Assessing the Impact of Swales on Receiving Water Quality

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Abstract
Swales are one type of sustainable drainage system (SuDS) which contribute to the management of water quality in receiving waterbodies. Using a semi-quantitative approach, an impact assessment procedure is applied to the residual water quality that is carried forward to surface waters and groundwaters following treatment within a swale. Both volumetric and pollutant distributions are considered as stormwater passes through the swale system. The pollutant pathways followed by TSS, nitrate, chloride, metals (Cd, Cu, Pb, Zn) and polyaromatic hydrocarbons (PAHs) are determined for a swale receiving highway runoff. For TSS, metals and PAHs between 20% and 29% of the total mean influent pollutant load is predicted to be directed to infiltration through the underlying soils compared to between 4% and 16% of chloride and nitrate. Although surface water impacts are deemed possible, the discharges of swales to groundwaters are assessed to represent a negligible impact for effectively maintained systems.

Keywords: Swales; stormwater; pollutant removal; receiving water quality; groundwater; impact assessment approach.

1. Introduction

To attenuate stormwater flows and to facilitate the removal of pollutants, sustainable drainage systems (SuDS) may utilise infiltration techniques either alone or in combination with other treatment processes. Swales are vegetated open channels designed to convey, treat and attenuate stormwater. They achieve this through sedimentation, filtration by plants and plant material, evapotranspiration and infiltration into the underlying soil. The incorporation of check dams and dense vegetation encourage many of these processes but must not impede the design flow rate. Where development density, topography and depth to water table permit, swales are preferable stormwater conveyance systems compared to concrete channels as they contribute to the reduction of impervious cover and possess the additional benefit of enhancing the natural landscape and providing aesthetic and biodiversity improvements (Kazemi et al., 2011). The ability of swales to transport stormwater makes them ideal components of treatment trains designed to efficiently enhance receiving water quality.

There are three main types of swales referred to as ‘standard conveyance swales’, ‘dry swales’ and ‘wet swales’ (Woods-Ballard et al., 2015). Standard conveyance swales typically possess broad shallow vegetated channels which provide an effective way of conveying runoff from a drainage area and facilitate infiltration into permeable soils. Dry swales are designed to incorporate a filter bed of prepared soil (overlaying an under-drain system) which provides additional treatment and conveyance capacity. Wet swales maintain wet and marshy basal conditions through a combination of shallow gradients and underlying impermeable soil or a liner to inhibit/prevent infiltration. There is therefore unlikely to be an impact on groundwaters from the use of wet swales but both standard conveyance swales and dry swales have the potential to contaminate groundwaters. To permit efficient pollutant retention by the underlying soil, the groundwater table should be between 1 and 1.5 m below the base of the swale. Infiltration swales are not recommended for brownfield sites or for runoff hotspots where a risk of groundwater pollution exists.

In this paper we present an analysis of how surface and sub-surface processes contribute to the performance criteria associated with ‘standard conveyance’ swales and examine the development of an impact assessment methodology to determine how swales may influence both adjacent ground- and surface-waters when receiving contaminated highway runoff containing total suspended solids (TSS), nitrate, chloride, metals (Cd, Cu, Pb and Zn) and PAHs (fluoranthene). An impact assessment procedure for surface water quality risk exposure proposed by Ellis et al. (2012), and recommended for application in the recent UK national SuDS manual (2015), initially identifies the roles played by different impermeable urban land uses in influencing the types and levels of pollutants flushed to the drainage network. This is achieved through the derivation of a pollutant index (PI), with values between 0 and 1, based on the reported event mean concentration (EMC) distributions for a given pollutant or pollutant group and the likelihood that the 50th percentile EMC values will exceed receiving water body environmental quality standards. Subsequently, consideration of the relative treatability of the runoff pollutants, such as would occur in a SuDS facility, enables the reduction in the PI values to be gauged at the different critical points in the treatment system and hence their impact on a receiving water body to be assessed. The extent of the risk to surface waters can be assessed by comparing the derived PI value with a recognised value for receiving water quality and ecological status to provide a first level screening procedure for risk assessment. In this paper, the approach is extended to groundwaters using swales as an example of SuDS which have the potential to compromise the status of both surface and sub-surface waters. The establishment of a relationship between a calculated PI value and the impact on groundwater quality is described in Section S1 of the Supplemental Material.

2. Methodology for the assessment of hydrological performance and pollutant removal mechanisms

2.1 Hydrological performance and modelling approaches

The volumetric performances of swales have been found to be related to the sizes of rainfall events with smaller events often producing no discharge compared to larger storms which may cause a swale to act as a conveyance device with a more limited pollutant reduction capacity. Davis et al. (2012) observed that incoming volume attenuation (27 - 63%) was mainly effective between mean swale water depths of 1.3 cm and 2.8 cm. This range of volume reductions compares reasonably well with other reported values of 30% (Rushton, 2001), 46% (Deletic, 2001), 33% (Bäckström, 2002), 47% (Barrett, 2005), 45% (Ackerman and Stein, 2008) and 48% (ISBMPD, 2011). The available data indicates that a mean volume reduction of 42% with an associated standard deviation of ±7.3% represents the typical performance of a grassed swale with respect to volume reduction.

The main water loss pathway within swales is by infiltration as water flows down the length of the swale with an additional contribution due to evapotranspiration which will be strongly influenced by local weather conditions. The recommended infiltration rate for optimum swale performance is 12.7 mm h^{-1} (US EPA, 1999; MDE, 2000) which corresponds to the soil characteristics associated with loams, sandy loams and sands. Although coarse sandy and gravelly soils would be better draining they are unlikely to support the required dense vegetation. Davis et al. (2012) have reported infiltration rates of between 15 and 30 mm h^{-1} for captured storm events. This compares favourably with saturated hydraulic conductivity values of 3.4 and 10.9 mm h^{-1} for loam and sandy loam soils, respectively. Fassman and Stokes (2011) have measured average dry weather evapotranspiration rates in lined grassed swales of <0.5 mm d^{-1} in winter and ~1 mm d^{-1} in early summer in the sub-tropical climate found in the north island of New Zealand. For considerably larger swale wetland systems in Canada, Carlson Mazur et al. (2014) estimated mean daily evapotranspiration rates of between 4.0 and 6.6 mm d^{-1}. Comparison of the quoted infiltration and evapotranspiration rates indicates that water losses by the latter route are typically less than 10% of those by infiltration. Therefore, on the basis that 42% of the incoming flow volume can be retained by a swale, the maximum amount lost due to evapotranspiration would be expected to be 4.2% with 37.8% being infiltrated.

2.2 Pollutant removal mechanisms and modelling approaches

Pollutant removal within swales is enhanced during small storm events (which produce the majority of annual runoff in most areas) and where contact time is increased by the presence of check dams to reduce flow velocity (≤ 0.5 m s^{-1}). Additional beneficial factors are gentle longitudinal slopes (<1%), the presence of permeable soils (infiltration rate ≥ 15 mm h^{-1}) and dense vegetation cover. Through the consideration of pollutographs, Stagge et al. (2012) demonstrated that for particulate associated pollutants, infiltration is the initial removal mechanism followed by sedimentation and vegetative filtration. Infiltration removes pollutants by the filtering of particles through the underlying soil matrix and the adsorption of dissolved contaminants to soil media, particularly in the first 20-50 cm below the surface (Weiss et al., 2010). Alkaline soils and sub-soils promote removal and retention of metals. In addition to infiltration, pollutants can be removed by surface processes including vegetative filtration (mainly particles), sedimentation (of solid particles) and plant uptake (particularly nutrients; some metals) (Abida and Sabourin, 2006; Bäckström, 2002; Schueler, 1987; Yu et al., 2001).

2.2.1. TSS removal

Field and laboratory scale studies suggest that the primary treatment mechanism for particulate associated pollutants within swales is sedimentation, with vegetative filtration playing a much smaller role (Deletic, 2001; Bäckström, 2002, 2003; Deletic and Fletcher, 2006). Sediment tracing techniques have shown that deposition follows a first order mechanism (Allen et al., 2015) with pollutant removal occurring predominantly in the first 20 m length of a vegetated swale (Barrett et al.,1998) although subsequent sediment resuspension and conveyance out of the swale has been reported (Allen et al., 2015). Therefore a range of pollutant removal responses can be expected with the reported removal efficiencies also being influenced by swale design and maintenance procedures, by monitoring practices and measurement techniques (comparison of influent and effluent EMCs being most common), by the dissolved or particulate associated status of the pollutant (size and density in the case of particles) and pollutant influent concentrations. The variabilities in reported TSS removal efficiencies by swales cover the range 22% to 98% (Table S3; supplemental material). The derived overall average is 74.3% ± 13.6% although this excludes the lowest values (22%-31%) reported by ISBMPD (2014) as these values do not seem to be compatible with the more elevated values for metals provided by the same database.

2.2.2. Nitrate removal

Both positive and negative removal efficiencies have been reported for nitrate with the negative values being attributed to initial nitrate retention followed by subsequent mobilisation in a following storm event. This has been reported to be most common for a small number of summer storms (Stagge et al., 2012). High variabilities for nitrate removal efficiencies within swales have also been partly attributed to the presence of extraneous organic matter, such as grass or other vegetation, which can leach significant quantities of nutrients (Yu et al., 2001). Despite the reported variations, the consensus is that swales have the ability to achieve moderate removal of nitrate during the majority of storm events, particularly when check dams are installed (Stagge et al., 2012). The same authors identified infiltration as an important removal mechanism for nitrate in swales through consideration of pollution-duration curves. A mean removal efficiency of 40.6% ± 14.0% is predicted from the available published data (Table S3; supplemental material) and this value and range have been used in the assessment procedure applied in this paper.

2.2.3. Chloride removal

The removal of chloride by swales has been less extensively investigated than nitrate despite high levels being observed in urban runoff, particularly after the application of de-icing salts. From a comprehensive investigation of the impact of 45 storms on swales, Stagge et al. (2012) concluded that the observed negative removal efficiencies (-4410% to –78%) were due to chloride release throughout the year following within-swale winter accumulation during non-recorded storm events. Clearly, the high solubility of chloride does not support its attenuation by the removal processes operating within swales but there is evidence of some retention particularly in fine grained roadside soils (Lundmark and Olofsson, 2007). Positive swale performances, although with low chloride removal efficiencies of less than 10%, have been predicted by US EPA (1999). Given the evidence for retention within a swale followed by subsequent remobilisation an overall average chloride removal efficiency of 10% (although without any associated variability) is proposed for use in this study.

2.2.4. Removal of PAHs

There is limited published data on the removal of hydrocarbons by swales (Table S3; supplemental material) and the selected average removal efficiency of 71.0% ± 12.7% is representative of the wide range of contributing compounds with differing physicochemical properties. The only data reported specifically for PAHs refers to a study conducted over 12 month periods using mesocosms simulating road-side swales (Leroy et al., 2015). By collecting effluents at a sub-surface outlet, percentage removals of total PAHs, phenanthrene, pyrene and benzo(a)pyrene were found to be consistently in excess of 99%. Hydrocarbon retention within a swale will be by sedimentation, infiltration, plant uptake and vegetative filtration with possible contributions by volatilisation for the lighter compounds. Microbial degradation will also become important during the infiltration process over longer exposure durations but will be less prominent during individual storm events, which are the main objective of this work.

2.2.5. Removal of metals

The design and location of swales have been demonstrated to influence the removal of metals with the incorporation of check dams and the presence of a prior filter strip enhancing the overall removal of Cd, Cu, Pb and Zn, which are commonly found in highway runoff (Stagge et al., 2012). These four metals have been selected for further study in this paper. Where metals occur in the dissolved form they will be preferentially removed by infiltration with sedimentation and vegetative filtration also becoming important for particulate associated metals. Cadmium and Zn have the greatest affinity for the soluble phase in highway runoff with Pb being most strongly bound to the solid phase (Morrison et al., 1990). The reported percentage treatment efficiencies for the selected metals (Table S4; supplemental material) indicate that Zn, which is found in the highest concentrations in highway runoff, demonstrates the most effective removal in grass swales. Consequently Zn is allocated the highest overall percentage removal efficiency of 70.7% ± 12.7% followed by Cu (62.2% ± 15.8%) and Pb (58.0% ± 18.5%) with Cd (51.0% ± 7.9%`) being the most difficult metal to remove in swales.

3. Methodological results and discussion

There is currently insufficient field data to support the development of an evidence based derivation of the different pollutant removal processes within swales. The premise adopted within the described theoretical approach is that the pollutant loads in the surface flow and those directed to the sub-surface will be proportional to the flow distribution within the swale. It is also assumed that the swale is operating at its design efficiency.

3.1 Outcomes for TSS

The typical mean distributions of flow volumes within a swale are shown diagrammatically in Figure S1a (supplemental material) with 58.0% of incoming flow leaving the swale and of the flow volume retained, 4.2% is assumed to be lost by evapotranspiration leaving 37.8% to infiltrate into the soil. It has been estimated that on average 74.3% of the TSS entering a swale will be retained by processes including infiltration, vegetative filtration and sedimentation. Given the distribution of flows, it can be deduced that of the incoming TSS load retained in the swale, 60.5% (58/95.8) will be carried within the horizontal flow and will be available for removal by surface processes (45.0%) including vegetative filtration and sedimentation. In comparison, 39.5% (37.8/95.8) of the TSS load (29.3%) will be directed to sub-surface flow and be removed during the infiltration process. These results are shown diagrammatically in Figure S1b (supplemental material).

If the swale is being used to treat the runoff from a major highway, a pollution index of 0.8 has been deemed appropriate to indicate the polluting potential arising from this source (Ellis et al 2012). The overall 74.3% reduction in the TSS surface load leaving the swale effectively diminishes the PI value for the discharged runoff to 0.21 (Table 1), which without further treatment, would be expected to exert a borderline high impact on a receiving water equivalent to a small reduction in pollution tolerant taxa (Ellis et al 2012).

To assess the impact of swale discharges to groundwater, it is necessary to consider only the infiltrating water and how this will be further decontaminated as it penetrates the sub-soil system. The PI for the highway runoff retained within the swale will be 0.59 (the difference between that entering the swale [0.8] and that discharged from the swale [0.21]) and when directed to infiltration would be reduced by 39.4% (29.3/74.3) to 0.23 prior to leaving the swale (Table 2). The TSS would be further attenuated by efficient filtering of the suspended particulates during passage through a minimum depth of 1-2 m of underlying permeable soil. Laboratory studies have

Table 1. Predicted impact of mean swale pollutant discharges on receiving surface waters following treatment of major highway runoff.

	Swale removal efficiency (%)	% removed by surface processes	% directed to infiltration	PI for major highway	PI for water discharged from swale	Potential impact for receiving surface waters
TSS	74.3	45.0	29.3	0.8	0.21	Borderline/ High
NO_3	40.6	24.6	16.0	0.1	0.06	Negligible
Cl	10.0	6.1	3.9	0.8	0.72	Severe
Cd	51.0	30.9	20.1	0.5	0.24	High
Cu	62.2	37.6	24.6	0.6	0.23	High
Pb	58.0	35.1	22.9	0.5	0.21	Borderline/ High
Zn	70.7	42.8	27.9	0.8	0.23	High
PAH	71.0	43.0	28.0	0.8	0.23	High

shown that gravel based soil columns are effective in removing TSS from stormwater in the first 0.5 m (Hatt et al., 2007) and therefore a pollution mitigation index (PMI) value of 0.1 is allocated for this process. The PMI value provides an indication of the level of treatment (Ellis et al., 2012) on a scale of 0.0 to 1.0 as shown by the allocated values for different pollutants in Table 2. Lower values represent higher treatment efficiencies. The PI value after infiltration is obtained by multiplying the PI for swale discharge directed to infiltration (0.23) by the PMI value (0.1) so that in the case of infiltrating water reaching the groundwater the PI for TSS is reduced to 0.02 (Table 2). Comparison of this very low predicted PI value with the values reported in Table S2 (supplemental material) would suggest TSS concentrations of the order of 4 mg l^{-1} which, in the absence of relevant environmental standards would not be expected to exert a detrimental impact on groundwater quality (Table 2).

Table 2. Predicted impact of mean pollutant levels on swale infiltration and on groundwater following treatment of major highway runoff.

	PI for water retained in swale	PI for water directed to infiltration (39.4% of that retained within swale)	PMI for infiltration	PI for ground-water impact	Potential impact for ground-water
TSS	0.59	0.23	0.1	0.02	No detrimental effect
NO_3	0.04	0.02	0.9	0.01	Negligible
Cl	0.08	0.03	0.9	0.03	Negligible
Cd	0.26	0.10	0.3	0.03	Negligible
Cu	0.37	0.15	0.2	0.03	Negligible
Pb	0.29	0.11	0.2	0.02	Negligible
Zn	0.57	0.22	0.2	0.04	Negligible
PAH	0.57	0.22	0.1	0.02	Negligible

3.2 Outcomes for nitrate, chloride, metals and PAHs

The results for these pollutants are summarised in Tables 1 and 2. The solubility of nitrate precludes any removal by sedimentation with plant uptake and vegetative filtration being responsible for the average 24.6% removal of the nitrate load from the horizontal flow through the swale. The additional 16.0% nitrate retention within the swale will be directed to infiltration where denitrification will be possible at those soil depths where the decomposition of organic material has resulted in a chemically reducing environment. Because a highway does not represent a major source of nitrate, a PI value of 0.1 has been assigned leading to a discharged PI of 0.06 which is equivalent to a negligible impact on a receiving waterbody. A reduced PI value (from 0.04 to 0.02) is also predicted for the infiltrating water although this will receive only minimal additional treatment (PMI 0.9) due to the affinity of nitrate for the dissolved phase. Nevertheless, the resulting PI (0.01), corresponding to a nitrate concentration of around 1.0 mg l^{-1} (see Table S2; supplemental material), would have a negligible impact on the groundwater quality even if it were to be used as a drinking water source when compared with a threshold value of 50 mg l^{-1} which exists for this purpose.

Chloride possesses similar solubility properties to nitrate but differs in that highways represent a major source of chloride during winter months, due to de-icing activities, when a pollution index of 0.8 is appropriate. The same physical removal processes are relevant for both nitrate and chloride but the chemically conservative nature of chloride results in less efficient removal by swales with an overall retention of only 10%. The contributions provided by the surface and sub-surface removal processes are shown in Table 1. The overall effect of swale treatment on chloride would be to only reduce the PI of the discharged water to 0.72 which would represent a potentially severe impact if discharged to a receiving water and ideally requires further treatment before release. Although only a small proportion of the incoming chloride (3.9%) is directed to infiltration within a swale, its inability to be attenuated by soils (Pitt, 1994) enables easy travel to shallow groundwater, and the allocation of a PMI value of 0.9, resulting in a PI value for chloride reaching the groundwater of 0.03. This is equivalent to a chloride concentration in the region of 20 mg l^{-1} (see Table S2; supplemental material) which would not be expected to have any detrimental impact given that a guideline of 250 mg l^{-1} exists for chloride in drinking water, above which there is the potential for taste problems to arise.

Metals in runoff may be either particulate associated or in dissolved form. Both forms will be removed in swales by vegetative filtering and infiltration with additional retention by sedimentation for particulate associated metals and by plant uptake for dissolved metals. Metal adsorption during infiltration occurs with increased efficiency at higher pHs (Elliot et al 1986) but adsorption to soil particles is believed to take place predominantly in the top 20-50 cm (Mikkelsen et al 1997; Dierkes and Gieger 1999; Dechesne et al 2004). Although preferential removal of particulate associated metals occurs in swales, in this paper we consider total Cd, Cu, Pb and Zn for which the representative swale removal efficiencies are shown in Table S4 (supplemental material).

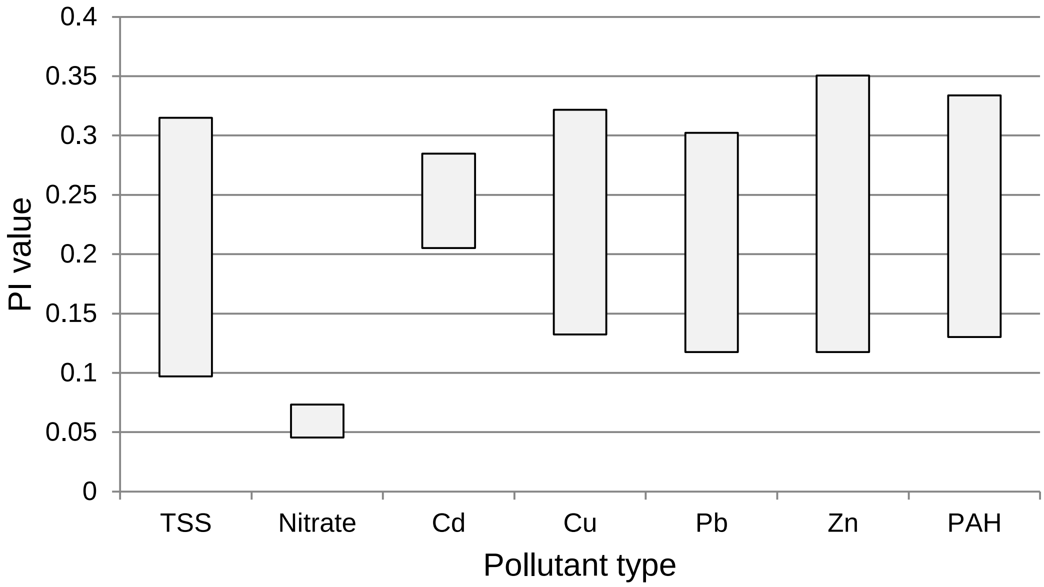
Consideration of the metal removal pathways within swales provides the results identified in Tables 1 and 2 indicating that their main impact is likely to be as a result of discharges directed towards surface receiving waters. Cd, Cu and Zn are predicted to represent a considerable threat to the impairment of receiving water quality with the lower impact allocated to Pb being partly associated with a lower PI value as a consequence of its phasing out as a petrol additive. It is clear that ideally where a swale acts as the first treatment component for runoff from a major highway, there should be further SuDS treatment of metals to protect the quality of surface receiving waters. In contrast, the infiltrating water within a swale is predicted to consistently have a negligible impact on groundwaters regardless of possible future interactions with surface waters or use as a drinking water source. The metal concentrations expected to be discharged to groundwaters located directly below swales are of the order of 0.03 µg l^{-1}, 1 µg l^{-1}, 2.5 µg l^{-1} and 12 µg l^{-1} for Cd, Pb, Cu and Zn (see Table S2; supplemental material), respectively and will be subject to further dilution during the mixing process.

Given the potential for the deposition of oils and grease on a major highway surface and the presence of PAHs in vehicle emissions, a PI of 0.8 for these compounds derived from this source is considered appropriate. Fluoranthene has been selected as a representative PAH given its molecular size and physicochemical characteristics and its known presence in vehicle emissions. Fluoranthene would be subjected to surface and sub-surface treatment during passage through a swale with the relative contributions to hydrocarbon removal being shown in Table 1. The overall effect of swale treatment on the discharged surface water would be a reduction of the PI to 0.23, which if passed directly to a receiving water would represent a substantial PAH impact such that further treatment would be beneficial. The fluoranthene pollution index for the runoff directed to infiltration would be reduced by 42.3% from 0.57 to 0.22 (Table 2) before entering the soil system where the hydrophobic fluoranthene would be expected to be efficiently removed during passage through a 1-2 m depth of permeable soil meriting a PMI value of 0.1. The resulting PI value for fluoranthene reaching the groundwater is 0.02 (Table 2) which represents a negligible polluting potential. The predicted equivalent concentration would be below 0.1 µg l^{-1} (see Table S2; supplemental material) which is consistent with the high removal efficiencies reported by Leroy et al. (2015) for simulated swale systems.

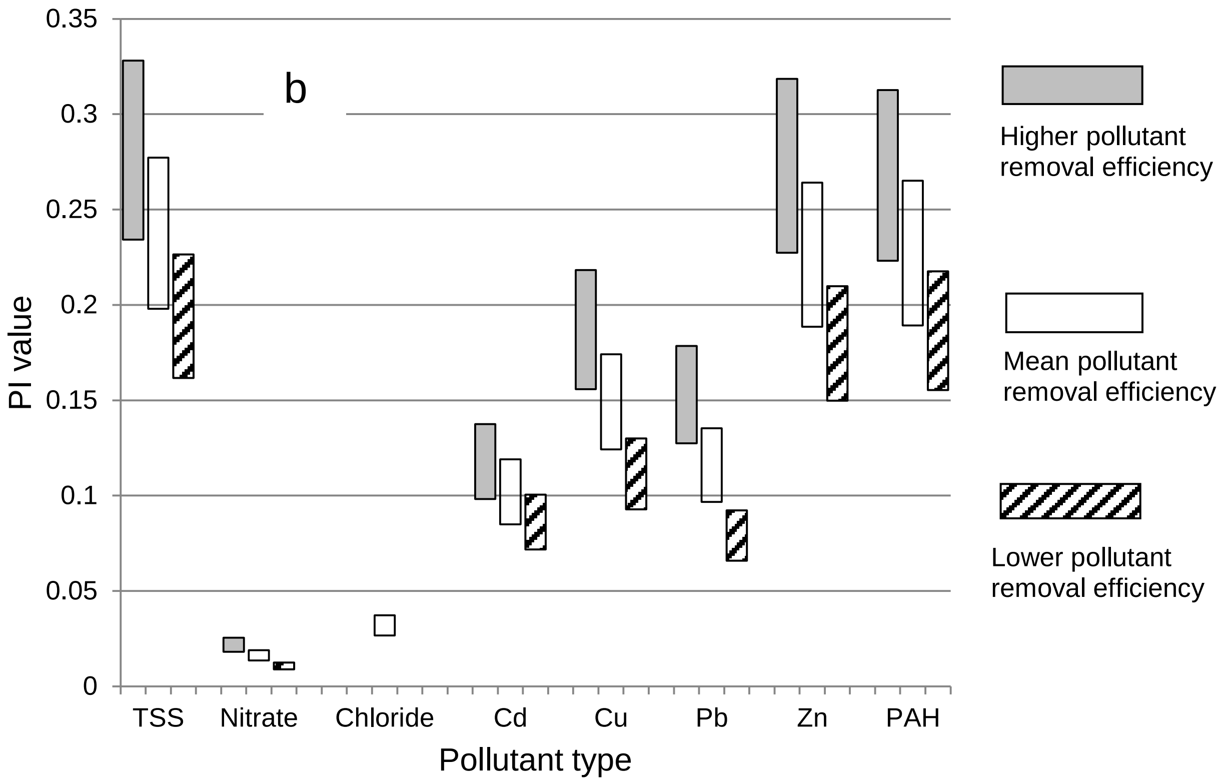
3.3 Bivariate probabilistic analysis for surface- and ground-waters

The analyses described in Sections 3.1 and 3.2 have identified that the two principal variables within swales which influence the quality of waters discharged to surface- and ground-waters are pollutant removal efficiency and volume reduction. To investigate the dependence of the results on the magnitude of these variables, the analysis has been extended to include their ranges expressed by means ± standard deviation (SD) and therefore incorporating 68% of the data spread. The model outcomes are assessed using all combinations of high (+1 SD), mean and low (-1 SD) volume reduction performances in combination with comparable pollutant removal efficiencies for each parameter. The predicted PI impact ranges to receiving surface- and ground- waters are shown in Figure 1 for all pollutants except chloride for which only a mean value for pollutant removal is available and therefore it only appears in Figure 1b as a single bar.

As can be seen from Figure 1a, the quality of water discharged from a swale above ground is dependent on the pollutant removal efficiency and is independent of the volume reduction. The higher pollutant removal efficiencies produce lower PI values and hence the expected improved effluent quality. Nitrate and Cd show the least variability in predicted PI values with the former always indicating negligible impact for surface waters. In contrast, Cd demonstrates a consistent potentially high impact over a pollutant removal efficiency range of 43.1% to 58.9% which is an area of concern for receiving surface waters. The other pollutants exhibit wider PI ranges spanning the minimal (PI; 0.1-0.2) and high (PI; 0.2-0.4) impact categories. To prevent swale surface effluents being in the high impact category it can be calculated that pollutant removal efficiencies of 75%, 67%, 60%, 75% and 75% would be required for TSS, Cu, Pb, Zn and PAH, respectively. Such high removal efficiencies



Key: Boxes identify ranges of PI values following application of higher and lower pollutant removal efficiencies



Key: Boxes identify ranges of PI values following application of lower and higher volume reductions

Figure 1. Ranges of PI values associated with water discharging from a swale to (a) surface waters and (b) infiltration

would suggest the need for swales to operate near their maximum design efficiency for their full lifetime duration and this emphasises the critical requirement for systematic and regular maintenance procedures. It also indicates that swales may need to be incorporated into treatment trains, particularly where final discharges are to sensitive receiving waters. Although none of the considered pollutants are predicted to produce a substantial impact (PI >0.4), this would be the case for Zn if the minimum lower pollutant removal efficiency were to reduce from 58% to 50%. Where the pollutant removal efficiency of a swale cannot be guaranteed it may be good practice to provide front-end protection in the form of a sedimentation pond to limit the entry of sediment associated pollutants to the swale channel.

In contrast to surface waters, the quality of water directed towards infiltration is dependent on both the swale pollutant removal potential and the volume distribution which takes place within the swale. The resulting trends in PI values (Figure 1b) show that pollutant removal efficiencies are the main controlling factor with higher retentions within the swale providing higher potentials for directing pollutants to infiltration. Superimposed on this, higher overall volume reductions for swales coincide with higher flows directed to infiltration and thus higher polluting potentials.

Examination of Figure 1b shows that it is possible to group the pollutants into three categories. Nitrate and chloride demonstrate predicted PI values for waters directed to infiltration which are lower than those for discharged surface waters (substantially so for chloride) indicating a negligible impact. Cd, Cu and Pb exhibit ranges of PI values which are associated with either negligible or minimal impacts and only Cu has the ability to reach the high impact category (PI >0.2) under a combination of high pollutant removal efficiency and high volume reductions within the swale. The greatest polluting impact for infiltrating waters is posed by TSS, Zn and PAH which under both high and mean pollutant removal conditions have the potential to exert high impacts (PI >0.2) regardless of flow reductions.

However, in ascertaining the impact on underlying groundwaters it is necessary to consider the remediation effect which occurs as infiltrating water passes through the sub-surface soil below the swale. Providing the soil retains its pollutant retention capabilities combined with well drained conditions, a low PMI value can be allocated for most pollutants (see Table 2) and the impact on the groundwater will be negligible. Over time the pollutant removal potential of the underlying soils can decrease and under this scenario the pollutants posing the greatest risk would be TSS, Zn and PAH. This is particularly the case under swale conditions characterised by high pollutant removal efficiencies combined with high volume reductions, when an increased PMI for infiltration to 0.6 or greater would result in a predicted high impact for the groundwater directly beneath a swale. Therefore it is important that swale design, especially sub-surface composition, and associated routine maintenance support effective infiltration processes.

4. Conclusion

The roles of swales in influencing the impacts of contaminated surface waters to both surface- and ground- waters have been assessed. As stand-alone treatment systems their ability to protect surface waters from less soluble pollutants is limited but their conveyancing capability allows them to be initial components of treatment trains involving additional pollutant removal facilities. The quality of surface waters discharged from swales is only influenced by pollutant removal efficiency with the outcome that all investigated pollutants (except nitrate) are capable of exerting a detrimental effect on the receiving water. Due to the involvement of infiltration processes, there are concerns that swales may pose a risk to underlying groundwaters. The water quality directed to infiltration is dependent on both pollutant removal efficiencies and volume reductions with high values of both posing the greatest problem, particularly for TSS, Zn and PAH. However, providing the soil permeability and pollutant removal capacities are maintained it has been shown that the level of risk posed to groundwaters from TSS, nitrate, chloride, Cd, Cu, Pb, Zn and PAH is negligible. Due to persistent adsorption of soluble pollutants the underlying surface soil layers may become saturated with pollutants over time limiting the removal efficiency of the infiltration process. Under these 'breakthrough' circumstances the predicted groundwater impacts of TSS, Zn and PAH could increase from negligible to high and Cd, Cu and Pb would also pose elevated concerns. It is unlikely that highly soluble pollutants, such as chloride and nitrate, would be affected although elevated inputs of nitrate could potentially impact on groundwater due to its higher retention in swales.

The continuous filtering of particles can lead to clogging of the surface soil pores and a shift in the flow balance towards the above ground route and consequently increased problems for receiving water quality. This situation can be exacerbated by the build-up of sediment in the base of the swale identifying the benefits of regular cleaning and/or careful swale design to prevent excessive sediment accumulation. The presence of a filter bed beneath dry swales provides a safeguard against this problem but in standard conveyance swales it is recommended that frequent checks are made to ensure that the permeability of the underlying surface soil is maintained. In making these predictions it is important to be aware of the numerous different site designs existing for swales and the highly variable operating conditions to which they can be subjected. Nevertheless, a scientific consideration of the unit operating processes responsible for pollutant removal in swales provides relevant insights into their potential impacts on groundwaters and could be usefully extended to other SuDS utilising infiltration as a pollutant removal mechanism.

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Supplemental Material

Assessing the Impact of Swales on Receiving Water Quality

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S1. Groundwater impact and pollution index (PI) methodology

Two approaches can be adopted for establishing a relationship between a PI value and the associated impact on groundwater quality according to the categories which have been used for surface waters i.e. negligible impact (PI <0.1); minimal impact (PI 0.1-0.2); high impact (PI 0.2-0.4); substantial impact (PI 0.4-0.7); extreme impact (PI >0.7). Based on the typical pollutant concentrations in the runoff from a motorway/major road (Crabtree et al, 2008; Kayhanian et al, 2007; Lundy et al, 2012) and the allocated PI value to this pollution source, the application of a linear relationship provides the concentrations corresponding to progressively decreasing PI values. However, these values need to be related to those which have been reported for pollutants in terms of their potential impacts on groundwaters. Defra and the Environment Agency (2011) have produced a comprehensive report identifying groundwater quality in the Thames river basin catchment based on minimum natural background limits, action levels (concentrations signifying the need to reverse an upward trend), threshold levels (an indication that the pollution threatens the groundwater status objective) and maximum detected concentrations. The process of matching predicted concentrations for surface runoff with groundwater 'trigger' concentrations to determine groundwater impact PI values and predicted concentration ranges is illustrated in Table S1 (supplemental material) for Cu for which a typical runoff concentration of 50 µg l^{-1} has been previously assigned a PI value of 0.6 (Ellis et al., 2012). The results for Cd, Cu, Pb, Zn, nitrate and fluoranthene (as representative PAH) are summarised in Table S2.

For chloride there is insufficient data available from the Thames catchment database (other than a minimum natural background concentration of 41 mg l^{-1}) and therefore relevant freshwater values have been used. US EPA (1988) reports acute and threshold chloride criteria for the protection of aquatic life in freshwaters of 860 mg l^{-1} and 230 mg l^{-1}, respectively which would have relevance to groundwaters where they discharge to surface waters. Similarly, where groundwaters are used for irrigation, a remedial goal of 350 mg l^{-1} exists and when used as a drinking water source a maximum chloride level of 250 mg l^{-1} is stipulated to avoid taste problems. Unfortunately, no relevant groundwater standards for TSS exist and therefore only the predicted ranges from highway runoff are reported in Table S2.

Table S1. Establishment of PI related pollution concentration categories for Cu in groundwater.

Groundwater impact category (PI value)	Predicted ranges based on highway runoff data (µg l^{-1})	Predicted 'trigger' concentration ranges based on from Thames Catchment groundwater data (µg l^{-1})	Predicted combined concentration ranges (µg l^{-1})
Negligible (<0.1)	<8.3	Below average minimum natural background level (<4.0)	<8.0
Minimal (0.1-0.2)	8.3-16.7	Average minimum natural background level– Average action level (4.0 – 10.27)	8.0 – 12.5
High (0.2-0.4)	16.7-33.3	Average action level - Average threshold level (10.27 – 13.70)	12.5 – 20.0
Substantial (0.4-0.7)	33.3-58.3	Average action level – Maximum detected concentration (13.70 –25.85)	20.0 – 40.0
Extreme (>0.7)	[a]	Above maximum detected concentration (> 25.85)	>40.0

[a]out of range

Table S2. Relationships between PI values and groundwater impacts

	PI values and associated impact categories				
	Negligible (< 0.1)	Minimal (0.1-0.2)	High (0.2-0.4)	Substantial (0.4-0.7)	Extreme (>0.7)
Cd (µg l^{-1})	<0.1	0.1-0.2	0.2-0.3	0.3-0.5	>0.5
Cu (µg l^{-1})	<8.0	8.0-12.5	12.5-20.0	20.0-40.0	>40.0
Pb (µg l^{-1})	<6.0	6.0-10.0	10.0-15.0	15.0-25.0	>25.0
Zn (µg l^{-1})	<30.0	30.0-75.0	75.0-135.0	135.0-250.0	>250.0
PAH (µg l^{-1})	<0.1	0.1-0.2	0.2-0.3	0.3-0.7	>0.7
Nitrate (mg l^{-1})	<7.5	7.5-15.0	15.0-30.0	30.0-50.0	>50.0
Chloride (mg l^{-1})	<75	75-250	250-400	400-875	>875
TSS (mg l^{-1})[a]	<19	19-38	38-75	75-131	>131

[a]Values predicted directly from highway runoff concentrations

Table S3. Swale percentage removal efficiencies for TSS, nitrate and PAHs.

	Percentage removal efficiencies	Average percentage removal efficiency (with standard deviation) calculated for this study
TSS	65-98; Schueler,1994 85-87; Barrett et al., 1998 68; Yu et al., 2001 79-98; Bäckström, 2003 48; Barrett, 2005 76; Deletic and Fletcher, 2006 44-83; Stagge et al, 2012 22[b]-31[c]; ISBMPD, 2014 66-98; Allen et al., 2015	74.3 ± 13.6
Nitrate	31±49; The Stormwater Manager's Resource Centre, 2015 38; US EPA, 1999 65; Caltrans, 2004 -25- 99; Fraley-McNeal et al., 2007 -25-89; Stagge et al., 2012 7[c]-8[b]; ISBMPD 2014	40.6 ± 14.0
PAHs (hydrocarbons)	67-93; Little et al., 1992 62; US EPA, 1999	71.0 ± 12.7

[a] based on pollutant loads; [b] median value; [c] based on event mean concentrations
Negative percentage removal efficiencies imply additions from within the swale.

Table S4. Swale percentage removal efficiencies for Cd, Cu, Pb and Zn

	Percentage removal efficiencies	Average percentage removal efficiency (with standard deviation) calculated for this study
Cadmium	41-72[a]; Stagge et al., 2012 42[b]; US EPA, 1999 48[b]-60[c]; ISBMPD, 2014	51.0 ± 7.9
Copper	14-67; Schueler, 1994 23–81; Rushton, 2001 89[b]; Caltrans, 2004 65[b]; Fraley-McNeal et al., 2007 62[b]; Barrett, 2008 42-81[a]; Stagge et al., 2012 14[b]-70[c]; ISBMPD, 2014	62.2 ± 15.8
Lead	60[b]; Wang et al.,1981 17–41[c]; Barrett et al. 1998 18–94[a]; Schueler, 1994; Rushton, 2001 67[b]; US EPA, 1999 85[b]; Caltrans, 2004 27-75[a]; Stagge et al., 2012 18[b]-78[c]; ISBMPD, 2014	58.0 ± 18.5
Zinc	75–91[c]; Barrett et al., 1998 71[b]; US EPA, 1999; Fraley-McNeal et al., 2007 18–93[a]; Schueler, 1994; Rushton, 2001; Bäckström, 2003; Stagge et al., 2012 89[b]; Caltrans, 2004 25[b]-59[c]; ISBMPD, 2014	70.7 ± 12.7

[a] based on pollutant loads; [b] median value; [c] based on event mean concentrations

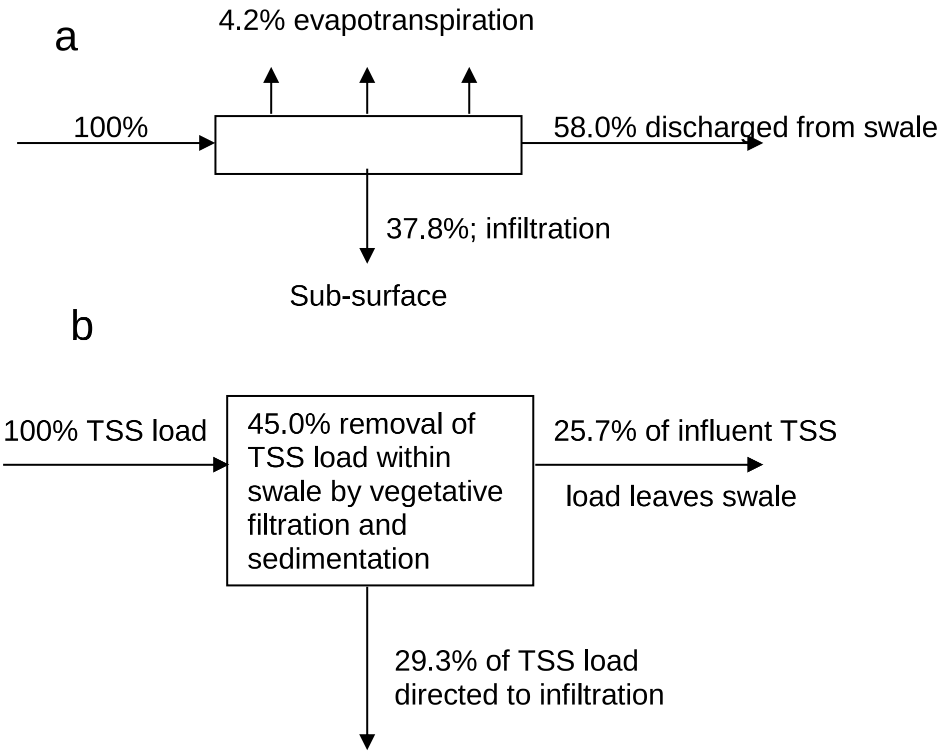

Figure S1. a) Diagrammatic representation of typical mean flow distributions within a swale. b) Diagrammatic representation of typical mean TSS loads during removal within a swale.

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