th>Effluent Flow (downstream-upstream)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Before June 27</td>
<td>After Sept. 8</td>
</tr>
<tr>
<td></td>
<td>(with orifice plate)</td>
<td>(without orifice plate)</td>
</tr>
<tr>
<td>Temperature(°C)</td>
<td>15.2 ± 3.1</td>
<td>12.0 ± 1.6</td>
</tr>
<tr>
<td>TDS (mg/L)</td>
<td>1419 ± 247</td>
<td>297 ± 57</td>
</tr>
<tr>
<td>TSS (mg/L)</td>
<td>8.1 ± 5.5</td>
<td>6.4 ± 3.0</td>
</tr>
</tbody>
</table>

Average base flow conditions

| Temperature(°C)              | 18.6 ± 2.4                 | 16.4 ± 2.3                           | NA                  | 16.4 ± 2.3             |
| TDS (mg/L)                   | 1211 ± 229                 | 263 ± 73                             | 27 ± 30             | 0.3 ± 7                |
| TSS (mg/L)                   | 10.7 ± 11.0                | 3.3 ± 1.6                            | 0.6 ± 1.1           | 4.3 ± 15.7             |
Table 2.4: Upstream and downstream difference of means (t-distribution) for TDS, Temperature, Temperature (4:00-4:30) and TSS with (Apr. 25, 2014 - June 25, 2014) and without (Sept. 8, 2014 – Oct. 30, 2014) SWP outlet orifice plate.

<table>
<thead>
<tr>
<th></th>
<th>Before June 27 (with orifice plate)</th>
<th>After Sept. 8 (without orifice plate)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Temperature (4:00-4:30)</td>
<td>TDS</td>
</tr>
<tr>
<td>n</td>
<td>17226</td>
<td>60</td>
</tr>
<tr>
<td>t-calc</td>
<td>-0.44</td>
<td>-0.05</td>
</tr>
<tr>
<td>P; one-tail</td>
<td>0.33</td>
<td>0.48</td>
</tr>
<tr>
<td>t-crit; one-tail</td>
<td>1.64</td>
<td>1.64</td>
</tr>
<tr>
<td>P; two-tail</td>
<td>0.66</td>
<td>0.96</td>
</tr>
<tr>
<td>t-crit; two-tail</td>
<td>1.96</td>
<td>1.96</td>
</tr>
</tbody>
</table>
Table 2.5: Upstream and downstream difference of means test (t-distribution) the pump-sampler TSS from the June 17, 2014 and Sept. 14, 2014 rainfall events.

<table>
<thead>
<tr>
<th>Pump-Sampling Event</th>
<th>June 17-18</th>
<th>Oct. 14-15</th>
</tr>
</thead>
<tbody>
<tr>
<td>t-calc</td>
<td>-4.23</td>
<td>-1.63</td>
</tr>
<tr>
<td>P; one-tail</td>
<td>0.00</td>
<td>0.06</td>
</tr>
<tr>
<td>t-crit; one-tail</td>
<td>1.72</td>
<td>1.71</td>
</tr>
<tr>
<td>P; two-tail</td>
<td>0.00</td>
<td>0.12</td>
</tr>
<tr>
<td>t-critical; two-tail</td>
<td>2.08</td>
<td>2.07</td>
</tr>
</tbody>
</table>
Figure 2.5: Discharge and Pump-sampler TSS correlation from the June 17, 2014 and Oct. 14, 2014 rainfall events. June 17 event produced a much smaller TSS yield with greater discharge compared to the Oct. 14 event.
Figure 2.6: June 17 rainfall event (mm), discharge (mm) and TSS (mg/L). Low TSS was recorded for an intense rainfall event.
Figure 2.7: Oct. 14 rainfall event (mm), discharge (mm) and TSS (mg/L). Although the rainfall intensity was low, with 4.5 mm in 2.8 hrs, antecedent conditions were dry as 0 mm of rain fell in the previous 5-days.
Figure 2.8: PW vegetation density on June 11 (a.) and Sept. 2 (b.). A lack of reed (~ 1.5 m tall in b.) establishment indicates possible clogging of the sub-surface media and leaves surface water vulnerable to radiation which warms surface flow.
Chapter 3.0 The temporal application of a Unified Stormwater Treatment Model for a treatment wetland in Brampton, Ontario

3.1 Abstract

The Unified Stormwater Treatment Model (USTM) is designed to test the effectiveness of stormwater treatment facilities. USTM is a physically based model using empirical measurements to simulate the natural water quality remediation of a pocket wetland (PW). The model simplifies the PWs hydrodynamic and water quality remediation processes to estimate the effluent concentration of total dissolved solids (TDS). The USTM uses a first-order kinetic decay algorithm with a hydrologic efficiency model to determine the effluent water quality concentrations. The PWs physical characteristics, water quality and influent discharge were monitored from May – October 2014. The hypothesis is the temporal application of the USTM will represent the response of effluent TDS from a PW on an event basis. The application of the USTM to this data set estimated effluent TDS concentrations poorly. The modelled effluent TDS responded predominantly to hydraulic loading, a function of influent discharge. Use and validation of the USTM is supported in the literature and is an important tool for stormwater management engineering and design. The input units suggest the USTM was developed as a time average model but this study proposes to investigate the potential of the model for instantaneous application. This model is used to determine the efficiency of a stormwater management facility to remediate water quality based on spatial constraints, hydrology and influent water quality concentrations before the final design to ensure a proper stormwater management designs.

3.2 Introduction

Stormwater management (SWM) systems are expensive to build and maintain so the design of these facilities is extremely important to ensure they provide suitable runoff control and improve water quality (Roy et al., 2008). Modeling the behaviour of SWM treatment approaches
is important when planning development or stream restoration (Bradford and Gharabaghi, 2004; Elliott and Trowsdale, 2007; Pitt, 1999; Wong et al., 2002). Through the combination of climate data and watershed measurements, stormwater models combine existing hydrological models to estimate the runoff for small catchments (Elliott and Trowsdale, 2007; Pitt, 1999). The estimated runoff discharge provides a basis for the SWM design knowing the volume and flow rates that need to be filtered and detained onsite (Brabec, 2009; Bradford and Gharabaghi, 2004; Konrad and Booth, 2005; Pitt, 1999). Models estimate hypothetical values for development of urban catchments that maintain existing hydrologic boundaries (Elliott and Trowsdale, 2007; Persson et al., 1999; Scholes et al., 2008; Wong et al., 2002, 1999).

The proportion of impervious surface area in a catchment increases with urban development. Impervious surfaces become important sources for sediment and other contaminants, which mobilise during rainfall events (Booth and Bledsoe, 2009; Brabec, 2009; Malaviya and Singh, 2012; Paul and Meyer, 2001; Sansalone and Cristina, 2004). Roads, lawns and buildings are sources for sediment, car residue (brake dust, tire abrasion, exhaust, etc.), lawn fertilizers, de-icing agents and other anthropogenic pollutants (Brabec, 2009; Bradford and Gharabaghi, 2004; Easton et al., 2007; Eriksson et al., 2007; Konrad and Booth, 2005; Paul and Meyer, 2001; Vaze and Chiew, 2004). Multiple studies suggest that fine debris contain nutrients, bacteria and heavy metals which alter the chemical composition of the local streams (Chazarenc and Merlin, 2005; Sansalone and Cristina, 2004; Wong et al., 1999). High concentrations of suspended solids increase water turbidity, degrade water quality and threaten habitats of aquatic species at risk (Booth and Bledsoe, 2009; Brabec, 2009; Lawler et al., 2006). Turbidity impairs visual hunters and blocks sunlight affecting the photosynthetic activity of aquatic vegetation, which can remediate chemical conditions within the water (Eriksson et al., 2007; Lawler et al., 2006;
Malaviya & Singh, 2012; Walsh et al., 2005). Turbid waters result in sedimentation on fish spawning habitats while fine sediment contains nutrients and heavy metals altering the chemical water properties (Eriksson et al., 2007; Paul & Meyer, 2001; Sear et al., 2010; Vaze & Chiew, 2004).

SWM systems are used to reduce the impacts of urban runoff and alleviate effects associated with impervious surfaces generated through development by attenuating flow and improving water quality (Booth and Bledsoe, 2009; Bradford and Gharabaghi, 2004; Burns et al., 2012; Pitt, 1999; Scholes et al., 2008). Low impact development (LID) techniques can minimize the impact urban runoff has at the watershed scale before SWM systems receive runoff. The LID objective is to design development with that maintain flow regimes and water quality objectives of the pre-developed conditions (Huber et al., 2006; Pyke et al., 2011). This is achieve by minimizing impervious surface areas to reduce surface runoff and incorporating LID feature, like the PW in available spaces, such as the channel flood plain(Huber et al., 2006). A pocket wetland (PW) is a stormwater treatment wetland which disperses stormwater runoff over a densely vegetated surface in effort to reduce sediment and nutrients loads through sedimentation and filtration (Davis et al., 2012; Hunt et al., 2011; Li and Davis, 2009; Malaviya and Singh, 2012; Roy-Poirier et al., 2010; Wadzuk et al., 2010). PWs also provide subsurface flow and filtration through porous media below, although surface flow is predominate through rainfall events (Malaviya and Singh, 2012; Van de Moortel et al., 2009). This PW is a modified LID feature incorporated as an end-of-pipe feature, acting as the final stage SWM within the channel flood plain.

Modeling SWM infrastructure is beneficial to planning, design and general understanding on how systems, such as PWs, function. The Unified Stormwater Treatment Model (USTM) is a
first-order kinetic decay algorithm, coupled to an hydraulic efficiency model for SWM treatment infrastructure (Kadlec, 2000; Persson et al., 1999; Wong et al., 2006, 2002). By coupling a kinetic decay algorithm to a hydraulic efficiency model, the user is able to estimate effluent water quality parameters based on input concentrations and hydraulic loading rates (Persson et al., 1999; Wong et al., 2006). The USTM is a model that simplifies the complex relations within the PW. While relying on simple regression equations to represent the function of a PW there remains a degree of variability for input parameters (Rousseau et al., 2004). The USTM has proven to be effective for a variety SWM systems (Wong et al., 2006) and the simplicity of the equations allow this model to be enhance or manipulated for a variety of treatment wetlands (Kadlec and Wallace, 2009; Kadlec, 2000; Rousseau et al., 2004).

This study uses the USTM to provide a reliable estimate of water quality improvements by a PW located at the end of a SWM system. The USTM is applicable to other stormwater management techniques such as gravel filters and vegetated swales (Wong et al., 2006, 2002), but the use of the USTM in this study will focus on the PW. The empirical measurements from a PW and the influent water discharge provided the basis to model the hydrodynamics and water quality variable decay through a PW. The USTM input parameter units suggest this model was design as a tie average model but recent literature has attempted to apply the USTM instantaneously (Wong et al., 2006). The working hypothesis for this study is the temporal application of the USTM will represent the response of effluent TDS from a PW on an event basis.

3.3 Background

The proposed application of the USTM combines a first-order kinetic decay algorithm with a hydraulic efficiency model to provide users with the ability to estimate pollutant concentrations
after passing through a PW treatment facility. The typical hydraulic efficiency of the PW is calculated using Equation 3.1 (Persson et al., 1999),

$$\lambda = \left( \frac{t_p}{t_n} \right)$$

(3.1)

where $\lambda$ is hydraulic efficiency (ratio from 0 to 1), $t_p$ is the mean residence time or time to peak flow and $t_n$ is the time of the nominal residence time distribution (Fig. 3.1). The hydraulic loading is a function of the wetland dimensions and discharge (Eq. 3.2) (Wong et al., 2006, 2002),

$$q = \frac{Q}{3.1536 \times 10^7 (A \lambda)}$$

(3.2)

where $q$ is hydraulic loading (m/year), $Q$ is influent discharge (m$^3$/s), $A$ is the area (m$^2$) and $\lambda$ is the hydraulic efficiency. The hydraulic loading is effectively the volume of water over the effective water area, as it encompasses the efficiency factor. The hydraulic loading is coupled to the decay algorithm (Eq. 3.3) (Wong et al., 2002),

$$\frac{(C_{in} - C^*)}{e^{-k/q}} = (C_{out} - C^*)$$

(3.3)

where $C_{in}$ is the influent concentration for a water quality parameter (mg/L), $C^*$ is the apparent background concentration for a water quality parameter within the PW (mg/L), $q$ is the hydraulic loading rate (m/year) and $k$ is the areal decay rate constant (m/year). The background concentration acts as the hypothetical concentration of a water quality parameter found within the wetland that is resistant to change, is present as sediment, or is a direct groundwater or rainfall input (Kadlec and Wallace, 2009). The USTM has suggested values for different treatment approaches which correspond to specific water quality parameters to estimate $C_{out}$, the effluent water quality parameter concentration from PW.
3.4 Methods

3.4.1 Study Site

The PW was installed as part of a channel re-alignment west of Chinguacousy Rd. and Queen St. W, in Brampton Ontario (Fig. 3.2, 3.3; 43°39’49” N, 79°47’07” W). This PW is the final stage of a stormwater management treatment train system that is responsible for the runoff from a newly developed 1.5 km² residential area (Fig. 3.2, 3.3). The PW is located within the Churchville tributary corridor and receives flow from stormwater pond outlet (SWP) (Fig. 3.2, a.). The stormwater pond provides space for sedimentation but primarily detains flow, while the PW functions as a polishing feature for stormwater quality (3.2, b). The Churchville tributary underwent a 640 m stream re-alignment in 2010, in response to proposed development and the protection of redside dace habitats (Credit Valley Conservation Authority, 2004) where PWs were incorporated within the Churchville tributary floodplain as part of the natural channel design. The Churchville tributary is located within the Churchville subwatershed of the Credit Valley watershed. The channel meanders through poorly drained Chinguacousy series clay loams overlying glacial till from calcareous limestone and shale (Ontario Geological Survey, 2010). The Churchville tributary width is ~3 m wide with bank full depths of 0.3 - 0.6 m and the subwatershed receives an average rainfall of 681 mm per year, as measured at the Toronto Pearson International Airport weather station from 1981 – 2010.

The PW is a densely vegetated shallow basin ~ 900 m², located ~ 5m from the Churchville tributary banks. The permanent pool was about 0.15 m deep, while the design incorporated an underlying porous media (60% rip-rap, 40% soil) that provided infiltration capacity and storage. It was assumed that porous media remained saturated as the PW remained wet through the entire field season. There was an incised channel (~0.20 m wide; ~0.15 m deep) along the berm.
between the PW and the Churchville tributary that developed throughout the field season (Fig. 3.2, c.).

3.4.2 Data Monitoring

Discharge and electrical conductivity were monitored at the SWP outlet while effluent conductivity monitoring took place at upstream and downstream locations from the PW. Onset HOBO water level loggers (range: 0 – 9 m; accuracy: ± 0.5 cm) and conductivity sensors (range: 0 - 1000 µS/cm; accuracy: 5 µS/cm) measured discharge and electrical conductivity at 5-minute interval from Apr. – Oct. 2014. An additional logger was located within the flood plain to measure atmospheric pressure (accuracy: ± 0.05 kPa). In channel conductivity sensors were housed in protective PVC capsules ~ 0.20 m from the bottom of the water column. Loggers located at the SWP outlet were attached to the bottom of the concrete conduit. Influent discharge at the SWP outlet was measured by the time that lapsed to fill a 40 L container. Site visits were performed weekly to gauge discharge and maintain and download data from the loggers. Additional visit took place during rainfall events to measure discharge for event flow conditions.

3.4.2 USTM Application

The USTM was applied using physical PW measurements, influent discharge and water quality to calculate effluent total dissolved solids (TDS). TDS concentrations were calculated from electrical conductivity with a coefficient of 0.67 (Atekwana et al., 2004; Marandi et al., 2013; Pellerin and Wollheim, 2008; Walton, 1989). Observed effluent flow was evaluated using the difference in TDS from upstream to downstream of the PW as it was difficult to pinpoint where the PW drains throughout rainfall events. The channel reach from the upstream to downstream monitoring location was ~ 26 m with minimal runoff input from the opposite bank.
as flow from a PW located on the other side of the Churchville tributary was bypassed downstream of the monitoring locations.

The USTM parameter inputs began with \( t_p \) and \( t_n \) which provide the model with hydraulic efficiency of 0.07 (Eq. 3.1). Electrical conductivity traced the \( t_p \) of the PW, while \( t_n \) was calculated from the wetland dimensions and average influent discharge. The influent discharge and wetland dimensions were used to calculate hydraulic loading rates (Eq. 3.2). The areal decay rate for TDS in the PW was estimated at 9250 m/yr using the average decay of TDS from influent to effluent concentrations (Chapter 2). A conservative background concentration of 1.0 mg/L was used. The final input parameter was the concentration of the water quality variable entering the wetland. The USTM was run temporally with the 5-minute TDS measurements and modelled TDS was evaluated against observed values.

The USTM was applied to influent TDS concentrations to estimate effluent concentrations for two periods during the field season. There was active construction dewatering upstream of the study area June 27 – Sept. 8, which resulted in a disruption in water quality monitoring. An orifice plate located at the SWP outlet was also removed at the beginning of upstream construction that altered the influent discharge to the PW. Therefore results presented in this study from before June 27 reflect the period when the orifice plate was in place, and results after Sept. 8 has no orifice plate. Upstream dewatering from June 27 to Sept. 8 altered water quality parameters so the time analysis during this period was disregarded.

3.5 Results

The PW provided adequate area for the remediation of TDS. There was a drop in TDS from influent to observed effluent TDS concentrations from the PW (Table 3.1). Influent
concentrations were larger (~5 times) before June 27 than those after Sept. 8. This change corresponded to changes in observed effluent, as TDS concentrations before June 27 were 32 ± 26 mg/L and 27 ± 30 mg/L, for base flow and event flow respectively. Observed effluent TDS concentrations were much less after the removal of the orifice plate, for both base flow and event flow conditions Sept. 8 (-5 ± 60 mg/L and 0.3 ± 7 mg/L, respectively). The observed effluent TDS for base flow conditions had a large standard deviation suggesting a release of TDS.

Modelled effluent TDS concentrations were highly influenced by discharge as it is a function of the hydraulic loading for the USTM (Fig. 3.4). The USTM did not respond to noise in the influent TDS concentrations and modelled effluent TDS which corresponded with influent discharge (Fig. 3.4, a., c.). A time series comparing the influent, observed effluent and modelled TDS concentrations (Fig. 3.4, b., d.) shows variability for observed effluent concentrations. Influent discharge at the SWP outlet only responded to rainfall events after Sept. 8 (without the orifice plate), as base flow from the PW was below the detection of the water level logger. There was a range of null data for the modelled effluent TDS as the USTM relied on influent discharge (Fig. 3.4; Table 3.1). The modelled effluent TDS which poorly represented the observed effluent concentrations and high RMSE (Table 3.1). The model reacted well to the initial peak or disturbance in TDS as discharge peaked (Fig. 3.4, b.) but lacked the response to antecedent conditions, or changes in $k$ and $C^*$. Multiple event flows indicate a dilution in observed effluent TDS, while the model modelled a positive increase in TDS concentration. The USTM relied on influent discharge greater than 0.00148 m$^3$/s in order to produce an output above the background concentration of one, suggesting this model does not represent low flow conditions well. Input parameter constants were tested to investigate the influence they had on the models output. The
increase and decrease of the constants ($\lambda$, $A$, $C^*$, and $k$) did not change the USTM response and only resulted in a change of magnitude for modelled effluent TDS.

A study by Credit Valley Conservation (1998) reported a normal TDS range of 150 – 500 mg/L within the West Credit subwatershed, while heavily industrialized areas can exceed TDS levels of 2000 mg/L. Influent TDS remained below the 2000 mg/L level with a maximum TDS of 1765 mg/L and 492 mg/L before June 27 and after Sept. 8, respectively. Average observed effluent TDS remained below the typical range. The USTM modelled a conservative TDS maximum of 105 mg/L and 134 mg/L before June 27 and after Sept. 8, respectively.

3.6 Discussion

The USTM was used to calculate the effluent TDS concentration of PW approaches. Previous studies support the USTM for specific water quality parameters such as total phosphorous, total nitrogen, total suspended solids, and oxygen demand (Brown et al., 2009; Kadlec and Wallace, 2009; Kadlec, 2000; Rousseau et al., 2004; Wong et al., 2006). There have been no previous studies using the USTM for the treatment of TDS. The USTM is better suited for a specific water quality parameters as it requires a decay coefficient ($k$) and uses this decay as a constant (Kadlec and Wallace, 2009; Kadlec, 2000). TDS is comprised of a variety of dissolved nutrients, which have individual decay coefficients. This study attempted to use observed influent and effluent TDS concentrations (Chapter 2) to estimate the apparent decay of TDS. The lack of knowledge and support for the treatment of TDS may have contributed to the poor modelled representation of effluent concentrations.

The USTM focuses on input and output concentrations, simplifying and neglecting the wetland characteristics and climate conditions (Rousseau et al., 2004). The Churchville PW case
study used the regression equations from Persson et al. (1999) and Wong et al. (2006) to test the modelled output against influent TDS concentration ranges of 3388 – 359 mg/L and 492 – 11 mg/L before June 27 and after Sept. 8, respectively. Previous studies suggest ideal water quality concentration ranges for the application of the USTM (Rousseau et al., 2004). The ideal influent range is unknown for the application of the USTM on TDS. This study tested the USTM for a wide range of input TDS concentrations but predominantly responded to influent discharge. Even though the input concentrations after Sept. 8 were lower, the USTM modelled similar effluent concentrations to those before June 27 (Fig. 3.4).

A tool such as the USTM is useful to designers and planners as it helps with decisions on the most efficient SWM design for specific spatial and hydrological conditions. Urban designs for development or stream restoration projects must work within the hydrological restraints defined by existing infrastructure (Brown et al., 2009; Elliott and Trowsdale, 2007; Persson et al., 1999; Scholes et al., 2008). If there is a limited space provided for SWM or a natural channel design, the USTM provides accurate estimates for system performance for specific size restraints and water quality parameters (Wong et al., 2006). If monitoring suggests that water quality parameter concentrations are high and there is not adequate space for a PW then other SWM systems must be considered.

3.7 Conclusion

The USTM did not effectively predict water quality parameters for the Churchville PW. The effluent modelled TDS concentrations produced by the USTM were primarily driven by influent discharge values. Because the model responded predominantly to influent discharge, the USTM did not work for base flow conditions. The USTM consistently peaked at ~ 120 mg/L although influent TDS concentration ranges were 5 times greater before June 27 (with orifice plate in
place) than after Sept. 8. The USTM performed poorly over the temporal application. It does not adjust to changes in antecedence and influent TDS. The USTM would benefit from future studies with more samples taken throughout the length of the PW to determine if there was a decreasing concentration trend to support the model’s decay algorithm and input parameterization. A tool such as the USTM will provide urban system planners and consultants with the means to test efficiency of stormwater management infrastructure before finishing designs. Urban restoration projects often consist of spatial restraints and the USTM can test whether water quality may benefit from features such as PWs.
Figure 3.1: The illustration for the application of a tracer concentration time distribution (modified from Persson et al., 1999). This study used EC to trace the mean detention time of 2.3 hrs (Chapter 2; Table 2.2).
Figure 3.2: The SWP outlet structure (a.), the ~900 m² Churchville PW (b.), the incised channel which flows into the Churchville tributary (c.) and their relative locations to the PW.
Figure 3.3: Influent and effluent discharge and water quality was monitored at the PW in the Churchville subwatershed from Apr. – Oct. 2014. This ~900 m$^2$ PW is part of a stormwater treatment train managing a ~0.15 km$^2$ area of residential runoff.
Table 3.1: Average influent, observed effluent, and modelled effluent TDS concentrations.

<table>
<thead>
<tr>
<th>Water Quality Parameter</th>
<th>Influent Flow</th>
<th>Observed Effluent Flow</th>
<th>Modelled Effluent Flow</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Before June 27</td>
<td>After Sept. 8</td>
<td></td>
</tr>
<tr>
<td></td>
<td>After Sept. 8</td>
<td>Before June 27</td>
<td>After Sept. 8</td>
</tr>
<tr>
<td><strong>Average base flow conditions</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>TDS (mg/L)</td>
<td>1419 ± 247</td>
<td>297 ± 57</td>
<td>32 ± 26</td>
</tr>
<tr>
<td><strong>Average event flow conditions</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>TDS (mg/L)</td>
<td>1211 ± 229</td>
<td>263 ± 73</td>
<td>27 ± 30</td>
</tr>
<tr>
<td></td>
<td></td>
<td>RMSE = 44.3</td>
<td></td>
</tr>
</tbody>
</table>
Figure 3.4: Influent discharge (a, c) and influent, observed effluent and modelled effluent TDS concentrations (b, d) from the Churchville PW.
Chapter 4.0 Conclusion

The primary objectives in this study were to evaluate the effective flow attenuation, and the remediation of water quality parameters from a PW. This data was also used for the validation of a SWM model, the USTM. Monitoring took place through predominantly summer and fall rainfall events from April – October 2014. Results demonstrate that urban stormwater runoff routed through a SWM system that ended in a PW had minimal influence on adjacent stream conditions. Monitoring data was also used to validate the USTM for the treatment of stormwater quality parameters. Input water quality parameters were low for the size of the PW, but the USTM effectively estimated output concentrations entering the stream. Below is a summary of the main findings from this thesis:

- The PW attenuated flow with an average residence time of 2.2 hours. Antecedence was the only limiting factor for residency, as the PWs basin has a limited storage capacity.
- The STT flowing into Churchville tributary resulted in low runoff ratios compared to those coming into the PW. This change was measured for runoff within the channel, after flowing through all SWM systems for the Churchville tributary.
- The change in runoff volume after August 1 suggests PWs are natural and may adjust to hydrologic conditions. Although the incoming hydrology change when the orifice plate was removed, output water quality parameters did not change even though there was a small increase in effluent water quantity. A PW will adapt to changes in rainfall intensity but remain efficient at water quality remediation.
- There was no measurable change in water temperatures due to effluent flow from the PW.
- The vegetation and subsurface flow through the PW effectively cool urban runoff before it enters the stream.
- Effluent TDS was lower and in some instances, TDS concentrations dropped from up to downstream due to dilution from incoming water.
- The TSS flux responded with an increase during rainfall events for the entire stream and PW. It depended on the magnitude of the rainfall event, as small rainfall events would flush the PW and not provide enough water to dilute TSS concentrations.
- The effluent TDS concentrations produced by the USTM corresponded poorly with the observed values. The model is dominated by the influent discharge and does not deal well with antecedence but may be better suited to time average predictions.

The monitoring results suggest the PW is performing effectively. The residency time of the PW in the current state is low. The PW would benefit from a longer residency period, roughly 10 – 15 hrs, for a 30 min rainfall event, to improve water quality further (Shutes et al., 1999). The incised channel from the Churchville PW reduced residency as it allowed surface water to drain continuously. The change in the depth and width of incised channel show how the PW is a dynamic system. It was a response to the changes in hydrologic inputs and it shows evidence of the PW expanding to an inline wetland. This expansion may increase residence time through due to an increase in the basin area. Long term monitoring of this system would indicate whether this expansion is a failure of the system or expansion would improve productivity.

An increase in the study area within the Churchville subwatershed would provide a better understanding on how the entire stream restoration is performing. The site is full of other end-of-pipe wetlands and inline features within the Churchville tributary realignment. An assessment on the entire systems would indicate the benefits stream restoration has on discharge and water quality. This study would also benefit from a control study for performance comparison. As the Churchville PW was in a ‘natural setting’, there was a lack of control for changed on the input
of water quality parameters. A control study would provide the means to provide known influent concentrations and collect all of the effluent water with no other source of dilution. Effluent flow from the Churchville PW was assessed under the assumptions that the change in stream conditions from upstream to downstream accurately reflects effluent conditions from the PW. Although this effectively measure the influence the effluent flow conditions had on the Churchville tributary, the true performance of the PW system is misled due to the mixing of upstream water quality inputs and the potential subsurface inputs between the upstream and downstream locations.

This study attempted the use of the USTM to predict effluent the response of water quality parameters. The USTM lacked precision for small influent concentrations. Much of this application was based on average input measured during the field season. An in depth study on the hydraulics through the PW and the exponential decay of water quality parameters along the length of the PW would provide the essential parameters to improve the performance of this application of the USTM. The USTM would also benefit from input parameterization and the addition of parameters to improve the models performance. An additional parameter to adjust the effective area, would deal with changes in water level and antecedence. This model is an important tool for consultants, engineers and designers for the proper applications of efficient SWM systems. Urban hydrological systems are often spatial restricted by infrastructure but this paper provide proof that PW within the flood plain will improve stormwater quality and the USTM is a useful tool in the design and decision making process.
References


Credit Valley Conservation Authority, 2004. Credit Valley Subwatershed Study: Huttonville Creek (7), Springbrook Creek (8a), Churchville Tributary (8b), 190 pages.


