Maximising the Ecological Benefits of Sustainable Drainage Schemes

Report SR 625
December 2003

HR Wallingford

dti
Department of Trade and Industry

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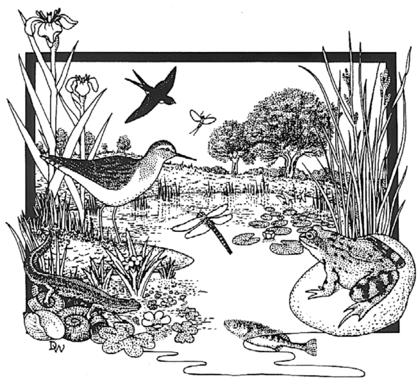

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Contract - Research

This report describes work funded by the Department of Industry, Partners in Innovation Scheme, under Research Contract39/5/137 cc2143, for which the DTI nominated officer was Richard Mellish, and the HR Wallingford nominated officer was Bridget Woods-Ballard.

Contributions to the project were also received from the project steering group:

- Babtie Group
- Binnie Black & Veatch
- Buchanan Consulting Engineers
- CIRIA
- English Partnerships
- Environment Agency
- Formpave Limited
- M C O'Sullivan & Co Ltd
- Marshalls Mono Ltd
- Montgomery Watson Harza Ltd
- Mott MacDonald Ltd
- Ponds Conservation Trust
- Robert Bray Associates
- SEPA
- Severn Trent Water
- T A Millard
- University of Bradford
- University of Abertay
- Welcome Break Group Ltd
- Westbury Homes
- White Young Green

DTI's management consultants for the project were WS Atkins Ltd., whose project officer was Mr J Leat. The report author was Jeremy Biggs of the Ponds Conservation Trust Policy & Research, and the work was undertaken as a sub-contract to HR Wallingford, who acted as project managers and report editors.

The HR job number was MCS 0430. The report is published on behalf of the DTI, but any opinions expressed in the report are not necessarily those of the funding department.

Contract continued

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Summary

Maximising the Ecological Benefits of Sustainable Drainage Schemes

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This report reviews and provides guidance on the ecological benefits of sustainable drainage systems (SUDS) compared to conventional systems. The report also suggests designs and maintenance regimes that will maximise the ecological performance of a range of SUDS components.

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1. INTRODUCTION

1.1 Scope and aims of the report

Sustainable Urban Drainage Schemes (SUDS) are increasingly being seen as a solution to environmental problems associated with the impacts of urban runoff. Particularly important in this context is the ability of SUDS schemes to protect downstream waters and provide additional still water habitats. However, although SUDS are increasingly common, data available describing their ecological performance is relatively limited, and often widely dispersed.

The objective of the present project was, therefore, to review the available ecological information on SUDS schemes, with particular emphasis on systems in the UK but drawing on other evidence as appropriate. These data were then used to provide guidance on the ecological benefits of sustainable drainage systems compared to conventional systems. The report also suggests designs that will maximise the ecological performance of a range of SUDS components.

Specifically the project aimed to:

1. *Review the literature on the ecology of sustainable drainage systems with particular reference to:*

    (a) their role in protecting downstream habitats and in providing new still water habitats (NOTE: it was not an objective of the project to review the extensive literature describing the precise mechanisms of pollutant removal by physical, chemical or biological processes in treatment systems)

    (b) landscaping practices and component designs to maximise the ecological benefits of SUDS schemes (in terms of downstream habitat protection and still water habitat provision)

    (c) the impact of conventional urban drainage on aquatic ecosystems.

2. *Collate and review ecological data on SUDS schemes in the UK gathered by the Environment Agency, SEPA and others in current or recently completed projects. This work focussed on two main areas:*

    (a) the ecological benefits of the SUDS schemes themselves (principally the pond and wetland systems created)

    (b) the mitigation of impacts on receiving waters.

3. *Use the above data to:*

    (a) describe the overall ecological benefits of SUDS schemes

    (b) make recommendations for further research on the ecological benefits of SUDS schemes (both for protection of receiving waters and for habitat benefits)

    (c) provide guidance on best practice in the design of SUDS schemes to maximise their ecological potential, both in terms of protection of downstream watercourses and in provision of new aquatic habitats.

1.2 The Water Framework Directive

The Water Framework Directive will play an increasing role in the protection of freshwater ecosystems over the next 15 years. Despite its wide range of potential influences, however, the precise effects of the WFD are still difficult to be certain about with a considerable amount of technical work currently in progress to implement the Directive.

In theory, the WFD is a major driver for SUDS. As noted in the recent Environment Agency consultation on a 'Framework for Sustainable Drainage Systems (SUDS) in England and Wales':

> *"The Water Framework Directive will require us to manage water resources sustainably. Sustainable drainage systems have a part to play in an integrated approach to water management. Rather than wait for the Directive to come fully into force we should act now to improve the management of water in the urban setting. It is time to look at water in the built environment in a different light: rather than see it as a threat we should take the opportunity to protect it more carefully. After all, there are many benefits, for us as individuals, for society as a whole, for industry and for wildlife"* (Environment Agency, 2003).

However the text of the WFD makes no specific mention of SUDS schemes or precise measures to control urban runoff. This suggests that the exact impact of the WFD will eventually be determined by technical discussions and lobbying from stakeholder groups including the construction industry, environmental groups and water users.

The role of the Environment Agency and SEPA in promoting SUDS schemes will obviously be critical in determining the extent to which SUDS are implemented. Careful reading of the consultation draft of the SUDS framework will probably enable users to judge the likely impact of the WFD. It is perhaps worth noting that the opening sentence of the document suggests that a relatively cautious approach to SUDS is likely, with the document noting:

> *"The need for sustainable drainage is not disputed, but SUDS if not properly designed and maintained, can lead to a number of problems"* (Line 1, Section 1.0, of the Framework for Sustainable Drainage Systems (SUDS), Environment Agency (2003)).

2. THE ECOLOGICAL IMPACTS OF CONVENTIONAL DRAINAGE SYSTEMS

The overall ecological impact of urbanisation on aquatic environments is multifaceted, with a range of physical, chemical and biological impacts. These are caused by activities which include: stream and river channel engineering, in-filling of ponds and wetlands, modifications to catchment hydrology and infiltration rates, increased pollutant levels, and the deliberate or accidental release of alien aquatic plants and animals (Paul and Meyer 2001, Arnold and Gibbons 1996, Booth and Jackson 1997, Line *et al.* 2002).

2.1 Hydrological impacts

The hydrological impacts of urbanisation on streams are well-known. As catchment imperviousness increases, as a result of the construction of roads, buildings and car parks, runoff increases, groundwater recharge decreases, floods peak more quickly (due to reduced lag time) and flood peaks are greater than in comparable non-urban catchments. In addition, urbanisation can also reduce unit water yield with more water leaving the catchment as runoff, although the effects may be partly ameliorated by leakage from potable and waste water pipes and treatment.

Urbanisation often also leads to increases in dry weather baseflow as a result of discharges from sewage treatment works. In the River Thames in London, for example, 40% of total river flow comes from waste water treatment plants in most summers but as much as 73% during droughts (DEFRA, 1999).

2.2 Physical impacts

Physical impacts in urban areas are a complex combination of engineering impacts (mainly caused by the shift from natural drainage paths with a high proportion of lateral and base flows, to piped networks draining paved surfaces to point source discharges), and ‘natural’ geomorphological adjustments to stream hydrology. Impacts on physical structure have been best studied in rivers and streams but urbanisation also affects the physical structure of standing waters, as discussed in the following paragraphs.

‘Natural’ geomorphological adjustment in newly urbanised streams and rivers usually begins with a phase of sedimentation, which is then followed by an erosional phase. This in turn is often followed by channel widening as channels begin to migrate laterally. Urban streams may also differ in other geomorphological features: for example, the distance between riffles may become greater, reducing the area available of shallow, fast-flowing well-oxygenated water available. Changes in sediment supply may also lead to changes in channel pattern with, for example, meandering channels becoming either braided or straighter (depending on local conditions) with increasing sediment supply. Sediment composition also changes in urban streams: less fine sediment, increased coarse sand fractions, and reduced gravel fractions have been observed, and these are likely to cause changes to the composition of the biota. There is also a general decrease in woody debris, particularly where urbanisation takes place on previously forested land. Urban channels have increased erosion around structures and may develop knickpoints, readily erodable points, which migrate upstream creating bank instability (Paul and Meyer 2001).

Changes in hydrology also affect velocity profiles in-channel and hyporheic/parafluvial dynamics, both of which are likely to affect biota distribution and ecosystem process, including carbon processing and nutrient cycling.

In some cases, temperature may increase in urban streams, although relatively little information is available about this important process. However, reduced cover from trees and woody vegetation, reduced groundwater discharge and the heat island effect of urban areas may all contribute to increased stream temperatures and therefore reduced dissolved oxygen concentrations (warmer oxygen dissolves less in warm water). This will be generally detrimental to the, predominantly riverine, aquatic animals which have a requirement for high levels of dissolved oxygen.

With increasing urbanisation, streams and rivers are also more likely to be embanked or over-deepened, both of which serve to detach the channel from floodplain and adjacent riparian habitats. In addition to physical alteration of channel structure the natural above-ground drainage network is normally greatly reduced in density as a result of urbanisation as channels are culverted, filled-in or paved over.

Standing water habitats are also much affected by urbanisation. Typically, a high proportion of small waterbodies are infilled during the urbanisation process. In the UK, for example, studies show urbanisation to have removed as many as 75% of pre-existing ponds in some areas. Remaining ponds and lakes often have their marginal habitats embanked and replaced by reinforced banks. Changes to runoff patterns may also alter the hydrology of still waters, either lowering water levels in groundwater-fed waterbodies or reducing water supply in surface water-fed waterbodies.

2.3 Chemical impacts

Chemically, urban runoff affects waterbodies through the uncontrolled release of a very wide range of pollutants. Most consideration is usually given to the effects of runoff on streams and rivers. However, still waters are also affected, particularly because of the common practice amongst highway engineers to route road runoff, often untreated, into existing ponds. Still waters can be particularly badly affected by this process because they act as sinks, building up an increasing pollutant burden in their accumulating sediments (Williams *et al.* 1998).

Untreated runoff from urban areas typically contains elevated concentrations of:

Pollutant	**Source**
Sediments	Sediments are primarily derived from eroding soils, erosion of built structures and breakdown of litter and other wastes.
Nutrients (Sonoda *et al.* 2001)	The origin of most nutrients is mainly fertilisers applied to gardens and amenity areas, eroded soils and waste organic matter from streets and gardens.
Metals (Liebens 2001, Heijerick *et al.* 2002, Gromaire *et al.* 2002)	Metals are particularly derived from vehicles, including brake linings (nickel, chromium, lead, copper), tyres (zinc, lead, chromium, copper, nickel) and engine alloys (nickel, chromium, copper, manganese).
Biocides (Ramwell *et al.* 2002)	Pesticides can be found in run-off as a result of municipal spraying programmes and domestic gardening.
Micro-organics (e.g. PAHs)	A wide range of micro-organic compounds have also been detected in urban runoff: Vehicles are a major source, mainly as a result of oil leaks. Studies in North America suggest surprisingly large yields of oil pollution from vehicles: in the 1980s the Los Angeles River was estimated to contribute 1% of the annual world petroleum hydrocarbon input to the Pacific Ocean.
Organics	

2.4 Ecological impacts

The complex effects of urbanisation, and the paucity of controlled studies, makes it difficult to isolate the specific effects of urbanisation impacts on ecology. Taken together, however, not only have the impacts of urbanisation on aquatic habitats been shown to be highly detrimental, but they begin to be ecologically significant at relatively low levels of catchment urbanisation. Thus a range of recent studies, mainly in North America, have shown that significant ecosystem impairment occurs when impervious surface cover

(ISC) reaches 10-20% of the total catchment surface area (Wang *et al.* 1997, 2000, Paul and Meyer 2001, Stepenuck *et al.* 2002). These effects are most obvious in catchments which are being converted from largely natural vegetation (for example, forest), a process which is very difficult to observe in Europe but quite common in North America. One consequence of this is that studies of the threshold effects of urbanisation are almost entirely limited to North America: in Europe urbanisation usually occurs on land already in agriculture, which itself creates considerable impacts on streams. Changes due to urbanisation may, therefore, be less obvious since aquatic ecosystems are usually already partly degraded in agricultural landscapes.

2.4.1 Effects of urbanisation on flowing waters

The net biological effect of urbanisation on flowing water habitats is to reduce species richness, eliminate species sensitive to elevated pollutant levels, change community composition and disrupt natural ecosystem processes (e.g. channel - floodplain recolonisation processes).

Such impacts are particularly significant where they affect small watercourses, where pollutant inputs may receive relatively little dilution and the additional volume may substantially increase stream discharge, with considerable implications for both hydrology and substrate.

Observed biological changes in North American catchments subject to increasing urbanisation mainly include studies of fish and invertebrates. In general, fish diversity decreased markedly as the ISC exceeded 10% of catchment area (Linburgh and Schmidt 1990). Studies in Maryland found that fish were absent completely when the ISC was greater than 30-50% of total catchment area (Klein 1979). Due to the lack of data, it is not possible to say whether such relationships would be representative for more impacted European streams and rivers.

Reductions in macroinvertebrate diversity also begin at relatively low levels of urbanisation, with marked changes in richness or Index of Biotic Integrity (IBI) with catchment urbanisation levels of between 6% and 33% ISC. All studies of gradients of urbanisation in single catchments show decreased invertebrate diversity, particularly in groups such as mayflies, stoneflies and caddis flies. Although these patterns are very widely observed, little is known of the specific mechanisms, largely because of the difficulty of designing field experiments that can separate out the different impact factors. Thus most urban streams are both physically modified and subject to chemical impacts, making it very difficult to assess the relative contribution of each of these factors to ecological degradation.

Although a considerable amount of information is available about the changes in invertebrate assemblage composition in urban streams and rivers, these data are almost entirely restricted to simple monitoring type surveys. Thus, for urban streams, little is known about invertebrate life-histories, drift, recolonisation potential, behavioural ecology or production - all common subjects of study by ecologists in less impacted watercourses (Paul and Meyer 2001).

For other groups (e.g. algae, macrophytes, zooplankton), relatively little is known of the effects of urbanisation, although in all cases reductions in diversity and changes to community structure have been observed (Paul and Meyer 2001).

2.4.2 Effects of urbanisation on still waters

Studies of urbanisation effects have mainly focused on flowing waters, with fewer data available from still waters. However, it is apparent that still waters are at least as severely impacted by urbanisation effects as running waters.

Loss and degradation of standing waters, particularly ponds, is important because it may have a particularly significant impact on aquatic biodiversity. Recent catchment-wide studies in predominantly agricultural landscapes suggest that ponds support a high proportion of the total aquatic biodiversity in any area. For example, in the south of England, studies in the catchment of the R. Cole on the

Oxfordshire/Wiltshire border showed that 70% of all aquatic macrophytes and aquatic macroinvertebrates recorded in a 10 x 10 km square were recorded from ponds (Figure 1 and Table 1). Reduction in pond density will decrease the extent of this habitat available to many species and may lead to extinction of species from some areas.

In addition to physical and chemical impacts, biological impacts on aquatic habitats have been of increasing concern recently, particularly through the release and establishment of non-native water plants which are initially introduced for water gardening or in amenity planting schemes (Williams *et al* .1998).

Although data are relatively limited in this area, there is some evidence that urban waterbodies are particularly vulnerable to alien species. This is particularly true of alien aquatic plants which are commonly thrown out of, or escape from, garden ponds. Naturally these are commonest in urban areas so this impact tends to be focussed on urban waterbodies.

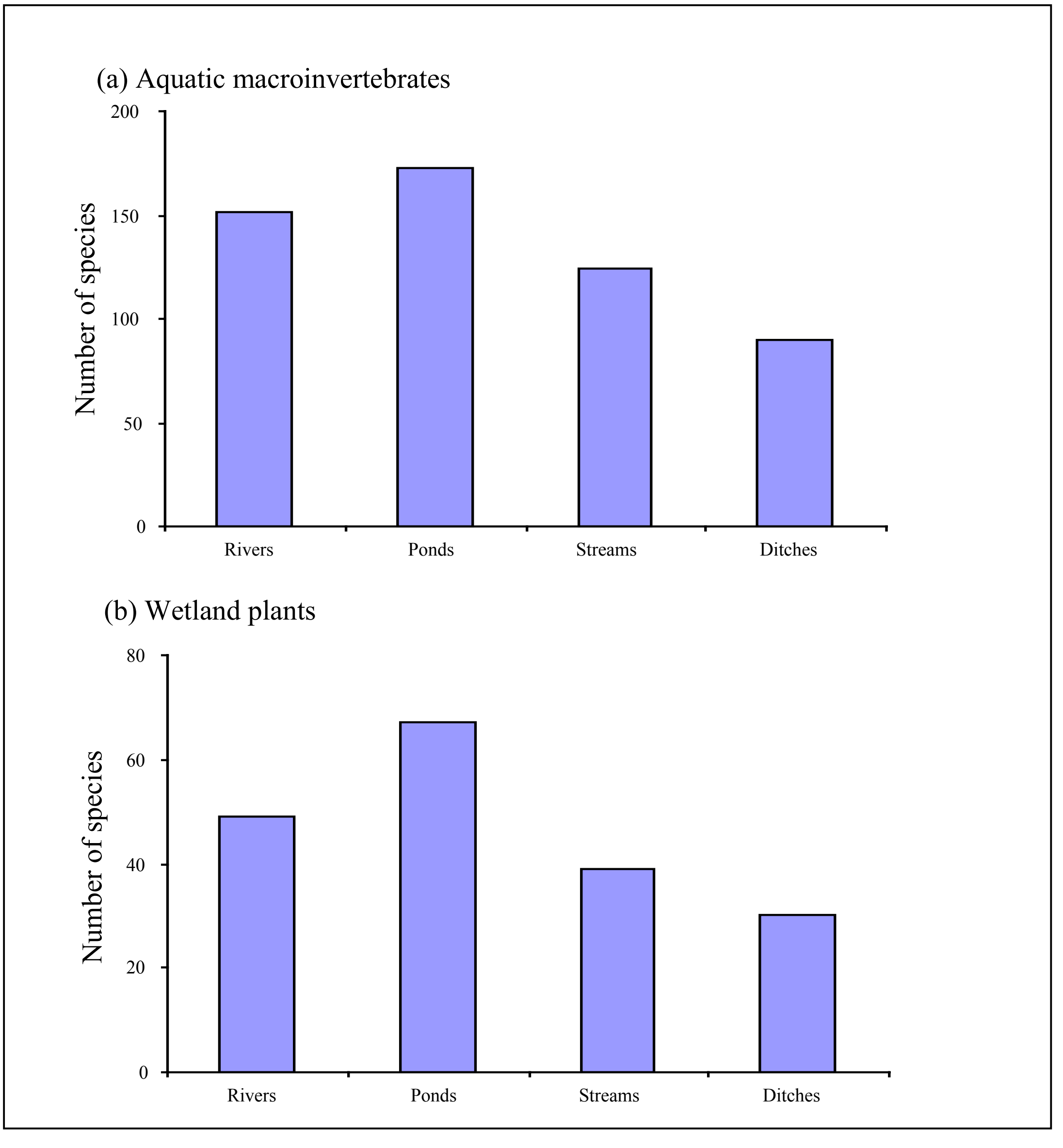

Figure 1. Total number of (a) aquatic macroinvertebrate and (b) wetland plant species (gamma diversity) in different waterbody types recorded in a 10 x 10 km square area of the R. Cole catchment

Table 1. Total number of wetland plant and aquatic macroinvertebrate species (gamma diversity) in different waterbody types in a 10 x 10 km square area of the R. Cole catchment

	Rivers	Ponds	Streams	Ditches	All habitats
Aquatic macroinvertebrate species:					
Total number of species	152	**173**	124	90	249
Proportion of total species richness	61%	**70%**	50%	36%	100%
Number of species unique to the waterbody type	26	**50**	6	8	-
Wetland plant species:					
Total number of species	49	**67**	39	30	88
Proportion of total species richness	56%	**76%**	44%	34%	100%
Number of species unique to the waterbody type	9	**24**	3	3	-

From Williams *et al*. (in press)

2.5 Conventional methods for mitigating impacts of urbanisation on aquatic ecosystems

Until recently, the main method for controlling urban impacts on aquatic ecosystems has been the construction of balancing ponds, which act as a temporary store for flood water. Short-term storage of water in this way can eliminate some pollutants by settling (e.g. suspended sediment, some heavy metals) and biological action (e.g. ammonia). There has also been considerable interest in wetland treatment systems which, at least initially, appeared to offer low cost, low maintenance options for treating urban runoff. In practice, however, studies of the effectiveness of both balancing ponds and wetlands have revealed wide variations in their efficiency at improving water quality both between sites and within the same site under different conditions (see Section 4.4).

In practice, therefore, treatment of urban runoff has still, to a large extent, relied on the ability of the receiving waters (streams, rivers, lakes or ponds) to dilute pollutants. Although this approach is probably effective (if not acceptable) where small volumes of effluent enter large diluting volumes in the receiving water it is much less likely to be successful for the smaller waterbodies which characterise most of the network length in the most catchments. Thus, although widely adopted, the approach of using the receiving water as the final treatment medium is increasingly seen as damaging and unsustainable.

3. REVIEW OF EXISTING GUIDANCE ON THE ECOLOGICAL DESIGN OF SUSTAINABLE DRAINAGE SYSTEMS (SUDS)

3.1 Data availability

Well founded ecological guidance on the design of SUD schemes is surprisingly limited. Amongst aquatic ecologists, investigations of the urban environment have remained a minority interest with little serious scientific attention. As a consequence, understanding of the effects of urbanisation remains at the most general level, and explanations of the patterns seen in the urban environment are generally still at the level of hypothesis rather than confirmed observations.

The design of SUD schemes themselves is based largely on experience of non-SUDS systems and on superficial observation or limited testing. Although a reasonable amount of information is available from studies of single components (swales, ponds, filters, porous surfaces etc), particularly of chemical efficiency, there are few studies of multi-element SUDS treatment trains, where these components are linked together.

Published information on techniques for improving the ecological value of SUDS ponds and wetlands as wildlife habitats is very recent indeed (e.g. SEPA 2000). However, although these recommendations are at least research based, they are again based almost entirely on information derived from studies of countryside ponds, rather than studies of SUD systems themselves.

3.2 Current recommendations

This section briefly reviews the existing design guidance on SUDS to maximise their ecological benefits. Fuller information on the optimum design and management of SUDS for maximising ecological value and minimising impacts in the light of current monitoring data is given in Section 6.

3.2.1 Introduction

SUD schemes may provide ecological benefits in two ways: firstly by protecting receiving waters and secondly by providing additional, mainly aquatic, habitat.

The primary components of SUD schemes which can provide useful habitats are the 'soft landscape' features, principally ponds and wetlands. Swales and surrounding terrestrial habitat areas (e.g. grassland, scrub) may also be useful, particularly if allowed to colonise naturally or planted with native species and managed extensively (i.e. without use of fertilisers and biocides). The 'hard landscape' components of SUDS (such as gravel trenches or porous paving) are not, based on present knowledge, likely to have significant value as habitats and function principally to protect existing ecosystems. They are not discussed further here.

3.2.2 Design and management of SUDS ponds and wetlands (including wet and dry detention basins)

Design for impact mitigation

The mitigation of impacts of urban runoff on receiving waters has, until very recently, been regarded as the primary objective of SUD systems. Because of this, guides on SUDS design generally cover this area quite thoroughly and a range of information is available in recently produced manuals (e.g. CIRIA 2000).

A number of regions in North America have also published guides to Best Management Practice devices which are generally more detailed than UK guidance.

Design for maximising ecological value

The only detailed advice available about the design of SUD schemes to maximise their value as habitats, particularly the pond and wetland component, is contained in the SEPA guide 'Ponds, pools and lochans (SEPA 2000), prepared by the authors of the current report (see Section 6 and Box 3, where this information is briefly summarised in the light of data on SUDS performance discussed in this report). For ecologists concerned with pond and wetland design, the more extensive information contained in (Williams *et al.* 1999) is also an important source, on which the SEPA guide is itself based.

Other manuals about SUDS design generally make only very brief comments on the ecological design of ponds and wetlands, often based on rather outdated ideas about pond ecology. North American guides to BMP systems contain similar levels of information about the value of BMP ponds and wetlands to that seen in European literature. This reflects the fact that, as in European literature, there have been very few studies of the design of new ponds and wetlands.

Management techniques for enhancing value

Little information is available about the management of SUDS ponds and wetlands to maximise their ecological value. This reflects the fact that, more generally, there have been virtually no technical studies of the effects of pond management on pond ecosystems, despite this being one of the most commonly undertaken environmental management activities (Biggs *et al.* 1994).

The starting point for ecological management in ponds, particularly dredging and vegetation removal, is that it is *not automatically necessary for maintaining or improving the value of ponds* (Williams *et al.* 1999). This is clearly in contrast to the requirements of SUDS ponds and wetlands where management is certainly required to maintain their performance function.

Managing SUDS ponds and wetlands from a wildlife perspective is essentially concerned with:

1. Maintaining, as far as possible, good water quality
2. Adjusting physical management regimes needed to maintain pond function (e.g. dredging sediments) to protect populations of species of conservation concern (e.g. Biodiversity Action Plan species) if they become established in ponds.

Application of these general principles will ensure that pond and wetlands habitat value is maximised. Detailed recommendations on good practice in pond management for the maintenance of ecological quality of ponds and wetlands is given in Section 6.

3.2.3 Design and management of filter strips and swales

Design for impact mitigation

Existing advice on the design of filter strips and swales for impact mitigation (i.e. treatment and removal of pollutants) is given in CIRIA 2000 and in various North American guides to BMP design, such as those published by the US EPA (1999b).

Further information on best practice, considered in the light of the results of the review presented here, is given in Section 6.

Design for maximising ecological value

There are no published guidelines on maximising the value of filter strips and swales as wildlife habitats. Best practice recommendations are given in Section 6.

Management techniques

Guidance on the general management of filter strips and swales is given in CIRIA (2000). No specific guidance is available on the management of these devices for maintaining habitat quality.

3.2.4 Design and management of terrestrial habitats (e.g. grassland, scrub)

Design for impact mitigation

The green spaces in which most SUDS schemes are located can play a role in impact mitigation, mainly through minimising the contributions that they themselves make to diffuse pollution and runoff rates. However, information on the design of surrounding landscapes in British SUDS literature is very limited, particularly the potentially negative effects of those landscapes (as sources of nutrients, biocides, sediments and organic matter). CIRIA (2000), for example, focuses entirely on the SUDS devices themselves and pays little attention to the design of the landscape in which the devices are placed.

In partial contrast to the UK manuals, some North American guides do provide guidance on landscape management, particularly with respect to non-structural BMP activities - although these are typically concerned with the management of whole catchments (e.g. SFWMD 2002).

Design for maximising ecological value of terrestrial habitats

Brief advice on maximising the ecological value of terrestrial habitats is given in SEPA 2000. Further notes on best practice are given in Section 6 of this report.

Management techniques for terrestrial habitats

No technical information is available on the management of terrestrial landscapes to maximise their wildlife habitat value. Further guidance on best practice is given in Section 6.

4. HOW ECOLOGICALLY EFFECTIVE ARE SUD SCHEMES?

4.1 Data availability

In the UK, data which can be used to assess the value of wetland habitats created as part of SUD schemes, and their effects on receiving waters, are available mainly from studies in central Scotland and from the Hopwood Park Motorway Service Area (MSA) on the M42 south-west of Birmingham. To date, all habitat value data relate to ponds or dry detention basins, with no information on other SUDS devices (e.g. swales, adjacent terrestrial habitat). Data on water quality relate to individual SUDS devices and to complete SUD systems. In addition to the data from Scotland and Hopwood Park MSA, a limited amount of plant and invertebrate data is also available from a group of balancing ponds in the Oxford area, although these sites are not part of fully developed SUD schemes.

In addition to data from specifically designed SUDS schemes, a considerable amount of information is available on the performance of individual urban water management structures such as ponds, swales and filters, particularly in North America. As these devices are commonly linked together in SUDS schemes, analysis of such data is useful for understanding the potential performance of multi-component SUDS.

Two approaches to assessing the quality of SUDS ponds as aquatic habitats are possible using currently available information:

(a) Pond biota can be described directly (particularly plants and macroinvertebrates - two of the most biodiverse groups)
(b) Water quality in SUDS ponds and wetlands can be compared to known biological water quality limits (e.g. the Total Phosphorus concentrations at which waterbodies may typically be regarded as impacted by eutrophication).

Both approaches are used in the review below, although most data are available on the water quality performance of SUDS features. Far fewer data are available which describe directly the plants and animals present in SUDS ponds, wetlands and other habitats.

Assessing the benefits of SUDS in protecting receiving waters relies, at present, on comparing typical SUDS output water quality with recognised environmental standards for flowing and still waters. The alternative approach would be the assessment of the effects of SUDS vs. conventional drainage systems on receiving waters. This latter approach would be extremely complex as it would depend on the existing quality and ecology of the receiving water, on the potential for dilution provided by the receiving water, and on isolating the conventional system impact and replacing it directly with a more sustainable alternative.

4.2 The value of SUDS ponds as aquatic habitats

In the following review, the value of SUDS ponds as aquatic habitats is assessed in terms of wetland plant and aquatic macroinvertebrate species richness and rarity. To make this assessment, data from SUDS ponds were compared with the results of larger national surveys, particularly the DETR Lowland Pond Survey 1996 (Williams *et al.* 1998) and the National Pond Survey (PCTPR, unpublished data).

4.2.1 Wetland plants

Studies of the wetland plant assemblages of SUDS ponds have been made at about 30 sites in Scotland and England (Table 2) (Pond Action 2000, Berrendt and Wilby 2002, PCTPR, unpublished data from Hopwood Park MSA).

Table 2. Wetland plant species richness in SUDS ponds in Scotland and England

Country	Site	Authors	Number of Ponds	Species richness[1]	
				Mean	Range
Scotland	Various	Pond Action (2000)	13	13	2-25
Scotland	Various	Behrendt and Wilby (2002)	25	11	3-20
England	Hopwood Park	PCTPR (unpublished data)	8	9.6	6-13

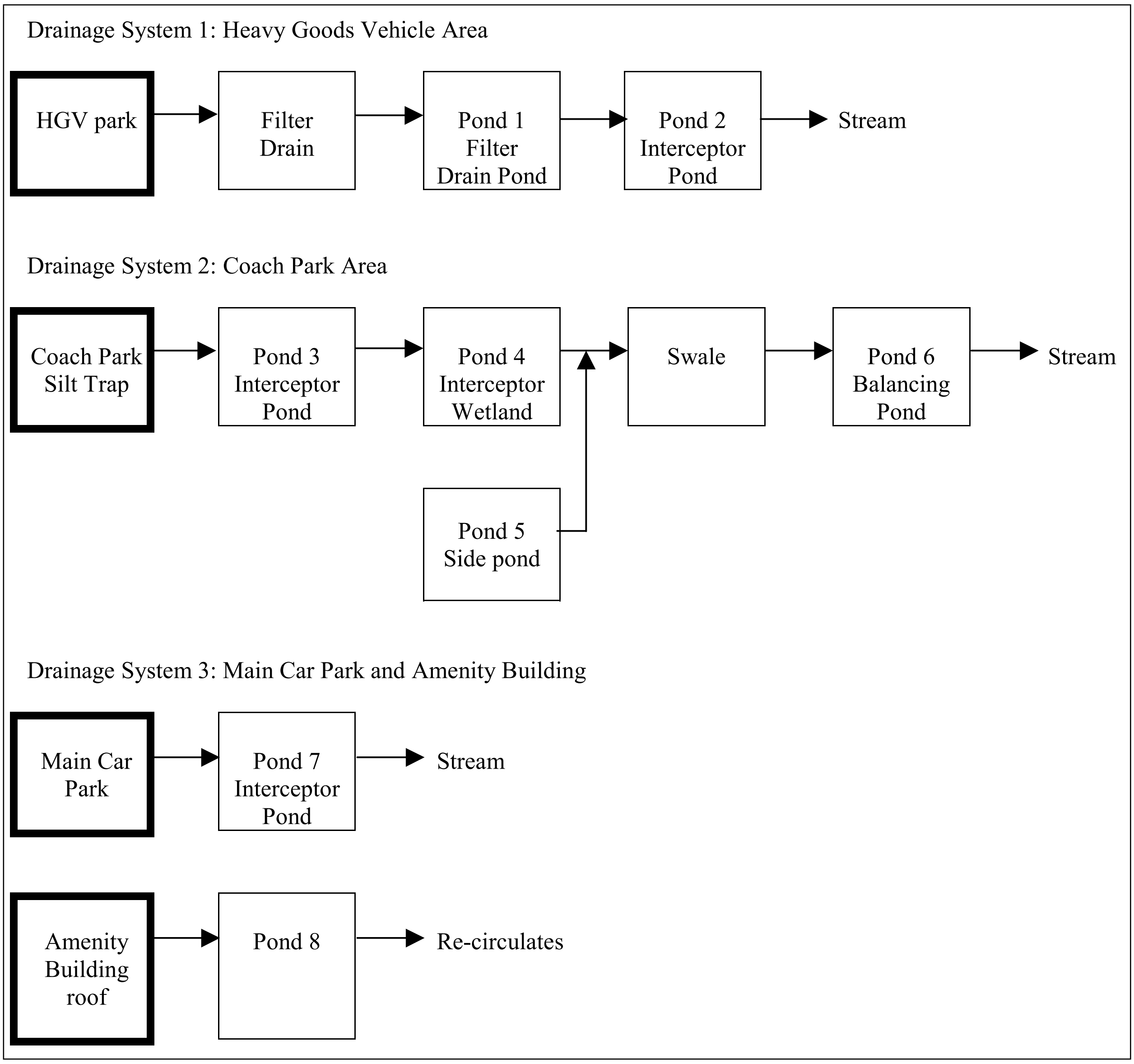

Figure 2. Layout of SUDS scheme drainage system at Hopwood Park MSA. Waterbodies with numbers are those on which plant and invertebrate surveys have been carried out.

[1]Naturally self-colonising species only; planted species excluded as far as possible.

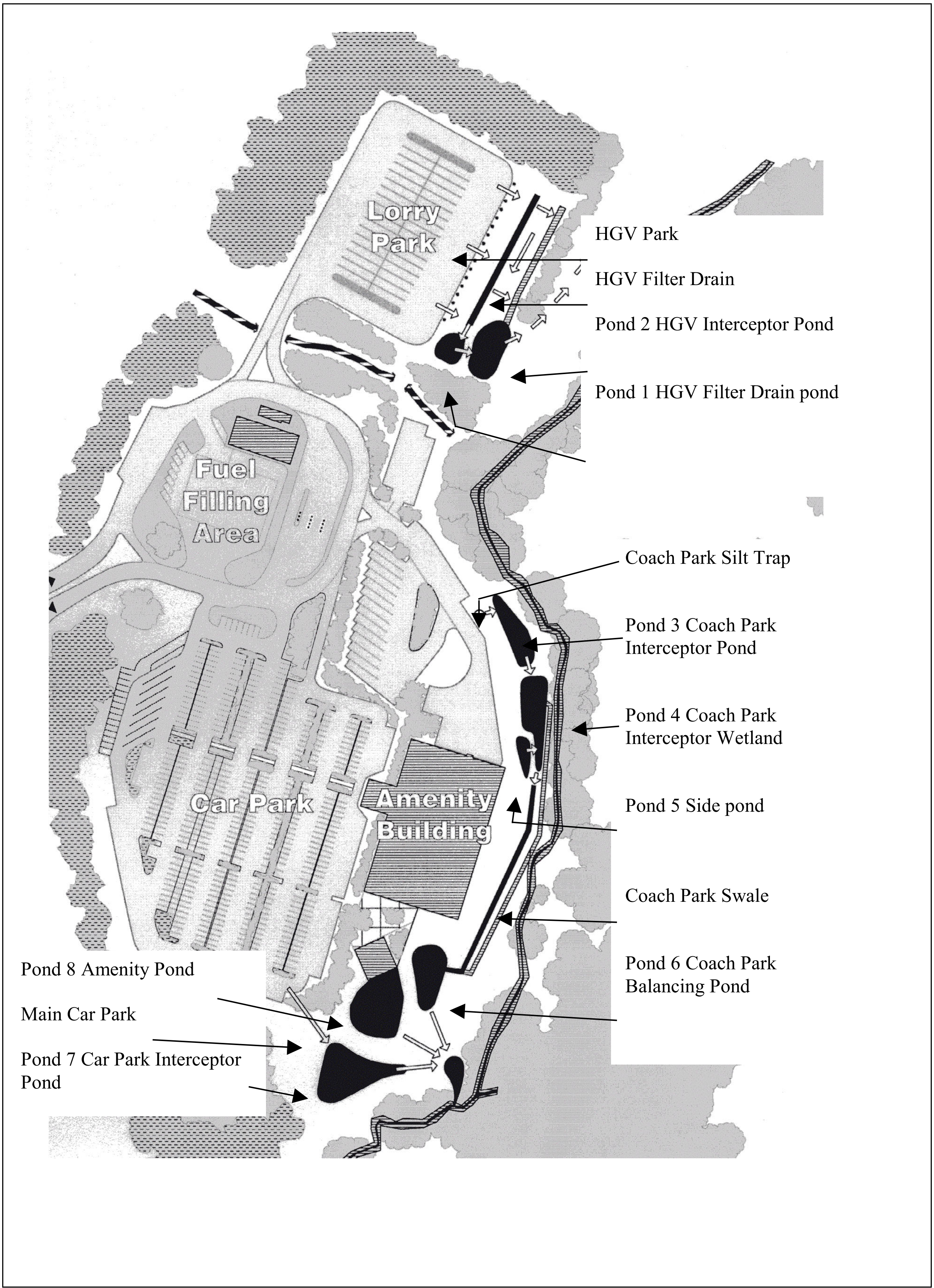

Figure 3. Location of SUDS elements at Hopwood Park motorway service area

Plant richness in Scottish SUDS ponds. Mean wetland plant species richness in Scottish SUDS ponds was 11-13 species/site. This is above the average of 10 species/pond for wider countryside ponds in lowland Britain (including Scotland). However, it should be noted that many ponds in the wider countryside are already impacted by pollutants and support, on average, only half of the wetland plant species that would be expected in minimally impaired ponds. For SUDS ponds, therefore, a more stringent comparison is with the minimally impaired ponds of the National Pond Survey which are as little impacted by anthropogenic stresses as is possible in the British landscape. Mean species richness of Scottish NPS sites is 15.4 species/pond, 20-40% greater than the mean richness recorded in the three Scottish SUDS studies (Table 2)[2].

Plant rarity in Scottish SUDS ponds. Data on species rarity in the Scottish ponds are available only from the Pond Action (now PCTPR) study of SUDS ponds in central Scotland. The mean number of uncommon species[3] recorded in the SUDS ponds (0.2 species/site) was significantly lower than the mean number of uncommon species in minimally impaired ponds in Scotland (1.24 species/site). In the Scottish SUDS ponds surveyed only one locally uncommon wetland plant species was recorded, the wetland grass Orange Foxtail (*Alopecurus aequalis*), which occurred in three of the 13 ponds.

Overall, taking account of species richness and rarity, the Scottish SUDS ponds surveyed by PCTPR were mainly of moderate to high conservation value, on a four point scale (Low, Moderate, High, Very High). Two of the thirteen sites were of Low value and none were of Very High value[4].

Plant richness and rarity in Hopwood Park MSA SUDS ponds. Mean wetland plant species richness in the Hopwood Park MSA ponds (south of Birmingham) was 9.6 species/pond, considerably lower than for the Scottish sites and reaching only the level of impacted lowland ponds. However, as the Hopwood Park ponds were only 12 months old at the time of survey, comparison with generally longer established ponds in the wider countryside may be misleading. A better assessment of the Hopwood Park ponds may be made by comparison with other new ponds. In practice, the only long-term data set available from minimally impaired new ponds comes from the wetland complex at Pinkhill Meadow, Oxfordshire, created in 1990/91 (Biggs *et al*. 1995). These ponds were established in semi-natural grassland on meadowland adjacent to the River Thames near Oxford and colonised completely naturally. Because the site is minimally impaired and within 75 miles of the Hopwood Park site, the species accumulation rate in the Pinkhill ponds can be assumed to represent a good target for the species richness to be expected from the Hopwood Park ponds.

At Pinkhill, ponds of a similar size to the medium and larger ponds at Hopwood Park supported about 10 species of wetland plants after 12 months of natural colonisation. This is similar to the number of naturally colonising species at Hopwood Park, suggesting that the Hopwood Park ponds are colonising well. After seven years, the Pinkhill ponds supported about 30 species, well above the average for minimally impaired ponds.

No nationally uncommon wetland plant species have so far been found to have naturally colonised the Hopwood Park ponds. Minimally impaired ponds in England and Wales support an average of 2.29 uncommon species/site. Overall, the Hopwood Park ponds were all of Low conservation value for wetland plants but this probably reflects their early stages of colonisation.

A more limited wetland plant dataset is also available from the five ponds surveyed by Ross (1999) in the Oxford area. These sites drained various road, housing and industrial developments but were not part of

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[2] Note: although Scottish SUDS ponds supported considerably fewer plant species on average than minimally impaired ponds in England and Wales (mean of 23 species/site, NPS, unpublished data) this probably reflected biogeographic factors, since Scottish freshwaters support fewer species than waterbodies further south.

[3] A 'locally uncommon' plant is defined here as a plant species which has been recorded from less than one quarter of the 10 x 10 km squares in the UK. It does not include Nationally Scarce or Red Data Book plant species which are much rarer.

[4] Conservation values were recalculated from the Pond Action (2000) report using new species richness limits for Scotland.

SUDS treatment train systems, simply receiving untreated runoff from the development areas. Mean species richness in these sites was 16.6 naturally colonising species/site, intermediate between the wider countryside mean (10 species/site) and the mean for minimally impaired ponds in England and Wales (23 species/site).

Overall, data on plants in SUDS ponds indicate that wetland plant species richness at least reaches or slightly exceeds that of 'ordinary' countryside ponds (i.e. ponds impacted by pollution and other damaging factors). However, SUDS ponds support significantly fewer plant species than would be expected in minimally impaired 'natural' ponds indicating that, as might be expected, many are still experiencing environmental impacts.

Although on average SUDS ponds are a little better than the typical damaged ponds of the British countryside, some individual SUDS ponds are of relatively high ecological quality, reaching the High conservation value category. To date, however, no SUDS ponds have been found to support Very High value wetland plant communities whereas amongst minimally impaired ponds, 12% of all sites reach this status in terms of their wetland plant assemblages (Williams *et al*. 1998).

4.2.2 Aquatic macroinvertebrates

Data on aquatic macroinvertebrate assemblages are available from studies undertaken by PCTPR at Scottish SUDS sites and from PCTPR/Environment Agency work at Hopwood Park MSA. Work at Stirling University on macroinvertebrates is currently being undertaken by MSc student Tricia Campbell but results have not yet been reported (Nigel Wilby, *pers. comm.*).

Table 3. Aquatic macroinvertebrate species richness in SUDS ponds in Scotland and England

Country	Site	Authors	Number of Ponds	Species richness[5]	
				Mean	Range
Scotland	Various	Pond Action (2000)	5	39.8	24-58[2]
England	Hopwood Park	PCTPR (in preparation)	8	36.9[6]	22-58[2]

As for wetland plants, the quality of aquatic macroinvertebrate assemblages in the PCTPR studies was assessed in terms of species richness and rarity.

The Scottish SUDS ponds supported a mean of 40 macroinvertebrate species in a three minute hand net sample. This is significantly above the mean for minimally impaired ponds (30 species/3 minute sample[7]). Nationally uncommon species[8] were rare in the Scottish SUDS ponds with an average of only 0.25 species/3 minute sample compared to the average of 2.6 uncommon species/3 minute sample in minimally impaired ponds.

Overall, three of the Scottish SUDS ponds surveyed by PCTPR were of High conservation value (on a four-point scale of Low, Moderate, High and Very High) with 1 each of Moderate and Very High value.

Assessment of invertebrate assemblage quality in the Hopwood Park ponds is constrained by methodological problems. PCTPR quality control of 3 minute invertebrate samples taken by the Environment Agency suggests that the Agency recorded approximately half the number of species found by PCTPR. Species richness data given in Table 3 have, therefore, had a correction factor applied to them to allow for this discrepancy. The analysis given below should, therefore, be treated with some caution.

[5] Number of species in a 3 minute hand-net sample.

[6] Based on corrected values.

[7] The mean number of invertebrate species in minimally impaired ponds Scottish NPS sites is slightly lower than this (27.2 species/3 minute hand net sample) but the England and Wales value has been used because the Scottish value is based on a small number of ponds (n = 25).

[8] Nationally uncommon species are those which are local, National Notable or listed in UK Red Data Books.

Estimated mean species richness in the Hopwood Park SUDS ponds was slightly greater than the mean value for minimally impaired ponds. Species rarity could not be adequately assessed because of the methodological problems noted above. Overall, based on species richness data alone, most ponds were of either Moderate or High ecological quality in terms of their invertebrate assemblages, with one site of Low value and one of Very High value.

4.2.3 PSYM analysis of Hopwood Park MSA pond quality

Background and aims

In addition to assessments based on species richness and rarity, both common measures of habitat quality in aquatic ecosystems, the Hopwood Park ponds were also assessed using PSYM, a system for assessing the overall ecological integrity of ponds, developed by the Environment Agency and the Ponds Conservation Trust.

PSYM predicts the plants and invertebrates that would be expected to occur in ponds that are little affected by human impacts (e.g. pollution, land drainage, unnatural numbers of fish or ducks). The predictions are made using simple environmental data about a pond (e.g. pond area, geology, pH). Comparing the predicted flora and fauna with the plants/invertebrates actually present in the pond provides an objective assessment of the extent to which the pond is reaching its biological potential.

The degree of impairment of the pond is described using three plant and three invertebrate biological measures (‘metrics’) known to be correlated with different types of environmental impact. These are:

<u>Plants</u>: Submerged and emergent plant species richness
Number of uncommon plant species
Trophic Ranking Score

<u>Invertebrates</u>: Average Score Per Taxon
Number of dragonfly and alderfly families
Number of beetle families

Results of a PSYM analysis are given as a percentage of the maximum score possible. Sites with scores below 50% are likely to be significantly below their full ecological potential.

Inspection of the individual PSYM metrics can also give an initial indication of the causes of degradation (e.g. high TRS values can indicate nutrient pollution and low scores for the number of water beetles families can indicate poor marginal habitat structure). A description of the PSYM methodology is available in Environment Agency and Pond Action (2000).

Results and discussion

A PSYM analysis of the summer samples from Hopwood Park was undertaken to assess the ecological integrity of the SUDS ponds.

The Hopwood Park ponds had PSYM scores between 33% and 61% of their full potential. As the ponds are only 12 months old these low scores are, almost certainly, in part, a reflection of the relatively early stages of colonisation of the ponds. At this stage, therefore, it is difficult to separate colonisation effects from site impairment.

Although the absolute values from the PSYM analysis should currently be treated with caution, some indication of the relative condition of the ponds is given by the results (Table 4). Overall, Ponds 4, 6 and 8 have the highest PSYM scores for Summer 2000 suggesting that these sites are generally closer to their full potential.

Analysis of individual metrics broadly confirms the general trends seen in the overall scores. Thus, the plant measures 'Number of Submerged and Marginal Species' and 'Number of Uncommon Species' were generally well below expectation, only one pond (Pond 6) having more than half the expected number of plants. However, it is, at present, impossible to determine whether these low values are due to lack of colonisation or to anthropogenic stresses.

All ponds, except Pond 8, have relatively high Trophic Ranking Scores indicating that nutrient levels are generally high. Pond 8 is the Amenity Pond fed by roof runoff alone and not exposed to car and lorry park contaminants. However, TRS scores also need to be interpreted with some caution at this stage because plant lists are, at present, relatively short.

The invertebrate metrics 'ASPT' and 'Number of Dragonfly and Alderfly Families' were generally below the expected values except in Ponds 6 and 7. These metrics are most sensitive to chemical water quality impacts but, again, it is difficult to be certain whether the low values are due to chemical impacts or to the current extent of colonisation.

The 'Number of Coleoptera Families' metric was generally only slightly below the expected value. Since a reasonably wide range of water beetle species had colonised the ponds the low values were, again, probably a reflection of the relative immaturity of the ponds.

Table 4. Results of PSYM analysis of ecological integrity of Hopwood Park ponds

Pond	**PSYM score (% of maximum possible score)**
1 HGV Filter Drain Pond	33%
2 HGV Park Interceptor Pond	44%
3 Coach Park Spillage Collector Basin	44%
4 Coach Park Interceptor Wetland	50%
5 Side pond	33%
6 Coach Park Balancing Pond	61%
7 Coach Park Interceptor Pond	44%
8 Amenity Pond	50%

4.2.4 Conclusions

Biological data on SUDS ponds collected to date needs to be considered with the following caveats:

1. Relatively small numbers of sites have been investigated.
2. Data on invertebrate richness at Hopwood Park must be regarded with particular caution as conclusions are based mainly on corrected data (following quality control of Environment Agency samples by PCTPR).
3. There have been no long-term investigations of SUDS ponds, all studies to date essentially being 'snapshots'.
4. Very few ponds have both biological and environmental data (particularly water chemistry) making it difficult to determine whether differences compared to benchmarks are due to contaminant impacts or other factors.
5. Little is known of the ways in which specific pollutants (e.g. heavy metals, nutrients) impact on pond biota, most studies of these interactions having been undertaken in either in the laboratory or in lakes and rivers.

Overall, studies of SUDS ponds biota to date suggest that, on average, SUDS ponds support better quality plant and invertebrate assemblages than impacted ponds in the 'ordinary' British countryside. In both Scotland and England, amongst the ponds surveyed (n=46 for plants, n = 13 for invertebrates) most ponds have communities of Moderate or High conservation value. A smaller number have Low value and two have been found with Very High value assemblages. The early stage of colonisation at Hopwood Park makes it difficult to interpret the results of PSYM assessment of overall ecological quality in these ponds.

In summary, the results of these studies suggest that SUDS ponds behave in a way typical of ponds generally i.e. ecological quality is greatest in ponds where water quality is highest and where ponds are located closest to established wetland habitats (e.g. river valleys). SUDS ponds therefore remain generally below the standard of minimally impaired ponds; this does not represent an impossibly high standard as techniques for creating ponds which reach Very High quality status are well-established and repeatable. Although they certainly add to the stock of ponds, there is relatively little evidence that as yet, SUDS ponds are adding significant numbers of *high quality* new ponds to the landscape.

4.3 Using chemical data to assess the ecological value of SUD systems

Observing the biota provides a direct measure of the value of SUDS as habitats. A more indirect, but still valuable, approach is to compare observed chemical quality with recognised optima and benchmarks for still and flowing water habitats.

As noted above, two types of data are available with which to make these assessments:

(i) data from the many separate systems, particularly ponds, which mostly do *not* have source control devices or treatment train devices upstream
(ii) a much smaller body of information from true SUDS systems which incorporate source controls and treatment trains.

4.3.1 Chemical quality benchmarks for the assessment of SUD schemes

Most studies of the ability of urban runoff treatment systems to remove contaminants emphasise percentage removal of contaminants. The underlying rationale behind this approach appears to be a general assumption that there is a linear relationship between the pollutant dose and its biological effect: i.e. that a (say) 75% reduction in pollutant load will lead to a 75% reduction in pollution impacts. In practice ecosystems do not function in this simple manner. For example, if Total Phosphorus levels in urban runoff are 5 mg/l, 75% reduction gives outflows of 1.25 mg/l, more than 10 times the level (0.1 mg/l) at which significant environmental impacts start to occur. It is important, therefore, to judge the success of SUDS systems on whether they deliver water that is clean enough at the outflow for appropriate biological functioning in the receiving waterbody.

There have been some limited attempts to relate SUDS outputs to recognised chemical standards but typically these:

(a) refer only to receiving waters
(b) deal with a very small number of contaminants
(c) only rarely is the chemical quality of the SUDS system itself considered against benchmarks. In addition, attempts to set benchmarks generally pre-date recent developments in European legislation, particularly the Water Framework Directive, which have shifted the emphasis of waterbody protection to maintaining conditions equivalent to the minimally impaired baseline condition. 'Minimally impaired' means the ecological characteristics associated with waterbodies which are only very slightly or not at all, impacted by human activities.

For this reason, a provisional set of chemical baselines is set out here referring to most commoner contaminants which are of concern in urban runoff (Table 5). The chemicals selected are those which are generally recognised as being of concern in urban runoff, such as BOD, suspended sediments, nutrients, metals, and for which sufficient suitable data are available. It was not possible to develop an analysis of hydrocarbons as no data were available on acceptable baseline levels.

To define chemical baselines, information from three main sources has been used:

(i) measured levels in waterbodies which are believed to be 'minimally impaired'
(ii) results of ecotoxicological studies (potentially relevant to all habitats)
(iii) limits set by national monitoring bodies (mainly lakes and rivers).

For running waters, data are given from both the Environment Agency of England and Wales and, for comparison, the Swedish Environmental Protection Agency. This includes data on thresholds, including quality limits agreed in national monitoring programmes. For ponds, the most relevant information comes from the UK National Pond Survey which provides an indication of the chemical conditions found in high quality ponds in the UK that are free from significant pollution impacts. Although data on lakes are also widely available they are less directly relevant to smaller SUDS waterbodies.

Biological benchmarks have been prepared or reviewed for 15 determinands: pH, Biochemical Oxygen Demand (BOD), Soluble Reactive Phosphorus (SRP), Total Phosphorus (TP), Total Oxidised Nitrogen (TON), Total Nitrogen (TN), suspended sediments, ammonia, copper, cadmium, chromium, iron, lead, mercury and zinc.

Benchmark levels for pH cannot easily be defined as there is considerable natural variation in pH between and within waterbodies, particularly still waters. For BOD, no data are available on 'natural' levels in ponds, although it can be expected that these will be relatively high in some natural waters (e.g. woodland ponds with heavy leaf litter inputs). For the nutrients (SRP, TP, TON and TN), levels likely to cause detrimental impacts are well-established and are based on OECD (Organisation for Economic Co-operation and Development) figures. These levels probably also apply in ponds, as minimally impaired sites have similar levels to those seen in the least impacted lakes. Nutrient levels in rivers have not been defined in the UK or Sweden but it is likely that the levels which cause damage in still waters also cause impacts in running waters. Levels for the common heavy metals are available for running waters and lakes; for ponds data on heavy metals are only available for copper, iron, lead and zinc.

Data on the minimum values known to cause observable ecological effects are mainly taken from the US EPA toxicological benchmarks programme (Suter and Tsao 1996). Values given are mainly Chronic Values (CVs) which are the geometric mean of the Lowest Observed Effect Concentration (LOEC) and the No Observed Effect Concentration (NOEC). It should be noted that levels known to cause observable effects on organisms are often below limits set by environmental protection agencies for the highest quality waters, and below those in minimally impaired sites.

The value of SUD schemes as habitats has been assessed by comparison with still water chemical baselines. Their impact on receiving waters has been assessed by comparison with flowing water chemical baselines.

Table 5. Biologically relevant levels of some commonly measured chemical determinands

Variables selected for inclusion include those which are biologically relevant for which sufficient suitable data for the analysis were readily accessible.

Contaminant	The Drinking Water Standards. The Water Supply Regulations 1989	Level in minimally impaired ponds in the UK	Environment Agency River Ecosystem Class 1 limits (RE 1)	Environment Agency River Ecosystem Class 2 limits (RE2)	Environment Agency River Ecosystem Class 4 limits (RE4)	Swedish EPA limits	Urban Waste Water Treatment Directive 91/271/EEC	Minimum concentrations causing observable biological effect
Biochemical Oxygen Demand		No data	2.5 mg/l (90 percentile)	4 mg/l	8 mg/l	No data	25 mg/l	Not applicable
Soluble Reactive Phosphorus (PO_4-P)		Median: 5 µg/l Mean: 69 µg/l n = 162	No definition			No definition		Not applicable
Total Phosphorus	2200 µg/l	Median: 77 µg/l Mean: 190 µg/l n = 49 Note that this value should be treated with caution owing to the relatively small number of sites for which data area available.	No definition			100 µg/l (May – October) Data derived from lakes; concentrations above this value are regarded as hypertrophic. Note that some ponds may naturally be hypertrophic.	2000 µg/l	Not applicable
Total Oxidised Nitrogen		Median: 13 µg/l Mean: 496 µg/l n = 158	No definition			No definition		Not applicable
Total Nitrogen		Median: 1.5 mg/l Mean: 2.9 mg/l n = 45 Note that this value should be treated with caution owing to the small number of sites for which data are available.	No definition			1.25 - 5.00 mg/l (May - October)	15 mg/l	Not applicable

Table 5. Biologically relevant levels of some commonly measured chemical determinands (continued)

Contaminant	The Drinking Water Standards. The Water Supply Regulations 1989	Level in minimally impaired ponds in the UK	Environment Agency River Ecosystem Class 1 limits (RE 1)	Environment Agency River Ecosystem Class 2 limits (RE2)	Environment Agency River Ecosystem Class 4 limits (RE4)	Swedish EPA limits	Urban Waste Water Treatment Directive 91/271/EEC	Minimum concentrations causing observable biological effect
Suspended sediments		Median: 9.3 mg/l Mean: 19.1 mg/l n = 103	No data			No data	35 mg/l	25 mg/l[9]
Ammonia	500 µg/l	Median: 0.067 mg/l Mean: 0.27 mg/l n = 103	0.25 mg/l (90 percentile)	600 µg/l	2500 µg/l	No data		1.7 µg/l (US EPA Chronic Value)
Copper	3000 µg/l	11.48 µg/l	Hardness (mg/l CaCO3) Copp (µg ≤10 5 >10&≤50 22 >50&≤100 40 >100 112	5 – 112 µg/l	5 – 112 µg/l	Rivers: 1 µg/l Lakes: 0.3 µg/l		6.54 µg/l 1.1 µg/l (predicted NOEC) 5.3 µg/l (US EPA Chronic Value)
Cadmium	5 µg/l	No data	No data			Rivers: 0.003 µg/l Lakes: 0.005 µg/l		0.66 µg/l (US EPA Chronic Value) 0.4 µg/l (Level below which 75% of European rivers fell[10])
Chromium	50 µg/l	No data	No data			Rivers: 0.2 µg/l Lakes: 0.05 µg/l		Cr (III): <44 µg/l (US EPA Chronic Value) Cr (VI): 2 µg/l (US EPA Chronic Value) 11.5 µg/l (Level below which 75% of European rivers fell[11])

[9]EIFAC (1964)
[10]Stanners and Bourdeau (1995)
[11]Stanners and Bourdeau (1995)

Table 5. Biologically relevant levels of some commonly measured chemical determinands (continued)

Contaminant	The Drinking Water Standards. The Water Supply Regulations 1989	Level in minimally impaired ponds in the UK	Environment Agency River Ecosystem Class 1 limits (RE 1)	Environment Agency River Ecosystem Class 2 limits (RE2)	Environment Agency River Ecosystem Class 4 limits (RE4)	Swedish EPA limits	Urban Waste Water Treatment Directive 91/271/EEC	Minimum concentrations causing observable biological effect
Iron		Median: 221µg/l Mean: 836 µg/l n = 96	No data			No data		1000 µg/l
Lead		Median: 15.7µg/l Mean: 20.6 µg/l n = 96	No data			Rivers: 0.05 µg/l Lakes: 0.05 µg/l		12.26 µg/l
Mercury		No data	No data			Rivers: 0.0001 µg/l Lakes: 0.001 µg/l		0.012 µg/l
Nickel			Hardness (mg/l CaCO3) — Nickel (µg/l) annual average 0-50 — 50 >100-150 — 150 >150-250 — 200					5 µg/l
Zinc	5000 µg/l	Median: 80.1 µg/l Mean: 97.0 µg/l n = 109	Hardness (mg/l CaCO3) — Zinc (µg/l) ≤10 — 30 >10&≤50 — 200 >50&≤100 — 300 >100 — 500	30 – 500 µg/l	300 – 2000 µg/l	Rivers: 3 µg/l Lakes: 1 µg/l		30 µg/l
pH	It is impractical to provide a 'natural' baseline pH as pH varies naturally over a very wide range; thus pHs from 4.0-10.0 can be encountered in minimally impaired waters.							

River Ecosystem Classification	General Quality Assessment	Likely uses and characteristics*
RE1	A - Very good	• All Abstractions • Very good salmonid fisheries • Cyprinid fisheries • Natural ecosystems
RE2	B - Good	• All abstractions • Salmonid fisheries • Cyprinid fisheries • Ecosystems at or close to natural
RE3	C – Fairly good	• Potable supply after advanced treatment • Other abstractions • Good cyprinid fisheries • Natural ecosystems, or those corresponding to good cyprinid fisheries
RE4	D – Fair	• Potable supply after advanced treatment • Other abstractions • Fair cyprinid fisheries • Impacted ecosystems
Re5	E – Poor	• Low grade abstractions for industry • Fish absent or sporadically present, vulnerable to pollution** • Impoverished ecosystems**
	F - Bad	• Very polluted rivers which may cause nuisance • Severely restricted ecosystems

* Provided other standards are met

** Where the grade is caused by discharges of organic pollution

The Hopwood Park SUDS scheme

The most comprehensive set of chemical water quality data available on a multi-element SUDS treatment sequence comes from Hopwood Park MSA. Data were collected as part of an Environment Agency monitoring project and by HR Wallingford. The SUDS scheme was completed in 2000 and chemical monitoring data are currently available from February 2000 to November 2002 for the Coach Park SUDS system and from February 2000 to February 2001 for the HGV Park and Main Car Park SUDS systems. This includes information on ten determinands: BOD, suspended solids, ammonia, organic carbon, cadmium, copper, chromium, lead, nickel and zinc. pH data have also been collected but are not discussed further here since all values to date, including those of the most contaminated areas, lie within the bounds for natural ponds. No data are available from Hopwood Park on nitrogen and phosphorus species although both have an important influence on habitat quality and receiving water impacts.

Hopwood Park MSA provides data on the reduction of physical and chemical pollutants from three treatment trains: the heavy goods vehicle (HGV) Park, the Coach Park and the Main Car Park (Figure 2 and 3). All three treatment sequences show significant reductions in some pollutant levels but outflow concentrations are, in some instances, comparatively high compared to natural, unpolluted, background levels.

The effectiveness of the Hopwood Park treatment trains in terms of receiving water impacts was assessed by comparing the number of determinands which reached minimally impaired levels in the final stage of the treatment train (Table 5). Results for individual determinands in each of the three treatment trains are given in Appendix 1.

The results show that, in terms of receiving water impacts, Hopwood Park performed very well for heavy metals in the first 12 months of monitoring. This good performance continued for the Coach Park system over the first three years. For biochemical oxygen demand (BOD), suspended sediments and ammonia, performance was more variable, with the HGV Park treatment train performing poorly for BOD and ammonia. The short Main Car Park treatment train also showed variable performance for BOD, suspended sediments and ammonia. Note that no data are available on sediment quality.

To assess the effectiveness of the SUDS systems in terms of providing habitats for aquatic biota the proportion of the ponds in all three systems which had chemical concentrations at or close to the minimally impaired level was assessed (Table 6). Overall, pond chemical quality was good for heavy metals, nearly all waterbodies showing minimally impaired conditions for the six metals measured. Aquatic habitat quality was more variable for BOD, SS and ammonia, with about half of the sites ponds having minimally impaired conditions in terms of these determinands.

In terms of ecological quality, the plant and invertebrate species richness and PSYM measures of ecological integrity largely reflected the chemical results (Figure 4): the Coach Park system ponds had the highest ecological integrity and species richness scores, followed by the Main Car Park and HGV Park systems. However, the PSYM assessments of ecological quality show the best pond (the lowest in the Coach Park treatment train) reached only 61% of its full potential. This may be the result of impairment, or it may reflect the relatively early stages of colonisation of the ponds. Further monitoring is required therefore to accurately assess the ecological quality of the ponds.

Table 6. Assessment of effectiveness of Hopwood Park MSA treatment trains in reducing receiving water impacts

	Chemical determinands measured								
Treatment train	**BOD**	**SS**	**Amm**	**Cd**	**Cr**	**Cu**	**Pb**	**Ni**	**Zn**
Coach Park	✔	✔	✔	✔	✔	✔	✔	❒	✔
HGV Park	✘	❒	✘	n/d	n/d	✔	✔	✔	✔
Main Car Park	❒	❑	❒	✔	❒	✔	✔	✔	✔
Number of treatment trains reaching minimally impaired levels	33%	33%	33%	100%	50%	100%	100%	66%	100%

Key to table

✔ = All samples collected from the final stage of the treatment train at or below minimally impaired levels.

❒ = Some samples collected from the final stage of the treatment train at or below minimally impaired levels.

✘ = No samples collected from the final stage of the treatment train at or below minimally impaired levels.

BOD = Biochemical Oxygen Demand, SS = Suspended Sediments, Amm = Ammonia, Cd = Cadmium, Cr = Chromium, Cu = Copper, Pb = Lead, Ni = Nickel, Zn = Zinc. n/d = indicate no data available.

Table 7. Number of ponds in Hopwood Park MSA treatment trains with 'minimally impaired' chemical quality for 9 measured determinands

	Chemical determinands measured								
Treatment train	**BOD**	**SS**	**Amm**	**Cd**	**Cr**	**Cu**	**Pb**	**Ni**	**Zn**
Coach Park (3 ponds)	❒	❒	❒	✔	✔	✔	✔	✔	❒
HGV Park (1 pond)	✘	✔	✘	n/d	n/d	✔	✔	✔	✔
Main Car Park (1 pond)	✔	✔	✔	✔	✔	✔	✔	✔	✔
Total number of ponds in all treatment trains	5	5	5	4	4	5	5	5	5
Number of ponds at or close to minimally impaired levels	2	3	3	4	4	5	5	5	4
% of ponds at or close to minimally impaired levels	40%	60%	60%	100%	100%	100%	100%	100%	80%

Key to table

✔ = All samples collected from the final stage of the treatment train at or below minimally impaired levels.

❒ = Some samples collected from the final stage of the treatment train at or below minimally impaired levels.

✘ = No samples collected from the final stage of the treatment train at or below minimally impaired levels.

BOD = Biochemical Oxygen Demand, SS = Suspended Sediments, Amm = Ammonia, Cd = Cadmium, Cr = Chromium, Cu = Copper, Pb = Lead, Ni = Nickel, Zn = Zinc. n/d = indicate no data available.

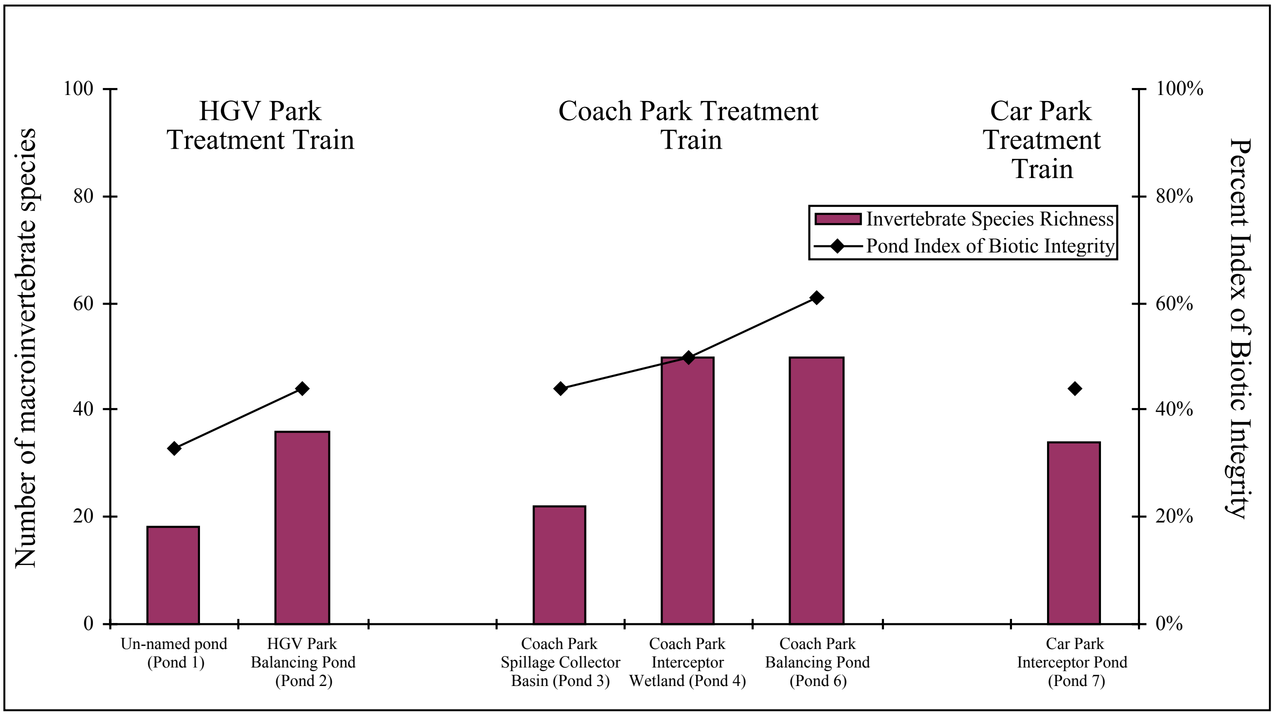

Figure 4. Pond quality in relation to position in the treatment train at Hopwood Park MSA

4.3.2 The Scottish SUDS Monitoring Programme

Data availability

Water quality data are available from several schemes in Scotland studied as part of the SUDS Monitoring Programme undertaken by the Scottish Universities SUDS Centre of Excellence (Jefferies 2001).

The studies primarily focus on individual SUDS devices rather than whole treatment trains, such as that at Hopwood Park. They therefore represent a conservative test of effectiveness since, inevitably, when SUDS components are combined in treatment trains the potential for water quality improvements is greater. Water quality data are available from eight sites including two swales, two areas of pervious surfacing and three retention ponds (Appendix 2).

Results and conclusions

An overall assessment of the effectiveness of the different SUDS devices monitored in the Scottish SUDS project was made by considering the number of determinands for which SUDS outputs exceeded minimally impaired background levels. This is a very stringent analysis and should be used for guidance only. This is because the devices in the Scottish study were all considered in isolation rather than as part of a treatment train. Thus devices which may be effective in removing one contaminant type may do poorly with other contaminants. In a treatment train such limitations may be mitigated where specific devices are incorporated to deal with a particular contaminant. However, the analysis does provide a simple overview of the effectiveness of different devices. The detailed data on which this analysis is based are given in Appendix 2.

The analysis shows that, for devices with more than one determinand measured, between 33% and 82% of the contaminants exceeded minimally impaired quality in the output from the SUDS device in terms of either the habitat quality of standing waters or the potential impact on receiving waters.

Table 8. Analysis of overall effectiveness of Scottish SUDS devices compared to minimally impaired chemical quality

Site	Swale: Emmock	Swale: West Grange	Pervious Surface: Royal Bank	Pervious Surface: NATS	Pond: Stenton	Pond: Halbeath	Pond : Linburn
Number of determinands measured	4	11	9	11	9	1	1
% meeting minimally background levels	25%	45%	67%	18%	67%	100%	0%

Table 9. Number of Scottish SUDS with 'minimally impaired' chemical quality for 9 measured determinands

	BOD	SS	NH3	Cd	Cr	Cu	Pb	Ni	Zn
Swale: Emmock	n/d	✘	✘	n/d	n/d	n/d	n/d	n/d	n/d
Swale: West Grange	✘	✘	✔	✘	✔	n/d	✔	✔	✘
Pervious Surface: Royal Bank	✔	✘	✘	✔	✔	✔	✔	✔	✔
NATS	✔	✔	✘	✘	✘	✔	✔	✔	✘
Pond: Stenton	✔	✔	✘	✘	✔	n/d	✔	✔	✔
Pond: Halbeath	n/d	✔	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Pond : Linburn	n/d	✘	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Pond Claylands	n/d	n/d	n/d	n/d	n/d	n/d	n/d	n/d	n/d
Pond Newbridge	n/d	n/d	n/d	n/d	n/d	n/d	n/d	n/d	n/d
% of SUDS at or close to minimally impaired levels	75%	50%	20%	25%	75%	100%	100%	100%	50%

Key to table
✔ = All samples collected from the final stage of the treatment train at or below minimally impaired levels.
✘ = No samples collected from the final stage of the treatment train at or below minimally impaired levels.

BOD = Biochemical Oxygen Demand, SS = Suspended Sediments, Amm = Ammonia, Cd = Cadmium, Cr = Chromium, Cu = Copper, Pb = Lead, Ni = Nickel, Zn = Zinc. n/d = indicate no data available.

4.3.3 Urban balancing ponds in the Oxford area

Ross (1999) investigated the ecology of six urban balancing ponds in the Oxford area as part of a MSc project at Oxford Brookes University (Table 9). Although the ponds studied are not part of fully developed SUDS schemes the data are of interest because both nutrient and biological data are available.

Water quality data

Water chemistry investigations undertaken by Ross (1999) focused on nutrient chemistry with measurement of total oxidised nitrogen (TON), ammonia and soluble reactive phosphorus (SRP). Measurements were made in May and August. Water column heavy metals were not measured in this study.

Mean TON values exceeded the mean for minimally impaired ponds (0.5 mg/l) in 5 out of 6 locations with only one site (Gavray Drive Pond) below the minimally impaired value, and the maximum reaching 50 mg/l in Thorney Leys Pond (Figure 5).

In contrast, mean total ammonia concentrations were well below the mean for minimally impaired ponds (0.27 mg/l) at four of the six sites, with only one site, Milton Park Pond, considerably exceeding the minimally impaired value (Figure 6).

Soluble reactive phosphorus concentrations were generally close to the mean value for minimally impaired ponds (69 µg/l). At two sites, mean values were a little above the minimally impaired mean (Figure 7).

Overall, all ponds in this study experienced nutrient concentrations which would be liable to cause biological impairment, particularly because of the generally high TON levels (Table 8).

Table 9. Urban balancing ponds in the Oxford area with ‘minimally impaired chemical quality for 3 measured determinands

Site	**Churchill Hospital**	**Milton Park**	**Bicester Park**	**Gavray Drive**	**Wendlebury M40 J9**	**Thorney Leys**
Total Oxidised Nitrogen (TON)	✘	✘	✘	✔	✘	✘
Total ammonia	✔	✘	✔	✔	✔	✔
Soluble Reactive Phosphorus (SRP)	✔	✘	✔	✘	✔	✔

Key to table

✔ = All samples collected from the final stage of the treatment train at or below minimally impaired levels.
✘ = No samples collected from the final stage of the treatment train at or below minimally impaired levels.

BOD = Biochemical Oxygen Demand, SS = Suspended Sediments, Amm = Ammonia, Cd = Cadmium, Cr = Chromium, Cu = Copper, Pb = Lead, Ni = Nickel, Zn = Zinc. n/d = indicate no data available.

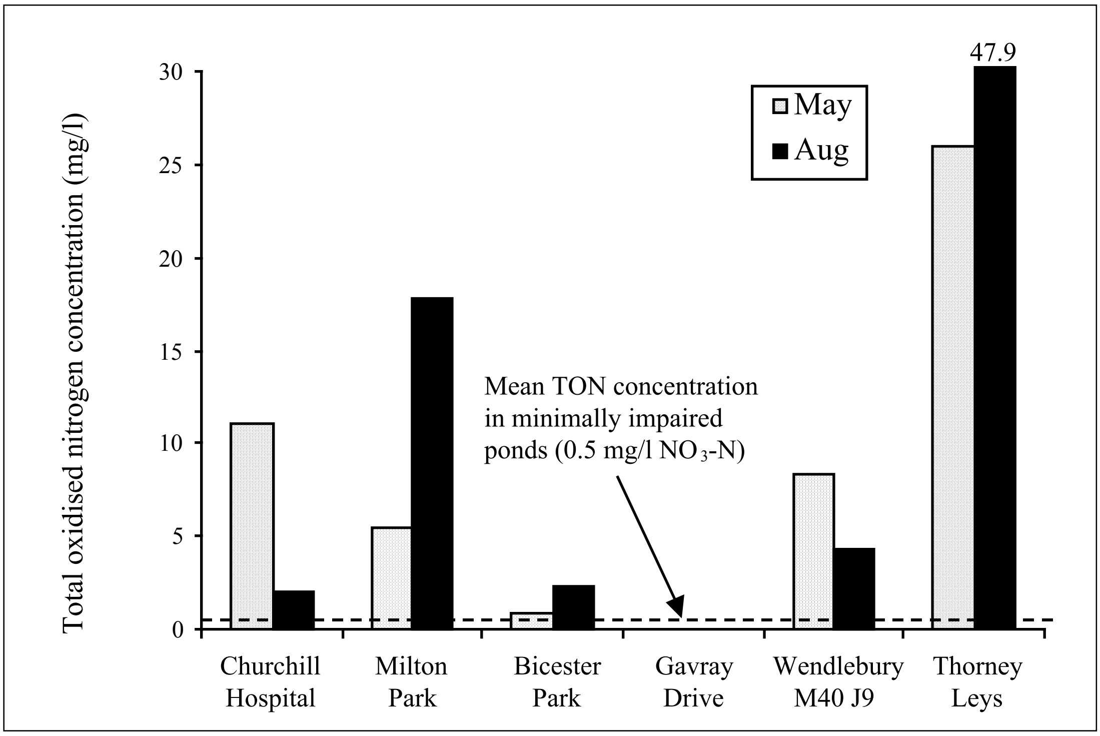

Figure 5. Total oxidised nitrogen concentrations in urban balancing ponds in the Oxford area

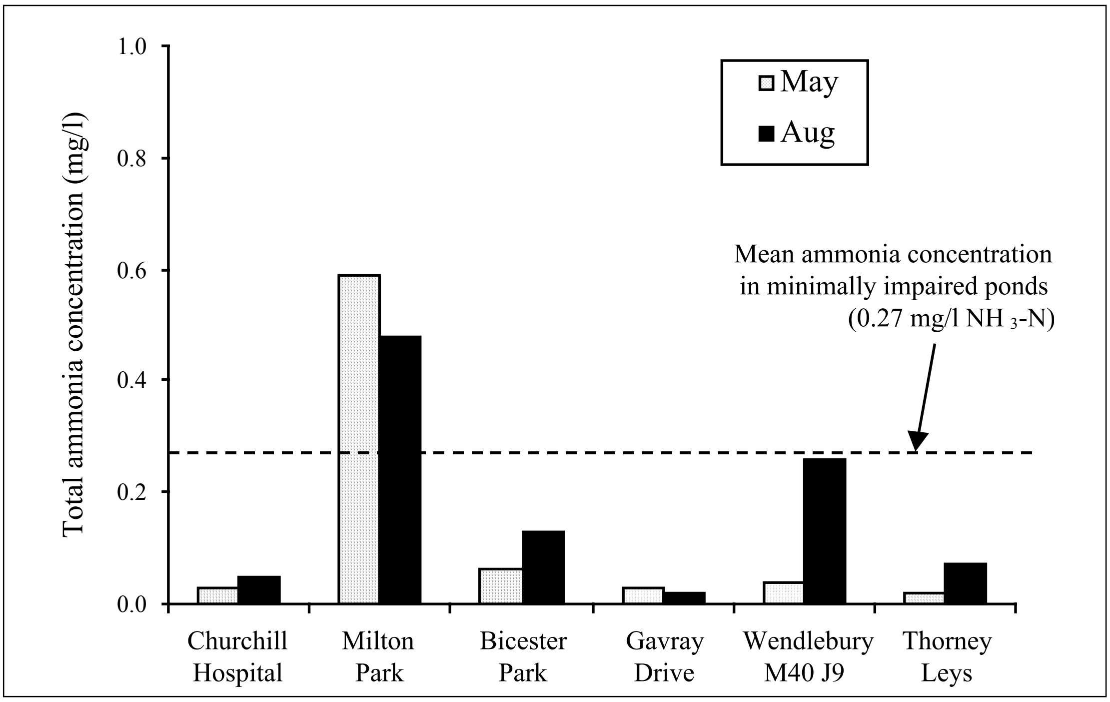

Figure 6. Total ammonia concentrations in urban balancing ponds in the Oxford area

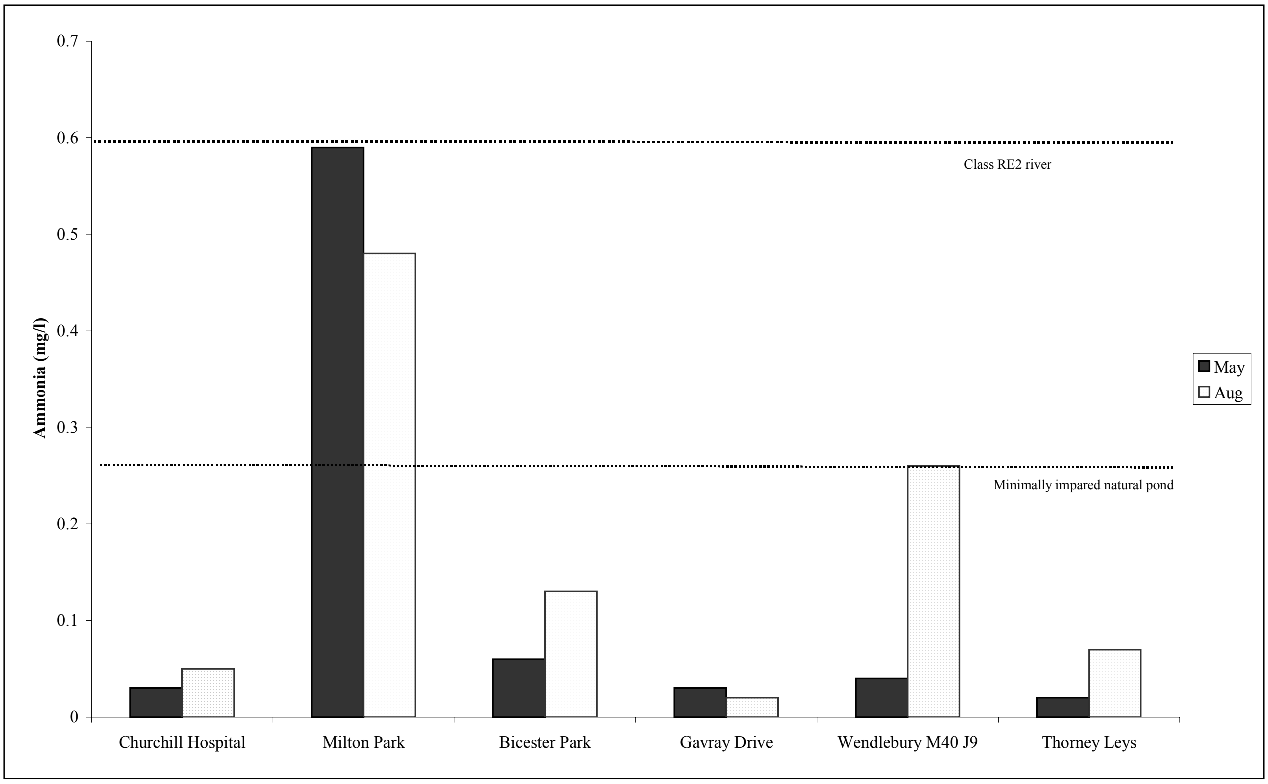

Figure 7. Soluble reactive phosphorus concentrations in urban balancing ponds in the Oxford area

4.3.3 The value of SUDS as habitat in the medium to long term

SUDS are a relatively new concept in the UK with few sites more than five years old. As a result, nothing is known of the medium or long-term value of these schemes.

Ponds and wetlands established as part of more traditional urban drainage systems (e.g. balancing ponds) have existed for considerably longer than five years and more is known about the effectiveness and quality of these habitats. However, it should be noted that, as most have operated under the pollutants loadings typical of traditional drainage schemes, they cannot be used as a guide to the medium or long-term performance of SUDS schemes which incorporate source control techniques (e.g. porous pavements, filter trenches). Indeed one objective of SUDS is to overcome the problems observed in ponds and wetlands which were installed as components in traditional engineered drainage schemes.

4.3.4 Analysis of the effectiveness of individual SUDS components using data from the American National Stormwater Best Management Practice database

The American National Stormwater Best Management Practice (BMP) database (http://www.bmpdatabase.org/) contains an extensive body of monitoring data on the effectiveness of individual SUDS devices, known in North America as Best Management Practice structures.

Information from this database was analysed using the same approach as adopted for Hopwood Park MSA, the Scottish SUDS project and urban balancing ponds in the Oxford area, comparing SUDS device outputs with minimally impaired levels.

Some caution needs to be exercised in the use of the database, owing to the wide range of climate zones from which the data are available. However, given the large body of data available, and the generality of biotic responses to pollutants, the information contained in the stormwater BMP database provides an unrivalled range of data.

Information is available from the database on about 20 determinands including nutrients and heavy metals. The largest body of data is available for retention basins (wet ponds) and wetlands. More limited data are also available for a range of other SUDS devices including filters, dry detention ponds, grass strips and swales. Detailed analysis of the effectiveness of different SUDS devices for each determinand for which sufficient data are available given in Appendix 3.

The effectiveness of North American BMP structures is summarised in Table 10. Devices were treated as achieving 'minimally impaired' status if the mean effluent concentration was within 50% of the baseline values identified in Table 4 (see Appendix 3). Overall, 57% of SUDS devices produced outputs which were within 50% of the minimally impaired mean values listed in Table 4 for ponds. This value disguises a high degree of variability. Thus, 91% of sites produced zinc concentrations which were reasonably close to minimally impaired levels whereas only 42% of sites produced reasonably good water quality in terms of Total Phosphorus. Effectiveness for BOD and cadmium was particularly low, although sample sizes were relatively small for these determinands.

Table 10. Effectiveness of North American BMP structures in providing 'minimally impaired' water quality (see Appendix 3 for detailed data)

		Chemical determinands measured								
Treatment train	**BOD**	**SS**	**TON**	**TP**	**Amm**	**Cd**	**Cr**	**Cu**	**Pb**	**Zn**
Number of devices for which data are available	10	36	15	38	20	11	7	15	26	22
Number of SUDS devices with water quality outputs at or close to minimally impaired levels	2	24	11	16	14	2	3	12	17	20
% of total	20%	66%	73%	42%	70%	18%	43%	80%	65%	91%

BOD = Biochemical Oxygen Demand; SS = Total Suspended Solids; TON = Total Oxidised Nitrogen; TP = Total Phosphorus; Amm = Ammonia; Cd = Cadmium; Cr = Chromium.

5. SUMMARY AND APPRAISAL OF AVAILABLE ECOLOGICAL MONITORING DATA

5.1 The value of SUD Systems as Habitats

5.1.1 The habitat value of different SUDS components

The primary value of SUD systems as habitats comes from the ponds and wetland components. Other SUDS devices (e.g. filter strips) are less likely to provide valuable habitats, although well-designed swales may contribute to biodiversity, particularly if low down in the treatment train.

The importance of ponds and small wetlands generally has, until quite recently, been overlooked. Thus although ponds have long been popular, and urban drainage engineers have been recommending their construction on nature conservation grounds for some years (e.g. Hall *et al.* 1993), it is only very recently that data have become available with which to assess the comparative importance of small standing waters and other freshwater habitats at a catchment level (see Section 2.4).

These studies suggest that the role of SUDS as new still water habitats may be as important as their role in protecting receiving waters, given that it is now becoming clear that ponds may support as large a proportion of total aquatic biodiversity, at a catchment scale, than rivers or streams.

Land around the SUD scheme could also make a useful contribution to habitat conservation, particularly in areas where there are opportunities to manage SUDS landscapes by establishment of new habitats, such as scrub and unimproved grasslands. Most importantly, non-intensively managed catchments around ponds and wetlands can provide (a) uncontaminated water sources and (b) high quality terrestrial habitat for aquatic biota which have both a freshwater and terrestrial component to their life cycle (e.g. amphibians, many aquatic insects).

Although SUDS ponds and wetlands have the potential to provide useful habitats, their position in the treatment train has a considerable influence on their value. As can be seen from data collected at Hopwood Park MSA, contaminant levels may be very high at the start of treatment. For example, total ammonia concentrations in the Hopwood Park Coach Park Interception Pond on 8th February 2000 were six times greater than the mean for minimally impaired ponds. On the same date, however, ammonia levels in the lowest pond in the Coach Park treatment train were well below the minimally impaired mean. See appendix 1.

5.1.2 Factors determining the quality of SUDS ponds as habitats

The ecological quality of SUDS ponds is determined by the same mechanism as all other ponds, with the key factors being (i) water quality, (ii) proximity to other wetland habitats and (iii) physical structure. The highest quality ponds are those with unpolluted water and, although the effects of pollutants are complex, in general the cleaner the water in a pond, the higher quality will be its biotic communities. Proximity to other wetlands (rivers, lakes, ponds, marshes) also affects pond quality since there is considerable exchange of biota between aquatic habitats: thus ponds on river floodplains tend to be particularly species-rich because of the network of aquatic habitats in such areas. The physical structure of ponds also affects their value as habitats, and well-designed ponds which incorporate, for example, extensive drawdown zones and sub-basins of varying degrees of permanence, are likely to support more species than those which are conventionally shaped.

5.1.3 Biological observations of the ecological quality of SUDS ponds

Observation on the biological quality of new SUDS ponds has been made in two main areas: central Scotland and at Hopwood Park, near Birmingham. Ponds in these studies have been found to be mainly of moderate to high ecological value (on a four point scale: low, moderate, high, very high) and of better quality than the average for the lowland British landscape. New SUDS ponds probably, therefore, lie somewhere between the impacted sites typical of most of the British countryside and high quality, minimally impaired, ponds characteristic of catchments free from significant pollutant impacts.

Within SUDS schemes (e.g. at Hopwood Park) the limited data available supports the hypothesis that the highest quality ponds ecologically are those least exposed to pollutants. Thus mildly polluted waters may support a limited range of common and tolerant species but to maintain the 'optimal' structure and function of pond communities, clean water is required. Thus at Hopwood Park, data so far suggest that the lower in the treatment train the waterbody was located, the higher the species richness of the plant and invertebrate assemblages and the higher the Index of Biotic Integrity values (Figure 46). It is also worth noting that the pond at Hopwood Park with by far the greatest macroinvertebrate species richness was the Amenity Pond, fed by roof water, which was not part of the three main contaminated water treatment trains.

5.1.4 Water quality data

Monitoring of chemical water quality in SUDS ponds now provides a considerable body of data with which to assess the value of SUDS as habitats. The present report has made a preliminary assessment of these data comparing output water quality with a range of baseline values equivalent to minimally impaired conditions for still and flowing freshwater habitats. This approach contrasts with the traditional approach to assessing SUDS effectiveness by comparing input and output levels to give a percentage efficiency of contaminant removal. Removal efficiency provides little information about the ecological effectiveness of devices since a high percentage removals does not guarantee that levels will be reduced to below biological impact thresholds.

Overall, water quality monitoring of SUDS devices clearly shows the considerable variations in water quality of the outputs generated from them. Thus for some determinands in some locations, water quality reaches minimally impaired (sometime pristine) levels. Elsewhere the same determinands are well outside acceptable standards. At present, therefore, implementation of SUDS devices does not guarantee that pond and wetland habitat quality will reach a particular chemical standard.

An important proviso of the analysis of SUDS device output water quality presented here is that, with the exception of the Hopwood Park study, none of the available data describe treatment trains with successive improvement of water quality in a range of source control devices.

The Hopwood Park dataset shows that there is still unpredictable variability in chemical quality in SUDS devices. Thus, two of the three treatment trains generally produced good quality outputs for the determinands measured. However, the third (the HGV Park treatment train) was generally of quite poor quality. It should also be noted that no measurements of nutrients are available for Hopwood Park although biological evidence strongly suggests that within waterbody nutrient levels are high in some parts of the system. This is consistent with the biological data from the HGV Park treatment train which had lower values for invertebrate species richness and overall biotic integrity than the other two treatment trains.

5.1.5 Other factors affecting SUDS habitat quality

As noted above, study of wider countryside ponds suggests that the physical structure and

location of SUDS ponds and wetlands play a role in determining their ecological quality. To date, however, little use has been made of modern pond design techniques in the structure and location of SUDS ponds (e.g. see SEPA 2000). As a consequence, it is impossible to assess the extent to which good pond design will increase the ecological value of SUDS ponds, although it is likely that considerable improvements could be made.

There is also a serious alien plant problem in SUDS ponds and wetlands. In PCTPR's study of Scottish ponds, 12% of plant species recorded were aliens, compared to 2% in lowland countryside ponds (Williams *et al.*1998, PCTPR 2000). In many cases, deliberate or accidental planting of alien plants has occurred during the construction of SUDS systems. Raised awareness of the problem of alien plants is needed to reduce this problem. However, in the longer term, even if all artificial introductions are stopped, SUDS will continue to be highly exposed to alien plants because of their proximity to urban areas. Ponds in urban areas generally have higher proportions of alien plants than more rural ponds probably because of the people's tendency to dump plants from garden ponds into the wild. Therefore, managing alien plants in SUDS ponds is likely to be a long-term problem, and there is a considerable possibility that SUDS ponds will themselves become a significant source of invasive alien plants and will enhance the spread of these species into the wild.

The occurrence of *Crassula helmsii*, New Zealand Pigmyweed, in SUDS ponds is particularly worrying. In Scotland, *Crassula* occurred in over a quarter of the SUDS ponds surveyed by PCTPR. (PCTPR 2000). This plant is currently spreading rapidly in Britain, causing very severe damage to natural plant communities in many areas and threatening some of our rarest plant species. It is estimated that at least £3 million is needed to control *Crassula* in the UK and many organisations have argued that the sale of this plant should be banned (Plantlife 2000). *Crassula* is unfortunately commonly transported to sites as seeds or small plants in the contaminated soil of other pot plants from aquatic plant suppliers, and this is likely to have been the source in all of the SUDS ponds.

In addition to the occurrence of alien species, a range of other plants that might be considered inappropriate were recorded from the Scottish SUDS ponds. These included:

1. Variegated cultivated varieties of native species including Reed Sweet-grass (*Glyceria maxima*).
2. Nationally uncommon or rare species, introduced without consultation with national authorities, well beyond their natural range. This included the Red Data Book species Hampshire Purslane (*Ludwigia palustris*) which is native only to the New Forest in Hampshire - but was found planted in Scotland.
3. Native species which were 'out of place' e.g. Arrowhead (*Sagittaria sagittifolia*) which is a common wetland species but would naturally be confined to rivers and closely associated floodplain pools.

Overall, therefore, there is a need to minimise unnecessary planting, beyond that needed for functional purposes, and as far as possible allow natural colonisation to occur, possibly with some aftercare to prevent the development of monodominant Bulrush (*Typha latifolia*) stands, where these are considered undesirable. In addition, where planting is needed there is a need to ensure that only native plants of local provenance are used.

5.2 Potential for the protection of receiving waters

SUDS offer considerable potential for the protection of receiving watercourses, and the observed reductions of pollutants reaching receiving waters must in general be beneficial. However, to date, analysis of the benefits of SUDS in protecting or improving urban watercourses, or the wider environment, have been undertaken in only the most general terms.

Consideration of SUDS outputs in relation to minimally impaired levels shows that urban drainage, after passing through SUDS devices, often still has considerable potential to cause damaging environmental impacts. In these cases, unless the watercourse is already of poor quality, or there is further dilution by the receiving water, some impairment of biological assemblages is likely to occur.

5.3 Unresolved issues needing further research

5.3.1 Treatment train research

It is widely recognised by researchers and practitioners working on SUDS schemes that, to be most effective, SUDS systems must consist of treatment trains with source control devices. In practice, however, as stressed above, very little of the monitoring data available on urban surface water management come from treatment train devices. Such data are essential if the true potential of SUDS is to be understood and developed. Key issues to investigate are:

- the extent to which different devices can operate together to effectively remove major contaminants, particularly the number and variety of devices in the treatment train which are needed to ensure high quality outputs
- the efficiency limits of treatment train systems – can SUDS ponds and wetlands provide conditions which are equivalent to those of minimally impaired waterbodies, or will some reliance on further dilution always be required?

Ultimately there is a need to establish the limits of effectiveness of SUDS: can all pollution problems be contained using SUDS or will it ultimately be necessary to remove the sources of contaminants completely?

***Recommendation*:** Experimental treatment train systems should be established which are representative of different urban catchment types (e.g. surface runoff vs infiltration dominated catchments, gentle slopes vs steep slopes etc).

Sites need to be developed where experimental observation can be designed free from the constraints of practical commercial developments. Although studies of commercial sites have provided valuable data, the lack of experimental control, inability to replicate experiments, inability to test particular pollution scenarios and often compromised designs (on the ground of cost) means that these studies do not provide the most effective way forward in understanding and developing SUDS.

Overall, one or two national facilities are needed at which researchers from different disciplines can undertake co-operative research to increase understanding of SUDS schemes. Critically, such facilities should have a very strong practical input from SUDS designers since there is little point in experimenting on SUDS systems that cannot be created in reality.

5.3.2 Long-term monitoring

Related to the need for research on well-designed test bed sites, is the need for long-term monitoring. All areas of SUDS performance need to be considered in the medium to long-term: at present, virtually no data are available on the performance of SUDS over more than two or three years. There is, as a result, very little information with which to assess whether SUDS systems have an age limit after which they become relatively ineffective, or how management techniques (e.g. dredging, reed cutting) can be used to mitigate this.

Establishing long-term monitoring studies is, however, difficult mainly for the practical reason that most research grants last for a maximum of three to five years whereas data on SUDS performance are required on a much longer timescale.

***Recommendation*:** Two or three long-term studies, collecting both biological and water quality data, are needed from SUDS systems with multi-element treatment trains. In particular, work at Hopwood Park MSA should be continued, and funded securely, as this is the only site in the UK with multiple element treatment trains. Sites in Scotland with treatment trains should also be identified and monitored using similar techniques to those applied at Hopwood Park.

5.3.3 More information needed on the contribution to aquatic biodiversity of different urban waterbody types

At present, little is known about the comparative importance of still and flowing waters for biodiversity in the urban environment. Studies of catchment scale patterns of aquatic biota are, therefore, necessary to underpin the objectives of SUD schemes. This is because, at present, creation of new habitats as part of SUD schemes is mainly seen as a desirable 'add-on' compared to the major objective of protecting receiving waters. Research on catchments suggests that the recreation of small standing waters may be as important for biodiversity protection as improving, or preventing further damage to, receiving waters.

5.3.4 Water quality thresholds for still and running water biota

The present study has made a provisional investigation of water quality thresholds in SUDS outputs for aquatic biota. In addition, a new approach has been adopted which has not previously been widely applied: that of comparison to baseline, minimally impaired or pristine, concentrations. This is an important issue, since it provides an objective benchmark against which the effectiveness of SUDS schemes can be judged.

***Recommendation*:** It is recommended that a more extensive review and analysis of available data, and benchmarking techniques, should be undertaken to produce a fuller review of SUDS systems water quality outputs. Ideally this needs a multi-disciplinary team, particularly involving aquatic ecotoxicologists, and an industry-wide consultation to ensure that recommendations are widely accepted.

5.3.5 Practicality of improved physical designs

Improved physical design of SUDS is essential if their ecological quality is to be maximised. At present, no such designs have been implemented and there are likely to be significant practical issues in doing so.

***Recommendation*:** It is recommended that two or three demonstration sites are identified where advanced pond design techniques can be applied, with design guidance provided by the Ponds Conservation Trust (who hold most knowledge and data in this area). Ideally these would be located at the same sites adopted for experimental treatment train research noted above.

5.3.6 The effect of SUDS on receiving waters

Understanding of the ecological effectiveness of SUDS is hampered by the almost complete absence of information on receiving waters which have been protected by SUDS schemes. To date, there have been no well-designed ‘before and after’ studies (known technically as BACI studies: Before, After, Control, Impact) of receiving water protected by SUDS with which to verify the improvements which are to be expected with the use of SUDS. Unfortunately, collecting such data is likely to prove challenging because of the difficulties of finding suitable study sites. Studies of the effectiveness of SUDS are also confounded by the effects of physical channel modification in receiving waters.

Two main types of study are possible to assess the protection given to receiving waters by SUDS:

i. studies of water quality, from which biological effects may be inferred
ii. direct observation of the biota before and after SUDS installation to determine whether effects have occurred.

In both cases, control receiving waters are also needed to determine whether changes would have occurred anyway in the absence of the development. Ideally, too, studies of the effects of the urban impacts without mitigation from a SUD system would also be useful, given the scarcity of such data in typical British greenfield sites, which already experience significant impacts from diffuse pollutants (particularly sediments, nutrients and biocides).

Given the technical difficulties of studying the ecological effects of SUDS in receiving waters directly, there is also a strong case for catchment modelling studies of water quality effects assuming different levels of SUDS implementation, and different levels of contamination in receiving waters.

The over-riding reason for this is that it may transpire that, counter intuitively, the maximum ecological benefits to be derived from SUDS may come from improvements in the *still water* aquatic environment. With the standing water component of SUDS it is possible to create near pristine chemical quality in some waterbodies. For receiving waters the best that can often be achieved is to prevent further deterioration, even where SUDS outflows are of the highest chemical quality, because of the existing condition of receiving waters.

***Recommendation*:** Well-designed BACI (Before, After, Control, Impact) studies of SUDS systems are developed. Such studies are relatively complex (cf. work on river restoration) and are unlikely to be possible at more than one or two sites. Secondly, catchment water quality modelling studies are required to assess the likely ecological effects of SUDS at a catchment scale, in terms of both standing water quality and receiving water quality. Large bodies of data are available with which to make realistic catchment assumptions about existing water quality in typical locations where SUDS are implemented.

5.4 The potential ecological benefits of SUDS

The overall ecological benefits of SUDS schemes can be summarised under two main headings: value of created habitats and protection of downstream receiving aquatic habitats.

5.4.1 Benefits deriving from creation of new, mainly aquatic, habitats

Overall, SUDS waterbodies are likely to be beneficial for aquatic biodiversity if:

1. SUDS schemes include some clean waterbodies and there is full implementation of treatment trains.
2. Improved physical designs are generally adopted.
3. Planting of non-native species is banned, and there are post establishment checks to ensure that invasive species have not been introduced by accident.

The spreading of non-native species around the landscape is potentially the single biggest adverse impact of SUDS schemes. Benefits gained from SUDS schemes could be significantly reduced by continued contribution to the spread of invasive non-native plants.

A provisional set of threshold concentrations for key pollutants indicates the treatment levels which should be achieved if SUDS are to truly replicate ‘green field’ runoff quality. Clearly, SUDS ponds which have concentrations of chemicals above the threshold for particular species (e.g. greater than 0.1 mg/l Total Phosphorus for nutrient sensitive plants) will not support those species.

5.4.2 Benefits deriving from protection of downstream habitats

The main benefits of SUDS schemes for receiving waters are:

1. Containment of the “urban footprint”.
2. Restoration of aquatic habitat in areas where natural stream and river drainage network has been damaged.

At present, urban runoff creates highly damaging impacts on the aquatic environment. Widespread implementation of SUDS schemes, particularly retrofitting, is likely to be needed if there is to be a significant improvement in the quality of urban watercourses and still waters receiving urban runoff.

6. BEST PRACTICE GUIDANCE ON MAXIMISING ECOLOGICAL POTENTIAL OF SUDS THROUGH GOOD DESIGN

6.1 Introduction

This section gives best practice guidance on designs for SUD schemes to maximise their ecological value. Section 6.2 considers the design and management of SUDS to maximise their value as habitats; Section 6.4 provides a short summary of best practice in SUDS design and management techniques for controlling downstream impacts.

The guidance given here on pond and wetland design is broadly similar to that previously given in SEPA (2000). Although that document was prepared before the review of research described, the conclusions are consistent with the present review.

6.2 Design and management of SUDS to maximise their value as habitats

6.2.1 Ponds

Introduction

As noted above, the three key factors which influence the value of ponds and small wetlands as habitats are water quality, physical structure and proximity to other wetland habitats (Williams *et al.* 1999). Manipulation of these three factors is a key requirement for maximising the value of SUDS ponds and wetlands.

Pond designs can also be considerably improved by ensuring that ponds are created in two phases: Phase 1 establishes the basic shape and structure of the pond with a follow-up Phase 2, 1-2 years later to undertake fine-tuning of the scheme. Examples of small-scale refinement that can be incorporated in Phase 2 which, although simple and cheap, add considerably to the habitat value of sites include:

- Addition of small scale topographic features (e.g. reprofiling of pond margins to increase the extent of seasonal drawdown zones, addition of small temporary and semi-permanent pools around the main SUDS waterbodies)
- Maximising the potential of unplanned habitats that occur on most sites, such as runoff from grassed slopes and natural seepages which can provide water for new ponds and pools.

Water quality

Ponds and wetlands in SUD schemes are, by their nature, likely to be exposed to pollutants. Since pollution of ponds and wetlands inevitably reduces their value as habitats, careful balancing of the habitat creation and pollution control functions of SUDS schemes is necessary.

Two main approaches to working with contaminated water to maximise the wildlife value of ponds and wetlands are available: (i) design the waterbodies or wetlands to maximise water quality in some areas and (ii) ensure that there are extensive shallow areas which are generally less affected by pollution than deeper water habitats.

In practice, four main techniques are available for maximising water quality:

(i) avoid mixing clean and contaminated water by keeping clean water (e.g. from roofs) separate from contaminated water (e.g. from car parking areas)

(ii) fully implement the SUDS treatment train: maximise source control and incorporate sufficient pools in the treatment train to give progressively cleaner water

(iii) Prevent nutrients leaching into ponds in the construction phase of SUDS projects by minimising soil runoff from surrounding slopes and avoiding use of fertilisers in the ponds' catchments

(iv) Use water draining from clean, non-urban, parts of SUDS sites (e.g. surrounding grassland) to provide additional high quality, unpolluted, ponds and wetland habitats. These additional ponds, which are not part of the SUDS treatment process, should not receive inputs from the SUDS system although they may ultimately drain into it. Clean water pools should be located above the level of the main SUDS ponds, and should be surrounded by semi-natural vegetation (e.g. rough grassland, scrub, woodland) which has been established on low nutrient status soils.

To work with polluted water, the main strategy for maximising the value of pond and wetland habitats is to create areas of shallow water which can support a range of wildlife that is less vulnerable to the effects of pollutants. Shallow waters and wetlands are dominated by emergent plants and air breathing animals, which are generally more tolerant of pollutants than submerged aquatic plants and those animals which live permanently under the water (such as mayfly larvae, dragonfly larvae and fish). In contrast, open water habitats very quickly become impacted by pollutants, particularly through the impact of eutrophication on algae and macrophytes.

Proximity to existing wetland habitats

Ponds in close proximity to existing wetland and freshwater habitats are generally richer in species than more isolated sites. This reflects the ability of wetland plants and animals to move between sites either actively (e.g. by flight) or passively (in flood water, or by wind). SUDS ponds located near to existing wetlands therefore have the potential to:

(i) colonise naturally very rapidly (see Section 4.2.1 above on natural colonisation rates)

(ii) add to the complex of habitats used by species found in the existing wetlands or aquatic habitats, thereby strengthening populations.

Where possible, opportunities should be taken to locate SUDS ponds close to locations known to support Biodiversity Action Plan species (e.g. Great Crested Newts, Water Voles, bats). Where populations of such species are known to occur locally, specialist advice on the requirements of these groups should be sought to ensure that the design (and, if possible, location) of SUDS ponds is beneficial for these groups.

New ponds and wetlands should not damage existing wetland habitats, and particular care should be taken to avoid digging up existing small or inconspicuous wetlands (flushes, wet grassland, springs). Linked to this, it is important to avoid incorporating existing ponds or wetlands into SUD schemes – either as functional wetlands or as receiving waters, unless there are certain to be improvements in environmental quality.

Waterbody design

Modern pond design techniques emphasise the creation of multiple basins to maximise habitat diversity. This contrasts with the traditional belief that maximising physical habitat diversity in ponds is achieved by creating a variety of water depths. In practice however, to achieve physical habitat diversity it is necessary to provide separate permanent, semi-permanent and seasonal waterbodies since it is the temporary-permanent hydrological gradient that is the dominant physical environmental driver shaping freshwater assemblages.

The main physical structural requirements of high quality ponds are:

- Multiple basins with mosaics of permanent, semi-permanent and seasonal ponds and pools (down to 1 m^2 in area and 5 cm deep)
- Very gently shelving margins around a high proportion of the waterbody edge

- Hummocky, undulating (not engineering-smooth) margins.

The creation of multiple basin pond complexes also assists in separating clean water from contaminated water as far as possible.

Design of pond margins

Pond margins should have large areas with slopes that are broad and very shallow (1:50) to maximise their wildlife value. Ideally these should be wet in winter and exposed by drawdown in summer. Typically, most newly created SUDS ponds currently have much more steeply sloping margins, with slopes of 1:10 to 1:5.

For SUD schemes, the creation of broad gently sloping margins can appear problematic because of (i) the additional space needed for such ponds, and (ii) potential conflicts with pond volume requirements. However, the benefits of shallow margins are considerable since maximum plant and animal diversity occurs in very shallow water (i.e. less than 15 cm deep).

As noted above, shallow water habitats, which are rich in vegetation are also generally less vulnerable to pollution impacts, compared to deeper water. For this reason, in terms of habitat design, the ponds receiving the most contaminated water should generally have large areas of shallow water habitat; water quality improvements down the treatment train means that, if deeper, steeper sided ponds are required, these should be located further down the treatment train.

Landscaping

Standard landscaping practices around SUDS ponds have considerable potential to add pollutants to the system. To prevent this:

- Landscaping within the catchment area of SUDS ponds should avoid the use of nutrient rich topsoil as far as possible. Linked to this, topsoil should never be added to the margins, or other areas, of waterbodies to encourage plant growth.

- During the SUDS establishment phase, runoff from bare soils should be minimised. For example: (i) green cover on slopes should be rapidly established (ii) base-of-slope trenches should be introduced to retain the inevitable runoff sediments, (iii) construction should be timed to avoid autumn and winter when high runoff rates are to be expected.

- The introduction of planting schemes, which need intensive management with biocides or fertilisers should be avoided. For example: slow release fertiliser applied to flower and shrub beds at Hopwood Park MSA are thought to have caused algae and duckweed problems in downstream treatment ponds.

Planting practices

Some planting of tall emergent species will be a requirement within most SUD schemes to ensure that they achieve their functional objectives. However, much planting of marginal, floating leaved and aquatic plant species in the SUDS ponds appears to be unnecessary in terms of either functioning or visual affect, and appears to have been done merely to help the ponds ‘colonise rapidly’. In practice it would be better to omit such planting, since ponds will colonise naturally, and the new pond stage is ecologically valuable in its own right in that it supports species which are not seen at later stages of colonisation. Planting up also fills up space in ponds that could otherwise be exploited by self-colonising local species, and in doing so reduces the potential ecological value of the pond.

In general it would be better to focus effort into developing a good design and location which will encourage natural colonisation of an appropriate range of plants at an appropriate rate.

Landscape consultants often request standard lists of suitable wetland plants. These specifications generally bear little resemblance to natural pond floras, and tend to generate a standard 'SUDS pond plant community', instantly recognisable and often out of place in the local environment. Rather than making standard specifications, consultants should develop local lists for different parts of the country comprising species found within 30 km of the development site. Such lists can easily be compiled in most areas from relevant country floras.

One of the most worrying findings of investigation of existing SUD schemes is the occurrence of *Crassula helmsii* in about one third of all SUDS ponds. This is a serious problem because the species is so highly invasive. The risk is that, through accidental transfer, SUDS ponds become a vector that helps to transfer *Crassula* around the country and encourage its spread into other semi-natural ponds and wetlands. In terms of wetland plants, the following species should be avoided completely, and contractors should have specific instructions to ensure that non-native aquatic or marginal plants are not included in planting schemes. There is particular risk from the species listed in Box 1.

Box 2 gives recommendations on the main issues associated with planting up SUDS ponds.

Box 1. Invasive alien wetland plants which pose a high risk to the environment. These plants should be excluded from all SUDS planting schemes

- New Zealand Pigmyweed (*Crassula helmsii*)
- Parrot's-feather (*Myriophyllum aquaticum*)
- Floating Pennywort (*Hydrocotlye ranunculoides*)
- Water fern (e.g. *Azolla filiculoides* and close relatives).

Box 2. Issues associated with planting-up SUDS ponds

- Ensure that the contract for *all* planting up of SUDS schemes specifies the requirement for 'native species of local provenance'
- Include only common species (unless the scheme is part of a recognised conservation project to protect populations of a particular uncommon plant)
- Include only species which are characteristic of local ponds
- Focus particularly on the more inconspicuous, but ecologically valuable, aquatic grasses, especially Creeping Bent and the Sweet-grasses (*Glyceria* species) which provide good invertebrate habitats
- Ensure that an experienced botanist assesses planting schemes before projects are signed-off to check what has actually been planted (as opposed to specified). Check again for the presence of invasive species after one year
- Contractors should be responsible for removing any unspecified material and make good any damage incurred to other plants
- Where possible work with local plant suppliers to develop appropriate ranges of native plant species of local provenance
- Check aquatic suppliers' premises in order to ensure that highly invasive species are not rampant and "growing wild" in their propagating areas (as has been observed at some sites).

Note that it is rarely worthwhile planting submerged aquatic species as they normally colonise by natural processes if the pond is suitable for them. If they do not occur this is usually because the waterbody is unsuitable.

Management

There have been exceptionally few published studies of the effects of management on the nature conservation value of ponds, either in natural conditions or as part of SUDS schemes. As a consequence, most recent pond management advice has emphasised the need for caution in traditional invasive management techniques (dredging, removal of vegetation) which often damage the wildlife value of ponds (Williams *et al.* 1999). Likewise, there is little available information on the frequency with which SUDS ponds and wetlands should be managed to retain their optimum function.

Management of SUDS ponds is more complex than normal pond management for wildlife purposes because it needs to combine removal of accumulated sediments and pollutants with retention of wildlife habitats. Because of this, SUDS pond management strategies will largely be determined by the need to periodically remove sediments and vegetation for water quality and functional purposes, rather than the habitat maintenance requirements.

There are, at present, no data to propose an optimum dredging frequency for wildlife: in practice, in ponds that are relatively free from pollutants, the longer the pond can be left undisturbed the better. In ponds that are exposed to a relatively high pollutant burden, removal of sediments may help to improve water quality and increase the value of the pond as a habitat. In such conditions, frequent dredging may be beneficial. This is especially true where it is possible to dredge out polluted sediments from deeper water areas, whilst leaving shallower wildlife-rich edges with little accumulated sediment intact.

Schedules for removing vegetation are also difficult to define. There is no ideal amount of vegetation from a wildlife perspective, although more is often better. For example, the greater the proportion of the pond that is vegetated the more plant species it will support. Where it is necessary to harvest plants to remove pollutants, it is probably best to accept the process.

In SUDS ponds that are well-protected from the contaminants, either by effective source controls or as a result of their position low-down in the treatment train, it may be possible to incorporate grazing. Many high quality ponds are grazed by low densities of cattle, sheep or horses (the equivalent of 1-2 cattle per hectare), with the low intensity disturbance that this causes creating physically varied, open ponds (i.e. not dominated by shrubs and trees or emergent plants). Where wetland vegetation may carry a considerable burden of pollutants, grazing may not be appropriate for animal welfare reasons. In many urban locations, grazing will usually be impossible to organise.

Where grazing is not possible, new ponds may become dominated rapidly by invasive native plants, particularly Common Bulrush (*Typha latifolia*). As it is not desirable for all new ponds to be bulrush dominated, it should be ensured that in the first five years, whilst vegetation is establishing, plants are controlled on at least some of the ponds in a SUDS complex. After this time, ponds can usually be allowed to develop naturally, recognising that, unless the margins are occasionally managed, they are likely to become dominated by trees and shrubs.

Box 3 summarises the advice above.

Box 3. 20 ways to maximise the value of SUDS ponds as wildlife habitats (SEPA, 2000)

This box describes ways to maximise the nature conservation value of new ponds in Sustainable Urban Drainage Systems (SUDS). Clearly SUDS schemes vary considerably in terms of their functions and constraints and not all of the features will be viable in all schemes. However, include as many as possible.

1. Maximise water quality reaching pond basins by fully implementing SUDS treatment sequences to prevent or ameliorate the export of pollutants into pond basins.
2. Where possible locate SUDS basins in, or adjacent to, non-intensively managed landscapes where natural sources of native species are likely to be good.
3. In particular, locate water treatment ponds near to (but not directly connected to) other wetland areas e.g. natural ponds, lakes and river floodplains. Plants and animals from these environments will be able to colonise the new ponds, and potentially recolonise after pollutant influx events.
4. Create habitat mosaics with sub-basins of permanent, temporary and semi-permanent ponds; vary these in size (from 1 ha down to $1m^2$) and depth (1m down to 5 cm).
5. Ensure that some ponds, or parts of basins, are not exposed to the main pollutant burden allowing many more sensitive animals and plants to exploit some parts of the site.
6. Create small pools around the margins of larger ponds which are fed by clean surface runoff from non-intensively managed grassland, scrub or woodland on the basin sides.
7. Create shallow grassy ponds along swales and floodways, particularly towards their cleanest ends - pools just 1 or 2 metres across and only 10 cm deep will be valuable for wildlife.
8. Maximise the area of shallow and seasonally inundated ground dominated by emergent plants - these are generally more tolerant of pollutants than submerged aquatic plants. To do this, create *very* low slopes at the water's edge (e.g. 1:50) and try to avoid fixing pond levels at a predetermined height.
9. Create undulating 'hummocky margins' in shallow water; these mimic the natural physical diversity of semi-natural habitats.
10. Avoid smoothly finished surfaces as traditionally used in ditch, drain and river engineering; although giving an impression of tidiness, they provide less physical habitat diversity for plants and animals.
11. Plant trees, scrub and wet woodland around ponds: these provide a valuable habitat for amphibians; a food source for invertebrates and tannins from decaying bark will help to suppress algal blooms.
12. Encourage development of open, lightly shaded and densely shaded areas or pools; this will add to the diversity of habitats available.
13. Encourage or install dead wood in ponds (anchor securely where necessary). Dead wood provides firm substrates for pond animals and can provide egg laying sites for dragonflies and other animals.
14. Encourage the development of mosaics of marginal plants (rather than single species stands) to maximise habitat structural diversity.
15. Avoid planting-up ponds (other than the plants needed for the water treatment function of the pond or the creation of safety barriers). This will allow native plants more opportunity to colonise.
16. Don't plant non-native water plants, trees, shrubs or grass mixes; take special care to avoid invasive alien plants such as *Crassula helmsii* by dealing with nurseries that only deal in native stock.
17. If planting is essential, stick to native plants of local origin. Include species which are wildlife friendly e.g. grasses such as *Glyceria fluitans* (Floating Sweet-grass) and *Agrostis stolonifera* (Creeping Bent).
18. Check planting schemes 1 and 2 years after establishment to ensure that specifications have been carried out and undertake immediate remedial action if invasive alien species are found.
19. Consider whether grazing livestock can be given access to ponds; grazing has been shown to be a viable and effective way of managing some SUDS schemes in agreement with conservation organisations or farmers.
20. Wherever possible include a brief post-implementation stage about 1 year after SUDS creation. Use this to (i) undertake fine-tuning of the pond design and (ii) capitalise on new opportunities that have arisen (e.g. pooling of natural areas of standing waters or natural seepage areas etc.). Fine tuning of this sort costs very little but will often greatly increase the biodiversity value of a SUDS scheme.

6.2.2 Wetlands

Although wetlands are often distinguished from ponds in SUDS literature, in reality there is little difference between the two ecologically. Indeed, what most SUDS guides term wetlands are simply ponds that are shallow enough for emergent plants to grow throughout their area. Although apparently dominated by emergent plants, they usually support the same types of plants and animals as can be seen in shallow ponds generally.

For this reason, broadly the same guidelines apply to the ecological design of wetlands as for ponds.

Specifically:

- Water quality should be as good as possible
- Physical structure of wetlands should be designed following the same principles as those described for ponds (i.e. multiple sub-basins, variations in permanence)
- Locate close to existing wetlands, without destroying valuable aquatic or terrestrial habitat
- Manage no more than is necessary for the SUDS function of the wetland
- Avoid planting-up beyond that necessary for SUDS functions.

6.2.3 Swales and filter strips

Introduction

Swales and filter strips typically occupy a relatively small area of SUDS schemes, but may be able to provide useful terrestrial and aquatic habitat. They are also likely to be highly exposed to contaminants as part of their interceptor function so will usually only be able to support assemblages of robust and tolerant species.

Design of filter strips and swales

Given the ecological restraints noted above, a number of recommendations can be made about their design:

- Where this does not compromise flow and infiltration requirements, create undulating depressions within shallow swales to allow the development of temporary pools, especially where grass is kept short
- Avoid the use of nutrient rich top-soil in creating swales and filter strips. Where swales convey water to ponds and wetlands, the use of nutrient rich top-soil to facilitate grass establishment is likely to lead to increased pollutant burdens in any downstream ponds and wetlands.

Management of swales and filter strips

From an ecological perspective, long, tussocky vegetation cut only periodically is preferable for swales. This will, however, interfere with the development of laminar flow, which is believed to maximise infiltration. Longer vegetation may, therefore, be mainly suitable where expected water volumes are low, or there is sufficient space to allow creation of long swales where a rapid infiltration rate is not essential.

6.2.4 Adjacent land

Adjacent land can provide clean catchments for off-line seasonal and permanent ponds which contribute to the ecological value of the overall scheme.

6.2.5 Definitive list of flora and fauna that are desirable / not desirable

A definitive list of undesirable plants can be specified easily: as a simple rule of thumb, all non-native plants should be excluded from SUD schemes. Because SUD schemes are part of the natural drainage system of a catchment, all planting should be regarded as *de facto* release to the wild. This means that there should be a general presumption against all forms of ornamental planting of aquatic and wetland plants. As a measure of SUDS effectiveness, each non-native species added should be regarded as a negative impact on the environment.

Box 4. Rules for planting up SUDS ponds

- Desirable plants should be identified region by region by local experts or consultants using the most up to date flora for the county in which the SUDS scheme is being developed. A good starting point is the list of plants which have occurred in National Pond Survey ponds
- Plants which are found in normally rivers should not be planted
- For most counties in Britain there is a local flora – use the NPS list to develop a list for the region checking whether the plant is common in that region; where no local flora is available the current distribution of plants can be checked in the New Atlas of the British & Irish Flora (Preston *et al.* 2002)
- Only common species should be planted; plants which have nationally local distributions (i.e. occur in 705 or less of the 2823 10 x 10 km grid squares mapped in the UK) or are nationally scarce or rare, should not be planted. PCTPR can provide a list of these species
- Avoid adding submerged and floating-leaved plants – these will generally colonise naturally if the pond is suitable. If it is not (because of poor water quality, particularly excess nutrients), added plants will die, wasting money and planting effort. There is little evidence that aquatics can soak up nutrients in ponds unless a large biomass of plants is already present before nutrients are added. In ponds with high nutrient levels most aquatics simply fail to grow
- Plants should be from sources local to the site; the ideal is to source plants from within 30 km of the SUDS scheme
- The practice of growers substituting plants which have not been specified when specified plants are unavailable should be stopped. It is better to plant fewer species than substitute undesirable species.

For desirable plants, it is *not* the ideal to work from a single national 'desirable plants' list. This practice has in part contributed to the poor ecological quality of most SUDS planting schemes. A strategy for developing good planting schemes in SUDS ponds is outlined in Box 4.

6.3 Best practice for SUDS Design to mitigate downstream impacts

The main aim is to make the SUDS scheme so effective that the urban footprint is minimised in terms of both water quality and quantity (and periodicity) leaving the scheme. This requires:

- Proper initial assessment of the natural base quality and flow rate of water leaving the site prior to development
- Ensuring that there are enough elements in the treatment train to maintain water quality and full source control
- A proper maintenance regime for the SUDS system
- Periodic checks on chemical and ecological quality and volume of outflow water, and the receiving water, to ensure that functioning is maintained.

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Appendices

Appendix 1

Hopwood Park MSA treatment train chemical analysis

Appendix 1 Hopwood Park MSA treatment train chemical analysis

The chemical water quality of the SUDS at Hopwood Park is compared to standards for flowing water and in some cases to minimally impaired ponds.

This report considers the ecological value of the SUDS themselves in providing habitats, because as discussed in Section 2.4.2- inland wetlands are a valuable habitat. To illustrate the value of the Hopwood Park SUDS as habitats they are compared to minimally impaired standards.

However the main function of the SUDS ponds is to attenuate and treat storm water before releasing it into the watercourse, so as to reduce the impact of the storm water on the receiving watercourse.

The impact SUDS ponds are likely to have on their receiving watercourse will be dependent on the quality of the receiving watercourse. As most SUDS are used in urban settings, in most cases it is unlikely that SUDS would be discharging into RE1 class rivers. For this reason, where data are available, the discharges from the SUDS ponds are compared to RE1, RE2 and RE4 class rivers to illustrate where the discharge from the SUD would be likely to impact on quality of the receiving watercourse in these cases.

Discharges from SUDS should also be put into the context of waste water treatment discharges. Where the levels were comparable, the recommended level for urban waste water, from EU directive 91/271/EEC, have been plotted.

The results provide data taken at different times of the year. There is not enough data to make robust assumptions about the impact of seasonality compared to varying pollutant loads that might occur on different days, for reasons completely unrelated to season.

A1.1 Biochemical Oxygen Demand (BOD)

Data on BOD treatment at Hopwood Park are available from the three systems shown in Figures 1-3. (Coach Park, Heavy Goods Vehicle Area, Main Car Park).

Outflows from the three treatment systems varied markedly in their BOD values. Two of the treatment trains (the Coach Park and Main Car Park) had final outflows with low BOD values, close to values expected in the cleanest rivers. In contrast, in the third train (HGV Area), BOD remained high at the final outflow.

Habitat quality: No data are available on the natural BOD levels in minimally impaired ponds in the UK. However, given that unpolluted ponds would be expected naturally to have higher BODs than flowing waters, values of less than 2.5 mg/l can probably be regarded as indicative of high quality ponds. Thus in terms of BOD, levels in the Coach Park Balancing Pond (Figure 1) and the Car Park Pond (Figure 3) were equivalent to those of minimally impaired ponds. Levels were higher in other ponds on the site and may have been above the levels associated with minimally impaired ponds.

Receiving water impacts: Two of the three systems at Hopwood Park produced outflow BOD levels which were close to the levels associated with the highest quality rivers and therefore would be expected to have little or no impact on receiving waters. However, the outflow from the HGV Area would fall into the worst class of the Environment Agency's river classification in terms of BOD (90%-ile = 15 mg/l BOD) and could, unless further diluted by the receiving water, cause significant environmental impact.

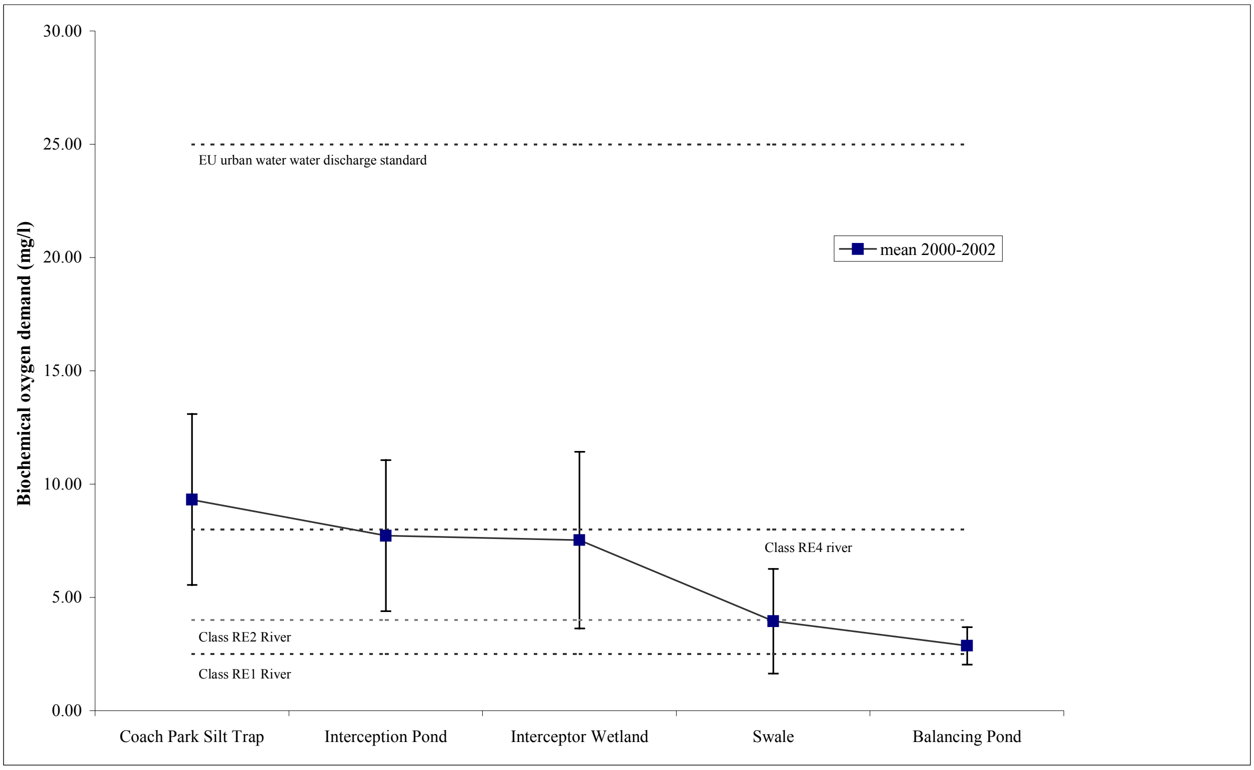


Figure 1 BOD values in the Coach Park SUDS system at Hopwood Park MSA

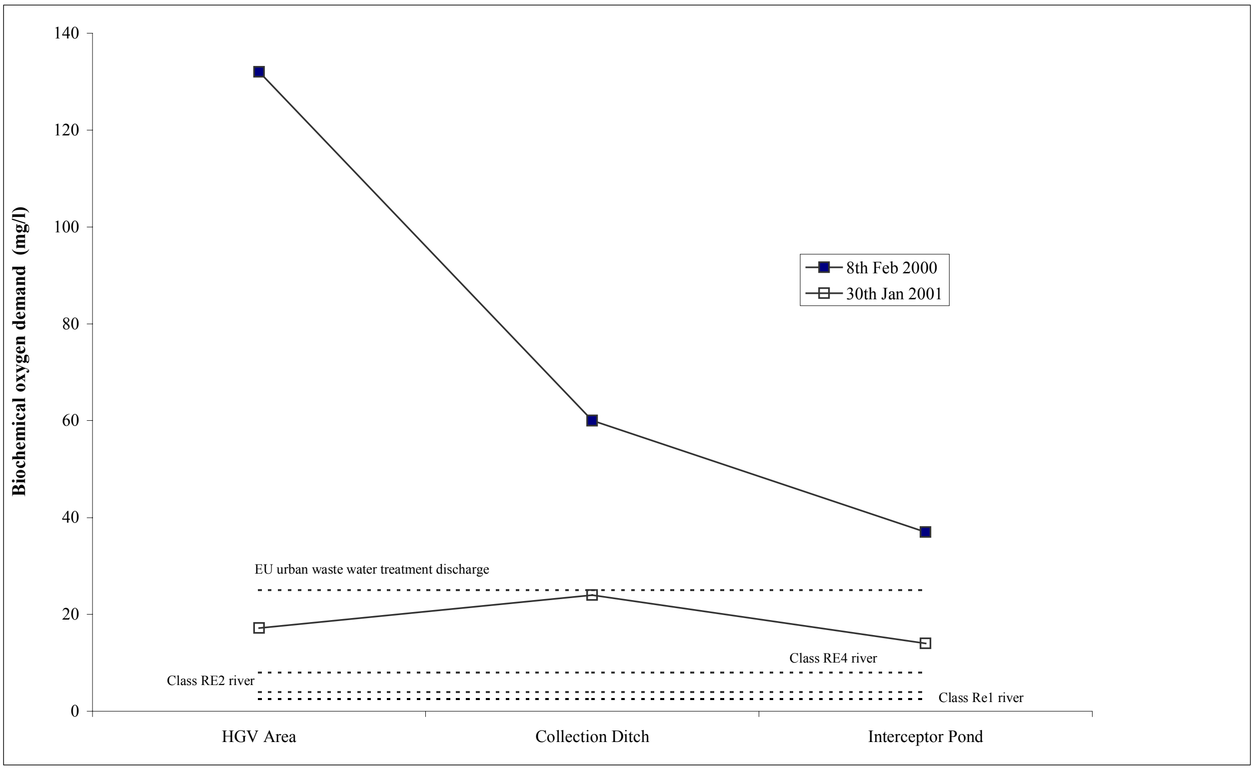


Figure 2 BOD values in the Heavy Goods Vehicle Park SUDS system at Hopwood Park MSA

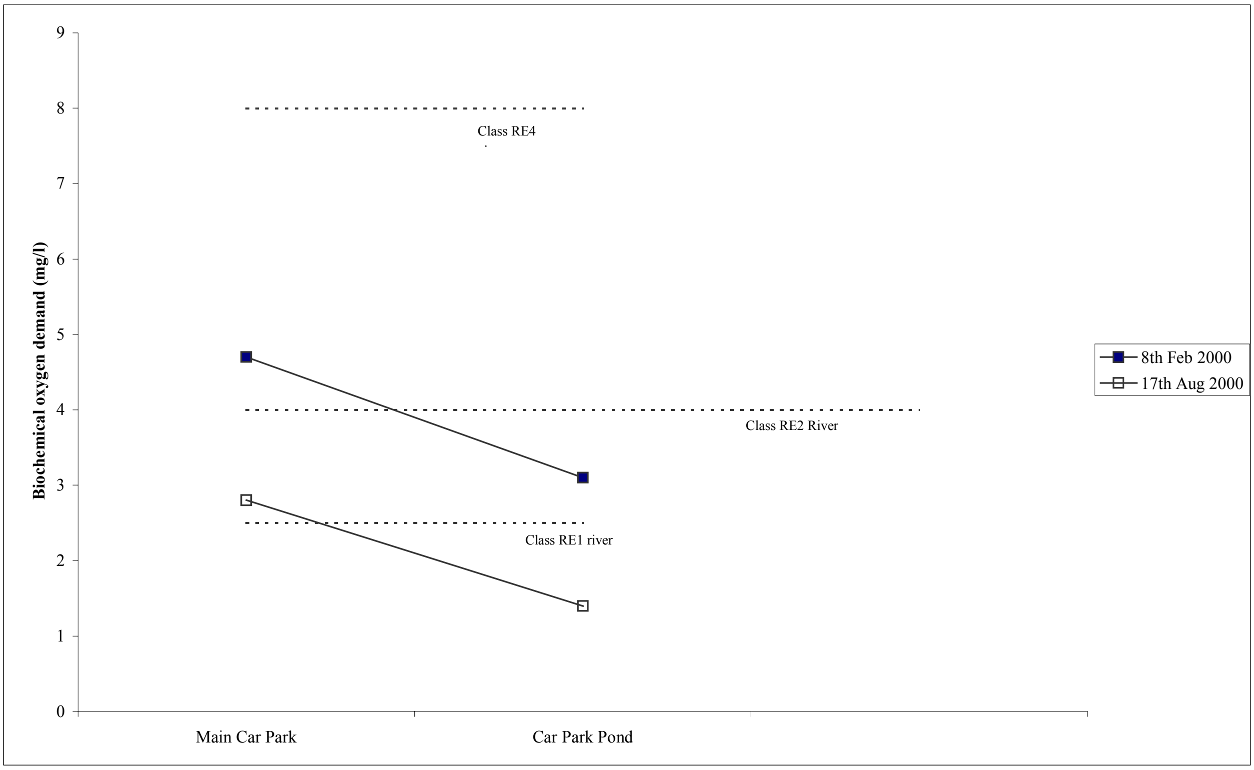


Figure 3 BOD values in the Main Car Park SUDS system at Hopwood Park MSA

A1.2 Suspended solids

Data on suspended solids are available for three systems at Hopwood Park from a maximum of three dates in 2000 and 2001 (Figures 4, 5 and 6).

Habitat quality: Ponds at the end of the treatment sequence generally had suspended sediment concentrations below the mean value for natural ponds. On one sampling date, levels in the Car Park Pond (which has only one pre-treatment stage) were significantly above the level seen in minimally impaired ponds.

Receiving waters impacts: No suspended sediment levels are set for UK rivers at present so it is not possible to compare the outflows from Hopwood Park with recognised UK limits. However, suspended sediment concentrations of less than 25 mg/l are regarded as unlikely to damage freshwater fish assemblages. Thus for two of the Hopwood Park systems, concentrations entering receiving waters were well below this. Concentrations on the 8th February 2001, were above the 25mg/l limit and exceeded the EU standard for urban waste water.

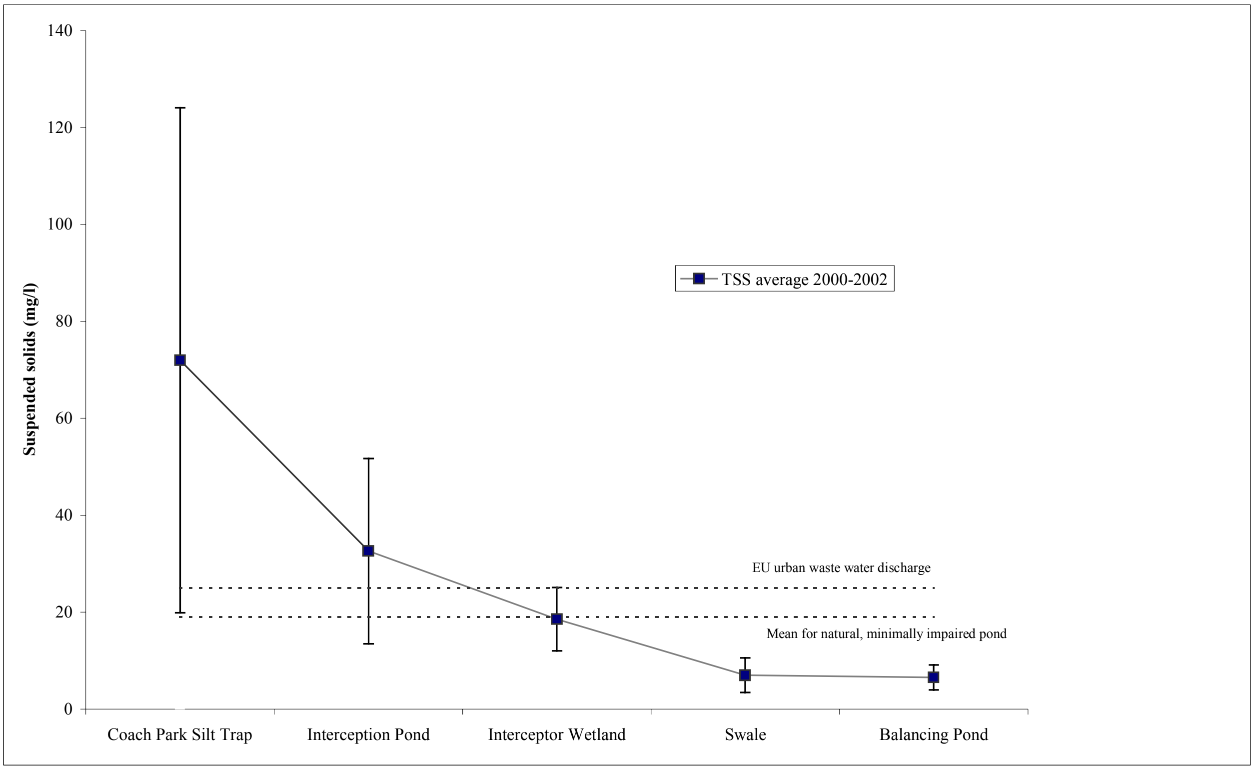


Figure 4 Suspended sediment concentrations in the Coach Park SUDS system at Hopwood Park MSA

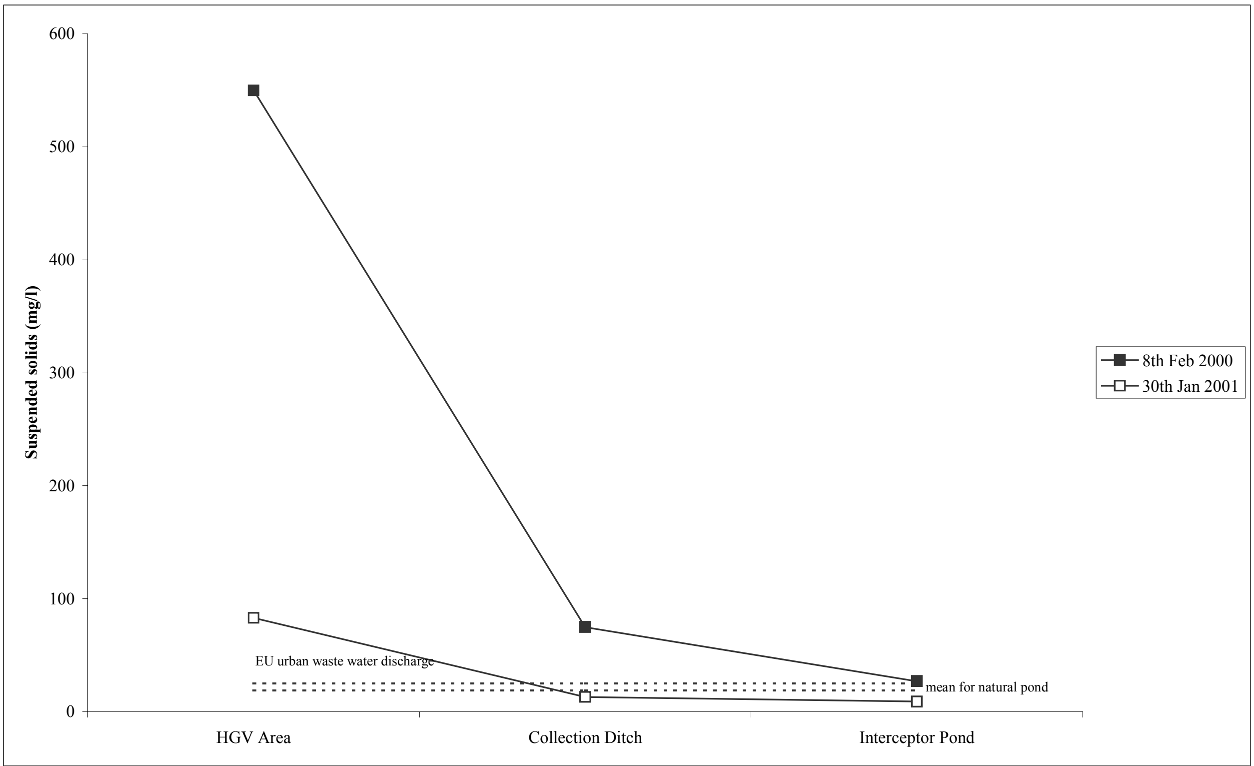


Figure 5 Suspended sediment concentrations in the Heavy Goods Vehicle areas SUDS system at Hopwood Park MSA

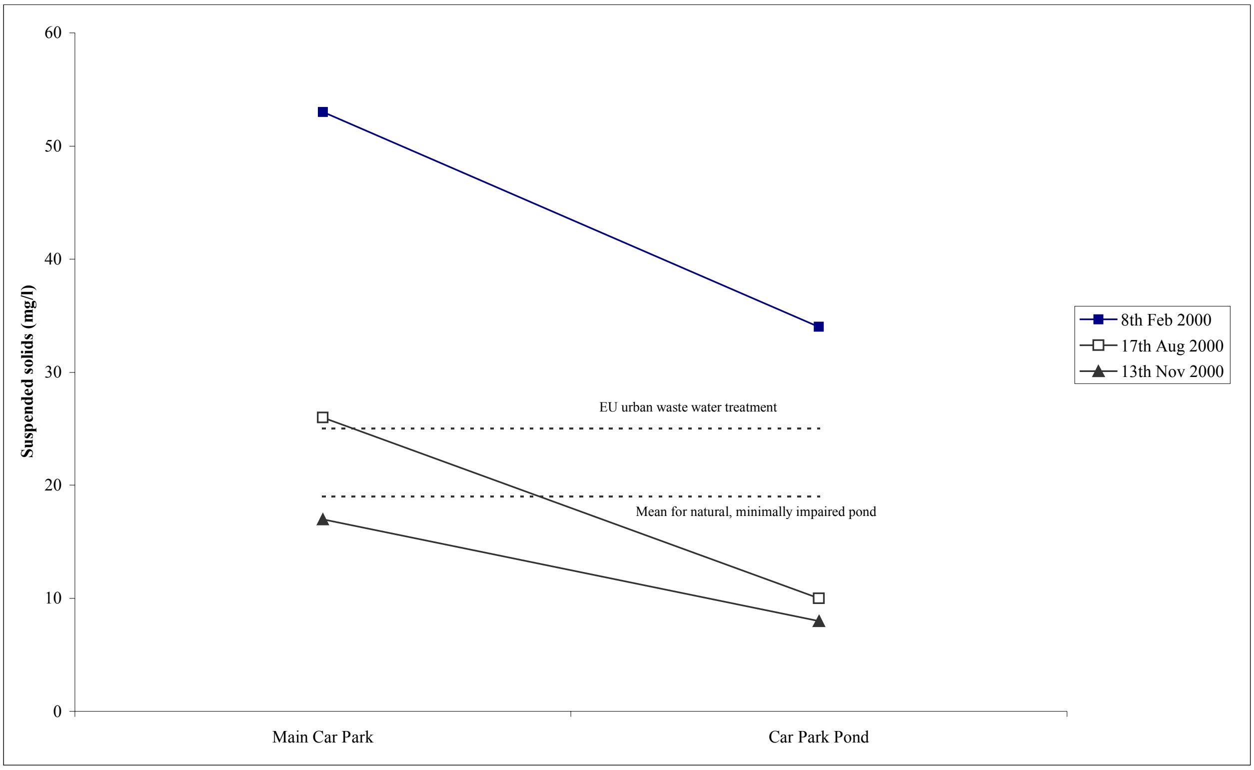


Figure 6 Suspended sediment concentrations in the Main Car Park SUDS system at Hopwood Park MSA

A1.3 Ammonia

Data on ammonia concentrations are available for 2000-2002 for the Coach Park system and between February 2000 and February 2001 for the HGV Area and Main Car Park systems.

Habitat quality: The ammonia concentrations in ponds at the end of the treatment train in the Coach Park and Main Car Park systems were generally below the mean concentration observed in minimally impaired ponds (Figures 7 and 8). However, the HGV Area system had exceptionally high ammonia concentrations both at the beginning and end of the treatment train, well above the natural baseline, although it should be noted that only one set of samples (January 2001) was available (Figure 9). This result is consistent with lorry drivers' common practice of urinating next to their lorries! Levels in ponds higher up the treatment train in the Coach Park and Main Car Park were also elevated above baseline levels. In general, therefore, the lower ponds provided a satisfactory environment in terms of ammonia concentrations whereas impacts could be expected further up the system.

Receiving waters impacts: The Coach Park and the Main Car Park SUDS systems (Figures 7 and 8) had final ammonia concentrations that were below the 90 percentile value for high quality rivers (0.25 mg/l total ammonia). These concentrations would create little impact on receiving waters. Concentrations in the HGV Park output would, however, have considerable potential for impact on receiving waters.

Effectiveness in relationship to Chronic Value: The ammonia Chronic Value given by the US EPA is 1.7 µg/l. The lowest ammonia concentration measured in the SUDS outputs at Hopwood Park was well above this, exceeding it by a factor of about 10.

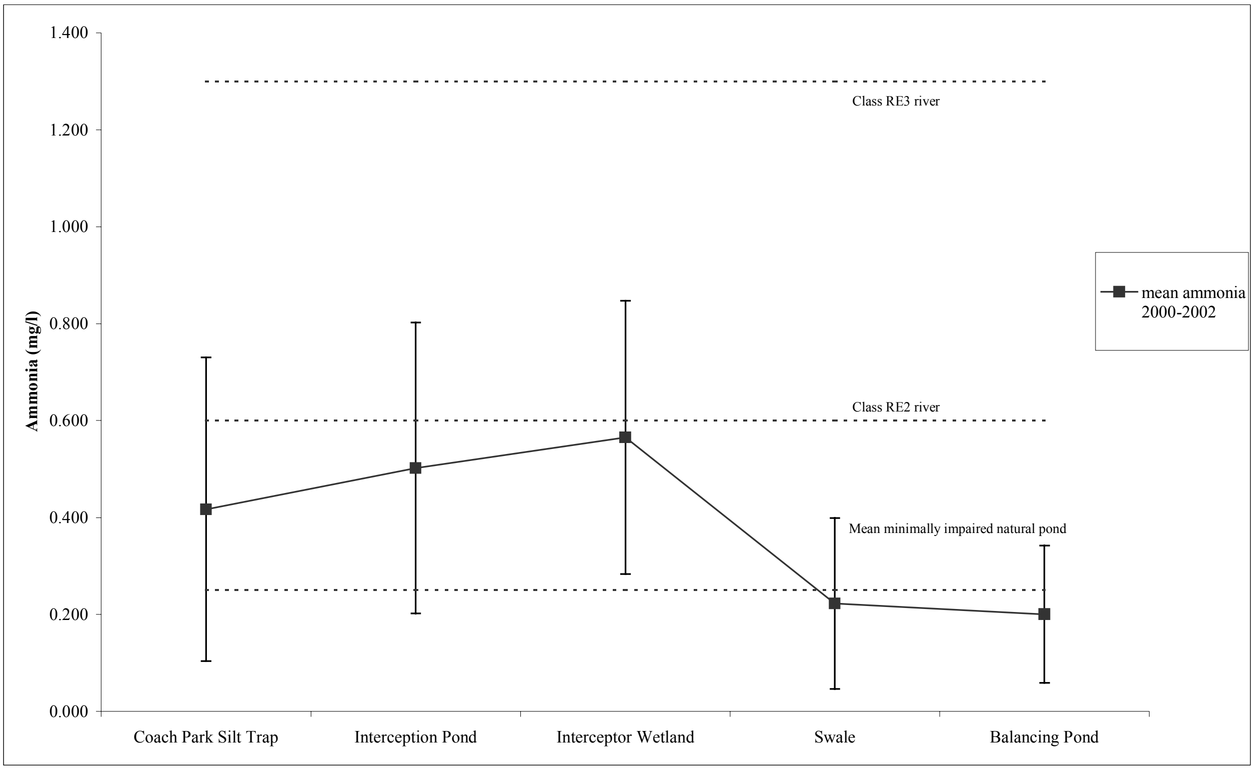


Figure 7 Ammonia concentrations in the Coach Park SUDS system at Hopwood Park MSA

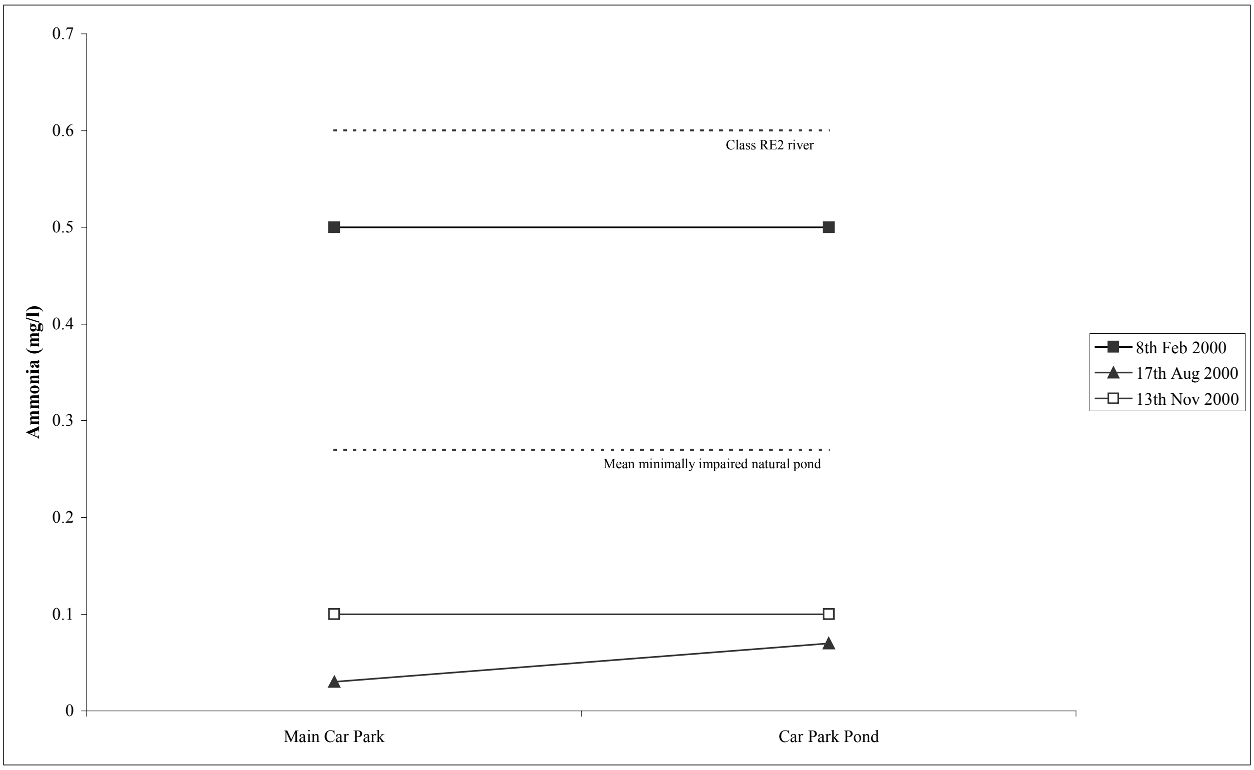


Figure 8 Ammonia concentration in the Main Car Park SUDS system at Hopwood Park MSA

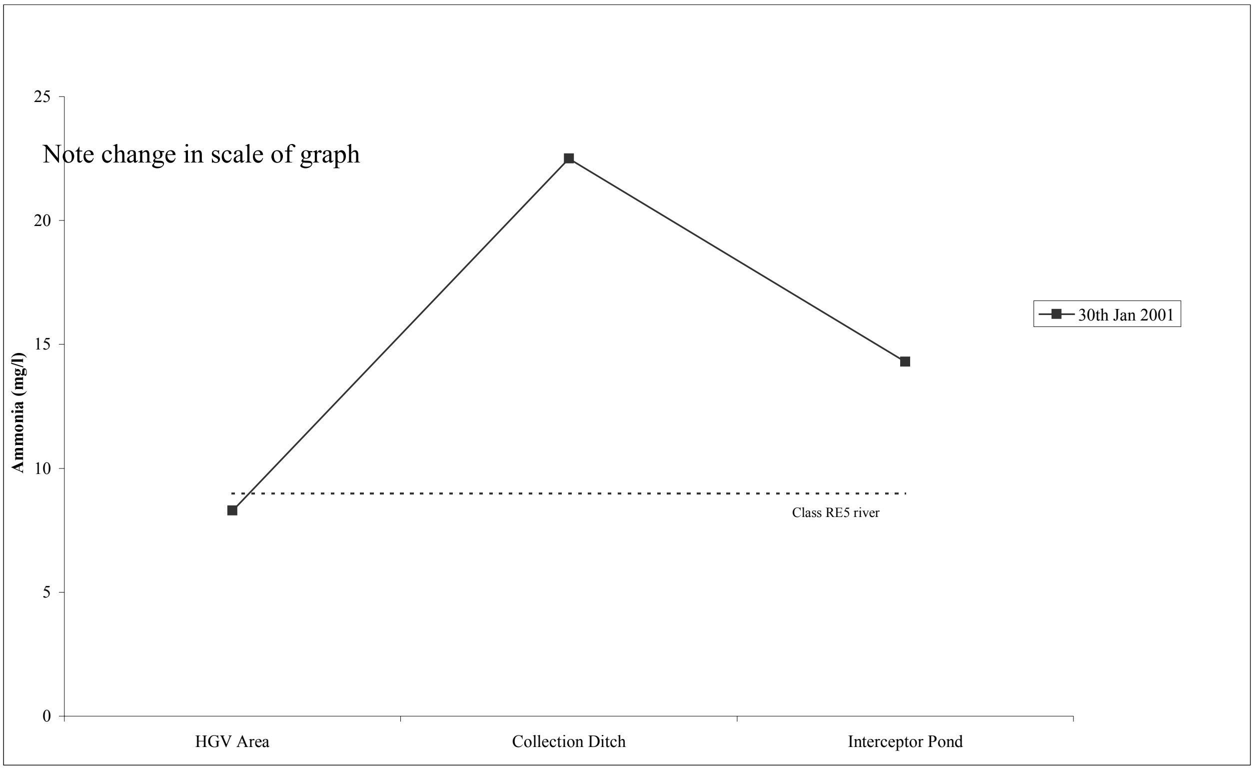


Figure 9 Ammonia concentrations in the HGV Park SUDS system at Hopwood Park MSA

A1.4 Cadmium

Data on cadmium concentrations at Hopwood Park are available for two of the treatment trains (Coach Park and Main Car Park) on two dates (Figures 10 and 11).

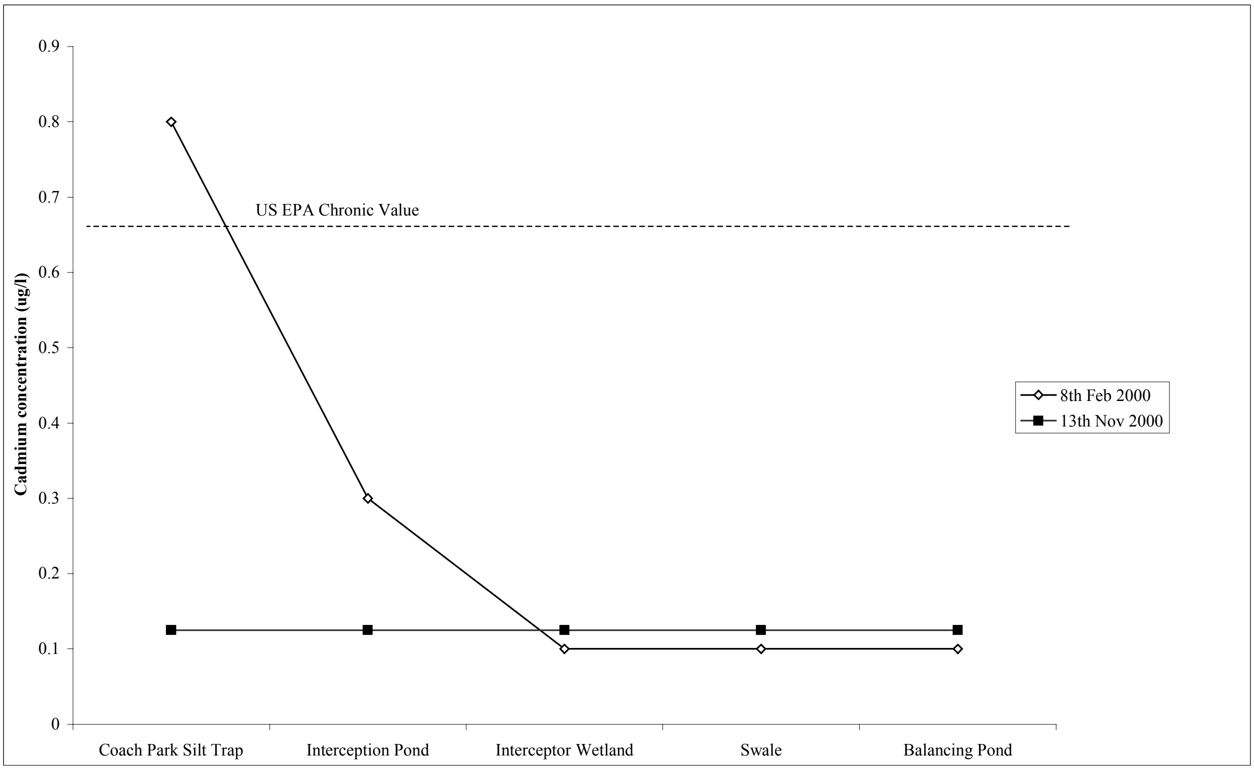


Figure 10. Cadmium concentrations in the Coach Park SUDS system at Hopwood Park MSA

Habitat quality: No data are available on Cadmium levels in minimally impaired ponds in the UK. However, water column concentrations in the Coach Park and Main Car Park ponds were all below the US EPA Chronic Value of 0.66 µg/l. Receiving water concentrations were, therefore, also below the US EPA chronic concentrations.

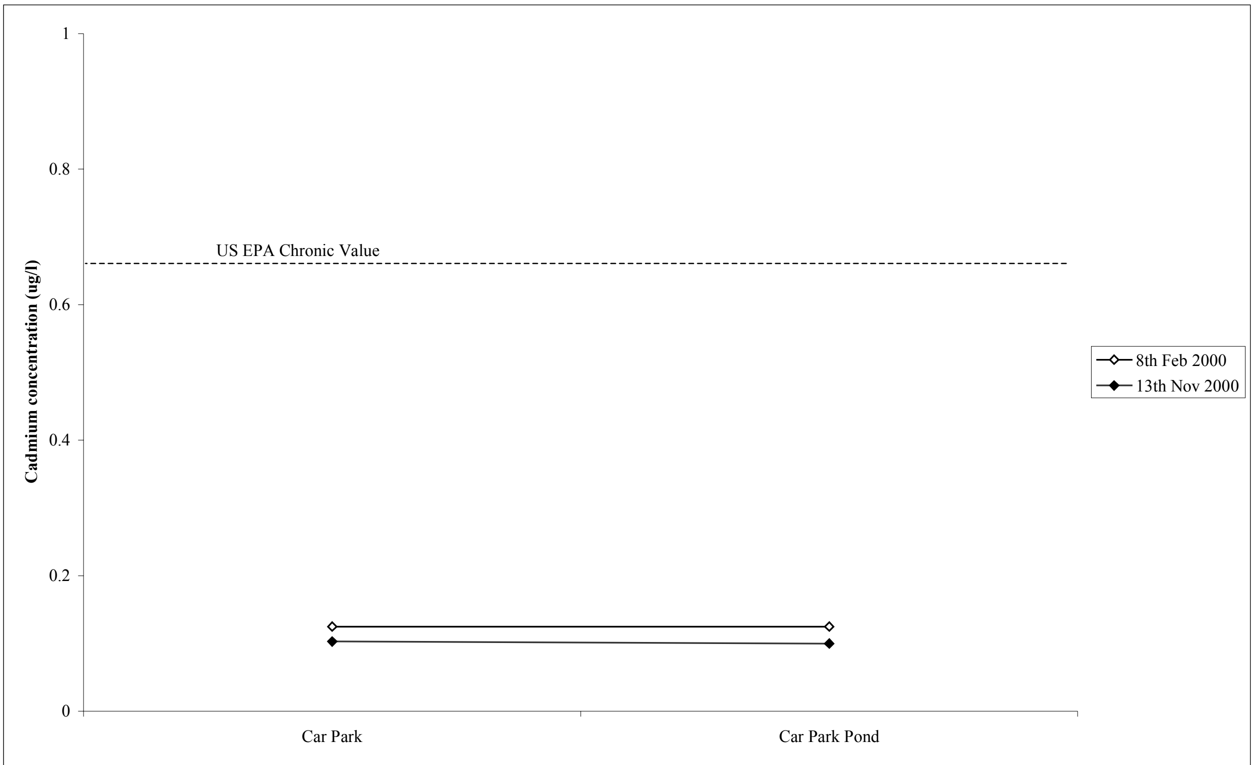


Figure 11 Cadmium concentration in the Main Car Park SUDS system at Hopwood Park MSA

A1.5 Chromium

Measurements of total chromium concentrations are available from the Coach Park system from 2000 to 2002 and from the Main Car Park systems on two dates between February 2000 and February 2001 (Figures 12 and 13). Note that interpretation of chromium toxicity is complicated by the existence of chromium in two forms: chromium III (trivalent) and chromium VI (hexavalent). The latter is more toxic to aquatic life but data from the Hopwood Park SUDS system gives only total chromium concentrations.

Habitat quality: Concentrations of chromium in the Coach Park pond systems are all below or very close to the US EPA Chronic Values for both chromium III and chromium VI. Levels are slightly higher in the Main Car Park pond so chromium VI could, theoretically, be at damaging concentrations in these ponds.

All receiving waters are subject to concentrations of chromium below the US EPA chronic value for chromium III although there is a theoretical risk from chromium VI.

Note that Swedish EPA values for background chromium levels are considerably below those found in the Hopwood Park ponds (Table 5).

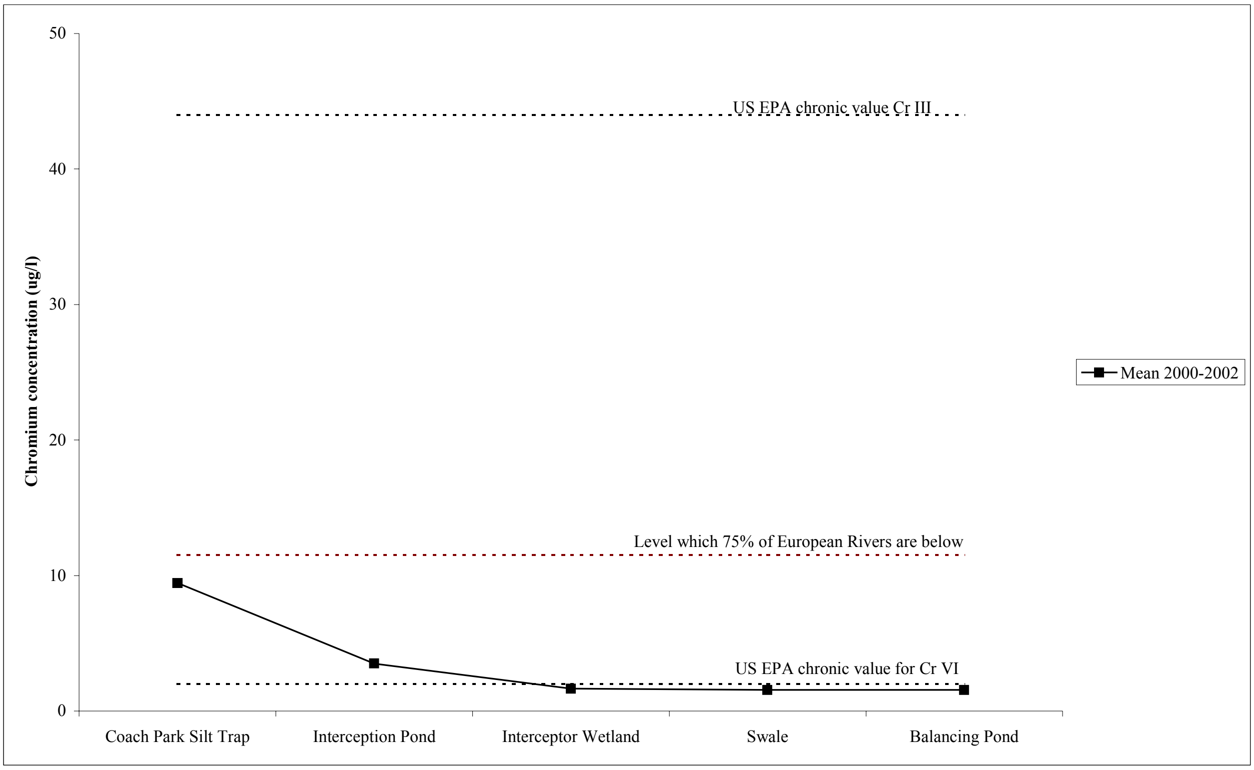


Figure 12 Chromium concentrations in the Coach Park SUDS system at Hopwood Park MSA

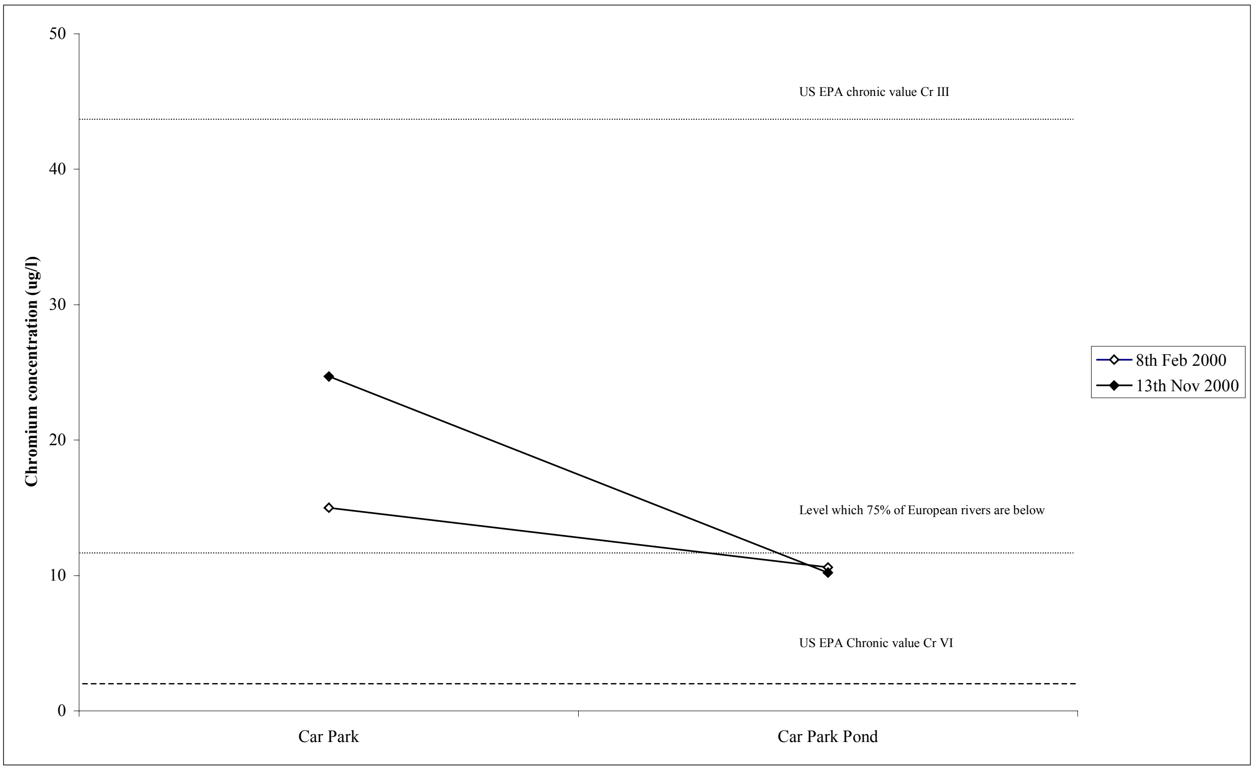


Figure 13 Chromium concentration in the Main Car Park SUDS system at Hopwood Park MSA

1.6 Copper

Data on copper concentrations are available for the Coach Park system from 2000-2002. For the HGV Park and the Main Car Park data are available from up to four dates in 2000 and 2001 (Figures 14-16).

Habitat quality: Copper concentrations in ponds at the end of the Hopwood Park treatment trains were below the mean for minimally impaired ponds in two of the systems (Coach Park and Main Car Park) and significantly above this level in the HGV Park system.

Receiving water impacts: Copper concentrations at the end of all three Hopwood Park treatment trains were below the 90%-ile value for River Ecosystem Class 1 rivers, as defined by the Environment Agency. No data are available on the calcium carbonate concentrations in the Hopwood Park SUDS systems and it is possible that, being rainwater fed and by definition close to the catchment source, low calcium carbonate levels may occur. Note that the values observed exceed by a factor of about 5 times the levels recognised as background by the Swedish EPA.

Effectiveness in relationship to Chronic Value: The minimum concentrations recorded in the Hopwood Park treatment trains exceeded the US EPA Chronic Value (5.3µg/l) by a factor of about 2.

1.7 Lead

Data on lead concentrations at Hopwood Park are available for the Coach Park system from 2000-2002. For the HGV Park and the Main Car Park, data are available from up to four dates in 2000 and 2001 (Figures 17-19). Note that lead toxicity is also dependent on water hardness.

Habitat quality: Water phase lead concentrations at the end of the SUDS treatment train are below the mean concentrations recorded in minimally impaired ponds. For biota directly exposed to lead in the water column levels are unlikely to cause significant impacts. For biota associated with the sediment phase (e.g. macroinvertebrates, rooted plants) relatively high concentrations of lead may be available and impacting biota. At present no sediment data are available from Hopwood Park to assess this potential impact.

Receiving water impacts: River water quality in England and Wales is not assessed in terms of lead concentrations so no values are available for comparison. However, concentrations in the outflows from the three treatment systems at Hopwood Park are quite close to the background levels used by the Swedish EPA (0.05 µg/l).

Effectiveness in relationship to Chronic Value: The US EPA Chronic Value for lead is 12.26 µg/l. Output values are below this level on all occasions.

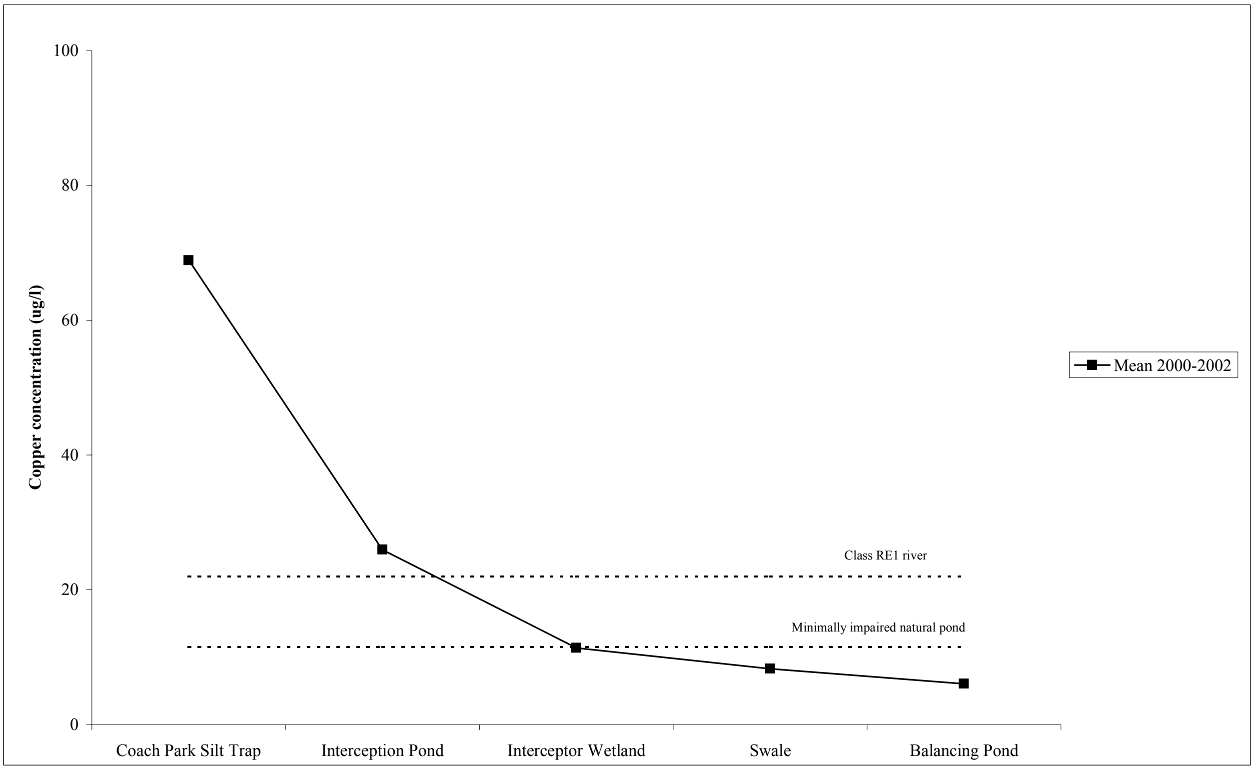


Figure 14 Copper concentrations in the Coach Park SUDS system at Hopwood Park MSA

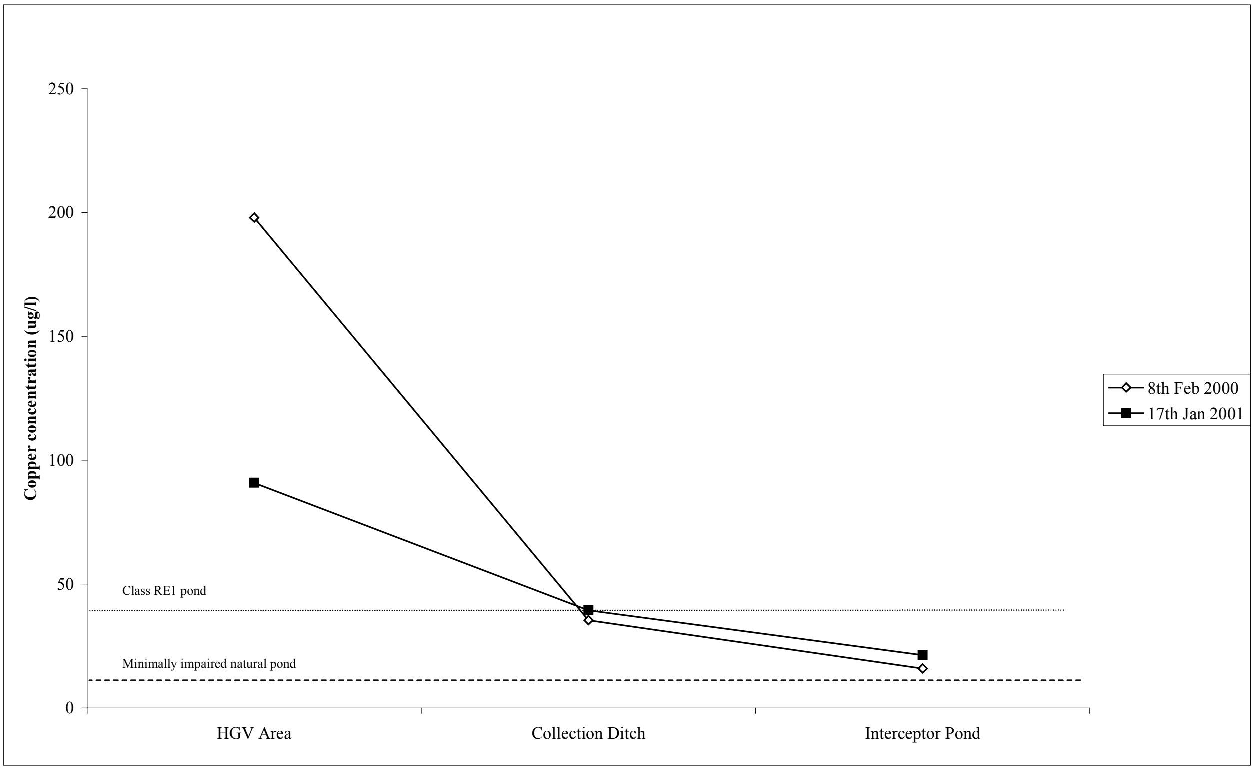


Figure 15 Copper concentrations in the HGV Park SUDS system at Hopwood Park MSA

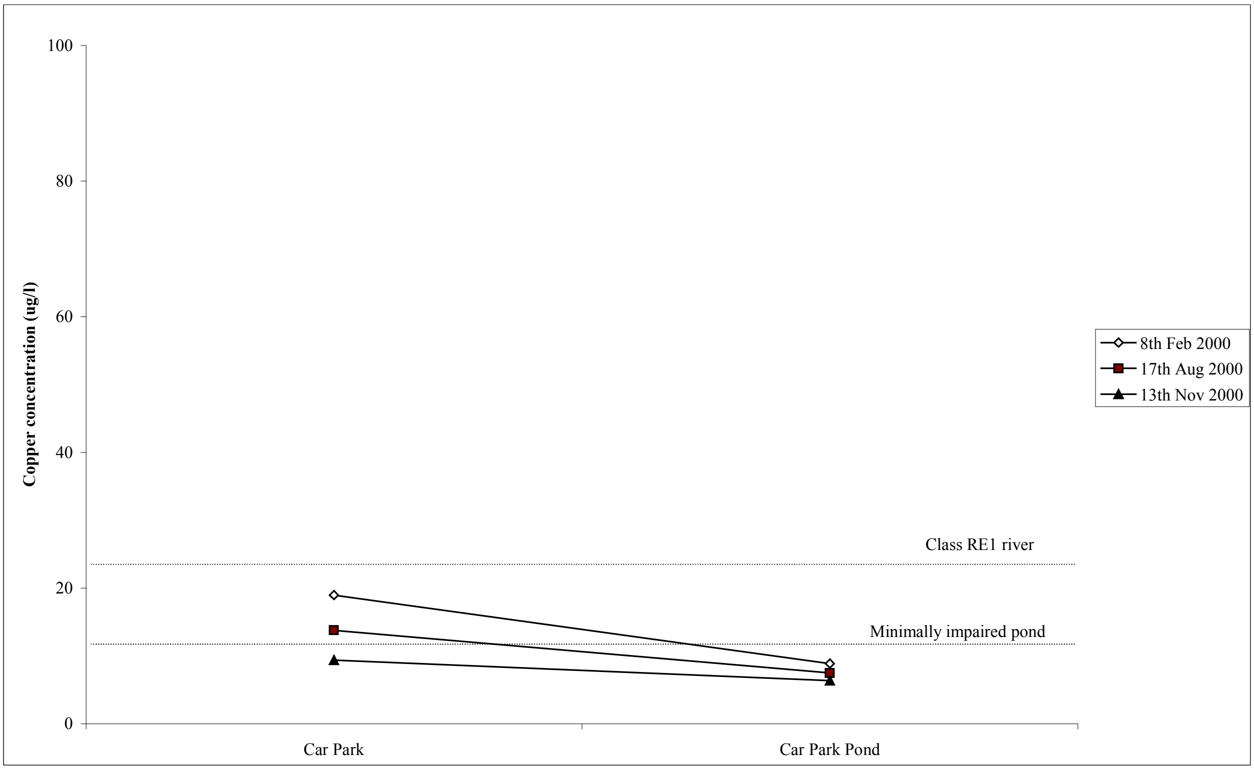


Figure 16 Copper concentration in the Main Car Park SUDS system at Hopwood Park MSA

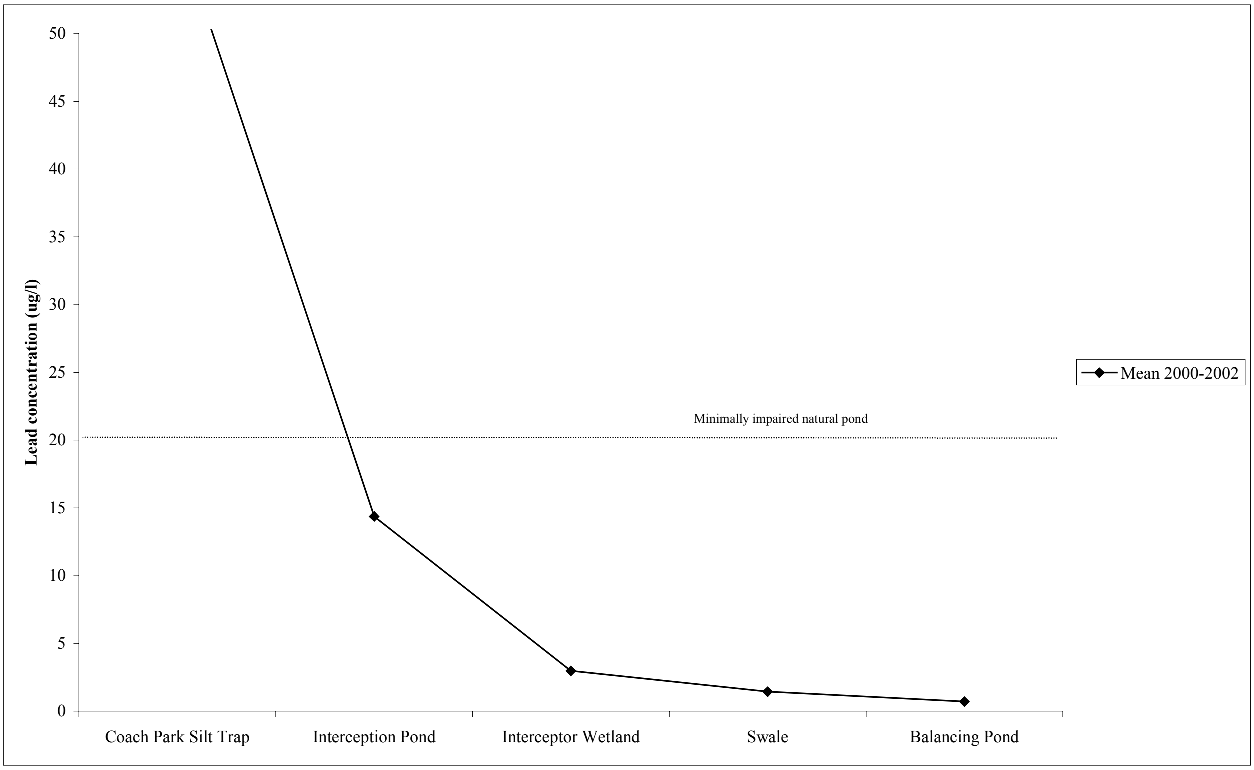


Figure 17 Lead concentrations in the Coach Park SUDS system at Hopwood Park MSA

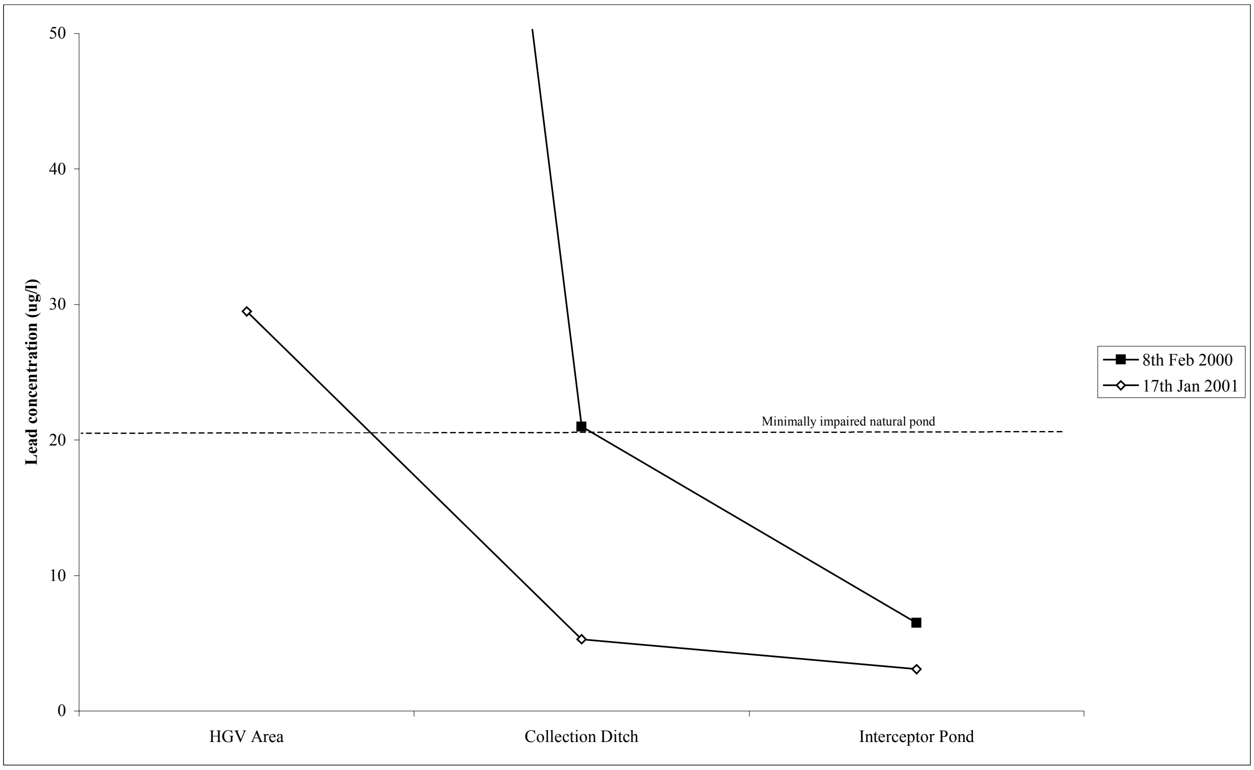


Figure 18 Lead concentrations in the HGV Park SUDS system at Hopwood Park MSA

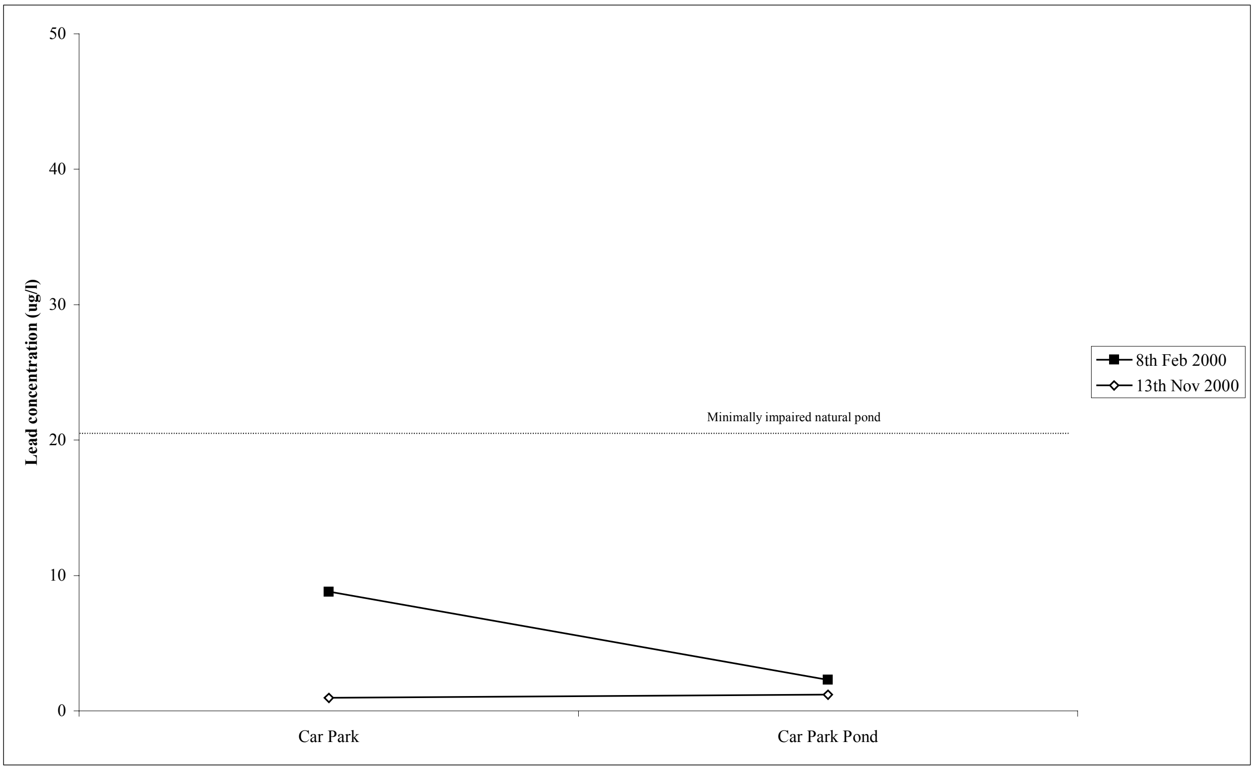


Figure 19 Lead concentration in the Main Car Park SUDS system at Hopwood Park MSA

1.8 Nickel

Data on nickel in the Hopwood Park ponds are available from up to three dates from the three systems (Figures 20-22). No data on nickel concentrations are available from minimally impaired ponds and the Environment Agency does not set river water quality objectives for nickel.

In all cases measured values are close to the US EPA Chronic Value for nickel (5 µg/l) suggesting that, both in terms of pond habitat quality and receiving water impacts, nickel concentrations pose a low risk to biota.

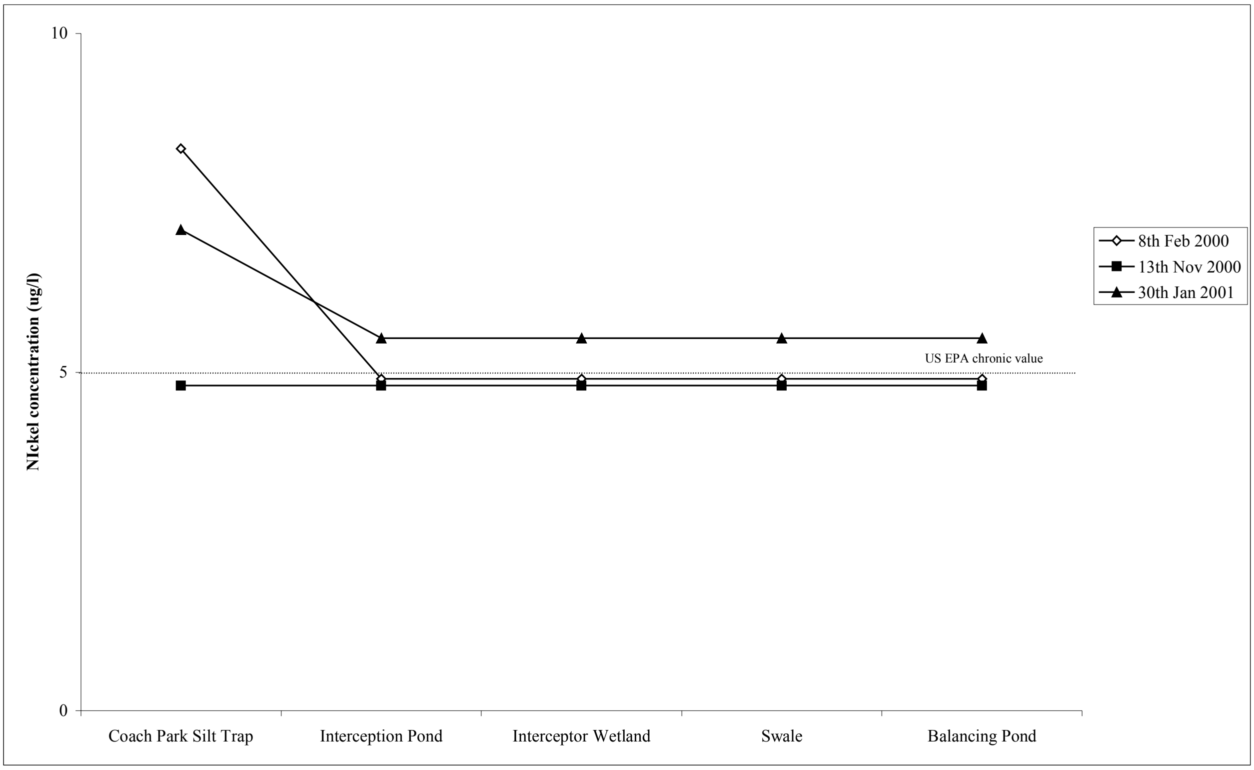


Figure 20 Nickel concentrations in the Coach Park SUDS system at Hopwood Park MSA

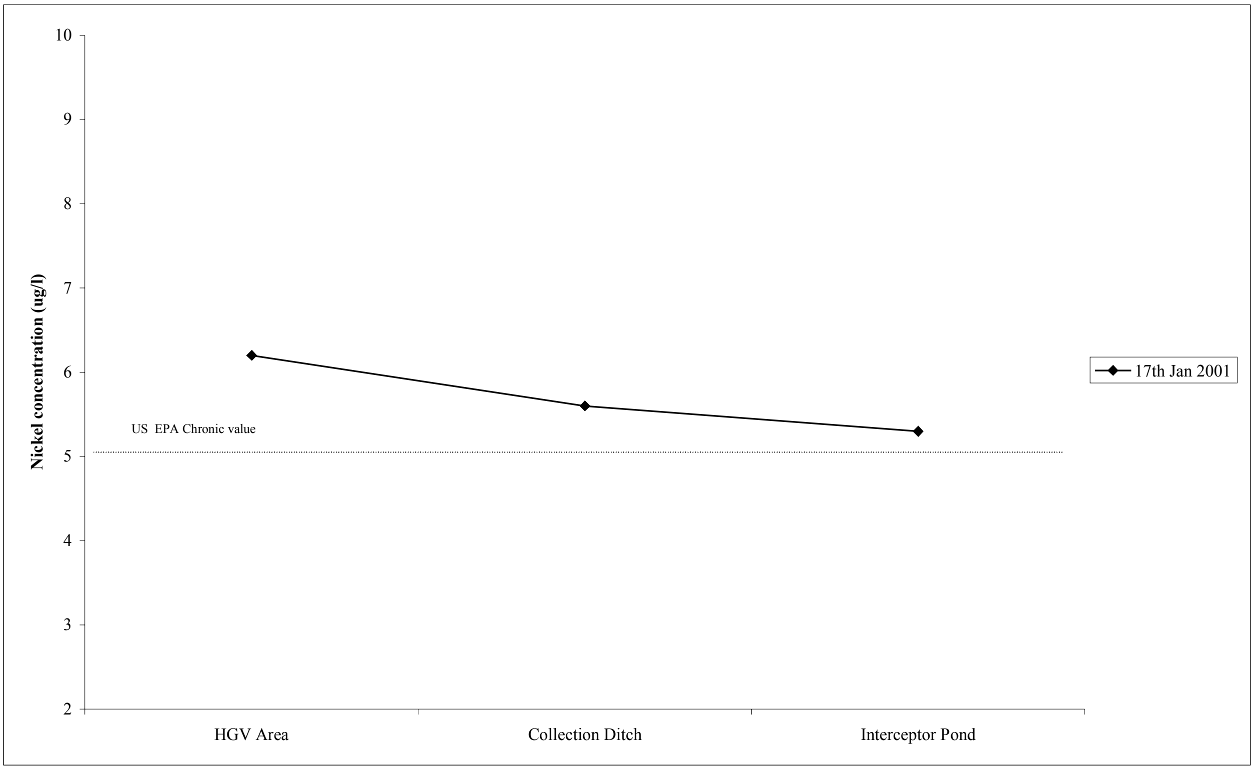


Figure 21 Nickel concentrations in the HGV Park SUDS system at Hopwood Park MSA

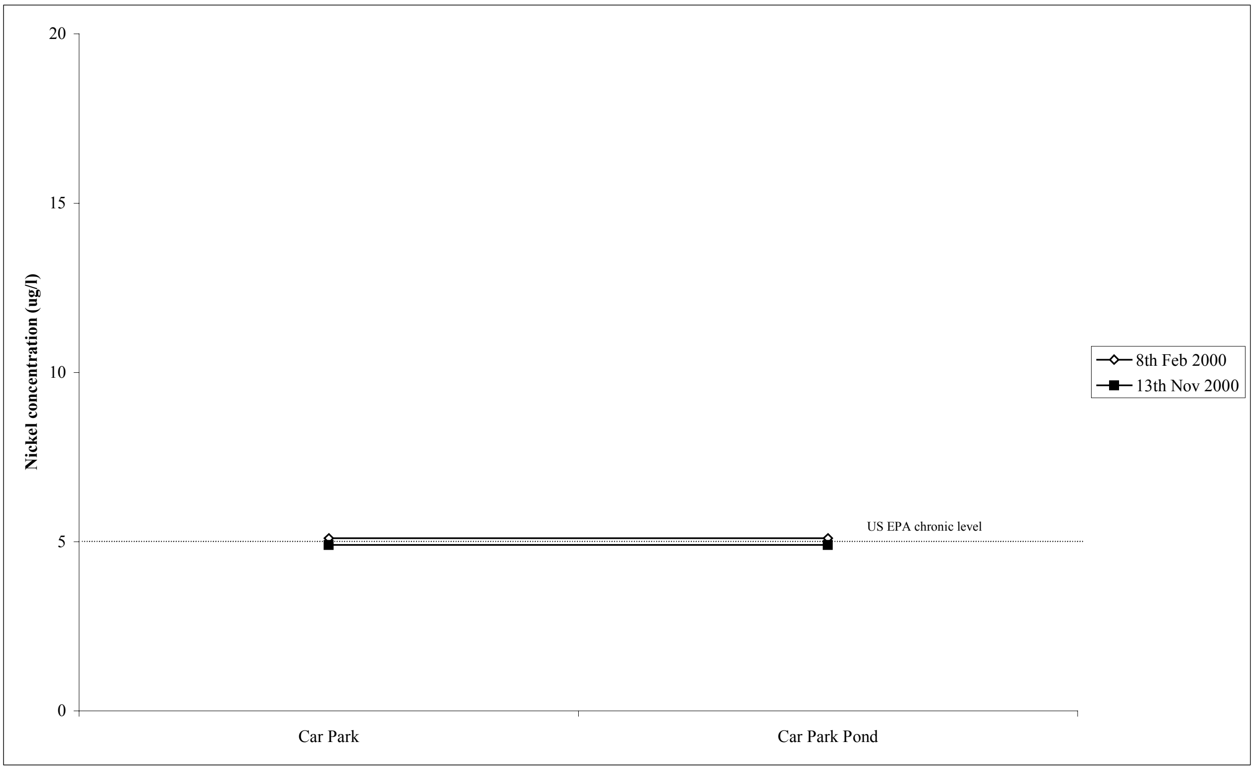


Figure 22 Nickel concentration in the Main Car Park SUDS system at Hopwood Park MSA

1.9 Zinc

Data on zinc concentrations in the Hopwood Park ponds are available for the Coach Park system from 2000-2002. For the HGV Park and the Main Car Park, data were available from up to four dates in 2000 and 2001 (Figures 23-25).

Habitat quality: Zinc concentrations in the ponds at the end of the Coach Park and Main Car Park treatment trains were well below the mean zinc concentrations seen in minimally impaired ponds. However, some of the upstream ponds (e.g. Coach Park Interception Pond) have concentrations which are likely to impact on some aquatic biota. Concentrations in the HGV Park system were generally higher than on other parts of the site, but were still equal to or below the minimally impaired pond mean at the end of the treatment train.

Receiving water impacts: Outflows from all of the Hopwood Park SUDS components were considerably below the 90%-ile value for fairly hard water high quality rivers in England and Wales, as defined by the Environment Agency. However, values were well above the Swedish EPA estimate of natural background levels in rivers of 3 µg/l (Table 5). As zinc toxicity is dependent on pH, SUDS systems could be vulnerable to relatively acid conditions exacerbating heavy metal impacts.

Effectiveness in relationship to Chronic Value: The US EPA Chronic Value for zinc is 30 µg/l and values were generally fairly close to this level in the lower parts of the treatment train at Hopwood Park.

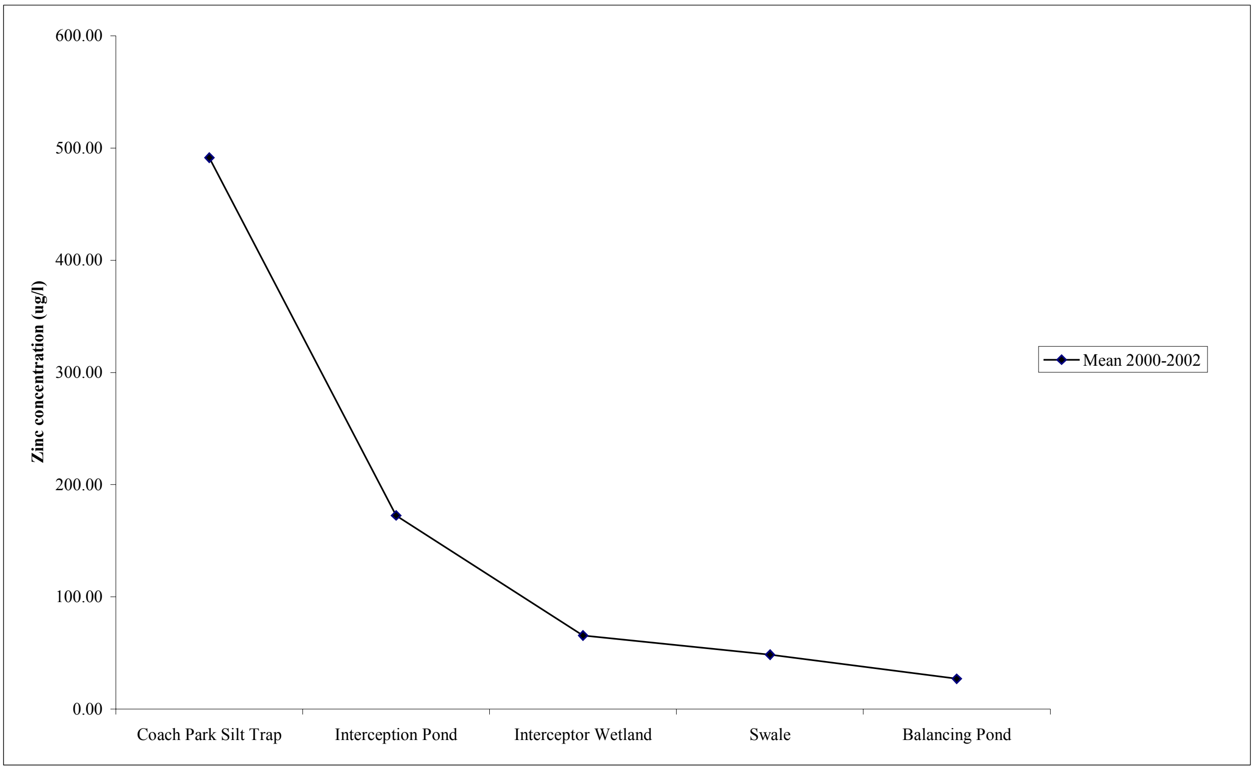


Figure 23 Zinc concentrations in the Coach Park SUDS system at Hopwood Park MSA

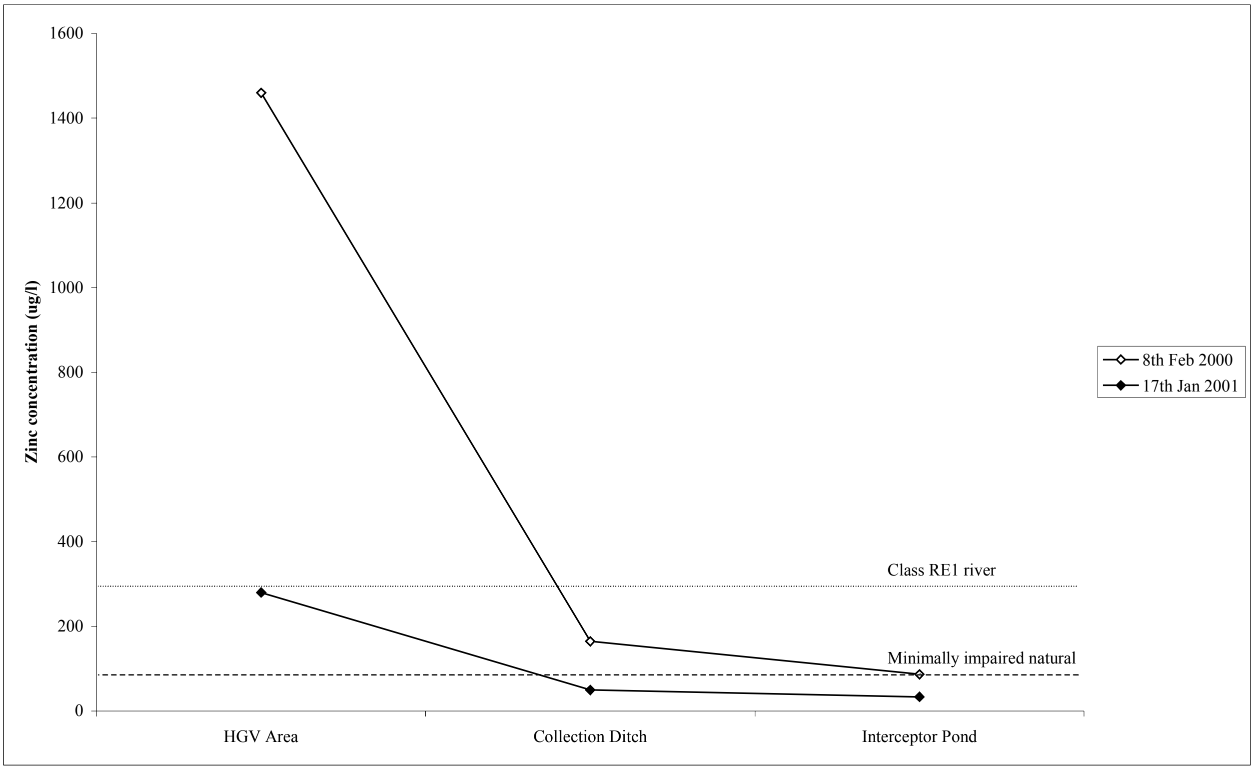


Figure 24 Zinc concentrations in the HGV Park SUDS system at Hopwood Park MSA

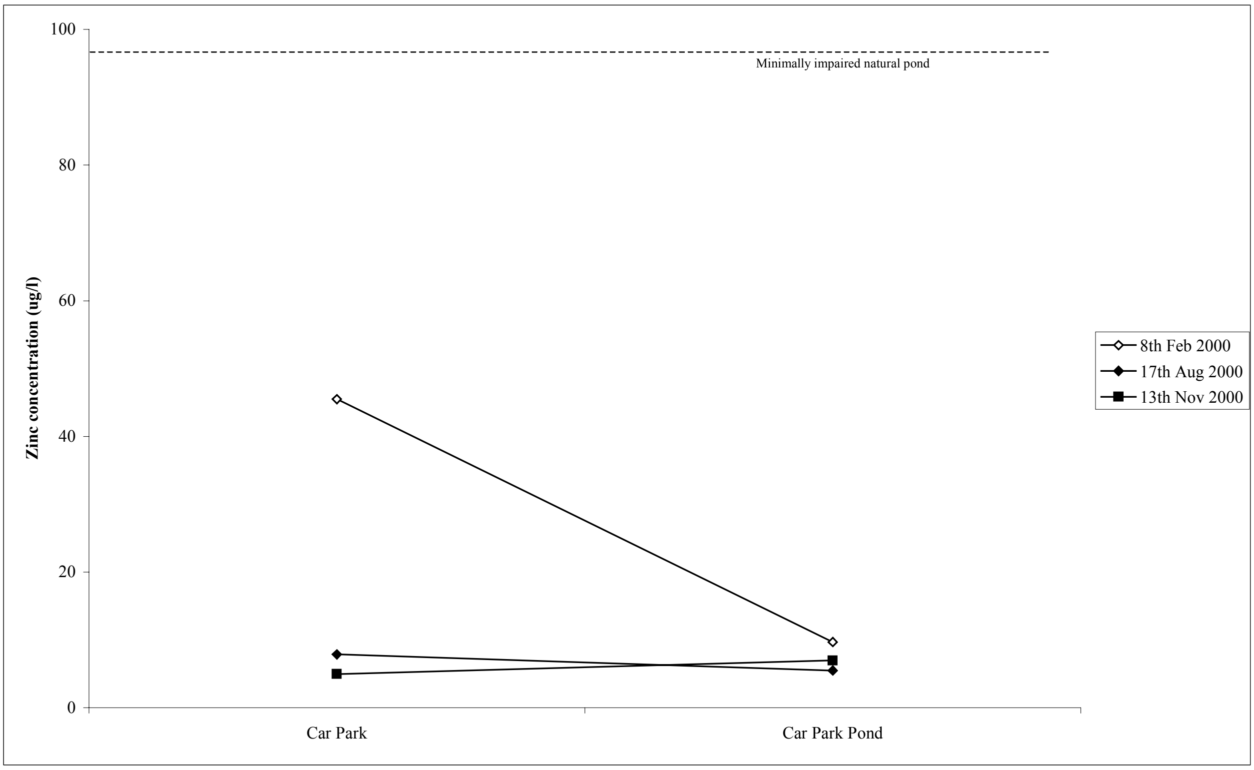


Figure 25 Zinc concentration in the Main Car Park SUDS system at Hopwood Park MSA

Appendix 2

Scottish SUDS monitoring programme chemical analysis

Appendix 2. Scottish SUDS monitoring programme chemical analysis

2.1 Suspended sediments

Data on suspended sediments were available from seven sites in the Scottish SUDS study (Figure 26). For the Royal Bank of Scotland porous paved car park, the mean suspended solids outflow concentration was derived by taking the mean of the maximum concentration for spot samples (16.1 mg/l) and for events (68 mg/l). No data are presented on the number of samples taken or events recorded. Note also that for Halbeath and Linburn Ponds no suspended solids means are presented, only maxima being reported. These values are shown in Figure 26.

The results show that sediment outputs are generally low from the sites' porous surfaces and ponds with values close to the levels seen in minimally impaired ponds. Swales, however, have considerably higher sediment outputs with levels well above those of minimally impaired waters.

Sediment concentrations of less than 25 mg/l are regarded as unlikely to damage fish populations. Outputs from ponds and porous surfaces are close to this level in the Scottish study; outputs from swales are considerably above this (Figure 26), and at levels likely to pose moderate or high risk to fish populations (100 mg/l and 200 mg/l suspended sediments respectively).

Appendix Table 1. Scottish SUDS sites at which performance data are being collected by the Scottish Universities SUDS Monitoring Programme

Device type	Description
Swales	
Emmock Wood	Swale on access road in Dundee housing estate
West Grange Swale	Swale on access road in Dundee housing estate
Detention Basins	
Duloch Park (Dunfermline Expansion Scheme)	No water quality data: a number of detention basins have been investigated in the Duloch Park area of the DEX Scheme but no water quality data have been collected.
Infiltration Systems	
Lang Stract Filter Drain	No water quality data available.
Aberdeen University soakaway.	
Pervious Surfacing	
Royal Bank of Scotland	Porous paved car park at the headquarters of the Royal Bank of Scotland in the South Goyle area of Edinburgh. Water quality data were collected over one summer.
NATS Car Park	Extension to car park of the National Air Traffic Control Service at Corstorphine.
Retention Ponds	
Claylands and Newbridge Ponds:	New ponds serving motorways west of Edinburgh. Claylands Pond is fed by drainage from an agricultural area; Newbridge Pond has limited habitat development owing to the need to reduce use by birds as the site is near to Edinburgh airport.
Halbeath and Linburn Ponds	New ponds in Duloch Park, constructed as part off the DEX development.
Stenton, Glenrothes:	A 17 year old pond which was originally constructed as a flood storage basin.

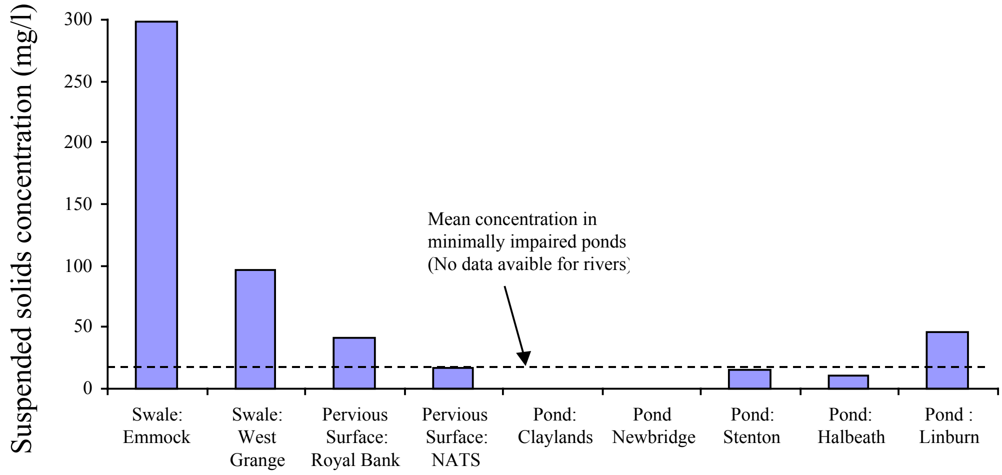


Figure 26 Suspended sediments concentrations in outflows from SUDS devices in Scotland

2.2 Biochemical Oxygen Demand

Data on biochemical oxygen demand (BOD) were available from five sites in the Scottish SUDS study (Figure 27).

Habitat quality: No data are available on the natural BOD levels in minimally impaired ponds in the UK. However, given that unpolluted ponds would be expected naturally to have higher BODs than flowing waters, values of less than 2.5 mg/l can probably be regarded as indicative of high quality ponds. Thus, with the exception of the West Grange swale, water quality in the Scottish SUDS devices can be regarded as likely to provide a good pond environment in terms of BOD.

Receiving water impacts: Outputs from all of the monitored Scottish SUDS devices, except the West Grange swale, were close to the 90%-ile value for high quality ponds in England and Wales. They are likely to have a minimal impact on receiving waters in terms of BOD. The output from West Grange, however, was of considerably poorer quality and would be equivalent to River Ecosystem (RE) Class 3 in the Environment Agency system. RE Classes run from 1 (highest) to 5 (lowest) quality.

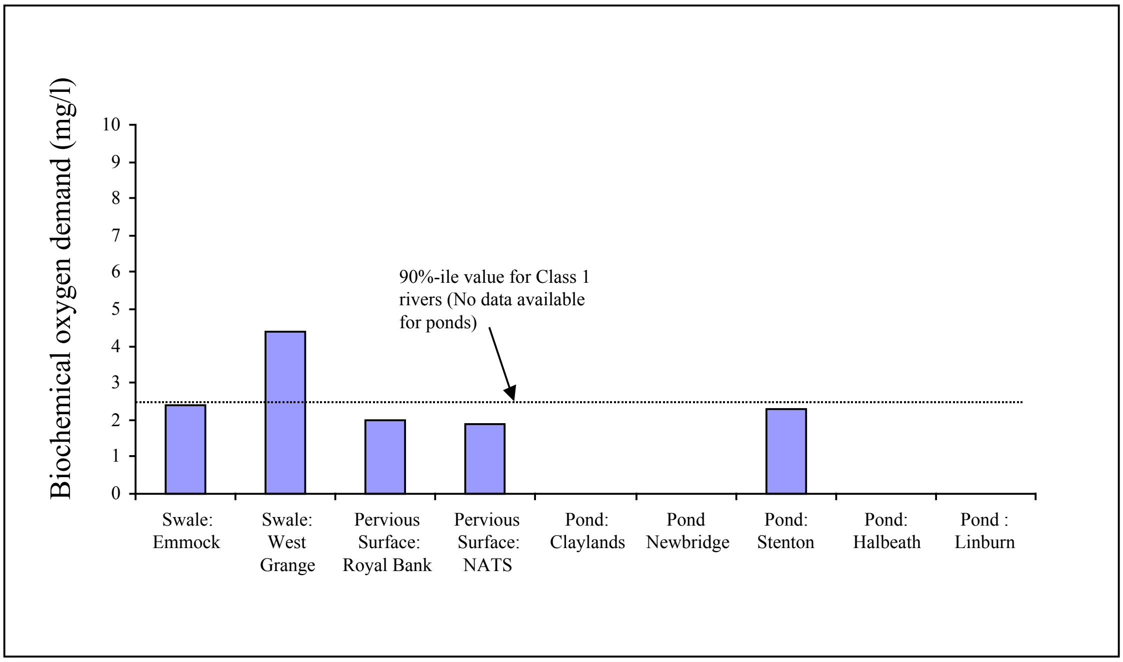


Figure 27 Biochemical oxygen demand of outflows from SUDS devices in Scotland

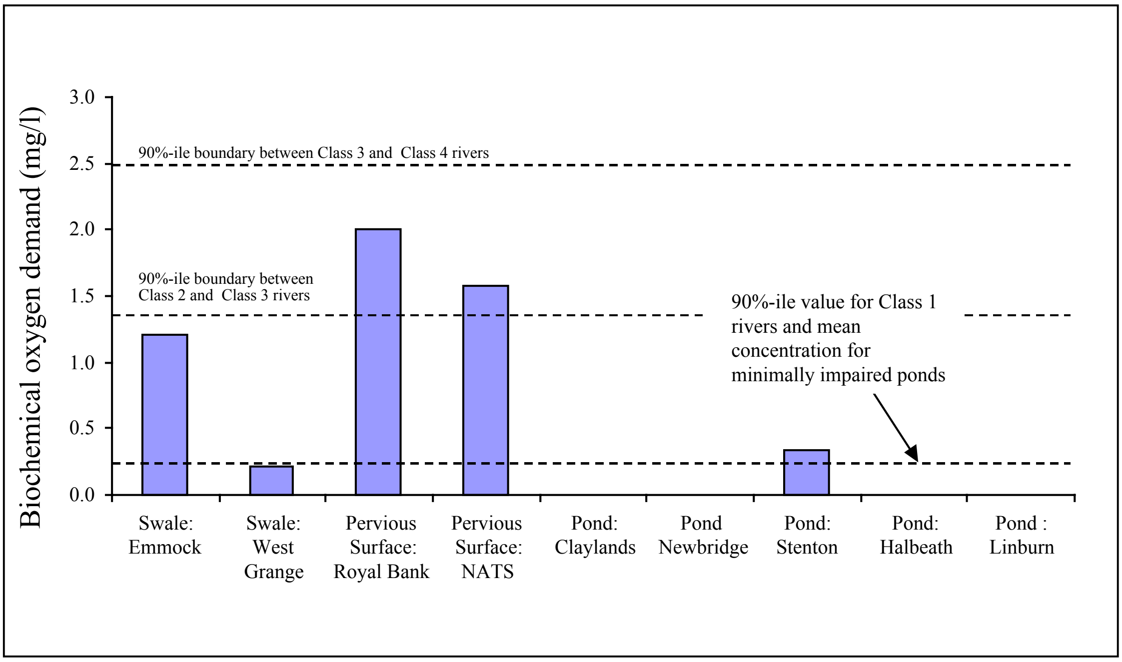


Figure 28 Total ammonia concentrations in outflows from SUDS devices in Scotland

90%-ile value for Class 1 rivers = 0.25 mg/l
90%-ile boundary for Class 2 and 3 rivers = 1.3 mg/l.
90%-ile boundary for Class 3 and 4 rivers = 2.5 mg/l

2.3 Ammonia

Data on total ammonia concentrations were available from five sites in the Scottish SUDS study (Figure 28).

Ammonia concentrations varied considerably in SUDS devices in the Scottish study. Two sites, one pond and one swale, produced low total ammonia outputs. The remaining three sites had considerably higher ammonia outputs which would lead to significant environmental impairment.

Habitat quality: The average total ammonia concentration in minimally impaired ponds is 0.27 mg/l. In practice, some SUDS devices in the Scottish study generated outputs well above this and habitat quality would almost certainly be impaired by such high ammonia concentrations.

Receiving water impacts: The higher ammonia concentrations produced by some of the Scottish SUDS devices would also be likely to cause significant impacts on receiving waters without considerable and rapid dilution. Note that both pervious surfaces produced high ammonia concentrations.

2.4 Cadmium

Data on cadmium levels in SUDS systems are available from four devices in the Scottish monitoring programme, two pervious surfaces, a swale and a pond (Figure 29).

Cadmium levels reported in the Scottish SUDS study are all below the drinking water limit identified as an Environmental Quality Standard for this substance in Jeffries (2001). However, in terms of habitat quality or receiving water protection the US EPA Chronic Value is a more relevant measure of potential impacts.

Cadmium levels in outputs from Scottish SUDS devices were relatively low but exceeded US EPA Chronic Values in three of the four sites for which data are available. Three sites were also close to or above the level for most (75%) European rivers. There was no obvious pattern in the effectiveness of devices.

Overall, these values indicate that in ponds, cadmium could cause chronic impacts on aquatic biota; in rivers, chronic impacts could occur unless there was further dilution.

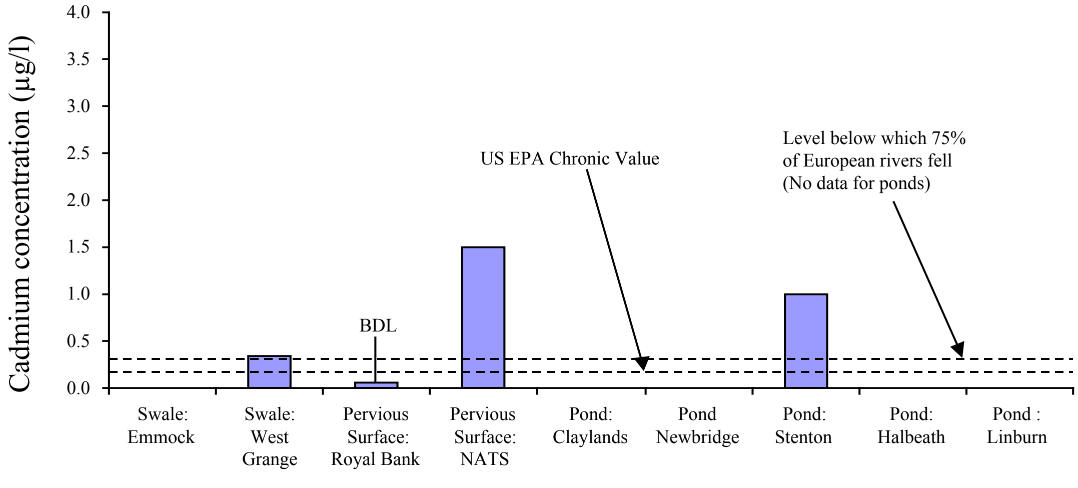


Figure 29 Cadmium concentrations in outflows from SUDS devices in Scotland

2.5 Lead

Data on lead concentrations are available from four Scottish SUDS devices (Figure 30).

Generally, lead levels were relatively low, with three out of four sites having values below the US EPA Chronic Value, and all four sites below the mean for minimally impaired ponds.

Overall these data suggest that lead outputs from the Scottish SUDS devices would generally be compatible with a high quality still water environment and would cause limited receiving water impacts.

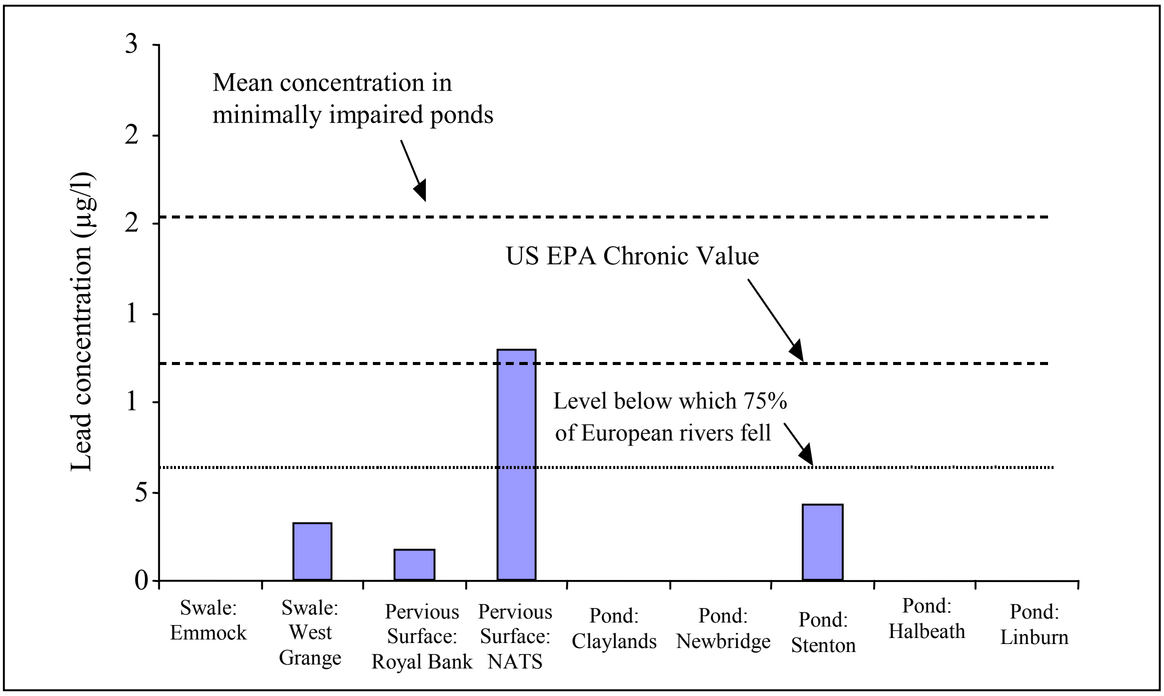


Figure 30 Lead concentrations in outflows from SUDS devices in Scotland

2.6 Chromium

Data on chromium concentrations are available from four sites in the Scottish study (Figure 31). Note that assessing the environmental concentrations of chromium is complicated by the differences in toxicity of chromium III and VI. The Scottish SUDS study presents only data on total chromium.

All measured values of total chromium were below the US EPA Chronic Value for chromium III. It can be assumed, therefore, that both still waters and receiving waters would not experience any impacts from SUDS systems with chromium III concentrations at these levels

For three of the four sites, measured total chromium levels are also close to the US EPA Chronic Value for chromium VI, suggesting that this would also pose little risk. Output from the pervious surface of the NATS Car Park could, theoretically, pose a risk from chromium VI.

All values measured are below the Environment Agency Environmental Quality Standard for total chromium (5 μg/l) in soft waters with calcium carbonate concentrations between 0 and 50 mg/l $CaCO_3$.

2.7 Copper

Data on copper concentrations in SUDS outflows are available from four Scottish sites (Figure 32). Note that, like other heavy metals, copper toxicity is related to water hardness. For this reason both soft and hard water limits are shown for river waters.

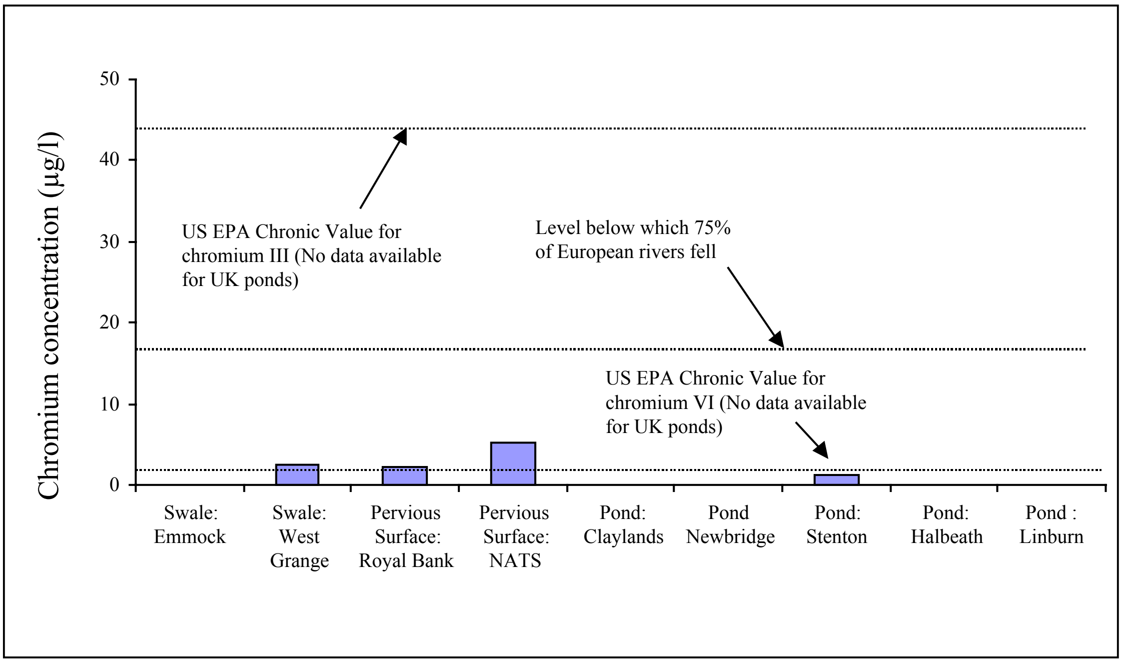


Figure 31 Chromium concentrations in outflows from SUDS devices in Scotland

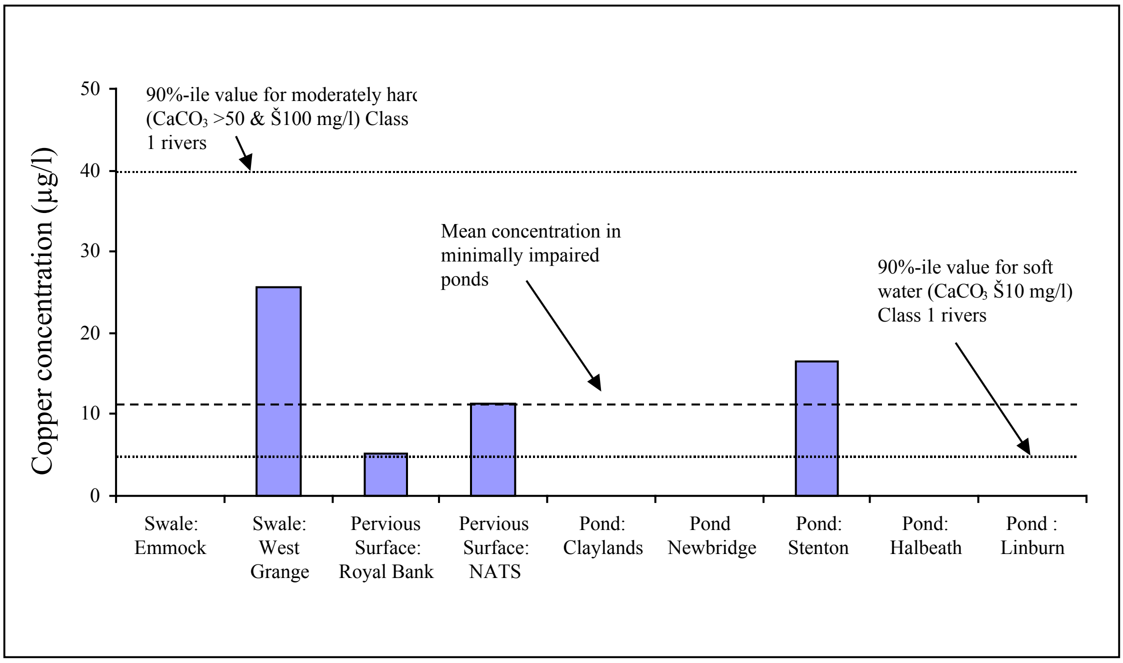


Figure 32 Copper concentrations in outflows from SUDS devices in Scotland

Habitat quality: In terms of habitat quality, two of the monitored SUDS systems, both pervious surfaces, had outputs which were equal to, or below, the mean for minimally impaired ponds. The two other monitored systems, a swale and a pond, had mean copper concentrations which were up to twice the minimally impaired pond mean. This indicates that some impacts on biota due to copper might be expected in ponds with water derived from SUDS.

Receiving water impacts: The impact of copper on receiving waters would depend on the hardness of the receiving water. Given that many Scottish rivers have low calcium carbonate concentrations, the lowest copper quality band is probably most relevant. For soft waters, outputs from all four SUDS systems would be greater than the 90%-ile for Class 1 rivers (i.e. the value below which 90% of measured values would be expected to fall). This suggests that SUDS outputs could still create some copper toxicity related impacts in receiving waters.

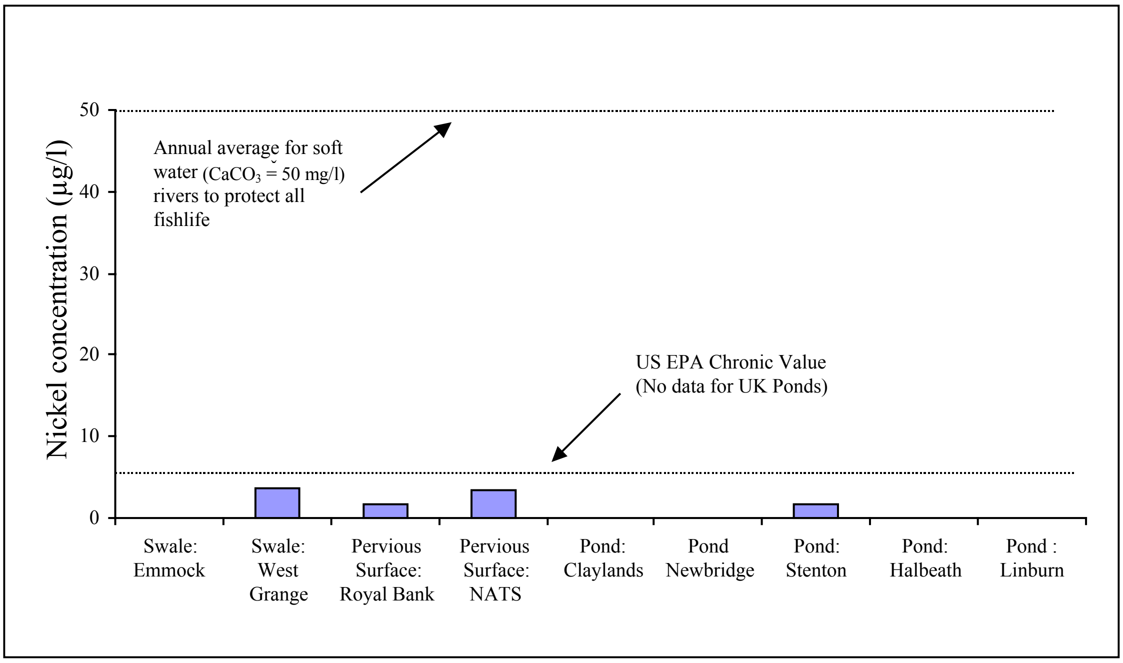


Figure 33 Nickel concentrations in outflows from SUDS devices in Scotland

2.8 Nickel

Data on nickel outputs are available from four Scottish SUDS sites (Figure 33). Mean values from all four sites were below the US EPA Chronic Value (5 µg/l) and well below the Environment Agency limit for the protection of fishlife in soft waters (50 µg/l).

Overall these data suggest that still waters or receiving waters would not experience impacts from nickel contamination after treatment by SUDS.

2.9 Zinc

Data on zinc concentrations from the Scottish SUDS project were available from four sites (Figure 34).

Habitat quality: Mean values from the Scottish SUDS sites all lie below the mean for minimally impaired ponds. This suggests that ponds with water of this quality would experience limited impacts from zinc contamination.

Receiving water impacts: Zinc toxicity is hardness dependent, and the potential impact of recorded zinc outputs from the Scottish SUDS sites therefore depends on the receiving water hardness. For soft water rivers, which are common in Scotland, two of the four sites had outputs with mean concentrations exceeding the 90%-ile for zinc for soft water Class 1 rivers (30 µg/l). Note that these sites also exceeded the US EPA Chronic Value. Two sites, one pervious surface and one pond, had zinc output values which were below the most stringent limits currently recognised. Overall, these data suggest that SUDS sites could cause zinc related impacts.

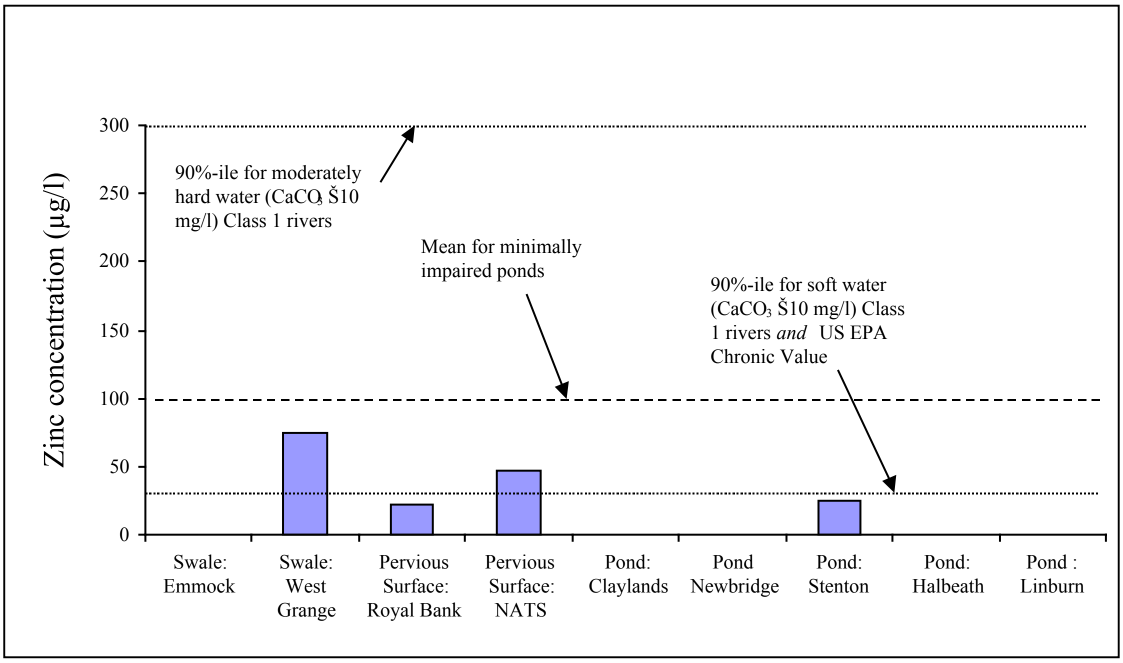


Figure 34 Zinc concentrations in outflows from SUDS devices in Scotland

Appendix 3

Analysis of effectiveness of individual SUDS components using data from American National Stormwater Best Management Practice database

Appendix 3 Analysis of effectiveness of individual SUDS components using data from American National Stormwater Best Management Practice database

3.1 Biochemical oxygen demand

Data on Biochemical Oxygen Demand are available from 10 sites in the North American Stormwater BMP database including retention ponds, wetlands, filters and swales (Figure 35).

No data are available on BOD levels in minimally impaired ponds so it is difficult to assess whether BOD levels in the outputs from North American SUDS devices would contribute to good quality still water habitats.

For receiving waters, mean BOD values in the SUDS devices are generally above the 90%-ile for Class 1 rivers in England and Wales with only two sites having mean BOD values below this level. This suggests that outputs from these SUDS devices would cause some impacts on receiving waters, unless there was significant dilution or waters were already degraded.

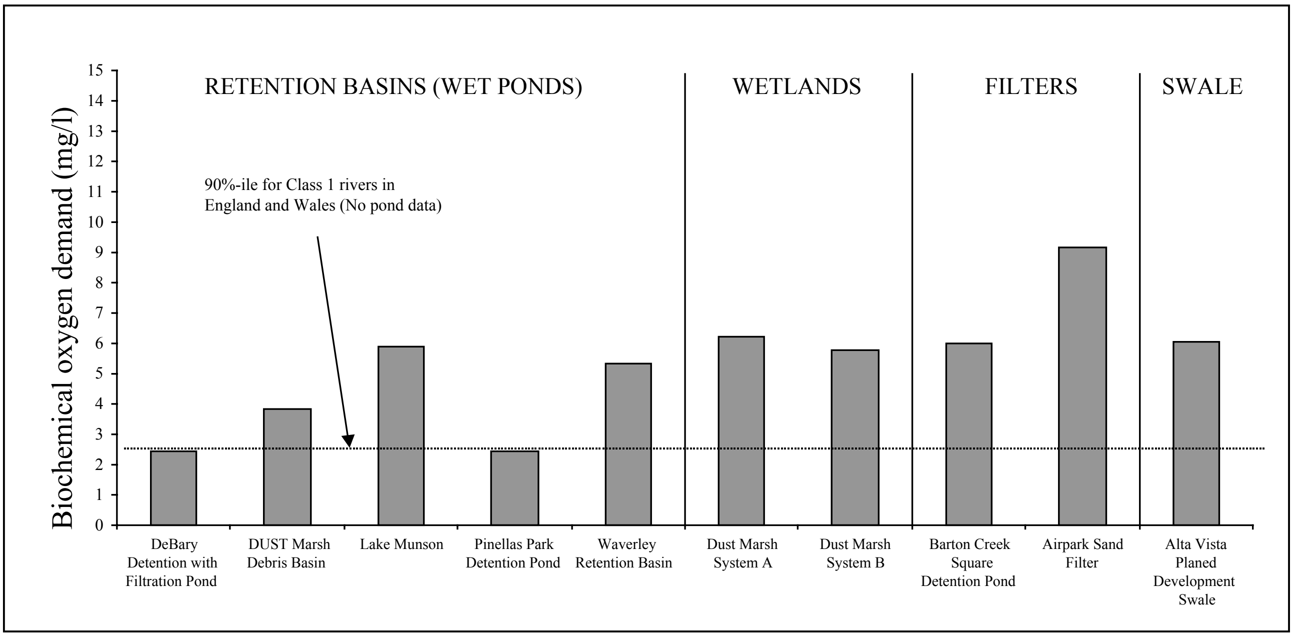


Figure 35 Biochemical oxygen demand concentrations in SUDS devices included in the North American Stormwater Best Management Practice database

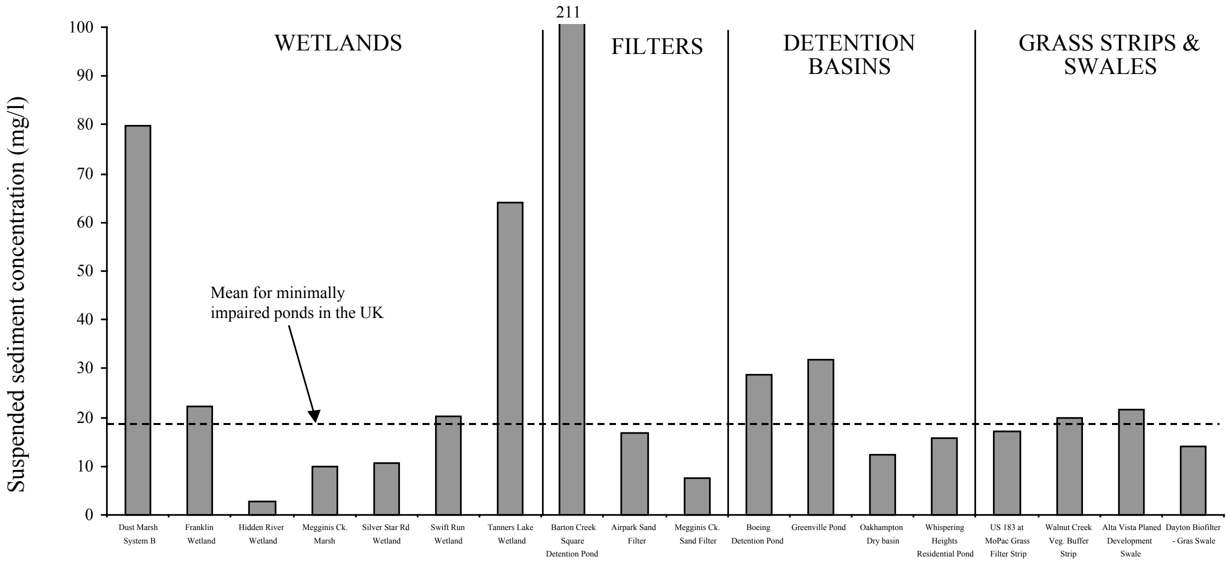
3.2 Total suspended solids

Data on total suspended solids are available from 36 sites in the North American Stormwater BMP database including retention ponds, wetlands, filters and swales (Figure 36).

Habitat quality: Total suspended solids concentrations are close to or below the mean for minimally impaired ponds in about one third of the sites studied. Remaining sites had higher than background suspended sediments concentrations, in a significant minority of cases markedly so. These data indicate that SUDS devices can produce good quality water in terms of suspended sediment but that, overall, they are very variable. It should be noted, as well, that data from the North American database is based on the outflow chemical quality. Since this is likely to be highest quality area in chemical terms, having passed fully through the waterbody and been exposed to most treatment, measurements here may give a rather optimistic impression of the water quality in the waterbody as a whole. Water quality at the upstream end of the pond may, therefore, be of poorer quality than in the outflow.

Receiving waters impacts: For receiving waters, no data are available in the UK to define minimally impaired conditions. However, EIFAC regard 25 mg/l of suspended sediment as the maximum at which unimpaired fish populations are likely to exist. This suggests that, in the absence of further dilution, SUDS outputs would cause minimal impacts on receiving waters in the majority of cases, although a significant minority of devices (around one third) would still produce potentially damaging effluents.

(a) Detention basins



(b) Other SUDS devices

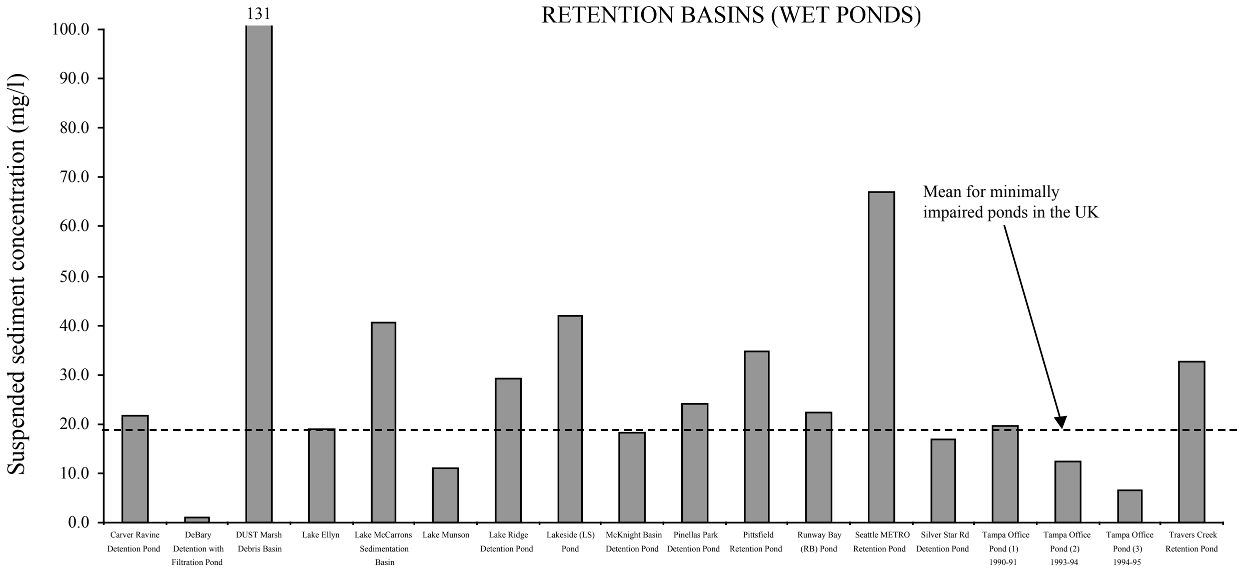


Figure 36 Total suspended solids concentrations in (a) detention basins (wet ponds) and (b) other SUDS devices included in the North American Stormwater Best Management Practice database

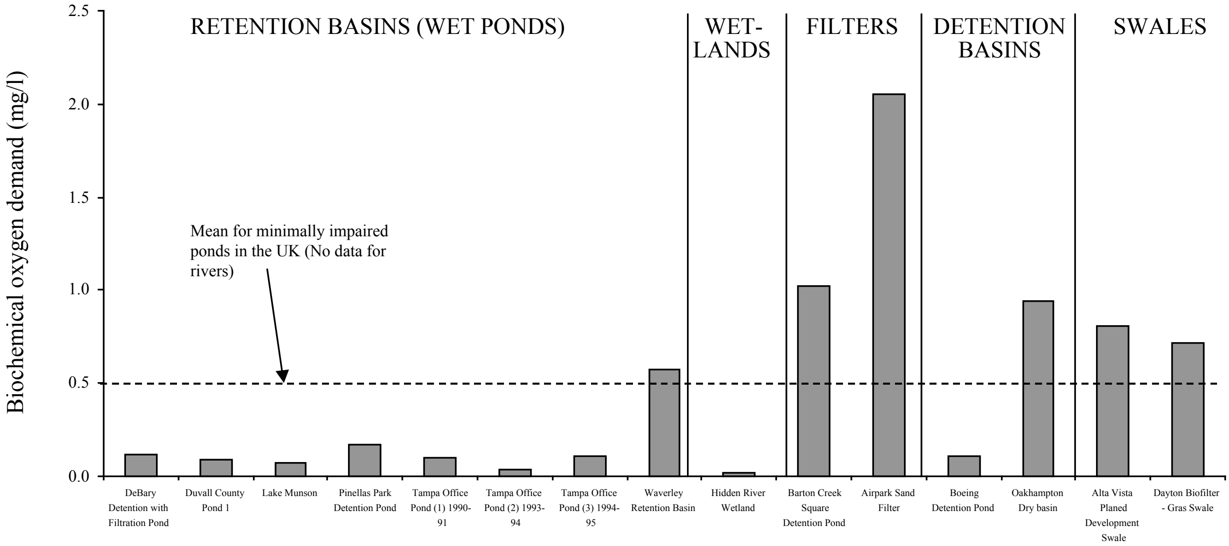


Figure 37 Total oxidised nitrogen concentrations in SUDS devices included in the North American Stormwater Best Management Practice database

3.3 Total oxidised nitrogen

Data on total oxidised nitrogen are available from 15 sites in the North American Stormwater BMP database. Most data are available from retention ponds, with more limited information from wetlands, filters, detention basins and swales (Figure 37).

Habitat quality: Total oxidised nitrogen concentrations in the outflows from SUDS devices were well below the mean for minimally impaired UK ponds in 60% of sites in the North American BMP database. The majority of these were retention basins (wet ponds). Remaining sites had higher than background TON concentrations, with the worst case 4x the minimally impaired mean value. Overall these data suggest that in terms of TON, some wet retention basins may provide good conditions (i.e. TON concentrations close to minimally impaired conditions), but a significant proportion of SUDS devices are likely to generate impacts due to TON.

For receiving waters, no data are available in the UK to define minimally impaired conditions in terms of TON.

3.4 Total phosphorus

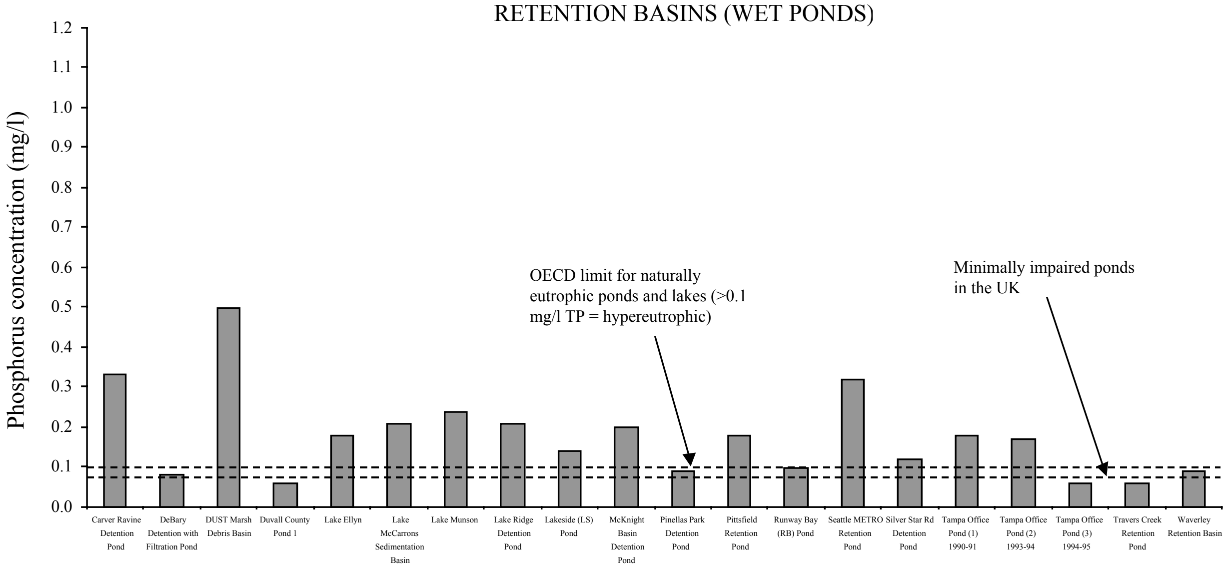
Data on total phosphorus concentrations are available from 38 SUDS devices, roughly half of which are ponds (Figure 38).

Habitat quality: In terms of habitat quality, most of the SUDS devices in the North American database had total phosphorus concentrations which were above the mean for minimally impaired ponds in the UK and above the internationally recognised OECD limit for naturally eutrophic waters (0.1 mg/l TP). Overall, the SUDS devices in the database had a mean TP output concentration of 0.26 mg/l, more than double the maximum value typically associated with naturally eutrophic waters. Only one site amongst those studied had TP outputs low enough to be regarded as oligotrophic or mesotrophic (Walnut Creek Vegetated Buffer Strip). In unimpacted catchments waterbodies with low total phosphorus levels should be quite common.

Receiving waters impacts: Potentially damaging levels of total phosphorus in receiving waters have not been defined in the United Kingdom but value in excess of the OECD limit for eutrophic standing waters can probably be regarded as potentially damaging in running waters unless further dilution occurs. This suggests that TP outputs from SUDS devices would be expected to cause damaging impacts on receiving waters in at least half of the cases considered in the North American BMP database.

There were no obvious patterns in the effectiveness of different device types with all apparently capable of producing both poor and good quality outputs in terms of TP.

(a) Retention basins



(b) Wetlands and other SUDS devices

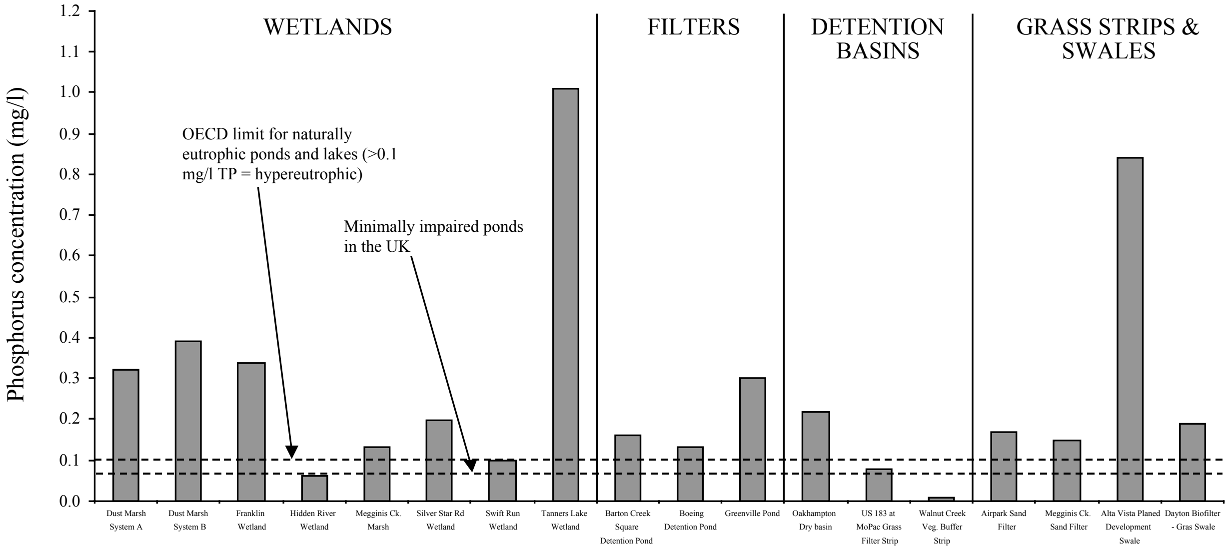


Figure 38 Total phosphorus concentrations in (a) retention basins (b) wetlands and other SUDS devices included in the North American Stormwater Best Management Practice database

3.5 Total ammonia

Data on total ammonia concentrations in SUDS device outputs are available from 20 sites in the North American BMP database. Most data are available from retention ponds and wetlands (15 sites), with more limited information from filters and detention basins (Figure 39).

Just over half of the devices produced total ammonia outputs below the level associated with minimally impaired ponds in the UK and Class 1 rivers. However, a significant minority of devices, mainly wetland, filters and detention basins, produced ammonia outputs which would be likely to cause biological impairment of either still water habitats or receiving waters.

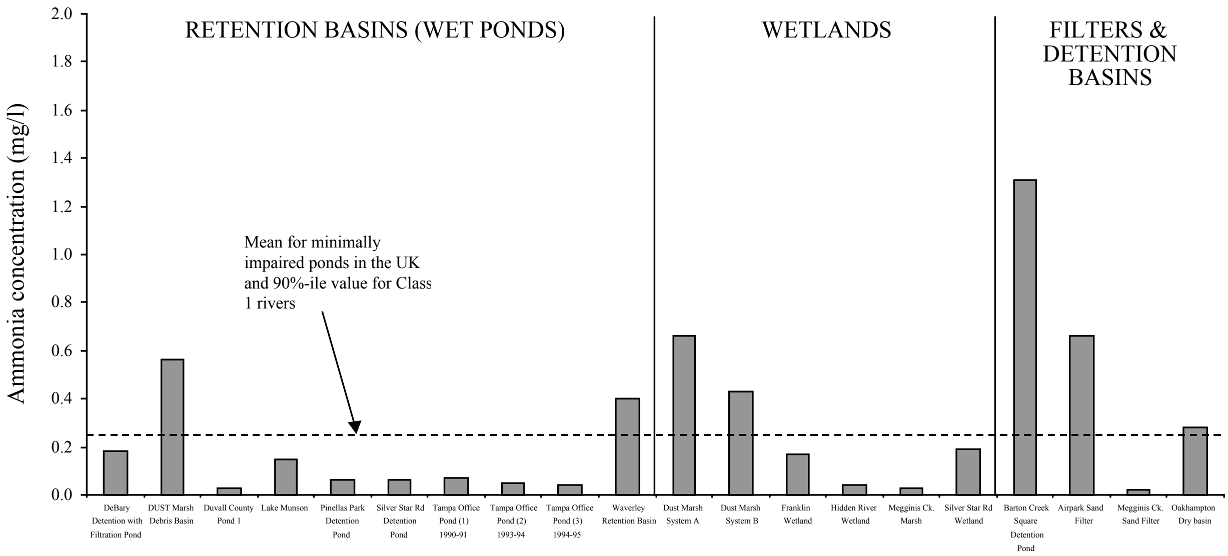


Figure 39 Total ammonia concentrations in SUDS devices included in the North American Stormwater Best Management Practice database

3.6 Cadmium

Data on cadmium concentrations in SUDS device outputs are available from 11 sites in the North American BMP database. Most data are available from retention ponds and wetlands (15 sites), with more limited information from wetlands, filters detention basins and swales (Figure 40).

Cadmium outputs from SUDS devices in the North American database are variable and generally relatively high. Only one site (Tampa Office Pond 3) had mean values below the US EPA Chronic Value. This suggests that in most sites in the database there would be a risk of biotic impacts due to cadmium toxicity in standing waters and, without further dilution, in receiving waters.

3.7 Chromium

Data on total chromium concentrations in SUDS device outputs are available from only seven sites in the North American Stormwater BMP database. Data are available from retention ponds, wetlands and detention basins (Figure 41).

Interpretation of chromium contamination in North American SUDS devices is complicated by lack of separate data on the two forms of chromium, Cr (III) and Cr (VI). It should also be noted that Cr (III) toxicity varies with water hardness, whereas Cr (VI) toxicity does not. With the exception of one site in the database (Dust Marsh System A), total chromium values were all less than 44 µg/l, the US EPA Chronic

Value for chromium (III). Four sites, however, had total chromium concentrations above 2.2 µg/l, the US EPA Chronic Value for chromium (VI), so could theoretically pose a risk for this compound.

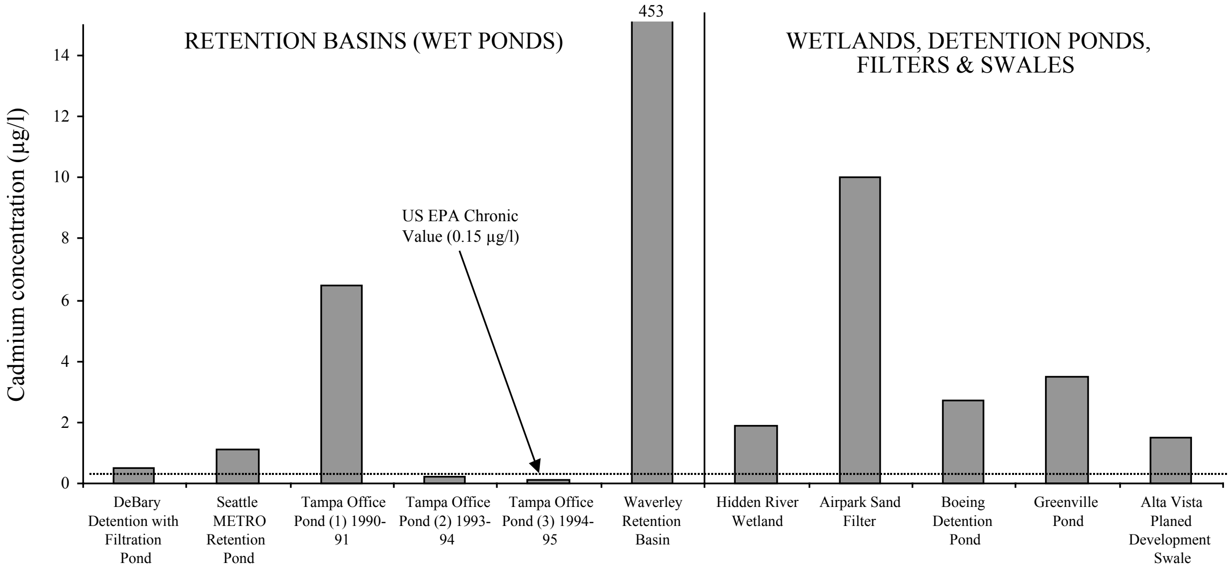


Figure 40 Cadmium concentrations in SUDS devices included in the North American Stormwater Best Management Practice database

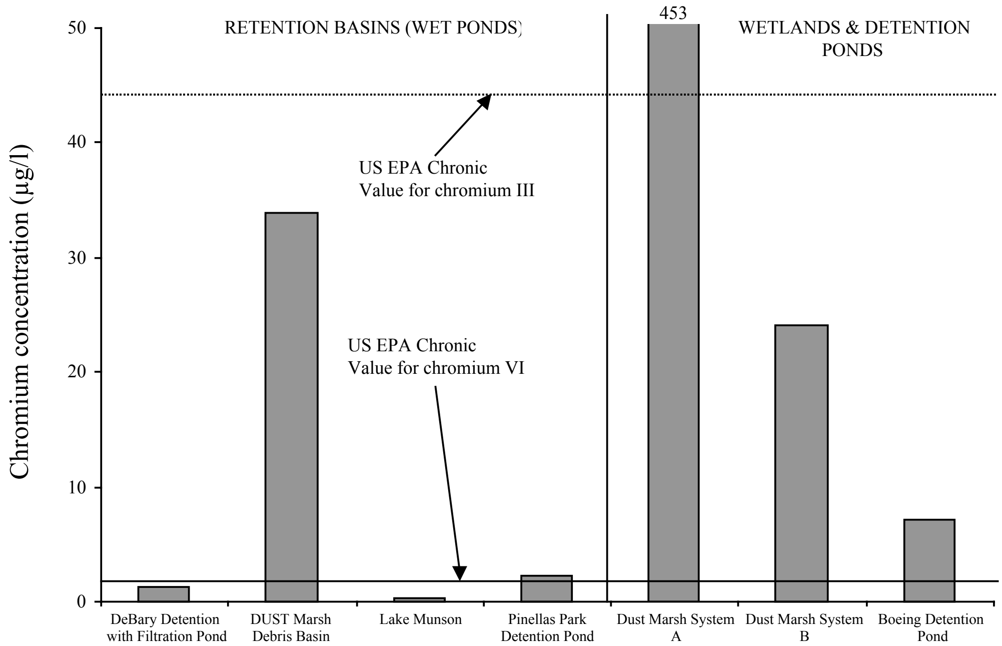


Figure 41 Chromium concentrations in SUDS devices included in the North American Stormwater Best Management Practice database

3.8 Copper

Data on copper concentrations in SUDS device outputs are available from 15 sites in the North American Stormwater BMP database. Data are available mainly from retention ponds and wetlands (Figure 42).

Habitat quality: Outflow concentrations from SUDS devices are generally quite low in retention basins (wet ponds) and wetlands: most being close to or below the mean value for minimally impaired ponds.

Other SUDS devices generally had somewhat higher outflow copper concentrations. This suggests that in terms of still waters, copper concentrations would probably cause relatively minor impacts. It should be noted that the relationship between copper toxicity and water hardness in ponds has not been studied but it is reasonable to assume that soft water ponds could experience some detrimental impacts from the copper levels recorded in the BMP database.

Receiving water impacts: Copper toxicity is dependent on water hardness and, for rivers, differential limits based on different water hardnesses are available. For soft waters, only four sites out of 15 are below the UK 90%-ile value for Class 1 rivers. In contrast, for moderately hard waters, all but one site had copper concentrations which were well within the limit for Class 1 rivers. This suggests that the copper outputs from SUDS devices could have an impact on receiving waters, depending on hardness and the extent of further dilution.

It should also be noted that all of the values for copper reported in the BMP database are *above* the US EPA Lowest Chronic Value which is only 0.23 µg/l copper (not shown on Figure 42).

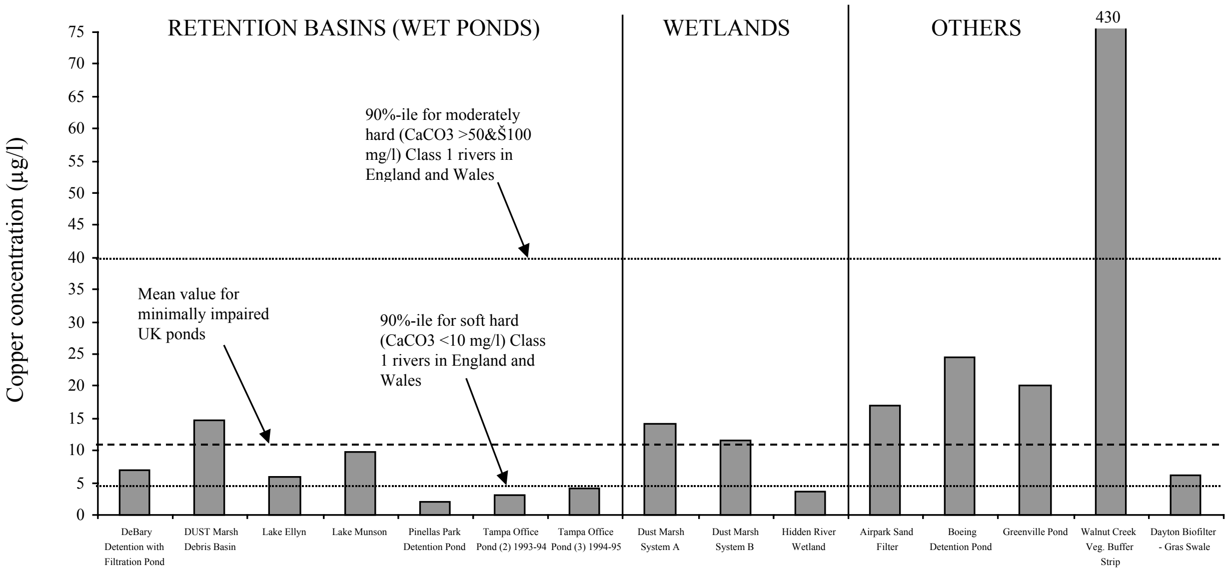


Figure 42 Copper concentrations in SUDS devices included in the North American Stormwater Best Management Practice database

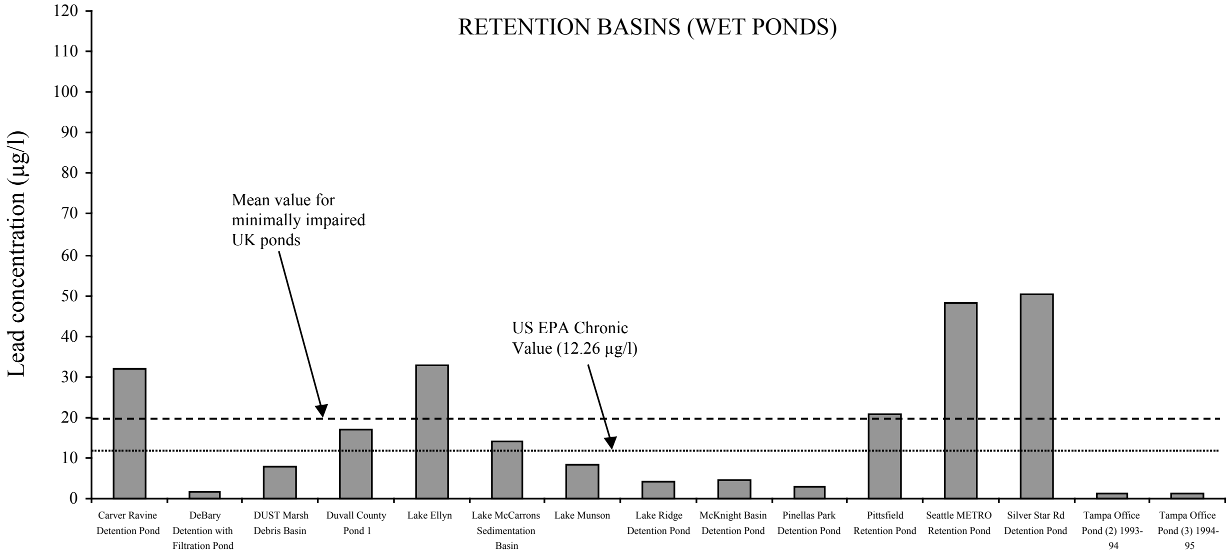
3.9 Lead

Data on lead concentrations in SUDS device outputs are available from 26 sites in the North American Stormwater BMP database of which over half are retention basins (wet ponds) (Figure 43).

Habitat quality: Lead concentrations in BMP database sites are very variable. Approximately half of the sites studied have mean output lead concentrations which are below the US EPA Lowest Chronic Value and also below the mean lead concentration in minimally impaired ponds in the UK. However, a significant minority of devices produced large outputs which would be liable to cause biological impairment. There is some indication that levels are lower in retention basins and wetlands than in other devices.

Receiving water impacts: There are no specific environmental limits for lead in UK rivers, although 90% of European rivers have mean lead concentrations of less than 11.0 µg/l lead (a value very close to the minimally impaired pond mean). This suggests that potential for receiving water impacts will be variable: for about half of the devices in the BMP database lead is likely to have little impact on receiving waters. However, for the devices with higher lead outputs biological effects might be experienced unless further dilution occurred in the receiving water.

(a) retention basins



(b) other SUDS devices

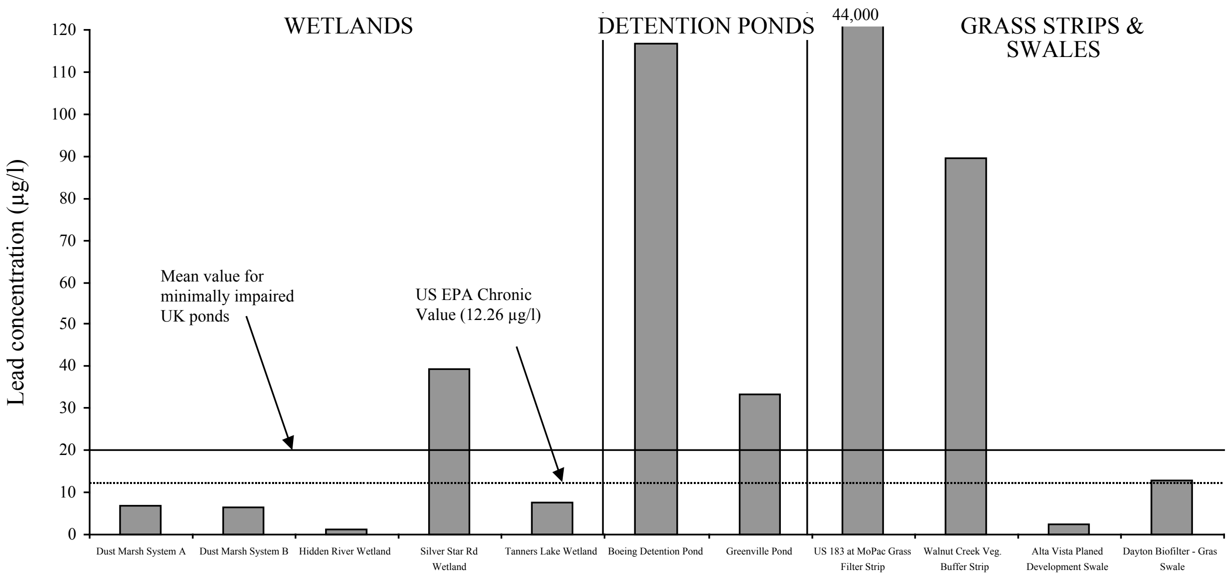


Figure 43 Lead concentrations in (a) retention basins (wet ponds) and (b) other SUDS devices included in the North American Stormwater Best Management Practice database

3.10 Zinc

Data on zinc concentrations in SUDS device outputs are available from 22 sites in the North American Stormwater BMP database of which over half are retention basins (wet ponds) (Figure 44). Interpretation of the data is complicated by the hardness related toxicity of zinc: generally, the levels seen in the outputs from the BMP database sites would be more problematic in soft than hard water.

Overall, zinc outputs from BMP database sites were relatively low compared to minimally impaired conditions. Most sites were below the mean concentration seen in minimally impaired ponds and about half were close to or below the 90%-ile for soft water Class 1 rivers in England and Wales (30 µg/l). This is also the US EPA Lowest Chronic Value. Virtually all sites in the North American database had zinc values which were below the limit for moderately hard Class 1 rivers.

This suggests that, in terms of habitat quality, zinc concentrations produced by SUDS devices would often be compatible with high quality habitats, although a significant minority of sites would be impaired biologically. In terms of receiving water impacts, these would be generally low or absent in hard waters but could occur in a significant minority of soft water sites.

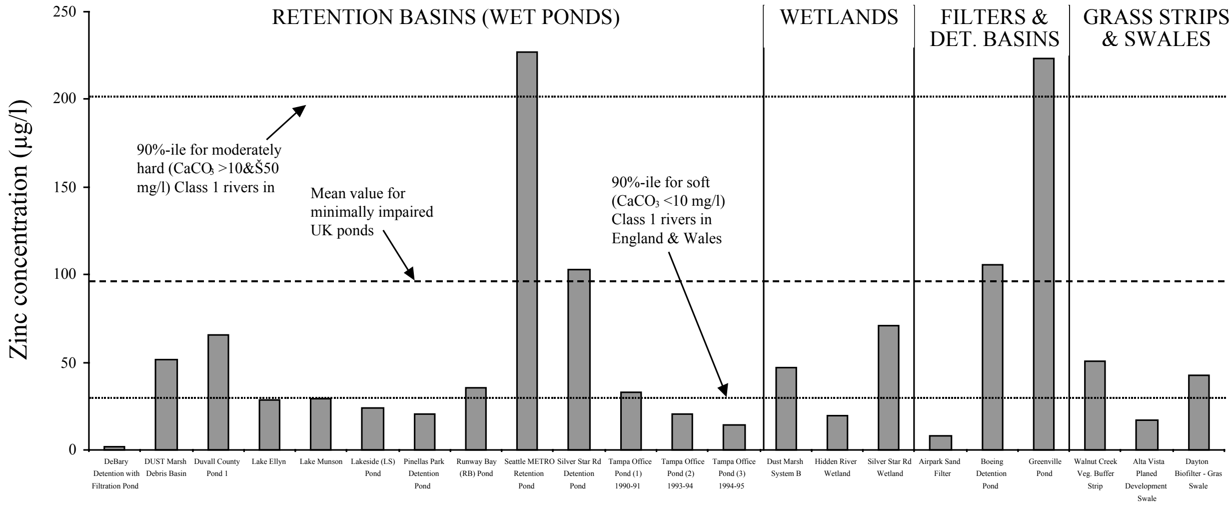


Figure 44 Zinc concentrations in SUDS devices included in the North American Stormwater Best Management Practice database

Appendix Table 2. Number of plant species and uncommon plant species in Scottish SUDS ponds. Revised data from Pond Action (2000) study

	M'rola Upper	*M'rola Middle*	*M'rola Lower*	*M'rola M'way*	*Freeport Upper*	*Freeport Lower*	*Houston Caw Burn*	*DEX Calais Marsh*	*DEX Halbeath*	*DEX Linburn*	*DEX Pond 5*	*DEX Retention 1*	*DEX Retention 2*
Number of native species	2	12	12	17	24	21	13	25	11	13	11	3	6
Number of alien or planted species	2	8	14	4	10	9	3	6	12	11	5	1	0
Number of uncommon species	0	0	0	0	0	0	0	2[1]	2[2]	1[3]	2[4]	0	1
Plant Conservation value*	Low	Mod	Mod	High	High	High	Mod	High	High	Mod	Mod	Low	Low

1. 0 species if stoneworts are excluded.
2. 1 species if stoneworts are excluded.
3. 0 species if stoneworts are excluded.
4. 1 species if stoneworts are excluded.

Appendix Table 3. Wetland plants: provisional categories for assessing the conservation value of ponds in Scotland

Low	Few wetland plants (≤6 species) and no local species.
Moderate	Equal to or below average number of wetland plant species (7-15 species) or 1-3 local plant species.
High	Above average
Very High	Supports at least one Nationally Scarce or RDB species or an exceptionally rich plant assemblage (≥28 species).

Appendix Table 4. Number of invertebrate species recorded from Scottish SUDS ponds surveyed by PCTPR (Pond Action 2000)

	Motorola Motorway	*Motorola Lower*	*Freeport Upper*	*Houston Caw Burn*	*DEX Calais Wood Marsh*
Number of species	40	37	58	24	40
Number of uncommon species	0	1	1	0	0
Conservation value*	High	High	Very high	Moderate	High

* see Appendix Table 4.2

Appendix Table 5. Number of invertebrate species recorded from Hopwood Park MSA ponds

Original data collected by Environment Agency

Pond	1		2		3		4		5		6		7		8	
Season	Sum	Aut	Sum	Aut	Sum	Aut	Sum	Aut	Sum	Aut	Sum	Aut	Sum	Aut	Sum	Aut
Number of species	9	10	18	12	11	10	25	12	13	12	25	19	17	24	29	37

Summer sample species richness with a 100% correction factor applied.

Pond	1	2	3	4	5	6	7	8
Season	Sum	Sum	Sum	Sum	Sum	Sum	Sum	Sum
Corrected number of species (corrected values obtained by increasing Environment Agency data by 100% are given in parentheses)	(18)	(36)	22	(50)	27	(50)	(34)	(58)
Corrected conservation value	Moderate	High	Moderate	High	Moderate	High	High	Very High