STREET TREE PITS AS BIORETENTION UNITS: ANALYSIS OF THEIR PERFORMANCE IN A RESIDENTIAL AREA OF MONTREAL, CANADA

By

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Abstract

Urbanization, increased surface impermeability, and climate change have resulted in changes in the quantity, intensity and quality of urban stormwater runoff. Urban stormwater runoff has ecological impacts. Generally, these impacts are related to the total and peak flow volumes, and the presence of contaminants in runoff. Low impact development (LID) techniques are emerging as alternatives to traditional stormwater management systems to mitigate these impacts. One such technique is to combine soil, plants and infrastructure in a bioretention unit. This technique involves the use of multiple smaller units spread across an area; street tree pits may be suitable as bioretention units.

The objective of this research was to analyze different designs of newly developed tree pits as bioretention units in the city of Montreal. These tree pits soil comprise the soil of the open part, where trees are planted, and the soil underneath the sidewalk. The two design factors were soil organic matter (SOM) content, and the permeability of the surrounding area (sidewalks and front lawns). A total of 24 tree pits were used in this study. The concentrations of trace metals and sodium were analyzed in soil solution and soil matrix. Using the estimated water flux mass flux of each contaminants was calculated.

The mean contaminant concentration increased from the surface to the deep sampling depths (e.g. 46% for Ni, and 18% for Cu) but taking into account the accompanying decrease in water volumes, mass flux of contaminants decreased with the increase in depth (e.g. 72% for Ni, and 81% for Cu). In addition, tree pits with higher SOM content presented a higher reduction of mass flux of contaminants than tree pits with lower SOM content between surface and deep

sampling depths. For example, tree pits with higher SOM content reduced the mass flux of Cr and Cu by 65%, and 86%, respectively, while tree pits with lower SOM content reduced the mass flux of Cr and Cu by 39% and 73% respectively.

Tree pits with higher SOM content presented higher concentrations of Cr, Cu and Pb in the soil matrix. For instance, in the soil matrix of tree pits with higher SOM, Cr and Cu concentrations were 19.9 mg kg^{-1} and 15.3 mg kg^{-1}, respectively, but were 17.4 mg kg^{-1} and 13.5 mg kg^{-1}, respectively, in tree pits with lower SOM. This corroborates the observed effect on mass flux of Cr and Cu. In addition, an overall increase of contaminants was observed over time. For example, the concentration in soil of Cr increased from 15.9 mg kg^{-1} to 20.0 mg kg^{-1} and of Ni from 11.9 mg kg^{-1} to 14.4 mg kg^{-1} representing increases of 26% and 21% after about 18 months of monitoring. The soil matrix contained approximately one third of the maximum permitted concentration of contaminants stipulated by the Canadian Council of Ministers of the Environment for Cr and Ni in residential and park areas.

High local permeability and high SOM bioretention units are recommended to mitigate the adverse effects of urbanization on runoff quality and quantity. In this study, tree pits with higher SOM retained contaminants better for all contaminants analyzed (except Pb). The increase in permeability of surrounding surfaces decreased the observed flux of water as well as the mass flux of contaminants observed in the open part of the tree pit. The reduced flux of water in the open part of the tree pit was likely a result of increased infiltration into the soil of the lawn and through the permeable sidewalk. The soil underneath the sidewalk is the same as, and contiguous with, the open area of the tree pit so it is designed to retain runoff contaminants. In this study, higher local permeability and higher SOM were both generally correlated with lower mass flux of contaminants and water. Tree pits can be used as bioretention units having their performance improved by increasing SOM and local permeability.

Résumé

L'urbanisation, l'augmentation de l'imperméabilité des surfaces et les changements climatiques ont entraîné des changements dans la quantité, l'intensité et la qualité des écoulements des eaux pluviales. Les techniques de développement à faible impact (DFI) émergent comme étant des alternatives possibles aux systèmes traditionnels de gestion des écoulements des eaux pluviales pour atténuer ces impacts. L'une de ces techniques est un système de biorétention dans lequel le sol, les plantes et les infrastructures sont combinés dans une unité de filtration. Cette technique implique l'utilisation de plusieurs petites unités réparties sur une zone donnée; les fosses d'arbres peuvent convenir à une unité de biorétention.

L'objectif de cette recherche était d'analyser différentes configurations de fosses d'arbre en tant qu'unités de biorétention dans la ville de Montréal. Les deux facteurs de configuration étaient la teneur en matière organique du sol (M.O.) et la perméabilité de la zone (trottoirs et pelouses) sur la performance des unités de biorétention. Au total, 24 fosses d'arbres ont été utilisées dans cette étude. La concentration de métaux lourds et sodium a été analysée dans la solution et la matrice de sol. Le flux d'eau a été estimé et utilisé pour estimer le flux de masse des contaminants.

La concentration moyenne en contaminants augmentait de la surface aux profondeurs d'échantillonnage (e.g. 46% pour Ni et 18% pour Cu), mais le flux massique de contaminants diminuait avec l'augmentation de la profondeur (e.g. 72% pour Ni, et 81% pour Cu). En outre, des fosses d'arbres avec une teneur en M.O. supérieure présentait une réduction plus importante du flux de masse de contaminants que les fosses arborées avec une teneur en M.O. inférieure entre les profondeurs d'échantillonnage de surface et profondes. Par exemple, des fosses d'arbres avec

une teneur en M.O. supérieure a réduit le flux de masse de Na et Ni dans 87% et 80%, respectivement, tandis que les fosses d’arbres avec une teneur en M.O. inférieure réduit le flux de masse de Na et Ni dans 66% et 62% respectivement.

Les fosses d’arbres avec une teneur en M.O. supérieure présentaient des concentrations plus élevées de Cr, Cu et Pb dans la matrice du sol. Par exemple, dans la matrice de sol des fosses d’arbres avec une teneur en M.O. supérieure les concentrations de Cr et de Cu étaient respectivement de 19,9 mg kg^{-1} et 15,3 mg kg^{-1}, tandis que dans des fosses d’arbres avec une plus faible teneur en O.M les concentrations étaient 17,4 mg kg^{-1} et 13,5 mg kg^{-1}, respectivement. Ceci corrobore avec l'effet observé sur le flux de masse de Cr et de Cu. De plus, une augmentation générale des contaminants au fil du temps a été observée. Par exemple, la concentration dans le sol de Cr a augmenté de 15,9 mg kg^{-1} à 20,0 mg kg^{-1} et la concentration de Ni de 11,9 mg kg^{-1} à 14,4 mg kg^{-1}, soit 26% et 21% d'augmentation après environ 18 mois de suivi, respectivement.

L'augmentation de la perméabilité locale et de le M.O dans les unités de biorétention sont recommandées pour atténuer les effets néfastes de l'urbanisation sur la qualité et la quantité des eaux de ruissellement. Dans cette étude, les fosses d’arbres avec une plus grande teneur en M.O. ont montré une meilleure performance dans la rétention des contaminants (sauf Pb). L'augmentation de la perméabilité des surfaces environnantes a diminué le flux de masse des contaminants prélevés dans la partie ouverte de la fosse. Le sol sous le trottoir fait partie de la fosse d'arbre et est conçu pour retenir les contaminants des eaux de ruissellement. Dans cette étude, l'augmentation de la perméabilité locale et l'augmentation de le M.O. dans le sol étaient corrèle avec une réduction générale du flux massique de contaminants et d'eau à travers la partie ouverte de la fosse d’arbre.

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Contributions of Authors

This thesis was written by the candidate following McGill University's Guidelines Concerning Thesis Preparation in manuscript format. It comprises a total of 8 chapters. There is one manuscript as part of this work. Dr. O. Grant Clark, as thesis supervisor, and Dr. Shiv O. Prasher, and Dr. Pierre Jutras as academic advisors and Dr. Maryam Kargar are co-authors of the manuscript. The manuscript (chapter 4) is under review for publication and has been used in this work with the authorization of the authors. Marcelo H. Frosi was responsible for all the field and laboratory work as well as the statistical analysis, data analysis, discussion of results and manuscript preparation and interpretation. Dr. O. Grant Clark and Dr. Maryam Kargar helped interpret the data and edit the manuscript and the thesis. Dr. Pierre Jutras and Dr. Shiv O. Prasher provided scientific advice about the data interpretation. The analyses were performed in the Environmental Soil Chemistry and Soil Nutrient Laboratory with the technical supervision and assistance of Hélène Lalande, and in the Soil Ecology Laboratory with the technical support of Hicham Benslim. This project was financially supported by the City of Montreal Transport Department and Large Parks, Greening and Mont-Royal Department.

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List of Abbreviations and Symbols

•	Statistical significance $p < 0.10$
*	Statistical significance $p < 0.05$
**	Statistical significance $p < 0.01$
***	Statistical significance $p < 0.001$
ABS	Acrylonitrile Butadiene Styrene
CCME	Canadian Council of Ministers of the Environment
Cd	Cadmium
CI	Confidence Interval
Cr	Chromium
Cu	Copper
DOC	Dissolved Organic Carbon
EC	Electrical Conductivity
FL	Front Lawn
HDPE	High-Density Polyethylene
ICP-MS	Inductively Coupled Plasma Mass Spectrometry
L.	Lower bound of 95% confidence interval
LID	Low Impact Development
Na	Sodium
Ni	Nickel
NS	Non-significant
O.M.	Organic Matter
Pb	Lead
SOM	Soil Organic Matter
Std. Dev.	Standard Deviation
TOC	Total Organic Carbon
U.	Upper bound of 95% confidence interval
WF	Water Flux
Zn	Zinc

Chapter 1. Introduction

1.1. Urbanization

Urban ecosystems are complex; they are comprised of human-made and natural features. Urbanization, or urban development, results from economic development and the evolution of regional and industrial infrastructure (Zhang et al. 2011). According to the United Nations (2014), urbanization has social, economic and environmental causes and implications. The increase in urban areas and their population is a matter of concern. These increases in urban areas and urban population present a challenge to urban planners as it leads to an increase in runoff, and water contamination.

1.2 Challenges of urbanization

1.2.1 Urbanization and population

Since the Industrial Revolution, the urban population has grown in contrast with rural population. Lyons (2014) stated that, globally, the number of cities with more than 1 million inhabitants has increased from sixteen to more than four hundred from 1900 to 2000. In the same period, the urban population increased over tenfold and will reach 67% of the global population by 2050 (Lyons 2014). For example, about 84% of Canada's total population lives in urban areas (Alberti 2016). According to the United Nations, the number of urban dwellers is expected to increase by 30% over the next 30 years (United Nations 2014). Increases such as these in urban population lead to increased urban development and expansion, creating challenges for city planners to deal with changes in land use and provide the resources needed by the populace.

1.2.2 Urbanization and climate change

Climate change is another issue which needs to be addressed in the urban environment. Cities are responsible for up to 70% of the total greenhouse gas emissions in the world (Sclar et al. 2013). Cities are also some of the areas most vulnerable to problems resulting from climate change, such as increasing temperatures, rising sea levels and extreme weather conditions (Malamis et al. 2016). An example of extreme weather is the change in rain patterns. The intensity and occurrence of heavy rain events are likely to increase in high and mid latitudes (Kirtman and Power 2014); in combination with increased impermeable surfaces as a result of urban development, the peak and total runoff also increase, augmenting the risk of flooding, and decreasing runoff water quality.

1.2.3 Urbanization and runoff quantity

Urban development traditionally involves the replacement of natural land cover by the expansion of roads, sidewalks, rooftops, and parking lots as well as lawns and parks (Finkenbine et al. 2000). This process decreases the permeability of the area and increases the total volume of runoff and the peak runoff rate during rain and thaw event, since less water is absorbed by the soil. More is directed to storm-water management systems. The increase in total runoff volume and peak rate of runoff ultimately increases the risk of flooding (Finkenbine et al. 2000).

1.2.4 Urbanization and runoff quality

Changes in land use due to urban development affect water (Tu 2013, Wang et al. 2008, Ren et al. 2003), and soil (Alberti 2016, Zhang et al. 2011), and generally a decrease in the quality of natural resources. According to Hartley et al. (2001), when the ratio of impermeability is higher than 60% of the total area in urban environment, the quality of surface water is reduced. Increased urban population also increases the demand for such resources. A decrease in the quality of urban water bodies is problematic because they are a source of water for consumption. Consequently, water quality problems increase the vulnerability of cities that rely mainly on surface water for their citizens' demands (Julie and Steven 2014).

Urban stormwater runoff is a major source of pollution for receiving waters (Davis et al. 2001). The source of the pollution in urban storm runoff is diffuse so it is called non-point source pollution. Urban runoff quality is affected by the complex interactions between atmospheric quality, urban land use types and intensity, surface composition and conditions, traffic types and intensity, municipal street cleaning practices, stormwater controls and precipitation. Urbanization tends to increase the concentration of pollutants such as trace metals, hydrocarbons, polychlorinated biphenyls, etc. (Torno et al. 1986). These contaminants, in particular trace metals, are interesting because of their potential toxicity, ubiquity and persistence (Davis et al. 2001).

1.2.5 Addressing the challenges of urbanization for stormwater management

According to Kamali et al. (2017), urban runoff is responsible for about 46% of the pollution of surface waters. In addition to its poor quality, urban storm runoff volume as well as peak flow are usually high. Thus, traditional stormwater management systems are often incapable of collecting and treating the stormwater before it is discharged into receiving waters (Givens et al. 2012).

It is necessary that city planners have information about the local urban runoff quality and quantity, so that they can devise strategies to manage it. However, getting this information requires extensive *in situ* studies due to the complexity of the urban environment. The urban environment is a mix of natural and “man-made” features, including buildings, streets, sidewalks, parking lots, and other infrastructure as well as green areas such as parks or treed boulevards.

Green coverage (e.g. trees, shrubs, etc.) improves urban water quality (Tu 2013). According to Oldfield et al. (2014), they mitigate storm water impacts and also reduce urban temperatures, and improve air quality. The use of trees and other plants is a common aspect of many low impact development (LID) practices. In this aspect, urban trees play an important function in creating better cities. However, the efficacy of trees depends on their health, which is limited by factors such as drought, salinity, potentially toxic metals, soil compaction, hypoxia and waterlogging (Mullaney et al. 2015).

Improving urban runoff quality and quantity is a main goal of LID. LID goals are to retain or regain local characteristics from pre-development stages of the land such as higher permeability and higher green coverage. Bioretention units and permeable pavements are two LID practices intended to decrease total runoff and improve infiltration. Using trees and vegetation in general, while expanding permeable surfaces, LID practices can mitigate the impact of urbanization and climate change, while improving the quality of urban life.

In light of the aforementioned problems, the City of Montreal and McGill University established a joint project to test the use of street tree pits as bioretention units. Following recommendations from previous studies conducted by these two partners, compost was added to tree pit soil to increase its organic matter (O.M.) content (Kargar, Clark, et al. 2015, Kargar et al. 2016, Kargar, Jutras, et al. 2015). The use of compost also aligns with the Montreal Waste

Management Master Plan to increase the recovery rate of waste materials to at least 60% by 2025 (Communauté métropolitaine de Montréal 2015).

The street tree pits were implemented in conjunction with a small-scale test of permeable pavement with the intention of further improving water quality and decreasing runoff.

1.2. Research objectives

Street tree pits were tested in the field as bioretention units to observe their affect on runoff water as measured from soil solution samples analyzed for contaminants (trace metals, DOC) and Na. The experimental factors were the amount of soil organic matter content and permeability of the surrounding area. Soil solution measurements were repeated over time at two horizontal locations in the open part of tree pits and at three depths. Soil samples were similarly repeated over time and depth. The hypotheses of this study were:

1. Does the soil organic matter content affect the concentration or the mass flux of contaminants in soil solution, and the concentration of contaminants in soil in the open part of the tree unit?
    a. H0: the organic matter does not affect concentration or mass flux of contaminants in soil solution or concentration of contaminants in soil in the open part of the tree unit.
    b. H1: the organic matter does affect concentration or mass flux of contaminants in soil solution or concentration of contaminants in soil in the open part of the tree unit.

2. Does the overall permeability of the area affect the concentration and the mass flux of contaminants in soil solution, and the concentration of contaminants in soil in the open part of the tree unit?
    a. H0: the permeability of the area does not affect concentration or mass flux of contaminants in soil solution or concentration of contaminants in soil in the open part of the tree unit.
    b. H1: the permeability of area does affect concentration or mass flux of contaminants in soil solution or concentration of contaminants in soil in the open part of the tree unit.
3. Are the concentration and the mass flux of contaminants in soil solution, and the concentration of contaminants in soil in the open part of the tree unit affected by time?
    a. H0: the concentration or mass flux of contaminants in soil solution or concentration of contaminants in soil in the open part of the tree unit is not affected by time.
    b. H1: the concentration or mass flux of contaminants in soil solution or concentration of contaminants in soil in the open part of the tree unit is affected by time.

Chapter 2. Literature review

2.1. Street tree pits

Street tree pits are sidewalk cut-outs or street side planting strips (Roman et al. 2013). Tree pit dimensions as well as soil properties might change greatly depending the local conditions such as sidewalk width, management practices, construction process, etc. One factor taken into consideration is the low interference with the structural needs of cities such as paved surface for transportation and pedestrians. Another factor is the capacity to sustain the life of the vegetation that is planted on its surface. One of the goals of street tree pits is to provide life support to the vegetation (e.g. trees) growing in the pit. Trees are, therefore, a key component of street tree pits.

2.1.1 Street trees

Street trees and green infrastructure provide a wide range of ecosystem services (Galenieks 2017, Berland and Hopton 2014, Hilde and Paterson 2014). Trees can mitigate adverse effects of urbanization on the environment. For instance, photosynthesis, transpiration and shade can attenuate urban temperature, improve air quality, reduce housing energy usage and fix atmosphere carbon (Dale and Frank 2014). According to Hilde and Paterson (2014), urban trees can provide wildlife habitat and, as part of green infrastructure, noise reduction, reduction of heat island effect, improvement of water quality and flood control. In addition, Berland and Hopton (2014) stated that stormwater runoff peak is affected by urban trees. Leaves and branches directly intercept and hold rainfall; a portion of the intercepted precipitation flows towards the soil. The interception and consequently decrease in flow intensity and quantity allows an increase in water evaporation, ground water infiltration and decreases the contribution of rainfall to runoff. (Elliott et al. 2018).

Street trees can improve the quality of life of urban population. According to Galenieks (2017) these improvements can range from improved mental health and memory, and decreased surgical recovery time. In addition, the practice of physical activity in urban areas with more trees positively reduces mood disturbances and boosts self-esteem (Pretty et al. 2007). The presence of natural elements, such as street trees, in urban areas reduces anxiety, blood pressure, mortality, physical inactivity, physician-assessed-morbidity while promoting physical activities, greater cardiovascular benefit and, improvement of mental health ultimately aiding the psychological and physiological restoration of population when compare to urban areas without street trees (Galenieks 2017). Roman et al. (2013) stated that the prevalence of childhood asthma decreased in urban areas with more street trees while contributing to create more livable neighborhoods. In addition to the improvement of urban population health, street trees can increase property value, improve economic development (Hilde and Paterson 2014).

The perception of the liveability of a city as well as the economic and environmental value of an urban area is affected by the presence of street trees. Street trees create more aesthetically pleasing and memorable areas (Galenieks 2017). This changes the perception of liveability of cities. According to Roman et al. (2013), street trees characterize many great boulevards and streets. The authors also state that street trees of business districts attract people and thus stimulate business. For instance, the economic and environmental value of public trees in the city of Montreal is around 648$ million (Jutras 2008). However, the benefits of urban forests, and thus urban trees, is mainly influenced by trees' longevity. One or more decades could pass, depending on the local conditions and evaluation methods, before the costs related with planting and maintaining trees is equal to the benefits they bring (Koeser et al. 2013).

Urban tree mortality is associated with environmental and social conditions. Examples of environmental conditions are: plant's species and size as well as planting location, and land use at the site (Roman et al. 2013). Other examples are nursery production and transplanting technique (Ferrini et al. 2000). Examples of social conditions are: vandalism, community involvement, and socioeconomical status of the neighborhood where the tree pits are located (Roman et al. 2014, Boyce 2010). The mortality of street trees is also related to the tree health (Roman et al. 2014, Kargar et al. 2013).

Trees' health in urban areas is negatively affected by several factors. Factors such as drought and waterlogging (Mullaney et al. 2015), extreme soil temperature and inadequate soil moisture during establishment (Koeser et al. 2013), construction damage, extreme weather events and, invasive pests and pathogens (Roman et al. 2013) reduce trees' health and longevity. In addition, some conditions may intensify the impact of these factors. For instance, the temperature in cities can be up to 10°C higher than nearby rural areas due to impervious surface covers, low vegetation coverage and anthropogenic heat sources (Dale and Frank 2014). The increase in temperature increases vapor pressure deficit and consequently lead to a greater atmospheric demand for water. In addition, water available to the root system might be limited because increased impermeable coverage (Dale and Frank 2014). Soil compaction excludes air and water from soil pore space while soil contamination with trace metals and deicing salt affects the availability of nutrients to the trees' root system (Kargar et al. 2013). Furthermore, the general limited rooting space available in tree pits intensifies the effect of other factors. With a smaller soil volume, the available water and nutrients is also reduced. Therefore, an increase in tree pits soil volume and an evaluation and monitoring of urban tree pit's soil quality are of high importance in order to improve urban tree's longevity.

Street trees are an essential part of the green infrastructure of cities (Roman et al. 2013). As part of green infrastructure, trees and soil improve the environmental quality particularly stormwater management of urban areas (Berland and Hopton 2014). Low impact development practices, such as rain gardens (also called bioretention system), are an example of urban green infrastructure (Jia et al. 2016). In this practice the vegetation (e.g. tree) is a used to reduce urban runoff generation while improving runoff quality.

2.2. Low impact development

Low impact development (LID) is an approach for land development that is intended to protect and conserve natural resources, such as urban waters, and decrease infrastructure costs. LID allows land to be developed while mitigating potential environmental impacts in a cost-effective way (United States Department of Housing Urban Development (Office of Policy Development and Research) 2003). This is possible by minimizing the urban development's impact on the quality and quantity of urban runoff, maximizing ecosystem services (Woods-Ballard et al. 2007). By providing these services, the use of LID practices results in an urban system that can manage urban runoff and its possible impact on the environment, while contributing to the enhancement of the environment. Low impact development is the term used in North America and New Zealand; in Australia, it is known as water sensitive urban design and in Europe, it is sustainable urban drainage systems in Europe (Fletcher et al. 2015).

LID makes use of existing natural systems or engineered systems to mimic natural processes (United States Congress House Committee on Transportation Infrastructure

(Subcommittee on Water Resources and Environment) 2010). The preservation or reestablishment of natural processes such as the self-regulatory properties of natural systems are an objective of LID practices. For example, LID systems can improve urban hydrology management through the reduction of runoff peak flow and, improve water quality through the control of the movement of pollutants (Liu et al. 2016), in a manner similar to natural systems. In natural systems, most of the rainfall is absorbed locally by the soil and vegetation (Figure 1), whereas in most urban systems with higher impermeable coverage, the locally absorbed rainfall is reduced, and the runoff generation increases (Figure 2).

LID systems are very versatile and can be blended with other urban design elements. According to the United States Department of Housing Urban Development (Office of Policy Development and Research) (2003), LID enhances stormwater management by increasing the proportion of permeable surfaces. Using LID techniques, it is possible to increase the proportion of permeable surfaces while keeping the elements of traditional stormwater design. This increase in permeable surfaces also enhances wastewater management. In cities with a combined sewage system (i.e. a system that conveys both sanitary sewage and stormwater), wastewater management is eased by increasing local absorption and treatment of stormwater which otherwise could overwhelm the system. Other urban design element is the road network. LID allows the necessary expansion of the road network for traffic and circulation while mitigating the possible increase of of impervious surfaces.

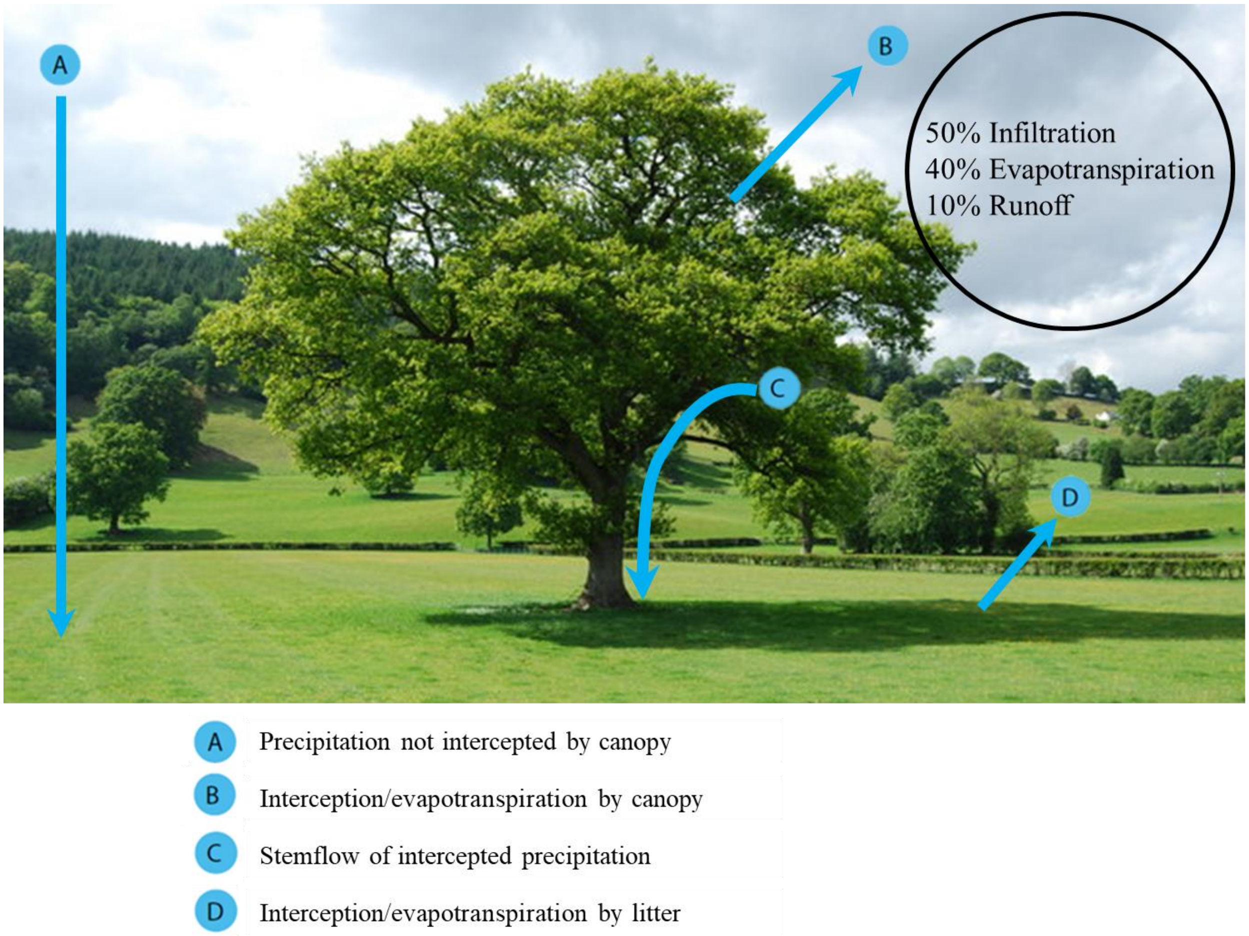

Fig. 1. Vegetation contribution to the water cycle. Adapted from Société Québécoise de Phytotechnologie (2018). Background photo © **John Haynes** free for use/modification under **cc-by-sa/2.0**.

The expansion of impervious surfaces increases urban runoff generation. For instance, in residential areas with 30-50% of impermeabilization of soil, runoff generation can increase three times as great as runoff in natural areas while in commercial areas runoff generation can increase more than seven times (Figure 2). Generally, urban runoff is collected by the traditional stormwater drainage systems. Theses systems can be separated or combined with the sewage system. The combined system predates the separate system (Iwugo et al. 2002). For instance, in the city of Montreal, two thirds of the system present is combined (Société Québécoise de Phytotechnologie

2018). According to Iwugo et al. (2002), in these systems where contaminants from wastewater and urban runoff are combined, it is unlikely that an increase in urban runoff can be accommodated.

Fig. 2. Modified water cycle in urban areas in (a) residential and (b) commercial areas. Adapted from Société Québécoise de Phytotechnologie (2018). Background photos without copyright under CC0 1.0 Universal (CC0 1.0).

Stormwater washes off and carries pollutants as it flows through the urban environment. The pollutants present in urban runoff water should be kept out of streams and rivers. It is possible to prevent pollutants from running into storm drains and ultimately to surface waters, whether treated or not, using low impact development practices which uses the local soil to trap these pollutants (United States Environmental Protection Agency (Office of Water) 2006). In urban areas, generally, the upper layers of soil have higher concentration of pollutants (Kargar et al. 2013). This happens as the upper layers of soil are the first to enter in contact with pollutants carried by urban runoff. As the upper layers sorption capacity is saturated, pollutants are trapped by the deeper layers. That is how the vertical distribution of pollutants is generally in urban soils.

LID is a possible solution for the volume of urban runoff and quality control. Traditional stormwater drainage systems and controls are designed to collect, convey, and discharge water quickly and efficiently (Bedan and Clausen 2009). However, these systems lose their design capacity or have their design capacity exceed over time (Miguez et al. 2013). Design capacity can be exceeded as the volume of stormwater increases with a constant expansion of surface impermeability and an increase in occurrence of severe weather conditions such as heavy rains. LID practices can address these limitations working as improvement in land development. The improvement is achieved as LID reduces runoff volume, peak flow and improves water quality. The awareness of the need of LID systems is becoming more common (Tedoldi et al. 2016). This increase ultimately leads to an increase in the use of LID systems. Tedoldi et al. (2016) reported that the use of such systems is becoming more common in urban areas as mitigation measurements to the increase of impervious cover and urbanization.

There is a wide variety of LID practices. They differ in scale, construction methodology, required maintenance and objectives. Some examples LID facilities are bioretention systems, permeable pavements, filter strips, vegetated swales, infiltration basins and soakaways (Tedoldi et al. 2016, Fassman 2010, Woods-Ballard et al. 2007). In this thesis, street tree pits with and without permeable sidewalks were used as bioretention system.

2.2.1. Bioretention systems

Bioretention systems are common LID facilities (Gülbaz et al. 2015). A bioretention system is made of mainly gravel, sand, soil, organic amendments (i.e. turf, mulch, compost), and vegetation (preferably indigenous) (Gülbaz and Kazezyılmaz-Alhan 2017) having between 0.7 m and 1 m of soil depth (Davis et al. 2009). A bioretention unit is located at a small depression area (< 2% of slope) to receive runoff water which must be totally infiltrated within 72h (Société Québécoise de Phytotechnologie 2018). The received water can either infiltrated locally (i.e. total infiltration system) or be redirect to the stormwater system (i.e. filtration system) if the units have an underlying impermeable layer and an underdrain connected to the stormwater system (Société Québécoise de Phytotechnologie 2018, Roy-Poirier et al. 2010).

A bioretention system is used to address nonpoint-source pollution (Davis et al. 2003). Such a system is often not designed as a single, large unit, but rather it is comprised of several smaller, integrated units spread across an urban area and designed to treat runoff as it flows through them (Chapman and Horner 2010). The contaminants in runoff are treated by ecological interactions between runoff and the local soil and plants as sheet flow before runoff is drained away from the area (Roy-Poirier et al. 2010). Several processes interact to reduce the pollutant load of the runoff, such as soil sorption, filtration, plant uptake, and microbial degradation (Gülbaz et al. 2015).

The design of a bioretention system depends on the project goals. For example, the maximization of infiltration can be obtained using soil with higher presence of sand and silt in its composition. This increase in infiltration would decrease runoff generation. In contrast, to improve the removal of contaminants, an increase of contact time between soil and runoff should be aimed (Société Québécoise de Phytotechnologie 2018). Using soil with higher proportion of clay can decrease the infiltration velocity, increasing soil-runoff contact time. Other design factors are the type and species of vegetation, the type and quantity of soil amendment, the unit size, etc.

Bioretention systems guidelines for the unit size follow different methods according to their location. For instance, Roy-Poirier et al. (2010) reported that in the USA there are five different sizing methods. These sizing methods take in consideration either the volume of runoff expected, the peak runoff, the load of pollutants expected, the percentage of total impervious drainage area, or local validated hydrological models. In Canada, no federal legislation oversees water discharge quality, rather the provinces are responsible to develop guidelines which can be more stringently adopted by municipalities (Roy-Poirier et al. 2010). The authors still point that there is little field experience in the guidelines current available in the country. A common point however, is that an increase in size of bioretention units, and overall permeable area, lead to an increase reduction in runoff generation and improvement in water quality. For instance, larger tree pits can receive and infiltrate more runoff than smaller sized tree pits. An additional use of an adjacent permeable sidewalk might further improve these effects.

Bioretention systems are good examples of how versatile LID practices are. Due to their small size, bioretention systems generally can blend in with urban infrastructure features (e.g. roads and sidewalks) making these systems adaptable to different urban conditions. According to

Trowsdale and Simcock (2011), not only their small size but also their aesthetic value allows this LID practice to blend into urban infrastructure while still able to accomplish stormwater management goals. In addition to facilitate stormwater management, other services provided by bioretention systems are noise reduction, wind coverage and, source of shading (Roy-Poirier et al. 2010).

Another beneficial characteristic of bioretention systems is the cost-efficiency associate to this practice. According to Roy-Poirier et al. (2010), the installation cost of a bioretention system, when compared with other pollutant treatment options, is very low; for instance, a bioretention facility for a 0.3 ha parking lot would cost $6,500 while an oil and grit separator could be three times more expensive. Thanks to bioretention system cost-efficiency and local services provided, these practices have become widely used (Davis et al. 2009).

As the use of bioretention system increases, the need of assessing its performance increases. Their performance is often evaluated by their impact on hydrologic cycle (peak flow, runoff volume received/retained), contaminants removal (heavy/trace metals, oil and grease, polycyclic aromatic hydrocarbons), total suspended solids removal, biological oxygen demand, pathogens removal, nutrients removal (nitrogen and phosphorus), as well as the effect on water temperature, pH, and dissolved oxygen (Roy-Poirier et al. 2010). The performance is also related to the local conditions as well as the specific characteristics of each system. The precision of the assessment can be improved when such conditions and specific characteristics are considered.

Bioretention units can reduce the hydrologic volume and flow peaks and delay peak timing at the field (Davis 2008). In a comparison between inputs and outputs of bioretention systems, Davis (2008) observed a reduction in the output peak flow volume and in some cases a completely retention of the generated runoff; The reduction observed varied between 49% and 58% while in 18% of rain events no outflow was observed. When outflow was observed, the bioretention system delayed the output peak flow. Output peak flow was reached between 5.8 and 7.2. times later than input peak flow. Similarly, a field study conducted by Hunt et al. (2006) reported that the annual estimated ratio between the volume of water leaving bioretention units versus the volume of runoff produced was 0.22 (i.e. 78% of water exfiltrated through the bioretention unit or was lost via evapotranspiration). The authors reported that this value range widely over the year being 0.07 during summer and 0.54 during winter. This could represent a seasonal effect on the performance of bioretention systems.

The removal of pollutants from runoff water is an important performance factor of bioretention systems. In laboratory tests of bioretention systems with different substrates, Liu et al. (2018) observed a removal of up to 85.6% of heavy metals from artificial precipitation events. The authors also observed that a higher pollutant concentration in the inflow and a higher antecedent dry period (i.e. time interval between precipitation events) improved the pollutants removal while the variation in temperature did not affect the performance of the bioretention systems tested. However this study used pollutants concentration to evaluate the performance of the bioretention units. Roy-Poirier et al. (2010) pointed that, to avoid misleading interpretation, mass removal of pollutants should be evaluated rather than concentration removal of pollutants. That is because mass pollutants removal takes in account changes in both concentration and volume of water. In a field study Davis et al. (2003) reported a removal in mass of pollutants greater than 97%. The authors also observed that the metal removal efficiency of bioretention units have little influence of runoff intensity, duration and pH as well as heavy metal concentrations in water. This last factor goes against of what observed by Liu et al. (2018). Davis et al. (2003) also pointed that, when comparing two different bioretention field units, a bioretention unit with coarser soil and lower mature vegetation might present poorer retention capacities. In their study, a more recently construct bioretention unit with coarser soil removed 43%, 70% and, 64% of copper, lead and zinc, respectively, in comparison with 97%, 95% and, 95% for the bioretention unit with less coarse and more mature vegetation.

The comparison of performances between bioretention units installed in different climate, geographical and local conditions is however complicated. Different rain patterns, temperatures, as well as urban, soil and vegetation condition might change results greatly. For instance, Roy-Poirier et al. (2010) state that there are still uncertainties on the implementation and performance of bioretention systems in cold climates, in special regarding snowmelt treatment and the use of deicing salt. In addition, studies in a field scale are necessary to confirm the findings of laboratory studies.

2.2.2. Permeable pavement

Permeable pavement is any paved surface intentionally designed to allow infiltration and thereby decrease the impermeability ratio of urban areas (Bean et al. 2007) while providing infrastructural needs (e.g. paved surface for transportation). Generally, permeable comprises a top porous surface, a base layer and a sub-base layer (Sañudo-Fontaneda et al. 2014). This design results in high surface infiltration rates and reduces runoff quantity, delays peak flow and reduces peak runoff rates. In addition, this practice can provide local storage, treatment and recharge of urban drainage.

The potential use of permeable pavement is mainly related to the top layer which enters in contact directly and firstly with the runoff. This first layer can be construct using a wide variety of material such as porous asphalt, porous concrete, cement brick, ceramic brick, sand base brick, and shale brick (Zhang et al. 2018). According to (Zhang et al. 2018), each material has different interactions with urban runoff. While some materials might eliminate or improve removal efficiency, other materials can increase pollutant concentration in runoff as the material is degraded or damaged.

The variety of the designs for the top layer also affects the potential use of permeable pavements. Some examples of permeable pavement designs are: porous concrete, interlocking permeable concrete pavers, concrete grid pavers, and plastic reinforcing grid pavers infilled with gravel or grass (Bean et al. 2007). Depending on the design, permeable pavement can be used in parking lots, fire lanes, walkways, and driveways to decrease surface runoff (Kamali et al. 2017). However, permeable pavements tend to clog over time. Each design has different limitation regarding clogging over time. Kamali et al. (2017) found that surfaces covered with paving blocks often clog after six or seven years, as coarse sediments fill the gaps between the blocks. Bean et al. (2007) reported that clogging can be prevented by regular maintenance.

2.3. Soil organic matter

The presence of organic matter can improve soil physical properties. SOM decreases the influence of soil particle size distribution in soil properties. Generally, water conductivity is positively correlated with an increase in coarseness in soil texture. However, increased texture coarseness is also correlated to poorer water retention. SOM can stabilize structural aggregates which improves soil macro-porosity while decreasing soil compaction and increasing water-holding capacity (Kargar, Jutras, et al. 2015). For example, Wesseling et al. (2009) amended different texture soils with 10% of organic matter. They observed that the total available water in soil increased between 144% in soils with a slightly coarse texture and 434% in soils with a very coarse texture. This increase in available water might improve the life span of vegetation as drought is one of the main adversities that urban vegetation faces.

The increase of SOM in urban areas is commonly assumed to improve soils ability to adsorb contaminants such as trace metals and deicing salt (Wuana and Okieimen 2011). One possible option for SOM amendment is compost. Compost is the biological stable, humified organic matter end-product of the microbial oxidation of raw sources such as green wastes (Kargar, Clark, et al. 2015). According to Bolan et al. (2014), humic acids present in compost have binding effects on metals such as Cd, Pb, Cu and Cr. Compost improves soil capacity of retention of contaminants by raising soil pH, cation exchange, complexation, sorption, the presence of phosphorus and aluminum compounds, and other inorganic minerals or a combination of them (Kargar, Clark, et al. 2015).

The amendment of SOM would result in an increase of dissolved organic carbon (DOC). DOC can be defined as an ensemble of organic molecules of different sizes and structures able to pass through a filter with a pore size of 0.45 μm (Koopmans and Groenenberg 2011). The two main groups of low-molecular weight hydrophilic compounds of DOC are humic acids and fulvic acids. Koopmans and Groenenberg (2011) report that concentration DOC positively correlated with dissolved Ni and Cu concentrations. Cu concentrations increase with DOC concentration due to Cu’s high affinity to dissolved organic compounds (Gartzia-Bengoetxea et al. 2009). Chahal et al. (2016) reported that, when using compost as a source of SOM, compost acts as source of observed Cu levels for the first rainstorm events and in later events, as a sink. This could be explained by the different fractions that form DOC. These fractions have their own capacity and affinity to bind metals. According to Pontoni et al. (2016), the increase in SOM in the soil could increase the mobility rather than the binding capacity of the soil by binding trace metals to colloids which are mobile in the soil profile.

Field scale studies should be conducted to assess the effect of SOM in the performance of street tree pits as bioretention system. In addition, a field scale study can improve the performance assessment precision of this system as local conditions are intrinsically part of a field scale study. The results can be used as guidelines for decision making and strategies of city planners. The performance of bioretention system has additional uncertainties in cities with cold climate due to occurrence of snowmelt and the use of deicing salt (Roy-Poirier et al. 2010). Therefore, cities such as Montreal are key locations to conduct field scale tests of bioretention systems.

Chapter 3. Methodology

To test the performance of street tree pits as bioretention units, soil and water samples were collected from the open part of tree pits in a residential area. These samples were analyzed for their contaminant concentration, observed sample volume and their relationship with the distinctive design factors of the tree pit. As a continuation of the study of Kargar (2015), the trace metals analysed in this experiment were Na, Cr, Ni, Cu, Cd, and Pb as well as DOC. The experimental treatments were the soil organic matter (SOM), the permeability of the adjacent sidewalk and presence/absence of front lawn. Tables 1 and 2 in section 3.4 present the experimental design for each tree pit. Due to design constraints, the experiment replicates were unbalanced.

3.1. Study site description

The experiment was conducted in the Hochelaga-Maisonneuve neighborhood of Montreal, QC, Canada, on Viau and St. Clement streets between 45°33'22.6"N 73°31'43.6"W and 45°33'37.0"N 73°32'19.7"W. Montreal has four distinct seasons. The summers are warm to hot while winters are cold and snowy (Environment and Natural Resources Canada 2017). The average daily temperature and monthly precipitation during the course of this study ranged from -9.7ºC in January to 21.2ºC in July, and from 62.7 mm in February to 96.4 mm in November, respectively. The sampling of soil solution started on 24 October 2016 and ended on 26 August 2017. Soil solution samples were collected every two weeks except during the winter season while the soil was frozen (December 2016 – May 2017). Soil matrix sampling occurred in three different moments, in January 2016, December 2016 and July 2017. The adjacent sidewalk is made of concrete. Some of the tree pits have porous concrete while other have impermeable concrete. The open part of the tree pit did not receive a top layer of mulch.

3.1.1. Street tree pits

The study was conducted using expanded street tree pits built into the sidewalk as experimental units. Standard street tree pits in Montreal (Figure 3) are generally 1.5 m long and 1.5 m wide, and 1 m deep, for a total of 2.25 m^3 of soil volume. The soil underneath the adjacent sidewalk is compacted and not available for root growth.

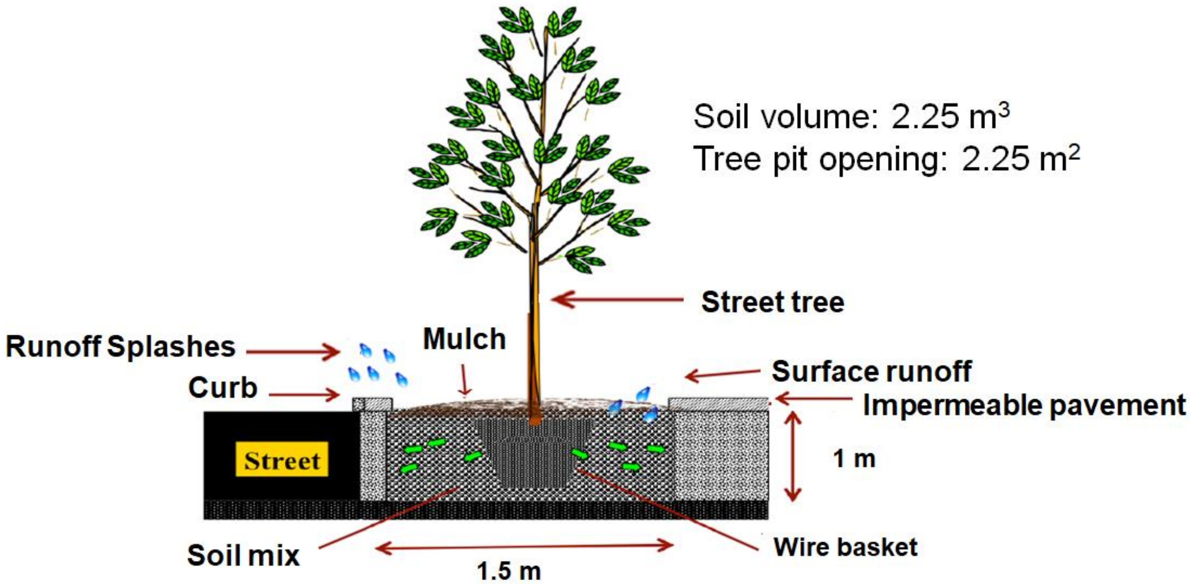

Fig. 3. Standard street tree pit in Montreal. Adapted from Kargar (2015).

The expanded tree pits are 3.0 m long, 1.5 m wide and 1 m deep (Figure 4) for a total of 4.5 m^3 of soil volume for the open part of the tree pit (i.e. that which is not covered by a sidewalk). The sidewalk was excavated, and uncompacted soil was used to fill the space beneath it, resulting in 3.15 to 3.78 m^3 of additional soil volume. Thus, the total volume available for root growth was between 7.30 and 8.30 m^3. The tree pits and the adjacent sidewalk were built during spring 2015. In the same period, the designed soil mix was used to fill the excavated area (i.e. soil and different amounts of organic matter). The trees were transplanted between September and October 2016. 36 tree pits were built separated in two areas. In this study, 24 of 36 available tree pits were used. 14 of 24 tree pits had impermeable sidewalk and the presence of nearby lawn. The 12 tree pits not used had the same configuration, thus, the addition of these units would increase the cost while not improving as much the quality of the data.

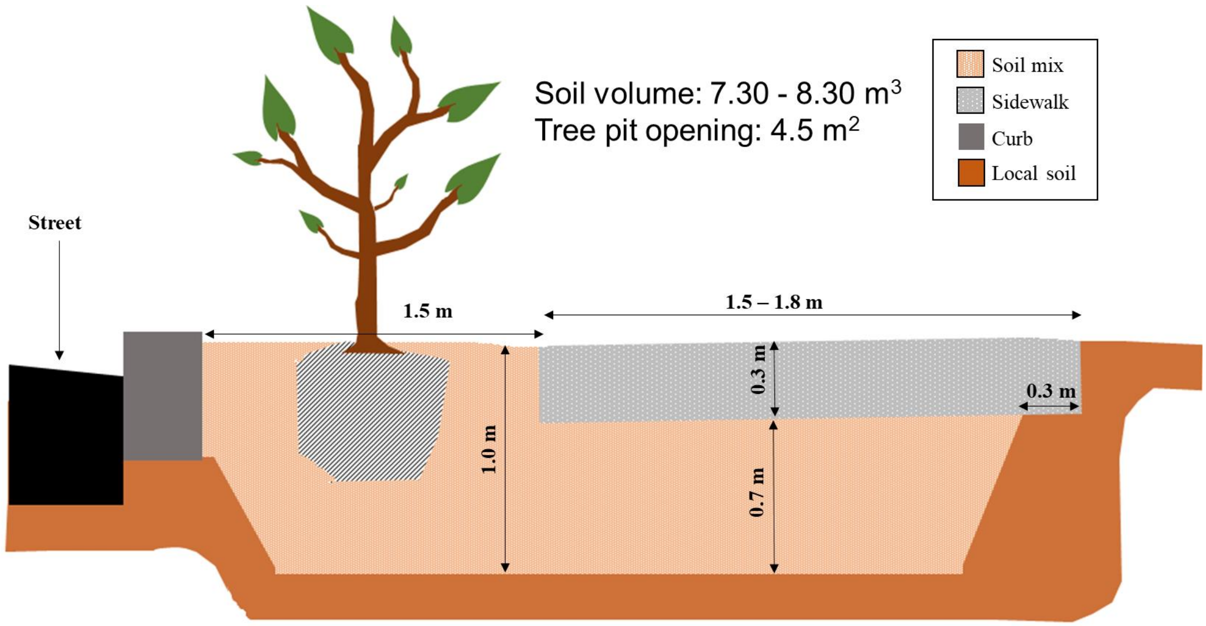

Fig. 4. Expanded street tree pit depicting the excavated area as well as the available soil for root growth.

The location of each tree pit can be seen in Figure 5. In the first area, 12 tree pits of 13 were used. Four of them are located on Saint Clement Street and nine on Viau street between Saint Clement and Saint Catherine streets. In the second area, 12 tree pits of 23 were used. All tree pits in this area are located on Viau street between Ontario and Rouen streets. As a residential area, most of the tree pits have housing buildings nearby. One tree pit in the first area has a deactivated gas station nearby while two tree pits are in the edge of a park in the second area.

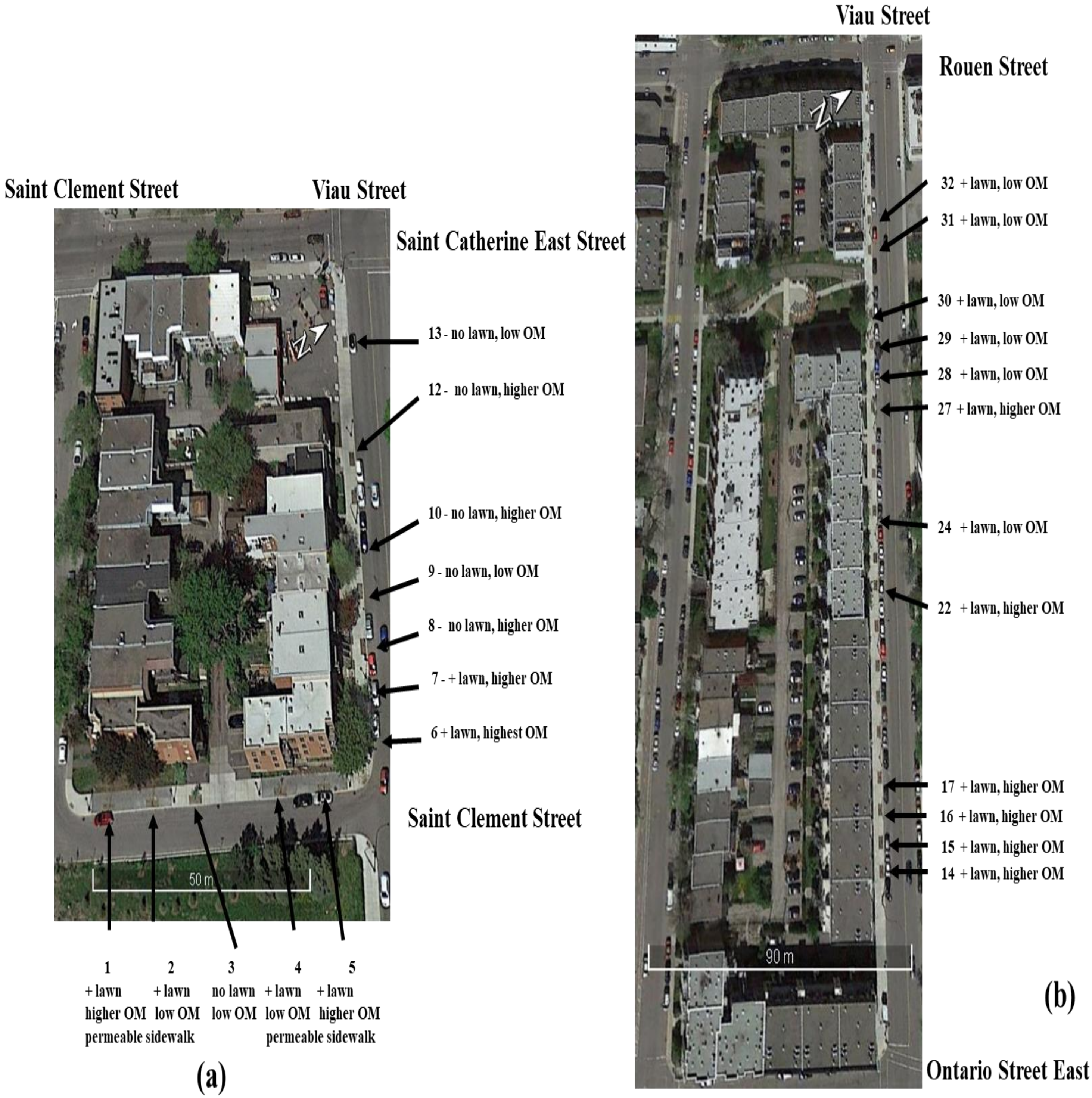

Fig 5. Tree pit location and configuration in the two nearby areas (a) and (b). Adapted from Google Earth Pro 7.3.2.5491 (2017)

3.2. Soil solution

3.2.1. Zero-tension lysimeter

Soil solution samples were collected using zero-tension lysimeters (Figure 6). These instruments were specially designed to sample the soil solution that moves through saturated soil (and some unsaturated flow) by the influence of gravity (Radin Mohamed et al. 2013, Thompson and Scharf 1994).

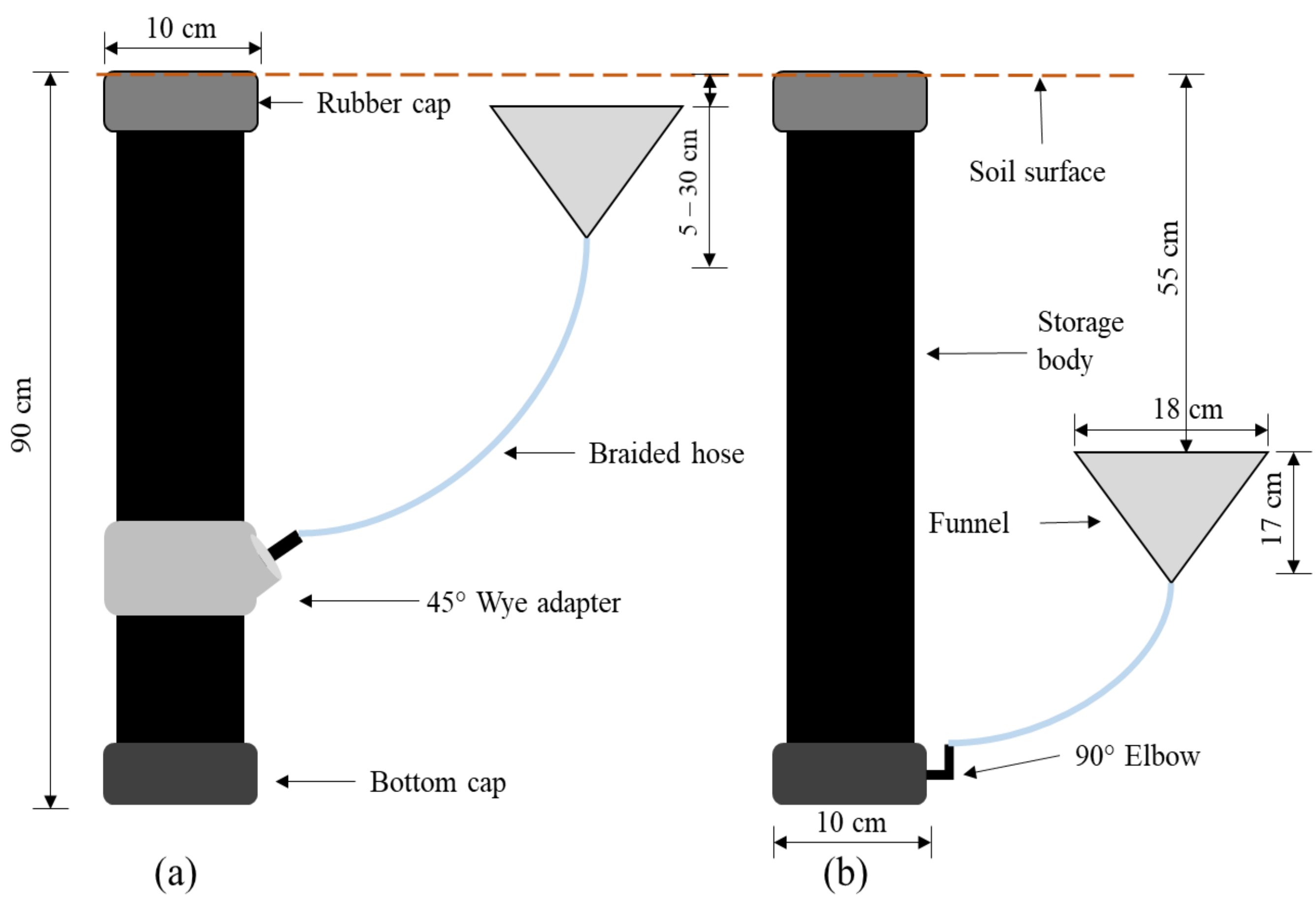

Fig. 6. Schematic diagram of lysimeter components. (a) for sampling from surface and middle depths, and (b) for sampling from bottom depth

The storage body was 0.9 m tall, 0.1 m in diameter and made of acrylonitrile butadiene styrene (ABS) pipe, with an ABS cap at the bottom and a rubber cap at the top. Each storage body had a 1.90 cm adapter connected to a hose and a funnel. Due to space limitation, two types of adapter were used, either a 45° Wye or 90 °elbow. The adapter type was determined by the depth on which the funnel was installed. Lysimeters with a funnel near the surface (≈ 5 cm of depth) and middle (≈ 30 cm of depth) had a 45° Wye adapter (Figure 6-a) while lysimeters with a funnel installed at the deep level (≈ 55 cm of depth) had a 90° elbow adapter (Figure 6-b). Vinyl hoses with a nylon braid were used to connect the funnel to the adapter.

The funnels had a top diameter of 18 cm, stem length of 7.2 cm and total height of 17 cm and were made of polyethylene. The funnels were lined with a nylon screen filled with silica sand that had been sieved (> 2 mm, mesh 10) and washed with deionized water. In order to prevent the funnels from moving during and after installation, high-density polyethylene (HDPE), 15-cm-diameter perforated tubes were used to support the funnels at the required depth (Figure 7). These tubes were filled using the soil presented in the tree pits.

Fig. 7. One of two lysimeter sets in tree pits as sampling units.

The lysimeters were installed in sets of three, with one at each of the aforementioned sample depths (5, 30 and, 55 cm) (Figure 7). These depths represent the sampling depth of soil solution. Two sets of lysimeters were installed in each tree pit one set closer to the street and one set closer to the sidewalk. (Figure 8).

Fig. 8. The layout of lysimeter installation in tree pits.

3.2.2. Soil solution sampling

Soil solution samples were collected from the lysimeters every 14 days. Every second week, the soil that covered the storage body was removed and the area surrounding the rubber cap was cleared to prevent any soil particles from falling into the storage body. A vacuum hand pump was used to collect and transfer all the available soil solution sample into a HDPE bottle, labelled according to each lysimeter. The sample bottles were placed on ice in a cooler for transport to the laboratory. The total volume sampled was noted and the excess sample not needed for analysis was discarded on the street. The tubing of the hand pump was then rinsed with deionized water to prevent any cross-contamination between samples. The sample bottles were acid washed and rinsed with double deionized water in laboratory between each sample round.

3.2.3. Pre-treatment of soil solution samples

The soil solution samples were pre-treated in the laboratory within 24h after collection. During the pre-treatment phase, samples were filtered using 0.45 µm nylon membranes and a vacuum filtration apparatus (Figure 9).

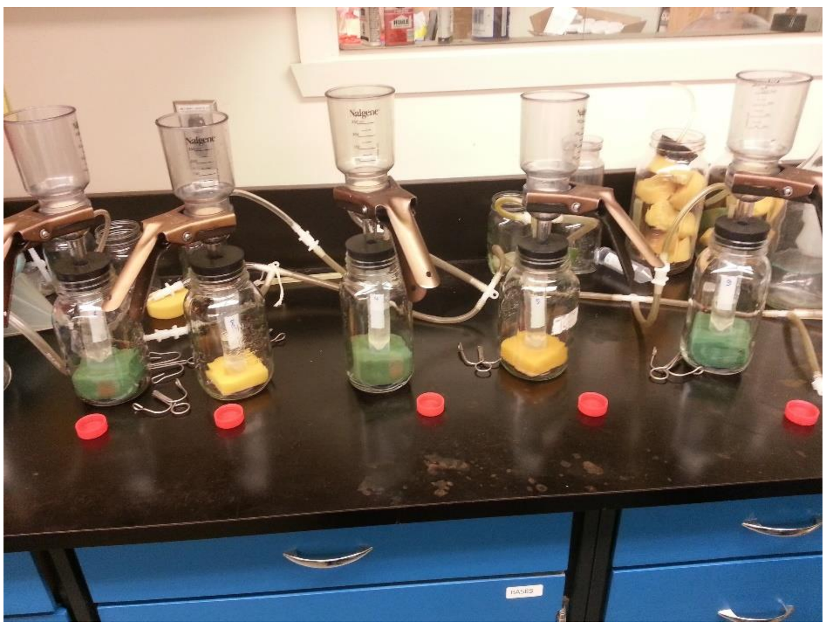

Fig. 9. Vacuum filtration apparatus.

This apparatus is composed of a collecting container, plastic core filter head with an outlet port to create the negative pressure, plastic core plate, and a cylinder funnel with a scale that is held together to the plastic core filter head using a metal clamp. Fifty (50) mL Falcon tubes are placed inside the collecting container where the filtered solution is deposited. The filtering membranes are placed on the plastic core plates. One membrane was used per sample, which was discarded after used. Each part of the vacuum filtration apparatus that had contact with the sample was rinsed properly with deionized water between each sample filtration. Before starting sample filtration, about 10 mL of each sample was filtered and discarded to prevent any remaining deionized water interfering with the final sample. The filtration process lasted until no more unprocessed sample was left or until the Falcon tube was filled with 50 mL of the soil solution sample. From the filtered sample, about 10 mL were separated and acidified with concentrated nitric acid and stored in the refrigerator (approximately at 4° Celsius) until the chemical analysis was performed. Blanks were also prepared for each round of filtration by passing double deionized (nanopure) water through the same pre-treatment process. More information on this method can be found in the work of Hendershot et al. (2008)

3.2.4. Soil solution analysis

After the maximum possible sample volume was filtered (up to 50 mL), any remaining sample was separated into two fractions for pH and electrical conductivity (EC) measurements. Soil solution pH was measured directly on the unprocessed sample using a pH meter (Accumet Research, AR 10, Fisher Scientific, Hanover Park, Illinois, USA) and its liquid-filled polymer body electrode (Fisher Scientific, Accumet, Hanover Park, Illinois, USA). The pH meter needs to be calibrated and standardized with pH buffers of 7.00 and 4.01 pH at room temperature. Electrical conductivity was measured to determine the amount of electricity that passes through the sample within the cell of the (CDM 83 conductivity meter, Radiometer, Copenhagen, Denmark) which will be directly related to the concentration of salts dissolved in the solution. The cell used to measure EC was rinsed between samples with the sample being analyzed 4 to 5 times, discarding the measured sample until a stable measurement was reached.

Trace metals and major nutrients were measured on the acidified samples, blanks and quality controls via inductively coupled plasma mass spectrometry (ICP-MS) equipment (Varian 820 MS, represented by Analytik-Jenna, Germany) (Hendershot et al. 2008). The quality controls used were from Sediment studies 104/107 (samples 3, 4, 5, 7 and/or 8) provided by the Environment Canada Proficiency Testing program. This equipment is also equipped with a collision reaction interface for measuring Cr, Arsenic, Selenium and Iron.

Dissolved organic carbon (DOC) was measured using a Sievers Innovox total organic carbon (TOC) analyzer (General Electric Power and Water, Water and Process Technologies, Boulder, Colorado, USA). The quality controls were made from the certified standard 1000 TOC ppm made by SCP Science Company.

3.3. Soil

3.3.1. Soil sampling

Soil was sampled three times: in January 2016, prior to the installation of the lysimeters in the tree pits, December 2016, about three months after installation; And July 2017. Soil samples were collected manually using a helical hand auger (Pansu et al. 2001). The samples were taken from the open area of the tree pit in the corner away from the locations of the lysimeters (Figure 10). After clearing the surface of any debris, samples, were collected by inserting the helical auger into the soil until the collecting compartment was totally buried and thenremoved. The compartment had 20 cm.

Three depths were sampled, 0 – 20 cm, 20 – 40 cm and 40 – 60 cm approximately. Subsamples were taken from two locations in each tree pit and combined in one labelled plastic bag for each depth of each tree pit.

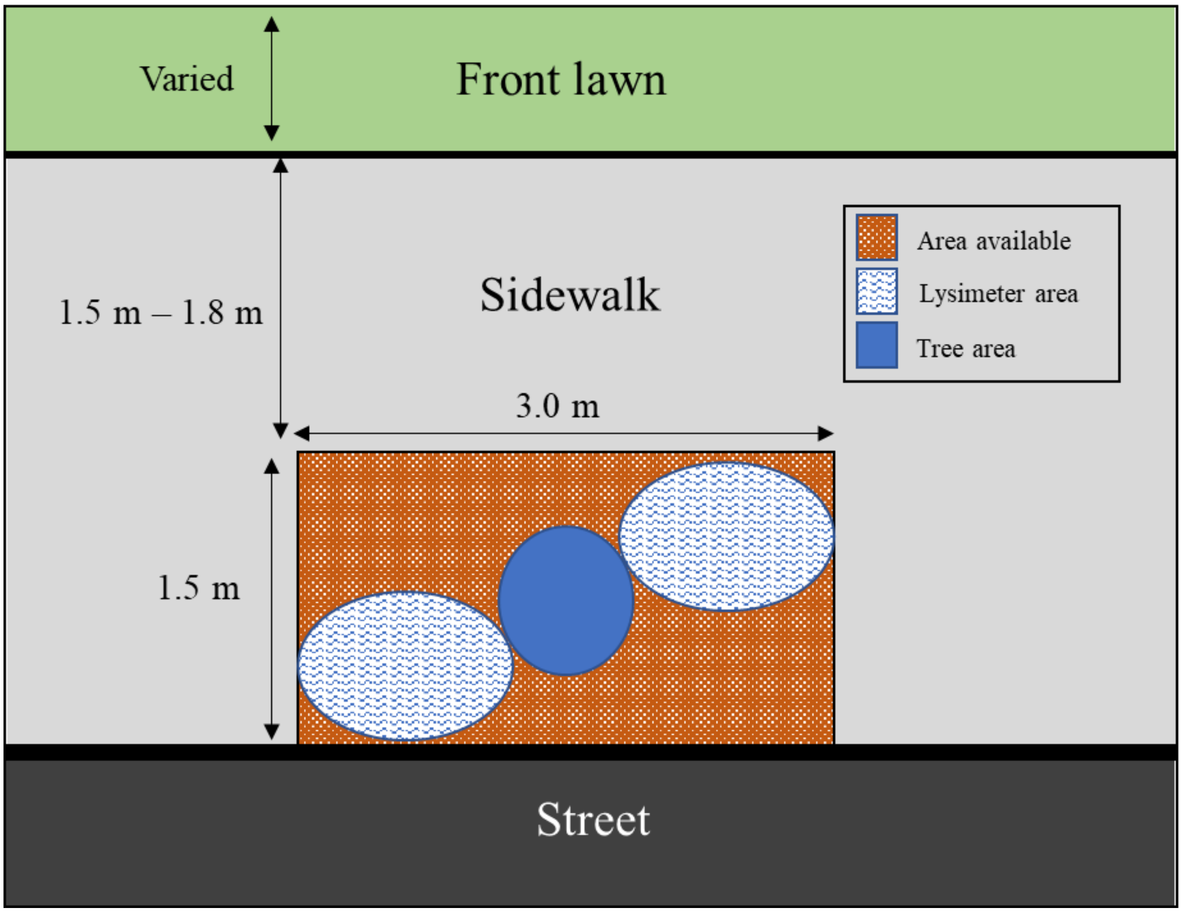

Fig. 10. Sub-sampling points in tree pits

3.3.2. Pre-treatment of soil samples

Before laboratorial analysis, the soil samples were pre-treated to improve their stability (storage life), the separation of aggregates and homogeneity (Pansu et al. 2001). The soil samples were air-dried at room temperature and sieved through a 10 mesh (2 mm) screen. The particles trapped in the sieve (> 2 mm diameter) were discarded. The fines were weighed and saved for analysis.

3.3.3. Soil samples physical analysis

Soil texture analysis were carried using the pipette method as described by Taubner et al. (2009). This method is based on the sedimentation process and the Stokes' equation. The equation follows the assumptions that all the particles are of spherical shape, the velocity is constant with laminar flow (Reynolds' number <1), no interaction occurs between particles and between particles and vessel wall, and no interaction occurs between particles and the liquid. Therefore, it is possible to separate particles in equivalent diameters such as silt and clay. At a given depth and time, an aliquot of the settling sample suspension is taken. This aliquot corresponds to a select particle diameter and is used for calculations on the mass base.

3.3.4. Soil samples chemical analysis

Soil chemical analysis were carried as follows: a portion of the air-dried fines of each sample was separated, ground with a mortar and pestle (0.160 to 0.200 g), and transferred to 15-mL digestion tubes. Two millilitres of nitric acid – trace metal grade (Hendershot et al. 2008) were added to digestion tube. The acidified samples were left overnight. The samples were then heated in a block heater at 130° C for 5 h together with blanks and quality control samples. Blanks were prepared using empty digestion tubes and 2 mL nitric acid. Quality control samples were from Sediments study 98, sample 5 provided by the Environment Canada Proficiency Testing program. After digestion, the samples were cooled. and transferred into labelled 50-mL Falcon tubes. The Falcon tubes were filled with nanopure water to 50 mL and capped. Falcon tubes were slowly inverted 3 to 4 times and left to settle overnight. The solubilized metals were then measured in the clear supernatant using ICP-MS equipment (Varian 820 MS, Analytik-Jenna, Germany).

3.4. Data analysis.

The data obtained from soil and soil solution samples were analyzed using the Mixed Procedure of the SAS™ statistical software version 9.4 (SAS Institute Inc., Cary, NC, USA). Each tree pit was considered as an experimental unit. The observed values were natural-log-transformed to improve the comparison of the estimates generated by the model. Correlation analysis was also conducted on the residual errors. The calculation of the Pearson Correlation Coefficients for each combination of different dependent variables was carried using SAS™ in the procedure CORR. The experimental unit of the study was each expanded street tree pit with a soil volume minimum of 7.30 m^3 compared to standard tree pits with 2.25 m^3. The experimental design employed repeated measures over time and depth. For the soil solution, there were also repeated measures with horizontal location, as each experimental unit was sampled in two locations (i.e. two sets of lysimeters in each tree pit). The independent variables were the SOM, presence of a lawn and the permeability of the sidewalk. SOM in the tree pit soil was tested by loss-on-ignition (Schulte et al. 1991). Tree pits were grouped into two categories according to their SOM. Eleven tree pits were considered to have soil with "lower" O.M. content ($< 5\%$ w/w) and 13 were considered to have "higher" O.M. content ($> 5\%$ w/w) (Table 1). The configuration and the replicates of the experimental units are presented in Table 2.

Table 1. Organic matter content of soil samples from tree pits and average of each group.

Lower O.M.		Higher O.M.	
Tree pit (ID)*	**Measured organic matter (% w/w)**	**Tree pit (ID)***	**Measured organic matter (% w/w)**
2	1.91	1	6.39
3	2.99	5	5.90
4	2.94	6	13.5
9	2.54	7	6.72
13	3.56	8	9.17
24	4.55	10	6.53
28	2.91	12	6.51
29	3.84	14	5.93
30	4.03	15	6.85
31	4.80	16	5.83
32	4.70	17	6.26
		22	5.54
		27	5.67
Average of organic matter content (%)	3.52±0.95	Average of organic matter content (%)	6.98±2.16

*Tree pit ID = individual identification of tree pits.

The sampling depth for the soil solution correspond to the depth of the lysimeter funnels: 5 cm, 30 and, 55 cm. For soil samples the sampling depths were 0 – 20, 20 – 40, and 40 – 60 cm. The two horizontal sampling locations for the soil solution were close to the sidewalk and close to the street. Soil solution was sampled every 14 days, weather permitting, mostly during the autumn, spring and summer. Soil solution sampling started in October 2016 and 12 sampling events were considered in this study until the end of August 2017. Soil samples were collected in January 2016, December 2016 and July 2017. Only surface samples were collected in January 2016 because the soil was frozen.

The dependent variables were the concentrations of Na, Cr, Ni, Cu, Cd, Pb in the soil solution and soil matrix. The concentration of DOC in the soil solution was also analyzed. Water flux was estimated using the sample volume of each lysimeter, normalized to one square meter, and the period normalized to one week. The mass flux of contaminants was estimated by multiplying the total mass of each contaminant sampled by the water flux.

Table 2. New configurations for experimental units.

Config. #	Lawn	Sidewalk	SOM*	Number of tree pits (n)	Tree pit ID#*
1	Absent	Impermeable	Low	3	3, 9, 13
			High	3	8, 10, 12
2	Present	Impermeable	Low	6	24, 28, 29, 30, 31, 32
			High	8	6, 7, 14, 15, 16, 17, 22, 27
3	Present	Permeable	Low	2	2, 4
			High	2	1, 5
Total				24	

*SOM = Soil Organic Matter. Low = SOM < 5% w.w., High = SOM > 5% w.w.
** Tree pit ID = individual identification of tree pits.

Connecting statement I

In the first two chapters, the research questions and information available in the literature were presented. Chapter 3 briefly explained the methods and equipment used to answer the research questions presented. In chapter 4, the performance of the street tree pits as bioretention units will be assessed as their effect on urban runoff and soil solution properties.

Chapter 4. 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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4.1. Abstract

The quantity, intensity and quality of urban stormwater runoff are changing as consequence of the urbanization and the climate change. Low impact development (LID) techniques (e.g. bioretention system) 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 contaminants 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 mm week^{-1} 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 are advised. Street tree pits have a higher potential as bioretention units to locally mitigate some of the impacts of urbanization. City planners could consider the use of street tree pits as biorientation units to help the management of urban runoff.

Keywords: Low impact development, bioretention system, street tree pit, permeable pavement, trace metals, de-icing salt.

4.2. 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 surfaces also change 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). Willuweit et al. (2016) stated that, in Ireland, climate change could increase monthly runoff by 30% during the winter season.

LID is an approach that also minimizes the impact of urbanization on the environment (Jia et al. 2016, Bedan and Clausen 2009), especially changes in runoff (Zahmatkesh et al. 2014). LID 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. Some practices comprise multiple instances of small-scale infrastructure. These structures 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. 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 aesthetic as well as functional value. This makes the deployment of such units possible in varied 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 plant 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 design might have different potential of use as bioretention units.

Most studies have focused on only one LID practice and few studies have followed their interactive effects (e.g. bioretention and permeable pavement) (Bedan and Clausen 2009). 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.

4.3. 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, Canada and McGill University. The mean daily temperature ranges from -9.7ºC in January to 21.2ºC 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 SOM (< 5% or > 5% w/w), sidewalk (permeable or impermeable) and lawn (presence or absence). Horizontal sampling location (near curb or near sidewalk) and sampling depth (surface, middle or bottom) (Figure 11), and sampling intervals were considered.

Soil solution samples were collected every two 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 of length, 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. 11. The layout of lysimeter installation in tree pits.

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 (Figure 11). 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 (Figure 12).

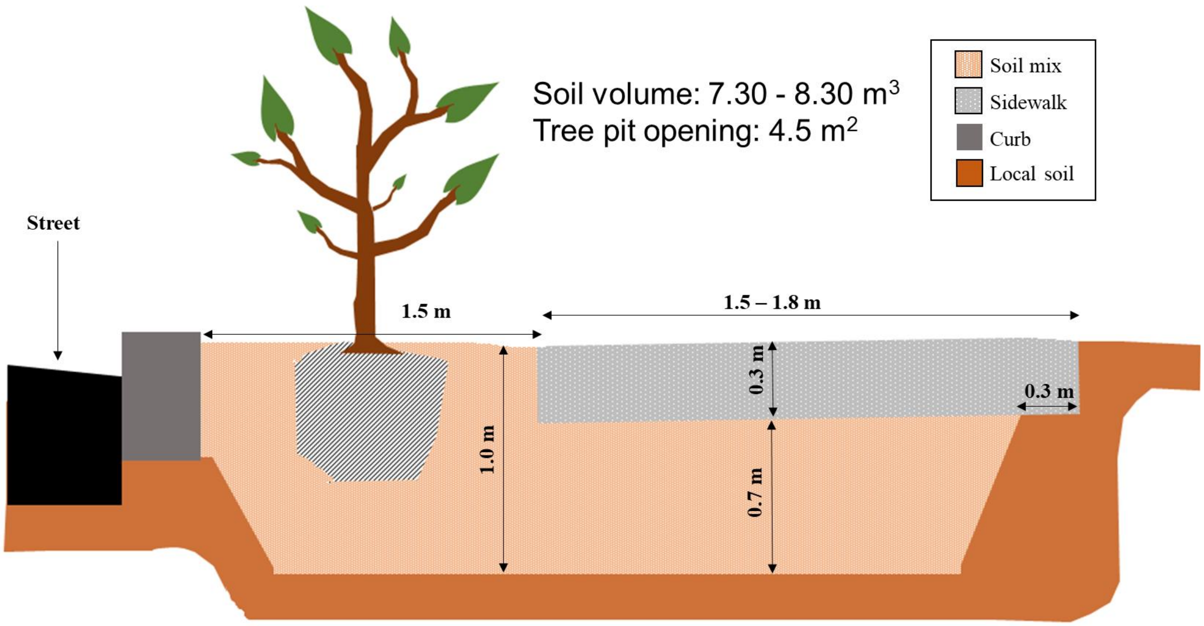

Fig. 12. Expanded street tree pit depicting the excavated area as well as the available soil for root growth.

Soil solution samples were collected from the lysimeters using a vacuum hand pump (Figure 11) and stored in labelled, high-density polyethylene bottles and placed in an ice box for transport. 50 ml of each sample was filtered using a 0.45 µm nylon membrane. In the case where unfiltered sample remained, pH was measured using a pH meter from Accumet Research and a liquid-filled electrode with a polymer body (AR 10, Fisher Scientific, Hanover Park, Illinois, USA). Electrical conductivity was measured with a CDM 83 conductivity meter (Radiometer, Copenhagen, Denmark). 10 mL was 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). ICP-MS 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 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 area to one square meter, and the period normalized to one-week. The mass flux of contaminants was estimated by multiplying the total mass of each contaminant sampled 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.

4.4. Results and discussion

The mean concentration of each contaminant, partitioned by treatment factor, can be seen in Table 3. The calculated water flux and estimated mean mass flux of each contaminant partitioned by treatment factor can be seen in Table 4. 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.

SOM had no statistically significant effect on the contaminant concentrations. However, 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. The increases in concentration, however, could have resulted from a decrease in the water volume. Table 4 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}.

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 could indicate that most contaminants come from the sidewalk, such as de-icing salt or weathered building materials. Chromium concentration, on the other hand, was higher near the street which could indicate that most Cr 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 of contaminant concentrations, 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 samples. Table 4 shows that water flux through the open tree pit was significantly reduced by the presence of a lawn ($p < 0.001$), 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.

Table 3. 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	8.0	**10.4**	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	**9.6**	11.8	7.9	**9.6**	11.6	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	**0.66**	0.83	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	**16.1**	20.8	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*** < 0.05), ** (***p*** < 0.01), *** (***p*** < 0.001), NS (non-significant)

Table 4. 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	**22.0**	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 sidewalk	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 event		**			**			**			**			**			**			**			**	
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 May 2017	683	**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)

0

The effects of time are shown in Table s 4 and 5. 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 were 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 considered possible sources of contaminants. Furthermore, abrasion of sidewalk surface by snow removal activity might also be a source 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.

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 concentration and volume of sample. In this study, mass flux was used as it takes in consideration mass of pollutants as well as the effect of area and time. The concentrations of contaminants (Table 3) were negatively correlated to the volume of sample, represented by the water flux (Table 4). For instance, the concentrations of all pollutants (except Pb) increased with depth while the water flux decreased with depth.

The variation in mass flux of contaminants could be directly related with the variation of water flux observed. The correlation matrix (Table 5) 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 with the mass flux 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 in the same proportion leading to a weaker correlation. The water flux is directly related to the precipitation. The frequency of precipitation events, therefore, influences the correlations values. Depending on the frequency, there might be a bigger effect of either dry or wet atmospheric deposition 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 snow thawing. Wet deposition is a result of precipitation scavenging effect on the atmosphere.

The variation in mass flux of certain contaminants might cause variation in others (e.g. interaction between contaminants, similar source of contamination, etc.). The concentrations of Ni and Cu are reported to positively correlate to DOC concentration. (Koopmans and Groenenberg 2011). Ni and Cu as well as Na mass fluxes, presented a high and positive correlation with DOC mass flux (Table 5). This indicates a similar behaviour of DOC, Cu, Na, and Ni. According to Gartzia-Bengoetxea et al. (2009), Cu concentration is expected to increase as DOC concentration increased due to Cu high affinity to dissolved organic compounds. This expected trend between

Cu and DOC, however, did not occur with depth (Table 3). 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 Cu affinity to colloids and the formation of a front of colloids as described by Pontoni et al. (2016).

Table 5. 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*** 1719	1 1721						
Cr	0.692*** 1719	0.688*** 1720	1 1721					
Ni	0.856*** 1711	0.919*** 1712	0.816*** 1712	1 1713				
Cu	0.844*** 1720	0.911*** 1721	0.765*** 1721	0.946*** 1713	1 1722			
Zn	0.800*** 1720	0.873*** 1721	0.682*** 1721	0.903*** 1713	0.912*** 1722	1 1722		
Pb	0.267*** 1720	0.202*** 1721	0.690*** 1721	0.441*** 1713	0.353*** 1722	0.325*** 1722	1 1722	
WF	0.661*** 1726	0.638*** 1721	0.872*** 1721	0.820*** 1713	0.746*** 1722	0.701*** 1722	0.730*** 1722	1 1728

[†] DOC = Dissolved organic carbon.
[††] WF = Water flux.

The mass flux of DOC, Zn and Cu presented a different behavior over time than other contaminants. Zn 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 contaminants fluxes. In addition, the highest water flux was during the first sampling. This fact could have been a consequence from the irrigation of the recently transplanted trees. Lysimeters were installed while trees were transplanted. After transplanting trees, the city of Montreal irrigates the plants to increase their acclimation and survival 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 contaminants mass and water flux was between May and June 2017. Except for DOC, the mass fluxes of all contaminants also decrease between the two sampling periods. This indicates an influence of the post-winter period. 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 decrease in water flux was not the same of the contaminants from 7.3 L to 4.7 mm week^{-1} (36%). Na, as a very soluble metal, is quickly washed out of the tree pit after winter. Cr and Cu reduction, on the other hand, is much slower. This could result from a high accumulation of the contaminants in soil during the winter period. These contaminants are gradually flushed out as the times passes after the winter.

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 higher increase in mass flux (59%, 43%, 47% and 44%, respectively) than the increase in water flux (39%). Thus, the increase in mass flux of these contaminants is not directly proportional to the increase in water flux. For the street side, the lower water flux observed could indicate little or no effect of sidewalk runoff. In this case, most of water is assumed to come from direct rain and water splashed from the street. The splashed effect in the study area can be considered low due to the fact that 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 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 has less influence of the sidewalks' permeability. When analysed the interaction effect of sidewalk permeability and horizontal sampling location, 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 sidewalk, 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 mm week^{-1} to 1.98 mm week^{-1}. The changes in mass flux of contaminants might not be only related 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 influence of SOM in the concentration and mass flux of contaminants varied depending on the analysis. The analysis of the singular effect of SOM on the concentration and the mass flux of the contaminants was not statistically significant. However, the interaction effect of the 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 and water fluxes ($p < 0.08$). All of those mass fluxes decreased more rapidly with depth in tree pits with higher SOM (Figure 13). As displayed in Figure 13, 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. SOM 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 the contaminants mobility.

The interaction effect of SOM and sampling depth affected the water flux. Similarly to the effect in the mass flux of contaminants, water flux decrease 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 (Figure 13), it is not the whole story. Water flux in tree pits with lower SOM decreased about 53% 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% by 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 observation (Kargar et al. 2016).

SOM appears to have affected moisture retention in tree pit's soil. The water fluxes of 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 (Figure 14). In tree pits with less SOM, the water fluxes were 5.0 mm week^{-1} and 2.4 mm week^{-1}. The differences in water flux between the two points were 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 S.O.M retained 1.5 mm week^{-1} more than tree pits with less SOM.

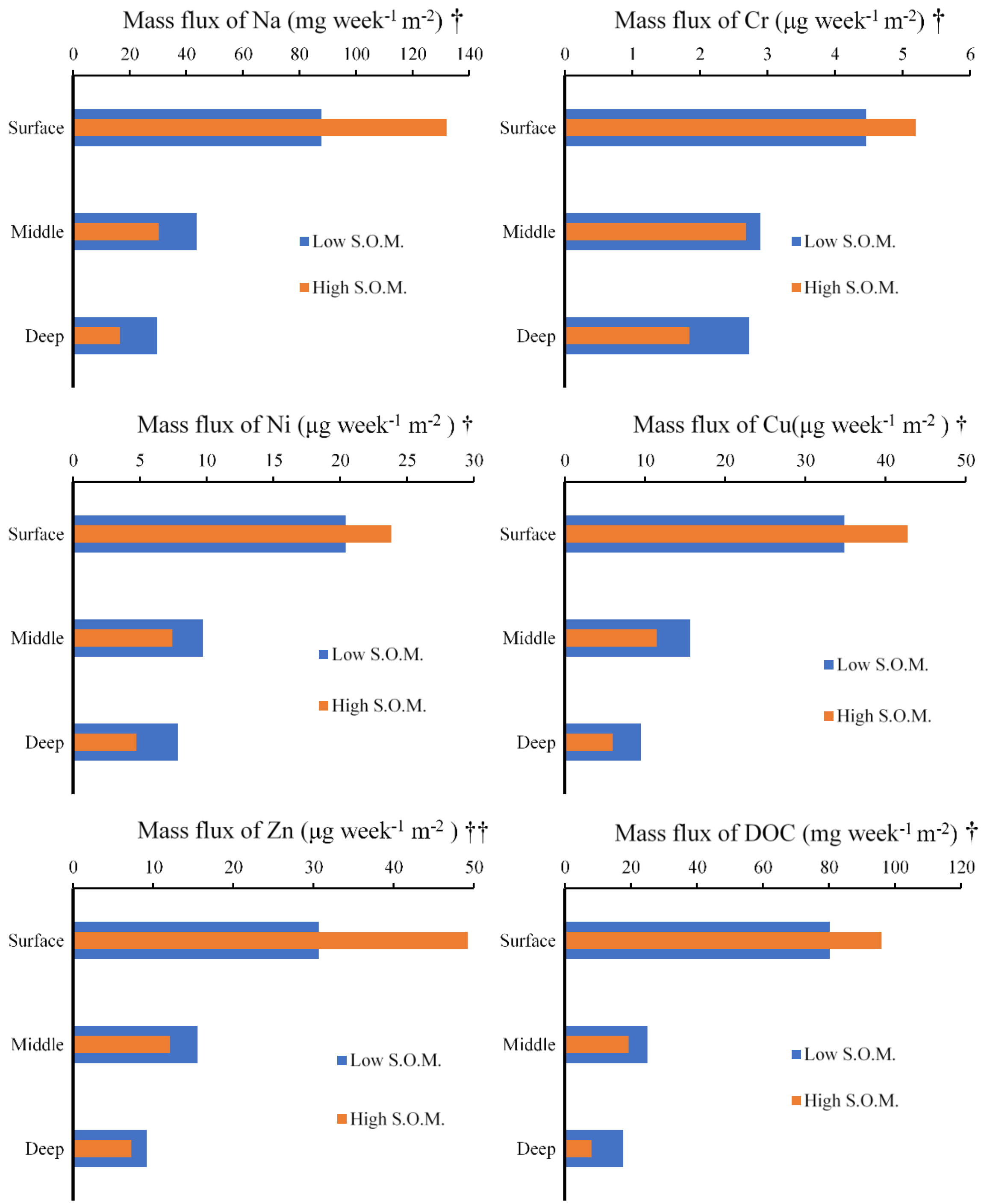

Fig. 13. Mass fluxes of contaminants in tree pits as related to soil organic matter and depth. † statistical significance $p < 0.05$, †† statistical significance $p < 0.01$

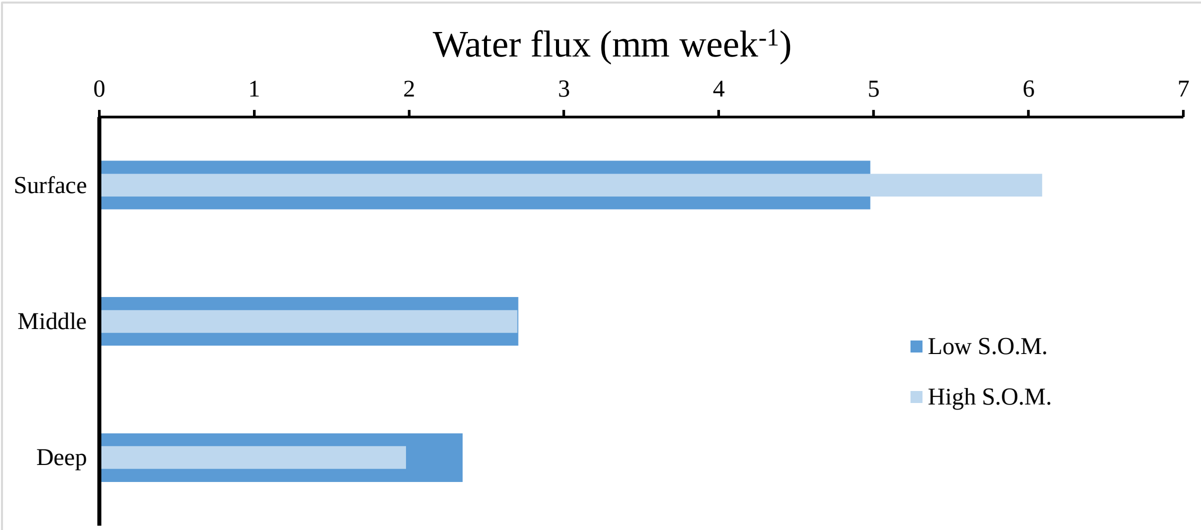

Fig. 14. Estimated water flux means in tree pits for the interaction of soil organic matter content and depth.

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) (Figure 15). At the sampling location near the sidewalk, reduced mass fluxes of Na, Ni, Cu, Zn and DOC was 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.

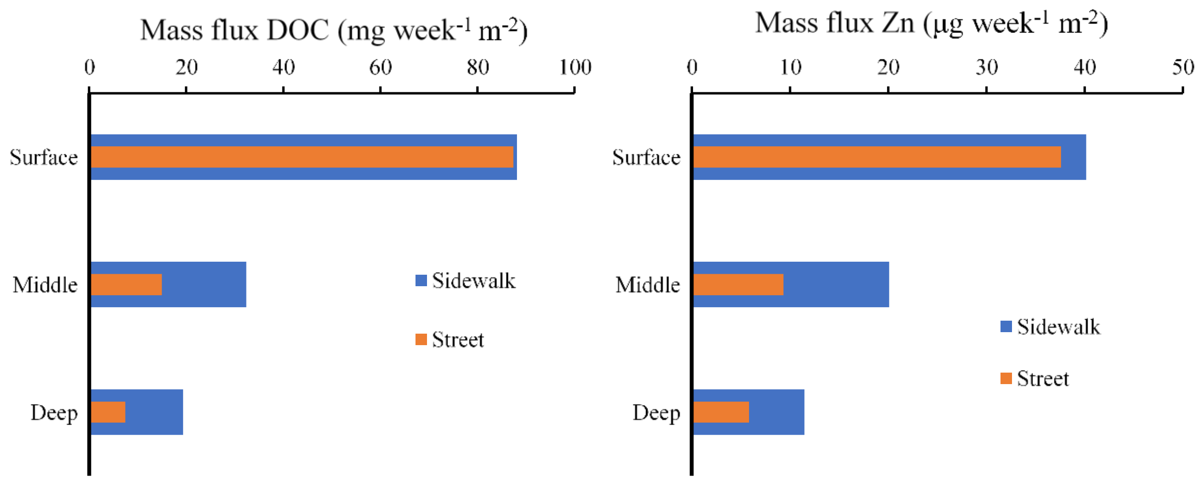

Fig. 15. Mass fluxes of contaminants in tree pits as related to horizontal sampling location and sampling depth.

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 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 (Figure 16) 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.

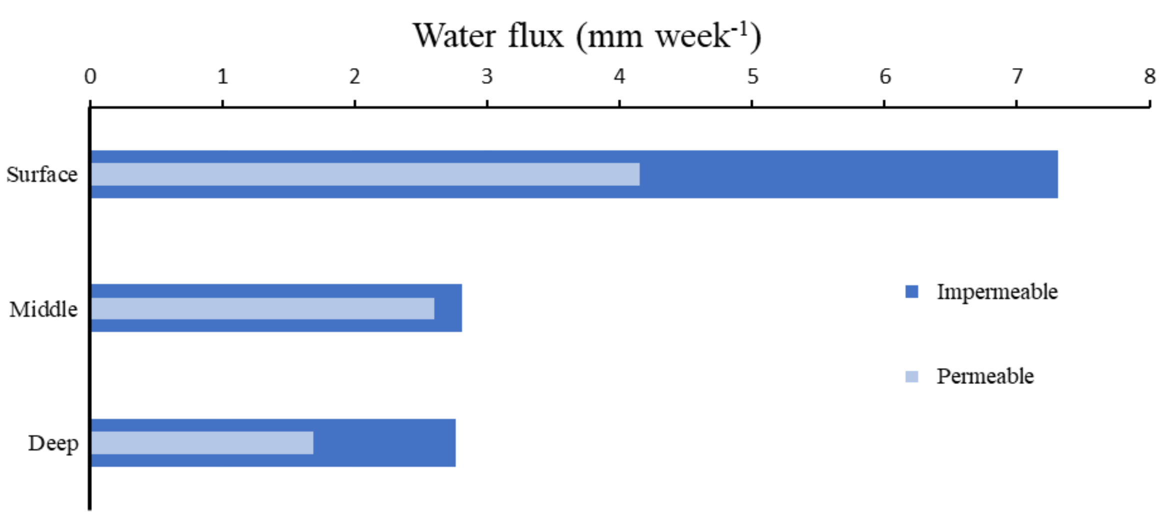

Fig. 16. Water flux in tree pits as related to horizontal sampling location and sampling depth.

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 to the trends displayed in Figure 15, 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 (Figure 17). This could be a result of a decrease in water flux in tree pits with nearby lawn.

Since the interaction of presence of lawn and sampling depth was not statistically significant different for water flux, the singular effect of lawn presence on the water flux should be considered. Table 4 shows that the presence of lawn represents 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 lawns rather than flowing into the open part of the tree pit. Thus, suggesting an improvement in local infiltration and expected 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 3).

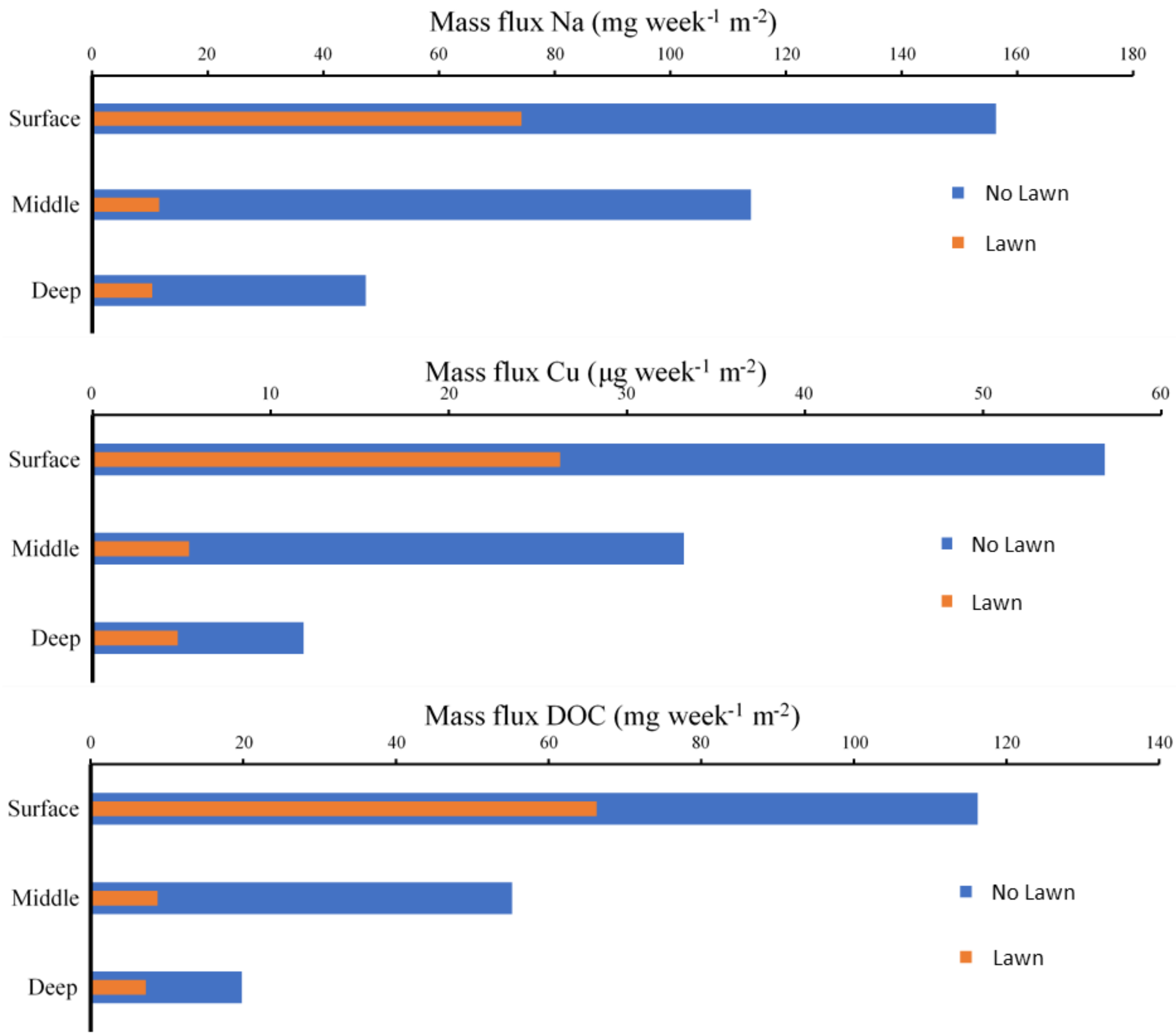

Fig. 17. Mass fluxes of contaminants in tree pits as related to presence of front lawn and sampling depth.

4.5. Conclusion

In this study, all street tree pits reduced mass flux of contaminants retaining urban 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 in the open part of the tree pit. The increased overall 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 course of a single season. The long-term performance of tree pits as bioretention units must be evaluated. The soil matrix was not analyzed in this study but could provide a better understanding of the tree pits as bioretention units.

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Chapter 5. Soil analysis

In chapter 4, the quality of the soil solution and its interactions with the different properties of the tree pits and their surroundings were discussed. The analysis of the soil matrix could provide a better understanding of the fate of the contaminants. This chapter includes a discussion of soil texture and the presence trace metals and Na in the soil.

The methodology used to determine soil texture is described in chapter 3. All tree pits' soils were classified as sand or loamy sand, implying high infiltration rates and low susceptibility to compaction. The complete soil texture analysis for each tree pit is available in Appendix H.

Soil sampling occurred on three different dates. On January 2016, before the installation of the equipment in the tree pits; frozen soil prevented the team from obtaining middle (20 – 40 cm) and deep (40 – 60 cm) samples. On November 2016 and July 2017 all three depths were sampled. During the instrumentation of the tree pits, on October 2016, underground infrastructure prevented the team from installing the instruments in two of the selected tree pits. Those two tree pits were switched for similar ones (27 and 32) but, as a result, there are no initial soil samples for those two tree pits.

Table 6. Soil texture analysis. Mean, standard deviation, minimum and maximum values are presented in percentages of total dry mass. See fig.5 for a list of tree pit I.D. numbers.

SOM	TreePit ID	n	Texture	Mean ± Std. Dev.	Minimum	Maximum
Low	**2, 3, 4, 9, 13, 24, 28, 29, 30, 31, 32**	**11**	**Sand**	89.5 ± 1.86	86	92
			Silt	5.3 ± 1.35	3	8
			Clay	5.3 ± 1.10	4	7
High	**1, 5, 6, 7, 8, 10, 12, 14, 15, 16, 17, 22, 27**	**13**	**Sand**	87.0 ± 1.58	84	90
			Silt	6.0 ± 0.81	4	7
			Clay	7.0 ± 1.29	5	9

The unbalanced data set precluded the use of SAS Proc Mixed on the entire data set. Two separate statistical analyses were therefore conducted, one comparing only the surface samples from all three dates (Table 7) and the other comparing the samples from all three depths for the second and third dates (Table 8). The estimated mean concentrations of Na and metals (Cr, Ni, Cu, Zn, Cd, and Pb) in the soil matrix were partitioned by treatment factor (SOM, sidewalk permeability, presence of lawn) and time

The tree pits with higher SOM (>5% w./w.) presented an overall higher concentration of Na in the surface sampling point than tree pits with lower SOM (<5% w./w.) (p = 0.03). This might have been due to adsorption of Na by the higher SOM It might also have been related to the fact that Na can be present in O.M. sources. In the study of Kargar, Jutras, et al. (2015), Na was found in compost used as O.M. source.

Table 7. Estimated mean and lower and upper confidence intervals (95%) of concentrations of contaminants in the surface soil matrix as partitioned according to the different levels of each variable (n = 72).

Variable	Na ($mg\ kg^{-1}$)			Cr ($mg\ kg^{-1}$)			Ni ($mg\ kg^{-1}$)			Cu ($mg\ kg^{-1}$)			Zn ($mg\ kg^{-1}$)			Cd ($mg\ kg^{-1}$)			Pb (mg kg-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	
Low	208	**263**	332	15.9	**17.3**	18.8	11.0	**12.9**	15.2	12.3	**13.8**	15.5	40.9	**49.3**	59.4	0.39	**0.45**	7.68	9.3	**47.6**	11.2
High	295	**372**	469	16.9	**18.4**	20.0	10.9	**12.9**	15.1	12.9	**14.5**	16.2	35.6	**42.9**	51.8	0.40	**0.46**	8.24	9.9	**39.1**	12.0
Sidewalk		NS			NS			NS			NS			NS			•			NS	
Impermeable	244	**278**	317	17.7	**18.6**	19.6	12.1	**13.2**	14.6	12.4	**13.3**	14.3	40.6	**45.7**	51.5	0.38	**0.41**	7.86	8.8	**48.0**	9.9
Permeable	261	**352**	475	15.2	**17.1**	19.1	10.1	**12.5**	15.5	12.7	**15.0**	17.6	35.3	**46.3**	60.6	0.42	**0.50**	8.08	10.5	**38.8**	13.6
Front lawn		NS			•			•			NS			NS			NS			NS	
Absent	228	**299**	390	15.4	**17.0**	18.8	9.7	**11.8**	14.3	12.2	**14.0**	16.2	34.9	**44.4**	56.5	0.40	**0.47**	0.55	8.0	**10.0**	12.7
Present	281	**328**	382	17.7	**18.7**	19.8	12.6	**14.1**	15.7	13.1	**14.2**	15.4	41.5	**47.7**	54.7	0.40	**0.44**	0.48	8.1	**9.2**	10.5
Sampling event		NS			**			*			*			NS			•			NS	
Jan. 2016	257	**328**	418	13.4	**15.9**	18.9	9.2	**11.9**	15.5	9.8	**12.7**	16.4	25.1	**42.0**	70.3	0.33	**0.40**	0.48	6.1	**9.0**	13.3
Nov. 2016	202	**270**	360	16.0	**17.2**	18.4	9.7	**12.4**	16.0	11.8	**13.3**	14.9	40.8	**45.2**	50.2	0.36	**0.43**	0.53	7.8	**9.3**	11.2
July 16	248	**346**	483	18.9	**20.8**	23.0	12.5	**14.4**	16.6	15.3	**16.6**	18.1	46.3	**51.3**	56.8	0.45	**0.55**	0.67	9.4	**10.6**	11.9

L = Lower bound of 95% confidence interval, U = Upper bound of 95% confidence interval
Sign over estimated mean denotes statistical significance of difference among levels of the factor: NS = Non-significant, • = $\boldsymbol{p} < 0.10$, * = $\boldsymbol{p} < 0.05$, ** = $\boldsymbol{p} < 0.01$

The presence of permeable surfaces around the tree pits (permeable sidewalk, and adjacent lawn) was associated with higher concentrations of some contaminants in the surface soil of the open area of the tree pit. Tree pits with a permeable sidewalk had about 21% more Cd in the surface soil than tree pits with an impermeable sidewalk (p = 0.04). Cr and Ni concentrations in the surface soil of tree pits next to a lawn were 10% (p = 0.06) and 19% (p = 0.06) higher than corresponding concentrations in soil from tree pits without a lawn. This difference might be due to a higher concentration in soil solution as seen in table 3. This effect was observed by Zeledón-Toruño et al. (2005) in the uptake of Cr and Ni from an aqueous solution. In chapter 4, the higher concentration of contaminants, such as Cr and Ni, in tree pits with a nearby lawn was associated with the reduction in water flux. In addition, a reduction in water flux might have caused an increased in contact time between water and soil (Table 4). The increase of contact time is reported to increase the sorption of Cd (Kamari et al. 2011), Cr and Ni (Zeledón-Toruño et al. 2005).

Table 8. Estimated mean and lower and upper confidence intervals (95%) of concentrations of contaminants in the soil matrix as partioned according to the different levels of each variable over three depths. (n = 144)

Variable	Na ($mg\ kg^{-1}$)			Cr ($mg\ kg^{-1}$)			Ni ($mg\ kg^{-1}$)			Cu ($mg\ kg^{-1}$)			Zn ($mg\ kg^{-1}$)			Cd ($mg\ kg^{-1}$)			Pb (mg kg-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			•	
Low	232	**295**	374	16.5	**17.4**	18.3	10.5	**13.0**	16.1	12.3	**13.5**	14.7	42.7	**46.7**	51.0	0.41	**0.46**	0.51	7.2	**8.6**	10.4
High	264	**335**	425	18.9	**19.9**	21.0	11.2	**13.8**	17.1	13.9	**15.3**	16.7	43.4	**47.4**	51.8	0.44	**0.49**	0.54	9.0	**10.8**	13.0
Depth		NS			NS			NS			NS			NS			NS			NS	
Surface	262	**337**	435	17.5	**18.3**	19.1	11.4	**13.5**	15.9	13.1	**14.4**	15.8	41.4	**46.0**	51.1	0.39	**0.45**	0.51	7.9	**9.4**	11.2
Middle	250	**306**	375	17.5	**18.6**	19.8	11.5	**13.6**	16.1	12.7	**13.9**	15.1	43.2	**47.2**	51.5	0.45	**0.49**	0.54	8.3	**9.6**	11.2
Deep	241	**301**	375	17.9	**18.9**	20.0	11.0	**13.1**	15.6	13.8	**14.8**	15.9	44.4	**48.0**	51.9	0.44	**0.48**	0.54	8.6	**9.9**	11.4
Sidewalk		NS			**			NS			•			*			*			NS	
Impermeable	288	**331**	379	19.0	**19.6**	20.2	11.9	**13.4**	15.1	12.9	**13.6**	14.3	41.3	**43.4**	45.7	0.41	**0.44**	0.47	7.9	**8.8**	9.8
Permeable	218	**299**	409	16.4	**17.6**	18.9	10.2	**13.4**	17.7	13.4	**15.1**	17.0	45.3	**50.9**	57.2	0.45	**0.51**	0.59	8.3	**10.6**	13.6
Front lawn		NS			*			•			•			NS			*			*	
Absent	218	**288**	381	16.8	**17.8**	19.0	9.9	**12.7**	16.2	13.5	**15.0**	16.7	44.1	**49.0**	54.4	0.45	**0.51**	0.58	8.8	**11.0**	13.7
Present	292	**343**	402	18.7	**19.4**	20.1	12.3	**14.2**	16.3	12.9	**13.7**	14.6	42.6	**45.2**	47.9	0.41	**0.44**	0.47	7.5	**8.5**	9.6
Sampling event		NS			**			NS			**			NS			**			•	
Nov. 2016	235	**291**	360	16.4	**17.2**	18.1	10.9	**12.8**	15.0	11.9	**13.0**	14.3	41.3	**44.9**	48.8	0.39	**0.42**	0.46	7.6	**9.0**	10.6
July 16	263	**340**	438	19.0	**20.0**	21.1	11.8	**14.1**	16.7	14.2	**15.8**	17.6	44.7	**49.3**	54.2	0.47	**0.53**	0.60	8.8	**10.4**	12.3

L = Lower bound of 95% confidence interval, U = Upper bound of 95% confidence interval.

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

Contaminant concentrations were not significantly affected by depth (Table 8). The tree pits were installed in the spring of 2015 and the soil was then disturbed again in October of 2016. It is, therefore, unlikely that any noticeable gradient in the contaminant concentrations would have had time to develop by time. This contradicts what was observed by Pontoni et al. (2016). In their study, laboratory scale, artificial soil was tested for the interaction with contaminated aqueous solution. The authors observed that trace metals, such as Cd, Cu, Ni, and Pb, were more concentrated in the top layers of soil 0 to 20 cm. Other studies also reported higher trace metal concentrations in the top 25 cm of soil in bioretention units (Sun and Davis 2007, Davis et al. 2003) similar to the tree pits in this study. Davis et al. (2003) carried their study in two parking lots for 5 and 1 year respectively. Sun and Davis (2007) study was a laboratory scale bioretention unit carried for approximately 8 months.

The results indicate that SOM improved the retention capacity of soil for some contaminants. For example, tree pits with higher SOM had higher concentrations of Cr ($p < 0.001$), Cu ($p = 0.04$) and Pb ($p = 0.07$) than tree pits with lower SOM (Table 8). Higher SOM was also associated with greater concentrations of Na ($p < 0.03$) in surface soil (Table 7). Kargar, Clark, et al. (2015) reported that an increase in SOM improved the soil retention capacity of Pb and Zn. However, in their study, SOM did affect retention of Cr, Cu, or Na. They associate the retention of Na and Cu to the soil rather than SOM.

The permeability of the surfaces around the tree pits had ambiguous effects in this study (Table 8). Design constraints prevented the statistical analysis of the interaction effect of permeable sidewalk and the presence of lawn; thus, the effects are evaluated on the single effects of these factors. The presence of a lawn increased the concentrations of Cr ($p = 0.01$) while it decreased the concentrations of Cu ($p = 0.09$), Cd ($p = 0.01$), and Pb ($p = 0.02$) in the samples from all three depths. On the other hand, the use of a permeable sidewalk increased the concentration of Cu ($p = 0.08$), Cd ($p = 0.01$), and Zn ($p = 0.01$) while decreasing the concentration of Cr ($p < 0.01$). Therefore, Cr, Cu and Cd presented different trends depending on which surface around the tree pit was permeable. The material used to build the permeable sidewalk could be a source of Cu and Cd observed. According to Zhang et al. (2018), the wear off of permeable sidewalk releases contaminants.

The sampling date had a significant effect on the concentrations of some metals (Cr, Ni, Cu and Cd) in the surface samples (Table 7). Kargar et al. (2013) stated that Ni, Cu, Zn, Cd, and Pb concentrations in Montreal's tree pits increased with time. The authors study tree pits with different ages, ranging from less than one to twenty-eight years. This increase could result from the deposition of traffic emissions and the accumulation of contaminants carried by runoff water. The analysis of data from all three depths also showed an increase in the concentration of Cr, Cu, Cd and Pb from the second to the third sampling date (Table 8).

Table 9. Mean concentrations of contaminants in tree pit soil compared with CCME standards.

Contaminant	L.	Mean	U.	Standard	Mean/Standard
	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	%
Na	311.00	335.00	361.00	-	-
Cr	18.90	19.40	19.90	64.00	30%
Ni	13.20	13.80	14.40	45.00	31%
Cu	13.20	13.70	14.20	63.00	22%
Zn	42.60	45.00	47.60	250.00	18%
Cd	0.41	0.43	0.45	10.00	4%
Pb	8.25	8.73	9.24	140.00	6%

n = 168.
L = Lower bound of 95% confidence interval of the mean.
U = Upper bound of 95% confidence interval of the mean.
CCME = Canadian Council of Ministers of the Environment.

The concentrations of contaminants in soil can pose a threat to human health. Samples analyzed in this study presented concentrations higher than the maximum recommended levels (Canadian Council of Ministers of the Environment 2001) (CCME). In the 18 months of this study, the overall mean concentrations of Cr in soil increased from 15.9 mg kg^{-1} to 20.0 mg kg^{-1} and of Ni from 11.9 mg kg^{-1} to 14.4 mg kg^{-1} which is almost a third of the maximum recommended levels for residential areas (Canadian Council of Ministers of the Environment 2001). This could indicate that the tree pits in the city of Montreal are serving their purpose as a sink for pollutants. Over the course of the study, the number of samples that exceeded the maximum recommended level for Zn increased from three to five.

The long-term performance of tree pits is suggested. As the concentrations of contaminants increase with time, contaminant saturation in the soil matrix might be reached. Soil saturation may hinder the adsorption capacities of the tree pits. Furthermore, a desorption process might be expected which would increase the release of contaminants into soil solution.

Chapter 6. Summary and conclusions

This research supports the idea that street tree pits can be used as bioretention units. Street tree pits with higher SOM better reduced the mass flux of contaminants with sampling depth. Soil matrix samples from the openings of tree pits with more SOM had higher concentrations of Cr, Cu and Pb as compared with tree pits with less SOM. This could indicate improved retention due to higher SOM.

The permeability of the surrounding surfaces affected fluxes of both water and the contaminants through the open part of the tree pits. A permeable sidewalk decreased the mass flux of Na but did not influence the water flux. The presence of a lawn decreased the mass fluxes of all contaminants through the open area of the tree pit, partly due to a decrease in water flux but the decrease in contaminant fluxes was proportionately higher than the decrease in water flux. Evaluation of the overall performance of the tree pits, including the soil volumes under the sidewalk, was not possible in this study because no samples were obtained from under the sidewalks or from the lawns. We assume that the overall flow of water through the entire system is greater because stormwater infiltrated into the ground through the lawn and permeable sidewalk, when present, before reaching the tree pit opening. Therefore, it is presumed that much less water ran off of the sidewalk into the gutter drains, that would otherwise have been the case. Overall, an increase in permeability of the surrounding area increased the concentration of some contaminants in the soil matrix of the open area of the tree pit although it is unclear why this was so.

The time effect varied with each contaminant. The mass fluxes of Zn and DOC were highest at the first date, possibly due mostly the disturbance of the soil when the trees were planted and the equipment installed. The highest mass flux of Cu and a high mass flux of Zn were observed on the last sampling date before the freezing of the soil. This could be a result of the pollutants present in snowfall as well as the wear of sidewalk material by machinery for dispersing de-icing salt. Na, Ni and Pb mass fluxes were highest values on the first sampling date after spring thaw, probably (for Na) due to the use of de-icing salt. The high values of Ni and Pb might have been a direct effect of the increased Na observed. Cr, Cu and Cd concentrations in the soil matrix increased over time, confirming that contaminants accumulate in urban soils.

The change in the mass flux of contaminants is suggested to characterize the performance of tree pits as bioretention cells because the contaminant concentrations might increase in a misleading way as the volumetric flux of water diminishes with depth.

Chapter 7. Recommendations for future research

A Sampling schedule based on rain events rather than at fixed times could provide different data. Another suggestion is to take into account rain intensity as well as the number of dry days between rain events. This is possible using weather stations. In this study, a weather station was available only for the last three sampling dates.

We could not test the interaction between sidewalk permeability and the presence of a lawn because of design constraints, all the units with permeable sidewalk also had lawn. This effect would be of interest to better understand the impact of increased surface permeability on local hydrology and the implications for stormwater management. Testing such an interaction would make it possible to compare a tree pit surrounded only by impermeable surfaces and a tree pit surrounded by permeable surfaces.

Studies similar to this one often use bioretention units with a top layer of organic matter. In this study, the soil was homogenized prior to the construction of the tree pits. An interesting study could be done comparing these two different methods of increasing SOM in tree pits.

The presence of lawn had positive impacts in this study. However, the lawns are private property, which presents challenges in adopting them as runoff control structures. The use of tree pits with permeable sidewalk seems promising to reduce the environment impacts of urbanization. A study including replicates of tree pits with permeable sidewalks would help to confirm this. Furthermore, the test of different construction materials and processes for the permeable sidewalk is suggested to investigate the sidewalk material as a source of contaminants.

These practices increase the adsorption and retention of water and contaminants. Adsorption and retention of contaminants and water is a feature of LID practices. Increased SOM and increased local permeability, therefore, can be considered in LID strategies. LID is a possible approach to mitigate the adverse effects on the environment of the unavoidable increase in urban areas.

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Appendices

Appendix A: F test summary for the effect of independent variables on concentration of contaminants in solution.

Table 10. Statistical significance table for contaminants concentration in soil solution.

	Na	Cr	Ni	Cu	Zn	Pb	DOC
Soil organic matter	ns	ns	ns	ns	ns	ns	ns
Sidewalk	***	***	ns	ns	•	*	ns
Depth	***	***	***	**	**	ns	*
Horizontal location	***	*	ns	•	ns	ns	ns
Front lawn	ns	***	*	•	*	ns	**
Sampling event	***	***	***	***	***	***	***
Soil organic matter *Depth	ns	ns	ns	ns	ns	ns	ns
Soil organic matter *Sidewalk	ns	ns	ns	ns	ns	•	ns
Soil organic matter *Sampling event	ns	***	•	**	*	*	**
Sidewalk*Depth	ns	*	•	*	ns	ns	•
Front lawn*Depth	***	ns	•	**	ns	ns	ns
Depth*Sampling event	***	***	ns	***	ns	***	***
Horizontal location*Sidewalk	*	ns	*	*	ns	ns	ns
Sidewalk *Sampling event	***	***	***	***	***	***	**
Depth*Horizontal location	ns	*	***	*	ns	ns	***
Horizontal location*Sampling event	***	*	**	ns	ns	***	•
Front Lawn*Sampling event	ns	***	**	***	*	ns	*

● ($p < 0.10$).
* ($p < 0.05$).
** ($p < 0.01$).
*** ($p < 0.001$).
ns (non-significant).

Appendix B: F test summary for the effect of independent variables on mass flux of contaminants and water flux in soil.

Table 11. Statistical significance table for mass flux of contaminants and water flux in soil.

	Na	Cr	Ni	Cu	Zn	Pb	DOC	WF
Soil organic matter	ns	ns	ns	ns	ns	ns	ns	ns
Sidewalk	**	*	*	*	ns	ns	*	ns
Depth	***	***	***	***	***	*	***	***
Horizontal location	***	***	***	***	**	***	*	***
Front lawn	***	*	***	***	***	***	**	***
Sampling event	***	***	***	***	***	***	***	***
Soil organic matter *Depth	*	*	*	*	**	ns	*	•
Soil organic matter *Sidewalk	ns	ns	ns	ns	ns	ns	ns	ns
Soil organic matter *Sampling event	ns	ns	ns	ns	ns	ns	ns	ns
Sidewalk*Depth	ns	ns	ns	ns	ns	ns	*	*
Front lawn*Depth	**	•	*	*	*	ns	**	ns
Depth*Sampling event	***	***	***	***	***	***	***	***
Horizontal location*Sidewalk	*	***	**	**	**	**	**	**
Sidewalk *Sampling event	***	***	***	***	***	***	**	***
Depth*Horizontal location	*	ns	*	*	*	ns	*	ns
Horizontal location*Sampling event	**	**	**	**	**	***	*	*
Front Lawn*Sampling event	**	ns	*	*	**	**	**	ns

● ($\boldsymbol{p}$ < 0.10).
* ($\boldsymbol{p}$ < 0.05).
** ($\boldsymbol{p}$ < 0.01).
*** ($\boldsymbol{p}$ < 0.001).
ns (non-significant).

Appendix C: F test summary for the effect of independent variables on contaminants concentration in surface soil for three sampling iterations.

Table 12. Statistical significance table for contaminants concentration in surface soil.

	Na	Cr	Ni	Cu	Zn	Cd	Pb
Soil organic matter	*	ns	ns	ns	ns	ns	ns
Sidewalk	ns	ns	ns	ns	ns	*	ns
Front lawn	ns	•	•	ns	ns	ns	ns
Sampling event	ns	**	**	*	ns	•	ns
Soil organic matter*Sidewalk	ns	*	ns	ns	ns	ns	ns
Soil organic matter*Sampling event	ns	ns	ns	•	ns	•	ns
Sidewalk*Depth	ns	ns	*	*	**	ns	ns
Front lawn*Sampling event	ns	•	**	**	**	*	**

● ($p < 0.10$).
* ($p < 0.05$).
** ($p < 0.01$).
ns (non-significant).

Appendix D: F test summary for the effect of independent variables on contaminants concentration in soil at three depths and two sampling iterations.

Table 13. Statistical significance table for contaminants concentration in soil at three depths for two sampling iterations.

	Na	Cr	Ni	Cu	Zn	Cd	Pb
Soil organic matter	ns	***	ns	*	ns	ns	•
Sidewalk	ns	**	ns	•	*	*	ns
Depth	ns	ns	ns	ns	ns	ns	ns
Front Lawn	ns	*	ns	•	ns	*	*
Sampling event	ns	***	•	*	ns	**	•
Soil category*Depth	ns	ns	ns	ns	ns	ns	ns
Soil organic matter*Sidewalk	ns	*	ns	ns	ns	ns	ns
Soil organic matter*Sampling event	ns	ns	ns	ns	ns	ns	ns
Sidewalk*Depth	•	*	ns	•	ns	*	ns
Front Lawn*Depth	ns	ns	ns	ns	ns	ns	ns
Depth*Sampling event	ns	ns	ns	ns	ns	ns	•
Sidewalk*Sampling event	ns	ns	ns	•	**	ns	ns
Front lawn*Sampling event	ns	ns	ns	**	ns	ns	•

● (***p*** < 0.10).
* (***p*** < 0.05).
** (***p*** < 0.01).
*** (***p*** < 0.001).
ns (non-significant).

Appendix E: General SAS Proc Mixed code for contaminant concentration in soil solution, mass flux of contaminants and water flux in soil.

```
DATA Na;
input TreePit Number Depth $ Source $ SType Permeable $ LawnAccess $ Subplot
$ Na1 Na2 Na3 Na4 Na5 Na6 Na7 Na8 Na9 Na10 Na11 Na12 ConcNa1 ConcNa2 ConcNa3
ConcNa4 ConcNa5 ConcNa6 ConcNa7 ConcNa8 ConcNa9 ConcNa10 ConcNa11 ConcNa12;

 Na1 = Log(Na1); ConcNa1 = Log(ConcNa1);
 Na2 = Log(Na2); ConcNa2 = Log(ConcNa2);
 Na3 = Log(Na3); ConcNa3 = Log(ConcNa3);
 Na4 = Log(Na4); ConcNa4 = Log(ConcNa4);
 Na5 = Log(Na5); ConcNa5 = Log(ConcNa5);
 Na6 = Log(Na6); ConcNa6 = Log(ConcNa6);
 Na7 = Log(Na7); ConcNa7 = Log(ConcNa7);
 Na8 = Log(Na8); ConcNa8 = Log(ConcNa8);
 Na9 = Log(Na9); ConcNa9 = Log(ConcNa9);
 Na10 = Log(Na10); ConcNa10 = Log(ConcNa10);
 Na11 = Log(Na11); ConcNa11 = Log(ConcNa11);
 Na12 = Log(Na12); ConcNa12 = Log(ConcNa12);


 MFNa=Na1; ConcNa=ConcNa1; Time=1; output;
 MFNa=Na2; ConcNa=ConcNa2; Time=2; output;
 MFNa=Na3; ConcNa=ConcNa3; Time=3; output;
 MFNa=Na4; ConcNa=ConcNa4; Time=4; output;
 MFNa=Na5; ConcNa=ConcNa5; Time=5; output;
 MFNa=Na6; ConcNa=ConcNa6; Time=6; output;
 MFNa=Na7; ConcNa=ConcNa7; Time=7; output;
```

```
 MFNa=Na8; ConcNa=ConcNa8; Time=8; output;

 MFNa=Na9; ConcNa=ConcNa9; Time=9; output;

 MFNa=Na10; ConcNa=ConcNa10; Time=10; output;

 MFNa=Na11; ConcNa=ConcNa11; Time=11; output;

 MFNa=Na12; ConcNa=ConcNa12; Time=12; output;


 drop Na1 Na2 Na3 Na4 Na5 Na6 Na7 Na8 Na9 Na10 Na11 Na12 ConcNa1 ConcNa2
ConcNa3 ConcNa4 ConcNa5 ConcNa6 ConcNa7 ConcNa8 ConcNa9 ConcNa10 ConcNa11
ConcNa12;



Datalines;

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;

Proc Sort Data=Na;
by SType LawnAccess Permeable TreePit Source SubPlot Depth Time;
Run;




ods output Lsmeans=Mean1;
ods output SolutionR=SolutionTreeTV;

Proc MIXED DATA=Na;
```

```
Class SType Permeable Source  LawnAccess Depth TreePit SubPlot Time;
Model MFNa = SType Permeable Depth Source  LawnAccess Time SType*Depth
SType*Permeable SType*Time Permeable*Depth LawnAccess*Depth Depth*Time
Permeable*Source Permeable*Time Source*Depth Source*Time LawnAccess*Time
/ddfm=kr outpred=PredNa;
Random TreePit(SType LawnAccess Permeable)/s;
Repeated Depth Time / Type=un@un Subject= SubPlot(SType LawnAccess Permeable
TreePit Source);

Lsmeans SType/diff cl;
Lsmeans Depth/adjust=scheffe cl;
Lsmeans Source/diff cl;
Lsmeans Permeable/diff cl;
Lsmeans LawnAccess/diff cl;
Lsmeans Time/adjust=scheffe cl;
Lsmeans SType*Depth/adjust=scheffe cl;
Lsmeans SType*Permeable/adjust=scheffe cl;
Lsmeans SType*Time/adjust=scheffe cl;
Lsmeans Depth*Permeable/adjust=scheffe cl;
Lsmeans Depth*LawnAccess/adjust=scheffe cl;
Lsmeans Depth*Time/adjust=scheffe cl;
Lsmeans Permeable*Source/adjust=scheffe cl;
Lsmeans Permeable*Time/adjust=scheffe cl;
Lsmeans Source*Depth/adjust=scheffe cl;
Lsmeans Source*Time/adjust=scheffe cl;
Lsmeans LawnAccess*Time/adjust=scheffe cl;
RUN;
```

```
Proc CONTENTS data = PredNA;
RUN;

Data PredNA2;
Set PredNA;
ResidualNA = Resid;
YhatNA = Pred;
drop Resid Pred;
Run;

Proc SORT data = PredNA2;
By  TreePit SType LawnAccess Permeable Depth Time;
Run;

Proc CONTENTS Data=Mean1;
RUN;

DATA Mean2;
Set Mean1;
EstimateEXP = exp(Estimate);
LowerEXP = exp(Lower);
UpperEXP = exp(Upper);

RUN;

Proc PRINT Data=Mean2;
Var DF Effect SType Depth Source Permeable LawnAccess Estimate Lower Upper
EstimateEXP  LowerEXP UpperEXP;
RUN;
```

Appendix F: General SAS Proc Mixed code for contaminants concentration at soil surface.

```
DATA NA;
input TreePit Number Depth $ SType Permeable $ LawnAccess $ NA1 NA2 NA3;

 NA1 = Log(NA1);
 NA2 = Log(NA2);
 NA3 = Log(NA3);


 NA=NA1; Time=1; output;
 NA=NA2; Time=2; output;
 NA=NA3; Time=3; output;

 drop NA1 NA2 NA3;

Datalines;

.
.
.

;



PROC PRINT DATA=NA;
RUN;
```

```
Proc Sort Data=NA;
by SType LawnAccess Permeable TreePit Time;
Run;



ods output Lsmeans=Mean1;
ods output SolutionR=SolutionTreeTV;

Proc MIXED DATA=NA;
Class SType Permeable LawnAccess TreePit Time;
Model NA = SType Permeable LawnAccess Time SType*Permeable SType*Time
Permeable*Time LawnAccess*Time /ddfm=kr outpred=PredNa;
Random TreePit(SType LawnAccess Permeable)/s;
Repeated Time / Type=un Subject= TreePit(SType LawnAccess Permeable);

Lsmeans SType/diff cl;
Lsmeans Permeable/diff cl;
Lsmeans LawnAccess/diff cl;
Lsmeans Time/adjust=scheffe cl;
Lsmeans SType*Permeable/adjust=scheffe cl;
Lsmeans SType*Time/adjust=scheffe cl;
Lsmeans Permeable*Time/adjust=scheffe cl;
Lsmeans LawnAccess*Time/adjust=scheffe cl;
RUN;


Proc CONTENTS data = PredNa;
RUN;
```

```
Data PredNa2;
Set PredNa;
ResidualNa = Resid;
YhatNa = Pred;
drop Resid Pred;
Run;

Proc SORT data = PredNa2;
By  TreePit SType LawnAccess Permeable Depth Time;
Run;

Proc PRINT data = PredNa2;
run;

Proc CONTENTS Data=Mean1;
RUN;

Proc PRINT Data=Mean1;
Var DF Effect Depth Permeable LawnAccess Estimate Lower Upper;
RUN;

DATA Mean2;
Set Mean1;
EstimateEXP = exp(Estimate);
LowerEXP = exp(Lower);
UpperEXP = exp(Upper);

RUN;
```

```
Proc PRINT Data=Mean2;
Var DF Effect Depth Permeable LawnAccess Estimate Lower Upper EstimateEXP
LowerEXP UpperEXP;
RUN;
```

Appendix G: General SAS Proc Mixed code for contaminants concentration in soil at three depths.

```
DATA NA;
input TreePit Number Depth $ SType Permeable $ LawnAccess $ NA2 NA3;

 NA2 = Log(NA2);
 NA3 = Log(NA3);


 NA=NA2; Time=2; output;
 NA=NA3; Time=3; output;

 drop NA2 NA3;

Datalines;

.
.
.

;



PROC PRINT DATA=NA;
RUN;

Proc Sort Data=NA;
```

```
by SType LawnAccess Permeable TreePit Depth Time;
Run;




ods output Lsmeans=Mean1;
ods output SolutionR=SolutionTreeTV;

Proc MIXED DATA=NA;
Class SType Permeable LawnAccess Depth TreePit Time;
Model NA = SType Permeable Depth LawnAccess Time SType*Depth SType*Permeable
SType*Time Permeable*Depth LawnAccess*Depth Depth*Time Permeable*Time
LawnAccess*Time /ddfm=kr outpred=PredNa;
Random TreePit(SType LawnAccess Permeable)/s;
Repeated Depth Time / Type=un@un Subject= TreePit(SType LawnAccess
Permeable);

Lsmeans SType/diff cl;
Lsmeans Depth/adjust=scheffe cl;
Lsmeans Permeable/diff cl;
Lsmeans LawnAccess/diff cl;
Lsmeans Time/adjust=scheffe cl;
Lsmeans SType*Depth/adjust=scheffe cl;
Lsmeans SType*Permeable/adjust=scheffe cl;
Lsmeans SType*Time/adjust=scheffe cl;
Lsmeans Depth*Permeable/adjust=scheffe cl;
Lsmeans Depth*LawnAccess/adjust=scheffe cl;
Lsmeans Depth*Time/adjust=scheffe cl;
Lsmeans Permeable*Time/adjust=scheffe cl;
```

```
Lsmeans LawnAccess*Time/adjust=scheffe cl;
RUN;


Proc CONTENTS data = PredNa;
RUN;

Data PredNa2;
Set PredNa;
ResidualNa = Resid;
YhatNa = Pred;
drop Resid Pred;
Run;

Proc SORT data = PredNa2;
By  TreePit SType LawnAccess Permeable Depth Time;
Run;

Proc PRINT data = PredNa2;
run;

Proc CONTENTS Data=Mean1;
RUN;

Proc PRINT Data=Mean1;
Var DF Effect Depth Permeable LawnAccess Estimate Lower Upper;
RUN;

DATA Mean2;
```

```
Set Mean1;
EstimateEXP = exp(Estimate);
LowerEXP = exp(Lower);
UpperEXP = exp(Upper);

RUN;

Proc PRINT Data=Mean2;
Var DF Effect SType Depth Permeable LawnAccess Estimate Lower Upper
EstimateEXP LowerEXP UpperEXP;
RUN;
```

Appendix H: Soil texture analysis per tree pit.

Tree pit No.	Sand (%)	Clay (%)	Silt (%)
1	84	7	9
2	88	6	6
3	90	5	5
4	90	6	4
5	86	6	8
6	90	5	5
7	86	7	7
8	88	4	8
9	92	4	4
10	88	6	6
12	88	6	6
13	92	4	4
14	87	7	6
15	88	6	6
16	87	6	7
17	85	6	9
22	86	6	8
24	90	6	4
27	88	6	6
28	86	8	6
29	89	5	6
30	88	6	6
31	91	3	6
32	88	5	7
Mean	**88.1**	**5.7**	**6.2**
Std. Deviation	2.1	1.1	1.5
Minimum	84.0	3.0	4.0
Maximum	92.0	8.0	9.0

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