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How has it changed? A comparative field evaluation of bioretention infiltration and treatment performance post-construction and at maturity

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ABSTRACT

A conventionally-designed bioretention cell, consisting of very sandy media meant to achieve high infiltration rates, was studied for its hydrologic and treatment performance over two monitoring periods: 2013–2014 (immediately post-construction) and 2017–2018 (4–5 years post-construction). Given the limited literature on mature performance of bioretention, the purpose of this study was to determine how the effluent from the bioretention cell and the hydrology changed over 5 years. The hydrologic performance was maintained 5 years post-construction, with median volume reductions of 100% in both monitoring periods, despite an under-utilization of the soil volume. Using censored data analysis and a 95% confidence level, the results revealed that the effluent water quality in 2017–2018 was improved compared to 2013–2014 for some parameters, e.g., dissolved solids and nitrogen species, and was maintained for phosphorus, metals, and suspended solids. These results suggest that for a reliable assessment of bioretention cell treatment performance, it is recommended to wait for soil and plant establishment, e.g., 2 years after construction. Between the inlet and outlet of the bioretention cell (2017–2018 data only), concentration decreased for nitrogen species and suspended solids, but did not significantly increase or decrease for alkalinity, hardness, dissolved solids, phosphorus and metals. Mass removal of all contaminants was very high, largely due to high volume reductions. Despite sustained hydrologic performance up to 5 years post-construction, there is a need for targeted bioretention design for enhanced treatment performance of dissolved contaminants.

1. Introduction

Stormwater management has evolved over the past two decades to meet the needs of flood protection and reduce environmental damage by responsibly managing flow rates and volumes in urban environments. Engineers began to shift the focus of stormwater management to mimicking pre-development hydrology, with technologies such as permeable pavement, infiltration chambers/trenches, bioretention cells, and green roofs. Bioretention cells are vegetated infiltration technologies made of a depression in the ground where the natural soil is replaced with high-infiltration media. The soil media is covered with mulch or other organic material and may have a gravel reservoir and underdrain.

Bioretention research is dominated by field studies on newly built systems, with 72% of field studies on systems < 2 years post-construction (Spraakman&Rodgers et al., 2020), which may be poor indicators of lifetime performance of bioretention. When Liu et al. (2014) compiled results for several newly built bioretention systems, the results showed significant variability in performance. For example, the authors showed that concentration decreases across bioretention cells ranged from −3 to 99% for total nitrogen (TN) and − 10 and 99% for total phosphorus (TP) (Liu et al., 2014). Poor bioretention cell performance has been attributed to media composition (Hunt et al., 2006) and/or system configuration/design (Li and Davis, 2009).

Compared to research on newly built systems, studies examining mature bioretention systems are scarce but tend to agree on an improvement in performance over time (Asleson et al., 2009; Komlos and Traver, 2012; Willard et al., 2017). Several studies on mature systems focused on uptake of contaminants within the soil media, with results showing accumulation in the top 20 cm of the media of fine particles (Jenkins et al., 2010), phosphate (Komlos and Traver, 2012), and metals (Kluge et al., 2018). Large surveys of multiple older systems have been conducted as well, mainly summarizing capacity for infiltration and soil media properties (Asleson et al., 2009; Kluge et al., 2018).

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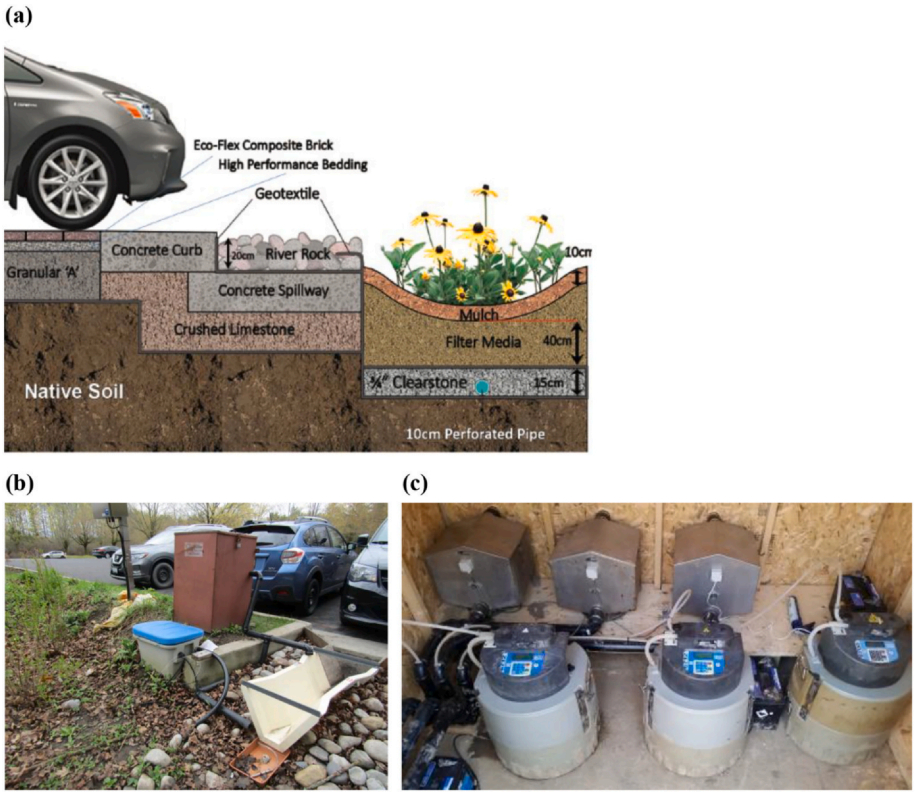

Fig. 1. Site schematic and monitoring equipment. (a) Schematic of bioretention cell (obtained with permission from TRCA STEP (2015)). (b) Inlet monitoring for 2017–2018 monitoring period. (c) Outlet monitoring for both monitoring periods.

Full water quality and water quantity field studies on mature systems are still quite rare. Willard et al. (2017) presented a comparative analysis of water quantity and water quality data seven years post-construction and immediately after construction, and found a median volume reduction of 100% in both monitoring periods, and median mass reductions of TSS, TP and TN above 90% in both periods. Kandel et al. (2017) studied water quality and quantity on a bioretention cell 7-years post-construction, but did not have comparison data collected immediately after construction. Johnson and Hunt (2019) presented data on mass and concentration of TN and TP in bioretention cell effluent 17 years post-construction and 14 months post-construction. The authors reported concentration reductions only in the mature system, at 26% and 39% concentration reduction for TN and TP, respectively, compared to the 14 month old bioretention cell, which had increases in concentration of 38% and 21% for TN and TP, respectively (Johnson and Hunt, 2016). Lucke and Nichols (2015) dosed several mature cells (10 years post-construction) in Australia with synthetic stormwater, and found excellent water quantity reductions, i.e. peak flow reductions above 80%, but very variable reductions in nutrient concentrations, with net exports of nutrients with low simulated runoff dosage and reduction of nutrients when using high dosages.

Bioretention cells are constructed globally at increased rates. To develop robust bioretention cell designs, engineers and practitioners must therefore understand the performance expected from these systems and the needs for maintenance or rehabilitation when performance degrades. To address this engineering challenge, full hydrologic and water quality studies need to be completed on mature systems.

The goal of this study was to compare the infiltration and treatment performance of a conventionally designed bioretention cell 4-years post-construction to its initial conditions. This study consisted of a full field-scale hydrologic and water quality analysis during the monitoring periods of 2013–2014 (immediately post-construction) and 2017–2018, using data from 40 sampling events and a comprehensive list of water quality parameters, including 25 metals and 20 polyaromatic hydrocarbons.

2. Materials and methods

2.1. Site description

The bioretention cell studied is located at the Kortright Center for Conservation in Vaughan, ON, Canada (43° 49′ 51.95“ N 79° 35’ 28.15” W). It receives runoff from a 265-m^2 impermeable parking lot made of rubber interlocking pavers, fabricated from recycled tires (Eco-Flex Churchill, Sturgeon County, AB, Canada). The bioretention cell was constructed as a demonstration site by the Toronto and Region Conservation Authority's Sustainable Technologies Evaluation Program (TRCA STEP) in November 2012 (Supporting Information (SI) Fig. S1). The cell was designed for a surface area of 30 m^2 and a 7.5-cm layer of hardwood mulch above a 40-cm deep layer of media (TRCA STEP 2015). The bioretention cell has an underdrain of 10-cm diameter perforated pipes placed on the native soil surface surrounded by 15-cm deep clear stone (20 mm in diameter) wrapped in a permeable geotextile (270R, Terrafix Geosynthetics, Toronto, ON, Canada). The inlet of the bioretention cell was a concrete spillway filled with rock, 20-cm deep covering a 2-m^2 surface area. The vegetation within the cell consisted of wildflowers, grasses and ground covers, including: *Rudbeckia hirta, Echinacea purpurea, Panicum virgatum, Eupatorium purpureum*. In 2013, a sample from the filter media had a composition of 62% sand and 38% silt and clay. In 2017, the media composition of the top 20 cm was 93–99% sand and 1–7% silt and clay (Gnanaraj, 2018; Rhodes-Dicker and Passeport, 2019). A prior study at this site described the native soils below the cell as clayey silt till, with typical soils in this area as glacial till soils, interspersed with cobbles, boulders and gravel inclusions (Drake et al., 2014).

The bioretention cell was surveyed using a Leica Geosystems

Multistation (MS60) in May 2017, when the vegetation was not fully developed and did not significantly obstruct surface points. Contours from the survey and areas of each contour were constructed in AutoCAD Civil 3D, and the volume of surface storage was calculated using the mean area method. The site survey revealed that the cell surface area was 33.2 m^2, close to the design surface area. The ratio of drainage area to bioretention area was 8:1, which is within the recommended range of 5:1 to 15:1 (CVC and TRCA 2010). A schematic of the bioretention cell and the monitoring setup is shown in Fig. 1.

2.2. Hydrologic monitoring

Monitoring was conducted in 2013–2014 and 2017–2018. Rainfall data was recorded every 5 min with an accuracy of ± 2% at a tipping bucket (TB3 0.2 mm, Hydrological Services, Lake Worth FL) located 500 m away from the site at TRCA weather station HY039 (Vaughan, ON, Canada).

The inlet flow in 2013–2014 was determined by monitoring an adjacent asphalt parking lot area with a single catchbasin draining to a 3-L tipping bucket (V2A Tipping Counter, Geneq Inc., Montreal QC) in an underground vault. The inlet flow in 2013–2014 was calculated in L/min as $Q_{out} = 3 \times tip\ count$, where the tip count was the number of tips per minute, recorded every minute.

The inlet of the bioretention cell was reconfigured in May 2017 to enable continuous monitoring of the flow and the collection of water samples during flow events. The inlet rock spillway was replaced with an extra-large 60°-V Parshall flume (Virtual Polymer Compounds LLC, Medina NY) fitted with a bubbler flow module (730 Bubbler Module, Teledyne ISCO, Lincoln NE) that measured water level every minute. The inlet flow rate (Q_{in}) was calculated in litres per minute (L/min) as $Q_{in} = 60.08\ H^{2.63}$, where H is the water level in the flume in meters. Because the flume was 30-cm tall, any values reported as above 30 cm by the bubbler flow module were excluded from the data set. In addition, water level values below 5 mm could not be measured accurately and were therefore eliminated as well.

During construction in 2012, the perforated underdrain pipe in the bioretention cell (hereafter referred to as the outlet) was directed to an HDPE pipe under the parking lot, then to a monitoring hut, which contained a calibrated 3-L tipping bucket (V2A Tipping Counter, Geneq Inc., Montreal QC). For both monitoring periods, the outlet flow rate was calculated in L/min as $Q_{out} = 3 \times tip\ count$, where the tip count was the number of tips per minute, recorded every minute. Monitoring experience from 2013 to 2014 showed that the flow from the perforated underdrains occasionally exceeded the capacity of the tipping bucket (60 L/min). As a result, the flow was throttled by partially closing a valve on the outlet pipe. During the 2017 and 2018 monitoring seasons, and unless otherwise stated, the valve was approximately 80% closed. Monitoring equipment photos are shown in Fig. 1 (b) and (c).

2.3. Water quality monitoring

Automated samplers (6712 Portable Sampler, Teledyne ISCO, Lincoln NE) were used to collect water samples. Over each 2-year long monitoring period, i.e. 6 summer months May – October each year, rain events led to either inlet and/or outlet samples that were analyzed for a range of water quality parameters such as alkalinity, conductivity, pH, nutrients, solids, metals, and polycyclic aromatic hydrocarbons. In 2013–2014, 22 outlet samples were collected. Due to the difference in monitoring methodology used in 2013–2014, inlet sample results from 2013 to 2014 are not used, and thus all inlet vs. outlet comparisons made below are for 2017–2018 only. In 2017–2018, 27 events were monitored, 13 of which had pairs of inlet and outlet samples.

In 2017–2018, the sampling intake at the inlet was done with a 3/8″ stainless steel ends with polypropylene center that was placed inside a plastic container at the outlet of the flume to allow water to pond above the sampling intake. The outlet sampling, taken from the end of the bioretention cell's underdrain, was unchanged between 2013 and 2014 and 2017–2018. The outlet monitoring equipment was located inside of a small hut. The outlet intake by a vinyl suction line was located immediately downstream of the tipping bucket, inside the U-bend of an HDPE pipe.

Outside of the monitoring season, i.e., from November to the following May, the automatic samplers were disconnected and stored indoors, the inlet was covered to prevent damage from snow equipment and vehicles, and the outlet flow was discharged directly to a low area in the forest. During the monitoring period, the site was visited weekly and after every runoff event to reset the autosamplers, collect autosampler data, clean the inlet of debris and check the functioning of the monitoring equipment.

The automatic samplers were installed to collect up to 24 time-dependent samples during each runoff event. The time-dependent samples were then combined by hand to make a flow-weighted composite sample for each of the inlet and the outlet. The samples were kept at 4 °C before being sent for analysis to the Ontario Ministry of the Environment, Conservation and Parks Laboratory Services Branch. The samples were analyzed for a range of traditional water quality parameters, including alkalinity, hardness, conductivity, chloride concentration, and pH, as well as for solids, nutrients, total metals, and polycyclic aromatic hydrocarbons (PAHs). The concentrations thus obtained are event-mean concentrations thereafter simply referred to as concentrations. Details on the compounds tested, analytical methods, and method detection limits (MDL) are presented in SI Table S1.

2.4. Data analyses

An independent flow event was defined as any event that began more than 3 h after the outflow from the previous event stopped. The volumes (V, in L) of water that entered (V_{in}) and exited (V_{out}) the bioretention cell were calculated from the flow rate data (Q_i, in L/min) and event time durations (t_i, in min). A peak flow rate (Q_p, in L/min) was defined as the maximum 1-min flow value observed during a flow event at the inlet ($Q_{p,in}$) and outlet ($Q_{p,out}$).

The volume during an independent flow event was calculated using Eq. (1). For each storm event, the volume reduced was calculated as the difference between the inlet and outlet volumes, the percent volume reduction was calculated by Eq. (2), the peak flow reduction was calculated by Eq. (3), the volume discharge ratio (VDR) was calculated by Eq. (4), and the lag time was defined as the difference between the start of the inflow and the start of the outflow.

$$V = \sum_{i=1}^{i=n} Q_i \times t_i \tag{1}$$

$$\frac{V_{in} - V_{out}}{V_{in}} \times 100 \tag{2}$$

$$\frac{Q_{p,in} - Q_{p,out}}{Q_{p,in}} \times 100 \tag{3}$$

$$\mathrm{VDR} = \frac{V_{out}}{V_{in}} \tag{4}$$

Davis et al. (2012) defined the performance metric of bioretention abstraction volume as the volume of influent runoff that is not returned to surface water, but is instead infiltrated or evapotranspired. This volume is based on porosity and soil moisture conditions, and therefore varies by storm event. The equations for average and low bioretention abstraction volumes (BAV) for underdrained bioretention cells from Davis et al. (2012) are as follows:

$$\text{Average BAV} = \mathrm{RZMS}\,(\mathrm{SAT} - \mathrm{WP}) + \mathrm{LMS}(\mathrm{SAT} - \mathrm{FC}) \tag{5}$$

$$\text{Low BAV} = \mathrm{RZMS}\,(\mathrm{SAT} - \mathrm{WP}) \tag{6}$$

where RZMS is the root zone media storage or the depth of media

(0.4 m depth by surface area, 13.27 m^3), LMS is the lower media storage (15 cm by surface area, 4.98 m^3), and SAT, WP and FC are the soil water contents at saturation, wilting point, and field capacity, respectively. The bioretention area and ponding volume were determined during the cell survey. SAT, WP and FC were 46%, 19% and 40%, respectively, determined using an evaporation experiment (UMS). The theoretical average and low bioretention abstraction volumes were 3.84 and 3.56 m^3, respectively.

For each water quality parameter except pH and conductivity, the volumes of water that passed at the inlet and outlet of the bioretention cell during the sampling period were multiplied by the corresponding event mean concentration to calculate the mass of each compound. When the concentration was below the detection limit for the laboratory, the value was flagged as censored (Helsel, 2012). As all contaminant mass data were not normally distributed, a non-parametric method for reporting statistics was chosen, thus the median and range are reported as median (interquartile range), e.g., 148 (120) µg/L.

Contaminant reduction was evaluated via percent concentration reduction (or "efficiency ratio"), percent mass reduction (or "summation of loads") and statistical difference between cumulative probability distributions of inlet and outlet data, as per recommendations presented in Geosyntec Consultants and Wright Water Engineers Inc. (2009). C_{in}, C_{out}, m_{in} and m_{out} represent median concentrations at inlet and outlet, and total masses ($V \times C$ for all events) at the inlet and outlet, respectively, for each contaminant quantified. All contaminant reduction data was not normally distributed (as determined by (i) histogram visualization, (ii) qqplot visualization, and (iii) Shapiro-Wilk test), and thus median values were used (Geosyntec Consultants and Wright Water Engineers Inc., 2009). The contaminant reduction equations were as follows:

$$\text{Percent Concentration Reduction} = \frac{C_{in} - C_{out}}{C_{in}} \times 100 \quad (7)$$

$$\text{Percent Mass Reduction} = \frac{m_{in} - m_{out}}{m_{in}} \times 100 \quad (8)$$

When both the inlet and outlet concentrations were below the MDL, concentration reduction and mass reduction were not calculated. Analyses and graphical representations (boxplots, x-y plots, cumulative distribution functions) of results for each contaminant were generated in R (R Core Team, 2017), using the following packages: graphics, dplyr (Wickham et al., 2019) and NADA (Lee, 2020).

Concentration and mass reductions were calculated for 2017–2018 only. Inlet data for 2013–2014 was not used in this analysis because the 2017–2018 drainage area pavement, composed of rubber pavers was different from that of 2017–2018, made of asphalt. To compare the two monitoring periods, the outlet concentrations were examined for statistical differences. Potential statistical differences for the inlet and outlet volumes, flow durations and peak flows were determined using the Wilcoxon Rank-Sum test. Potential statistical differences for inlet and outlet concentrations of each water quality parameter were tested using the Peto & Peto modification of the Gehan-Wilcoxon test (Helsel, 2012). Summary descriptive statistics and differences between inlet and outlet or between monitoring periods for water quality data containing censored values was done using the Regression on Statistics (ROS) method (Helsel, 2012). A confidence level $\alpha = 0.05$ was applied for all statistical analyses.

3. Results

An overview of the observed rainfall and runoff and completed sampling during the two monitoring periods (2013–2014 and 2017–2018) is shown in Fig. 2. Additional information on the total number and total reliable runoff events monitored in this study is presented in SI Table S2.

3.1. Rainfall

Rainfall statistics for monitored events are summarized in Table 1. The rainfall depth and intensity in 2013–2014 and 2017–2018 exhibited a low statistical difference (p-values of 0.045) and the number of antecedent dry days was similar between the two monitoring periods, indicating that data from the two monitoring periods are similar. The measured rainfall data was compared to long-term averages from (i) climate normals, i.e. three-decade averages, obtained from Environment Canada Woodbridge monitoring station (located 5.3 km from the site) for 1981–2010 and (ii) on-site averages, available for 2010–2018 (see Fig. S2). The climate normals show a total rainfall for the monitoring period of 457 mm. The 2013 and 2014 monitoring period had more precipitation, with totals of 615 and 507 mm, respectively. July 2013 was the wettest July of the four years, in part due to a major storm event on July 8, 2013 that caused flooding in nearby urban areas. In 2017, rainfall depths were similar to climate normals, with a total of 446 mm, but with a particularly wet spring whereas, 2018 was drier, at 401 mm. The rainfall depths in 2017 had large variability with wet May and June and drier August and September than climate normals. The 2018 May and June were drier than the long-term averages.

3.2. Hydrology

Volumes, durations, and peak flows were measured at the inlet and outlet of the bioretention cell, and hydrologic performance was characterized using the following parameters: portion of events that were completely retained, cumulative inflow and outflow depths, volume reduction, peak flow reduction, lag time, bioretention abstraction volume, and volume discharge ratio. Volume reduction and bioretention abstraction volume were examined for all monitored events and for the design storm event. From 2013 to 2014 to 2017–2018, the outlet volume statistically increased (p-value $< 10^{-6}$), the percent volume reduction decreased (p-value $< 10^{-6}$), and the volume discharge ratio increased (p-value $< 10^{-6}$). Conversely, the inlet volume and bioretention abstraction volumes of the two monitoring periods were not statistically different (p-value = 0.19 and 0.21, respectively). Due to the differences in flow monitoring methodology, results for duration, peak flow or lag time were not available for 2013–2014. All water quantity descriptive statistics are presented in Table 2 and SI Tables S3 and S4, and boxplots are available in SI Fig. S3.

The majority of rainfall events, i.e., 84% in 2013–2014 and 51% in 2017–2018, were retained by the bioretention cell, i.e. no outflow was produced during a single event. Total outflow volume accounted for only 5% and 8% of total inflow volumes in 2013–2014 and 2017–2018, respectively (see Table 2).

The volumes exhibited a statistically significant decrease between the bioretention cell inlet and outlet in 2017–2018 with p-values $< 10^{-13}$ (Table 3 and SI Table S3). In 2017–2018, the event durations and peak flows were also significantly reduced between the inlet and the outlet (p-values $<10^{-12}$). The lag time, or the time between the beginning of flow at the inlet and the beginning of flow at the outlet, had a median of 0.6 (2.9) h. Note that the lag time was only measured for 36 events, as the other 38 events did not produce any outflow.

Though conventionally-designed bioretention cells often have a retention volume target for the 25-mm event (Aquafor Beech Ltd., 2016), this specific bioretention cell was installed in an existing parking lot with limited area, and was therefore not designed to retain the entire 25-mm event. The bioretention cell survey indicated that the bioretention cell should be able to contain events less than or equal to 15 mm. According to the monitoring data, 93% of events less than 15 mm in 2013–2014 and 59% in 2017–2018 were completely retained. The volume reduction for < 15-mm events was a median of 100 (0)% in 2013–2014, and 100 (9)% in 2017–2018.

With the majority of events being retained in the bioretention cell,

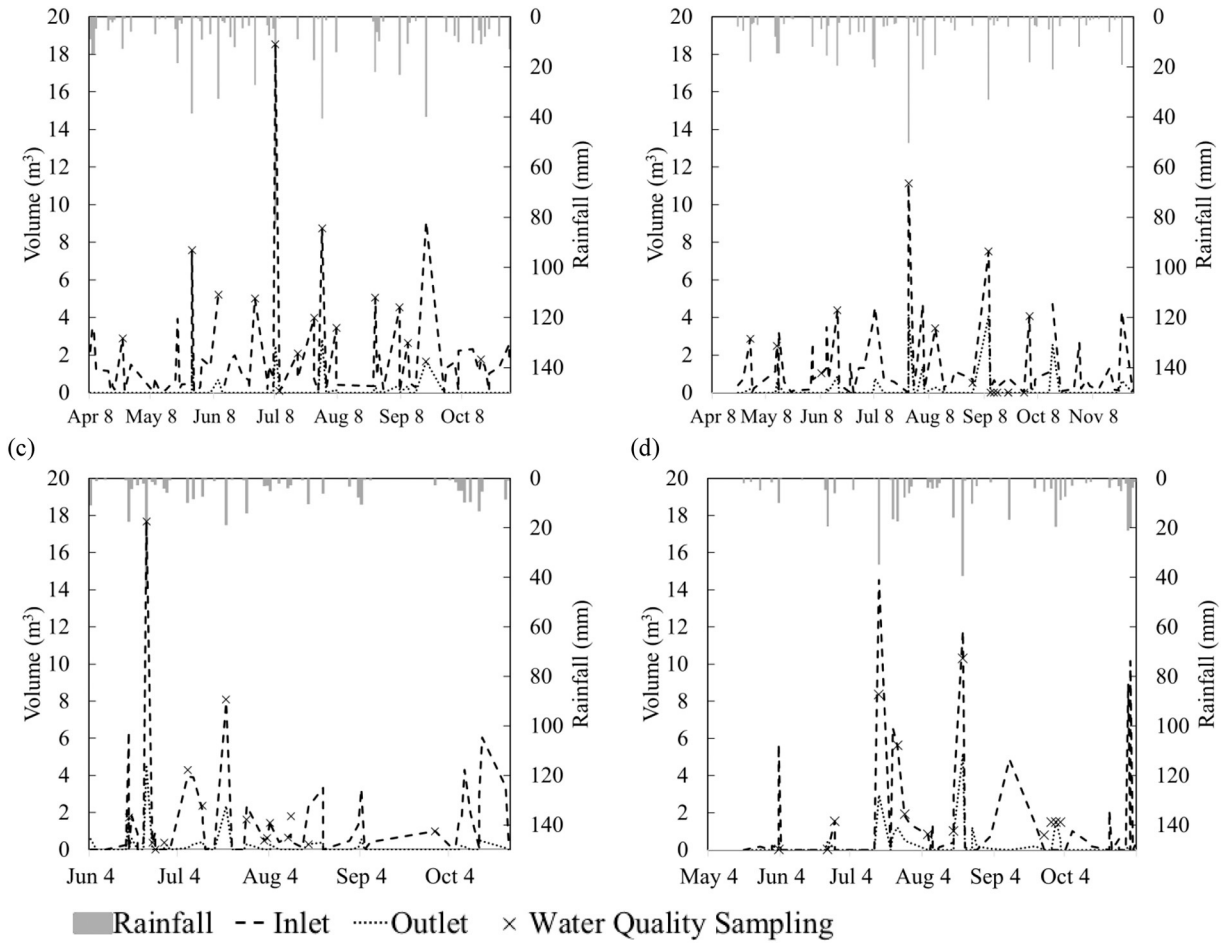

Fig. 2. Rainfall events, inlet and outlet volumes, and sampling events for 2013 (a), 2014 (b), 2017 (c) and 2018 (d).

Table 1
Rainfall statistics for the April–October monitoring periods.

Parameter	Min[a]	Mean	Median	IQR[b]	Max[c]
2013–2014					
Rainfall depth (mm)	0.60	7.6	3.6	7.6	83
Peak 5-min rainfall intensity (mm/h)	1.2	17	9.6	14.4	139
Antecedent dry days (d)	0.15	2.6	1.6	3.4	18
2017–2018					
Rainfall depth (mm)	0.60	7.0	4.0	7.4	44
Peak 5-min rainfall intensity (mm/h)	0	16	9.6	13.2	62
Antecedent dry days (d)	0.13	2.7	1.8	2.9	22

[a] Min: minimum.
[b] IQR: interquartile range.
[c] Max: maximum.

this cell exceeds the performance criteria for volume discharge ratio. The volume discharge ratio for 2013–2014 and 2017–2018 are shown in Table 2 and SI Table S4. The exceedance probability chart in Fig. 3 shows that there was only a 15% chance of producing any outflow (VDR > 0) in 2013–2014, and a 48% chance of producing any outflow in 2017–2018, showing agreement with the significant increase of VDR between monitoring periods (p-value $< 10^{-6}$). Despite the increase of VDR between monitoring periods, the suggested threshold of bioretention efficacy of VDR < 0.33 (Davis, 2008) had less than a 3% chance of being exceeded in both monitoring periods.

3.3. Water quality

The water quality results are presented as inlet-outlet concentration and mass comparisons in 2017–2018, outlet concentration comparisons between the two monitoring periods (2013–2014 and 2017–2018), and outlet concentration comparisons to target water quality concentrations in local regulations (Fig. 4). Overall, mass reductions were very high for all compounds analyzed, due to the high volume reduction. Descriptive statistics, boxplots and probability plots for all water quality parameters are shown in SI, Tables S5-S7, and Figs. S4-S9. Inlet vs. outlet plots for only paired events in 2017–2018 are shown in SI Fig. S10.

3.3.1. Water chemistry

The water quality values for alkalinity, conductivity, hardness and pH exhibited a large statistically significant increase at the outlet compared to the inlet (p-values < 0.05, SI Table S6). The pH increase (< 1 pH-unit) was within guidelines and therefore not likely to produce significant effluent toxicity due to changes in metal speciation. There are no water quality guidelines for alkalinity and hardness (see Table S8). Chloride concentrations were not significantly different (p = 0.1) between the inlet at 1.8 (0.9) mg/L and the outlet at 1.5 (0.4) mg/L, and the outlet concentrations were independent of the inlet concentrations (as shown in the inlet-outlet plots in SI Fig. S10), indicating no reduction in chloride concentration due to the bioretention cell. Chloride concentrations were always much lower than the Canadian guidelines of 120 mg/L. The mass reduction for alkalinity, chloride and hardness was 60% or greater.

Outlet alkalinity, conductivity, chloride concentration, and hardness were significantly lower (p-value $< 10^{-4}$ except for chloride at 0.001) in 2017–2018 than in 2013–2014; whereas, the total mass of these parameters over the whole monitoring season remained very low in both years, < 4 g each, due to the large volume reductions. The

Table 2
Hydrologic Performance Results and Summary Statistics.

Parameter	2013–2014			2017–2018			p-value, between monitoring periods [b]
	Mean	Median	IQR[a]	Mean	Median	IQR[a]	
Volume at inlet (m^3)	1.80	0.82	1.82	2.40	1.32	2.60	0.18
Volume at outlet (m^3)	0.18	0.00	0.00	0.36	0.00	0.38	2.05E-07
Event duration at inlet (h)	–	–	–	4.26	2.47	3.96	NA
Event duration at outlet (h)	–	–	–	1.17	1.17	1.40	NA
Peak flow rate at inlet (m^3/min)	–	–	–	0.152	0.049	1.614	NA
Peak flow rate at outlet (m^3/min)	–	–	–	0.01	0.002	0.015	NA
Calculated Parameters							
Volume reduction (%)	98	100	0	93	100	12	2.61E-07
Peak flow reduction (%)	–	–	–	95	100	8	NA
Lag time (h)	–	–	–	2	0.6	2.9	NA
Bioretention abstraction volume (m^3)	1.61	0.82	1.80	2.06	1.04	1.97	0.21
Volume discharge ratio	0.03	0	0	0.07	0.00	0.12	2.24E-07
Probability to produce an outflow (VDR [c] > 0) (%)	15			48			NA
Probability to produce VDR > 0.33 (%)	1.9			2.7			NA
For events less than 15 mm (depth of theoretical average bioretention abstraction volume)							
Volume reduction (%)	100	100	0	95	100	9	NA

Notes: (a) IQR = interquartile range; (b) P-value is calculated using the Wilcoxon rank sum test, comparing the results from the two monitoring periods; (c) Volume discharge ratio.

median chloride concentration in 2013–2014 was 24.0 (5.7) mg/L, with the three largest values, at 27.4 mg/L, 33.1 mg/L, and 83.0 mg/L measured in April and May. The median outlet chloride concentration in 2017–2018 was much lower, at 1.5 (0.4) mg/L, likely because no data was available in April and May when snowmelt-induced spikes in chloride concentrations might have occurred. The bioretention cell effluent pH value did not change significantly (p = 0.25) between the two monitoring periods.

3.3.2. Solids

The bioretention cell effectively trapped particles, with total suspended solids (TSS) concentration decreasing significantly at the outlet (p-value $<10^{-3}$), with a 63% concentration decrease across the cell. Total dissolved solids (TDS) concentration, on the other hand, increased significantly, by 292% (p-value $< 10^{-5}$) and overall total solids (TS) increased significantly, by 145% (p-value $< 10^{-4}$) at the outlet. Despite fairly consistent concentrations of solids at the outlet, the mass reduction was quite high. The outlet concentrations of TSS were similar in the two monitoring periods (p-value = 0.2); whereas, TDS and TS concentrations decreased significantly between 2013 and 2014 and 2017–2018 (p-value $< 10^{-6}$) (Fig. 4).

3.3.3. Nutrients

Phosphate and total phosphorus concentrations did not significantly change between the inlet and the outlet (p-value 0.3 and 0.2, respectively) in 2017/18. Phosphate accounted for 80% of TP in both the inlet and the outlet. Total phosphorus concentration water quality targets in Canada depend on the trophic status of the receiving freshwater environment, and range from < 4 to > 100 µg/L. These recommendations were exceeded in most inlet and outlet samples. Mass reductions of phosphate and total phosphorus were 93% and 92%, respectively.

Concentrations in nitrogen species were mostly reduced between the inlet and the outlet. Total nitrogen, nitrite, and ammonia/ammonium had concentration decreases of 24, 80, and 56%, respectively, with statistically significant differences between the inlet and the outlet (all p-values $< 10^{-3}$). Concentrations in nitrate + nitrite was not reduced significantly (p-value = 0.2). The inlet median total nitrogen concentration, at 0.8(0.7) mg/L, was made of approximately 19% ammonia/ ammonium and 34% nitrate + nitrite, suggesting that approximately half of the total nitrogen entering the cell was in the form of organic nitrogen. In the outlet, total nitrogen median concentration was 0.6(0.2) mg/L, composed of 42% of nitrate + nitrite and 10% of ammonia/ammonium. Mass reduction for all nitrogen species were above 90%.

Between the two monitoring periods, there were no significant differences in concentrations for phosphate and total phosphorus (p-values 0.2 and 0.3, respectively), though the interquartile range for both compounds was larger in 2013–2014 than in 2017–2018. For total nitrogen, nitrite, and nitrate + nitrite, outlet concentrations were lower in 2017–2018 than in 2013–2014 primarily because some very high concentrations (above the range detected in 2017–2018) were reported in 2013–2014. The 2013–2014 outlet nitrogen spikes corresponded to samples with similar spikes in concentrations of phosphate and total phosphorous. These were likely associated with flushing of organic material from the freshly laid mulch. Ammonia/ammonium at the outlet did not change significantly between the monitoring periods.

3.3.4. PAHs

PAHs were analyzed in 2013–2014 and 2017–2018, but there was insufficient data in 2013–2014 to allow for statistical analysis. Compounds that were analyzed but never detected at the inlet and outlet include acenaphthene, benz(*a*)anthracene, benzo(*g,h,i*)perylene, benzo(*k*)fluoranthene, dibenz(*a,h*)anthracene, indeno(1,2,3-c,d)pyrene, and perylene. Compounds that were detected at the inlet but never at the outlet include 2-methylnaphthalene (detected at the inlet 20 times), acenaphthylene (23 times), benzo(*b*)fluoranthene (24 times), benzo(*e*) pyrene (23 times), chrysene (23 times). The only compounds that were detected at the inlet and outlet in 2017–2018 were 1-methylnaphthalene, anthracene, benzo(*a*)pyrene, fluoranthene, fluorene, naphthalene, phenanthrene, and pyrene. The results are consistent with the most

Table 3
Summary of cumulative inflow, exfiltration/evapotranspiration (ET) and outflow depths over the monitoring period in different years, relative to the area of the bioretention cell.

	Precipitation (mm)	Inflow (mm)	Total Inflow (mm)	Outflow (mm)	Exfiltration + ET (mm)	Outflow (%)	Exfiltration + ET (%)
2013–2014	1121	931	2052	105	1947	5%	95%
2017–2018	847	680	1527	121	1405	8%	92%

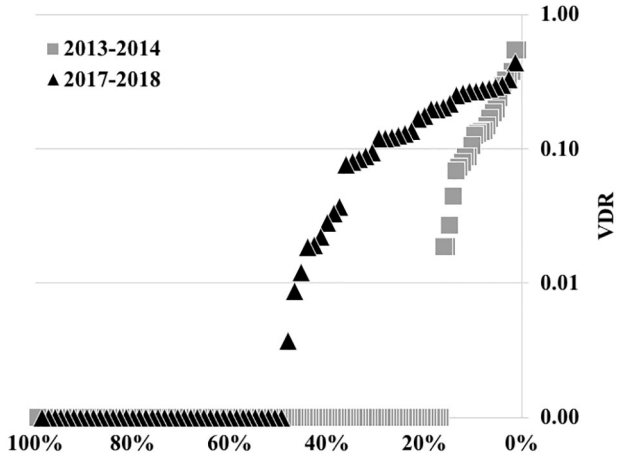

Fig. 3. Cumulative probability plot for Volume Discharge Ratio (VDR, or outflow volume divided by inflow volume) for the 2013–2014 and 2017–2018 monitoring periods.

soluble compounds (also associated with the lowest octanol-water partitioning coefficients) usually being detected the most frequently and in both the inlet and the outlet (Table S9). Variable reductions in concentration were seen among the PAH compounds, such as no change between inlet and outlet for acenaphthylene, anthracene, benzo(*b*) fluoranthene, benzo(*e*)pyrene and chrysene, statistical increase at the outlet for 1-methylnapthalene, benzo(*a*)pyrene, and statistical decrease at the outlet for fluoranthene, fluorene, naphthalene, phenanthrene and pyrene. Only the inlet concentrations of pyrene exceeded the Canadian Water Quality criterion of 0.025 µg/L in some of the samples, while phenanthrene inlet concentrations sometimes exceeded the interim Provincial Water Quality Objectives of 0.03 µg/L. However, all outlet concentrations were below these thresholds, suggesting that overall, the bioretention cell was able to protect downstream ecosystems from PAHs contamination. Mass reduction was high for all detected PAHs, at 60–90%, due to high volume reduction.

3.3.5. Metals

Concentration reduction was statistically insignificant for most metals, mostly due to the fact that inlet concentrations were already very low for many of the metals. Statistically significant decreases in concentrations of total metals at the outlet were observed for 5 out of the 25 metals analyzed, i.e. chromium, cobalt, copper, manganese and zinc with *p*-values ranging from 0.01 to 0.03. Compounds that were detected but in less than 6 samples during 2017–2018 were cadmium, chromium, lead, lithium, nickel, silver, tin and zirconium. All metals except aluminum, copper and zinc (as well as chromium, iron, lead and cobalt for one inlet sample in 2017–2018 each) had concentrations below the corresponding Canadian Water Quality criteria at the inlet and outlet in 2017–2018. Both the inlet and outlet concentrations of copper were above the criteria of 2 µg/L (Canadian) and 5 µg/L (Ontario), while those of aluminum exceeded the Canadian Water Quality criteria of 100 µg/L. Zinc concentration was above the criteria of 7 µg/L (Canadian) and 20 µg/L (Ontario) for all inlet and outlet concentrations in 2017 and 2018. Mass reductions for all metals were very high, with all mass reductions between 86 and 96%.

The outlet concentrations decreased between the two monitoring periods for aluminum, barium, calcium, chromium, cobalt, copper, iron, magnesium, manganese, nickel, potassium, sodium, strontium and titanium (p-values < 0.05); they increased for beryllium, cadmium, uranium and zinc (p-values < 0.05); and they did not change significantly for lead, molybdenum and vanadium (p-values 0.1–0.2). The decrease in calcium concentration at the outlet may be due to the gravel underdrain layer, which likely leached more immediately post-construction than after 5–6 years. Iron, lead and cobalt had outlet concentrations that infrequently (< 2 times) exceeded water quality criteria in 2013–2014 that were not exceeded at the outlet in 2017–2018. Aluminum, copper and zinc had concentrations frequently exceeding the guidelines at the outlet in both 2013–2014 and 2017–2018. The majority of metal concentrations had a larger interquartile range in 2013–2014 than in 2017–2018.

4. Discussion

4.1. High hydrologic performance maintained over years

The design of this bioretention cell theoretically allowed to retain 15 mm runoff events, which was accomplished for most rain events. Overall, the bioretention cell performed very well for water quantity management even four years post-construction (see Table 2 for details). The abstraction volume and volume reduction remained very high. Four years post-construction, the mean VDR was much less than 0.33 (Davis, 2008), the reduction in outflow volume was greater than 90%, and more than 50% of 15-mm storm events, i.e. the equivalent storm depth of the bioretention abstraction volume, were completely retained and produced no outflow. The volume reduction of greater than 90% found in this study is comparable to past hydrologic bioretention cell studies on mature systems. For example, a 4-year-old bioretention cell in Oslo, Norway with a partially clogged underdrain had a volume reduction of 100% (Paus et al., 2016). A 7-year old bioretention cell with a conventional underdrain in Oklahoma, referred to as the "ECP" bioretention cell, had a mean volume reduction of 73%, and two cells from this same study had volume reductions of 71% and 60% as long as no inflow from the groundwater was entering the bioretention cells (Kandel et al., 2017). Finally, a 7-year-old bioretention cell in Virginia had a median volume reduction of 84% when only accounting for events that produced some outflow (Willard et al., 2017). In field studies of bioretention cells, the probability of meeting the VDR target of 0.33 typically lies between 42 and 83% (Davis et al., 2012; Johnson, 2019; Winston et al., 2016). The bioretention cell in our study met the VDR target of 0.33 in 97% of the cases in both monitoring periods.

This high performance is likely due to continuous maintenance of the bioretention cell, with good vegetation establishment and replenishment of organic material on a regular basis, which are both known to increase soil hydraulic conductivity (Barzegar et al., 2002; Materechera et al., 1993). Because the bioretention cell was in a region with cold winters, the generation of larger and more connected soil pores due to freezing and thawing cycles might have contributed to maintaining a high infiltration capacity over time (Denich et al., 2013; Ding et al., 2019).

The high hydrologic performance can also be explained by the reduced-flow outlet valve, which forced water to remain within the filter media longer, thus inducing exfiltration from the sidewalls and vertically below the bottom of the cell. Winston et al. (2016) found that the drawdown rates for bioretention cells were much higher than the measured vertical saturated hydraulic conductivity of clay soils. The authors attributed this difference to increased sidewall exfiltration because of high head, high lateral soil saturated hydraulic conductivity, and the presence of a saturated zone. The effect of the presence of a valve and different valve designs and placements on the underdrain deserves further attention as it has the potential to offer a flexible control of bioretention cell hydrology. For example, the effect of further reduction of the valve orifice, beyond the 80% used in this study, must be tested. When regular inspection and maintenance of a bioretention cell is possible, manual control of such valves is possible. With growing interest in smart stormwater systems (Kerkez et al., 2016), automatic control could help manage a system's hydrology remotely when in-person visit is not possible. While automated control of stormwater bioretention systems is a seductive idea, it is yet to be demonstrated as feasible, cost-effective, and reliable in the field.

The field-derived bioretention abstraction volume was much less

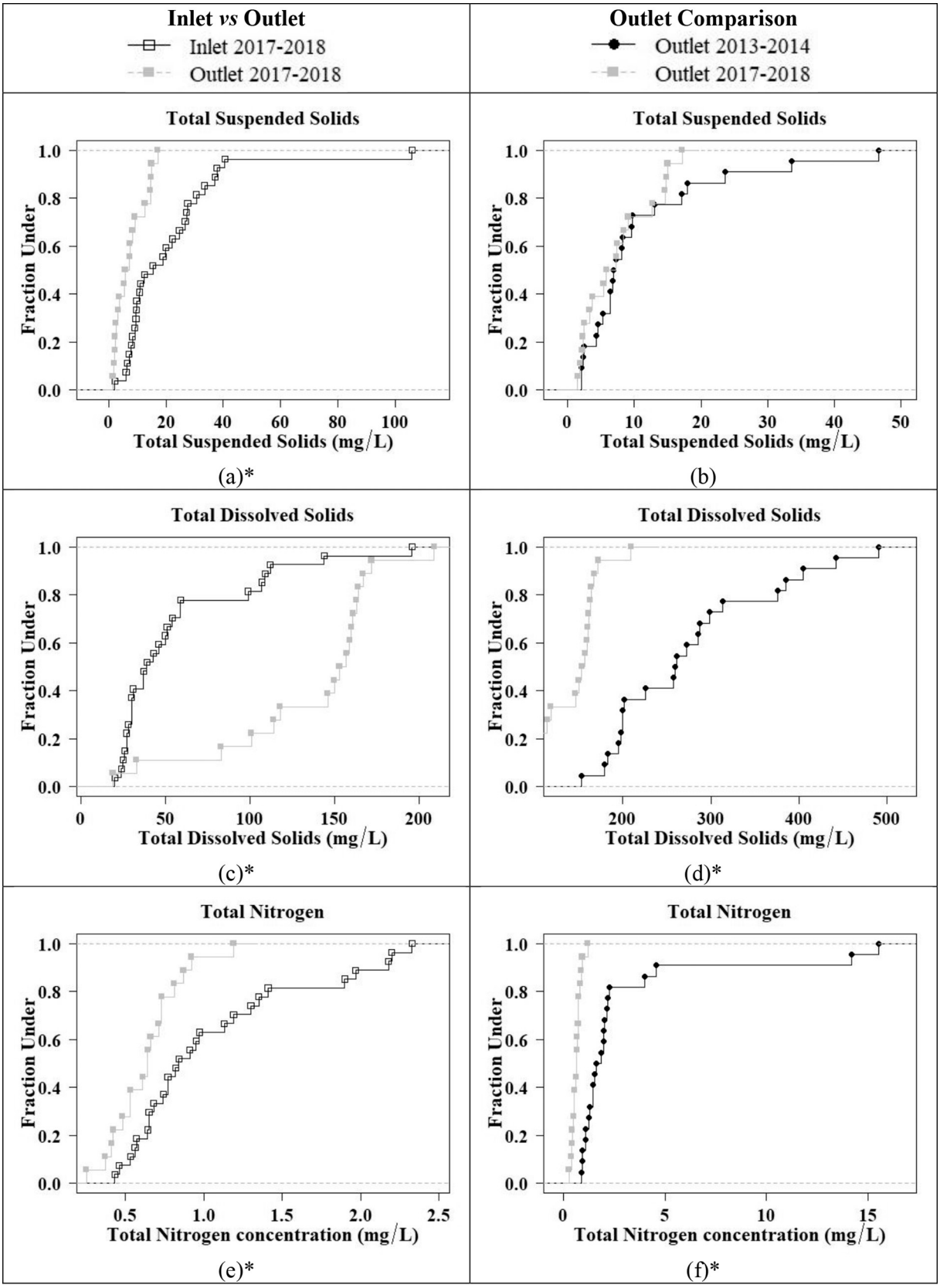

Fig. 4. Cumulative distribution functions (CDF) for outlet comparison (2013–2014) and inlet to outlet comparison (2017–2018). * denotes significant change (at $\alpha = 0.05$).

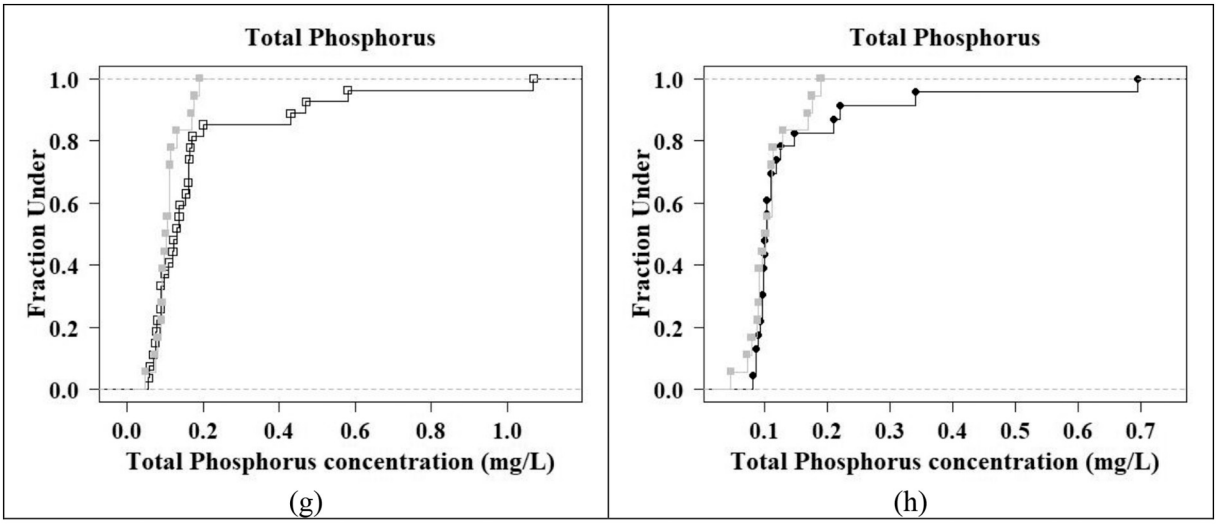

Fig. 4. (*continued*)

than the theoretical bioretention abstraction volume (see Table 2). The concept of theoretical bioretention abstraction volume was first proposed by Davis et al. (2012). In the three field studies reported by the authors, the field-derived bioretention abstraction volume was 10–25% less than the theoretical BAV (Davis et al., 2012). The difference between field-derived and theoretical average BAV in our study was 58% in 2013–2014 and 46% in 2017–2018. This suggests that the entire bioretention cell was not used for storage and dead volumes likely occurred. Methods for reducing dead volumes and encouraging contact within the bioretention cell include increasing the flow length to the outlet and increasing the media depth.

Finally, even though high hydrologic performance was maintained after 4 years of operation, it decreased noticeably between 2013 and 2014 and 2017–2018. For example, the amount of stormwater exiting the bioretention cell increased in 2017–2018 compared to 2013–2014. This was partly due to the slightly higher inlet volumes observed in 2017–2018 compared to 2013–2014. In addition, the portion of the < 15 mm events resulting in outlet flow was larger in 2017–2018 (41%) than in 2013–2014 (7%). This comparatively poorer performance after 4 years might be due to decreased exfiltration, lower storage volume within the bioretention cell media, and/or short-circuiting due to macropore formation. In a comparable study of monitoring water quantity post-construction and 6 years later, Willard et al. (2017) found statistically significant decreases in volume reduction and peak flow reductions between the two monitoring periods. Contrary to our study though, Willard et al. (2017) had more precipitation in the later monitoring period, whereas the difference between rainfall in our two monitoring periods was not significant. Therefore, whether or not this was representative of an actual trend should be further investigated. Regardless, the bioretention cell was still able to provide excellent volume and peak reductions thus continuing to reduce the amounts of chemicals reaching downstream aquatic systems.

The design of the bioretention cell presented here was conventional and the cell was a good model for many systems built in various part of the world, including North America (CVC and TRCA 2010; MPCA, 2018; Prince George's County Maryland, 1997), Europe (CIRIA, 2015), and Australia (Victorian Stormwater Committee, 1999). Conventional designs involve the use of 50–125 cm of highly sandy media, at 60–90% sand, containing vegetation and organic matter. Conventionally designed bioretention cells rarely incorporate media additives for enhanced contaminant retention or biotransformation (CVC and TRCA, 2010). The steady hydrologic performance observed in this study up to 5 years post-construction suggests that many existing bioretention cells are likely to continue to infiltrate water adequately.

4.2. High mass reductions but low concentration reductions in the effluent

The bioretention cell showed high mass reductions of more than 80% for the majority of contaminants, with no releases of contaminants to surface water. This was entirely due to the high volume reductions. While runoff volume from impervious areas, which is a lead cause of urban stream syndrome (Walsh et al., 2005), can be reduced with bioretention, it may not improve surface water quality, as much of the shallow groundwater will reach surface water without receiving treatment. For example, Fischer et al. (2003) found elevated levels of volatile organic compounds and pesticides as well as lower dissolved oxygen levels in groundwater underneath infiltration practices. However, the long travel time in soil and groundwater can allow for some slow natural attenuation, which can contribute to protecting groundwater-fed ecosystems. Natural attenuation includes biological, chemical, and physical processes such as biotic and abiotic transformation of PAHs, microbiological nitrogen reactions (nitrification, denitrification, annamox, DNRA), and adsorption of some metal and phosphorous species to the substrate.

Though achieving high mass and volume reductions, the bioretention cell often led to larger outlet than inlet concentrations. Outlet water quality in terms of nutrients, PAHs and metals was little improved by the presence of the bioretention cell. For example, with nitrogen species, ammonia/ammonium concentrations were significantly reduced by the bioretention cell, while the concentrations of the oxidized inorganic nitrogen forms, nitrate and nitrite, did not consistently decrease or increase through the cell. This suggests that both production of nitrate and nitrite through aerobic ammonia nitrification occurred in the cell and was sometimes followed by denitrification under saturated conditions. Ammonification of dissolved organic nitrogen likely occurred; however, the relatively steady proportion of organic nitrogen over time was explained by input from vegetation. The clear dominance of nitrification over denitrification was similar to results from other conventionally designed bioretention cells (Bratieres et al., 2008; Brown and Hunt, 2012; Hunt et al., 2006; Johnson and Hunt, 2019; Khan et al., 2012; Li and Davis, 2009; Lucke and Nichols, 2015; Willard et al., 2017). Concentrations of phosphorus (total and phosphate) were not reduced by the bioretention cell likely due to a combination of phosphorus adsorption onto and desorption from the soil as well as inorganic phosphorus formation from organic matter degradation.

This study shows that conventionally designed bioretention cells, while useful for reducing suspended solid concentrations and particulate contaminants by physical filtration, are not capable of reducing

dissolved contaminants. These results were not surprising. Dissolved contaminants can be eliminated via transfer processes, such as adsorption onto the soil media and plant uptake, and transformation mechanisms, such as phytotransformation (i.e. by plants) and biotransformation (i.e. by microorganisms present in the soil media). Conventionally designed bioretention cells focus on maintaining high water infiltration, resulting in short retention times, at approximately 2 h for this cell (Gu et al., 2020). This leaves few opportunities for dissolved contaminants to be transformed during a runoff event, even though transformation can occur between events, even for trace organic contaminants (Gu et al., 2020 To achieve higher concentration reduction for nutrients, trace organics and metals, an increase in hydraulic retention time would need to be combined with changes to media and vegetation. For example, using media with enhanced sorption capacity can increase removal of metals (Lim et al., 2015) and phosphorus (Marvin et al., 2020). Vegetation can increase infiltration capacity via macropore creation (Le Coustumer et al., 2012), support phosphorus sorption around roots (Muerdter et al., 2018), and some plant species can enhance nitrogen uptake (Bratieres et al., 2008; Waller et al., 2018). The retention time can be increased with adaptive controls on the outlet structure or the creation of a temporary saturated zone within the media to support denitrification. Soil media incorporating bioavailable organic carbon are also critical for denitrification of oxidized inorganic nitrogen species (Waller et al., 2018). While promising research is being done to improve the design of bioretention cells for enhanced treatment performance, in the meantime, many conventionally-designed bioretention cells have been implemented and will be in operation for decades to come. Therefore, further research should evaluate how to retrofit existing bioretention systems to enhance their treatment efficiency while maintaining acceptable hydrologic performance.

The majority of outlet concentrations were significantly different between the two monitoring periods, and overall lower in 2017–2018 than in 2013–2014. These differences are likely due to the establishment of media over time. New media and mulch that are in the process of settling and compacting in 2013–2014 occasionally released soil particles and nutrients. In 2017–2018, settled and compacted media had stable properties. The spread in outlet concentrations (SI Figs. S10) was much higher in 2013–2014 than in 2017–2018 for most metals, PAHs, all nutrients and suspended solids. This is in line with previous research on mature systems that showed maintained or improved hydrologic and water quality performance after maturity (Johnson and Hunt, 2019; Lucke and Nichols, 2015; Willard et al., 2017). Johnson and Hunt (2019) found lower overall effluent concentrations of total nitrogen and total phosphorus 17 years post-construction than immediately post-construction; concentrations of heavy metals and hydrocarbons in the media were found to be within acceptable soil limits for Australia 10 years post-construction in Lucke and Nichols (2015); and Willard et al. (2017) found the effluent loads of TN and TP seven years post-construction were not significantly different from those measured immediately post-construction. Even though enhanced performance was observed with this mature system, it is still uncertain how long it can be sustained. Guidelines for maintenance of bioretention systems recommend replacement of media after 25+ years (e.g., Ontario, Canada estimates rehabilitation needed at 25 years (Toronto and Region Conservation Authority and University of Toronto, 2013), New Zealand estimates a lifespan of 25 years (Lewis et al., 2015), and Singapore estimates a design life of at least 35 years (Wang et al., 2016)). However, field validations of the expected lifetime of bioretention cell media are lacking. Large-scale visual inspections of systems 11–22 years of maturity in Germany showed acceptable hydraulic conductivities for 44/48 sampled soils (Kluge et al., 2018). Johnson and Hunt (2019)'s water quality study reported improved total nitrogen and total phosphorus concentration reductions after 17 years of operation. These are promising results that suggest that bioretention systems might provide acceptable water infiltration and treatment performance beyond 20 years. The changes in hydrologic and treatment performance observed in this study also suggest that for a reliable evaluation of bioretention treatment performance, monitoring should start at least 2 years post-construction, to allow for soil media settlement and vegetation establishment.

5. Conclusion

Conventionally designed bioretention systems are meeting the performance goal of reducing runoff volumes entering surface water. This has a net benefit of reducing erosion and decreasing the loads of contaminants in the downstream surface water receiver. However, when outflow is produced from the bioretention, limited improvement in water quality is observed. Effluent water quality for this system improved over time, which is likely correlated with soil media and plant establishment. It is not yet known for how long this improvement in effluent water quality and high volume reductions will last, and the current guidance of 20–25 years lifetime for a bioretention system still needs to be supported by more field evaluations of old systems.

We recommend revisiting the performance targets of bioretention with respect to effluent water quality, and set more stringent requirements, that can likely only be achieved with less conventional designs, i.e., more engineered design and/or operation strategies. This can be achieved by hydraulic controls, targeted media amendments, and proper accounting for all sources (e.g., organic N and P from plants) and sinks (via adsorption and bio- and phyto-transformation) of contaminants within the system. It appears that mature systems have the same hydrologic performance and improved effluent quality as newly built systems. Therefore, a reliable assessment of the treatment performance of a bioretention cell can be obtained once its soil media and vegetation are established, approximately 2 years post-construction.

Contributions

All authors have contributed to the work. Sylvie Spraakman designed and carried out most of the experiments and analyzed the water quantity and quality data. Water quality samples were collected by technical staff at the Toronto and Region Conservation Authority, and analyzed at the Ontario Ministry of Environment, Conservation and Parks. Tim Van Seters, Jennifer Drake and Elodie Passeport supervised the work and helped interpret the data. All authors contributed to the preparation of the manuscript and have approved its final version.

Declaration of Competing Interest

None

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoleng.2020.106036.

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