Journal of Hydrology 574 (2019) 1053–1061

Contents lists available at ScienceDirect

Journal of Hydrology

journal homepage: www.elsevier.com/locate/jhydrol

Research papers

Measurement and conceptual modelling of retention of metals (Cu, Pb, Zn) in soils of three grass swales

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ARTICLE INFO

Keywords:
Grass swale
Heavy metals
Roadside soils
Planning-level modelling
Metal transport

ABSTRACT

Grass swales are important elements of the urban green infrastructure that convey and attenuate urban runoff and improve its quality mostly through stormwater infiltration into, and retention of conveyed pollutants by, swale soils. The retention of metals by grass swales was addressed in this study investigating the enrichment of swale soils by three common traffic-related metals: Cu, Pb and Zn. Three swales of various characteristics (L1, L2, L3) were selected for study and their soils were sampled by coring the top 30 cm and dividing the cores into 5 cm thick layers. Cumulative metal burdens were compared to those modelled by the proprietary StormTac Web model, which estimates annual loads of specific constituents for the given land uses and stormwater treatment. The comparisons of measured (MBm) and simulated (MBs) metal burdens retained by swales showed that the measured values exceed the simulated ones, as described by average ratios MBs/MBm = 0.64, 0.50 and 0.59, for swales L1, L2 and L3, respectively. The measured burdens were calculated after subtracting the native soil metal concentrations, assumed equal to those found in the deepest sampled layer, 25–30 cm below the surface. The results suggest the feasibility of assessing performance of grass swales by modelling metal (Cu, Pb, Zn) retention by swales, however for older facilities considered for rehabilitation, the simulated results should be supplemented by soil chemistry sampling.

1. Introduction

Modern or redeveloped urban areas include stormwater management measures in the form of green elements attempting to restore the original hydrological features of natural catchments and thereby reduce negative impacts of urbanisation. Among such elements, grass swales are widely used examples of Sustainable Urban Drainage Systems (SUDS) serving to convey and attenuate urban runoff and improve its quality, mostly through stormwater infiltration and retention of conveyed pollutants by swale soils (Ahiablame et al., 2012). In urban drainage practice, grassed swales are typically placed along roads and parking lots in the upper parts of catchments, and are exposed to traffic-generated pollutants including contaminated solids, metals and polycyclic aromatic hydrocarbons (PAHs). Much of the past research focused on retention of metals by grass swales, because such metals may occur in stormwater at toxic levels and their traffic-related sources are relatively ubiquitous and well-known in urban areas (Huber et al., 2016). As reported in the literature, short-term water quality enhancement performance of grass swales was investigated by comparing the quality of runoff entering and exiting swales, either from samples of actual runoff generated by rainfall (Stagge et al., 2012), or by sampling runoff simulated by irrigation (Deletic and Fletcher, 2006). In most studies, runoff quality was described by total suspended solids (TSS), nutrients (species of N and P), and metals (Cu, Pb, Zn) (Boger et al., 2018; Gavrić et al., 2019). Removals of total suspended solids (TSS) in grass swales were reported as high as 98% (Bäckström et al., 2006), but the removals of metals and nutrients were much lower (Boger et al., 2018). Overall, the previous research recognised that grass swales enhance runoff quality mostly through infiltration into swale soils (i.e., preventing pollutant transport) and settling.

Kondo et al. (2016) compared the soils of different types of green stormwater infrastructure (GSI) to non-GSI sites and showed that although the former had lower or similar concentrations of elements of concern to human health than the latter, the mean metal concentrations were generally higher than background concentrations and guidelines in all studied samples. In infiltration-based SUDS, metals accumulate in top layers of grassy areas (10–30 cm deep), with the highest concentrations occurring near the runoff inflow area (Tedoldi et al., 2017).

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https://doi.org/10.1016/j.jhydrol.2019.05.002
Received 20 December 2018; Received in revised form 28 April 2019; Accepted 1 May 2019
Available online 04 May 2019

Such concentrations decrease with the distance from the runoff inlet and with the soil depth, even though in older infiltration facilities (> 30 years old) the contamination can reach deeper layers as reviewed by Tedoldi et al. (2016). Particulate fractions of removed or immobilized pollutants remain either on the bottom of swales, or are incorporated into the swale soil matrix and may represent in-situ sources of pollution (Lind and Karro, 1995; Rushton, 2001). The strength of such sources depends on the runoff pollution sources and swale effectiveness in retaining the incoming pollutants, as determined either from short-duration field studies (Lind and Karro, 1995) or by modelling (Larm, 2000).

Field studies of pollution abatement by grass swales are complicated by difficulties in measuring low flows and pollutant concentrations, and analytical costs limiting the number of analysed samples and constituents. Such challenges can be overcome by modelling studies and the feasibility of modelling swale flows and their quality was addressed by a number of researchers (e.g., Larm, 2000; Deletic and Fletcher, 2006, Larm and Alm, 2014). While the modelling of swale flows was found feasible, with some limitations (e.g., Rujner et al., 2018), the detailed modelling of water quality in swales remains to be a challenge. The relevant models reported in the literature are either research models (Deletic and Fletcher, 2006), or represent integrated conceptual models derived directly from experimental data (Larm, 2000).

The objective of the study presented herein is to examine the feasibility of estimating the long-term grass swale performance in retaining metals (represented by the model substances Cu, Pb, Zn) from swale soil chemistry and conceptual modelling with the StormTac Web model. Such a comparison should provide a long-term assessment of grass swale performance in runoff pollution mitigation and the feasibility of conceptually modelling the annual metal balance of swales.

2. Materials and methods

2.1. The study sites

Three swale catchments were selected for study in the City of Luleå in Northern Sweden (65° 35′ 4″ N 22° 10′ 14″ E). The local climate is characterised by long winters; the snow season usually starts in October–November and snow remains on the ground until April. The lowest and highest monthly average temperatures occur in January (−11.5 °C) and July (15.5 °C), respectively, and the average annual precipitation is 506 mm (SMHI, 2014).

The first swale (L1) is located in a small commercial catchment and receives lateral runoff from a parking lot, a small part of the roof of an adjacent building, and a single-lane road with the average daily traffic (ADT) ~ 2750. The second swale (L2) is located close to the downtown and on one side drains a two-lane road with the highest ADT in Luleå ~11,650. The other side of the swale L2 is delineated by a continuous curb and does not receive any stormwater runoff; consequently, that side of the swale was excluded from investigations. The third swale (L3) is located in a small residential catchment and receives lateral runoff from a parking lot, a roof of an adjacent building, a grassed area, and a two-lane road with ADT ~ 2500. The footprint areas of the studied swales L1, L2 and L3, including the upstream non-sampled sections and the downstream sampled sections, were 225, 357 and 516 m^2, respectively. According to the Luleå municipality, the road and swale at L2 is 57 years old, and the roads and swales at L1 and L3 are 38 years old. Swale catchment characteristics are summarized in Table 1.

In Sweden, there are no specific guidelines for type of soil to be used for swale construction, and therefore Luleå swales are built using native soils. Soil infiltration rates of three swales corresponded to the soil texture of a loamy sand.

2.2. Soil sampling

Soil sampling was performed in October 2017 with a 5 cm diameter stainless steel corer, 30 cm long. At each site, a 20 m long swale section receiving direct runoff from only roads and parking lots was selected and sampled along three cross-sections 10 m apart. Samples were collected in each cross-section at distances 40 and 80 cm from the edge of the contributing drainage area and at the deepest point of the cross-section. Before extracting the soil cores, the grass was cut. Each core sample was divided into 5 cm slices, using a stainless steel knife, placed in a plastic bag, refrigerated and kept in cold storage until analysed (up to 7 days). At each swale, samples representing topsoil layer (0–5 cm) were subject to laboratory analysis. In swale L3, the side draining a parking lot could not be sampled, because of the presence of gravel from the construction of the parking lot. At all three swales, additional deeper soil layers from the swale bottom section (i.e. 5–10, 10–15 cm and the deepest layer obtained) were subject to laboratory analysis. Lastly, additional deeper soil samples (i.e. 5–10, 10–15 cm and the deepest sample obtained) from the swale side draining road were also analysed, except for swale L3, where only the samples from the top 5 cm were analysed. Some deeper soil layers could not be sampled because of high soil compaction. During the sampling campaign, location data (x-y-z coordinates) were collected at numerous points along the swale using a real-time kinematic-GPS device (model GeoMax Zenith35 Pro TAG) with the precision of 1.5 cm (for x and y) and 2 cm for z, in order to investigate the swale topography and estimate runoff contributing areas.

Table 1
Characteristics of studied swales.

Swale	L1	L2	L3
Runoff contributing surface areas [m^2]	241	728	520
Road traffic (ADT)	~2750	~11,650	~2500
Parking lots [m^2]	408	0	287
Roof of adjacent building [m^2]	5	0	812
Green areas [m^2]	0	0	726
Total area [m^2]	654	728	2,345
Ratio[1]	0.34	0.49	0.22
Swale age [year]	38	57	38

[1] Ratio between the swale area and runoff contributing total area.

2.3. Laboratory analyses

Samples collected in swales L1-L3, were preprocessed in the university laboratory according to the standard ISO 11464, and analysed for conductivity and pH. The conductivity was measured according to the standard ISO 11 265 (2016) using the CDM210 conductivity meter (MeterLab Radiometer analytical), with the accuracy of ± 0.2% of reading ± 3 of the least significant digit. Measurements of pH were done according to the standard ISO 10390 (2005), using the WTW pH 330 instrument with the accuracy of 0.01 pH unit. Air-dried samples were dry sieved in a vibratory sieve shaker (Retsch AS200) using a stainless sieve with the mesh size of 2 mm; soil lumps (> 2 mm) were crushed and mixed with the finer material (< 2 mm) (ISO 11464, 2016). The sample mass < 2 mm and > 2 mm was measured after sieving. Material > 2 mm comprised stones and grass roots. Analyses of total metals (Cu, Pb, and Zn) and loss on ignition (LOI) were done by an accredited commercial laboratory (ALS Scandinavia AB in Luleå). The total metal content was analysed using Inductively Coupled Plasma Sector Field Mass Spectrometry (ICP-SFMS) (Element, Thermo Fisher) according to SS EN ISO 17294-1, 2 and EPA-method 200.8. All metal concentrations were reported in mg/kg of dry weight (DW). Analytical uncertainties in the reported metal (Cu, Pb, Zn) concentrations were 19–21% of the reported values. The LOI analysis was associated with the metal analysis and was made after melting the material with lithium metaborate (L_iBO_2) at 1000 °C.

2.4. Calculating metal burdens using soil chemistry data

The mean pollutant concentration (C) [mg/kg] in each layer was multiplied by the mass of swale soil [kg] (in particles < 2 mm) in that layer, which was estimated from the soil density [kg/m^3] and the layer volume [m^3]. A more detailed explanation of the method follows.

For each 5 cm layer of swale, a mean metal concentration and standard deviation (SD) were calculated, using all the samples collected in the layer. In the case of swales L1 and L3, and the layers 15–20 cm and 20–25 cm, where no samples were analysed, the mean metal concentrations and SD were assumed to be equal to those of layers 10–15 and 25–30 cm, respectively.

From the measured soil mass (< 2 mm and > 2 mm) a mean total soil mass in a 5 cm slice (m_{sample} [g]) was calculated as well as the share of the material < 2 mm [%]. The volume of a sample (V_{sample} [m^3]) was calculated from a 5 cm high cylinder with a 5 cm diameter, representing the 5 cm slice from the soil corer. Soil density (ρ) [kg/m^3] in each layer was then calculated from the measured mass of the soil sample (including both materials smaller and larger than 2 mm) and the sample volume.

Each 5 cm layer volume (V_{layer}) [m^3] of swale soil was calculated from the swale area [m^2] and the layer thickness (5 cm). The area included the upstream swale section (which was not sampled), assuming that the metals were evenly distributed upstream of the sampled section. Mass of soil in each 5 cm layer ($M_{soil,layer}$) [kg] was calculated from ρ [kg/m^3] and layer volume [m^3]. Since the metal concentrations were analysed in sieved material (< 2 mm), mass of soil < 2 mm in each 5 cm layer ($M_{<2mm}$) [kg] was calculated by multiplying $M_{soil,layer}$ by the share of the material < 2 mm [%]. Finally, the obtained mass of soil $M_{<2mm}$ [kg] was multiplied by C [mg/kg] in order to estimate the metal mass [kg] retained in that layer of swale soil (M_{metal}).

2.4.1. Soil background concentration

Metal burdens in swale soils comprise two components: (i) naturally occurring (background) burdens in swale soils, and (ii) burdens contributed by anthropogenic sources (i.e., mostly road runoff and atmospheric deposition). The knowledge of the chemistry of native soils in the three studied swales would allow for subtracting the native metal burdens from those calculated from sampled chemical concentrations. The idea of obtaining the native soil chemistry information from Swale L2, which does not receive stormwater runoff on one side, was rejected, because only top layer (0–5 cm) samples from this side were analysed and were potentially contaminated by stored snow. Another way of accounting for native soil metal concentrations would be to use the mean concentration in the deepest soil layer (25–30 cm deep) as the best estimate, since the studied swales were built with native soils. The metal concentrations in the deepest soil layer were compared to the background levels from Johansson et al. (1995) who investigated regional patterns of metals (Cu, Zn, Pb, Hg, and Cd) in lake sediments and the mor layer in forest soils in Sweden. For Cu and Zn, the authors adopted the soils data from remote areas of Northern Sweden (Andersson et al., 1991) as the background concentrations of 55 and 6.3 ppm, respectively. Concerning Pb, the mean concentration of 15 mg/kg was reduced to 8 mg/kg, on the basis of the historical record of the anthropogenic effect on Pb in lake sediments. Table 2 shows the mean metal concentration values from the deepest soil layer at the three sampled swales in this study (L1–L3), together with the metal concentrations from remote areas of Northern Sweden and the estimated background levels of metals by Johansson et al. (1995).

Table 2 shows that the background levels of Pb (8 mg/kg) and Zn (55 mg/kg) are higher than the concentrations sampled in the deepest layers at L1 and L3, while those of Cu were comparable. The background levels of all the metals studied were lower than those in the deepest layer of L2. For estimating the background soil metal burdens (M_{back}) in this study, the mean metal concentrations in the deepest soil layer of individual swales were used as the best estimates of native soil metal concentrations.

Table 2
Mean metal concentrations and standard deviation in the deepest sampled layer, the values from remote areas in Northern Sweden, and estimated background concentrations from Johansson et al. (1995).

	Cu [mg/kg DW]	Pb [mg/kg DW]	Zn [mg/kg DW]
L1 (25–30 cm)	5.7 ± 1.1	2.9 ± 0.8	16.6 ± 3.0
L2 (25–30 cm)[1]	19.3 ± 1.8	80.1 ± 11.1	61.4 ± 8.5
L3 (25–30 cm)	7.1 ± 0.7	4.4 ± 1.3	17.3 ± 4.5
remote areas of N. Sweden (mor layer)	6.3	15	55
background levels	6	8	55

[1] Because only two samples were collected from the deepest layer in swale L2, instead of the standard deviation, the differences of the actual concentrations from the mean of the two samples available for the layer were shown.

Cumulative metal burden (M_{tot}) for a 30 cm thick soil layer was calculated as a sum of metal masses in individual layers (M_{metal}), reduced by the background metal burden in each layer (M_{back}).

Total uncertainty of the cumulative metal burden (s_{tot}) was calculated using the law of uncertainty propagation and considering the uncertainty of the mean metal concentrations for the layer (SD) (uncertainty in estimating $M_{<2mm}$ was considered negligible) (Eq. (1)).

$$s_{tot} = \sqrt{s_{metal,1}{}^2 + s_{metal,2}{}^2 + \ldots + s_{metal,6}{}^2 + s_{back,1}{}^2 + s_{back,2}{}^2 + \ldots + s_{back,6}{}^2} \quad (1)$$

where,

$s_{metal,\,i}$ – standard deviation of metal concentration for the layer
$s_{back,\,i}$ – standard deviation of background metal concentration
index i = 1,…,6 denotes the number of layer e.g. 1 (0–5 cm), 2 (5–10 cm), … 6 (25–30 cm)

2.5. Modelling methods

The second study objective was to examine the feasibility of estimating the total input loads of Cu, Pb, and Zn with runoff into the tested swales, and their partial retention in swale soils, by runoff modelling. Recognizing the challenges of swale flow modelling and the need to model long-term pollutant fluxes, for modelling the swale mass balances of the metals studied, a source-based urban runoff model (StormTac Web) was selected to simulate the annual pollutant mass retained in the swale soils. This proprietary conceptual model requires little input data for performing long-term calculations of annual runoff fluxes and pollutant loads, which can be used for planning and design of stormwater treatment facilities (Larm, 2000). The model inputs include: (i) precipitation (rain + snow), (ii) the land-use data, including the corresponding runoff contributing area and the volumetric runoff coefficient, and (iii) the StormTac Web database standard pollutant concentrations for each land use, which represent annual averages based mainly on long-term flow proportional sampling.

The annual stormwater pollutant load entering the swale (L_{in} [kg/year]) is calculated as a product of annual flow volume and standard pollutant concentrations for each land use. The annual runoff volume (Q [m^3/year]) is calculated from annual precipitation, a sub-catchment area of specific land use, and the land-use specific volumetric runoff coefficient (Larm, 2000). The total annual runoff is then obtained as a sum of annual runoff volumes from individual sub-catchments:

$$Q = 10\,p \sum_{i=1}^{N} (\varphi_i A_i) \left[\frac{m^3}{year}\right] \quad (2)$$

where,

p – The average annual precipitation (rain + snow) [mm/year], corrected by a factor 1.1 to account for sampling errors such as wind, evaporation and surface wetting

Constant 10 serves as a unit conversion constant φ – Volumetric runoff coefficient (0.85 for roads and parking lots, 0.9 for roofs, and 0.1 for grass areas)

A – Land use area [ha]

i = 1, 2, …, N (number of land use areas)

The model produces estimates of the average annual pollutant loads entering and leaving the swales, from which one can determine the annual loads retained in the swales, using site-specific functions. For more than 15 different types of stormwater treatment facilities (STFs) (e.g., swales, biofilters etc.), StormTac Web database contains treatment efficiencies [%] derived from flow-proportional input and output data from such facilities. Furthermore, the database includes site-specific data of the STFs, e.g. the ratio of the footprint area of different facilities (e.g. biofilters and swales) to reduced watershed area (i.e., the watershed area multiplied by the runoff coefficient). Based on this data regression equations (RE [%]) are determined for calculating STF reduction efficiencies, which are updated continuously as more data is added to the database (Larm and Alm, 2014). The regression equation for swale includes the ratio of the swale area to the reduced drainage area, empirical field data of the studied substances removal by the swale compiled in the StormTac Web database, and additional site-specific characteristics such as inflow concentrations (Eq. (3)) (Larm and Alm, 2016).

$$RE\% = [k_1 \ln(n_o) + k_2] * f_{c_{in}} * f_{c_{irr}} * f_{bypass} \tag{3}$$

where

k_1 and k_2 – regression coefficients

n_0 – share facility area of reduced watershed area [%], where reduced watershed area is calculated by multiplying land use areas (Ai) by the volumetric runoff coefficient ($\varphi_i < 1$).

f – factor

C_{in} – inlet concentration

C_{irr} – irreducible concentration, the minimum outflow concentrations which cannot be further reduced by the STF

Bypass – bypass level can be set to 0 or 1, indicating if a portion of the flow will overflow or bypass the STF without treatment. If there is a bypass, the flow entering the facility is set as an input. This flow representing a part of the total annual runoff volume is also calculated in StormTac Web.

The annual load leaving the swale after treatment (L_{out} [kg/year]) is calculated from Eq. (4).

$$L_{out} = L_{in} - \frac{RE}{100} * L_{in} \tag{4}$$

The annual pollutant mass retained by the swale is calculated from the pollutant runoff input loads and the loads leaving the swale after treatment (L_{in}-L_{out}). The StormTac Web model calculates only the annual loads added (kg/year) to the soil from the polluted stormwater and groundwater, without consideration of the metal mass native to the soil. Thus, the load retained by the swale represents the load added to the existing pollutant burden resulting from pollutant background concentrations in soils. The annual loads produced by the model were multiplied by swale ages in years (38, 57 and 38 years, for L1, L2 and L3, respectively) to obtain the total loads.

2.5.1. Modelling uncertainty

For estimating the modelling uncertainty, each pollutant concentration has been categorized by the model developers into three levels of uncertainty, based on the coefficient of variation (CV) of input data (i.e., CV < 0.5, 0.5–1.25, and > 1.25, indicating high, intermediate, and low certainties, respectively (Table 3)) (Burton and Pitt, 2002). Based on these criteria, standard concentrations of Cu, Pb, and Zn were ascribed low uncertainty for the reduction efficiencies in swales, average uncertainty for the concentrations from parking lots and grass areas, but high uncertainty for roofs. For the road runoff, different assessment criteria were used, and according to such criteria the pollutant concentrations in road runoff were considered as containing low uncertainties (R^2-value > 0.75) (Table 3). In this study, the data from Swedish studies were used for calculating standard concentrations in road runoff.

Table 3
Coefficient of variation (CV) and number of studies (n) from StormTac Web database used for calculating standard stormwater pollutant concentration for each land use.

Metal	Roads		
	Pb	Cu	Zn
R^{2*}	0.94	0.97	0.97
n	7	7	8
	Parking lots		
CV	1.20	0.57	0.58
n	8	10	8
	Roofs		
CV	1.9	2.0	2.5
n	8	9	9
	Green areas		
CV	1.1	0.56	0.54
n	4	4	4

* Regression coefficient of determination (R^2) for the road runoff.

StormTac Web “standard pollutant concentrations” (Larm, 2000) are estimated for each land use based on an overall assessment of available literature data, time trends, calibrations based on case studies and comparisons with data from similar land uses. Therefore, the standard concentration is not equal to median value from the flow proportional sampling presented in StormTac Web database.

The standard concentrations for different pollutants and land uses are based on investigations where extensive flow or volume proportional sampling of runoff from catchments with one type of land use have been conducted. These measurements cover all seasons of the year generating good estimates of the site mean concentrations and are conducted in a similar climate as where the model is applied. The standard concentrations are updated continuously. When new reliable data for a specific land use is incorporated into the database, data for other comparable land uses are re-evaluated. The standard concentrations are also calibrated to consider different time trends. StormTac Web uses factors (0–10) for each land use, except the roads, to calculate land use specific concentrations. A factor 5 indicates standard conditions of the land use, while factors < 5 and > 5 indicate that the levels are adjusted towards the minimum or the maximum values in the database, respectively. For the roads, the standard concentration is calculated on the basis of the ADT.

To further investigate the modelling sensitivity and uncertainty, two scenarios were formulated with a minimum factor (0) and a maximum factor (10) to obtain minimum and maximum land use input concentrations and thus the yearly input loads (L_{in}) from the model. Moreover, the swale pollutant reduction efficiency (RE) was set to the minimum and maximum values estimated from StormTac Web database. The impact of these two parameters was investigated, because they were considered the most uncertain and directly related to the stormwater quality modelling. This resulted in two scenarios for estimating the minimum annual load retained in the swale soil [kg/year] (RE_{min} and C_{min}) and the maximum annual load retained in the swale soil [kg/year] (RE_{max} and C_{max}). The metal mass from these two scenarios was compared to the standard scenario.

2.5.2. Leaded gasoline scenario

In order to investigate the scenario with higher Pb concentrations reflecting the use of leaded gasoline in the past, the median concentration of lead in runoff prior to 1995 was calculated from concentration data from the older road and parking lot data (Larm, 1994). To estimate median concentrations of Pb in road runoff after 1995, the

Table 4
Soil properties of the three studied swales: mean values from the samples analysed and standard deviations are shown in the brackets.

Swale	LOI [% DW]	pH	Conductivity [μS/cm]
L1 (37 samples)	6.6 (3.8)	6.5 (0.3)	28 (17)
L2 (35 samples)	4.4 (1.0)	7.1 (0.3)	22 (7)
L3 (18 samples)	6.6 (3.1)	6.5 (0.2)	32 (17)

StormTac Web database was queried ignoring the data prior to 1995. This resulted in Pb concentrations about seven times lower than those prior to 1995. Thus, two Pb concentrations were used in computations of metal burdens entering the swales studied, the concentration currently available in the model for the period starting in 1995, and the seven times higher concentration for the period prior to 1995. For Cu and Zn, the current estimates of concentrations were used throughout the period studied.

3. Results

Soils properties of the three swales were characterized by measuring their LOI, pH and conductivity (Section 2.3) and the results are presented in Table 4.

The mean total metal concentrations in various soil layers at each swale site (L1–L3) are listed in Table 5. The coefficient of variation (CV) and the number of samples collected (n) are shown in the brackets.

The data in Table 5 show distinctly higher metal concentrations (particularly in the case of Pb) in soil samples from the oldest swale L2, which is exposed to runoff from the site with the highest ADT intensity.

To assess the environmental significance of measured metal concentrations, the calculated mean metal concentrations (mean from all samples at the site) and the maximum measured concentrations were compared to the Swedish Environmental Protection Agency guideline for polluted soils (SEPA, 2016). The guideline is meant to protect people's health taking into account the direct exposure (i.e. contact with the contaminated soil) and indirect exposure (i.e. transport of contaminants to air, groundwater and plants) (SEPA, 2009). There are guideline values for two different types of land use: sensitive land use (S) and less sensitive land use (LS), respectively. Both guideline values were used here for comparison, as illustrated in Table 6, listing the mean total metal concentrations, the maximum measured concentrations, and the two target values from the guideline. Exceeding the guideline values for less sensitive land use (LS) means that the soil quality limits the choice of land use to e.g. offices, industries or roads (SEPA, 2009).

Table 5
Mean concentrations of Cu, Pb and Zn in four soil layers of three swales (in brackets, the coefficient of variation and the number of samples).

Soil layer depth [cm]		0–5	5–10	10–15	25–30
Cu [mg/kg DW]	L1	25 (0.23; 15)	17 (0.26; 9)	14 (0.33; 9)	6 (0.20; 4)
	L2[1]	38 (0.18; 9)	45 (0.17; 9)	32 (0.23; 9)	(19 ± 2)[2]
	L3	20 (0.33; 9)	11 (0.15; 3)	10 (0.33; 3)	7 (0.10; 3)
Pb [mg/kg DW]	L1	8 (0.16; 15)	8 (0.26; 9)	8 (0.18; 9)	3 (0.26; 4)
	L2[1]	33 (1.02; 9)	50 (0.96; 9)	104 (0.42; 9)	(80 ± 11)[2]
	L3	9 (0.28; 9)	9 (0.17; 3)	9 (0.42; 3)	4 (0.30; 3)
Zn [mg/kg DW]	L1	81 (0.18; 15)	53 (0.34; 9)	47 (0.36; 9)	17 (0.18; 4)
	L2[1]	98 (0.13; 9)	104 (0.12; 9)	95 (0.23; 9)	(61 ± 8)[2]
	L3	49 (0.28; 9)	30 (0.06; 3)	25 (0.21; 3)	17 (0.26; 3)

[1] The table does not contain 6 samples that were taken in the layer 15–25 cm.

[2] Concentrations of only two samples were available for the layer.

The maximum measured Cu concentrations were 36.7, 57.9, and 30.4 mg/kg for L1, L2, and L3 respectively, below the guideline maximum permissible concentrations of 80 (S) and 200 mg/kg (LS) (SEPA, 2016). Similarly, the maximum measured Zn concentrations were 111, 136, and 64.2 mg/kg for L1, L2, and L3 respectively, and none of them exceeded the guideline maximum permissible concentrations of 250 mg/kg (S) and 500 mg/kg (LS) (SEPA, 2016). Finally, the maximum Pb concentrations measured in soil samples were 13.3, 188, and 14.5 mg/kg for L1, L2, and L3 respectively. While the Pb concentrations at sites L1 and L3 complied with the SEPA guideline 50 (S) and 400 mg/kg (LS) (SEPA, 2016), Pb concentrations at site L2 exceeded the sensitive threshold (S) by the mean concentration and the maximum Pb concentration Fig. 2 shows the Pb burdens estimated from soil samples in three locations. The lowest Pb burden was calculated for swale L1, and the loads at L1 and L3 were low compared to the much higher load in swale L2, which is next to the road with the highest ADT and thus exposed to more traffic pollution. Figs. 1 and 3 show Cu and Zn burdens, respectively, which were larger in L1 and L3 than those of Pb in the same swale, and otherwise followed the same trend as noted above for Pb (L2 metal load > L3 metal load > L1 metal load).

StormTac Web simulations of annual inputs and the retained loads are listed in Table 7, together with the estimates of the total pollutant mass, which has been retained since the swale construction. The calculations were performed for the current catchment layout and the mean ADTs, which were corrected to represent only the road lanes draining into the swales (Table 1). The annual input load was determined for the standard concentrations. The retained load [kg/year] is presented in three variants, based on standard, minimum and maximum concentrations (as described in Section 2.5.1).

In road runoff pollution calculations, the StormTac Web model accounts for traffic intensity. In order to examine the calculated metal loads sensitivities to ADT, historical ADT values were obtained from the Luleå municipality for the period of swale operation. The available data included three measured ADTs during 2008–2016 for L1 and L3, and more detailed measurements for L2 (1980–2016). The yearly retained metal load was simulated using three ADT values (the mean, and the mean ± standard deviation). The modelled values were little sensitive to historical variations in ADTs at the study sites thus, the mean ADT was used for the load calculations.

Fig. 4 shows the comparison of cumulative metal loads from soil samples (without background concentrations) and from the model for the standard scenario and the scenario giving the minimum and maximum annual retained load.

The differences between the measured and modelled Pb burdens can be further examined by accounting for phasing lead out of gasoline in the early 1990s. Analysis of historical trends in measured Pb concentrations in road runoff over the course of five decades, 1970–2010, showed a sharp decline in Pb concentrations after 1990, corresponding to the phasing out of Pb from gasoline (Huber et al., 2016). The lead-free gasoline was introduced in 1985, and from 1995, the sales of leaded gasoline were discontinued in Sweden (SPBI, 2014). Fig. 5 shows the comparison of the Pb burdens calculated from: (1) soil sample chemistry data, (2) model outputs without modifying Pb inflow concentrations prior to 1995, and (3) model outputs with increased Pb concentrations prior to 1995.

The calculated metal burdens for the leaded gasoline scenario were about 4, 5 and 3 times higher than those neglecting the higher lead concentration in gasoline prior to 1995, for swales L1, L2 and L3, respectively. Overall, Fig. 5 shows that the cumulative soil metal burdens do not exceed the modelled burdens for L2 and L3 when the higher past lead concentrations are considered. In the case of L1, the opposite is true, i.e. metal burden from the model output is around 3 times the cumulative mass from soil samples.

Table 6
Mean and maximum concentrations of Cu, Pb, and Zn in the three swales studied, and the SEPA guideline values for contaminated soils (sensitive (S) and less sensitive (LS) land use).

		Cu [mg/kg DW]	Pb [mg/kg DW]	Zn [mg/kg DW]
L1 (37 samples)	mean ± SD	18.4 ± 7.9	7.5 ± 2.2	59 ± 26.0
	Max	36.7 ± 7.7[1]	13.3 ± 2.7[1]	111 ± 21.0[1]
L2 (35 samples)	mean ± SD	35.5 ± 10.0	74.0 ± 54.3	94.6 ± 18.2
	Max	57.9 ± 12.1[1]	188 ± 39[1]	136 ± 26[1]
L3 (18 samples)	mean ± SD	14.5 ± 7.1	8.3 ± 2.9	36.4 ± 16.5
	Max	30.4 ± 6.4[1]	14.5 ± 3.0[1]	64.2 ± 12.3[1]
SEPA	S	80	50	250
	LS	200	400	500

[1] Analytical uncertainty in the reported values.

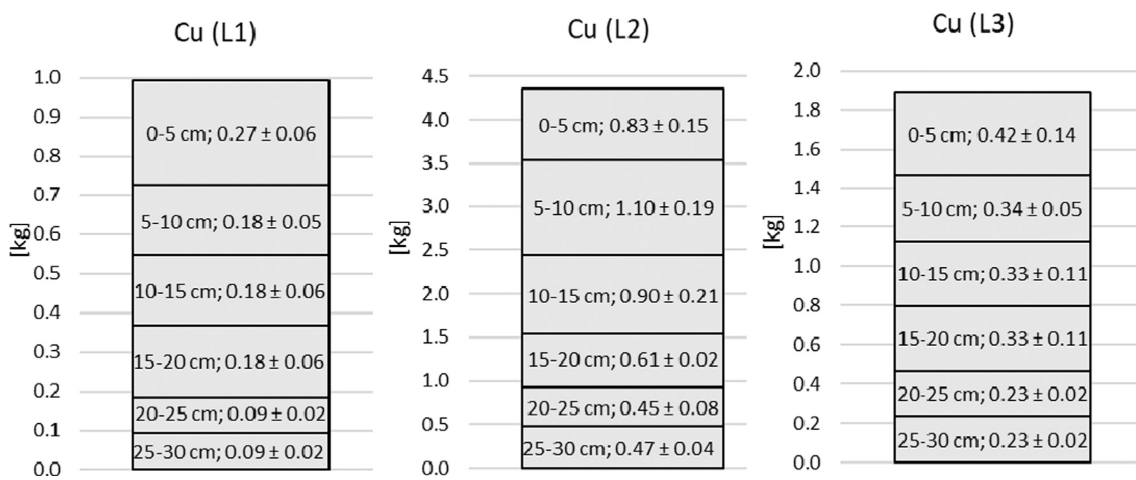

Fig. 1. Calculated Cu mass in each layer of the soils in the three swales studied, where the ± value represents the uncertainty calculated from the SD of metal concentrations in that layer (Section 2.4).

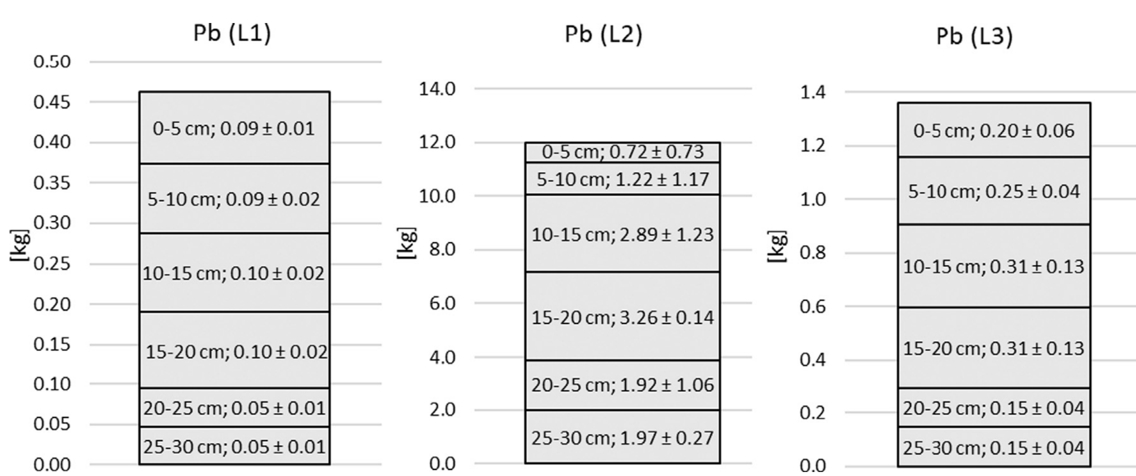

Fig. 2. Calculated Pb mass in each layer of the soils in the three swales studied, where the ± value represents the uncertainty calculated from the SD of metal concentrations in that layer (Section 2.4).

4. Discussion

Higher metal concentrations were measured in the soil samples from the oldest swale L2, which was exposed to the highest ADT (Table 5). Such a finding agrees with the previous research: investigations of soil quality of 35 swales (depths 20–30 cm) showed increased soil metal concentrations (Cu, Cd, Cr, Pb and Zn) for higher average daily traffic (ADT) intensity and frequent stop-and-go traffic (Horstmeyer et al., 2016). Moreover, the metal content in swales of various ages (1–34 years old) showed significantly higher content of Pb in older facilities (Horstmeyer et al., 2016).

High Pb concentrations in soils next to a highway road built in 1964 (with ADT of 40,000–50,000) were measured by Norrström and Jacks (1998). Along the line at distances of 0.5 and 2.5 m from the road, 20 soil cores were sampled and divided in 5 cm slices. The samples representing the same layer were composited, and Pb concentration measured in the 0–5 cm layer was 542 mg/kg and in the 5–10 cm layer 502 mg/kg. Moreover, Carrero et al. (2013) measured high Pb concentrations in roadside soil samples even though the leaded gasoline has been banned since 2001 at the study site location.

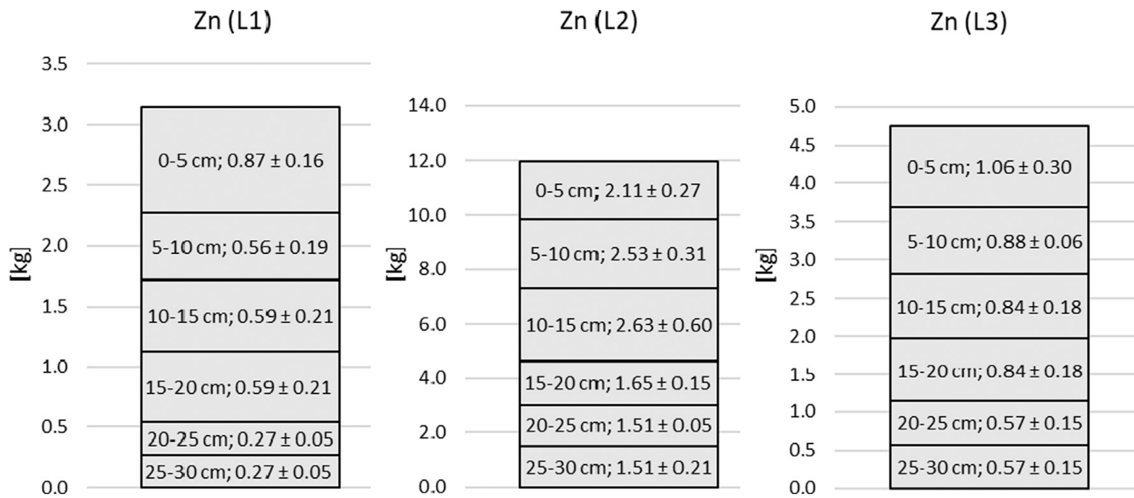

Fig. 3. Calculated Zn mass for each layer in the soils in the three swales studied, where the ± value represents the uncertainty calculated from the SD of metal concentrations in that layer (Section 2.4).

Table 7
The table presents annual metal input load and retained metal loads calculated from model StormTac Web. For each swale, annual retained loads for three scenarios are presented (Section 2.5.1). Lastly, calculated metal burdens that have accumulated in the three studied swales since the construction are presented (the calculations are based on model outputs of annually retained metal loads Section 2.5).

		Cu	Pb	Zn
L1	Annual input load [kg/year] standard	0.011	0.0065	0.036
	Annual retained load [kg/year] standard	0.0076	0.0049	0.028
	Annual retained load [kg/year] min	0.0033	0.0014	0.0075
	Annual retained load [kg/year] max	0.0096	0.0078	0.046
	Retained burden (*38 years) [kg] standard	0.3	0.2	1.1
L2	Annual input load [kg/year] standard	0.014	0.0046	0.064
	Annual retained load [kg/year] standard	0.011	0.0034	0.054
	Annual retained load [kg/year] min	0.0055	0.0021	0.025
	Annual retained load [kg/year] max	0.012	0.004	0.064
	Retained burden (*57 years) [kg] standard	0.6	0.2	3.1
L3	Annual input load [kg/year] standard	0.016	0.0068	0.050
	Annual retained load [kg/year] standard	0.0098	0.0046	0.037
	Annual retained load [kg/year] min	0.005	0.0018	0.013
	Annual retained load [kg/year] max	0.016	0.0086	0.096
	Retained burden (*38 years) [kg] standard	0.4	0.2	1.3

The total pollutant load entering the swale is partly incorporated into the soil matrix, and partly passes through the swale unabated. Among the metals studied, Pb is mostly associated with the particulate fraction, while Cu and Zn may also occur in appreciable quantities in the dissolved fraction (Huber et al., 2016). Fig. 2 indicates the lowest Pb concentrations in the two top layers of swale L2, followed by the peak in the fourth layer, and gradual stabilisation in deeper layers. Pb layer burdens in L1 and L3 were more or less uniformly distributed. Recognizing the high Pb affinity to solids, high Pb concentrations in the fourth layer of L2 (15–20 cm below the surface) would suggest migration (flushing out) of fine particles with Pb from the top layer into the deeper soil horizons. Such vertical Pb transport would confirm the previous observations that older stormwater facilities may have higher metal content and that Pb may migrate into deeper soil layers (Rushton, 2001; Horstmeyer et al., 2016; Tedoldi et al., 2016).

There are multiple sources of uncertainties in the metal burdens presented in the results section: (i) Metal uptake by vegetation, (ii) metal burden variation along the swale (in the flow direction), (iii) transport of metals into deeper layers, (iv) background metal concentrations in native soils, and (v) uncertainties in modelling results. A brief discussion of all such uncertainties follows.

In the procedure applied in this study, the metal mass calculated from the soil chemistry data represented the mass retained in the top soil layer (30 cm deep) of the swale (the soil material < 2 mm). In the study by Leroy et al. (2017), after two-year period, the plant uptake (including both bellow and above-ground plant parts) was 0.4–4.6% of the initial soil metal mass. The study included a control section and the swale section spiked with PAHs and metals producing Cd, Pb, and Zn surface soil content of 2, 100, and 300 mg/kg respectively. The mean Pb and Zn mass plant uptake was significantly higher for the artificially contaminated swale section (spiked with a mix of PAHs and metals), compared to the control section (non-contaminated) (Leroy et al., 2017). However, the use of artificially spiked pollutants (Leroy et al.,

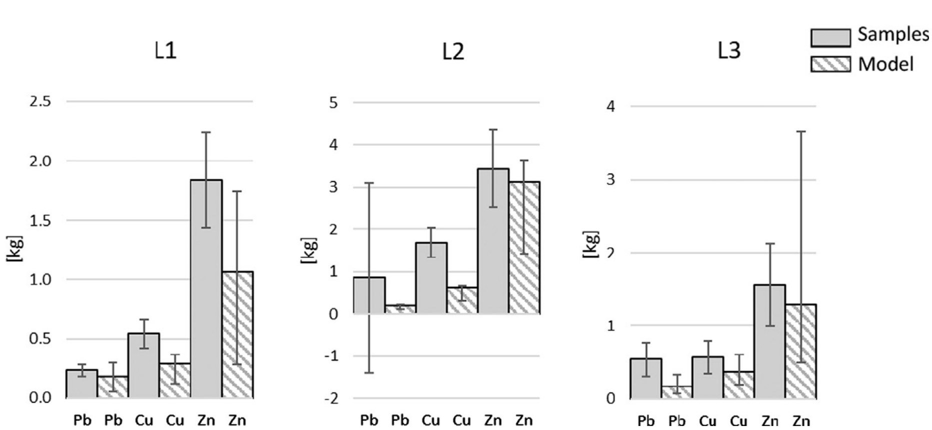

Fig. 4. Metal mass without the native soil burden, and metal mass calculated from the model output. Uncertainty bands for metal burdens calculated for soil samples represent ± standard deviation (described in the method Section 2.4.1. (Eq. (1))) Uncertainty bands for the modelled burden display the burdens calculated for the two extreme scenarios (method Section 2.5.1)

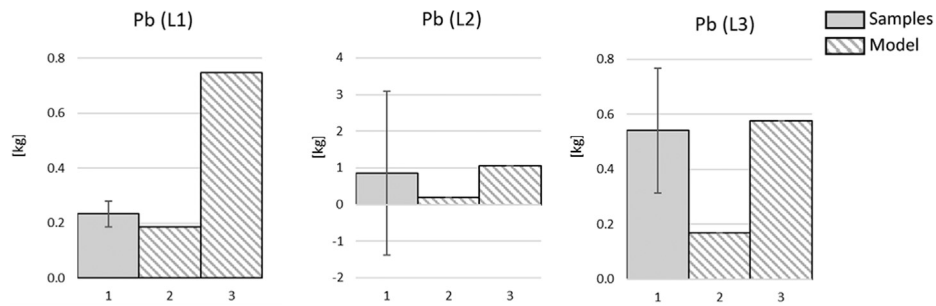

Fig. 5. Comparison of Pb influx masses into the studied swales calculated from: (1) soil samples; (2) model output with no modification of Pb concentrations, and, (3) the model output considering the leaded gasoline before 1995.

2017) might have increased the plant uptake. Considering that plants also contribute to metal (Zn, Cd, Pb) retention in swales through metal uptake (Leroy et al., 2017), the inclusion of such a metal uptake (out of scope of the present study) would have yielded higher retention of metals by the swales studied.

Moreover, we assumed that the mean pollutant concentration in each layer of the sampled 20-m long swale section was representative of the entire swale. This could lead to a load overestimation, if the pollutant concentrations in the swale were increasing in the downstream direction. By comparison, model outputs represent the annual pollutant burden retained by all soil layers, without distinguishing the burden parts retained by individual soil layers. The model calculation also does not include transfer of pollutants into the groundwater. Furthermore, as suggested above, dissolved metal burdens (in the case of Cu and Zn) could be transferred deeper than the 30 cm depth considered here and possibly even into the groundwater.

Fig. 4 shows a comparison among the cumulative metal burdens from soil samples, reduced by the native soil burdens, and those calculated from model outputs. Higher uncertainty can be observed for calculation of cumulative Pb mass in swale L2. At swale L2, mean Pb concentrations and the metal burdens in the deeper layers were higher than those in the top layers (0–5 and 5–10 cm) (Fig. 2). Metal accumulations in swale soils are difficult to estimate for older facilities because of difficulties in estimating the effects of historical changes (maintenance and reconstruction).

Moreover, simulated results for swale L1 and L3 had significantly higher Zn burden for the scenario with maximum input concentrations. This can be explained by the higher variation in input concentrations for the roofs (Table 4).

Fig. 5 shows that the cumulative soil metal burdens do not exceed the modelled burdens for the three swales when the use of leaded gasoline in the past is considered. In fact for swale L1, the metal burden from the model output is 3 times higher than the cumulative mass from soil samples. One explanation of this could be the overestimation of the swale age. The swale L1 is located at a commercial site, which was redeveloped after the initial construction of the road and this could have affected the swale soils. Moreover, the history of lead phasing out of gasoline is more complex than the simple margins prior and after 1995. The Pb content in gasoline was subject to a gradual decrease from the maximum of 0.7 g/L in 1970, to 0.15 g/L in 1980, and the total lead ban on all motor fuels in 1995 (SPBI, 2014). Although the computational method used here represents a rough estimation with high uncertainty, Fig. 5 offers some insight into the early years of operation of the swales studied, when higher Pb loads were entering the swales and contributed to the magnitude of Pb burdens currently found in swale soils.

The accuracy of conceptual (planning) models that calculate annual pollutant loads based on annual runoff volumes cannot be assessed by standard approaches (Marsalek, 1991). Rather, such models are subject to sensitivity analysis, as done here for the minimum and maximum stormwater concentrations, and assessed for applicability to specific problems, e.g., when conducting planning analyses of future developments (Larm, 2000). The StormTac Web has been used in Swedish municipalities for planning and design of STFs and their maintenance. As applied in this paper, StormTac Web served for planning-level analysis of the contamination of swale soils and the model results were further verified by soil chemistry data. This method assessed the soil chemistry of STF exposed to road runoff during a long-term (such as swales) and offered an opportunity to relate field results to the guideline values for contaminated soils. The method presented in this study could be used for planning grass swale maintenance, which would include screening the contamination of swale soils, in order to decide upon the removal of contaminated soils. This is important because the long-term performance of infiltration swales in pollution mitigation can degrade over time, because of soil saturation with pollutants (Leroy et al. 2016). Thus, the scalping of the top layer of bottom of the swale is recommended as a regular maintenance by some municipalities in order to restore the soil ability to mitigate pollutants through infiltration (Leroy et al. 2016).

5. Conclusions

The feasibility of estimating the long-term retention of three metals (Cu, Pb and Zn) by three grass swales was investigated by applying two methods: (i) sampling swale soils for the metals studied, and (ii) modelling the metal influx and retention by a source-based runoff model. In the former method, the runoff contributed metal burdens retained in swale soils were assumed to equal the current burdens reduced for the metal content in native soils. In general, the metal concentrations and metal burdens in swales reflected the quality of road runoff draining into the swales; the soils in swale L2 with the highest traffic density (ADT = 11,650) contained the highest metal concentrations and burdens. Furthermore, Pb concentrations in L2 exceeded the Swedish EPA permissible Pb levels for sensitive land use specified as 50 mg/kg. All the other metal concentrations in the swales studied ranged from 0.15 to 0.44 of the permissible metal levels for sensitive land uses, defined for Cu, Pb and Zn as 80, 50, and 250 mg/kg, respectively. The background levels of the metal studied in swale soils were assumed equal to the metal concentrations in the deepest layer studied (25–30 cm below the surface). In the modelling approach, the annual metal mass retained in swales was simulated by a source-based runoff quality model, StormTac Web. The annual loads produced by the model were multiplied by swale ages in years (38, 57 and 38 years, for L1, L2 and L3, respectively) to obtain total loads. Comparisons of measured and modelled HM loads retained by the swales indicated a fair agreement, described by ratios of MBs/MBm (metal burden simulated/metal burden measured) varying from 0.2 to 0.9, and average values for the three metal studied in swales L1, L2, L3 equal to 0.64, 0.50 and 0.59, respectively. Additional uncertainties in modelled burdens were caused by changes in environmental practices during the lifespan of swales (38–57 years), including the phasing of Pb out of gasoline in the mid-1990s, storage of used urban snow in swales, and maintenance done on swales. The study results indicate that the modelled data should be supplemented by soil chemistry data when addressing rehabilitation of older swales.

Declaration of interests

None.

Acknowledgement

This study was funded by the Swedish Research Council Formas, project 2015–778 and DRIZZLE–Centre for Stormwater Management, funded by the Swedish Governmental Agency for Innovation System (Vinnova), project 2016-05176. The authors would also like to thank the Technical and Urban Planning Departments of the Luleå municipality, especially Anders Stenlund, Mikael Nilsson, Maria Emanuelsson and Kari Kenttämaa.

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