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Street Tree Pits as Bioretention Units: Effects of Soil Organic Matter and Area Permeability on the Volume and Quality of Urban Runoff

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Abstract The quantity, intensity, and quality of urban stormwater runoff are changing as a consequence of urbanization and climate change. Low impact development (LID) techniques (e.g., bioretention systems) are emerging to manage runoff quantity and quality. Street tree pits were used as bioretention units in Montreal, Canada. The concentration and mass flux of contaminants (Na, Cr, Ni, Cu, Zn, Cd, Pb) and dissolved organic carbon (DOC) were measured in soil solution samples from the tree pits. The soil organic matter (SOM) and the permeability of the area nearby the tree pit (sidewalk and front lawn) were tested. The SOM did not affect contaminant concentrations. However, tree pits with higher SOM reduced the mass flux of contaminants more than tree pits with lower SOM. Sidewalk permeability decreased the concentration and mass flux of contaminants observed (e.g., Na and Cr). The estimated water flux in the open part of the tree pit changed from 6.15 to 1.64 mm week^{-1} from the less permeable units (absence of lawn + impermeable sidewalk) to the more permeable units (presence of lawn + permeable sidewalk). Urban runoff quality and quantity were locally affected by the tree pits. This indicates that the increase in surface permeability and SOM in street tree pits is advisable. Street tree pits have the potential as bioretention units to locally mitigate some of the impacts of urbanization. City planners could consider the use of street tree pits as bioretention units to help the management of urban runoff.

Keywords Low impact development · Bioretention system · Street tree pit · Permeable pavement · Trace metals · De-icing salt

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1 Introduction

Non-point-source pollutants found in urban runoff water include dust, weathered building material, and industrial and vehicular combustion products. These contaminants may be present as airborne particles or as deposits on surfaces. Precipitation can transport these pollutants. According to the United States Environmental Protection Agency (Office of Water) (2002), precipitation and runoff transportation of pollutants can be a source of non-point pollution. In the urban environment, the fate of contaminants is directly related to the fate of runoff.

Urbanization intensifies the importance of runoff. During urban development, the expansion of impervious areas increases runoff volume and reduces ground water recharge (Liu et al. 2016; Davis et al. 2010; Bedan and Clausen 2009). Increased impervious surface also changes the quality of runoff as pollutants accumulate and are transported to nearby water bodies as runoff is collected by stormwater drains (Davis et al. 2010). According to Kamali et al. (2017), about 46% of the

pollution of water bodies is the result of urban runoff. The quality of urban runoff directly affects receiving waters, ecosystems, and even human health. Trace metals are of particular concern, due to their prevalence and persistence in the environment (Joshi and Balasubramanian 2010). However, the fate of such pollutants was not usually considered in the design of traditional urban drainage systems.

In addition, the increase of impervious surfaces increases the accumulation of water on these surfaces, and hence the peak and total volume of runoff in urban areas, increasing the risk of flooding. To address this issue, traditional urban drainage systems are intended to collect, convey, and discharge water quickly and efficiently (Bedan and Clausen 2009) which can intensify the impact of urban runoff if it is not treated in timely fashion. As the impermeable urban areas increase, the pressure on these drainage and treatment systems increases. Flooding will result if the system capacity is exceeded, causing economic loss, pollution, traffic interruptions, and health issues. A conventional response is to expand and upgrade the drainage system to reduce the probability of such scenarios. However, this response is costly and may be impractical, especially in more urbanized areas. In addition, climate change is increasing the intensity and frequency of heavy rain events (Kirtman and Power 2014). For example, Willuweit et al. (2016) stated that, in Ireland, climate change could increase monthly runoff by 30% during the winter season.

Low impact development (LID) is an approach to minimize the impact of urbanization on the environment (Jia et al. 2016; Bedan and Clausen 2009), especially regarding changes in runoff (Zahmatkesh et al. 2014). Low impact development practices can be used in the design of parking lots, streets, and highways in residential, commercial, or industrial areas (Tedoldi et al. 2016). The increased implementation of LID attenuates peak runoff, contributes to groundwater recharge, and reduces combined sewer overflows (Tedoldi et al. 2016). The use of LID structures improves water quality and controls the movement of pollutants (Liu et al. 2016).

Different practices are considered as LID. These practices aim to mimic the natural hydraulic functions of the area prior to urban development or to retain the hydraulic functions before urban development, such as higher local infiltration and evapotranspiration and lower generation of runoff. Some examples of LID are bioretention units, grassed swales, green roofs, and permeable pavements (Liu et al. 2016; Bedan and Clausen 2009). These practices are generally scattered across an area with multiple instances of small-scale infrastructure. This makes it possible to use these practices to address the impact of nonpoint-source pollution because they act on urban runoff rather than on the source. This is achieved by improving local runoff retention and infiltration and overall water management processes (Jia et al. 2016).

Bioretention units are generally small in size, and they tend to have esthetic as well as functional value. This makes the deployment of such units possible in a wide range of local conditions while still accomplishing stormwater management goals (Trowsdale and Simcock 2011). According to Davis et al. (2009), bioretention units are widely used but the performance of such practices in terms of water quality under different conditions and varied weather is still not fully understood.

Street tree pits and bioretention units have similar features. In general, they are small scale, include plants and soil, and are replicated across urban areas. However, they are designed with different purposes. Bioretention units are designed to receive urban runoff water and improve its quality as the water passes through the soil. The soil of bioretention units may be amended with additional organic matter to improve its capacity to retain water and contaminants. Street tree pits are designed to provide life support for the trees and plants that are part of the pit. Street tree pits are generally smaller and do not have soil amendments with the purpose of retaining contaminants. Thus, the performance of street tree pits as bioretention units is still not fully understood. In addition, different street tree pit designs might have different potential for use as bioretention units. Information about the use of tree pits as bioretention units is valuable for city planners who are working towards greener cities.

According Bedan and Clausen (2009), most studies on the performance of LID practices have focused on only one practice, and few studies have followed their interactive effects (e.g., bioretention and permeable pavement). An improved understanding of the performance of such management practices on urban runoff quality and the fate of its pollutants is therefore necessary. This study was conducted to explore the effects of design factors of street tree pits as bioretention units on the quality of urban runoff and the fate of the pollutants that infiltrate the tree pits. As a continuation of the study by Kargar et al. 2016), the trace metals analyzed in this experiment were Na, Cr, Ni, Cu, Cd, and Pb as well as DOC.

2 Material and Methods

This study was conducted in the Hochelaga-Maisonneuve neighborhood of the city of Montreal (Canada) located at 45° 30′ N 73° 34′ W as a joint effort of the City of Montreal and McGill University. The mean daily temperature ranges from − 9.7 °C in January to 21.2 °C in July, and monthly precipitation from 62.7 mm in February to 96.4 mm in November, respectively. Montreal has four distinct seasons with warm to hot summers and cold, snowy winters (Environment and Natural Resources Canada 2017).

The experiment followed a split-split-plot design with double-repeated measures over time and depth with the experimental unit sampled in two locations. Twenty-four tree pits were used. The design factors were soil organic matter (SOM) (< 5% or > 5% *w/w*), sidewalk (permeable or impermeable), and lawn (presence or absence). Horizontal sampling location (near curb or near sidewalk), sampling depth (surface, middle, or bottom) (Fig. 1), and sampling intervals were considered.

Soil solution samples were collected every 2 weeks from October 2016 to August 2017 except when the ground was frozen from December 2016 to April 2017. The samples were collected using zero-tension lysimeters, adapted from MacDonald et al. (2004). The main parts of the lysimeters were a storage body and a funnel. The storage body was a 0.9 m, 0.1-m-diameter acrylonitrile butadiene styrene (ABS) pipe, with an ABS cap at the bottom and a rubber cap at the top. Each storage body had a ¾ inch, 45-degree wye or 90-degree L-shaped adapter, connected to a hose and the funnel. The adapter type was determined by the depth at which the funnel was placed, to allow the gravimetric flow of the soil solution from the funnel into the storage body.

Fig. 1 The layout of lysimeters installed in a tree pit

The lysimeters were placed in groups of three, to collect soil solution from near the surface, at 30 cm depth and at 55 cm depth. In each tree pit, one set of lysimeters was close to the street and the other was close to the sidewalk (Fig. 1). This study used expanded street tree pits with 4.5 m^2 of open area and soil volume ranging from 7.3 to 8.3 m^3. The available soil extends underneath the sidewalk (Fig. 2).

Soil solution samples were collected from the lysimeters using a vacuum hand pump and stored in labeled, high-density polyethylene bottles and placed in an ice box for transport. About 50 ml of each sample was filtered using a 0.45-μm nylon membrane. Ten milliliters were separated from the filtered solution, acidified with grade nitric acid, and stored in the refrigerator at 4 °C until the chemical analysis (Hendershot et al. 2008). Inductively coupled plasma mass spectrometry equipment (Varian 820 MS, Analytik-Jenna, Germany) equipped with a collision reaction interface was used to analyze the concentration of trace metals (Cr, Ni, Cu, Zn, Cd, and Pb) and Na. Thirty milliliters of the remaining filtered solution were used to determine the concentration of dissolved organic carbon (DOC). The total mass of each contaminant in each lysimeter was estimated using the measured concentration and the sample volume. Water flux was estimated using the sample volume of each lysimeter, normalized to an area of 1 m^2, and the period normalized to 1 week. The mass flux of contaminants was estimated by multiplying the total mass of each sampled contaminant by the water flux.

To determine the performance of street tree pits as bioretention units, the effects of the design factors were estimated using SAS™ statistical software, version 9.4 (SAS Institute Inc., Cary, NC, USA) with the Mixed Procedure. A correlation analysis was run on the residual errors. The Pearson correlation coefficient was calculated for each combination of dependent variables with the SAS™ CORR Procedure.

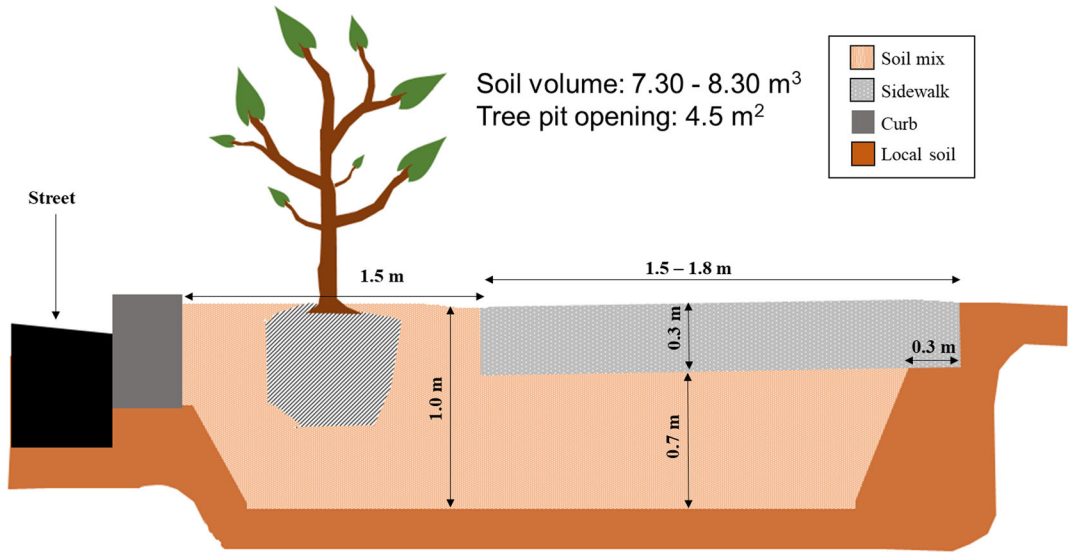

Fig. 2 Expanded street tree pit depicting the excavated area as well as the available soil for root growth

3 Results and Discussion

The mean concentration of each contaminant, partitioned by treatment factor, can be seen in Table 1. The calculated water flux and estimated mean mass flux of each contaminant partitioned by treatment factor can be seen in Table 2. Cadmium values were excluded from the analysis due to the fact that the analyzed values were below the minimal detection limit of the equipment used.

3.1 Analysis of Contaminant Concentration

The horizontal sampling locations in the tree pit influenced the concentrations of Na ($p < 0.001$), Cr ($p < 0.01$), and Cu ($p = 0.08$). Both Na and Cu concentrations were higher in samples collected near the sidewalk as compared to those collected near the street. This indicates that the bigger proportion of these contaminants came from the sidewalk area, from sources such as de-icing salt, used in cities with cold climates, or weathered building materials. Chromium concentration, on the other hand, was higher near the street, which could indicate that most Cr in solution is from the splashing caused by cars that pass along the street. Chromium plating is commonly used to provide wear and corrosion resistance to the braking system of cars which can yield toxic waste over time (Bogdanova et al. 2002).

The permeability of the surrounding surfaces was related to the concentrations of contaminants sampled in the tree pits. The concentrations of Na ($p < 0.001$), Cr ($p < 0.001$), and Pb ($p = 0.04$) were lower in the solution of tree pits with permeable sidewalks. However, Zn concentration increased ($p = 0.06$) in tree pits with permeable sidewalks. The presence of a lawn was associated with increased concentrations of Cr ($p < 0.001$), Ni ($p = 0.03$), Cu ($p = 0.06$), Zn ($p = 0.01$), and DOC ($p = 0.01$). This effect could have been caused by a reduction in volume of the water. Table 2 shows that water flux through the open tree pit was significantly reduced by the presence of a lawn ($p < 0.01$), from 5.09 to 1.98 mm week^{-1}. Some of the water and the contaminants in it were likely infiltrating the lawn and sidewalk before they reached the open part of the tree pit where the lysimeters were located.

The effects of time are shown in Tables 1 and 2. The first four sampling events were prior to the freezing of the soil in the tree pits (December 2016). Some snow fell during sampling period 4 (December 2016) but did not persist on the ground. Snowfall, like rain, can capture contaminants and then deposit them on the ground (Takeda et al. 2000; Nguyen et al. 1979). Copper concentration in the soil solution of the tree pits was highest in early winter. Similar observations were made by Muthanna et al. (2007), who reporting high values of Cu, Zn, and Pb in snow from three different urban areas in Norway. The material inherent to the manufacturing of snow removal machinery might be a possible sources of contaminants. Furthermore, abrasion of the sidewalk surface by snow removal activity might also be a source

Table 1 Contaminant concentration in soil solution of tree pits

Variable	Na (1125)†† mg L^{-1}			Cr (1097)†† μg L^{-1}			Ni (1117)†† μg L^{-1}			Cu (1126)†† μg L^{-1}			Zn (1126)†† μg L^{-1}			Pb (1126)†† μg L^{-1}			DOC (1001)†† mg L^{-1}		
	L.	Mean	U.	L.	Mean	U.	L.	Mean	U.	L.	Mean	U.	L.	Mean	U.	L.	Mean	U.	L.	Mean	U.
Soil organic matter	NS			NS			NS			NS			NS			NS			NS		
Low	39	50	64	0.81	1.03	1.34	5.5	6.5	7.8	8.9	11.7	15.3	9.5	11.6	14.2	0.18	0.21	0.25	34.5	47.6	65.7
High	42	53	68	0.63	0.81	1.05	4.7	5.5	6.7	8.01	0.4	13.7	11.5	14.1	17.5	0.17	0.20	0.24	28.3	39.1	54.2
Depth	**			**			**			**			**			NS			*		
Surface	24	30	37	0.65	0.79	0.96	4.3	5.0	5.7	7.8	9.61	1.8	7.9	9.61	1.6	0.17	0.19	0.23	28.3	36.5	47.1
Middle	45	55	69	0.74	0.92	1.15	5.2	6.0	7.0	9.9	12.5	15.7	11.9	15.4	20.1	0.17	0.20	0.25	33.1	43.7	57.8
Deep	65	84	107	0.87	1.07	1.32	6.1	7.3	8.8	8.9	11.3	14.4	10.9	14.3	18.8	0.18	0.22	0.27	37.7	50.4	67.5
Horizontal location	**			**			NS			•			NS			NS			NS		
Near the sidewalk	50	62	76	0.82	0.84	1.24	5.4	6.2	7.3	9.5	11.8	14.6	10.1	12.3	14.9	0.18	0.21	0.25	32.5	42.2	54.9
Near the street	35	43	53	0.68	1.33	1.02	5.0	5.7	6.7	8.4	10.4	12.9	11.1	13.4	16.2	0.17	0.20	0.24	33.9	44.1	57.3
Sidewalk	**			**			NS			NS			•			*			NS		
Impermeable	72	82	95	1.15	1.33	1.53	5.9	6.5	7.2	9.3	10.8	12.7	10.0	11.2	12.5	0.21	0.24	0.26	40.0	48.0	57.7
Permeable	23	32	45	0.46	0.64	0.89	4.4	5.5	7.0	7.9	11.2	16.1	11.2	14.7	19.4	0.14	0.18	0.23	25.2	38.8	59.7
Front lawn	NS			**			**			**			*			NS			*		
Absent	35	46	62	0.52	0.70	0.95	4.4	5.4	6.6	7.0	9.6	13.1	8.8	11.1	14.0	0.17	0.21	0.26	22.6	32.9	47.8
Present	49	57	68	1.01	1.20	1.42	6.0	6.7	7.6	10.7	12.8	15.4	12.9	14.8	17.1	0.18	0.20	0.23	45.4	56.7	70.7
Sampling event	**			**			**			**			**			**			**		
24 Oct. 2016	37	48	63	0.53	0.6	60.83	3.8	4.5	5.4	8.0	10.3	13.3	12.1	15.3	19.3	0.09	0.11	0.14	36.8	50.3	68.8
7 Nov. 2016	21	28	39	0.49	0.62	0.79	3.2	3.9	4.8	8.2	10.9	14.3	17.5	22.9	30.0	0.19	0.26	0.34	26.3	38.5	56.2
21 Nov. 2016	18	24	33	0.41	0.53	0.68	3.4	4.1	5.1	9.4	12.6	16.8	13.4	17.7	23.5	0.18	0.24	0.32	19.3	30.5	48.0
6 Dec. 2016	29	38	49	0.67	0.84	1.05	6.5	7.7	9.1	17.8	22.9	29.4	14.0	17.6	22.1	0.19	0.24	0.31	26.1	35.4	48.0
24 May 2017	178	232	301	0.89	1.12	1.41	7.8	9.2	10.9	6.4	8.3	10.6	7.2	9.1	11.4	0.22	0.28	0.35	22.5	30.7	42.0
05 Jun 2017	46	60	79	1.19	1.51	1.90	4.1	4.8	5.8	12.4	16.12	0.8	7.9	10.0	12.7	0.16	0.20	0.26	49.1	67.9	93.7
19 Jun 2017	54	74	101	0.79	1.01	1.29	6.4	7.8	9.6	9.4	12.5	16.7	6.0	8.0	10.6	0.10	0.14	0.18	27.3	40.5	60.1
03 Jul 2017	39	56	80	0.78	1.02	1.34	5.3	6.6	8.3	7.0	9.6	13.2	10.1	13.9	19.3	0.14	0.20	0.28	33.5	49.8	74.1
15 Jul 2017	39	56	79	0.73	0.95	1.23	6.0	7.5	9.3	11.5	15.9	21.8	8.0	11.2	15.6	0.24	0.34	0.48	28.9	43.0	64.0
29 Jul 2017	38	53	74	0.79	1.03	1.34	4.6	5.7	7.1	5.0	6.8	9.2	6.1	8.5	11.7	0.11	0.15	0.21	30.8	45.3	66.7
12 Aug. 2017	28	41	59	0.71	0.92	1.20	4.4	5.6	7.1	6.8	9.5	13.3	10.2	14.6	20.8	0.16	0.22	0.32	33.7	51.1	77.5
26 Aug. 2017	29	41	58	0.98	1.27	1.64	6.0	7.5	9.3	4.7	6.4	8.7	9.4	13.3	18.8	0.14	0.20	0.28	33.0	48.2	70.4

† Values are estimates of the mean, †† Element (sample size), *L* lower bound, *U* upper bound, *DOC* dissolved organic carbon

Sign over estimated mean denotes statistical significance of difference among levels of the factor: • ($p < 0.10$), *($p < 0.05$), **($p < 0.01$), NS (non-significant)

Table 2 Mass flux of contaminants and water flux through tree pits

Variable	Na (1721)†† mg week^{-1} m^{-2}			Cr (1649)†† μg week^{-1} m^{-2}			Ni (1713)†† μg week^{-1} m^{-2}			Cu (1722)†† μg week^{-1} m^{-2}			Zn (1722)†† μg week^{-1} m^{-2}			Pb (1722)†† μg week^{-1} m^{-2}			DOC (1726)†† mg week^{-1}m^{-2}			WF (1728)†† mm week^{-1}		
	L.	Mean	U.	L.	Mean	U.	L.	Mean	U.	L.	Mean	U.	L.	Mean	U.	L.	Mean	U.	L.	Mean	U.	L.	Mean	U.
Soil organic matter	NS			NS			NS			NS			NS			NS			NS			NS		
Low	26	49	90	2.13	3.30	5.05	7.1	11.6	18.9	11.1	17.3	27.0	9.9	16.4	27.1	0.81	1.14	1.59	18.1	32.9	59.8	2.1	3.2	4.8
High	22	40	75	1.92	3.00	4.54	5.8	9.4	15.4	9.2	14.3	22.3	9.9	16.3	26.9	0.86	1.20	1.68	13.6	24.7	44.9	2.1	3.2	4.8
Depth	***			***			***			***			***			*			***			***		
Surface	64	108	180	3.35	4.80	6.91	14.7	22.1	33.0	26.4	38.6	56.6	25.5	38.9	59.3	1.06	1.42	1.89	51.9	87.8	148.6	3.9	5.5	7.7
Middle	20	36	65	1.94	2.80	4.00	5.5	8.5	13.1	8.8	13.4	20.3	8.6	13.7	21.9	0.81	1.08	1.45	12.8	22.00	37.8	1.9	2.7	3.8
Deep	12	22	40	1.55	2.20	3.24	4.0	6.1	9.3	5.1	7.5	11.2	5.2	8.2	12.8	0.78	1.04	1.40	7.1	12.0	20.1	1.5	2.2	3.0
Horizontal location	**			**			**			**			**			**			*			**		
Near the side walk	41	69	115	2.83	4.00	5.64	9.4	13.9	20.6	14.9	21.6	31.3	13.9	21.0	31.8	1.10	1.45	1.90	23.1	38.1	62.9	2.9	4.1	5.7
Near the street	17	28	48	1.72	2.40	3.43	5.3	7.9	11.7	7.9	11.5	16.6	8.4	12.7	19.2	0.72	0.94	1.24	12.9	21.3	35.2	1.8	2.5	3.5
Sidewalk	**			*			*			*			NS			NS			*			NS		
Impermeable	61	87	124	3.49	4.50	5.73	11.1	14.7	19.5	16.7	21.6	28.0	15.7	21.0	28.1	1.15	1.39	1.69	30.6	43.1	60.9	3.0	3.8	4.9
Permeable	10	23	50	1.23	2.20	3.81	3.9	7.4	14.1	6.4	11.4	20.6	6.6	12.7	24.6	0.63	0.98	1.53	8.6	18.8	41.3	1.5	2.6	4.5
Front lawn	**			*			**			**			**			**			**			**		
Absent	46	94	194	2.55	4.20	7.02	10.2	18.0	31.9	16.7	28.2	47.6	16.2	29.2	52.8	1.15	1.70	2.53	25.0	50.3	101.4	3.1	5.1	8.3
Present	14	21	31	1.71	2.30	3.05	4.4	6.1	8.4	6.5	8.8	11.9	6.5	9.1	12.8	0.64	0.80	1.01	10.8	16.2	24.1	1.5	2.0	2.6
Sampling even	**			**			**			**			**			**			**			**		
24 Oct 2016	181	368	750	5.68	8.80	13.57	26.1	43.2	71.7	52.2	88.1	148.7	68.6	120.2	210.6	1.21	1.78	2.63	130.9	284.2	617.1	5.3	8.0	12.1
7 Nov. 2016	10	20	40	1.08	1.70	2.59	3.2	5.3	8.7	6.6	11.2	18.9	8.2	14.3	25.1	0.58	0.85	1.26	6.1	13.3	28.8	1.5	2.3	3.4
21 Nov. 2016	5	11	22	0.79	1.20	1.90	2.3	3.9	6.4	4.5	7.6	12.8	4.8	8.4	14.6	0.47	0.70	1.03	3.6	7.7	16.8	1.2	1.8	2.7
6 Dec. 2016	85	173	352	3.52	5.40	8.42	24.8	41.1	68.1	66.2	111.6	188.3	48.3	84.7	148.4	1.11	1.63	2.41	68.1	147.7	320.8	4.0	6.0	9.0
24 May2017	633	1394	2846	5.47	8.50	13.08	36.7	60.9	100.9	32.1	54.1	91.2	34.6	60.7	106.3	1.58	2.34	3.45	40.7	88.4	192.0	4.8	7.3	10.9
05 Jun 2017	70	143	293	4.53	7.00	10.86	10.7	17.8	29.5	27.9	47.1	79.6	18.0	31.6	55.5	0.95	1.41	2.08	59.6	129.4	280.9	3.1	4.7	7.1
19 Jun 2017	19	38	77	1.72	2.70	4.11	5.3	8.8	14.7	7.1	12.0	20.2	5.3	9.3	16.3	0.52	0.77	1.13	5.9	12.8	27.7	1.7	2.5	3.8
03 Jul 2017	10	20	41	1.53	2.40	3.67	3.9	6.5	10.8	4.5	7.7	13.0	4.9	8.7	15.2	0.66	0.98	1.45	9.1	19.8	43.0	1.6	2.4	3.6
15 Jul 2017	11	22	45	2.02	3.10	4.84	5.1	8.4	14.0	7.0	11.8	20.0	5.7	10.0	17.5	1.23	1.82	2.69	9.1	19.7	42.7	2.1	3.1	4.6
29 Jul 2017	7	13	27	1.12	1.70	2.68	2.6	4.3	7.1	2.6	4.5	7.5	2.9	5.0	8.8	0.46	0.68	1.00	4.8	10.5	22.9	1.2	1.8	2.8
12 Aug.2017	4	9	18	1.24	1.90	2.96	2.4	4.0	6.6	2.7	4.5	7.7	3.2	5.6	9.8	0.75	1.10	1.63	4.2	9.1	19.7	1.4	2.1	3.1
26 Aug.2017	5	11	22	1.40	2.20	3.34	3.3	5.4	9.0	2.9	4.9	8.3	3.9	6.9	12.1	0.76	1.11	1.65	5.7	12.3	26.7	1.5	2.3	3.4

† Values are estimates of the mean, †† Element (sample size), *L* lower bound, *U* upper bound, *DOC* dissolved organic carbon, *WF* water flux

Sign over estimated mean denotes statistical significance of difference among levels of the factor: *($p < 0.05$), **($p < 0.01$), ***($p < 0.001$), NS (non-significant)

of contaminants. According to Zhang et al. (2018), damage to permeable sidewalks can contaminate runoff. Lead concentrations were highest in late winter after the thawing of accumulated ice and snow (May 2017). The high values of Na concentration, observed in May, were also likely due to de-icing salt being washed by the spring thaw into the tree pits.

The concentrations of Na ($p < 0.001$), Ni ($p < 0.001$), Cu ($p < 0.01$), Zn ($p = 0.01$), and DOC ($p = 0.02$) all increased as the sampling depth increased. This contradicts some literature (Trowsdale and Simcock 2011; Muthanna et al. 2007; Davis et al. 2003) which reported that the concentration of contaminants decreased as the solution passed through the soil. However, the increasing concentrations in this study could have resulted from a concurrent decrease in the water volume. Table 2 shows that the effect of depth on water flux was statistically significant ($p < 0.001$), decreasing from 5.51 mm week^{-1} to 2.16 mm week^{-1}.

As pointed out by Roy-Poirier et al. (2010), the evaluation of pollutant concentration can be misleading. The authors suggested that to better evaluate the performance of bioretention units, the mass of pollutants removed should be evaluated. That is because mass removal takes into account both sample concentration and volume. In this study, mass flux was used as it takes into consideration the mass of pollutants as well as the effect of area and time. The concentrations of contaminants (Table 1) were negatively correlated to the volume of the sample, represented by the water flux (Table 2). For instance, the concentrations of all pollutants (except Pb) increased with depth while the water flux decreased with depth.

3.2 Analysis of mass flux of contaminants

The variation in mass flux of contaminants could be related directly to the variation in water flux. The correlation matrix (Table 3) shows that all the possible correlations between the mass flux of each contaminant and water flux were positive and statistically significant ($p < 0.001$). The correlations between water flux and the mass fluxes of Na, DOC, and Zn were weaker than those for other contaminants. For instance, the correlation of water flux with the mass flux of Na was 0.638 while the correlation between the mass fluxes of Na and Cu was 0.911. The weaker correlation of Na mass flux with water flux could be explained by their seasonal variation. For example, the use of de-icing salt, as one of the main sources of Na in urban areas, is greatest during winter, generating the highest Na mass flux after spring thaw (May 2017). The variation of water flux over time did not follow the pattern of the variation in Na, leading to a weaker correlation. The correlation of contaminant flux and water flux is directly related to the precipitation. Depending on the precipitation frequency, there might be a bigger effect of either dry or wet atmospheric deposition of contaminants over time (Hong et al. 2017; Connan et al. 2013). Dry deposition occurs as pollutants accumulate over urban surfaces. The deposited pollutants are transported by runoff from rain or snowmelt. Wet deposition is a result of the precipitation scavenging effect on the atmosphere.

The variation in mass flux of certain contaminants might be correlated with variation in others due to interaction between contaminants or similar sources of contamination. The concentrations of Ni and Cu are reported to positively correlate with DOC concentration (Koopmans and Groenenberg 2011). Mass fluxes of Ni, Cu, and Na presented high and positive correlation with mass flux of DOC (Table 3). This indicates a similar behavior of DOC, Cu, Na, and Ni. According to Gartzia-Bengoetxea et al. (2009), Cu concentration is expected to increase as DOC concentration increases due to the high affinity of Cu to dissolved organic compounds. This trend between Cu and DOC concentration, however, did not occur with depth (Table 1). A possible reason could be the effect of the different proportions of the fractions that composed DOC (low-molecular weight hydrophilic compounds such as humic acids and fluvic acids). These fractions have different capacities and affinities to bind metals (Koopmans and Groenenberg 2011). Another reason could be the affinity of Cu to colloids and the formation of a front of colloids as described by Pontoni et al. (2016).

The mass flux of DOC, Zn, and Cu presented a different behavior over time than that of other contaminants. Zinc and DOC presented the highest flux during the first sampling period (October 2016). This could be attributed to the disturbance of the soil when the lysimeters were installed. The mass flux of Cu was the highest during the pre-freezing period (Dec. 2016) while Zn and DOC mass fluxes were the second highest even with less solution sampled (water flux). Except for Cu, Zn, and DOC, the highest fluxes of all other contaminants were measured after the thaw (May 2017). Snowfall and de-icing salt could have influenced the contaminant fluxes. In addition, the highest water flux was during

Table 3 Pearson correlation coefficients for contaminant fluxes and water flux

Pearson correlation coefficients								
Number of observations								
	DOC	Na	Cr	Ni	Cu	Zn	Pb	WF
DOC	1							
	1726							
Na	0.840***	1						
	1719	1721						
Cr	0.692***	0.688***	1					
	1719	1720	1721					
Ni	0.856***	0.919***	0.816***	1				
	1711	1712	1712	1713				
Cu	0.844***	0.911***	0.765***	0.946***	1			
	1720	1721	1721	1713	1722			
Zn	0.800***	0.873***	0.682***	0.903***	0.912***	1		
	1720	1721	1721	1713	1722	1722		
Pb	0.267***	0.202***	0.690***	0.441***	0.353***	0.325***	1	
	1720	1721	1721	1713	1722	1722	1722	
WF	0.661***	0.638***	0.872***	0.820***	0.746***	0.701***	0.730***	1
	1726	1721	1721	1713	1722	1722	1722	1728

DOC dissolved organic carbon, *WF* water flux, ***($p < 0.001$)

the first sampling. This fact could have been a consequence of the irrigation of the recently transplanted trees. The lysimeters were installed when the trees were transplanted. After transplanting trees, the city of Montreal irrigates them to improve their acclimation and survival chances in the tree pit. The water collected by the lysimeters after irrigation was discarded, but water remaining in the soil could have contributed to the sample volumes collected later.

The greatest variation in contaminant mass and water flux was between May and June 2017. Except for DOC, the mass fluxes of all contaminants decreased between the two sampling periods. However, the change in mass flux was not the same for every contaminant. For instance, the mass flux of Na decreased from 1394 to 143 mg week^{-1} m^{-2} (approximately 90%), the mass flux of Cr decreased from 8.5 to 7.0 μg week^{-1} m^{-2} (19%), and the mass flux of Cu from 54.1 to 47.1 μg week^{-1} m^{-2} (13%). Likewise, the water flux decreased from 7.3 to 4.7 mm week^{-1} (36%). Sodium, as a very soluble metal, was quickly washed out of the tree pits after winter. Chromium and copper movement, on the other hand, was much slower.

The mass flux of all contaminants decreased with the sample depth: Na ($p < 0.001$), Cr ($p < 0.001$), Ni ($p < 0.001$), Cu, ($p < 0.001$), Zn ($p < 0.001$), Pb ($p = 0.04$), and DOC ($p < 0.001$). The estimated water flux decreased over 60% ($p < 0.001$) between the surface and the deep sampling points while the mass flux decreased for Na (79%), Cr (53%), Ni (72%), Cu (81%), Zn (79%), Pb (27%), and DOC (86%). This reduction was possibly related to decreased water flux with depth and possibly due to adsorption of contaminants by the soil since the decrease was proportionally different.

The mass fluxes of all contaminants were higher near the sidewalk. Similarly, water flux was higher near the sidewalk. Some contaminants such as Na, Ni, Cu, and DOC presented a larger difference in mass flux (59%, 43%, 47%, and 44%, respectively) than the water flux (39%). Thus, the difference in mass flux of these contaminants was not directly proportional to the difference in water flux. For

the street side, the lower water flux observed could indicate little or no effect of sidewalk run-off. In this case, most of the water is assumed to come from direct rain and water splashed from the street. The splashing in the study area was probably minimal because there were parking spots along the street.

Except for Zn and Pb, the mass fluxes of all contaminants in the open part of the tree pit were lower when the sidewalk was permeable. For

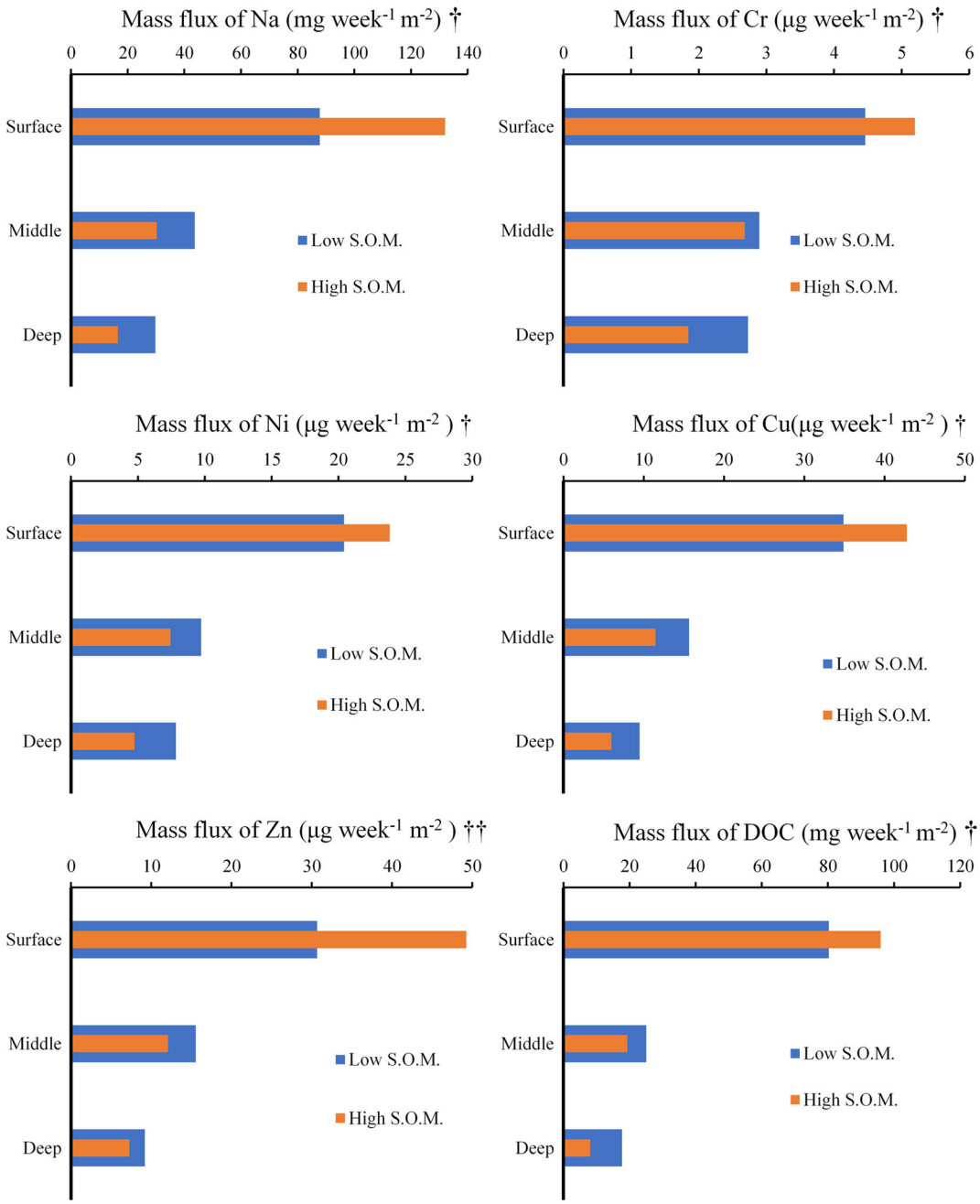

Fig. 3 Mass fluxes of contaminants in tree pits as related to soil organic matter and depth. †Statistical significance $p < 0.05$, ††statistical significance $p < 0.01$

example, mass fluxes of Cr ($p = 0.02$), Ni ($p = 0.04$), Cu ($p = 0.04$), and DOC ($p = 0.04$) in the open part of the tree pit were 52%, 50%, and 47% lower, respectively, with the use of permeable sidewalk. Likewise, the mass flux of Na was about 75% lower ($p < 0.01$) in tree pits with permeable sidewalk. These reductions are likely a result of infiltration through the permeable sidewalk, decreasing the fluxes in the open part of the tree pit. Water flux was not significantly affected by the single effect of the permeability of sidewalk. This effect was likely attenuated since half of samples were obtained from the area near the street which was less influenced by the sidewalk's permeability. When the interaction effect of sidewalk permeability and horizontal sampling location was analyzed, the attenuation effect is clearer. The estimated water flux near the street presented no difference between tree pits with permeable and impermeable sidewalks, with values of 2.42 and 2.55-mm week^{-1} respectively. The estimated water flux near sidewalk was significantly different ($p < 0.01$) for tree pits with permeable and impermeable sidewalks, with values of 2.85 and 5.78-mm week^{-1} respectively.

Similarly, the presence of a lawn adjacent to the tree pit was associated with a decrease in the mass flux of all contaminants through the open part of the tree pit: Na ($p < 0.001$), Cr ($p = 0.02$), Ni ($p < 0.001$), Cu ($p < 0.001$), Zn ($p < 0.001$), Pb ($p < 0.001$), and DOC ($p < 0.01$). Water flux through the open part of the tree pit was also reduced when a lawn was adjacent ($p < 0.001$), from 5.09 to 1.98 mm week^{-1}. The changes in mass flux of contaminants might not be related only to the change in water flux. For example, the reduction of mass flux of Na, Ni, Cu, Zn, and DOC ranged from 66.3 to 78% while water flux reduction was 61%.

The effect of SOM alone on the concentration and the mass flux of the contaminants was not statistically significant. However, the interaction effect of SOM and sampling depth was statistically significant for Na ($p = 0.01$), Cr ($p = 0.02$), Ni ($p = 0.02$), Cu ($p = 0.03$), Zn ($p < 0.01$), and DOC ($p = 0.01$) mass fluxes and water flux ($p < 0.08$). All of those mass fluxes decreased more rapidly with depth in tree pits with higher SOM (Fig. 3). As displayed in Fig. 3, there was less mass flux of contaminants at the deepest sampling point in tree pits with higher SOM, even though mass fluxes near the surface of these tree pits were proportionally higher than other tree pits. On the other hand, Pontoni et al. (2016) speculated that an increase in SOM could increase the mobility of Cu, Ni, and Cd. Soil organic matter oxidation results in water-soluble low-weight molecules such as humic acids and fulvic acids (Koopmans and Groenenberg 2011). According to Pontoni et al. (2016), these molecules interact with the contaminants and increase their mobility.

The interaction effect of SOM and sampling depth affected the water flux. Similar to the effect on the mass flux of contaminants, water flux decreased with depth, which varied depending on the SOM level. Although this interaction effect was likely one driver of the reduction of the mass flux of contaminants with depth (Fig. 3), it is not the whole story. Water flux in tree pits with lower SOM decreased 53% on average between the surface and deep sampling points, while the mass fluxes of Na and Cu decreased by 66% and 73%. In tree pits with higher SOM, the water flux decreased by 67% between the surface and deep sampling points, while the mass fluxes of Na and Cu decreased by 87% and 86%. The reduced mass flux of contaminants was disproportionate to the change in water flux. The metal binding properties of SOM could be the cause of this effect (Kargar et al. 2016).

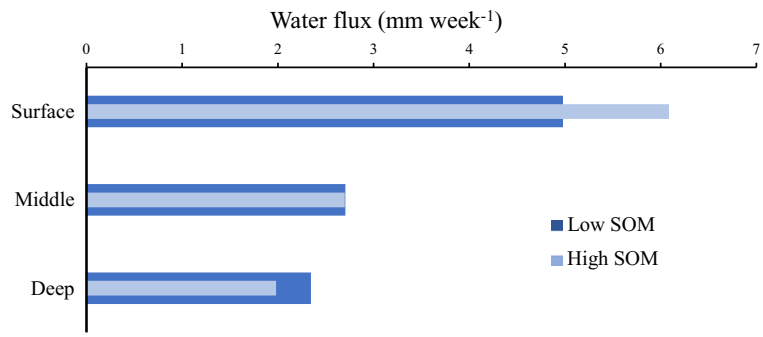

Fig. 4 Estimated mean water flux in tree pits for the interaction of soil organic matter content and depth

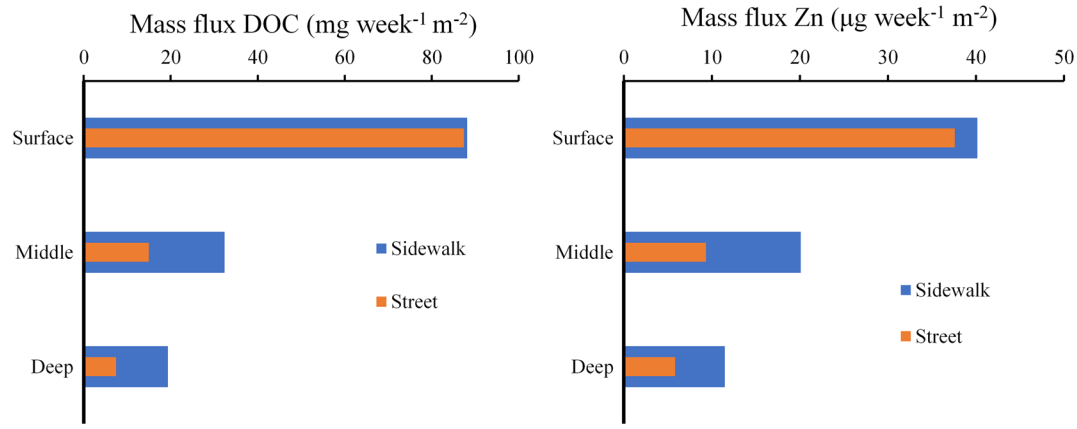

Fig. 5 Mass fluxes of contaminants in tree pits as related to horizontal sampling location and sampling depth

Soil organic matter appears to have affected moisture retention in the tree pit soil. The water fluxes in tree pits with more SOM were 6.1 mm week^{-1} near the surface and 2.0 mm week^{-1} at the deep sampling points (Fig. 4). In tree pits with less SOM, the water fluxes were 5.0 mm week^{-1} and 2.4 mm week^{-1}. The difference in water flux between the two depths was 4.1 mm week^{-1} for tree pits with more SOM and 2.6 mm week^{-1} for tree pits with less SOM. Tree pits with more SOM retained 1.5 mm week^{-1} more than tree pits with less SOM.

The area near the street showed a higher mass retention of contaminants than the area near the sidewalk. The effect of the interaction of horizontal sampling location and depth was significant for Na ($p = 0.05$), Ni ($p = 0.02$), Cu ($p = 0.03$), Zn ($p = 0.02$), and DOC ($p = 0.01$) (Fig. 5). At the sampling location near the sidewalk, reduced mass fluxes of Na, Ni, Cu, Zn, and DOC were observed between surface and the deep sampling points. At the sampling location near the street, the mass flux of the contaminants was further reduced, e.g., the above contaminants decreased between 63 and 78% from the surface to the deepest sampling point. At the sampling location near the street, the mass flux of the above contaminants decreased between 79 and 92% from the surface to the deepest sampling point.

The interaction of the sidewalk permeability and sampling depth had a significant effect on the DOC ($p = 0.04$) and Cu ($p = 0.07$) mass fluxes and the water flux ($p = 0.05$). An increase in permeability was associated with decreased fluxes in the open area of the tree pit. The water flux at the middle depth was similar in tree pits with impermeable and permeable sidewalks (Fig. 6) but different near the surface. This could indicate that a portion of the water that infiltrated the permeable sidewalk was not sampled at the surface; however, this water might have been sampled at the middle depth as water flowed laterally through the soil from the sidewalk towards the lysimeters in the open part of the tree pit.

The interaction of depth and the presence of a lawn had a significant effect on the mass fluxes of Na ($p < 0.01$), Cu ($p < 0.001$), DOC ($p < 0.01$), Ni ($p = 0.02$), Zn ($p = 0.02$), and Cr ($p = 0.08$). Similar

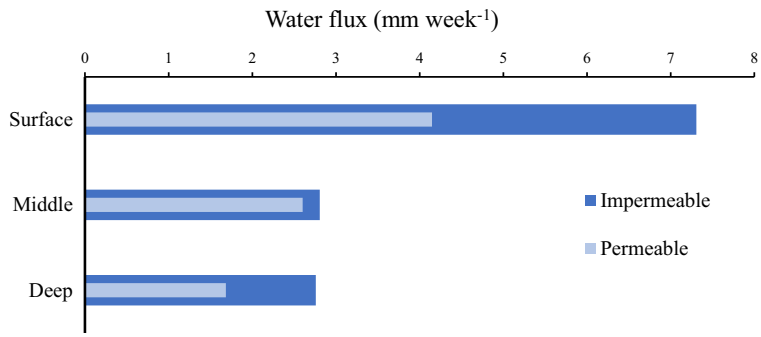

Fig. 6 Water flux in tree pits as related to sidewalk permeability and sampling depth

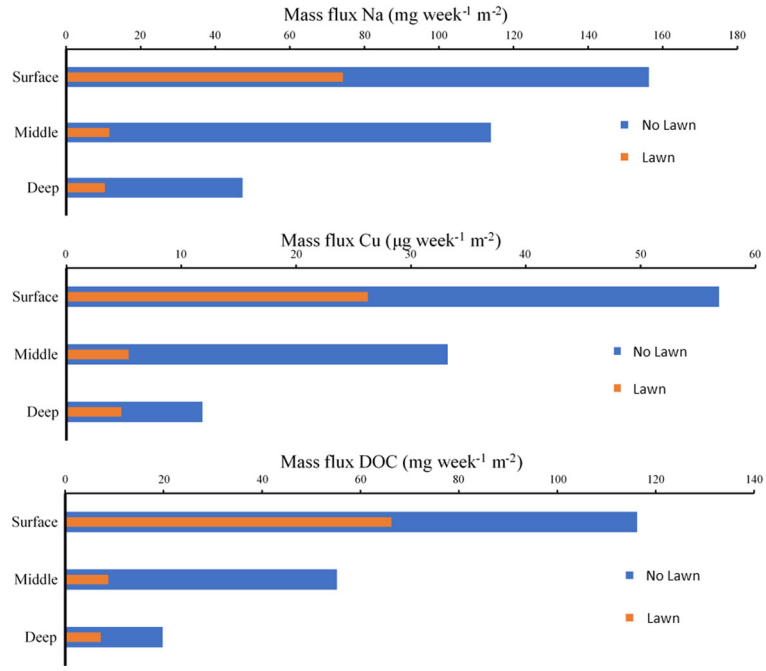

Fig. 7 Mass fluxes of contaminants in tree pits as related to presence of a lawn and sampling depth

to the trends displayed in Fig. 5, the mass fluxes of these contaminants decreased over depth for all tree pits. The flux of contaminants decreased more rapidly with depth in tree pits near a lawn (Fig. 7). This could be a result of a decrease in water flux in tree pits with a nearby lawn.

Table 2 shows that the presence of a lawn was associated with a reduction of over 61% of the water flux in the open part of tree pit ($p < 0.001$). This suggests that some runoff infiltrated into the lawn rather than flowing into the open part of the tree pit, thus suggesting an improvement in local infiltration and a decrease in runoff.

Pontoni et al. (2016) mentioned that although all dissolved contaminants follow the flow of water in soil, their velocity is not the same. Some metals are more strongly absorbed to colloids and move more slowly through zones with higher colloid concentration. Contaminant concentrations in the soil solution were affected by their solubility. The concentrations of more soluble contaminants such as Na, Ni, and DOC increased with depth even as water flux decreased with depth. Other contaminants, such as Cu and Zn, however, were more concentrated at the middle depth than at the deepest sampling point (Table 2).

4 Conclusion

In this study, all street tree pits reduced the mass flux of contaminants. The mass fluxes of Na ($p < 0.001$), Ni ($p = 0.04$), Cu ($p < 0.001$), Zn ($p < 0.001$), and DOC ($p < 0.001$) all decreased by more than 70% with depth. Tree pits with higher SOM showed a better capacity to decrease the mass flux of contaminants. For example, the mass flux of Na and Cu decreased with depth by 66% and 73% in tree pits with less SOM and by 87% and 86% in tree pits with more SOM.

Permeable surfaces near tree pits decreased the movement of contaminants and water through the open part of the tree pit. The increased local permeability likely increased infiltration, thus decreasing the volume of water collected by the stormwater drainage system. The mass fluxes and concentrations of contaminants varied with time. Higher values were observed after periods of disturbance in the system such as after the installation of the sampling equipment, early winter, and during spring thaw. However, these results were obtained in a period of less than 2 years. The long-term performance of tree pits as bioretention units must be evaluated.

The results of this study show that tree pits can indeed be used as bioretention units. As urban areas continue to expand, tree pits can play an important role

in making cities greener and healthier. The authors suggest that an analysis of the soil matrix could provide a better understanding of the tree pits as bioretention units. Moreover, since the results obtained in this study are representative of cities with climates similar to that of Montreal, Canada, validation studies should also be conducted in cities with different climates.

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