Sediment yield and BMP control strategies in urban catchments

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Abstract The sediment yield of urban catchments is quantified together with a variety of estimation methods for predicting discharge loads. Structural best management practices (BMPs) for urban storm water runoff control and treatment are identified and the criteria for their selection described. A brief overview is presented of BMP performance with the most effective sediment control devices being identified as wet detention basins and filtration systems.

INTRODUCTION

Sediments discharged from both surface water outfalls (SWOs) and combined sewer overflows (CSOs) are known to be key contributors to the pollution of urban receiving waters in chronic, accumulative terms as well as in terms of their acute storm event impacts. The specific association of toxic micro-pollutants, including bacteria, with the fine (<63 μm) particulate phase makes this behaviour especially detrimental to the quality and ecology of the receiving water (House *et al.*, 1993). In-pipe sedimentation can also lead to hydraulic surcharging, surface flooding and premature CSO overflow. The deposited sediment, especially if mixed with sanitary sewage or industrial effluent in a reduced environment, provides an ideal site for anaerobic biodegradation of any included organics. Subsequent scouring of the pipe invert during storm flow conditions can then yield high dissolved oxygen content (DOC) as well as BOD, COD and NH_4-N loadings to the receiving water (Ellis, 1991). Delayed in-stream impacts are normally associated with accumulated nutrients, metals, hydrocarbons and bacteria in sediment as well as being the result of benthal sediment oxygen demands (SOD). Typical undisturbed SOD levels vary between 0.15 and 2.75g m^{-2} day^{-1}, which can account for a permanent deficit of about 1.5-2.5 mg l^{-1} in the dissolved oxygen (DO) regime of the receiving water. Bed disturbance during storm flow conditions can elevate the SOD levels into the range 240-1500 g m^{-2} day^{-1} and depress normally near saturation in-stream DO levels down to 2 mg l^{-1} or less (House *et al.*, 1993).

A UK survey (CIRIA, 1986) has estimated that the annual cost of sediment related problems in urban drainage systems is of the order of £50-60M year^{-1}. By comparison, annual costs for urban sewer management in Norway are placed at £30M, whilst in Sweden sewer investment amounts to some £38M year^{-1} with operating and maintenance costs estimated at about £40M year^{-1} which rises to nearly £55M year^{-1} in West Germany. Sediments do, therefore, present a major management problem to the urban drainage engineer and influence the fate of many toxic and bioaccumulative substances in the aquatic ecosystem. They serve both as pollution sinks and potential contaminant sources to the overlying water column and require that best management practices (BMPs) are developed and implemented to achieve cost-effective removal performance.

SEDIMENT YIELD

Flow conditions in gravity sewers are extremely variable and self-cleansing is rarely achieved throughout the below-ground drainage system, so that in-pipe sedimentation is a common occurrence. Design procedures to ensure optimum hydraulic conveyance of sediment through the system require information on sediment type and concentration. A recent review of UK data (Ackers *et al.*, 1994), has recommended the values of relevant sediment characteristics given in Table 1. Alternative high and medium values are given depending upon how severe the sediment problems are judged to be. The figures represent the average noted for both SWO and CSO conditions and thus must be used with some caution when applied to estimating sediment outflows to receiving waters from separately sewered surface water systems.

Table 1 Typical sediment characteristics for in-pipe UK conditions.

Sediment transport mode	Parameter	Category	
		Medium	High
Suspended	Average concentration (mg l^{-1})	350	1000
	Median diameter (d_{50}, μm)	60	100
	Standard deviation (s)	2.0	2.5
Bed load	Average concentration (mg l^{-1})	50	200
	Median diameter (d_{50}, μm)	750	750
	Standard deviation (s)	2.6	2.6

after Ackers *et al.*, 1994.

Sediment discharges

Typical Event Mean Concentrations (EMCs) of suspended solids (SS) associated with UK urban surface drainage systems average 190 mg l^{-1} and range between 21 and 2582 mg l^{-1} with loads per unit area averaging 487 kg imp.ha^{-1} year^{-1} (with a range of 347-2340 kg imp.ha^{-1} year^{-1}). Table 2 provides data on the relative sediment concentrations and loadings associated with urban catchments for a number of EU member states and for the US. The data are reasonably consistent although the national average EMC values for both France and Scandinavia appear to be about double those recorded elsewhere in Europe. The data do, however, demonstrate the inherent variability of urban runoff quality which the design engineer and water manager needs to be aware of when estimating the quality efficiency of any control device and the likely receiving water effects. Table 3 presents typical loading ranges for various urban land use types with constructional sites, industrial/commercial and trafficked sources providing the highest sediment yields. Such published yield values can be used to derive a crude first-order screening procedure for the estimation of cumulative catchment loading:

Table 2 Suspended sediment values for urban runoff.

Country	Load per unit area (kg imp.ha^{-1} year^{-1})	EMC (mg l^{-1})	Median EMC (mg l^{-1})	90th percentile value (mg l^{-1})
UK	487 (347-2340)	190 (21-2582)		
France	*1460 (800-2650)	364 (15-3780)		
Germany	*1035 (263-1499)	170 (46-2700)		
Scandinavia		323 (5-1040)		
USA		150 (2-2890)	100 †(1.0-2.0)	300

Note: figures in parentheses give range of observed mean values; † figures are coefficients of variation (*CV*); * figures in kg ha^{-1} year^{-1}. Source: Ellis, 1989; Athayde *et al.*, 1983; EWPCA, 1987.

Table 3 Sediment loading rates for various urban land uses (figures in kg ha^{-1} year^{-1} except where stated).

	Highways	Industrial commercial	Residential low density	Residential medium density	Residential high density	Car parking areas	Grassed parkland	Construction sites
Median	502	865	200	322	434	440	346	67 415
Range	821-723	242-1369	60-340	97-547	133-755	124-762	80-588	22 000-84 000
EMC (mg l^{-1})	250	280	100	187	250	-	-	-

$$L = \Sigma a_i Al_i \tag{1}$$

where: L = total loading, a_i = area in land use type i and Al_i = area loading from land use i.

The probability distributions of flow-weighted SS EMCs are known to follow log-normal patterns (Brizio *et al.*, 1989) such that linear fits reflect the frequency of occurrence of concentrations at the lower end of the range of values with only very small percentages occurring at high concentrations. Control and treatment devices intended to capture sediment flows must take into account this distributional regime and have operational designs primarily intended to remove sediments associated with flow volumes equal to or less than the annual event.

First flush

The maximum pollutant EMCs are consistently observed to occur in response to the initial 12-15 mm of effective rainfall-runoff with significantly lower runoff concentrations occurring thereafter (Ellis, 1991). Therefore, the pollutant rate which discharges to a receiving water system can be a better indicator of the acute impact of individual storms. A "first flush" of sediment and sediment associated pollutants typically occurs during the initial periods of storm flow with some 65-75% of the total SS load being discharged with the first 25-30% of the runoff volume (Verbanck *et al.*, 1994). If pollutographs from different storm events are superimposed so that the recession limbs coincide, it is possible to classify the adjusted pollutographs into categories based on the peakedness of the storm profile ($k = Pint_{max}/Pint_{avg}$). The timing and value of peak SS concentrations can then be described as a function of the peakedness (k), antecedent dry weather period ($ADWP$) and time of concentration (t_p) of the drainage system. Thus the shape of the pollutograph can be related to a non-dimensional form of the hydrological parameters identified as being important for urban storm sewer and detention storage design.

Estimating sediment yield

A variety of methods are available to estimate sediment loadings for urban catchments which include simple empirical procedures such as indicated above or other similar methods which use EMC values. One such empirical approach appropriate for planning level decisions within urban catchments of less than 2500 ha uses a simple two step procedure:

Step 1 Estimate the annual storm water volume, Q_{year} as:

$$Q_{year} = a.A_{imp}(P_{AN} - b) \times 10^{-3} \qquad (\mathrm{m}^3) \tag{2}$$

where: a is a unit conversion factor representing urban land use type; A_{imp} is total impermeable surface area (m^2); P_{AN} is total annual precipitation (mm) and b is total depression storage loss (mm). Advice on default values for a and b is given in Hall *et al.*, 1993.

Step 2 The derived Q_{year} volume is multiplied with an appropriate EMC value such as those in Table 2. More detailed calculations can be made using EMC values adjusted for differing land use types within the catchment as given in Table 3. Although the method appears crude and must be used with caution, the loading ranges estimated from this empirical approach almost always bound estimates made independently by more complex mechanistic modelling and either approach will normally produce the same management conclusions.

A number of workers have utilized multiple regression equations for the estimation of runoff volumes and pollutant concentrations and loadings expressed as a function of various independent variables, with standard errors providing uncertainty bounds to the predicted values (Hemain, 1986; Driver & Troutman, 1989). These empirical equations tend to be site-specific, with the principal controlling variables being cumulative runoff volume (Qt), time (t) elapsed since commencement of the storm, total rainfall depth (Pd_{tot}), 5 min rainfall intensity ($Pint_5$) and $ADWP$. When applying such regression techniques it must be remembered that an increase in the standard error of the estimate by inclusion of another variable indicates that the additional information given by the extra variable is offset by the loss in degrees of freedom i.e. the regression is better without the extra variable.

Site-specific estimates can also be derived empirically for planning level decisions using the assumed EMC lognormal distribution (Ellis, 1986; Marsalek, 1991). The natural log of the EMC value is first taken and the mean (x) and variance (s^2) of the natural logs computed. Then the mean EMC value (EMC_{mn}) is:

$$EMC_{mn} = e^{(x + s^2/2)} \tag{3}$$

The confidence interval (CI) can be calculated for the mean EMC estimate as:

$$CI = EMC_{mn} . e^{\pm\theta . [s^2/n + 2.(s^2)^3/(n-1)]^{0.5}} \tag{4}$$

where: $\pm$ is used for the upper and lower confidence limits; $\theta = 1.96$ for 95% CI and 1.69 for 90% CI; and n = number of EMC values used to find x. A site (or local) flow record can be consulted to obtain the total flow volume (Q_{tot}) for the loading estimate period. This volume is then multiplied by the mean EMC to get the loading, and then multiplied by the upper and lower confidence limits to get the estimate bounds.

A number of comprehensive mechanistic models are also available for estimating sediment concentrations and loadings discharged from urban drainage systems. All however, require substantial local data to set variable parameters in the calibration step and to verify them for the intended application. In the UK, the National Rivers Authority have identified a strategic modelling framework for implementing the intermittent discharge requirements of the EU Urban Wastewater Treatment Directive. The framework advocates the use of a suite of deterministic sewer quality models (MOSQITO, QM) which can model various SS fractions as well as BOD/COD and total ammonia. An alternative stochastic modelling approach (MOUSETRAP) based on EMC lognormal assumptions is also widely used throughout Europe and Australasia whilst the US-based models ILLUDAS, STORM and SWMM have been in wide global use over the last decade. These models structure the water quality components on a mass balance framework with sediment additions (deposition) computed as a linear function of time and with losses represented by a first-order wash off function.

TREATMENT PRACTICES

BMP options

Urban runoff management involves controlling both the quantity and quality of runoff and best management practices (BMPs) include a range of both structural and non-structural measures to achieve control and treatment. Structural practices rely on three basic mechanisms: detention storage, infiltration and filtration.

(a) Detention storage: Detention BMPs such as wet and dry ponds, extended detention dry ponds, inlet devices, tanks etc., temporarily impound storm water to control runoff rates and to settle and retain SS and associated pollutants. Constructed wetlands and multi-purpose basin systems will also remove a range of micro-pollutants through enhanced gravitational settlement and biofiltration of SS and secondary biological treatment as a result of adsorption and microbial decomposition. Inlet chambers (or catch basins) can also be effective in capturing coarse solids and oils washed off impermeable surfaces. A number of inlet devices are now coupled with oil/grit separation facilities, but all are of restricted volume and require frequent cleaning out.

(b) Infiltration: Infiltration BMPs, such as trenches, basins, french drains, sand filters, perforated pipe systems, porous and block pavements, rely on absorption of runoff to control and treat urban storm water discharges. The storm water is percolated through the porous media of the device and surrounding soil where filtration and biological action removes the pollutants.

(c) Filtration: A range of vegetative BMPs, such as grassed swales and filter strips (as well as buffer zones and urban forestry) can be used to filter and settle sediment and associated pollutants. The objective of a vegetated treatment device must be to achieve a grass or plant stand that serves as an effective filter. Therefore, the ideal characteristics will be the development of a dense, uniform growth of thin-stemmed indigenous plants. As the vegetative sward will essentially serve as a filter, pollutant uptake properties are of secondary significance.

BMP selection criteria

Success in applying any storm water practice will initially depend on selecting devices appropriate to the treatment objectives and site conditions. Evaluation criteria might include whether quantity and/or quality control is required; which pollutants are to be treated, as well as any secondary objectives related to environmental or land enhancement; ability to meet regulatory requirements; site conditions and costs. BMP selection should also address the primary management objectives in terms of whether the control device is intended for new or existing developments; for highway runoff control; for on-site disposal systems; as stand-alone construction site controls; or as part of a wider catchment protection strategy. A further group of selection criteria relate to post-construction operation and maintenance (O & M) requirements. Proper O & M of structural facilities is critical to their long term effectiveness in mitigating adverse impacts of urban runoff. The proper installation and maintenance of various BMPs will often determine their success or failure and both aspects require rigorous and aggressive field inspection checks and enforcement procedures. A final element in the selection criteria is that relating to public acceptance and institutional attitudes which require wider public education on urban runoff programmes and their purposes as well as local administrative structures which satisfactorily interface with regulatory agencies.

These criteria can be used as a basis for BMP screening with a ranking (e.g. 1 to 5) assigned to each practice for every parameter identified for each criteria; these can be differentially weighted if necessary. Florida and California have each produced BMP Handbooks which attempt to refine the process of practice selection using such scoring techniques and a number of other US states are currently drafting similar municipal guidance. Figure 1 provides a planning level screening approach to the applicability of

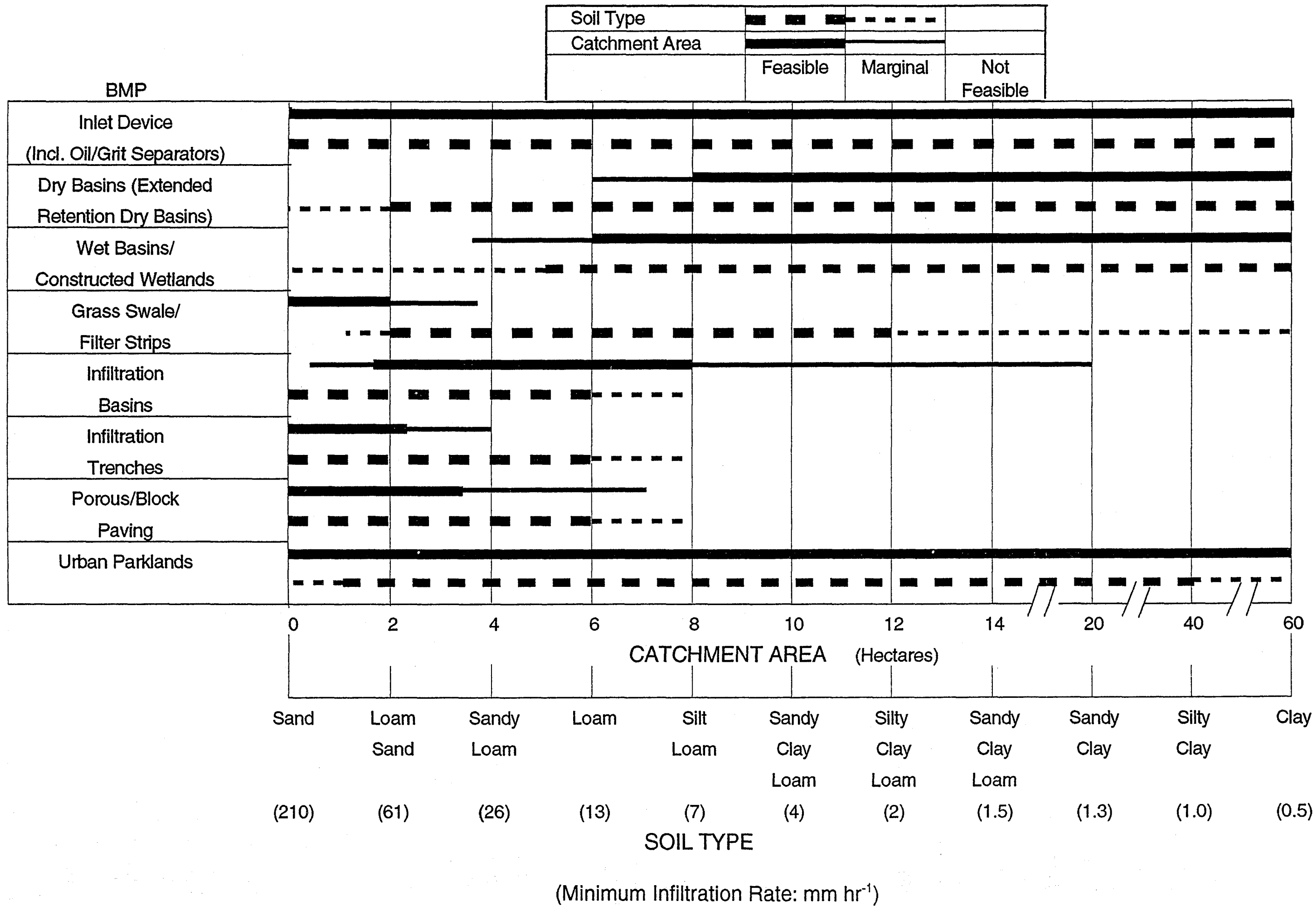

Fig. 1 Applicability of treatment practices.

the various BMP options in terms of catchment area and soil type/infiltration rates. It should be noted that extremely high sediment inputs (e.g. from construction sites) can be a major constraint on the use of most BMPs, apart from dry/wet ponds and constructed wetlands which need very careful site design if they are to function effectively under such input criteria. Likewise, high water tables or steep gradients can preclude the use of infiltration and porous paving systems as well as restricting the effectiveness of grass swales and filter strips.

BMP performance

The effectiveness of BMPs to provide flow and sediment control is indicated in Table 4 together with information on costs, annual O & M burdens and expected operational lifetimes. The data essentially refer to new and existing urban developments but can also be extended to include highway locations and are derived from North American and European data bases. Table 4 provides the median and ranges reported for SS removals and it appears that most BMPs are capable of achieving greater than 75-80% efficiency levels, when properly designed, operated and maintained. The most consistent high-level

Table 4 BMP effectiveness and cost.

BMP	SS removal efficiency (%)	Peak discharge control			Costs (US $)		
		2 year storm	10 year storm	100 year storm	Construction cost	Annual O & M	Total annual cost
Extended detention dry basin	35 (5-90)	✓	✓	✓	0.002-$014/m^3$	3%-5% of capital cost	0.002-$0.009/m^3$
Constructed wetland	68 (-20-100)	✓	✓	✓	N/A	N/A	N/A
Wet pond	75 (-30-91)	✓	✓	✓	0.001-$0.03/m^3$	0.1%-5% of capital cost	0.0002-$0.002/m^3$
Infiltration basin	75 (45-100)	?	x	x	0.006-$0.04/m^3$	3%-13% of capital cost	0.001-$0.012/m^3$
Infiltration trench	75 (45-100)	✓	?	x	0.03-$0.26/m^3$	5%-15% of capital cost	0.009-$0.03/m^3$
Vegetative filter strip	70 (25-80)	?	x	x	0.01-20 000/ha	20-365/ha	40.5-567.0/ha
Grass swale	65 (0-100)	?	x	x	1.4-2.6/ linear metre	0.15-150/ linear metre	0.3-0.6/ linear metre
Filtration basin	80 (60-95)	✓	?	x	0.03-$0.3/m^3$	7% of construction cost	0.003-$0.02/m^3$
Inlet device (including oil/grit interceptor)	15 (0-25)	x	x	x	445-8100/ drained hectare	2.0-40.5/ drained hectare	42-405/ drained hectare
Porous/block paving	85 (65-95)	✓	?	x	0.09-$019/m^2$	0.001-$0.004/m^2$	$0.014/m^2$

performance, in terms of the inter-quartile range, is given by wet ponds and filtration basins (sand/gravel filters). Infiltration systems including porous/block paving can also give good performance levels and such source controls are frequently viewed as being optimal systems for achieving sustainable development in urban catchments, since they focus on prevention (rather than cure) of water quantity and quality problems downstream. Unfortunately, these practices have the highest failure rates among all urban storm water BMPs, primarily due to early sediment clogging and microbial growths infilling the void spaces. Schueler *et al.* (1992) have reported 5 year failure rates for trenches and porous pavements of 50% and 75% respectively.

Both infiltration and (bio)filtration systems can serve as very effective first-level BMPs in a treatment train with the filtered flows passing on to second and tertiary level devices. Alternatively, one BMP mode can incorporate other modes, e.g. wet detention ponds can be designed to include sediment forebays with constructed wetlands introduced to provide a secondary polish (Hall *et al.*, 1993). However, the effectiveness of a treatment train system will not be additive, since the first device in the series will trap the pollutant fractions easiest to remove, making subsequent reductions more difficult. It is possible to provide a rough estimate of the combined efficiency of a treatment train by applying the following equation:

$$E_s = 1 - X(1 - E_1)(1 - E_2) \qquad (5)$$

where: E_s = combined series efficiency ie pollutant fraction remaining; X = "penalty" representing the performance reduction of the second device ($X > 1$); and $E_{1/2}$ = efficiency of first and second control devices if used alone. The problem is to accurately establish the penalty (X) value as insufficient field data are available, but assuming a 50% stand-alone capture, combining two devices would yield an uplift to 69% for a given X value of 1.25.

Regionalization of BMP systems by installing and maintaining BMPs for more than one development site, might also prove to be more efficient and cost-effective due to the economies of scale of operating one large system against several smaller systems. Within existing urban developments where land availability is scarce and costly, retrofitting of structural controls may be the only feasible alternative for improving water quality. A range of retrofit options are possible, including dry pond conversions to provide extended detention, fringe wetland creation in wet ponds and at SWO outfalls to receiving waters, additional storage capacity within the open channel or flood plain, first flush diversion to sand-peat filters etc. In addition, the use of non-structural approaches such as offset buffer zones at the riparian edges of receiving waters will also help alleviate the impact of diffuse pollution loads from urban surfaces.

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