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Analysis of swale factors implicated in pollutant removal efficiency using a swale database

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Abstract
Swales are traditional basic open-drainage systems which are able to remove stormwater-borne pollutants. In spite of numerous case studies devoted to their performances, parameters influencing the reduction of pollutant concentrations by swales remain elusive. In order to better characterize them, a database was set up by collecting performance results and design characteristics from 59 swales reported in the literature. Investigations on correlations among pollutant efficiency ratios (ERs) indicated that total trace metals (copper (Cu), zinc (Zn), cadmium (Cd), and lead (Pb)), total suspended solids (TSS), total phosphorus (TP), and total Kjeldahl nitrogen (TKN) exhibited many cross-correlated ERs. High ERs were observed for pollutants including a particulate form such as TSS (median ERs = 56%) and total trace metals (median ERs ≥ 62%), suggesting that these pollutants are efficiently trapped by sedimentation in swale bed and/or filtered within swale soil. Medium to high ERs were found for dissolved trace metals (median ERs ≥ 44%), whereas ERs for nutrient species were lower (median ERs ≤ 30%). The inflow concentration was identified as a major factor correlated to ER for most pollutants. For some pollutants, there is also a trend to get higher ER when the geometrical design of the swale increases the hydraulic residence time. Overall, this database may help to better understand swale systems and to optimize their design for improving pollutant removal.

Introduction

There is a growing trend in stormwater management to deal with the harmful effects of urbanization on runoff quantity and quality. This has led to the implementation of techniques that reduce stormwater discharges and pollution at subcatchment scale (Rodriguez et al. 2007;

The original version of this article was revised: The original publication of this paper contains an error. Correct presentation of Equation 1 is presented in this paper.

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Ahiablame et al. 2012; Hamel et al. 2013; Petrucci et al. 2013; Sage et al. 2015). Among these new techniques, the implementation of swales is becoming increasingly attractive for practitioners (Dierkes et al. 2000; Revitt et al. 2003; Caltrans 2004; Lucke et al. 2014; Woods Ballard et al. 2015). Over the past 30 years, swales have evolved from basic, shallow, open ditches used to attenuate peak flows (Wang et al. 1980; Clar et al. 2004), to systems capable of removing a wide range of stormwater-borne pollutants (Stagge et al. 2012). These contaminants are distributed between the dissolved and particulate phases in the runoff (Hvitved-Jacobsen et al. 2010; Gasperi et al. 2014). They comprise organic or inorganic substances such as (i) total suspended solids (TSS), (ii) nutrients, (iii) trace metals (e.g., lead (Pb), copper (Cu)), cadmium (Cd) and zinc (Zn)), and (iv) other micropollutants (e.g., polycyclic aromatic hydrocarbons (PAHs) and pesticides) (Ingvertsen et al. 2011; Kayhanian et al. 2012). The removal of these pollutants by swales involves a combination of distinct mechanisms generally related to hydraulic processes. Transport of stormwater over the swale bed promotes the sequestration of particulate pollutants

through sedimentation and trapping by vegetation (Deletic 1999; Purvis et al. 2018). Subsurface flows within the swale materials are involved in the implementation of retention, degradation, and transport mechanisms of pollutants in soil (LeFevre et al. 2014; Tedoldi et al. 2016).

The performance of swales for improving stormwater quality can be assessed using indicators such as efficiency ratios (ERs) (Urbonas 1994). The ER is a ratio between the mean pollutant input and output and provides a long-term estimation of overall treatment performance compared with the mean or median removal efficiencies (Roseen et al. 2009). Consequently, this parameter is useful to evaluate the potential treatment efficiency of swales (Fassman 2012), as well as to identify the factors influencing swale ability to remove pollutants.

Hydraulic processes within swales have a paramount role on contaminant transport. Hydraulic characteristics of the swale, such as the hydraulic residence time (HRT), are therefore considered as influential variables for pollutant removal (Yu et al. 2001; Johnson et al. 2003). Factors influencing HRT, such as swale geometry, swale materials, and swale vegetal cover, have been investigated with respect to pollutant removal in some studies (Yousef et al. 1985; Bäckström 2002; Roseen et al. 2009; Stagge et al. 2012; Winston et al. 2012a; Lucke et al. 2014; Powell 2015; Leroy et al. 2016). Results suggest that several design variables could be important for pollutant removal, such as the type of swale (Winston et al. 2012a), the swale length, the centerline slope (Yu et al. 2001), the presence of check dams (Stagge et al. 2012), the vegetation (Mazer et al. 2001), and the composition of soil materials (Hood et al. 2013). However, the contribution of these design parameters to swale efficiency remains to be fully established and there are still some knowledge gaps on how to design swales to remove pollutants efficiently.

The goal of this work was to gain insight into the pollutant removal capabilities of swales and the design factors that could affect them. For this purpose, the relationships between ERs and factors related to swale design and runoff water quality were investigated, using a swale database, established by collecting performance results and design features from various distinct swales characterized in the literature. The potential relationships between ERs for various pollutants were additionally studied to ascertain the effectiveness of specific treatment processes, such as sedimentation and physical filtration, on pollutant removal by swales. The results will provide useful information to scientists, engineers, and practitioners on the following: (i) the ability of swales to treat stormwater, (ii) the potential effect of relevant factors (e.g., design and incoming water quality) on pollutant removal by swales, and (iii) the future research needs to advance knowledge regarding swale design and treatment processes.

Materials and methods

Overview of the method used for collecting and analyzing data

The methodology used to collect and analyze data is summarized through a diagram (Fig. 1). This conceptual scheme gathered four steps together, presented below: (1) raw data collection, (2) data filtration to feed the final database, (3) data range representations, and (4) statistical approaches to analyze the data.

Data collection

To produce a database containing indicative information about pollutant removal in swales, a literature search was conducted using standard research databases (e.g., Web of Science, ScienceDirect, Google Scholar). For this purpose, we used keywords related to the pollution removal (e.g., "swale pollutant removal" and "swale performances"), the types of swale (e.g., "swale", "grassed swale", "dry swale", "wet swale", "bioswale", "bioretention swale"), and the inputs (e.g., "swale stormwater"). In addition to peer-reviewed journal papers, reports, technical notes, proceedings, master thesis, and Ph.D. dissertations were considered. This online search comprises studies conducted up to May 1, 2018. The produced data set has been filtered again to exclude the studies either with (i) weak number of monitoring rainfall–runoff events (i.e., $n < 3$) or (ii) several locations of outflow (i.e., two distinct outlets to discharge surface and subsurface outflows) or (iii) median values without access to raw data. Owing to their prevalence in the data set, we chose to consider mean concentrations as an estimator of water quality.

All selected studies used one of the three sampling designs that are often considered: flow interval, time-interval, and random grab sampling (Leecaster et al. 2002). Continuous auto-sampling using time-weighted or flow-weighted methods is usually considered to be a robust method to collect a representative sample of the complete runoff, which gives direct access to the event mean concentration (EMC) (Lee et al. 2007). Although the mixing of grab samples does not necessarily provide access to the EMC, studies using this sampling method were also included in the database. Among the selected studies, ER was used as the reference metric to assess swale performance (Roseen et al. 2009). The ERs were either retrieved from the screened documents or calculated from the concentrations using the following equation:

$$ER(\%) = \left(1 - \frac{AverageEMC_{outlet}}{AverageEMC_{inlet}}\right) \times 100 \qquad (1)$$

where EMC_{inlet} and EMC_{outlet} are the EMC measured at inlet and outlet of a swale, respectively.

Fig. 1 Diagram describing the
following methods used to filter
and analyze the database
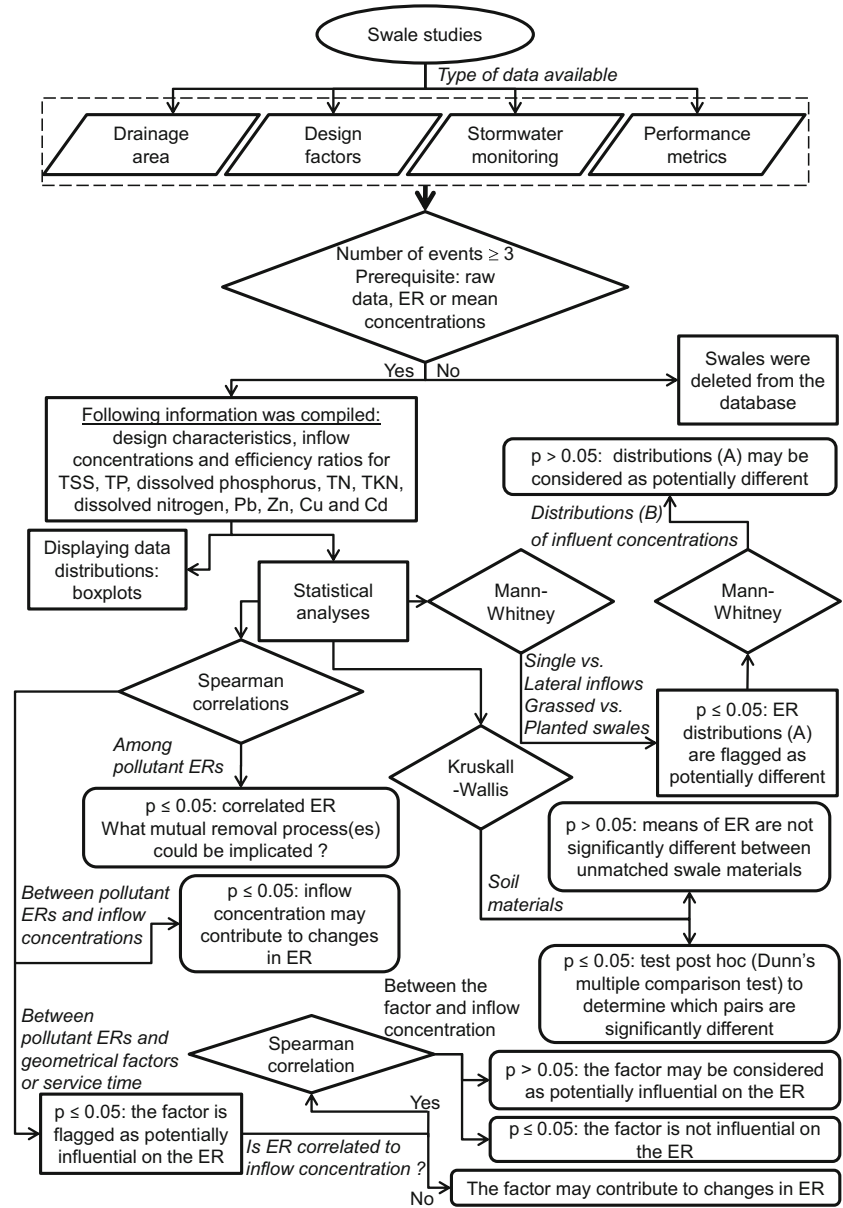
Several swales can be studied in a single document; the largest contributing document fed the database with six swales (Caltrans 2004). The whole selection process finally yielded 59 swales fulfilling the aforementioned criteria, against 103 swales at the beginning of the procedure. One important step was to ensure that each swale was only considered once, although it might have been described in distinct documents. Exceptions to this rule have been taken for either discontinuous monitoring program involving the same swale or swale design modifications (e.g., length divided by two (Larcombe 2003), retrofitting swale with check dams (Powell 2015)).

Following the database constitution, each swale has been referenced with an identification number (ID). The generic characteristics of the swales used for this study, such as the site name, the location (country), and the land use (i.e.,

catchment specifications), are summarized in Table S1. The facility scale was defined through three categories (Table S1): (i) *field-scale swale* refers to a monitoring facility which is already in operation (i.e., not designed by the investigators or for the purpose of a monitoring study), (ii) *technical field-scale swale* means that the facility or the contributing area has been designed or retrofitted by researchers and implemented on the field for the purpose of the study, and (iii) *technical-scale swale* defines a facility which has been designed and implemented in a pilot system, e.g., a test bed system (Hood et al. 2013) or any other type of device structures (Bäckström 2002; Li et al. 2016; Nara and Pitt 2006). Furthermore, the water supply of swales is also considered as a parameter, since runoff could be generated by a storm event or a simulated event.

Other relevant factors, including the surface of the drainage catchment (i.e., discharge area), the discharge ratio and the type of swale are also indicated in Table S1. When not reported, the swale area and the discharge ratio were determined by multiplying the swale top or average width by its length and by dividing the swale area by the discharge area, respectively. In addition, the active discharge area and active discharge ratio were calculated, when possible, by multiplying the impervious surface ratio by the discharge area and by dividing the swale area by the active discharge area, respectively (Table S2). Swales were additionally classified in four categories: "standard", "dry", "wet", and "bio-" swales. Standard swales may refer to natural grassed depressions or to graded grassed channels. Hence, all facilities called "swale", "grass swale", "grassy swale", "grassed swale", "planted swale", "vegetated roadside swale", and "grassy median" were indexed as standard swales. Dry swales can refer either to swales with a sandy soil, meaning that they are generally able to completely drain stormwater between two successive storm events (Winston et al. 2012b), or to facilities provided with an engineered media to promote better infiltration and filtration of pollution load (Revitt et al. 2017). Wet swales are facilities where wetland conditions dominate like ponded water or soil moisture near water saturation (e.g., swales built on soil with a high water table) and emergent vegetation (Winston et al. 2012a). Thereupon, swales described as wet or dry swales in the documents are indexed as such, as well as the sandy soil swale monitored by Hood et al. (2013), indexed as dry swale, given its good capacity of drainage. The remaining facilities were indexed as bioswales; this is the case for facilities called "bioswale", "biofiltration swale", "bioretention swale", "infiltration swale-trench", and "stone-lined swale". This latter type of swale brings together facilities that employ an engineered filter soil media below vegetal cover that may overlay an underdrain or be lined with an impervious material to prevent any groundwater contamination by controlling seepage flows (Toronto and Region Conservation Authority 2008).

The geometry of swales (swale length, top or average width, bottom width, cross-section shape, centerline slope, side slopes, and swale area) is schematically represented in Fig. S1. Information about the geometrical characteristics of each swale of the database is available in Table S2. Furthermore, when possible, the soil material of each swale was indexed according to the United States Department of Agriculture (USDA) textural triangle (Erickson et al. 2013) (Table S2). The rocky materials (e.g., gravel and limestone) were also indexed as such (Table S2). Concerning the vegetal cover, swales were merely indexed as "bare soil", "grassed swale", "planted swale", or "vegetated swale" (i.e., grassed or planted swale) (Table S2). An extensive description of the vegetal cover, when available in the reviewed studies, was also included in Table S2. In addition, the service time was taken into consideration to assess whether it could affect the

performances of swales. For this purpose, when possible, the ages of swales were estimated using the dates of their construction and the dates of the sampling periods (Table S2). An exception was made for swales studied by Leroy et al. (2016). Indeed, the soil used to build these swales has been excavated from a 6-year-old swale already in service; therefore, their service times were set to 6 years. Finally, inlet and outlet mean concentrations, as well as ERs and design considerations are gathered together for each pollutant in Table S3 to Table S17 (see Supplementary Material).

Data analysis

The efficiency of swales to treat stormwater was evaluated through ERs. In order to evaluate the effective capacity of swale to remove a specific pollutant, data were examined through box and whisker plots to show the shape of the distribution, the central value, and its variability. The censored or range values were not reported in the graphical representations.

The quality of incoming runoff, the service time, and possibly important design variables, such as active discharge area, active discharge ratio, soil material, vegetal cover, length, centerline slope, and side slope, have been selected to potentially highlight relevant effects on ERs. Because of the non-normal distribution of most parameters, non-parametric statistical approaches were used in this study, following the examples of Huber and Helmreich (2016) and Winston et al. (2012a).

The correlations between pollutant ERs were conducted using the Spearman's ρ correlation. Each coefficient obtained from this analysis measures the strength and direction of a monotonic relationship between two paired data (Pavlineri et al. 2017). Hence, for correlated data, degree of correlation was analyzed through ρ interpretation as a small ρ value ($\leq$ 0.39) means a weak correlation, a medium ρ value ($0.40 \leq \rho \leq$ 0.59) means a medium correlation, and a high ρ value (≥ 0.60) indicates a strong correlation (Huber and Helmreich 2016). Similarly, potential relationships between ERs and selected factors, i.e., influent quality, geometrical parameters, and service time, were investigated using the Spearman method. When the inflow concentration and a design parameter or the service time were both correlated to one pollutant ER, the potential relationship between these factors, i.e., whether they are dependent or independent, was subsequently examined using Spearman tests.

The comparisons of the distributions of two unmatched groups were done with the Mann–Whitney test (Mann and Whitney 1947). This concerns in particular the distributions of ERs between swales fed by single inflow and swales receiving lateral runoff, as well as those between grassed swales and planted swales. When multiple unmatched groups had to be compared (e.g., three or more distinct soil materials), the Kruskal–Wallis test was used (Kruskal and Wallis 1952). If significant differences were found between means of three or

more groups, further explorations were completed using the Dunn's multiple comparison post hoc test for comparing all possible pairs of groups.

All statistical calculations were performed using the GraphPad Prism 5.03 software (GraphPad Software, San Diego, CA, USA). A statistical test was considered as significant when the alpha p value is smaller than 0.05.

Results

Database elements and swale design parameters

Standard swales and bioswales each account for about half of the database (Table 1). The majority of swales were implemented on the field (97%) and supplied with natural storm events (93%). All swales from the database appear to be artificial, i.e., man-made. In terms of contributive area specifications, roadside swales are the most prevalent facilities in the database (64%), far before parking lot swales (27%), and suburban/urban residential swales (9%). This partly explains why swales in the database are predominantly fed by direct diffuse runoff (61% of the database), since roadside swales or parking lot swales generally receive lateral inflow from the edge of pavement (Fig. S1c). Pollutant removal processes occurring in the soil of swales are less often considered in the studies, as the surface waters were mainly sampled (78%). Furthermore, the database shows variability concerning the nature of analyzed pollutants; classification of main pollutants according to the number of collected data is the following: TSS > total phosphorus (TP) > total zinc (Zn_t) > Total Nitrogen (TN), total copper (Cu_t) > total lead (Pb_t) > total cadmium (Cd_t). Although TSS is nearly ubiquitous in the database (documented for 98% of the swales), data available for dissolved nutrients (nitrate and nitrite (NO_x^--N) and nitrate (NO_3-N)) are scarcer (documented for 32% of the swales), as well as those for dissolved trace metals (i.e., metals contained in water filtered through 0.45-μm membrane filters), documented for 8 to 11% of the swales.

Some geometrical parameters of swale design, such as the structurally interrelated parameters length, swale area, and discharge ratio, exhibit rather important dispersions, as demonstrated by the boxplot representation of the data (Fig. 2) and the determination of basic descriptive statistical parameters (Table S2). In the case of the swale length for example, its distribution displays a median of 51.5 m but it ranged between 4 and 1055 m (Fig. 2), with a coefficient of variation of 152% (Table S2). The discharge ratio ranges between 0.1 and 192%; its median (20%) is however at the upper boundary or slightly higher than the intervals (e.g., 5–10% or 5–20%) suggested in technical manuals (Beenen and Boogaard 2007; Credit Valley Conservation and the Toronto and Region Conservation Authority 2010). Conversely, the coefficients of variation for the side slopes and the top and bottom widths (about 60–100%), were smaller than for those for length and swale area (Table S2). This is likely due either to the narrow intervals which are suggested in the manuals to prevent erosion (for slopes) or to the visual definition of swales — a shallow ditch (for width). Regarding the shape of the cross section, most of the swales are either triangular (16 swales) or trapezoidal (24 swales) (Table S2). This again may reflect the content of design handbooks where the trapezoidal shape is advocated for its traditional good hydraulic performance and longer operation life (Woods Ballard et al. 2015).

Concerning the soils of the swales, only 37% of the facilities could be indexed according to their materials (Table S2). Sand was the most important soil separate. For example, 59% of the indexed swales were built with more than 45% of sand.

Table 1 Characteristics of the swale database

Parameter	Description (number of swales)
Total number of analyzed swales	59
Type of swales	Standard swales (24), dry swales (6), wet swales (5), bioswales (24)
Country	Australia (1), Canada (1), China (2), Denmark (2), France (2), New Zealand (2), Spain (2), Sweden (1), Taiwan (1), the UK (1), the USA (44)
Facility scale	Field (24), technical field (33), technical (2)
Water supply	Storm events (55), simulated events (4)
Flow direction	Single inflow (21), lateral inflow (36), lateral and single inflow (1), *n. a.* (1)
Sampling flow origin	Surface flow (46), subsurface flow (10), surface and subsurface flow mixture (3)
Catchment specifications	Road (38), parking lot (16), urban or suburban residential area (5)
Vegetal cover	Bare soil (1), grassed swale (44), planted swale (11), vegetated swale, i.e., grassed or planted facility (3)
Pollutant with available ER	TSS (58), TN (32), TKN (26), NO_x^--N (19), NO_3-N (19), TP (49), DP (27), Pb_t (31), Zn_t (37), Cu_t (32), Cd_t (20), Pb_d (10), Zn_d (11), Cu_d (11), Cd_d (5)

n. a. not available

Fig. 2 Design characteristics of swales registered in the database. **a** Boxplots for the design factors characterizing swale database. **b** Legend for boxplots
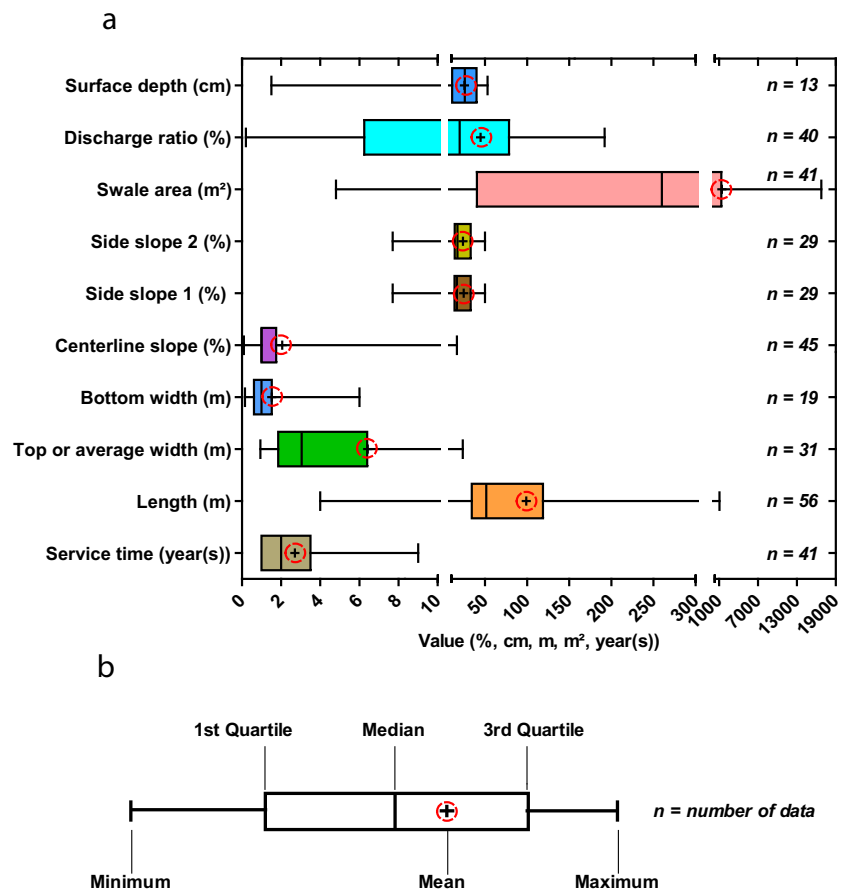
Regarding the service time, an age could be estimated for 69% of the swales. It is noteworthy that 75% of these facilities were less than 3.5-year-old and can be considered as quite young.

Most of the swales referenced in the database were covered by vegetation. Although grass was traditionally selected, mix of plants is increasingly incorporated into swales (planted swale) either to be adapted to the site conditions (e.g., hydric soils, deicing salts in stormwater) or to favor the sedimentation and improve the biodegradation of the contaminants (Leroy et al. 2017). As presented in Table 1 and Table S2, grassed swales prevail over planted swales and constitute at least 75% of the database. Among grassed swales, grass mixtures contribute to almost half of the vegetal covers (Table S2). Planted swales are quasi exclusively referred to wet swales or bioswales (Table S2).

Pollutant removal in swales

The proportions of positive ERs emphasized important variations (Fig. 3). Thus, for TSS, total Kjeldahl nitrogen (TKN), nitrate, total trace metals (i.e., Pb_t, Zn_t, Cu_t and Cd_t), dissolved trace metals (i.e., dissolved lead (Pb_d), dissolved zinc (Zn_d), dissolved copper (Cu_d), and dissolved cadmium

(Cd_d)), ERs are positive for over 60% of the involved swales. By contrast, 40 to 74% of the swales did not reduce NO_x^--N, TN, TP, and dissolved phosphorus (DP).

The pollutants including a particulate form (total trace metals and TKN) appear to be mainly retained in swales. In agreement with this conclusion, the ERs for TSS are found to be rather strongly positively correlated ($\rho \geq 0.60$) with those for TKN, Cu_t, and Cd_t (Table 2). TSS ERs are also correlated to those for TP, TN, Pb_t, and Zn_t, but to a lesser extent ($\rho < 0.60$). Conversely, the ERs for pollutants found in the dissolved phase (NO_x^--N, NO_3-N, DP, and dissolved trace metals) are not correlated to TSS ERs (Table 2).

Additionally, the high ERs reported for the dissolved trace metals suggest that they could be well trapped by swales. When the number of data is sufficient to perform correlation tests, the ERs of a dissolved form and the corresponding total form are strongly correlated. It is the case for Cu and Zn (Table 2). Furthermore, the ERs for the trace metals are often found strongly positively correlated between them, regardless of their form in runoff (Table 2).

Regarding nutrients, significant Spearman correlations are found between ERs for the nitrogen species TN and TKN and ERs for TP (Table 2). As expected, the ERs for TN are strongly positively correlated to those for other N-related parameters

Fig. 3 Range of ER values for pollutants from the database. Boxplots were used to display data spread (see Fig. 2b for the legend of a boxplot)
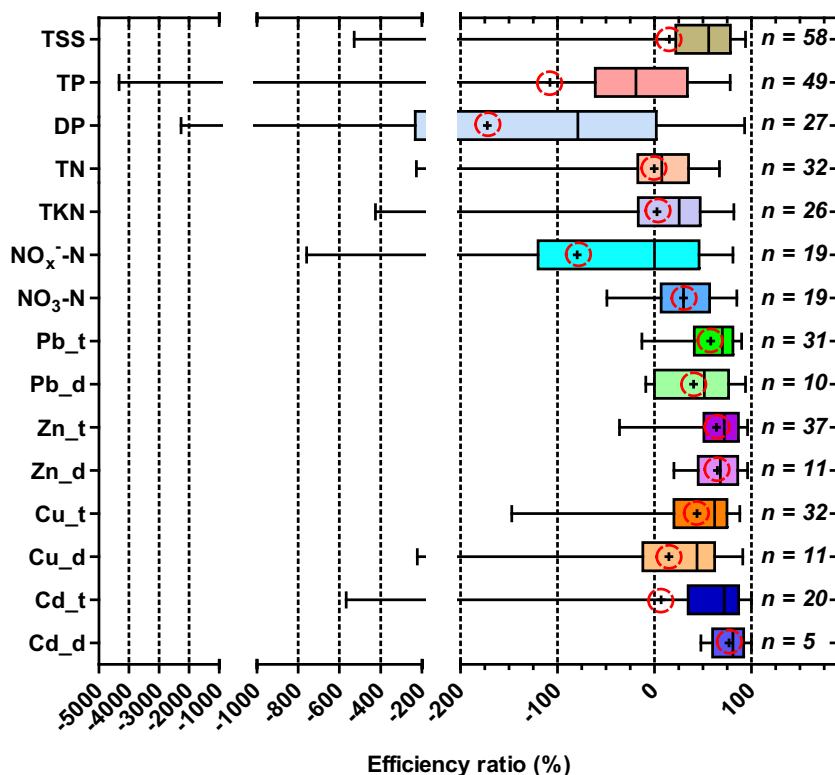
such as TKN, NO$_x^-$-N, and NO$_3$-N ($\rho \geq 0.60$). They are also strongly correlated with the ERs of Pb_t, Zn_t, and Cu_t (Table 2). The ERs for TP are correlated to those for DP and Cu_t (Table 2). Other significant positive correlations are found between ERs of TKN and Zn_t, ERs of Cu_t and Cd_t, and ERs of NO$_3$-N and Zn_t (Table 2).

A schematic graphical representation of the significant correlations between the ERs clearly illustrates the fact that TSS, TN, TKN, and the trace metals Zn and Cu exhibited a higher number of significant correlations compared with other pollutants (Fig. 4).

Factors involved in swale efficiency

The implication of various factors, i.e., inflow pollutant concentrations and various swale design characteristics, in swale efficiency was firstly studied through a correlation analysis. All pollutant ERs, except ERs for NO$_3$-N, Pb_t, Cd_t, and Zn_d, are significantly and positively correlated to their related inflow concentrations (Table 3). Swale length is correlated with ERs for TP, TN, NO$_x^-$-N, Zn_t, and Cu_t. Additionally, centerline slope is correlated with ERs for NO$_x^-$-N and Pb_t and side slope with ERs for TN (Table 3). While the active discharge area is positively correlated with ERs for TSS, TN, and TKN, pollutant ERs and the active discharge ratio have no significant relationship (Table 3). Furthermore, the service time is not significantly correlated with any pollutant ERs, except for TKN (Table 3).

To determine whether a correlation involving a design parameter could be biased by a similar distribution pattern between this parameter and the site mean concentration at swale inlet, additional Spearman correlations were performed (Table 4). Results indicated that inflow concentrations of TN and NO$_x^-$-N, unlike those of TP, Zn_t, and Cu_t, were correlated to swale length; it was also the case for inflow concentrations of TN and the side slope. Consequently, the swale length constitutes a statistically robust inflow-independent parameter implicated in pollutant removal for TP, Zn_t, and Cu_t, but not for TN and NO$_x^-$-N. This is also the case for the active discharge area for TSS, TN, and TKN because of the lack of significant correlations between this factor and inflow concentrations of these pollutants. Owing to the lack of correlation between the service time and the inflow concentrations of TKN, the service time can be considered as an inflow-independent parameter for TKN removal. Conversely, the side slope cannot be considered as an inflow-independent parameter for the removal of TN (Table 4).

Among swales, the differences of ER due to the direction of inflow or the swale soil materials were investigated through statistical comparisons between unmatched groups. Only ERs for Pb_t were found to statistically differ between swales fed by single inflow and those fed by lateral inflow (Table S18). Influent concentrations of Pb_t were by contrast not statistically different between the two types of swale feeding. No significant differences of ER were next observed among
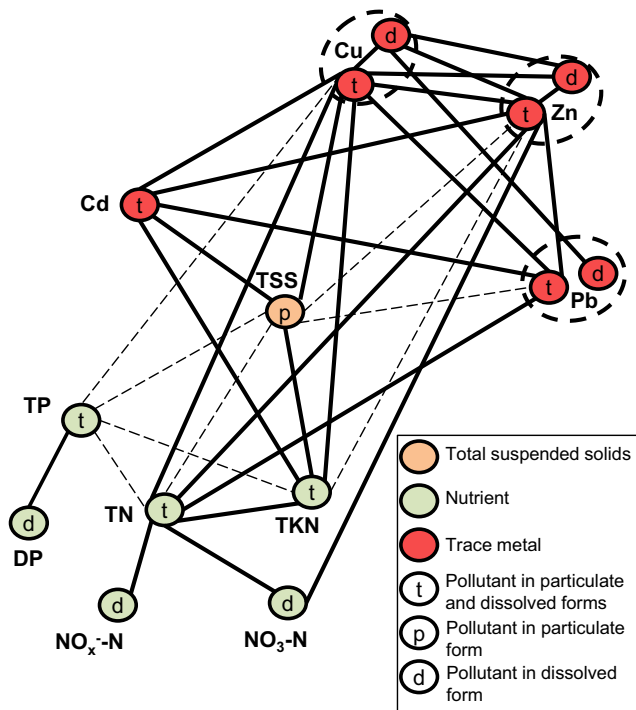
Table 2 Analysis of correlations between swale ERs for various pollutants

ER[a]		TSS	TP	DP	TN	TKN	NO$_x^-$-N	NO$_3$-N	Pb_t	Zn_t	Cu_t	Cd_t	Pb_d	Zn_d	Cu_d
TP	ρ	0.40													
	p	0.0054**													
	(n)	(48)													
DP	ρ	0.15	0.77												
	p	0.4547	<0.0001***												
	(n)	(26)	(26)												
TN	ρ	0.43	0.40	−0.13											
	p	0.0161*	0.0257*	0.6155											
	(n)	(31)	(31)	(17)											
TKN	ρ	0.66	0.45	0.12	0.80										
	p	0.0002***	0.0206*	0.7074	0.0003***										
	(n)	(26)	(26)	(13)	(15)										
NO$_x$-N	ρ	−0.20	−0.17	−0.33	0.77	0.07									
	p	0.4206	0.4912	0.2706	0.0007***	0.8346									
	(n)	(18)	(18)	(13)	(15)	(12)									
NO$_3$-N	ρ	0.27	0.02	−0.01	0.62	−0.24									
	p	0.2645	0.9233	0.9910	0.0478*	0.4434									
	(n)	(19)	(19)	(12)	(11)	(12)									
Pb_t	ρ	0.36	0.15	0.38	0.81	0.42	ND[b]	0.47							
	p	0.0484*	0.4618	0.1521	0.0008***	0.1187		0.0509							
	(n)	(31)	(25)	(16)	(13)	(15)		(18)							
Zn_t	ρ	0.58	0.31	0.15	0.85	0.54	ND[b]	0.66	0.67						
	p	0.0002***	0.0944	0.5435	<0.0001***	0.0238*		0.0028**	<0.0001***						
	(n)	(37)	(31)	(18)	(16)	(17)		(18)	(31)						
Cu_t	ρ	0.70	0.39	0.27	0.64	0.88	ND[b]	0.45	0.75	0.79					
	p	<0.0001***	0.0487*	0.2902	0.0076**	<0.0001***		0.0812	<0.0001***	<0.0001***					
	(n)	(32)	(26)	(17)	(16)	(15)		(16)	(29)	(32)					
Cd_t	ρ	0.74	0.15	0.20	ND[b]	0.74	ND[b]	−0.35	0.64	0.63	0.74				
	p	0.0002***	0.5567	0.5837		0.0023**		0.3129	0.0033**	0.0028**	0.0002***				
	(n)	(20)	(17)	(10)		(14)		(10)	(19)	(20)	(20)				
Pb_d	ρ	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]			
	p														
	(n)														
Zn_d	ρ	0.35	0.01	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]	0.82	0.78	ND[b]	0.50		
	p	0.3129	0.9730							0.0058**	0.0105*		0.1440		
	(n)	(10)	(10)							(10)	(10)		(10)		
Cu_d	ρ	0.62	−0.07	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]	0.92	0.98	ND[b]	0.88	0.66	
	p	0.0603	0.8382							0.0005***	<0.0001***		0.0016**	0.0306*	
	(n)	(10)	(10)							(10)	(10)		(10)	(11)	

*$p < 0.05$; **$p < 0.01$; ***$p < 0.001$ ($p < 0.05$, $p < 0.01$ and $p < 0.001$ mean higher level of significance than $p < 0.05$)

[a] Correlations between pollutant ERs were analyzed using the non-parametric Spearman rank method. The Spearman's rank correlation coefficient ρ, as well as the p value (italicized value) and the number of swales analyzed, are indicated in bold when the correlation is significant ($p < 0.05$); these data are additionally underlined when the correlation is considered to be rather strong ($\rho \geq 0.60$)

[b] ND, not done ($n < 10$)

Fig. 4 Graphical representation of correlations between pollutant ERs; strong correlations ($\rho > 0.60$) are represented by thick lines and weaker correlations ($\rho < 0.60$) by dashed lines. Pollutants were grouped by type (i.e., suspended solids, nutrient and trace metal), and speciation was indicated for each substance (i.e., total (t), particulate (p), and dissolved (d))

groups of soil materials through Kruskal–Wallis test analysis, except for TKN (Table S19). The Dunn's post hoc test however failed to highlight significant differences for TKN ERs between the different groups of soil materials (Table S19). Furthermore, ERs between grassed and planted swales were not statistically different, except for DP (Table S20); the distributions of DP inflows were however found statistically different between grassed and planted swales, suggesting that the type of vegetal cover is not an inflow-independent factor influencing DP ERs.

Discussion

The original database designed for this study provides representative information on the variability of swale performances through a data compilation from geographically and climatically diverse sources. Although three quarters of the swales were from the USA, our database is more extensive than the International Stormwater BMP Database (ISBD; Clary et al. 2017) which quasi exclusively contains swale case studies from the USA. Furthermore, our database compiles very diverse information in terms of swale design, installation or maintenance, as well as sites exposed to contrasted levels of runoff pollution and hydrological conditions. All these aspects

have enabled us not only to assess swale efficiency in removing pollutants but also to investigate the factors influencing this efficiency.

Processes driving the removal of pollutants in swales

The greatest swale efficiencies were mainly observed for the pollutants which tend to attach to the solid particles, i.e., TSS, total trace metals, and nutrients including a particulate form. These results are in accordance with previous findings from the ISBD; for example, Barrett (2008) has reported median concentration attenuations of 60%, 60%, and 62% for TSS (56% in our database), Zn_t (72% in our database), and Cu_t (62% in our database), respectively. In addition, median ERs of total trace metals fluctuate between 60 and 75% in our database and thus are close to the recurring removal threshold of 75% mentioned by Leroy et al. (2016). Concerning the effluent quality at surface outlet, TSS, TP, TN, TKN, and total trace metals are effectively removed by swales via physical mechanisms such as sedimentation and trapping by vegetation (Stagge et al. 2012; Boger et al. 2018), which are likely to be the main treatment processes occurring as water flows within the facility (Deletic 1999; Bäckström 2002; Deletic and Fletcher 2006). Additionally, positive correlations observed between ERs of total trace metals and TSS can be explained through their physical speciation in runoff, which means pollutants fully or partially attached to particles (Djukić et al. 2016). Subsequently, this may emphasize the importance of efficient removal of suspended solids, through settling or physical filtration, in mitigating total trace metal concentrations in swale outflows (Stagge et al. 2012).

The removal hierarchy of pollutants including a particulate form needs to be examined through their occurrence and partitioning in urban runoff. Concerning the removal of total trace metals in the present study, the grading established from the median ERs and interquartile ranges (IQR) follows the order: Zn_t > Pb_t – Cd_t > Cu_t. This ordering is slightly distinct from other swale studies where Cu_t may be ranked either ahead of Zn_t (Leroy et al. 2016) or Pb_t and Cd_t (Stagge et al. 2012; Revitt et al. 2017). According to previous studies, Pb should be mostly in the particulate phase (Legret and Pagotto 1999; Ingvertsen et al. 2011; Kayhanian 2012), whereas the trend for Cd is to be free or associated with dissolved solids in stormwater (Makepeace et al. 1995), as well as for Zn and Cu, which may have the highest dissolved concentrations in urban runoff (Gromaire-Mertz et al. 1999; Mosley and Peake 2001; Huber et al. 2016). Additionally, Cu is known to be mostly attached to organic particles and colloids smaller than 0.45 μm (Bäckström et al. 2006; Béchet et al. 2009). Since swales preferentially remove the particles larger than about 20 μm (Bäckström et al. 2006; Deletic and Fletcher 2006; Nara and Pitt 2006; Lucke et al. 2014), this may explain why Cu_t concentration is attenuated in swales to a lesser degree

Table 3 Analysis of correlations between ERs for various pollutants and swale parameters

Pollutant[a]		Swale Parameter						
		Inflow Concentration	Length	Centerline slope	Side slope	Active discharge area	Active discharge ratio	Service time
TSS	ρ	**0.57**	0.18	0.25	0.17	**0.37**	-0.28	0.04
	p	**0.003****	0.1799	0.1037	0.3686	**0.0068****	0.1071	0.8185
	(n)	**(57)**	(55)	(45)	(29)	**(53)**	(35)	(40)
TP	ρ	**0.46**	**0.38**	0.10	-0.11	0.27	0.10	-0.13
	p	**0.001****	**0.0078****	0.5255	0.6081	0.0758	0.6051	0.4733
	(n)	**(49)**	**(47)**	(39)	(24)	(43)	(30)	(33)
DP	ρ	**0.46**	0.20	0.39	-0.07	0.00	0.42	-0.34
	p	**0.0152***	0.3384	0.0651	0.8232	0.9947	0.1400	0.1076
	(n)	**(27)**	(25)	(23)	(13)	(26)	(14)	(24)
TN	ρ	**0.54**	**0.62**	-0.25	**0.51**	**0.49**	0.08	-0.14
	p	**0.0017****	**0.0003*****	0.2143	**0.0418***	**0.0069****	0.7041	0.5415
	(n)	**(31)**	**(30)**	(26)	**(16)**	**(29)**	(23)	(21)
TKN	ρ	**0.69**	0.28	0.32	0.41	**0.39**	0.05	**-0.62**
	p	**<0.0001*****	0.1721	0.1395	0.0621	**0.0467***	0.8272	**0.0075****
	(n)	**(26)**	(26)	(23)	(21)	**(26)**	(19)	**(17)**
NO$_x^-$-N	ρ	**0.52**	**0.51**	**-0.56**	0.32	0.22	0.17	0.12
	p	**0.0231***	**0.0248***	**0.0184***	0.2607	0.3875	0.5071	0.7132
	(n)	**(19)**	**(19)**	**(17)**	(14)	(18)	(18)	(12)
NO$_3$-N	ρ	0.17	0.43	-0.10	0.17	0.20	ND[b]	0.20
	p	0.4796	0.0667	0.6819	0.6321	0.4100		0.4555
	(n)	(19)	(19)	(18)	(10)	(19)		(16)
Pb_t	ρ	0.26	0.08	**0.47**	0.26	-0.04	0.23	-0.13
	p	0.166	0.6853	**0.0195***	0.3684	0.8338	0.4188	0.5065
	(n)	(31)	(28)	**(24)**	(14)	(30)	(15)	(27)
Zn_t	ρ	**0.67**	**0.49**	-0.05	0.24	0.30	0.18	-0.07
	p	**<0.0001*****	**0.035***	0.8085	0.3489	0.0797	0.4292	0.7080
	(n)	**(37)**	**(34)**	(28)	(17)	(36)	(21)	(32)
Cu_t	ρ	**0.52**	**0.56**	0.07	0.49	0.31	0.45	-0.03
	p	**0.0022****	**0.0017****	0.7465	0.0738	0.0902	0.0695	0.8912
	(n)	**(32)**	**(29)**	(25)	(14)	(31)	(17)	(29)
Cd_t	ρ	0.22	0.30	0.43	ND[b]	0.36	-0.06	-0.18
	p	0.3924	0.2469	0.1401		0.1357	0.8439	0.4680
	(n)	(17)	(17)	(13)		(19)	(13)	(19)
Pb_d	ρ	**0.82**	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]
	p	**0.0058****						
	(n)	**(10)**						
Zn_d	ρ	0.50	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]
	p	0.1440						
	(n)	(10)						
Cu_d	ρ	**0.82**	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]	ND[b]
	p	**0.0047****						
	(n)	**(10)**						

Correlations between pollutant ERs were analyzed using the non-parametric Spearman rank method. The Spearman's rank correlation coefficient ρ, as well as the *p* value (italicized value) and the number of swales analyzed are indicated in bold when the correlation is significant (p<0.05); these data are additionally underlined when the correlation is considered to be rather strong (ρ≥0.60 or ρ≤-0.60). [b] ND, not done (n<10). * p < 0.05; ** p < 0.01; *** p < 0.001

than those of other metals. Zn_t is generally found in higher concentration in water runoff compared to other trace metals (Kayhanian et al. 2012; Huber et al. 2016); this could partially explain the greater treatment efficiency found for Zn_t (Stagge et al. 2012). At first sight, the degrees of correlations between ERs of total trace metals and TSS are a bit surprising because

they deviate from the aforementioned traditional observations. TSS is weakly correlated with Pb_t, whereas Cu_t and Cd_t are strongly associated with suspended solids. These dissimilarities might be partly explained by the variability of particle size distributions (PSD) of water samples that likely populate the database used in this study (Ingvertsen et al. 2011; Kayhanian

Table 4 Analysis of correlations between inflow concentrations and various swale parameters

Pollutant	Tested correlation	Results
TSS	Inflow concentration vs. active discharge area	$\rho = -0.02$
		$p = 0.8771$
		$n = 52$
TP	Inflow concentration vs. swale length	$\rho = 0.07$
		$p = 0.6345$
		$n = 47$
TN	**Inflow concentration vs. swale length**	$\boldsymbol{\rho = 0.45}$
		$\boldsymbol{p = 0.015}$
		$\boldsymbol{n = 29}$
	Inflow concentration vs. swale side slope	$\boldsymbol{\rho = 0.59}$
		$\boldsymbol{p = 0.0165}$
		$\boldsymbol{n = 29}$
	Inflow concentration vs. active discharge area	$\rho = 0.36$
		$p = 0.0604$
		$n = 28$
TKN	Inflow concentration vs. active discharge area	$\rho = 0.12$
		$p = 0.5516$
		$n = 26$
	Inflow concentration vs. service time	$\rho = -0.38$
		$p = 0.1331$
		$n = 17$
NO_x^--N	**Inflow concentration vs. swale length**	$\boldsymbol{\rho = 0.46}$
		$\boldsymbol{p = 0.0465}$
		$\boldsymbol{n = 19}$
	Inflow concentration vs. swale centerline slope	$\rho = 0.433$
		$p = 0.0820$
		$n = 17$
Zn_t	Inflow concentration vs. swale length	$\rho = 0.33$
		$p = 0.0599$
		$n = 34$
Cu_t	Inflow concentration vs. swale length	$\rho = 0.22$
		$p = 0.2476$
		$n = 29$

Parameters analyzed correspond to inflow concentrations and swale design parameters or service time previously found correlated to ER for the same pollutant (see Table 3). Correlations were analyzed using the non-parametric Spearman rank method. The Spearman's rank correlation coefficient ρ, as well as the p value (italicized value) and the number of swales analyzed, are indicated in bold when the correlation is significant ($p < 0.05$)

et al. 2012). Considering the fine particle fraction often carries the largest load of particle-associated pollutant (Sansalone and Buchberger 1997; Lau and Stenstrom 2005), a predominance of small particles (e.g., < 63 μm) in stormwater inflow likely makes their removal by sedimentation difficult. To explain the aforementioned fluctuations of ERs, the resuspension processes of fine sediment after multiple rainfall–runoff events should be also carefully considered (Allen et al. 2017).

Our database indicates that trace metals in dissolved form could be efficiently removed by swales. Considering only the substances for which at least ten ERs are available in the database, the grading established from the median ERs and IQRs follows a similar order of removal rates as total trace metals: Zn_d > Pb_d > Cu_d. This ordering can be explained by the predominant speciation of these compounds in stormwater. To predict some metal species distributions in water runoff, LeFevre et al. (2014) used numerical computations; the results showed that Zn may be predominantly expected under the form of ions, whereas Pb and Cu could be more present as metal-organic or metal-inorganic complexes in water runoff. Similar assumptions were also reported by Yousef et al. (1987) to try to explain the removal efficiency higher for Zn_d than for Cu_d. In swales, metal species under the form of ions could be highly removed by adsorption processes (i.e., sorption to the swale media, adsorption on the grass, and uptake by plant roots). Conversely, the metal–organic and metal–inorganic complexes potentially carry a diffuse charge or zero charge making them less amenable to treatment by adsorption (Yousef et al. 1987). These elements may be proposed to explain the ERs higher for Zn_d than for Cu_d in our database. The few results available for Cd_d suggest that swales could remove it very well. Since Cd_d may be predominantly expected under the form of ions in water runoff (LeFevre et al. 2014), the aforementioned pattern underlines again that the speciation of dissolved metals likely has a substantial role in their removal by swales. It is noteworthy that all these expectations require further investigations, in particular concerning the treatment of infiltrating water which has only been sparsely investigated in the studies included in our database. A change in the physicochemical characteristics of the solute (e.g., pH, redox state and electrical conductivity) may also lead to poor sorption capacities of swale soil and to metal leaching (Ingvertsen et al. 2012; Tedoldi et al. 2016); these potential effects on the dissolved metals ERs should be therefore carefully examined.

The removal of nutrients in swales requires also to be discussed through their speciation. Regarding phosphorus, Morquecho et al. (2005) found that TP could be highly removed from stormwater through retaining particles taller than 20 μm; therefore, its attenuation in swales may mainly be driven by sedimentation (Boger et al. 2018). This hypothesis tends to be confirmed by two results of our database: (i) the significant correlation between ERs of TP and TSS and (ii) the wide extent of TP ERs. Large fluctuations of TP removal have been reported in other studies (Mazer et al. 2001; Yu et al. 2013; Jiang et al. 2017). They could be partly explained by the trend for phosphorus to be adsorbed to very fine particles (< 50 μm) (Stagge et al. 2012), which could hardly settle when they are smaller than 20 μm; efficiency of sedimentation is consequently reduced in this case. Meanwhile, TP contains a dissolved form (dissolved phosphorus) which remains more

challenging to remove according to the quasi-exclusive negative DP ER values stored in the database. DP attenuation could be attributable to geochemical processes occurring after infiltration, such as sorption to aluminum (Al) and iron (Fe) oxides, and precipitation (Li and Davis 2016). Nonetheless, the aforementioned poor performances suggest that swales could easily enrich stormwater with soluble phosphorus. For surface runoff, this export of phosphorus may result from the resuspension or scouring of organic matter; for percolating water, leaching of organic matter is generally incriminated (LeFevre et al. 2014; Li and Davis 2016). Such increases in discharge of DP should be a matter of concern for the preservation of aquatic ecosystems because it is more bioavailable than TP (Cederkvist et al. 2016).

Similar to phosphorus, the distinctive results for nitrogen species suggest that their treatment patterns depend on the nature of each component. The strong correlations between the ERs of TKN and TSS may be the consequence of the settling of TKN, caused by its particulate organic component (Stagge et al. 2012; Boger et al. 2018). By contrast, the slightly lower removal of TN coupled with a medium correlation with TSS ER may be explained by its predominant dissolved form in runoff (Taylor et al. 2005), which would involve a weaker removal by sedimentation than for TKN (Stagge et al. 2012). Although the ER distribution of oxidized nitrogen compounds (nitrate and nitrite) displays frequent positive values, the absence of correlation with TSS ER suggests that sedimentation should not be a particular contributor to their removal. The performances reported in this study could thus be expected to depend on infiltration, plant uptake and denitrification (Stagge et al. 2012; Winston et al. 2012a). Additionally, the variability of NO_x^--N ER validates the consensus regarding the complexity to predict nitrogen species removal in swales (Revitt et al. 2017). When soluble and organic nitrogen enrichment of stormwater occurs within the facility, this suggests that there is a supplementary source of nitrogen that is readily mobilized and mixed with runoff. The runoff enrichment in nitrogen could be due to the degradation of organic elements or discharge of high quantities of grass deposited on swale bed after frequent maintenance periods (Yousef et al. 1985; Yu et al. 2001).

Factors governing efficiency ratios

For investigating factors potentially modulating ERs in the present study, the preference was given to variables already be highlighted in previous studies. As reported by others (Yu et al. 2001; Barrett 2005; Stagge et al. 2012; Winston et al. 2012a; Leroy et al. 2017), the swale performances for pollutant removal can be affected by some design parameters and the water quality at the facility inlet. In this study, the potential influential variables on swale efficiency were classified into two categories: (i) the factors related to the drainage area such

as the active discharge area, the active discharge ratio, and the mean concentration at the swale inlet, and (ii) the major factors related to the swale itself, such as the swale length, the centerline slope, the side slope, the type of soil, the vegetation and the service time. In addition, as flow path of incoming water (i.e., single or lateral inflow) could drive the spatial distribution of pollutants in the swale bed (Tedoldi et al. 2017; Evans et al. 2019), we have investigated its potential effect on swale efficiency.

Our analyses point out that pollutant inflow concentration is the main factor correlated with the ER. Swales receiving stormwater rich in TSS, TP, DP, TN, TKN, NO_x^--N, Zn_t, Cu_t, Pb_d, and Cu_d tend to have high ERs. This suggests that their removal mechanisms are more efficient when the pollutant concentrations in water runoff are high. Such data also support the idea that low inflow pollutant concentrations may result in low ERs. This hypothesis is fully supported by the fact that negative ERs were often observed in low influent concentration conditions. For instance, negative ERs can be observed when the influent TSS concentration is in the range of 0–50 mg/L (Bäckström et al. 2006; Andrés-Valeri et al. 2014; Purvis et al. 2018). This enrichment of particles may be explained by the scouring of sediments during water flow in swales (Lucke et al. 2014).

With respect to the swale design factors, length was identified as a strong inflow-independent parameter positively correlated to the ER for TP, Zn_t, and Cu_t. These contaminants are partly in particulate form in runoff; therefore, increasing length probably results in an increase of the flow residence time and facilitates the settling of particles. By the way, swale length has repeatedly been considered as an influential factor on the trapping of particles (Yu et al. 2001; Bäckström 2002; Winston et al. 2016). However, sediment reduction as a function of length seems to reach an asymptote in sloped grassed areas (Deletic 1999), and most of our results show that installation of long swales (e.g., length > 15–30 m) is not necessary to efficiently remove particles (Lucke et al. 2014).

Some previous studies showed that increasing centerline slope could slightly attenuate TSS reduction in swales (Yu et al. 2001; Winston et al. 2016). Although it was difficult to ascertain this trend because of the small range of centerline slopes in our database (75% of the centerline slopes are below 2%), the ERs of a trace metal predominantly bound to particles, Pb_t, are positively correlated to the centerline slope. This may contradict the fact that lower slopes lead to lower flow velocities, which benefits to the sedimentation. By contrast, the negative correlation found between the ERs of dissolved nitrogen (NOx^--N) and the centerline slope validates the consensus that low flow velocities provide a better dissolved nitrogen removal (Yousef et al. 1987).

The active discharge area was positively correlated with the ERs of TSS, TN, and TKN. For a given storm event, an increase of the active drainage area will produce more water at

swale inlet. However, the active discharge ratio (defined by the surface of the swale over the active discharge area) seems to be a better predictor of the stormwater management capacities of swales because it links the sources of runoff with the surface of treatment. Our database however shows that this ratio has no significant effect on water treatment efficiency.

Concerning the vegetal cover, a comparison between the ERs of grassed swales and those of planted swales does not provide any significant inflow-independent difference. Therefore, our results do not support that a planted swale, with its potential deeper root network limiting the in-depth transfer of particles in soil (Leroy et al. 2016), could provide higher removal of pollutants. This conclusion may however be challenged by the relative low number of planted swales in the database.

Concerning the soil materials, results from the database do not display any difference related to the pollutant removal efficiency. This conclusion may however be challenged by (i) the relative low number of soil material data in the database and (ii) the fact that surface runoff was generally collected at swale outlet. In the same way, the ER for pollutants might be independent from the direction of inflow. Nonetheless, no test has been carried out on the same swale to ascertain this trend.

Swale efficiency can be expected to decline over time, due to the accumulation of pollutants in swale systems by settling and filtering processes (Leroy et al. 2016). Nonetheless, our data indicate that no discernible pattern emerges with respect to the service time, except for the removal of TKN. Since most of the swales in the database are young (i.e., 75% of the swales ≤ 3.5 years), further experimental investigations and modeling of swale systems could however be required to more definitively assess their long-term performances.

Limitations of the study

The present study suffers from various limitations arising from (i) the heterogeneity of monitoring methods used in each study to sample events and analyze pollutant concentrations and (ii) the adopted methodology to analyze the database. Concerning the first point, studies of the database exhibited various types of sampling method, which may affect the representativeness of resultant concentrations in terms of pollutant discharges by storm events. On this aspect, flow-weighted or time-weighted sampling might give access to EMC whereas manual (grab) sampling produces several subsamples which can be subsequently mixed to estimate an average concentration of pollutant discharge. Since most of the database studies used the first type of sampling, concentrations used to calculate the ERs might be regarded as representative of entire storm event discharges. Further differences may be attributable to sample preparation and sample analysis (Huber et al. 2016). For example, determining the concentration of a pollutant could be highly dependent on detection limits, notably for stormwater

contaminants found in trace concentrations (e.g., trace metals). In this study, very low concentrations of Cd in inflows and outflows (0.1–3 μg/L) may suggest that a considerable uncertainty could affect Cd ERs. Furthermore, the adopted methodology to analyze raw data from literature represents a significant part of overall uncertainty. The number of sampling events as well as the duration of the monitoring period (e.g., a case study can cover a part of a year or a single year or several years) differ between the studies, but the same weight was given to each ER. Since level of contamination as well as hydrological conditions could be highly variable for incoming runoff in swales, an ER calculated from a large number of sampling events would generally be more representative in the potential assessment of a site to treat stormwater (Fassman 2012). Previous studies suggested that reliable TSS, TN, and Zn site mean concentrations could be achieved through monitoring for 15 to 20 runoff events (Thomson et al. 1997; Drapper et al. 2000), knowing that this minimum threshold could be dependent on the nature of pollutant (McCarthy et al. 2018). In the database, several studies monitored a limited number of storm events (i.e., < 15); their site ERs may be therefore considered as less relevant than those of sites with large number of sampling events (i.e., ≥ 15). Additionally, the selection process of studies to feed the database did not consider hydrological conditions, i.e., inflow flow rates, initial soil moistures, and durations of events, which could directly impinge on ER. While these data were not systematically available in the studies, such information could be useful to explain high ERs for particulate pollutants in cases where most of the sampling events for a site had low flow rates, which are likely to favor sedimentation.

In addition, this study dealt with pollutant removal in swales only from the perspective of ER. Nonetheless, using this ER-based approach may mislead the evaluation of swale effectiveness because of its strong reliance on additional inputs, e.g., inflow concentrations and influent flow rates. This is why percent removals, notwithstanding their historic prevalence, are increasingly being criticized as stand-alone parameters (Barrett 2005; Wright Water Engineers and Geosyntec Consultants 2007). There is also a current need to investigate the management practice of sustainable urban drainage systems (SUDS) in terms of quality assessment to remove pollutant (facility efficiency) without relating to achievements in terms of project goals or effectiveness (Scholes et al. 2008; Fassman 2012). For this reason, other analytic metric tools have been developed such as the probability plots (Fassman 2012; Stagge et al. 2012), which do not rely on an arbitrary starting value. For evaluating effluent stormwater quality variability in the sight of water quality standards, probability plots focus on whether the discharges are likely to adversely affect the receiving ecosystems (Fassman 2012). A complementary method to investigate SUDS efficiency is to examine EMC distributions. This leads to explore whether the distribution of

effluent concentrations differs statistically significantly from the distribution of influent concentrations (Geosyntec Consultants and Wright Water Engineers 2011). Specific data from theses distributions (e.g., mean, median, IQR) could be also selected as a basis for a comparison procedure against regulatory environmental quality standard (EQS). This may help to assess the environmental risks due to stormwater discharges (Ellis et al. 2012). Nonetheless, applying one of these recent methods requires treating raw data, which were not always available in the studies included in our database.

Conclusion

In the present study, the efficiency of swales to remove TSS, nutrients and trace metals was investigated by establishing an original database, including data on the ERs, the swale design characteristics and the inflow concentrations. Concentrations of pollutants including a particulate form can be highly attenuated in swales, e.g., median reduction exceeds 50% for TSS, Pb_t, Zn_t, Cu_t, and Cd_t. In terms of removal processes, the strong statistical relationships between ERs of TSS and other contaminants that may be under particulate form support the idea that sedimentation and physical filtration are key processes explaining their sequestration in swales. The concentration reduction of dissolved contaminants is pollutant-dependent. The concentrations of dissolved trace metals can be efficiently reduced in swales (median reduction $\geq 44\%$), likely through adsorption processes, but the removal of dissolved nutrients, such as NO_x^--N and DP, displays very high fluctuations (IQR > 160%), which may constitute a challenge for further improvement related to swale design. Moreover, instability of pollutant ER may partly be explained by variations of the site mean concentration, emerging as the most striking influential factor towards ER. By contrast, various geometrical design factors, such as the length, centerline slope, side slope, and discharge ratio, were not correlated to the ERs for most of pollutants. Practitioners should however consider all design options that enhance the hydraulic residence time in swales, since it could improve the removal of dissolved nutrients and some particulate pollutants. In addition, the absence of significant differences of ERs among soil materials is likely due to the limited information collected in the database. In the same way, drawing conclusions related to the long-term efficiency of swales is tricky because of the young ages of the swales indexed in the database. Future works are required to gain insight on the swale capacities to remove pollutants present in infiltrating water and to assess the long-term performances of swales. In addition, further studies could investigate the capacities of swales to remove micropollutants, such as PAHs and pesticides, which are a matter of growing environmental health concern.

Compliance with ethical standards

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