
**THE FATE OF STORMWATER PRIORITY
POLLUTANTS IN BMPs**

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EXECUTIVE SUMMARY

This report assesses the fate, behaviour and removal potential of a range of stormwater priority pollutants during their passage through structural Best Management Practices (BMPs). It has been prepared as part of DayWater, an EU 5th Framework funded urban stormwater management project, using contributions from several partners covering both practical and theoretical areas of expertise. The stormwater priority pollutants were identified in D4.3 (Selected Stormwater Priority Pollutants) and the evaluated BMPs are those identified in DayWater Deliverable 5.1 (Review of the Use of Stormwater BMPs in Europe). The report gives comprehensive descriptions of the primary removal processes, which take place within BMPs, and discusses how these processes affect the stormwater priority pollutants based on their characteristic properties. This information is then combined to enable an evaluation of the removal of stormwater priority pollutants by BMPs.

The primary removal routes within BMPs are identified as being due to physical (e.g. sedimentation), biological (e.g. microbial degradation) and chemical (e.g. precipitation) processes. The mechanisms of removal are described in detail and relevant measurements are identified to enable the removal of stormwater priority pollutants by each process to be considered and compared in a clear and quantifiable manner. For example, volatilisation is considered using the K_h value, a constant calculated using Henry's Law and quantified with the units $\text{atm}\cdot\text{m}^3/\text{mole}$.

The selection process used to identify the stormwater priority pollutants is outlined, along with a justification of their inclusion in the list. The selected 21 stormwater priority pollutants are sub-divided into 5 sub-groups representing general water quality parameters, metals, polycyclic aromatic hydrocarbons, herbicides and miscellaneous compounds. The latter category incorporates nonylphenoethoxylate (NPEO), pentachlorophenol, polychlorinated biphenyl 28, di-(2-ethylhexyl) phthalate (DEHP) and methyl tert-butyl ether (MTBE).

Pollutant removal performance data for each type of BMP is presented and discussed, with important operation and maintenance issues being highlighted. However, field data for the removal of several of the stormwater priority pollutants (e.g. the herbicides pendimethalin and phenmedipham) by BMPs is limited and this is identified as an important area for further work.

This report provides a unique overview of the use of BMPs based on their individual removal processes and the subsequent potential of these processes to remove a variety of pollutants. It forms the basis for the development of a model to enable the use of BMPs to be considered in terms of the removal of identified chemicals of concern. The use of such an approach will enable end-users concerned about a particular pollutant to rank BMPs based on a combination of the ability of the identified processes to remove the targeted pollutant and the relative importance of these processes within a particular BMP.

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1 Primary Removal Mechanisms in BMPs and their Quantification

The removal of pollutants by BMPs can occur due to a complex combination of biological, chemical and physical processes. The primary removal mechanisms which participate in pollutant removal within BMPs are categorised in Table 1-1, together with the appropriate measures by which they can be quantified.

Table 1-1. Primary Removal Processes in BMPs

Removal Processes	Relevant Measurements and Units
Sedimentation	Settling velocity (m/s)
Adsorption	K_d (L/g); associated chemical fraction
Microbial degradation	Biodegradation rate ($t_{1/2}$ life in days)
Precipitation	Solubility (mg/l)
Filtration	Function of K_d (L/g) and precipitation (mg/l)
Volatilisation	K_h (atm-m ³ /mole)
Photolysis	Photodegradation rate ($t_{1/2}$ life in days)
Plant uptake	Bioaccumulation (K_{ow} (ratio))

Key: K_d = adsorption coefficient = partitioning of a substance between the solid and dissolved phases at equilibrium = ratio of the concentration of a pollutant in the solid phase to its concentration in the dissolved phase at equilibrium
 K_h = Henry's Law constant (based on the relationship that at a constant temperature the mass of gas dissolved in a liquid at equilibrium is proportional to the partial pressure of the gas)
 K_{ow} = octanol-water partition coefficient = a measure of the potential for organic compounds to accumulate in lipids = ratio of the concentration of a pollutant in octanol to that in water at equilibrium

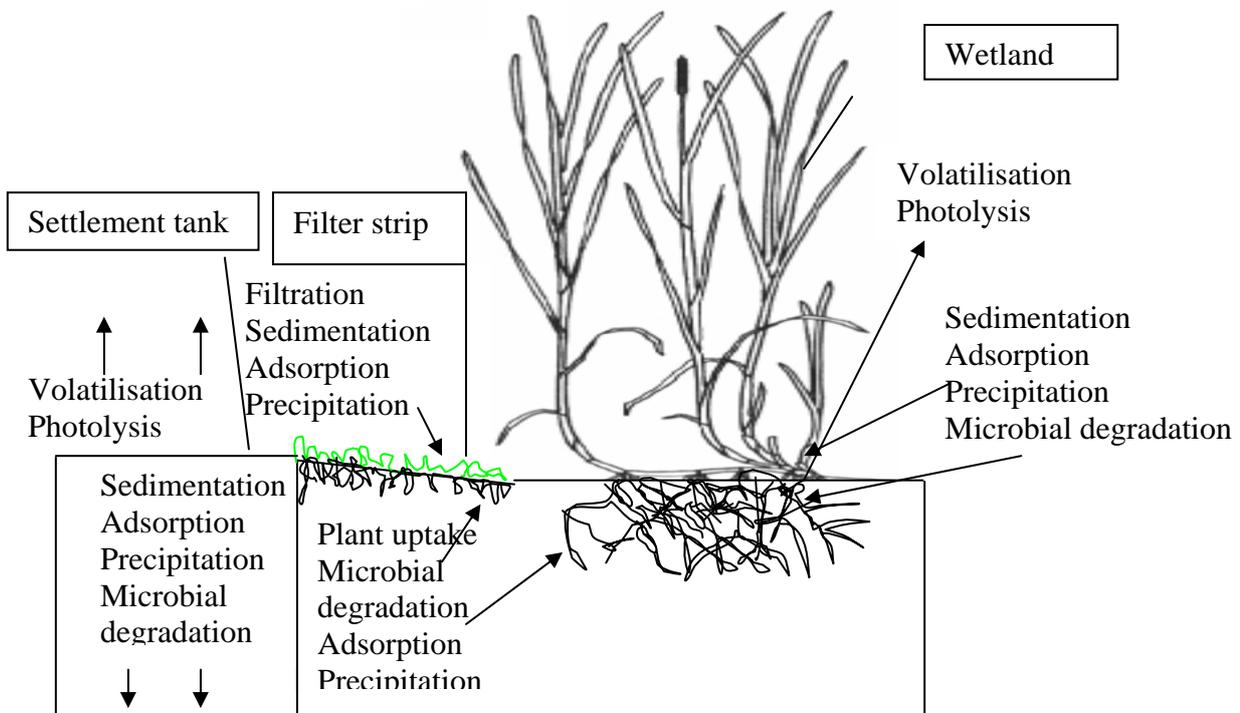


Figure 1.1. Diagrammatic representation of primary stormwater pollutant removal processes in BMPs.

The different removal mechanisms and their quantification, as defined in Table 1.1 and illustrated in Figure 1.1 (for a treatment train consisting of a settlement tank, filter strip and constructed wetland), are described in detail in the following sections.

1.1 Sedimentation

Sedimentation can be defined as the sinking or settling of discrete or agglomerated suspended sediment particles to the base of a water column (Ellis *et al.*, 2004). The process of discrete particle settling is governed by Stokes' Law which states that the frictional force on an object moving through a medium is dependent on the radius and velocity of the particle and the viscosity of the medium through which it is moving.

The principle forms of sedimentation can be subdivided into Type I and Type II processes. Type I sedimentation refers to the settlement of discrete particles, such as coarse, inorganic sands, which may be mobilised during first flush conditions, with Type II referring to the settlement of flocculated or aggregated particles which less readily remobilise. Sedimentation can be further enhanced in the presence of vegetation due to increased frictional resistance slowing the hydraulic flows. It is reported to be most effective on particles $>10\mu\text{m}$ with a specific gravity >1.0 , and begins to occur when the flow rate is $<0.2\text{m/s}$ (Merritt, 1994).

1.2 Adsorption

Adsorption involves the binding of dissolved or particulate substances onto the surface of suspended solids, micro-organisms, vegetation, settled bottom-sediment or other type of substrate material. It can be either a physical or chemical process or, as is more generally the case in the field, a combination of both processes. Physical adsorption involves the binding of ions due to electrostatic forces or Van der Waals forces (Harrison and de Mora, 1996). A physically adsorbed ion may be displaced by a similarly charged ion which has a greater electrostatic attraction for the binding site (e.g. by a process known as cation or anion exchange), and therefore physical adsorption is rarely a permanent process. In chemical adsorption, adsorbed substances are held in place by chemical bonds resulting in the formation of a more stable complex than that formed by physical adsorption. The propensity of a pollutant to be removed through this process can be quantified using the K_d ratio, or through an experimental determination of the chemical fraction of the solid phase with which the pollutant is associated e.g. exchangeable fraction, carbonate fraction, hydrous metal oxide fraction, organic fraction etc.

The adsorption of pollutants to suspended solids and their subsequent removal from the water column by sedimentation (see Section 1.1) is one of the most important removal mechanisms in many BMPs.

1.3 Microbial degradation

Micro-organisms are involved in many important pollutant removal processes such as the breakdown of organic matter, nitrification, denitrification and the precipitation and binding of metals (Scholes, 2000). Both aerobic and anaerobic micro-organisms, including bacteria, fungi and actinomycetes, may be involved in these pollutant removal processes.

Metal-oxidising bacteria, found in aerobic environments, are known to oxidise ferrous iron to ferric iron, which then precipitates as iron (III) hydroxide (Webb *et al.*, 1998). The

precipitated iron hydroxide can then act as an adsorption site (see Section 1.2) for the removal of further pollutants from the water column, hence metal-oxidising bacteria can play both a direct and indirect role in pollutant removal. Similar mechanisms are also understood to be involved in the oxidation and subsequent precipitation of a range of other metals such as Pb, Cu, Zn and Cd. In anaerobic zones, the reduction of sulphate by bacteria such as *Desulfotomaculum* results in the production of H₂S. H₂S ionises to give S²⁻ which readily reacts with dissolved metals precipitating them as insoluble sulphides.

Many organic substances are microbially degraded under both aerobic and anaerobic conditions. For example, ethylene and propylene glycol (major components of aircraft de-icers) are reported to be degraded under both aerobic (Verschueren, 1996) and anaerobic conditions (Veltman *et al.*, 1998). The complete mineralisation of a complex organic compound is understood to involve a consortium of micro-organisms (Jones, 1995), with different groups or species involved in different stages of the biodegradation process.

Nitrification and denitrification are both microbially-mediated processes, which occur under aerobic and anaerobic conditions, respectively, and involve several different groups of bacteria. Nitrification is an important mechanism in removing ammonia, with denitrification resulting in the release of nitrogen to the atmosphere, mainly as free nitrogen gas (Cooper *et al.*, 1996).

Although anaerobic degradation processes tend to proceed at a slower rate than aerobic processes, anaerobic processes, such as the precipitation of metal sulphides, may represent a more long term sink as they tend to occur within sediments or other confined conditions reducing the likelihood of the re-release of pollutants (Cooper *et al.*, 1996). The presence of both aerobic and anaerobic conditions (and the resulting dramatic changes in redox potential) found in many BMPs increases the potential for the establishment of a diverse microbial consortium, effectively enhancing the potential of biodegradation as a pollutant removal process. Microbial degradation can be quantified using biodegradation rates (often expressed as a half-life) or more generally considered through noting the presence or absence of aerobic and anaerobic zones within a BMP.

1.4 Precipitation

The formation of insoluble precipitates followed by sedimentation can be an important pollutant removal mechanism, both directly via the precipitation process itself and indirectly through the adsorption of soluble components by the precipitate as it settles out of the water column. Precipitation varies with factors such as pH, water hardness and the presence of competing ions. It can result in the formation of both insoluble metal and organic complexes and can be quantified using the solubility of the pollutant being considered.

1.5 Filtration

Filtration can refer to both the physical sieving of stormwater as it passes through a porous media, such as porous asphalt, and the bio-filtration of stormwater as it passes through vegetation. The former process directly traps particles and associated pollutants, with the latter increasing the potential for sedimentation by reducing the rate of flow. The efficiency of the 'sieving process' is obviously dependent on the pore size of the material or substrate selected: the smaller the pore size, the higher the degree of particle retention and the greater the susceptibility to blocking or clogging. Smaller pore sizes also decrease the filtration rate,

or hydraulic conductivity, of the substrate. The use of a substrate with too small a pore size can result in the backing-up of stormwater and surface flooding. The efficiency of bio-filtration depends on vegetation density, with flow velocity reduction increasing with an increase in plant density. However, at flow velocities greater than 2 m/s, the vegetation stand will be flattened and thus filtration potential is substantially reduced.

The potential of a pollutant to be removed from the water column via filter 'sieving' can be quantified as a function of the parameters for precipitation and adsorption (see Sections 1.1 and 1.3). Bio-filtration may be quantified through a consideration of the presence or absence of vegetation and the flow rate.

1.6 Volatilisation

All solids and liquids produce vapours of atoms or molecules via a process known as volatilisation. The amount of a substance that can be volatilised is dependent on its vapour pressure with higher vapour pressures indicating a greater potential to be volatilised. This is also influenced by local temperature and wind conditions.

The volatility of a substance can be quantified by using Henry's Law to calculate the K_h value, a partition coefficient, which is the ratio of a substance's concentration in air and in solution at equilibrium. This provides an indication of a substance's potential to volatilise (ATSDR, 1995). Volatilisation tends to be a more important process in the removal of organic rather than inorganic substances, with the lighter and more soluble organic compounds being the first to be removed via volatilisation (Jones, 1995) under the prevailing wind and temperature conditions.

1.7 Photolysis

Photolysis involves a chemical reaction initiated by the exposure of a substance to light and is therefore a more important process in the degradation of atmospheric pollutants than those found in the water column. However, photolysis may still be a significant process in the removal of pollutants from the surface layers of a water body. Photolysis can be quantified through the determination of whether a substance is photodegradable and, where data is available, it is represented by the photo-degradation half-life.

1.8 Plant uptake and bioaccumulation

The amount of a chemical which is potentially available for biological interaction is known as the bioavailable fraction, with the uptake of this bioavailable fraction referred to as bioaccumulation (Ellis *et al.*, 2004). Plants are capable of bioaccumulating a range of compounds such as those containing nitrogen, phosphorous (Brix, 1996) and heavy metals (Mungur *et al.*, 1997). Terrestrial plants take up pollutants from the soil whereas aquatic vegetation, including algae and phytoplankton, can take up pollutants from both the sediment and the water column. Contact time between vegetation and pollutant is obviously an important factor, with the amount of pollutant bioaccumulated increasing with increasing contact time. Roots and rhizomes are the main storage areas in rooted plants, with lesser amounts being stored in the leaves (Kadlec and Knight, 1996), in an approximate ratio of 25:10:1 for roots, rhizomes and leaves, respectively. For plant uptake to occur, plants must be actively growing, and therefore both climate and season can determine the importance of bioaccumulation as a pollutant removal process. Plant uptake may be quantified using K_{ow} , a

partition coefficient representing the distribution of a substance between octanol and water at a specified temperature. This value is used to indicate the potential of a substance to accumulate in lipids and organic fluids relative to water, and thus indicates its potential to be bioaccumulated.

2 Stormwater Priority Pollutants and their Primary Removal Mechanisms in BMPs

2.1 Selection of stormwater priority pollutants

A list of stormwater priority pollutants has been identified within WP4 of the DayWater project using a decision tree analysis based on a range of physico-chemical properties, such as volatilisation, persistence and sorption, for both the sediment and aquatic phases (see DayWater Deliverable 4.3). The decision tree based tool was initially applied to a list of over 600 substances identified from the literature as being potentially present in stormwater. A final list of selected stormwater priority pollutants was generated based on their environmental concern. From this ‘concern list’, 19 substances were short-listed for further work, containing representative pollutants which are either illustrative of the behaviours of a group of pollutants (eg pyrene for 4 ring PAH compounds) or, because of similarities in properties, are able to act as indicators for other substances of concern. The more general water quality parameters (suspended solids, BOD/COD, bacteria, nitrate and phosphate) were added to the list to make a total of 24 stormwater pollutants to be considered with respect to their behaviour within BMPs. It should be noted that bacteria were not included in DayWater Deliverable 4.3 but are included here to represent concerns over microbiological contamination of stormwater. These are given in Table 2.1 together with a brief justification for their inclusion in the list.

2.2 Removal of pollutants representative of general water quality parameters

For the purposes of this study, these general pollutants are defined as biological oxygen demand (BOD), chemical oxygen demand (COD), suspended solids (SS), nitrate, phosphate and bacteria (i.e. faecal coliforms/pathogens). Of these, BOD is an indirect measure of the level of degradable organic pollution (i.e. both soluble and particulate material). Both soluble and insoluble organics may be removed from the water column through processes of adsorption to particulate matter followed by sedimentation. However, these processes are thought to be more important in the removal of particulate BOD. A similar relationship is anticipated for filtration, with the removal of particulate BOD being more significant than for soluble BOD. The removal of BOD via microbial degradation is also a major process, which occurs under aerobic conditions in both the water column and on the surface layers of substrate/vegetation. Photolysis may be responsible for the removal of both soluble and particulate BOD if it is located close to the surface layer. However, it is considered to be a much less significant removal process than either adsorption and sedimentation or microbial degradation. Processes of precipitation, volatilisation and plant uptake do not significantly reduce the levels of BOD.

COD is an indirect measure of the amount of oxygen required to oxidise all the organic and inorganic compounds in an aqueous sample. COD may also be particulate or soluble in nature and is thought to be affected by removal processes in a similar manner to that described for BOD above. In addition to the above processes COD may also be removed under anaerobic conditions, for example, the breakdown of sulphates by sulphate-reducing bacteria (see Section 2.3).

Sedimentation and filtration are obviously the prime removal processes for suspended solids. The agglomeration of particles into larger particle “clumps” through flocculation processes will enhance the efficiency of both these processes, as larger particles settle out from the

water column at a faster rate than fine particulate matter and are also more easily retained through filtration. Bacteria can be considered as suspended particulate matter, and are therefore subject to the same primary removal processes as sedimentary particulate matter.

The nutrients (phosphate and nitrate) tend to be affected by removal processes in very different ways. Phosphates readily adsorb to suspended particulate matter, and in particular to particles of an inorganic nature, and are therefore readily removed from the water column via processes of adsorption, sedimentation and filtration. Phosphates readily precipitate out of solution as insoluble Fe, Al and Ca phosphates and as insoluble organic compounds in the pH and concentration ranges found in natural waters and BMPs (Hemond and Benoit, 1988). Phosphates are not microbially degraded (other than through the uptake of essential nutrients) and are not removed from the water column by volatilisation or photolysis. However, phosphates are readily bioaccumulated by plants where they are used for growth as an essential nutrient.

Table 2-1 Stormwater priority pollutants and justification for their selection (Eriksson *et al*, 2004).

Type	Constituent	Justification
Common water quality parameters	BOD/COD	General water quality parameter
	Suspended solids	General water quality parameter
	Nitrate	General water quality parameter
	Phosphate	General water quality parameter
	Faecal coliforms/pathogens	General water quality parameter
Metals	Cadmium	P, CMR
	Chromium	P, occurs widely as anionic form in natural waters
	Copper	P, T
	Nickel	P, CMR
	Lead	P, CMR
	Platinum	P, CMR
	Zinc	P
PAH	Naphthalene	Representative indicator of PAH (2 ring compound), PB, CMR
	Pyrene	Representative indicator of PAH (4 ring compound), PB
	Benzo [a] pyrene	Representative indicator of PAH (5 ring compound) PB, CMR
Herbicides	Glyphosate	BT
	Pendimethalin	PB
	Phenmedipham	PB
	Terbutylazine	Representative indicator of triazines, PB
Miscellaneous	Di (2-ethylhexyl) phthalate	PB
	Methyl tert-butyl ether	Technical problems (odour)
	Nonylphenolethoxylates	CMR
	Pentachlorophenol	PB
	Polychlorinated biphenyl 28	PB; water solubility

Key: CMR = Carcinogenic/mutagenic/hazardous to reproduction and/ or endocrine-disrupting

P = Persistent

B = Bioaccumulating

T = High acute aquatic toxicity

Nitrates can form soluble salts and complexes with a range of metals and are therefore less susceptible to processes of adsorption, sedimentation, precipitation and filtration. Nitrates are readily removed by denitrifying bacteria (microbial degradation) under anaerobic conditions.

Denitrification reduces nitrates to nitric oxide, nitrous oxide but mainly free nitrogen gas, which is permanently lost to the atmosphere, and therefore this process can continue indefinitely without damaging the system. As with phosphates, nitrates are not removed via volatilisation or photolysis but they are readily bioaccumulated by plants as essential nutrients for growth.

2.3 Removal of metals

The metals included in the list of priority pollutants are Cd, Cu, Ni, Pb, Zn, Cr (as Cr(VI)) and Pt. Because there are numerous combinations of binding mechanisms and associated sediment fractions (see Section 1.2), these metals behave differently under the same set of environmental conditions, and therefore their potentials to be removed by the various mechanisms also vary. For example, Zn, Cr and Cd tend to have only a loose association with particulate matter in comparison to Pt and Cu, with Ni and Pb having the greatest binding affinity out of the metals listed (Revitt and Morrison (1987), ATSDR (2000), Ballach (1997)). Therefore the general trend in the order of potential for removal through adsorption, sedimentation and filtration processes is $Pb = Ni > Pt = Cu > Zn = Cr = Cd$.

The propensity for removal via precipitation is determined by the solubility of a particular metal compound, with the more soluble metal species possessing lower removal potentials and vice-versa. Based on this characteristic, Cd, Ni and Zn generally have a lower potential for removal compared to Pb, Cr, Cu and Pt out of the considered metals. A removal mechanism that is thought to be associated with all the listed metals is their precipitation as insoluble sulphides as a by-product of the reduction of sulphate by sulphate-reducing bacteria (see Section 1.3). This process only occurs under anaerobic conditions.

All of the metals listed have been reported in the literature to be bioaccumulated by plants (Scholes *et al.*, (1999), Lesniewska *et al.*, (2004), ASTDR, (2000)). Uptake is generally highest in the roots and rhizomes, with lesser amounts stored in the leaves (Kadlec and Knight, 1996). Plants can bioaccumulate metals at concentrations comparable to those accumulated by soils and sediments. However, in terms of overall metal loading, soils and sediment generally tend to be more important metal sinks or storage zones due to the greater number of binding sites available in soils and sediments as opposed to plants. None of the listed metals are removed from the water column to any great extent via processes of volatilisation or photolysis. Volatile organo-metal compounds can be produced under anaerobic conditions within sediments. However, atmospheric release of these compounds is not thought to be easily achieved as the need for anaerobic conditions would suggest that this process could only occur below a certain sediment depth, with the result that any volatile compounds produced would effectively remain 'locked' within the sediment unless the sediment was later disturbed.

2.4 Removal of polycyclic aromatic hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) are organic compounds that contain two or more benzene rings. Naphthalene contains two benzene rings and is used in the manufacture of plastics, dyes, solvents and insecticides (ATSDR, 2003). Pyrene and benzo(a)pyrene are four and five-ringed PAHs, respectively. Both compounds are formed as a result of the incomplete combustion of organic materials and are found in vehicle exhaust fumes and many types of industrial and domestic smoke, including cigarette smoke (ATSDR, 1995). PAHs can also be grouped by molecular weight into high, medium and low molecular weight classes, with

PAHs in each class reported to generally have a similar environmental fate (ATSDR, 1995). Naphthalene, with a molecular weight (MW) of 128.19 is classed as a low molecular weight PAH, pyrene (MW = 202.3) as a medium molecular weight PAH and benzo(a)pyrene (MW = 252.3) as a high molecular weight PAH.

Due to their generally low solubilities, all hydrocarbons readily sorb to solid surfaces (Jones, 1995). However, the K_d values of aqueous pyrene and benzo(a)pyrene are greater than that of naphthalene by at least 2 orders of magnitude (Eriksson *et al.*, 2004) and it is therefore anticipated that their removal by adsorption and sedimentation will be more significant. This assumption is supported by the approach of using molecular weight to predict PAH behaviour. For example, it is reported that adsorption and volatilisation are primary removal processes for medium and high molecular weight PAHs, whereas the primary processes for low molecular weight PAHs are reported to be biodegradation and volatilisation (ATSDR, 1995). However, as all three PAHs are considered to be insoluble it is predicted that the potential for removal by processes of both filtration (determined as a function of solubility and adsorption) and precipitation (based on solubility) will be equally important for all three compounds.

Volatilisation is known to be a significant process in the removal of many hydrocarbons from the water surface (Kadlec and Knight, 1996). All three selected PAHs are relatively volatile ($K_h = 10^{-7}$ - 10^{-3}) as indicated by Henry's Law constant values (Eriksson *et al.*, 2004). Therefore it is anticipated that their removal by volatilisation will be a significant process from BMPs which possess an aerated open water surface.

Plant uptake can be quantified using the octanol/water coefficient (K_{ow}) constant. As for the K_d values, the K_{ow} values for pyrene and benzo(a)pyrene are higher than for naphthalene (Eriksson *et al.*, 2004), which indicates that pyrene and benzo(a)pyrene have a greater potential for removal via plant uptake, although the plant uptake of PAHs from soils is reported to be generally quite low (ATSDR, 1995).

As a representative of the low molecular weight group of PAHs, microbial degradation is predicted to be a more important removal process for naphthalene than for either pyrene or benzo(a)pyrene. In addition, the comparatively lower K_d of naphthalene would also suggest that it should be the most susceptible of the three PAH compounds to microbial degradation as adsorption to sediment reduces bioavailability. However, these assumptions do not appear to be supported by the literature, with the microbial degradation of all three PAHs reported to be comparably low under both aerobic and anaerobic conditions (Eriksson *et al.*, 2004, ATSDR, 1995). In fact, the microbial degradation of naphthalene is not considered to be a significant process under anaerobic conditions whereas both pyrene and benzo (a) pyrene continue to be microbially degraded, all be it at a reduced level to that reported for these compounds under aerobic conditions.

The breakdown of PAHs by photolysis is thought to be a potentially important removal mechanism for many PAHs (Jones, 1995), although there does not appear to be any correlation between the extent or rate of photolysis and PAH structure. As expected, photodegradation rates are highest in surface waters and decrease with increasing water depth. For example, the photolytic half-life of naphthalene is estimated to be 71 hours in the surface layer but rapidly increases to an estimated 550 days at a water depth of 5m (ATSDR, 2003). Photolysis is also reported to be an important removal mechanism for pyrene and benzo(a)pyrene, with photolytic degradation rates reported to increase in the presence of certain sensitisers e.g. humic acid.

2.5 Removal of herbicides

In addition to the use of herbicides in agriculture for crop protection, industry, local and transport authorities also use herbicides for a variety of non-agricultural applications to roads, railways, pavements, paths and recreational areas (DEFRA, 2000). In the UK, for example, non-agricultural applications account for approximately 3% of total herbicide use. The herbicides included on the list of stormwater priority pollutants are pendimethalin, phenmedipham, terbutylazine and glyphosate. Pendimethalin is a dinitroaniline-based selective herbicide used for the control of broad-leaved weeds in a range of crops. Common trade names include 'Herbadox', 'Prowl' and 'Stomp'. Phenmedipham is a carbamate systemic herbicide that inhibits photosynthesis. It is typically applied to sugar beet and strawberries and its main tradenames include 'Betanal', 'Alegro' and 'Vanguard'. Glyphosate, pendimethalin and phenmedipham appear on the list of the fifty most widely used pesticides in the UK (DEFRA, 2000), and glyphosate and pendimethalin are also listed in the top 50 pesticides used in uncultivated non-agricultural areas in California (PAN Pesticide Database, 2002). Terbutylazine is a triazine herbicide that is absorbed mainly by the roots. It is used for broad-spectrum weed control in a range of crops, under trade names such as 'Gardoprim', 'Folar' and 'Topogard'. Glyphosate is a non-selective systemic herbicide which can be used on both crops and non-cropland. Glyphosate itself is an acid but it is commonly used in the salt form as an isopropylamine salt (Extonet, 1996). Trade names of products containing glyphosate include Gallup, Landmaster and Pondmaster.

The potential for the selected herbicides to be removed from the water column by adsorption and sedimentation is thought to be greatest for glyphosate as its adsorption coefficient is approximately 2 orders of magnitude greater than the adsorption coefficients reported for either pendimethalin, phenmedipham or terbutylazine (Extonet, 1996 ; Eriksson *et al.*, 2004) ref). In comparison, the potential for glyphosate to be removed by precipitation is lowest out of the selected herbicides as it has a much greater solubility. With regard to filtration, the potential for removal is considered to be greatest for phenmedipham, as it is both readily adsorbed and relatively insoluble, followed by glyphosate and pendimethalin and then terbutylazine. The relatively lower potential for glyphosate to be removed by filtration, despite its high potential to adsorb to particulate matter, is associated with its significantly greater solubility in comparison to the values reported for the other herbicides (Extonet, 1996).

As anticipated, all of the selected herbicides are readily bioaccumulated (medium bioaccumulation, $\log K_{ow} = 3 - 4.5$, high bioaccumulation, $\log K_{ow} > 4.5$ (Eriksson *et al.*, 2004), with the greatest potential for bioaccumulation associated with pendimethalin and glyphosate. The potential for the selected herbicides to be removed via volatilisation is greatest for pendimethalin, followed by terbutylazine and phenmedipham, with the volatilisation of all three of these herbicides being significantly greater than that reported for glyphosate (Eriksson *et al.*, 2004 ; Mensink and Janssen, 1994). A comparison of the volatilisation of pendimethalin and terbutylazine from two different soil types reported pendimethalin to be approximately 10 times more volatile than terbutylazine (Schroll *et al.*, 1999), which is consistent with the higher K_h value reported for pendimethalin. However, the percentage of pendimethalin applied which volatilised ranged from only 1.24 – 2.78% over a 28 day period and as pendimethalin has the highest potential for volatilisation out of the selected herbicides, it is therefore suggested that this process will not play a significant role in the removal of any of these herbicides.

Microbial degradation is generally considered to play a major role in the removal of many organic compounds within both the sediment and the water column (Kadlec and Knight, 1996). Pendimethalin, phenmedipham and glyphosate can be microbially degraded under both aerobic and anaerobic conditions (Mensink and Janssen, 1994 ; ARSUDA, 1995 and 1995a). Terbutylazine is also microbially degradable (Pesticide Manual, 2003), although it is not clear whether this process can occur under both aerobic and anaerobic conditions. The degradation of both glyphosate and phenmedipham is reported to be comparable under both aerobic and anaerobic conditions (Mensink and Janssen, 1994; ARSUDA, 1995a). In contrast, pendimethalin is reported to have a greater susceptibility to biodegradation under anaerobic conditions (ARSUDA, 1995).

Photolysis is a major abiotic degradation pathway for many compounds, and is reported to play a dominant role in the conversion and degradation of pesticides in aquatic systems (Mansour *et al.*, 1997). However, although pendimethalin, glyphosate and terbutylazine are reported to be broken down by photolytic reactions (Mansour *et al.*, 1997; Mensink and Janssen, 1994), these herbicides are considered to only have a low susceptibility to photodegradation processes. In contrast, phenmedipham is reported to have a much greater susceptibility to photodegradation making this process more important in its removal (Vialation *et al.*, 2001). However, it should also be noted that photochemical degradation processes (along with other degradation processes) may result in the formation of metabolites with properties, which are more hazardous than those of the parent compound, and the formation and environmental behaviour of these metabolites must also be considered.

2.6 Removal of miscellaneous compounds

The miscellaneous compounds included in the list of stormwater priority pollutants are nonylphenoethoxylate (NPEO), di-(2-ethylhexyl)-phthalate (DEHP), pentachlorophenol (PCP), polychlorinated biphenyl (PCB) and methyl tert-butyl ether (MTBE). NPEO is a man-made compound mainly used in the production of industrial and domestic detergents (Environment Agency, 2002). DEHP is a member of the 'phthalate' family that is commonly added to plastics to make them flexible (e.g. some vinyl materials may contain up to 40% DEHP) (ATSDR, 2002), and is ubiquitous in urban surface runoff. MTBE is a volatile organic compound added to petrol to increase the octane rating and reduce carbon monoxide emissions (ATSDR, 1996). Pentachlorophenol (PCP) is a chlorinated fungicide and insecticide, widely used to protect timber from fungal rot and wood-boring insects. However, PCP is no longer authorised for use in the UK (Environment Agency, 2002a). PCBs are synthetic organic which were widely used as coolants and lubricants in electrical equipment due to their inflammable nature and insulating properties (ASTDR, 2000).

DEHP and MTBE are reported to be degradable by both aerobic and anaerobic micro-organisms (ATSDR, 2002, ATSDR, 1996). Although the anaerobic degradation of DEHP is a slow process, the aerobic degradation is comparatively much more efficient and is therefore considered to be an important removal mechanism. The microbial degradation of MTBE is thought to be a relatively insignificant removal process under both aerobic and anaerobic conditions (ATSDR, 1996). With regard to NPEO, acclimation of microbes to the substance is an important determinant of the level of persistence (Maguire, 1999). For example, although not readily biodegradable, NPEO will ultimately be broken down over time to a variety of metabolites. Many of these metabolites, e.g. nonylphenol, are reported to be more persistent than the parent compound NPEO, although over time all metabolites will

eventually biodegrade. Pentachlorophenol (ARSUDA, 1995) and PCBs (ATSDR, 2000) can be microbially degraded under both aerobic and anaerobic conditions although biodegradation is considered to be more important in the removal of the former.

Both DEHP and PCBs have low solubility and high K_d values, favouring removal by both sedimentation and filtration processes (Williams *et al*, 1995 ; ATSDR, 2002 ; ATSDR, 2000). The solubility of NPEO varies with chain length and therefore the potential for removal via these processes will also vary in a corresponding manner. MTBE has a greater solubility than DEHP by several orders of magnitude (ATSDR, 1996), and therefore it is anticipated that its removal via sedimentation and filtration will be significantly lower. Like MTBE, PCP has a low tendency to adsorb however it has a much greater tendency to precipitate (ARSUDA, 1995), and is therefore considered to have a slightly higher overall potential for removal by sedimentation and filtration processes. The adsorption of PCP in soils is reported to be pH dependent, with no adsorption occurring above a pH of 6.8 (ATSDR, 2001).

The potential for removal through uptake by plants can be assessed through the K_{ow} value which increases in the order MTBE<NPEO<PCP<PCB<DEHP (ATSDR, 1996, 2000, 2001 and 2003), and this can be related to a general trend of increasing potential for each compound to be bioaccumulated. MTBE is reported to rapidly volatilise from surface waters and surface soils, and volatilisation is also considered to be an important process in the removal of PCBs due to a comparable K_h value (ATSDR, 2000 and 2002). The values reported for PCP and a range of NPEO metabolites are similar to or less than the K_h value of DEHP (Montgomery-Brown and Reinhard, 2003 ; ATSDR, 2001 and 2002) suggesting that volatilisation is not a significant process in the removal of these three compounds.

MTBE does not absorb sunlight in the range above 210nm, and therefore photolytic decay is not considered to be a significant process in its removal from the water column (ATSDR, 1996), nor is it reported to be an important process in the removal of DEHP (ATSDR, 2002). NPEO is known to be broken down by photodegradation processes, but further research is necessary before its importance as a pollutant removal mechanism can be quantified (Montgomery-Brown and Reinhard, 2003). Pentachlorophenol is reported to be broken down by photolytic reactions (Chi and Huang, 2004). This was reported to be higher in surface water in comparison to subsurface water, which the authors suggested could be associated with the correspondingly higher concentrations of dissolved organic matter in the surface layers which have UV-absorbing properties (Chi and Huang, 2004). Salinity and pH are also reported to alter the photolytic degradation rate, with the photodegradation rate of pentachlorophenol reported to decline below pH 6 and to decrease with increasing salinity (Chi and Huang, 2004).

3 Removal of Stormwater Priority Pollutants by BMPs

Taking into consideration both the removal processes associated with BMPs (Section 1) and the potential for removal of a list of identified stormwater priority pollutants by these mechanisms (Section 2), the fate of these pollutants through different BMPs can be evaluated by considering the importance of each removal mechanism within each BMP. The BMPs evaluated within this section are those that were identified and described in DayWater Deliverable 5.1 (Review of the Use of BMPs in Europe, 2003). These BMPs, together with brief descriptions, are listed in Table 3.1.

Table 3-1. Descriptions of types of structural BMPs

System type	Description	Section of this report describing removal processes
Swales	Vegetated broad shallow channels for transporting stormwater.	3.1
Filter strip	Grassed or vegetated strip of ground that stormwater flows across.	3.1
Filter drains	Gravelled areas where stormwater can drain through the gravel to be collected in a pipe.	3.1
Soakaways	Underground chamber or rock-filled volume; stormwater soaks into the ground via the base and sides.	3.2
Infiltration trench	A long thin soakaway.	3.2
Infiltration basin	Detains stormwater above ground which then soaks away into the ground through the base.	3.2
Sedimentation tank (also known as silt trap)	A symmetrical structure composed of synthetic material capable of containing the appropriate depth of water to assist the settling of suspended solids under quiescent conditions.	3.3
Lagoons	Pond designed for the settlement of suspended solids	3.3
Retention ponds (balancing ponds)	Contain some water at all times and retains incoming stormwater.	3.3
Detention basins	Dry most of the time and able to store rainwater during wet conditions.	3.3
Extended detention basin	Dry most of the time and able to store rainwater during wet conditions for up to 24 hours.	3.3
Constructed wetlands	Artificial vegetated system with extended retention time.	3.3
Porous asphalt	Open graded powdered/crushed stone with binder; high void ratio.	3.4
Porous paving	Continuous surface with high void content, porous blocks or solid blocks with adjoining infiltration spaces; an associated reservoir structure provides storage.	3.4
Combined/hybrid system	Combination of two or more of any of the above measures (but could also include a conventional drainage system as one of the elements).	

3.1 Filter Strips, Swales and Filter Drains

Primary removal mechanisms in filter strips and swales include adsorption and precipitation. The presence of vegetation in both BMPs contributes to the overall pollutant removal process through:

- enhancement of sedimentation through flow retardation
- physical filtration of the stormwater increasing the retention of particulate matter
- plant uptake of pollutants

These biofiltration processes are generally thought to be more efficient in swales than in filter strips, as the former typically detain flow for longer periods of time assisting the particle trapping efficiency, which is reported to increase exponentially with residence time (Bäckström, 2002). The important role that vegetation plays in solids removal has also been demonstrated in various research studies carried out in Sweden (Bäckström, 2002, Bäckström *et al.*, 2002). These studies reported that swales with the thinnest vegetation had the lowest removal efficiencies, and that there was no significant relationship between suspended solid removal and swale design in grass channels that were densely turfed. A strong relationship between suspended solid removal and particle size was reported, with particles $>25\mu\text{m}$ being retained in a swale to a significantly higher degree than particles $<25\mu\text{m}$. However, the study also suggested that a proportion of the latter fine particles, which would preferentially be mobilised by low flows at the start of a storm event, may have been removed in the swale via infiltration (see Section 3.2).

Several studies have reported an exponential decrease in pollutant concentrations with increasing swale length, suggesting that the majority of pollutants are trapped in the first few metres of the swale or filter strip (Wang *et al.*, 1981, Coyne *et al.*, 1998, Deletic, 1999). Research has shown that a majority of stormwater pollutants are trapped in the upper soil layers (Mikkelsen *et al.*, 1997, Dierkes and Geiger, 1999), and filter strips and grass swales are therefore classified as stormwater infiltration facilities under some schemes. Stormwater pollutants that do infiltrate into the upper soil layers would also be subject to further pollutant removal processes in the soil through microbial degradation. However, there is also potential for pollutants accumulated on the surface of swales and filter strips to be re-entrained by subsequent storms. For example, filter strips have been reported to generate nutrient-enhanced outflows for intense storm events (see Section 4.1.2).

As both filter strips and, to a greater extent, swales detain stormwater flow, there is the potential for photolysis and volatilisation to occur in both systems. However, the contributions of these processes to overall pollutant removal are considerably less than for the other removal processes described above. In addition, both swales and filter strips have only a low removal potentials for soluble and loosely adsorbed species as these are not influenced by the main prevalent removal processes, other than plant uptake.

The primary removal processes in filter drains are adsorption to substrate, microbial degradation and filtration. The typically gravel composition of filter drains allows close contact between stormwater and substrate contributing to the relatively high efficiencies of adsorption and filtration processes, with the gravel substrate also providing abundant attachment areas for the growth of micro-organisms. The gravel also provides attachment sites for algal growth, with associated potential for plant uptake, although this process is not considered to be as important as microbial degradation. Due to the comparatively low volume

of quiescent water within a filter drain in comparison to e.g. a retention pond, the potential for removal by sedimentation is considered to be relatively lower. Photolysis is not thought to occur in filter drains as stormwater generally rapidly infiltrates into the body of the system out of reach of direct sunlight. In contrast, volatilisation is thought to have a low potential to occur as this process requires exposure of stormwater to atmospheric conditions (as opposed to requiring direct sunlight), which occur within unsaturated filter drains.

3.2 Infiltration Systems

The general category 'infiltration systems' includes soakaways, infiltration trenches and infiltration basins, none of which encourage surface stormwater flow. Swale-trench systems that are widely applied in central Europe could be included in this category or in the previous biofiltration classification. Adsorption, precipitation, microbial degradation and filtration are the main removal processes reported to occur in infiltration systems. Sedimentation is important in infiltration basins, which because they detain stormwater above ground are also subject to volatilisation and photolysis. These processes are less prevalent in soakaways and infiltration trenches where the stormwater is detained underground within the permeable structures. Following storage either above or below ground, stormwater infiltrates into the surrounding ground. Barbosa and Hvitved-Jacobsen (2001) suggest that the main processes occurring in an infiltration basin are the sedimentation of suspended solids and particulate pollutants, and the retention of dissolved and colloidal pollutants that infiltrate into the soil. These processes can also occur in soakaways and infiltration trenches. However, in these systems filtration takes over as the major pollutant removal process.

Nitrates and phosphates are primarily removed in infiltration systems by precipitation or adsorption processes. However, as nitrates are highly soluble, nitrogen-containing compounds may be transported to a greater soil depth in comparison to phosphates, which readily precipitate out at, or adsorb onto, soil surfaces. This adsorption of phosphate on to particulate matter is reported to be a two-step process (Pitt *et al.*, 1999), involving initial sorption to the soil surface followed by transformation into phosphorous-containing mineraloids or minerals. However, in the presence of competing anions or if the pH is not favourable the degree of phosphate sorption will be reduced.

Microbial degradation is an important process in many infiltration systems. Both organic compounds and metals may be removed/immobilised via direct and indirect microbial processes, although Mikkelsen *et al.* (1994) has noted that organic degradation products may not necessarily be less toxic than the parent compound. Laboratory studies involving sand columns inoculated with the soil bacteria *Pseudomonas putida* reported that the presence of these micro-organisms influenced the retention of Zn (Hebrard and Delolme, 1999).

The ability of micro-organisms to degrade oil is described later in this review in the section on alternative road surfaces (see Section 3.4) and similar processes of microbial oil degradation can be present in infiltration systems. Microbial degradation is also thought to be a primary process in the removal of a range of other organic compounds, such as pesticides, by infiltration devices (Pitt *et al.*, 1999).

Organic compounds which volatilise to a significant degree, such as naphthalene and pyrene, may be removed from the water column in infiltration basins particularly when there is prolonged surface detention of stormwater. This may also be a factor for compounds, such as pendimethalin and phenmedipham, which are susceptible to photolytic degradation processes.

For both processes, removal rates are highest within surface waters and decrease with increasing depth in the water column. However, the photolysis within the water column is not thought to be a highly significant process, due to the relatively limited opportunity for waterborne pollutants to absorb light.

As for other permeable materials (see Section 3.4), the clogging of the voids in infiltration devices has been reported and concern expressed that this can lead to reduced efficiency as well as acting as an eventual pollutant source. However, in contrast to this concern, Mikkelsen (1995) has reported that rather than being a pollutant source, a clogged topsoil layer in fact can act as a pollutant sink, presumably primarily through processes such as adsorption and filtration. Nevertheless, clogging can lead to a 'sealed' surface, which prevents or substantially reduces the ability of stormwater to infiltrate into the substrate. Further work by Mikkelsen *et al.*, (1996) also suggested the potential of accumulated runoff sludge in sub-surface infiltration systems to retain pollutants.

Concern has also been expressed over the possibility of infiltration devices resulting in the contamination of groundwater. However, this can usually be avoided through the use of careful site selection, correct design and appropriate soil type. This assertion is supported by a study by Datry *et al.*, (2003) which investigated the pollutant content of bed sediments collected from a 30-year-old infiltration basin. The authors reported that the sediments contained high concentrations of organic carbon, nitrogen, phosphorus, total hydrocarbons and metals, and identified 33 organic compounds including 15 polycyclic aromatic hydrocarbons and 10 polychlorinated biphenyls. Of particular interest were the results of the sediment pore water analyses which found that, despite high concentrations in the sediment, hydrocarbons and metals were rarely detected in pore water. This study clearly demonstrates three important points, namely that :

- infiltration basins can remove a wide range of organic and inorganic pollutants from the water column
- a range of pollutants strongly adsorb to particulate matter and that once adsorbed these pollutants are no longer mobile
- infiltration basins may perform effectively over an extended period of time.

However, as discussed in Section 3.3, it is also important to bear in mind that little if any work has been conducted in infiltration systems under charged wet conditions when hydraulic gradients will be high and large pressure differences are generated at the sediment-water interface, as well as within sediment pores. Under such conditions, there may be a more rapid downward transmission of pollutants, as well as enhanced exchange due to micro-turbulence in the macro-pores. In addition, most research work to-date has assumed Darcian flow in the upper soil and substrate layers. This ignores the potential effect of rapid fissured flow via macro-pores, cracks, and fissures in the subsoil and by widened joint planes in the underlying stratum. Such fissured flow occurs readily in calcareous rocks such as limestone, chalk and massive sandstones, as well as in soils that have been subjected to prolonged drying conditions. The propensity for pollutants to diffuse and flow outwards from infiltration structures in the long-term needs to be the subject of further research and field work before a full understanding of the processes and issues can be attained.

3.3 Storage Facilities

The BMP category of storage facilities includes sedimentation tanks, retention ponds, detention ponds, extended detention ponds, lagoons and constructed wetland systems. Major potential removal processes in sedimentation tanks, retention ponds, detention ponds and extended detention ponds include sedimentation, adsorption, precipitation and volatilisation. The primary removal mechanisms differ slightly for constructed wetlands, which are generally more complex environments, as they also include enhanced microbial degradation, filtration and plant uptake.

Sedimentation tanks, lagoons, retention ponds and constructed wetlands generally maintain a permanent open water body whereas, detention ponds and extended detention ponds are generally dry, being designed to only store stormwater for a limited time following a rainfall event. The water capacity of these treatment systems typically allows a reduction in stormwater velocity with the induced quiescent conditions promoting the sedimentation of particulate matter and associated pollutants. Appropriately designed ponds can therefore provide ideal conditions for the removal of many stormwater priority pollutants via adsorption, precipitation and sedimentation.

As well as providing treatment of stormwater through the processes described above, constructed wetlands also enhance and incorporate further removal processes, primarily due to the presence of dense stands of vegetation. Wetland plants can both directly and indirectly increase treatment potentials due to:

- buffering of incoming flow thereby enhancing hydraulic conditions for the settlement of suspended solids
- diffusing of oxygen from plant roots (hence their ability to grow in saturated soils) resulting in the presence of oxygen-rich zones in the otherwise anaerobic sediment which enables wetlands to support both aerobic and anaerobic microbial populations
- along with the sediment, provision of a diversity of attachment sites for micro-organisms, with the plants also providing ready sources of carbon and nutrients for microbial growth.
- development of an extensive network of roots, which can help stabilise sediment and reduce erosion.
- bio-accumulation of nutrients, metals and organics from both the water column and the sediment (although in terms of the overall mass of metals and organics removed, bioaccumulation is thought to be a significantly less important process than adsorption and sedimentation)
- seasonal die-back of plants resulting in the formation of a litter layer, which may physically filter stormwater and retain particulate matter

3.4 Alternative Road Structures

Adsorption and filtration are the primary physical removal mechanisms in porous paving and porous asphalt (Legret *et al.*, 2004, Stotz and Krauth, 1994, Noutsopoulos *et al.*, 2004). For example, porous paving is reported to be a highly efficient trap for stormwater particulate pollution (Legret *et al.*, 2004), and also therefore for the pollutants that strongly adsorb to particulate material such as BOD, COD, phosphates, Pb, Ni, pyrene, benzo(a)pyrene and DEHP. Removal of dissolved metal species is also reported, and is understood to be due to a

combination of adsorption to suspended solids that have been retained through filtration and adsorption to the porous material itself (Legret and Raimbault, 2003).

The majority of pollutants are accumulated in the top layers of permeable surfaces, which may lead to clogging of the porous structure. The trapped material is able to physically remove fine particulate material and also to adsorb pollutants suggesting that the clogging of voids may not always initially have a negative impact on treatment efficiency. However, clogging can reach a level leading to a reduction in treatment performance, either through the saturation of available binding sites or the physical blocking of pores. In these situations, research has shown that maintenance with high pressure spraying and/or suction can significantly improve performance (Legret *et al.*, 2004). The availability of metals bound up in the trapped material has been investigated and, although some studies have suggested that these metals are unlikely to be released under normal field conditions, the strongly bound metals can be released using more intensive extraction procedures (Legret *et al.*, 2004). This raises important issues regarding the disposal of both the clogging materials and the permeable structure at its end-of-life.

Permeable surfaces with a reservoir structure are capable of retaining stormwater for extended periods of time. Retention and the associated potential for microbial degradation are two of the main processes associated with the removal of PAHs. Hydrocarbon retention is reported to depend on the size and volume of substrate material. As well as physically or chemically retaining hydrocarbons, the granular materials used to build the permeable structure also provide substrate attachment sites for microbial growth. Studies have shown that such permeable structures can maintain a viable microbial population, suggesting that they may effectively be considered as *in-situ* bioreactors (Pratt *et al.*, 1998). Conditions for microbial growth within porous structures are thought to be generally aerobic, with the presence of anaerobic conditions possibly developing during stormwater retention and within the clogging materials. A study by Newman *et al.*, (2001) compared the oil removing performance of a permeable surface inoculated with a commercial oil-degrading microbial population to that of a permeable surface with its own indigenous microbial population. This study reported that there was no significant difference between the abilities of the microbial populations to degrade oil when sufficient nutrients were supplied. Pratt *et al.*, (1998) reported the removal of hydrocarbons by permeable surfaces due to a combination of microbial degradation and retention by the construction materials. A range of metals as well as nitrates are also known to be removed from the water column by microbial degradation (Scholes, 2000), so that the presence of an oil-degrading microbial consortium may also have a positive impact on the reduction of these parameters. However, it is important to note that biodegradation can only be effective when nutrient levels, above those normally found in stormwater runoff, are available to support bacterial activity.

Further treatment potential is available if the stormwater is allowed to infiltrate into soil underlying the porous structure (see Section 3.2). Concern over the possibility of contamination of underlying soils has been expressed (personal communication ; J B Ellis). Field and modelling studies carried out in France on a permeable surface with reservoir structure receiving road runoff (traffic density of 1,600 vehicles/day) concluded that reservoir structures did not result in soil pollution (Legret *et al.*, 2004). However, studies on infiltration basins (where infiltration into underlying soils forms an integral part of the treatment process) have reported annual increases in soil metal concentrations, with an exponential mobilisation of soluble species with depth (e.g. Ellis *et al.*, 2002). The extent of pollutant movement

within underlying soils, particularly with regard to pollution diffusion in relation to the wetting front and fissure-flow, are areas highlighted as requiring further work.

4 BMP Removal Efficiencies

4.1 Swales, Filter Strips and Filter Drains

4.1.1 Swales

Table 4.1 shows the range of pollutant removal efficiencies that have been noted by various European workers for grass swales (Scholes *et al.*, 2003). It is clear that whilst in general, good removal rates can be achieved by such systems, there is still considerable variability in performance. Investigations of soluble metal species, nutrients or bacteria suggest that direct removal of these contaminants is limited and it may be that swales can only provide an efficient performance for solids and heavy organics such as leaf litter (and their attached pollutants) as well as oils. Solids removal performance increases as total suspended solids (TSS) concentrations increase, and with inflow concentrations below 30-40 mg/l, little reduction can be expected.

Table 4-1. Swale Pollutant Concentrations, Loadings and Removal Efficiencies.

Pollutant Parameter	EMC and Range (mg/l)	Load (kg/ha/yr)	% Removal Efficiency
TSS	25.0 (7.0 - 47.0)	–	86 (55 - 91)
Total Zn	0.032 (0.011 - 0.143)	7.05 (1.85 - 9.2)	83 (63 - 93)
Total Pb	0.079 (0.014 - 0.144)	0.78 (0.25 - 2.61)	54 (17 - 76)
COD	39.0 (2.0 - 76.0)	–	62 (53 - 74)

The 2001 Scottish SUDS (Wild *et al.*, 2002) database lists a total of 161 swales but only four have any substantial monitored data. Two sites at Dundee estimate initial wetting losses at 1.2 to 5.0 mm in comparison to 0.3 – 0.4 mm for the adjacent road surfaces. The corresponding pairs of percentage runoff rates are 6.5% - 37% and 41% - 53% respectively. At both grass channel sites, increased lag times before runoff and reduced peak flows have been observed. Outlet pollutant concentrations have been generally very low, with significant removal of SS and other chemical determinants (e.g hydrocarbons reduced by 36%), although there is some evidence of increased metal outflows (Jefferies *et al.*, 1998).

Irrespective of these reservations, comparison of the data in Table 4.1 and Table 4.6 implies that pollutant loadings in conventional swale channels are generally well below most national criteria for biosolid disposal to land. However, the relatively low loading limits specified for Cd might provide a more critical restriction. Nevertheless, swales serve as effective pre-treatment systems prior to further downstream SUDS facilities such as detention basins or wetlands.

The results of the grab samples taken in three Swedish roadside swales during snowmelt in March-April 2000 are presented in Table 4.2. Even though no flow measurements were obtained at the time of sampling, the results indicate that suspended solids and total metals are retained to a large degree in a snow-covered swale (78-99% removal). Dissolved Cu and Zn showed negative removal rates, thus the dissolved species of these metals were washed out with the melt water even at low to medium flow rates. Suspended solids removals in the range of 79 to 98 % were observed during simulated runoff event experiments. The swales with thin vegetation had the lowest suspended solids removals.

Table 4-2. Pollutant Concentrations in Snow and Melt water in Three Roadside Swales in Luleå

		pH	SS (mg/l)	Cu (ug/l)	Total Pb (ug/l)	Zn (ug/l)	Dissolved		
							Cu (ug/l)	Pb (ug/l)	Zn (ug/l)
Site A	Snow	6.91	1800	214	212	525	4.78	0.177	13.6
Bodenv.	Melt	6.89	13	15.3	2.44	33.4	7.00	0.105	18.0
2000-03-29	water Reduction	-	99%	93%	99%	94%	-46%	41%	-32%
Site B	Snow	6.69	1000	83.7	55.9	275	1.43	0.137	13.5
Hertsöv.	Melt	6.70	12	5.00	1.93	60.5	1.84	0.097	56.7
2000-04-10	water Reduction	-	99%	94%	97%	78%	-29%	29%	-320%
Site C	Snow	6.99	5400	520	189	1240	2.00	0.143	6.37
Lulsundet	Melt	7.03	240	21.9	7.43	72.8	3.28	0.090	16.0
2000-04-10	water Reduction	-	96%	96%	96%	94%	-64%	37%	-151%

M Bäckström (2003)

At the Södra Hamnleden site in Luleå, N Sweden, removals of suspended solids were observed for three out of seven events (Bäckström *et al*, 2002). It seems that no significant removal occurred in the swale when the influent concentrations of suspended solids were below approximately 40 mg/l which is approaching the irreducible background level for these systems. Removal efficiencies of more than 50% were generally observed when influent suspended solids concentrations were above 100 mg/l. Studies of particle size removal in passage through the swales showed that particles larger than 25 µm were retained in the swale whilst particles in the size interval 9 to 15 µm seemed to be most easily transported out of the swale. Smaller particles (4 to 9 µm) showed a lower net transport to the swale outlet. Removal of fine particulate matter close to the swale inlet will reduce the potential contact time between pollutants and particles, and thus it might be expected that such systems could have low efficiencies for adsorbable micro-pollutants as well as for soluble species.

Road runoff and swale runoff samples from four events at the Södra Hamnleden site were analysed for total and dissolved concentrations of Pb, Cu and Zn. In general the highest removal rates were found for total Zn, with mean concentrations in road runoff ranging from 50-135 µg/l, in comparison to a range of 67-97µg/l in runoff from the swale. During events with low influent concentrations the swale acted as a source for all three metals. However, the concentrations of both total and dissolved Cu were lower in road runoff compared with swale runoff for all four events. The event mean concentrations of dissolved Cu were two to

four times higher in swale runoff than in road runoff, and it suggested that this might be related to colloid-facilitated transport as a result of complexation. Thus, the Södra Hamnleden swale appeared to contribute to the transport of Cu during the time of the study. In the case of Pb, no significant differences in concentrations were found between road runoff and swale runoff samples.

Table 4.3 presents the calculated mass loads and load reductions for water volume, suspended solids, Cu, and Zn based on four rain events with a total precipitation of 47.4 mm for one of the swale sites having a length of 110 m. The mass load reductions of pollutants give valuable information on the long-term performance of a grassed swale. At the Södra Hamnleden swale, the overall mass load reduction of suspended solids was 70% even though negative removals were observed during some rain events. Mass load reductions were also obtained for total Cu and total Zn. However, heavy metal removal was lower than suspended solids removal, with negative removal efficiencies reported for dissolved Cu.

Table 4-3. Total Mass Flows at the Södra Hamnleden Swale Site During Four Rain Events in May-June 2000

	Water (m ³)	Suspended solids (kg)	Cu		Zn	
			Total (g)	Dissolved (g)	Total (g)	Dissolved (g)
From road to swale	19	2.1	0.56	0.11	2.2	0.59
From swale to recipient	8.7	0.63	0.37	0.15	0.73	0.20
Load reduction (%)	54	70	34	-27	66	66

Bäckström (2003)

The results of the elemental analyses of grab samples of surface sediment removed by a sweeping machine from the swale at Bodenvägen, Luleå are presented in Table 4.4. The results indicate that none of the ten heavy metals and PAHs presented here are likely to exist in very high concentrations in surface sediments in the swales of Luleå.

Table 4-4. Pollutant Concentrations (mg/kg) in Surface Sediment in a Grassed Swale (Bodenvägen, Luleå)

Element	Surface sediment at Bodenvägen, Luleå 2001-05-17	Guidelines from the Swedish EPA (1997), sensitive soil use
As	2.6	15
Cd	0.12	0,4
Co	12	30
Cr	30	120
Cu	32	100
Ni	22	40
Pb	15	80
V	86	100
Zn	86	400
Carcinogenic PAHs ¹	0.03	0,3
Non Carcinogenic PAHs ²	0.9	20
Sum of sixteen PAHs according to USEPA	0.9	-

¹Sum of benzo(a)anthracene, chrysene, benzo(b)fluoranthene, benzo(a)pyrene, indeno(1,2,3-cd)pyrene, and dibenzo(a,h)anthracene.

²Sum of naphthalene, acenaphthalene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, and benzo(ghi)perylene.
Bäckström (2003)

A previous study in southern Sweden (Scholes *et al.*, 2003) has characterised the sediment quality of roadside swales at 19 sites and found that a majority of the investigated top sediments close to highways contained high amounts of oil, Pb and Cd which exceeded the Swedish EPA guidelines (see Table 4.6). Therefore it cannot be expected that all grass swales will inevitably maintain levels below the appropriate threshold disposal limits.

The US EPA National BMP Database has records for 32 filter strips and 14 swales which indicate that such biofilters showed an average of 20% less runoff volume on a storm-by-storm basis and performed consistently better in terms of runoff control than other BMP options. However, their performance in water quality terms was less satisfactory in that they demonstrated a wide data scatter, removing between 42% to 89% of total suspended solids for some 300 recorded storm events. The nature of flow through swale channels may well lead to entrainment and flushing of loosely bound pollutants from the grass and surface soil layers. This inefficient performance may be exacerbated by poor maintenance and flow channelling as well as by over-frequent mowing activities.

4.1.2 Filter Strips

There are few reports of the operation or performance of filter strips in the UK, Scandinavia or continental Europe but the pollutant removal efficiency of a filter strip receiving runoff from a car park in the US has been described by Yu *et al.* (1987). Optimum removal rates were achieved within 18 to 25 m slope lengths but the density and height of the grass sward were also important factors influencing pollutant removal effectiveness. Sheet flow over the grass surface was achieved using a level spreader and the associated average percentage removal efficiencies were 71, 38, 10, 25, and 50% for TSS, P_{tot} , N_{tot} , Pb and Zn, respectively.

A 10 m grass filter strip has been installed as an initial receptor for surface runoff from a heavy goods vehicle (HGV) parking area at the Hopwood Park Motorway Service Station on the M42 in Oxford. The first 10 mm first-flush volume is directed over the filter strip to intercept silt prior to discharge into an infiltration trench. No problems have been reported (although the grass surface is occasionally heavily oiled) and the combined filter strip and infiltration trench collectively show reductions of 94% for total Zn, 82% for total Cu and 97% for Pb as well as 99% and 98% reductions for SS and BOD, respectively (Bray, 2001).

A 1000 m² grassed filter strip was included as part of a hybrid stormwater control system for a 81.5 ha urban development draining to Lake Magelungen in Sweden. A two year monitoring programme indicated that there was a 70% - 80% summer reduction in phosphorus washout but increases were noted in nitrogen and heavy metal concentrations across the grass filter. These increases were largely explicable in terms of surface channelling of the filter strip which was difficult to control and as a result the filter strip was taken out of use (Rosen, 1996).

German studies (Dierkes and Geiger, 1999) of stormwater runoff from 5 highway sites over adjacent grass filter strips in the vicinity of Essen showed heavy metal removal efficiencies of 60%, 75%, 58% and 59% for Pb, Zn, Cu and Cd respectively. Comparison of the upper 5 cm soil layer beneath the filter strip with the lowest 30 cm layers also showed considerable trace metal retention capacity. These German studies are therefore in conflict to some degree with the observations of the performance of grass swales. It may be that the longer contact time

and surface area provided by filter strips (as well as more uniform flow), yields greater removal potential than for grass swales, which frequently exhibit channelled flows.

Table 4.5 indicates the range of pollutant removal efficiencies that might be expected for a grass filter strip based on long term data from generally similar climatic regions such as Western Europe, New Zealand and the eastern Atlantic seaboard of the United States. It can be seen that in general the removal efficiency is only modest, but nevertheless there does appear to be potential for the removal of total metals and hydrocarbons.

Table 4-5. General Range of Removal Efficiencies for a Grass Filter

	TSS	N _{Total}	Hydrocarbons	Bacteria	Metals	
					Total	Dissolved
Filter Strip	10 - 40	10 - 30	60 - 75	30 - 40	70 - 80	10 - 20

Table 4.6 shows heavy metal loading criteria for direct disposal of contaminated sediment to land which might be applied to the toxicity assessment of sedimented deposits occurring in biofiltration and retention BMPs. The EU Soil Directive and the Swedish EPA standards are the most generally stringent criteria, but even these would not normally be exceeded within the expected 25-30 year lifetime of a swale or filter device. For many retention facilities however, such as detention/retention basins, infiltration basins and wetlands, contaminated sediment accumulations could increment mass loads which might fall at or above the acceptable limits and thus require disposal as a classified hazardous waste.

Table 4-6. Loading Criteria for Biosolid Disposal to Land

Pollutant	UK ICRL (mg/kg)	EU 1986 Directive				Dutch Ministry of Public Housing (mg/kg)	Swedish EPA "Moderate pollution" mg/kg	US EPA 503 Regs kg/ha/yr	Ontario Ministry of Env. (Lowest Effect Level) mg/kg
		Biosolids (mg/kg)	UK 90% 1996/97 Biosolids Limit	Soil mg/kg	Application loading 10 yr avge (kg/ha/yr)				
Zn	300	2500 - 4000	1076	150 - 300	30	720	175 - 300	140	110.0
Pb	2000	750 - 1200	288	50 - 300	15	530	30 - 100	15	31.0
Cd	15	20 - 40	3.4	1.3	0.15	12	1.7 - 2.0	1.9	1.0
Cu		1000 - 1750	758	50 - 140	12		25 - 50		25

4.1.3 Filter drains

The results of an experimental study carried out adjacent to the M1 motorway, north of Luton in SE England, are shown in Table 4.7. The effective removal of both conventional and toxic pollutants by a 55 m length of filter drain is clearly indicated although removal efficiencies varied considerably between storms and seasons. The filter drain yielded pollutant concentrations of 28 mg/l and 3.7 mg/l for suspended solids (SS) and total oil respectively in comparison to 92 mg/l and 30 mg/l for the untreated motorway runoff.

Table 4-7. Mean Percentage Annual Removal Efficiencies for a UK Motorway Filter Drain Treatment System.

Pollutant	Percentage removal efficiency
TSS	85
Pb _{tot}	83
Zn _{tot}	81
Zn _{diss}	56
COD	59
Oil	70
PAH	70

Perry and McIntyre (1986).

A 750 m filter drain in Aberdeen, Scotland receiving inflows from 44 roadside gullies alongside the A944 highway has been monitored as part of the data collection process for the Scottish SUDS database. The percentage runoff outflow from the perforated pipe under-drain ranged from 0.85 – 196.0%, with a mean value of 42% (the events over 100% were mainly due to snowmelt). Although reductions in TSS of 74.3% have been recorded, water quality was highly variable showing strong seasonal differences (Wild *et al.*, 2002).

4.2 Infiltration Systems

Improvements in water quality during the infiltration process are achieved through physical filtration, adsorption of pollutants by the infiltration media (infill and surrounding soils) and microbiological removal of pollutants due to reactions on the surface of the media. The treatment efficiency is dependent on the contact time between the drainage waters and the infiltration media. Mikkelsen *et al* (1997) have demonstrated the effective pollutant trapping potential of these systems whilst also highlighting that a solid waste disposal problem could eventually result. The typical ranges of pollutant removal efficiencies which have been monitored for infiltration systems are shown in Table 4.8. The variability of the data is not unexpected given the size and scale of the infiltration systems for which results have been obtained.

Table 4-8 Ranges of pollutant removal efficiencies for infiltration systems (soakaways, infiltration trenches, and infiltration basins).

Pollutant	Range of percentage removal efficiencies
Suspended solids	60-90
COD/BOD	70
Oil/Hydrocarbons	70-90
Nutrients	20-50
Total metal	50-90
Dissolved metal	20-35

In addition to the use of natural soil, constructed filter systems containing different sized sand or gravel can be used to remove contaminants from stormwater. There have also been investigations of more specialised filter materials, which are able to provide increased sorption of dissolved constituents and hence improve effluent quality. Steiner and Boller

(1997) have reviewed the different substrates available and identified three possible adsorbent classes which in order of their effectiveness are metal hydroxides (such as iron and aluminium), activated carbon, and zeolites. However, this study did not relate specifically to pollutant removal from stormwater.

Recently a number of new media types have been advocated for possible use in infiltration systems as stormwater treatment filters. One of these is iron-oxide coated sand (IOCS), which Møller *et al* (2002) have shown, using column experiments, is capable of metal removal efficiencies of over 90% for Pb, Cu and Zn compared to values of >95%, 35% and 5%, respectively under the same conditions using an uncoated filter sand. The coating of goethite was found to be at least twice as effective as ferrihydrite with respect to the adsorption capacity of Cu and Zn. Sansalone (1999) has advocated the use of oxide-coated sand (OCS) within so-called partial exfiltration trenches (PET) as an efficient removal system for Cd, Cu, Pb and Zn within urban drainage waters. Such systems performed effectively with regard to the adsorption of dissolved metals as well as acting as filters for particulate-bound fractions. OCS had a greater capacity than untreated silica sand for all dissolved fractions and this improved as the pH was raised from 6.5 to 8.0.

In Sweden, Farm (2002) has investigated the performances of three selected natural filter materials (calcium silicate rock (opoka), zeolite and pine bark) with regard to the removal of Cu and Zn from highway runoff under field conditions. The average reductions of these metals for infiltrated stormwater were between 63 and 81%. Pitcher *et al* (2004) tested the abilities of two zeolites (synthetic MAP and natural mordenite) to remove dissolved heavy metals from simulated and spiked motorway stormwater using batch experiments. Synthetic zeolite MAP showed almost complete removal (>91%) of the studied heavy metals (Zn, Cu, Pb, Cd) but its use was implicated in possible serious environmental consequences as a result of increasing sodium levels to 295 mg/l, removing calcium and increasing the pH of the spiked motorway stormwater to 8.5. Mordenite was less effective at reducing the levels of heavy metals (42-89% in synthetic solution, 6-44% in motorway stormwater) and exhibited a preference for Pb>Cu>Zn similar to Cd.

4.2.1 Soakaways

Soakaways are widely used in the UK for the attenuation and treatment of highway runoff through gradual infiltration into the surrounding soil after passing through a coarse sediment trap and then either a chamber or stone filled system. The majority of research on soakaways (albeit very limited in extent) would suggest that pollutant concentrations peak in the 0.4 - 0.5 m soil levels immediately below the base of the soakaway, declining exponentially with depth to background levels (Price, 1994). However, there are concerns about the behaviour of highly soluble contaminants such as certain zinc and cadmium species, selected herbicides and MTBE as tracer studies have indicated that trace levels could reach abstraction wells at a distance of 3 km away from a soakaway injection point (Barker *et al*, 1999). The observed depth-concentration profiles suggest that pollutant decay rates are strongly influenced by both the available total organic carbon and the clay-silt percentages.

The accumulation of polluted sediments in soakaways receiving highway runoff in Switzerland has been demonstrated by Mikkelsen *et al*, (1997). Two sites were studied. The first, which was adjacent to a road carrying 37,000 vehicles per day, showed the

accumulation of a 50 cm depth of runoff sludge containing high concentrations of metals, PAHs and halogenated organics over a 34 year period. The second site involved much lower vehicle densities (2,300 vehicles per day) and the depth of runoff sludge collecting in the 3 m deep soakaways was 20 cm. Similar observations were made by Pratt (1996) for two soakaways at Brandon, Suffolk, UK where high concentrations of metals and organic carbon were associated with the fine, organic material which accumulated in the first 40 cm of sediment in the base of the soakaways. Below 40 cm the pollutant levels approached those of expected background levels. Studies of a motorway soakaway on the M25 London orbital motorway showed total quantified PAHs in the basal soakaway sediments to be in excess of 100 mg/kg with an estimated annual loading of 0.20 – 0.33 kg per annum being delivered to the soakaway (Barker *et al.*, 1999). Below the base of the M25 soakaways, the pollution levels appeared to approach those of background levels. The significant build-up of metals, PAHs and halogenated organics in the sludge layer at the base of the soakaway is important in retaining pollutants by filtration and adsorption. However, once these sites become saturated with the pollutant species, they would be the potential for migration within the runoff water to the unsaturated zone and to the groundwater. In addition, soluble components such as pesticides and de-icing salts could pass straight through the infiltration system and reach the groundwater. Thus in the long term, without regular maintenance, the performances of such soakaways can progressively decline and basal accumulation of sedimented toxic sludges can occur.

Two soakaways in Valence, France serving similar catchment areas (road network and associated urban areas) but having been in existence for 2 years and 30 years, respectively have been compared in terms of the pollution impacts on the soil and groundwater (Barraud *et al.*, 1999). It was concluded that over the longer term (e.g. 30 years) metals and mineral oils could contaminate the soils to within distances of 1 m around the installations. Although the groundwater impact was perceived to be low, metal concentration increases suggested that retention is not complete when the depth of the unsaturated soil is around 40 cm.

Soakaways have also been widely used in the UK for treating runoff from the impervious surfaces and roofs of individual properties. They are also used for the disposal of highway drainage, particularly when connected to single or multiple gully pots. The installation of multiple soakaways (or soakaway fields) has also been used to increase the storage and infiltration capacity for highway runoff prior to overflow but the practice could lead to downstream build-ups of persistent pollutants.

4.2.2. *Infiltration Trenches*

Infiltration trenches are essentially a linear version of soakaways and operate in an identical way. Infiltration trenches are filled with stone or rubble and in comparison to soakaways require lower volumes of infiltration material to treat equivalent volumes of stormwater. However, the ability to maximise the infiltration surface area in these systems enables higher treatment efficiencies to be achieved. Very little data has been reported on the performance of such systems but silt infilling of a stone-filled infiltration trench has been encountered at the M42 Hopwood Park Motorway Service Area on the M42 motorway in Worcestershire, UK. Despite this, the trench has performed well in treating oil and other pollutants carried across a filter strip by the 10 mm first flush run-off generated from a heavy goods vehicle lorry park. Impressive pollutant reductions of 94% for total Zn, 82% for total Cu, 97% for total Pb, 98% for BOD and 99% for TSS have been monitored for this system which also incorporates a spillage interceptor basin (Bray, 2001). A trench containing sand modified

with an oxide coating, referred to as a partial exfiltration trench, was able to remove metals from highway runoff at efficiencies above 80% through adsorption-filtration mechanisms (Sansalone *et al*, 1998).

During an investigation of the ability of infiltration trenches to reduce peak flows within a Belgrade sub-catchment, Todorovic *et al* (1999) modelled the degree of clogging produced by suspended solids with differing bulk densities. Complete clogging was predicted in 109 years for solids with a bulk density of 2.8 but this decreased to 25 years for bulk densities as low as 0.8. The performances of two parallel infiltration trenches receiving discharges from roofs and paved surfaces in Copenhagen have been compared over a 3 year operational period (Warnaars *et al*, 1999). Over the observation period the soil infiltration rate appeared to decrease from 30 to 70% although this was less than the original difference in side wall infiltration between the two systems (a factor of 10). A full-scale model of a modified infiltration trench, known as an ecology ditch, has been tested for its ability to reduce the time to peak flow and the peak discharge under simulated storm conditions. The ditch, which was composed of compost, sand and gravel achieved a peak flow reduction of 10 to 50% for the larger storms (Barber *et al*, 2003).

4.2.3. Infiltration Basins

Infiltration basins have the ability to store runoff and subsequently to allow it to percolate through either the soil base (sandy loams, sands, sandy gravels with an infiltration rate exceeding 15 mm/hour) or a specially constructed under-drainage system composed of gravel or sand filter beds. To achieve total solids removal efficiencies of up to 90%, an overall filtration rate of 5 m³/ha/m² is needed. Schueler (1987) has studied a number of on-line and off-line infiltration basins in the USA and while supporting the removal efficiencies quoted in Table 4.8 raises concerns regarding their long term operational viability due observed failure rates of 50% within 5 years of installation. Factors such as lack of sediment pre-treatment, unsuitable soils, inadequate underdrainage and poor maintenance were cited as causes of failure and a current view is that infiltration basins are best used as final polishing systems.

Norrstrom and Jacks (1998) found elevated concentrations of Cd, Pb, Cu, Zn and PAH in soils beneath an infiltration pond and incremental annual accumulations of Zn, Cu and Cd (averaging between 0.8-1.5 mg/kg for Zn) were found in the basal sediments of an infiltration basin in Luton, UK receiving peak discharges of 2.4 m³/s from a 26 ha residential site (Ellis, 2000). Thus the long-term performance of such basins could prejudice groundwater quality unless fitted with an impermeable liner or compacted clay base. In addition, such systems can accumulate toxic sludge and will need regular cleaning and final off-site disposal of any contaminated sediment.

Dechesne *et al* (2002) have studied the performance of 4 French infiltration basins with respect to the contamination and clogging potential of the top soil. One of these basins (Centre Routier) has been receiving runoff from a truck parking lot since 1987 and has a calculated clogging indicator of 0.878 hours which compares with a value of 45 hours for a completely clogged system. The majority of metals (e.g. Cd, Cr, Cu, Ni, Pb, Hg and As) were found to concentrate in the surface soil layer and to reach sustainable levels at a 30 cm depth. The exception was Zn for which the mobility enabled concentrations to exceed those found in background soils at depths down to 110 cm. More recent studies on an infiltration basin in suburban Lyon have investigated metal, nutrient and hydrocarbon concentrations at different depths. Pollutany concentrations were found to decrease rapidly with depth whereas

pH, mineralization and grain size increased. After 14 years of operation, sustainable metal concentrations were reached at a 30 cm depth although this was deeper for hydrocarbons (Dechesne *et al*, 2004).

Studies of infiltration basins in France containing both stone and sand filter layers have indicated that clogging is due to a combination of relatively slow mechanical processes (surface deposit and pore clogging) and a superimposed variable and reversible clogging due to biological processes (Gautier *et al*, 1999). One of these infiltration basins receiving runoff from a 380 ha mixed industrial, agricultural and residential catchment is located at Venissieux and the geotextile and sand layer were renewed in 1990. Examination of soil cores below the infiltration basin showed that solids less than 2 mm were seriously contaminated by metals and mineral oils, although this decreased with depth until the base of the geotextile and 0.5 m into the subgrade the pollution was slight (Chocat *et al*, 1999). The pollutant removal efficiencies of the same infiltration basin were investigated between 1990 and 1995 and values of 25%, 48% and 59% determined for Zn, Cu and Pb respectively (Bardin *et al*, 2001). These values were very similar to those obtained for the settling basin located immediately before the infiltration basin and this was interpreted by the authors as being due to clogging of the permeable material in the infiltration basin so that settling became the primary removal process.

Where the naturally occurring soil is used as the filter medium, infiltration basins are referred to in New Zealand as soil adsorption basins. Watts *et al* (1999) have predicted that the use of such systems within a new development in Christchurch would produce pollutant reductions of >90% for suspended solids, total hydrocarbons, nutrients (except nitrate-N) and faecal coliforms at the influent concentrations expected. For the metals Pb, Zn and Cu the expected removal efficiencies were between 80 and 95%.

The International Stormwater Best Management Practices (BMP) Database reports the performances of two infiltration basins in Austin, Texas with regard to their pollutant removal capabilities. The Seton Pond facility incorporates a two layer filter involving an upper layer of washed sand (45 cm) which is separated by geotextile fabric from the under-drain medium of washed gravel (1.3-5.1 cm diameter). Monitored pollutant removal percentages were 89% for TSS, 66% for COD, 3% for nitrate, 51% for phosphorus, 81% for Zn and 75% for Fe although the poor hydraulic performance of the sand filter reduced the facility's capture capacity. The Barton Creek Square Shopping Centre Pond is equipped with three layers of filter media (fine sand, coarse sand and gravel) covering a surface area of 0.7 ha which represents about 85% of the pond floor area. Water quality data collected over a 2 year period indicate average pollutant removal efficiencies of 77% for TSS, 70% for COD, -28% for nitrate, 63% for phosphorus, 67% for dissolved Zn and 100% for dissolved Pb.

A BMP design in California for which performance data are frequently reported in the International Stormwater Best Management Practices (BMP) Database involves a sedimentation basin followed by a separate filtration basin fitted with a sand layer on top of a gravel layer. These systems provide the same pollutant removal processes as in an infiltration basin but in two separate stages. Based on monitored event mean concentrations at the inlets and outlets to four similar systems located in Carlsbad, Monrovia, Whittier and Norwalk, the following ranges of pollutant removal efficiencies have been reported for TSS (79-94%), total Cd (23-42%), Cr (4-70%), Cu (34-61%), Pb (70-94%), Ni (0-33%) and Zn (48-93%).

4.3 Storage Facilities

4.3.1 Lagoons

A study conducted on the M1 motorway north of Luton in SE England investigated the performance of a 28 m long by 2 m wide polythene lined lagoon for the treatment of runoff from a 3172 m² area of road surface. The high pollutant removal efficiencies recorded for this system are shown in Table 4.9 and were attributed to over-design of the system with contributing factors being water loss due to seepage.

Table 4-9. Mean Percentage Annual Removal Efficiencies for a UK Motorway Lagoon Treatment System.

Pollutant	Percentage Removal Efficiency
TSS	92
Pb _{tot}	90
Zn _{tot}	71
Zn _{diss}	25
COD	54
Oil	>70
PAH	>70

Perry and McIntyre (1986)

Soakage lagoons at the junction of the A1(M) and M25 motorways to the north of London have shown persistent surface oil and scum problems despite the installation of booms and gate valves, and recently some £60,000 (€90,000) was spent de-silting both lagoons.

4.3.2 Extended Detention Basins

A series of French motorway studies (Balades *et al*, 1985; Cathelain *et al*, 1981; Ruperd, 1987) have investigated the performance of dry detention basins and removal efficiencies in the ranges of 0-73%, 0-75%, 0-73%, 1-67%, 0-48% and 7-86% were reported for BOD, COD, TSS, total Pb, total Zn and oil, respectively. The removal efficiencies were generally at the lower end of these ranges and the reasons given for the poor performances were the re-entrainment of solids during high flows (2.5 - 3.1 m³) through the basins. The use of front-end oil separators, sediment chambers and shallow basal marshes were found to be beneficial to overall pollutant removal performance yielding solids removal efficiencies of at least 50%. Stahre and Urbonas (1990) quote long term efficiencies for extended detention basins having 48 hour detention times of 50 - 70% for TSS and hydrocarbons, 20 - 40% for BOD, 75 - 90% for Pb and 30 - 60% for Zn. Even 4 - 10 hours detention is alleged to offer up to 50 - 60% TSS removals. However, as most dry basins often have less than 2 hours detention times, the pollutant removal efficiencies are usually rather mediocre with TSS in the range of 15 - 20% and BOD/COD generally less than 10%.

4.3.3 Retention Basins

The United States EPA National Stormwater BMP Database contains data for a total of 33 retention ponds and carries nearly 15,000 water quality records for this BMP type. By comparison the UK/European database is relatively small and highly variable in terms of the

type and number of pollutants studied as well as in terms of the monitoring protocols employed.

The time-based trap efficiency of a 25,000 m³ balancing pond receiving discharges from a 60.7 ha residential development in NW London is given in Table 4.10 together with bench-top determination of settleability. It is clear that even a few hours detention can provide a base level treatment of stormwater runoff but extended detention times are required to remove nutrients and organic loadings. For a retention time of 12 to 15 hours, the reductions which can be expected include some 60% of total SS, hydrocarbons, total coliforms and Pb; 40% of BOD, phosphates and Cu; and 10 – 20% of other pollutants such as Cd, nitrate and ammoniacal nitrogen.

Table 4-10. Trap Efficiency of Retention Basins

Pollutant	Imhoff Settleability (24 hrs average %)	Balancing Pond 2 hr removal (%)	Balancing Pond 6 hr removal (%)	24hr Balancing Pond removal (average %)
TSS	68	34	82	46 - 84
BOD	32	13	48	29 - 53
Pb	62	30	66	46 - 78
Oil/Hydrocarbons	69	18	62	20 - 78
Total P	46	20	58	20 - 70
Total Coliforms	71	60	72	54 - 73

Hall *et al*, 1993.

Two wet retention basins within the Dunfermline DEX development receiving highway runoff are included in the Scottish SUDS database monitoring programme. Using a vortex flow outlet control, good peak flow reduction is achieved unlike a simple pipe outlet, which yields a much poorer performance. Visual observations imply a good sedimentation performance especially for coarse and sand size solids. However, problems of litter and sludge shoal accumulations have been encountered confirming the potential for inlet silting. The (retrospective) inclusion of marginal vegetation (to a maximum of 25% cover) and islands (to improve flow characteristics) within balancing ponds has been shown to markedly increase the priority pollutant treatment efficiency without prejudicing the overall peak storage capacity.

A five year (1997–2002) monitoring study of highway runoff in South East England included two sites having retention basins, one of which (the A34 Newbury-Bypass) was planted with marginal reeds. Accumulation of PAHs and heavy metals occurred in the outlet sediment of both ponds, sediment concentrations being twice those of recorded untreated inflow samples. The average percentage reduction in key pollutant groups recorded across the two basins (inlet to outlet) is given in Table 4.11 and Figures 4.1 to 4.3 show the final treatment concentration reductions achieved for BOD, TSS, individual PAH and heavy metals for 10 recorded storm events within the M40 Souldern Brook retention basin.

Table 4-11. Treatment Efficiency of Highway Retention Basins

Site	Nominal Retention Time (hours)	Pollutant	% Reduction
M40 Souldern Brook, Oxford	1.5	Metals	11
		PAH	99
		TSS	73
A34 Newbury	0.75	Metals	35
		PAH	50
		TSS	62

Moy, *et al* (2002).

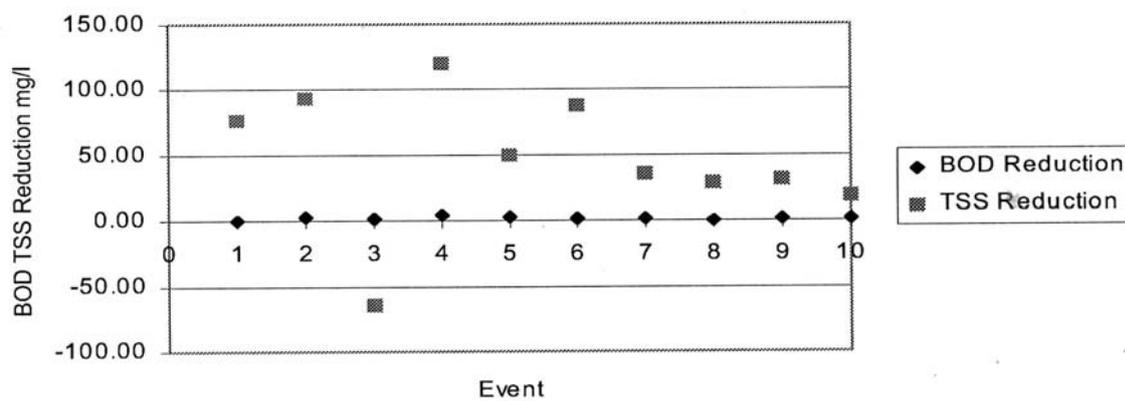


Figure 4.1. M40 Wet Retention Basin Outlet Concentrations for BOD and TSS

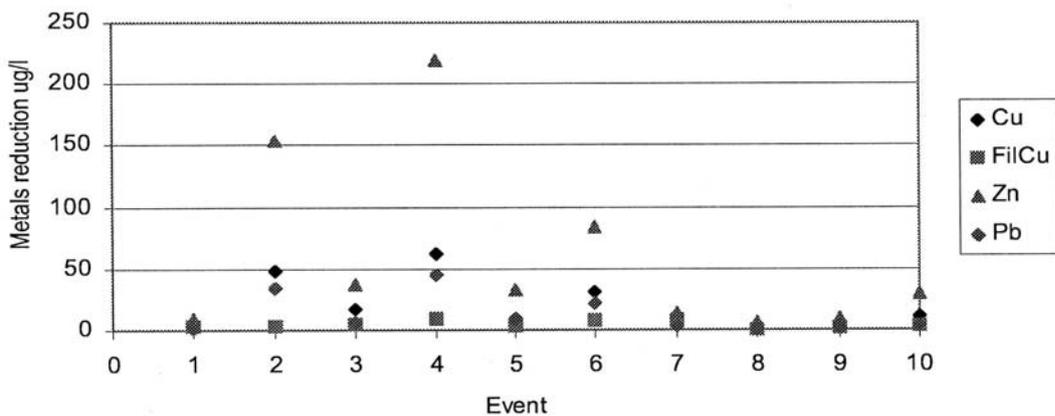


Figure 4.2. M40 Wet Retention Basin Outlet Concentrations for Heavy Metals

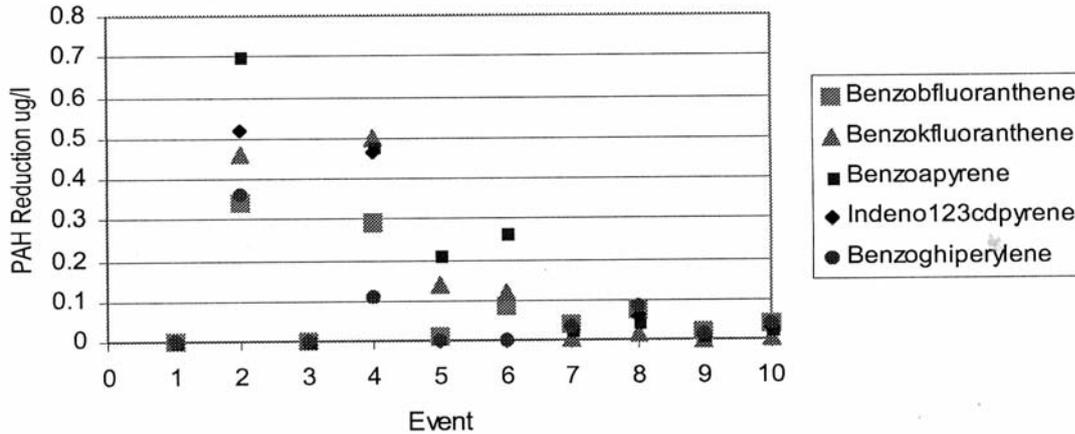


Figure 4.3. M40 Wet Retention Basin Outlet Concentrations for PAH

Examination of the data spread for Figure 4.1 shows that removal of BOD is near zero with variable TSS removal rates. Apart from Zn and Pb, metals also show only limited removal. However, some storms (e.g. events 2, 4 and 6) achieved good removal rates for nearly all pollutants, whilst others (e.g. events 7, 8 9, and 10) show only rather limited improvement in water quality. This storm dependency may be partly related to varying event and site characteristics (e.g. rainfall intensity, antecedent dry period, traffic intensity etc.), but poor removal efficiency appears primarily related to low inflow concentrations.

A more detailed view of pollutant removal efficiencies in the two wet retention basins is given in Table 4.12, which confirms the low nutrient and organic loading (represented by BOD/COD) improvements achieved by these flood storage ponds and the variable performance in respect of metal removal. However, it is clear that the retention facilities are providing a high removal efficiency for TSS, most herbicides and PAHs apart from naphthalene and phenanthrene.

Table 4.13 presents a synthesis of average concentrations and annual loads into five retention basins in France covering 312 individual storm events gathered as part of a national monitoring campaign during the 1982 –1987 period. The average annual runoff concentration to all basins and storm events noted for hydrocarbons was 5.3 mg/l and the average annual load was 17 kg/impervious hectare/year, of which between 82% – 99% was associated with the solid particulate phase. However, it should be noted that considerable intra-storm variation in concentration occurred, as noted by the maximum observed values shown in brackets in the table.

Table 4-12. Treatment Efficiencies for Highway Retention Basins

Pollutant	M40 Souldern Brook (% reduction)	A34 Newbury (% reduction)
BOD	10.24	28.72
COD	28.10	21.34
TSS	62.09	73.21
NH ₄ -N	-7.53	0.00
Cu	55.14	29.91
Zn	56.01	-11.67
Cd	4.63	53.71
Pb	69.87	5.84
Simazine	94.24	29.82
Glyphosate	60.38	ND
Atrazine	39.68	50.00
Naphthalene	-23.91	ND
Phenanthrene	22.15	ND
Fluoranthene	66.55	93.24
Pyrene	63.60	90.83
Benzo(a)anthracene	36.14	95.27
Benzo(b)fluoranthene	71.64	94.78
Benzo(k)fluoranthene	74.56	ND
Benzo(a)pyrene	94.45	95.60
Indeno(123cd)pyrene	91.20	94.15
Benzo(ghi)perylene	83.66	ND

ND = Not Detectable
Moy *et al* (2002).

Table 4-13 Performance of French Wet Retention Basins

Location	Average Annual Concentration (C; mg/l) & Load (L; kg/ha/yr)	TSS	COD	BOD	N _{Total}	Pb
Maurepas (78ha)	C	190 (890)	77 (590)	12	3.3	0.085 (0.44)
	L	940	380	55	16.0	0.41
Ulis Nord (91ha)	C	440 (2480)	190 (2720)	34	6.1	0.12 (0.73)
	L	1100	460	85	17	0.30
Aix ZUP (13ha)	C	300 (2010)	200 (1220)	38	5.4	0.16 (0.84)
	L	630	430	75	12	0.35
Velizy (78ha)	C	190 (760)	90 (500)	17	3.8	0.47 (1.4)
	L	400	190	36	8.0	1.0
Nice (06ha)	C	130 (550)	120 (400)	28		
	L	540	530			

Bertgue and Ruperd. (1994).

For all the experimental basins, the average storm removal for TSS and Pb varied between 65% to 90%, and removal ranges for other pollutants are given in Table 4.14. It is also interesting to note that in laboratory two hour settling (decantation) tests, the average COD removal efficiencies achieved were 55%, 44% and 48% for Maurepas, Aix ZUP and Velizy, respectively. These figures are much less than achieved in the field monitoring programme for the three urban catchments and illustrate the effectiveness of pollutant removal by adsorption in the presence of continually turbid flow conditions (Bertgue and Ruperd 1994).

Table 4-14. Reduction in Pollutants within French Retention Basins

	TSS	COD	BOD	N _{Total}	Hydrocarbons	Pb
% Removal Efficiency	80 - 90	75 - 90	75 - 90	40 - 70	35 - 90	65 - 80

Bertgue and Ruperd (1994).

Table 4.15 provides more detailed information on the overall pollutant removal efficiency of the Ulis Nord basin in Paris observed over the duration of the monitoring programme and Table 4.16 compares the removal efficiency of the Ulis Nord and Ulis Sud retention basins.

Table 4-15. Removal Efficiency of the Ulis Nord Retention Basin

Pollutant	Average Concentration in Influent Runoff (mg/l)	Average Concentration in Outflow from Basin (mg/l)	Average % Reduction in Pollutant	Removal Range	% Reduction following 2 hour retention
BOD	17	8	52.9%	54 % - 99%	76%
COD	134	32	76.1%	10% - 56%	
TSS	498	11	97.8%	60% - 97%	88%
TVS					84%
N _{Total}	3.4	3.3	2.9%	30% - 54%	
Detergents	0.57	0.23	56.9%		
P _{Total}	2.9	1.08	62.8%	12% - 39%	
Hydrocarbons	2.9	0.63	78.3%		
Pb	0.19	7.310 x 10 ⁻³	96.2%		65%
Cu					69%
Zn					77%
Cr					68%

Cottet (1980).

It should be noted that some storm events recorded negative removal efficiencies for nutrients and this may be related to either flushing and entrainment of adsorbed nutrients from sediments and macrophyte vegetation or to very low inflow concentrations which are close to the irreducible concentrations of the permanent retention pool. At such low relative concentrations, the reliability of the calculated efficiency is highly questionable.

Table 4-16. Removal Efficiency Comparison for Ullis Nord and Sud Basins

Pollutant	Ullis Sud (70 ha; 48% impermeable)			Ullis Nord (47ha; 42% impermeable)		
	Input Load (kg/imp.ha)	Output Load (kg/imp.ha)	Average % Removal	Input Load	Output Load	Average % Removal
BOD	829	107	87.1%	359	157	56.1%
COD	2598	521	79.9%	163	977	40.2%
TSS	3902	387	90.1%	2592	678	73.8%
Hydrocarbons	65	4	94.2%	50	5	90.1%
Total Nitrogen	285	168	35.6%	256	88	68.4%
Total Phosphorus	44	22	50.6%	34	8	75.8%
Pb	0.893	0.054	94.0%	0.824	0.061	92.6%
Zn	5.12	0.66	87.1%	2.37	0.77	67.7%
Cd	0.031	0.005	83.7%	0.016	0.006	62.5%

Adler (1993)

Table 4.17 illustrates the pollutant removal range of a further retention basin at Molette in the Seine St-Denis region of Paris, which emphasises the characteristic variability in performance recorded by typical retention basins. The improvement in water quality achieved in a number of in-series retention basins at Melun-Senart in the Marne-la-Vallee region of SE Paris is also illustrated in Table 4.18.

Table 4-17. Pollutant Removal Efficiency of the Molette Wet Retention Basin

	TSS	TVS	COD	BOD	Pb	Zn	Cd
% Removal Range	55.7 – 89.7	48.6– 89.5	45.7– 79.6	37.3– 75.9	69.1– 81.1	79 – 81.9	33 – 39.9

Dept. de Seine St Denis (1987)

Table 4-18. Comparison of Inlet and Outlet Concentrations for Retention Basins in Marne-La-Vallee

Sampling Location	BOD (mg/l)	COD (mg/l)	TSS (mg/l)
Inflowing Water	10 – 100	50 – 600	Variable
Outflowing Water	2 - 10	10 - 50	10 - 30
	N _{Total} (mg/l)	Cl ⁻ (mg/l)	P _{Total} (mg/l)
Inflowing Water	3 – 6		
Outflowing Water	1 - 4	10 - 60	0.05 – 1.0

SAUVETERRE. (1981)

Table 4.19 details the inflow-outflow concentrations for a small wet retention basin located upstream of two wetlands in Halmstad, Sweden. Little water quality improvement during passage through the basin is observed but this may be partly due to the rather low inflow concentrations, which are approaching the irreducible background level of the permanent wetland pool.

Research in Sweden on the performance of two retention basins in Gothenburg and Örebro concluded that the ponds generally provided an effective pollution control (Table 4.20) with removal efficiencies increasing with the specific storage capacity of the ponds up to approximately 250 m³ /impervious ha (equivalent to a depth of 2.5 m). However, the pond

sediments were found to be highly contaminated and required disposal via a controlled landfill site (this confirms the data given in Table 4.6). In addition, the outflow concentrations of some heavy metals exceeded Swedish guidelines.

Table 4-19. Pollutant Efficiency of a Small Retention Basin in Halmstad, Sweden.

	TSS	Cd	Cu	Zn	Pb
Inflow (mg/l)	9.7	0.06	4.06	29.93	1.91
Outflow (mg/l)	14.8	0.04	3.17	18.46	1.39

Svensson *et al* (2002).

Table 4-20. Pollutant Removal Efficiencies for Two Swedish Retention Basins

Pollutant	Jarnbrott Basin, Gothenburg Basin			Krubban Basin, Örebro		
	Inflow EMC (Range and average)	Outflow EMC (Range and average)	Removal (%)	Inflow EMC (Range and average)	Outflow EMC (Range and average)	Removal (%)
TSS (mg/l)	6.3-820 (55.2)	6.0-33 (16.6)	70	33-360 (153)	12-65 (25.0)	84
VSS (mg/l)	1.6-180 (16.0)	1.0-11 (6.47)	60	7.9-57 (26.9)	3.1-15 (6.54)	76
Zn (µg/l)	42-520 (118)	9.0-180 (82.8)	30	62-240 (135)	0-57 (24.1)	82
Cu (µg/l)	16-210 (53.3)	18-72 (37.4)	30	13-89 (33.5)	7.0-9.3 (8.52)	75
Pb (µg/l)	2.1-77 (13.1)	1.4-16 (6.76)	48	8.8-60 (26.4)	1.3-6.9 (4.69)	82
Cd (µg/l)	0.15-1.3 (0.55)	0.2-1.1 (0.483)	11	0.17-0.59 (0.4)	0.03-0.47 (0.05)	88
Nitrogen (mg/l)	0.63-5.3 (2.0)	0.82-3.6 (1.86)	7	0.52-1.4 (0.92)	0.50-0.77 (0.61)	33
PO4-P(µg/l)	20-560 (70.2)	16-89 (42.3)	40	24-180 (93.6)	9.0-5.0 (24.5)	74

Pettersson *et al* (1999).

It is interesting to compare the above retention performance studies with the data contained in the US EPA National Stormwater Database, which determines system performance using pollutant outlet concentrations in relation to receiving water standards. Retention basins not only achieve lower outlet concentrations for TSS than other BMPs (median value 8.5 mg/l; 1st quartile of 2 mg/l and 3rd quartile of 18 mg/l), but also on average remove 70% of total TSS in the incoming stormflows. According to the EPA, database retention basins are also particularly effective in removing faecal coliforms, reducing the 2400MPN/100ml median inflow level to 491.5 MPN/100ml, and also in damping the inflow variability.

4.3.4 Constructed wetlands

Wetlands (both natural and constructed) have been used for treatment of sewage and for urban, industrial and agricultural runoff within the UK and Europe but experience of their use for highway runoff has been relatively limited until recently. Sub-surface flow wetlands provide very limited flood storage and therefore are rarely used for stormwater control and treatment. Table 4.21 indicates the average range of pollutant removal efficiencies that have been reported in the literature for constructed wetlands receiving highway runoff in the UK, France, Canada and the United States.

Table 4-21. Percentage Pollutant Removal Rates in Constructed Wetlands

Wetland Type	TSS	Faecal Coliforms	N _{tot}	P _{tot}	Total Metals	Pb _{tot}	Zn _{tot}	BOD/TOC
Subsurface flows	85 (67-97)	88 (80-97)	44 (25-98)	50 (20-97)		83 (5-94)	42 (10-82)	-
Surface flows	73 (13-99)	92 (76-99)	33 (10-99)	43 (2-98)	(40-75)	69 (41-83)	58 (31-75)	15 (5-32)

Ellis *et al.* (2003).

A 3900 m² biofiltration (surface flow) wetland receiving runoff from the M25 London Orbital Motorway recorded consistent metal removal efficiencies of about 90% confirming the potential treatment effectiveness of such systems. Solids removal performance increased as flow TSS concentrations increase but with inflow concentrations below 30 - 40 mg/l, performance efficiencies were influenced by the attainment of irreducible background concentration levels. A 1050 m³ constructed wetland on the A34 Newbury Bypass in SE England draining 16,000 m² of road surface showed similar good TSS (90%) removal rates but metal removal efficiencies were highly variable particularly for Zn and Cu. This variability in performance, which can also be noted from Table 4.21, has variously been attributed to short-circuiting, short detention and contact times, pollutant speciation mechanisms and remobilisation, seasonal vegetation effects etc. Table 4.22 illustrates the pollutant removal efficiencies for two storm events recorded at a sub-surface wetland at the Newbury site, which shows the highest removal efficiencies to be for Pb and Cd. The negative efficiencies recorded for Cu appear to be anomalous with other studies and no easy explanation can be offered although the higher mass loads recorded for Pb and Zn in storm 2 may be due to the elevated chloride levels associated with this storm which occurred in winter in comparison to the short, intense summer event of storm 1.

Table 4-22. Pollutant Removal Efficiencies for Sub-surface Flow Wetland at Newbury, UK

Pollutant	Storm 1		Storm 2		% Removal Efficiency	
	Inlet Load	Outlet Load	Inlet Load	Outlet Load	Storm 1	Storm2
TSS	11.3 kg	2.83 kg	5.06 kg	3.02 kg	75.0	40.3
Cd	0.114 g	0.014 g	1.74 mg	0.10 mg	90.3	99.4
Cr	0.338 g	0.174 g	0.12 g	0.091 g	48.5	24.2
Cu	1.4 g	2.76 g	0.666 g	1.67 g	-97.1	-88.4
Ni	0.796 g	0.179 g	0.656 g	0.10 g	77.5	84.8
Pb	1.07 g	0.022 g	4.12 g	0.097 g	97.9	97.6
Zn	4.79 g	1.62 g	5.49 g	2.21 g	66.2	59.7

Shutes *et al* (2000).

A separate study investigating the treatment of runoff from the Newbury Bypass also showed good pollutant removal performances with 69% and 81% average TSS removals recorded for two separate surface wetland systems (Table 4.23). Considerable metal concentrations accumulated in the basal sediments of the wetland and were particularly associated with the finer sediment fractions < 63 µm in size.

Table 4-23. Pollutant Removal for a Surface Flow Wetland at Newbury, UK

	DO (mg/l)	TSS (mg/l)	COD (mg/l)	BOD (mg/l)
System C	4.2 – 12.6	14 – 73	1 – 108	– 11.2
System H	3.2 – 13.1	30 - 96	2 - 73	0.6 – 18.3

Pontier *et al* (2000).

Figure 4.4 shows the influent and effluent quality of both constructed wetlands and retention basins for total Cu as recorded in the US EPA/ASCE National Stormwater Database. The median (50%), 1st quartile (25% percentile) and 3rd quartile (75% percentile) values are shown for the box plots. Retention basins show a considerable pollutant reduction in comparison to constructed wetlands, and have a similar superior performance in respect of dissolved Cu.

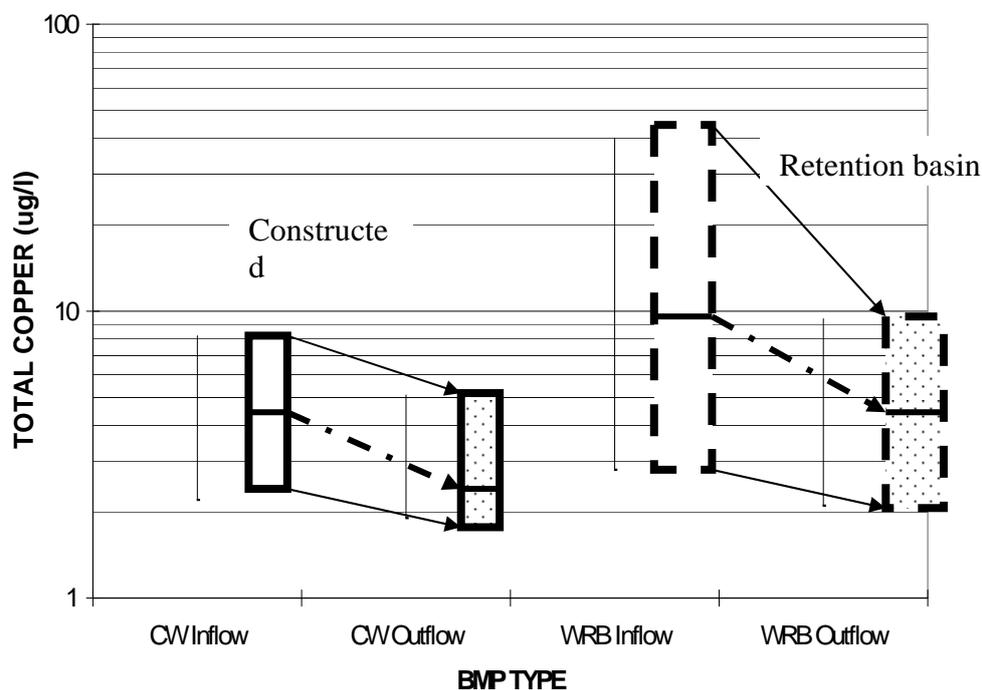


Figure 4.4. Box Plots for BMP Copper Removal

However, using the EPA approach of basing system performance on outlet concentration in relation to water quality standards, an evaluation of the database shows that constructed wetlands are by far the best BMP for removal of dissolved Zn species, with a median outlet concentration of 14 µg/l. In comparison, the median outlet concentrations for retention basins and swales and filter strips are 19.5 µg/l and 36 µg/l, respectively.

The following tables provide performance summaries for a range of UK stormwater constructed wetlands, which confirm the overall effectiveness of such biofiltration systems for the treatment of contaminated surface water runoff generated by various urban land uses.

4.3.4.1 Residential (and mixed) Catchment Runoff

Table 4-24. Details of stormwater wetland in Anton Crescent, Surrey

Design & discharge	System purpose & components	System specification	Site data
1:50 design; 165.5 m ³ /d.	Flood storage, 1.3 ha: Retention basin; SF wetland with pond at outlet; Typha	Maximum design storage capacity = 10,000 m ³ Wetland area; 13, 125 m ² Mean retention time; 10.8 days	Wildlife conservation area; local amenity; educational facility Average removal %: N -19%; SS 56%; Pb 7%; Zn 37%; E. Coli 78% High levels of metals reported in sediment (Cd 1.5;Cu 40.0;Pb 126.6;Zn 120.7 mg/kg) Plant tissue (leaf) metal levels also recorded

Cutbill (1994).

Table 4-25. Details of stormwater wetland in Great Notley Garden Village, Essex

Design & discharge	System purpose & components	System specification	Site data
1:100 design Maximum discharge; 272.4 l/s Catchment =188.18 ha	Flood storage; Retention basin; SF wetland with sediment trench, 16,000m ² pond. Typha, Phragmites, Iris , Scirpus	Maximum design storage capacity = 30,000 m ³ Wetland area = 7, 900 m ²	Final open water recreational pond. % removal range: Cd 10-99;Cu 94-97; Pb 89-97;Zn 10-99. Sediment metal levels: Cd 2±1.9; Cu 24±19.4; Pb 31±10.3; Zn 65±49.7 Plant tissue and rhizome metal levels also recorded

Mungur *et al* (1998).

Table 4-26. Details of stormwater wetland in Keytec 7 Business Park, Worcestershire

Design & discharge	System purpose & components	System specification	Site data
1:5 year design Catchment = 10.9 ha	Flood balancing, pollution control; Retention basin with SF marginal constructed wetland; <i>Typha</i>	Maximum design storage volume = 1500 m ³ . Retention time = 15-20 hours	Consents; SS 100 mg/l, BOD 20 mg/l and Oils/HCs 5 mg/l; all have been successfully met over past 5 years. One major oil spillage but reedbed recovered within a few months. Landscaped to provide amenity feature/wildlife habitat.

Tucker (1999).

Table 4-27. Details of stormwater wetland in DEX, Fife

Design & discharge	System purpose & components	Site data
1:100 design Catchment = 600 ha	Flood balancing, pollution control and amenity. Dry and Wet retention basins; SF constructed wetlands and marginal vegetation to wet basins	Landscaped wetland treatment area with swales, filter drainage for road/car parking runoff and flood storage basin % metal removals recorded in wetland: Cd 0; Cu 33; Pb 25;Zn 65. Mean metal sediment levels (mg/kg): Cd 0.09; Cu 13; Pb 10.5; Zn 30.2

McKissock *et al* (1999).

Table 4-28. Details of stormwater wetland in Brentwood, Essex

Design & discharge	System purpose & components	System specification	Site data
Catchment = 400 ha	Flood storage; SSF constructed wetland (<i>Phragmites</i>) and SF natural wetland (<i>Typha</i>)	Wetland area; 204 m ² Mean retention time; 50 mins	Front-end settlement basin. Elevated inflows of Pb (195 µg/l), Zn (132 µg/l) and high BOD (75 mg/l). Mean % removals: <i>Dry weather</i> ; Zn 19; Cd 33; Pb 30; Cu -15; BOD 17; SS 18; Total Amm 50. <i>Wet weather</i> ; Zn 22; Cd -4; Pb 16; Cu 8; BOD 15; SS 4; Total Amm 59. Plant tissue metal levels also recorded.

Revitt *et al* (1999).

4.3.4.2 Highway Catchment Runoff

Table 4-29. Details of highway runoff wetland on the A34, Newbury Bypass, Wiltshire

Site description & discharge	System purpose & components	System specification	Site data
A34 bypass. 13.5 km dual two lane trunk road; Discharges: 20-120 l/s for 1:50 and 1:25 design	Flood storage and balancing; 9 x wet detention basins with SF and SSF constructed wetlands. Front-end oil/silt bypass interceptors, grass filters and reedbeds <i>Phragmites/Typha</i> .	Maximum design storage volumes = 121-676 m ³ Retention times = 30-120 hours	% metal removals recorded: BOD -63-64; Cu -58-83; Cd -89-83; Zn -56-76. Wet weather % removals recorded: SS 40-75; Cd 90-99; Cu -88-97; Pb 98; Zn 59-66. Metal sediment levels (µg/g); Zn, 20-28; Cd, 3-7; Pb 17-18; Cu, 4-12. Metal levels in plant tissue also recorded

Shutes *et al* (2000).

Table 4-30. Details of highway runoff wetland on the M42, Hopwood Park Service Station

Site description & discharge	System purpose & components	System specification	Site data
Surface drainage for car/HGV parking and fuelling areas (9 ha) 5l/s/ha design runoff rate	Series of small wetland cells following either filtration trench or hydrodynamic separators. Typha/Phragmites with other plants	10mm first-flush capture. 48 hour "filtering" period	95 - 99% retention for heavy metals, TSS and BOD for HGV parking area. 79 - 83% retention for heavy metals for fuelling/car park areas with 89% BOD and 97% TSS. High nature conservation value.

Bray (2001).

4.3.4.3 Airport Runoff

Table 4-31. Details of airport runoff experimental wetland at Heathrow Airport, London

Site description & discharge	System purpose & components	System specification	Site data
245 ha 8.6 m ³ /day	Pollution control (de-icing removal); SF constructed wetland, SSF constructed wetland, Rafted lagoon (Typha/Phragmites)	Retention time: 16 hours	Elevated BOD inflows: 270 mg/. Glycol inflows: 1180-6326 mg/l. % Removals recorded: Glycol (SF system), 40-60%; (SSF system), 26-99%; BOD (SF system), 18%; (SSF system) 22%; <i>E. Coli</i> (SF system), 97%; (SSF system), 98%.

Revitt *et al* (1997).

4.3.4.4 Other Surface Water Runoff

Table 4-32. Details of urban runoff wetland at Mount Farm Lake, Milton Keynes

Site description	System purpose & components	System specification	Site data
Residential, highway and industrial catchment	Flood balancing; Marginal and submerged macrophyte vegetation – <i>Typha</i> , <i>Phragmites</i> , <i>Juncus</i> , <i>Nymphaea</i>	Surface area = 95 ha	Front-end oil interceptor. Measured inflow concentrations (mg/l): Cl 300-3000; PO ₄ 0.07-0.12; Zn 0.01-0.06; Pb 0.03-0.02. <i>E. Coli</i> ; >10,000MPN/100ml. Fish kills recorded

Milton Keynes Development Corporation (1992).

Table 4-33. Details of urban runoff wetland in Dagenham, Essex

Site description	System purpose & components	System specification	Site data
Residential, industrial and commercial. 440 ha	Flood storage and pollution control; 3 x 250m SF linear constructed wetland cells, <i>Typha</i> and <i>Phragmites</i>	Wetland area: 1750 m ²	Front-end sedimentation basin. Elevated inlet concentrations: Pb 285 µg/l; Zn 550 µg/l, BOD 69.4 mg/l and THMs. Wetlands targeted for 50% removals. Mean % removals: <i>Dry Weather</i> ; Zn 31; Cd 48; Pb 37; Cu 23; BOD -23; SS 35; Total Amm 6; TMHs 32. <i>Wet weather</i> ; Zn 71; Cd 72; Pb 69; Cu 7; BOD 24; SS -16; Total Amm 38. Plant tissue metal levels also recorded

Revitt *et al* (1999).

4.4 Alternative road surfacing

The design of porous paving includes continuous surfaces (such as porous macadam/asphalt), porous or solid blocks separated by open joints or castellated "grasscrete" blocks with their central voids filled with soil or gravel.

4.4.1 Porous Block Paving

Work in the 1980s on a concrete block-surfaced car park at Nottingham Trent University, in north central England, using differing types of sub-base materials demonstrated 34% - 47% reductions in total runoff discharges with initial wetting loss before drain discharge being 2.4 – 3.2 mm. Considerable improvements were also achieved in water quality with the porous paving limiting TSS discharges from near zero to 50mg/l maximum with hydrocarbons only being detected at trace levels. The work showed that pollutant retention occurred in the 50 mm layer immediately above the geotextile liner on which the porous blocks were placed. Observations on a similar concrete block surfacing at Shire Hall, Reading in SE England indicated a mean infiltration rate of 2600 mm/hour, six years after installation without any maintenance (Pratt, 1995). The surfacing allows the immediate infiltration of rainfall-runoff into the construction with the sub-base providing storage, treatment and pathways for downward percolation into the underlying soil or to perforated underdrains. The presence of dirt and oil spillage on the paved surface significantly reduced the infiltration rates for both the blocks and the gaps between them.

A 1500 m² porous paved car park installed at the Scottish Civil Aviation Authority HQ in Edinburgh, Scotland, whilst costing out some 15% more expensive than conventional "blacktop" asphalt, has shown significant attenuation (22% flow reduction) of the outlet hydrograph with first discharge only occurring several hours after the start of rainfall. In addition, TSS, COD and BOD outflow values were consistently below 20, 10 and 2 mg/l respectively with hydrocarbons below detection levels. Studies conducted on permeable pavements at the M40 Wheatley, Oxford service station and at the Wokingham Tesco car park site, demonstrated average peak storm intensity reductions of 88% and 81% at the outlet respectively, with attenuation extended on average by a factor of 14 (Abbott *et al.* 2003). Newman *et al.*, (1998) have also shown that properly installed permeable paving with the gravel bedding reinforced with lightweight clay aggregates or porous concrete granules, can provide considerable oil and water retention. Experimental block paving surfaces have given infiltration rates of some 4500 mm/hour with overall oil retention capacities of 9542 g/m², which would give an operational life of over 40 years and an outflow effluent from the pavement of about 0.6 - 0.8 mg/l. The use of pre-fabricated plastic/polypropylene geocellular units under the sub-base bedding layer has been shown to optimise oil and sediment removal capturing some 20 – 40% of the oil applied to the paved surface (Newman *et al.*, 2003).

A porous concrete block-surface car park located at the Bank of Scotland, South Gyle, Edinburgh showed outflow discharge to be under 50% of total rainfall-runoff with initial wetting loss being 1.65 mm. Both heavy metal (<0.068 – 1.7µg/l) and hydrocarbon (<3.5mg/l) concentrations in the porous paving effluent were markedly reduced over raw surface runoff concentrations. A 6250 m² porous concrete-block surfaced car park at the M40 Wheatley Motorway Service Area near Oxford has provided similar observations to those in Scotland with total outflows from the porous pavement being only 4% – 47% of the total rainfall volume, with lag times of up to 2 – 3 days. Block infiltration was assessed as varying between 250 – 14,000 mm/hour, although after 10 months of operation this had reduced to zero. Infiltration through the gaps was however maintained at rates of 11,000 – 229,000 mm/hour.

Grasscrete type modular pavement is well suited to overflow car parks, which require a grass surface that must be sufficiently hard wearing to withstand regular vehicle use. Such grass-

concrete surfaces possess infiltration rates between 0.2 to 1.0 mm/s; well in excess of most design storm rainfall intensities and can hold up to 5 mm rainfall within one hour for subsequent evaporative loss. Filtration-sedimentation and adsorption processes within the structural reservoir of the surfacing material can limit TSS outflows from near zero to 50 mg/l and typically remove between 40 - 60% bacteria, 70 - 90% heavy metals and hydrocarbons.

4.4.2 Porous Surfacing and Reservoir Structures

Studies on porous highway surfaces in Nantes, NW France, showed that the reservoir structure only discharged on average 3.3% of total rainfall received over a 4-year monitoring period. Similar studies of 40 storm events at experimental sites in Reze, France showed that concentrations of SS, Pb, Zn and Cd were reduced by 61%, 81%, 67% and 62%, respectively, by the highway sub-structure illustrating the efficient performance of the infiltration system (Table 4.33). On average, 96.7% of the stormwater volume infiltrated into the soil below the reservoir structure.

The percentage improvement in performance achieved by the reservoir structure over a conventional asphalt surface is also given in Table 4.34, which is based on the pollutant loading rates for the two sites, and the performance efficiency of the porous reservoir structure is clearly demonstrated.

Table 4-34. Comparison of Runoff Quality from a Highway Reservoir Structure with that of a Conventional Separately Piped Catchment Surface

	pH	COD (mg/l)	TSS (mg/l)	HC (mg/l)	Pb (µg/l)	Cu (µg/l)	Cd (µg/l)	Zn (µg/l)
Reservoir structure								
Number of samples	28	31	31	8	31	32	31	28
Mean	7.5	<22	13	<0.02	3.7	8.7	0.5	52
Range	6.1-8.7	<20-36	0.6-139	-	0.6-33	3.5-22	0.04-3.5	11-340
Std. Deviation	0.7	5	27	-	6.2	4.5	0.8	69
Reference catchment								
Number of samples	16	18	19	3	19	19	19	19
Mean	7.2	<23	33	<0.02	20	9.6	1.3	158
Range	6.2-8.1	<20-43	5-86	-	6-63	4.5-44	0.3-3.2	110-229
Std. Deviation	0.6	6	25	-	15	8.7	0.8	34
Mean Difference (%) (Based on loading, kg/ha of contributing area)	59%	-	-	-	84%	-	77%	73%

Colandini (1997).

Balades *et al* (1992) observed pollution load reductions of 50%, 93% and 89% for suspended solids, Pb and COD respectively, for a 56 cm thick reservoir structure in comparison to a conventional carriageway pavement. Ranchet *et al* (1993) in their comparison of the performance of a conventional pavement with a 16 cm thick porous concrete surface, reported reductions of 70% for suspended solids, 78% for Pb and 54% for COD. A study undertaken in Sweden, which investigated the effect of snowmelt on a porous paving surface, reported a reduction of 95% for suspended solids and between 40% - 50% for Pb (Hogland *et al.*, 1987).

Table 4.35 summarises the results for several experimental studies undertaken in France on various types of reservoir structures and inspection of the data shows that infiltration of stormwater runoff through such structures improves water quality regardless of surface permeability. The presence of porous surfacing material to the sub-grade reservoir only serves to enhance the water quality improvement. Infiltration into the porous asphalt over a period of more than 8 years did not cause any reduction in overall performance or result in any migration of particulate-bound metals within the reservoir.

Table 4-35. Comparison of the Pollution Reduction (%) Performance of Different Types of Reservoir Structures Receiving Stormwater Runoff in France

	Characteristics	TSS	BOD	COD	Pb	Zn
Reze: Classerie Street	Porous asphalt + porous bituminous-bound, graded aggregate.	61	-	-	81	67
Bordeaux: Le Caillou Park Verneuil ZAC: Zone I	Porous asphalt + porous concrete	36	-	79	86	
	Porous asphalt + porous bituminous-bound, graded aggregate.	81	45	63	76	35
	Zone II Conventional asphalt + porous bituminous-bound graded aggregate.	68	39	48	77	45
Zone III	Different reservoir structures	1	7	14	50	16

Agence de L'Eau Seine-Normandie, Suez Lyonnaise des Eaux, LCPC, SAFEGE, LROP (1999).

Table 4.36 illustrates the typical pollutant reduction with depth noted in the various French studies with metals retained in the top 50 mm of the soil layers. The data reflect a substantial exponential pollutant decay with depth with removal being effected within 50 cm of the surface. This performance pattern is similar to that noted in the UK soakaway and infiltration basin studies (Sections 4.2.1 and 4.2.3)

Table 4-36. Effects of Porous Surfacing and Reservoir Structure on Reduction of Metals with Depth.

Depth (cm)	Reference Material	Pb (mg/kg)	Cu (mg/kg)	Cd (mg/kg)	Zn (mg/kg)
Surface	Surface material collected by sweeping.	179	18	0.52	87
Porous Asphalt	Material collected from pores.	178	40	1.5	165
30-45	Cobblestone reservoir sub-grade	18	8	0.1	28
45-50	Sand layer	189	47	0.83	367
45-50	Drain at base of cobblestones	672	147	3.74	1780
50-52	Sand layer	29	6	0.13	67
52-60	Geotextile	8	3	0.04	12
160-175	Sub-soil	11	4	0.05	17
175-190	Sub-soil	8	3	0.03	13
190-205	Sub-soil	8	5	0.04	14
205-220	Sub-soil	8	4	0.04	14
Reference Soil		4 - 12	3	0.01 - 0.03	8 - 16

Legret *et al* (2004).

5 Conclusions

5.1 BMP pollutant removal processes and efficiencies.

The characteristics of the different structural BMPs with regard to their most significant pollutant removal processes are summarised in Table 5.1 together with the ranges of the predominant pollutant removal efficiencies reported in the literature. There is limited data available for PAHs, herbicides and miscellaneous organic compounds (although oils have been included in this category). For the pollutants defined as basic water quality parameters, BOD/COD and TSS are generally efficiently removed whereas nitrate is more resistant and only constructed wetlands and porous paving show removal efficiencies above 25%. The performances of individual metals are not identified in Table 5.1 but, where possible, the behaviours of total metals and dissolved metals have been described. It is clear that the dissolved metal fractions are more difficult to remove by the processes normally found in structural BMPs. Table 5.1 also contains brief comments on the pollutant removal characteristics of individual BMPs which may assist end-users in their selection of a BMP for a specific application. Fuller details of the behaviours of specific BMPs can be found in Sections 3 and 4 of this report.

5.2 Factors controlling pollutant removal capabilities.

The general water quality parameters (BOD, COD, TSS, nitrate, phosphate, bacteria) are subject to different controlling influences on their removal from the water column due to their differing physico-chemical characteristics. Microbial degradation is an important process for the removal of BOD, and to a lesser extent COD, in aerated water systems. Sedimentation and filtration are clearly prime removal processes for TSS and bacteria, of which faecal coliforms and pathogens are water pollutants of concern. Nitrates and phosphates are both taken up and utilised by plants but the higher solubility and lower adsorption tendency for nitrates makes them more difficult to remove by physical processes.

Within the aquatic environment, metal loadings in sediments exceed those in plants due to the higher rates of accumulation by physico-chemical compared to biological processes. For the metals discussed in this report (Cd, Cr, Cu, Ni, Pb, Pt and Zn), Pb demonstrates a highly efficient behaviour with respect to adsorption, sedimentation and filtration compared to Zn which is least responsive to transfer from the water to the sediment phases by these processes.

The PAH grouping includes three compounds which are representative of the low, medium and high molecular weight classifications. This parameter influences the preferred removal processes with the higher molecular weight PAHs being most susceptible to adsorption in contrast to the lower molecular weight PAHs which are preferentially removed by biodegradation. The low water solubilities of PAHs and the significance of volatilisation and photolysis also influence the overall removal.

The herbicides described in this report (pendimethalin, phenmedipham, glyphosate and terbutylazine) are broad-spectrum weed control agents for which the characteristic properties are derived from different active functional groups. Glyphosate is similar to the other herbicides in possessing a high adsorption potential (which supports sedimentation) but it also has a relatively high solubility (which inhibits removal by filtration). All the selected herbicides demonstrate efficient bioaccumulation. Microbial degradation and photolysis can

Table 5-1 Summary of BMP removal processes and the potentials for the removal of different pollutants.

BMP type	Primary removal processes	Pollutant removal characteristics and efficiencies					Comments on the application of this type of BMP
		Basic water quality parameters	Metals	PAHs	Herbicides	Miscellaneous organics	
Filter Strip	Adsorption; precipitation; plant uptake.	BOD/COD; **** TSS; *** Bacteria; ** Nitrate; * Phosphate; **	Total metals; **** Dissolved metals; *	No data	No data	Oils; ***	Optimum removal rates achieved with 18-25 m slope lengths and influenced by density and height of the grass sward.
Swales	Adsorption; precipitation; filtration; volatilisation; plant uptake.	COD; *** TSS; *** Bacteria; * Nitrate; *	Total metals; *** Dissolved metals; *	No data	No data	Oils; **	Particle sizes >25 µm are most efficiently removed; exponential decrease in pollutant removal observed with increasing swale length.
Soakaways	Adsorption; precipitation; microbial degradation; filtration.	BOD/COD; *** TSS; *** Nitrate; *	Total metals; *** Dissolved metals; **	No data	No data	Oils; ***	Typically, pollutant concentrations peak in the 40-50 cm soil layer below the soakaway but concerns for the groundwater exist for very soluble pollutants and when pollutant saturated soil conditions are reached.
Infiltration Trench	Adsorption; precipitation; microbial degradation; filtration.	COD; *** BOD; **** TSS; **** Nitrate; *	Total metals; **** Dissolved metals; **	No data	No data	Oils; ***	Little pollutant removal data exists and the high quoted values are mainly based on one system; clogging can be a problem for runoff containing oils.
Infiltration Basin	Adsorption; sedimentation; precipitation; microbial degradation; filtration; volatilisation.	BOD/COD; *** TSS; **** Nitrate; * Phosphate; **	Total metals; *** Dissolved metals; **	No data	No data	Oils; ***	Permeable media used for infiltration include gravel, sandy gravels, sands, sandy loams or naturally occurring soils; most pollutants retained within a 50 cm depth below the basin; accumulated toxic sludge may have to be removed.
Filter Drain	Adsorption; microbial degradation; filtration.	COD; ** TSS; ****	Total metals; **** Dissolved metals; ***	PAH; ***	No data	Oils; ***	Performances found to vary with respect to storm characteristics and seasonal factors.
Lagoons	Sedimentation.	COD; ***	Total metals;	PAH; ***	No data	Oils; ***	The relatively high removal efficiencies reported

		TSS; ***	**** Dissolved metals; *				here are for one system which was reported to have been over-designed.
Retention Ponds	Sedimentation; microbial degradation; volatilisation.	BOD/COD; *** TSS; *** Bacteria; *** Nitrate; * Phosphate; **	Total metals; ***	PAH; ***	Herbicides; ***	Oils; ***	Wet ponds for which retention times of at least 24 hours are desirable; removal efficiencies often relate to storage capacities; accumulation of pollutants may produce highly contaminated sediments.
Detention Basins	Adsorption; sedimentation; volatilisation.	BOD/COD; ** TSS; **	Total metals; **	No data	No data	Oils; **	Dry basins designed to store stormwater for a limited time after rainfall; re-entrainment of solids possible during high flows; the use of front-end treatment systems (eg.oil separators;sediment chambers) beneficial to overall pollutant removal.
Extended Detention Basins	Adsorption; sedimentation; microbial degradation; volatilisation.	BOD; ** TSS; ***	Total metals; ***	No data	No data	Oils; **	Stormwater may be retained for up to 48 hours whereas it can be as low as 2 hours for normal detention basins.
Constructed Wetlands	Adsorption; sedimentation; microbial degradation; filtration; volatilisation; plant uptake.	BOD/COD; ** TSS; *** Bacteria; **** Nitrate; ** Phosphate; **	Total metals; ***	No data	No data	No data	Sub-surface flow systems provide limited storage but higher potentials for pollutant removal; surface flow systems have the opposite characteristics; generally less variability in pollutant removal efficiencies than retention ponds.
Porous Paving	Adsorption; microbial degradation; filtration.	TSS; *** Nitrate; **	Total metals; ***	No data	No data	Oils; ***	Installations with reservoir structures are typically capable of 50% reductions in runoff volumes (can be as high as 95%); reservoir provides potential for microbial degradation when sufficient nutrients present.
Porous Asphalt	Filtration.	COD/BOD; ** TSS; ***	Total metals; ***	No data	No data	No data	Operation for up to 10 years reported with no maintenance although subject to clogging; regeneration of the surface can be achieved by high pressure spraying and/or suction.

* Low potential for pollutant removal (typically less than 25%)

*** Medium-high potential for pollutant removal (typically between 50% and 75%)

** Medium-low potential for pollutant removal (typically between 25% and 50%)

**** High potential for pollutant removal (typically between 75% and 100%)

participate in the removal of all four herbicides from the aquatic environment giving rise to metabolites, which may be of environmental concern before further degradation.

Five miscellaneous organic compounds (nonylphenoethoxylate (NPEO), di-(2-ethylhexyl)-phthalate (DEHP), pentachlorophenol (PCP), polychlorinated biphenyl (PCB) and methyl tert-butyl ether (MTBE)) have been identified as stormwater priority pollutants as a consequence of their industrial usage. Microbial degradation by both aerobic and anaerobic pathways occurs with differing degrees of efficiency except for MTBE which is resistant to biodegradation. Removal by sedimentation and filtration is most prevalent for DEHP and PCBs due to their low water solubilities and high K_d values. The high solubility of MTBE limits removal by these processes but it is the most volatile. PCP and NPEO are susceptible to photolytic reactions for which the reaction rates are elevated in surface waters.

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