Stormwater Detention and Infiltration Devices Treating Road Runoff

By

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Statement of Authorship

I, Sara Kazemi Yazdi, hereby declare that except where explicit reference is made in the text of the thesis, this thesis contains no material published elsewhere or extracted in whole or in part from a thesis by which I have qualified for or been awarded another degree or diploma. No other person's work has been relied upon or used without due acknowledgment in the main text and bibliography of the thesis.

This research was conducted under the supervision of Dr Maiklas Scholz, as the principal supervisor and Dr Kate Heal as the second supervisor.

This thesis has been prepared to conform to the guidelines provided by The University of Edinburgh and is based upon the style recommended in the Publications Manual of Postgraduate Assessment Regulations for Research Degrees (Sep. 2006).

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Abstract

This thesis comprises five individual projects involving innovative solutions for the problems facing stormwater management in urban areas, problems like flood attenuation and pollution control (i.e. microbial contamination within stormwater runoff). These approaches include belowground stormwater detention systems, stormwater infiltration devices and bio-filtration.

The first study 'The Glasgow Sustainable Urban Drainage System (SUDS) Management Project' satisfies the first phase of the Glasgow Surface Water Management Project. This was Glasgow City Council's contribution to the Transformation of Rural and Urban Spatial Structure (TRUST) project, one of the European Union's (EU) interregional (INTERREG IIIB) funded research projects. The remit of this EU project comprised also other representative regions in Europe. The project showed also how SUDS can contribute to the overall catchment dynamics of cities such as Glasgow, ultimately relieving stress on the current predominantly combined sewer system. Fifty-seven sites within 46 areas of Glasgow were identified for investigation. A detailed soil chemistry analysis, a preliminary SUDS feasibility assessment and a desk study relating to historical planning issues that may be relevant for subsequent future development and regeneration options were undertaken. Detailed design and management guidelines were then drafted for selected representative demonstration areas (Belvidere Hospital and Celtic FC Stadium Areas) of high public and property developers' interest, and education

value. Combinations of infiltration trenches or swales with ponds or belowground storage were the most likely SUDS options for the majority of the demonstration areas. Soil contamination issues were considered when selecting SUDS because heavy metals such as lead and zinc can cause environmental health problems.

During the second study, 103 sites within Edinburgh were identified to assess the applicability of SUDS being integrated into future development, regeneration and retrofitting plans. A practical SUDS Decision Support Model based on a matrix and weighting system, incorporating the Prevalence Rating Approach for SUDS Techniques (PRABT) has been developed. The findings indicate that ponds (or lined ponds) and permeable pavement are the most likely individual SUDS techniques, and ponds combined with swales (or shallow swales) are the most recommended dual SUDS combination.

The aims of the third study were to assess constraints associated with the planning, design and operation of stormwater infiltration systems, the influence of aquatic plants on water quality and the overall water treatment potential. Runoff from a lightly trafficked road within The King's Buildings campus, mixed with dog faeces was used to simulate the real life conditions. The experimental site comprising a silt trap, a below-ground detention tank and two infiltration ponds (one planted and one unplanted) was fed by road runoff. Concentrations of suspended solids were frequently above international secondary wastewater treatment standards during the system set-up period that was mainly as a result of construction materials entering the system during the construction period. An analysis of variance indicated that the systems were significantly different in terms of most of their treatment performance variables. The system has shown higher reduction of microbiological contaminants in colder in comparison to warmer seasons. The nutrient analyses of the water samples showed significant reductions of nutrient concentrations within the infiltration ponds.

Next study examined whether multiple regression analysis and neural network models could be applied successfully for the indirect prediction of the runoff treatment performance with water quality indicator variables in an experimental stormwater detention system rig. Five mature experimental stormwater detention systems with different designs treating concentrated gully pot liquor were assessed in this study. The systems were located on The King's Buildings campus at The University of Edinburgh and were monitored for a period of eighteen months. Multiple regression analyses indicated a relatively successful prediction of the biochemical oxygen demand, and total suspended solids for most systems but due to a relatively weak correlation between the predictors, and both microbial indicators, multiple regression analyses were not applied for the prediction of intestinal enterococci, and total coliform colony forming units. However, artificial neural network models predicted microbial counts relatively well for most detention systems.

And finally the fifth study aimed to assess the performance of an experimental combined planted gravel filter, stormwater detention and infiltration tank system treating runoff from a car park and feeder road. An overall water balance of the system was compiled, which demonstrated that 33% of the rainfall volume left the system as evaporation, while of the remaining 67% approximately 8.9% was infiltrated and 11% was discharged into the sewer system. These findings highlighted the importance of evaporation in source control, and show that infiltration can be applied successfully even on man-made urban soils with low permeability. The assessment of the system's hydrological efficiency yielded mean lag times of approximately 3.6 h (gravel filter) and 8.0 h (entire system). Mean flow volume reductions of 73% and mean peak flow reductions of 80% were achieved compared to conventional drainage systems. The assessment of the pollutant removal efficiency resulted in promising removal efficiencies. Pollutant removal rates for the gravel filter were found to be high, ranging from 66% for nitrate-nitrogen to 95.83% for total solids. In contrast, with the exception of biochemical oxygen demand and suspended solids, the tank was associated with negative removal efficiencies. Ortho-phosphate-phosphorus concentrations considerably increased in the tank. Despite the generally poor performance of the tank, the proficiency of the filter assured that removal rates for the entire system were all positive. The lowest removal rate was for total dissolved solids (28%) and the highest for biochemical oxygen demand (98%). Despite of problems regarding hydraulics (i.e. clogging), the filter provided a valuable function with respect to water quality improvement. Finally, the Stormwater Management Model was applied to gain a deeper understanding of system processes and flow pathways. The model helped to quantify the runoff escaping the system due to flooding during strong storm events. The overall system performed well for low and moderate rainfall events, but inadequate for strong storms, if the filtration trench was clogged.

During the course of this PhD, the possible SUDS solutions for the problems arising from conventional drainage systems within the two largest cities in Scotland (Edinburgh and Glasgow) were investigated. Therefore, the lessons learned from the Glasgow and Edinburgh SUDS management projects were employed to design the experimental rigs within the King's Buildings campus. This resulted in gathering realistic laboratory-based data to be used in further studies such as the bio-filtration/ detention system investigated in this thesis. The outcomes of the first 4 projects lead to a better understanding in design and operation of stormwater detention/infiltration systems. As a result the experience earned was considered when planning for the construction of the life-sized stormwater bio-filtration, detention and infiltration device in the campus.

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Abbreviations

ANN	Artificial Neural Network
BMP	Best Management Practice
BOD	Five day at 20°C N-Allylthiourea Biochemical Oxygen Demand
CFU	Colony Forming Units
COD	Chemical Oxygen Demand
DO	Dissolved Oxygen
DSM	Decision Support Matrix
DST	Decision Support Tool
HRT	Hydraulic Retention Time
MSE	Mean Square Error
NTU	Nephelometric Turbidity Unit
PAH	Polycyclic Aromatic Hydrocarbons
PRAST	Prevalence Rating Approach for SUDS Techniques
Redox	Redox potential
SS	Suspended solid (SUDS Sustainable Urban Drainage System)
SWMS	Stormwater Management Model
TDS	Total Dissolved Solids
TS	Total Solids
TSS	Total Suspended Solids

Nomenclature

Alphabet Characters

A cross sectional area through the porous medium perpendicular to the flow,

a output of the transfer function,

b a scalar bias,

b_i bias input to the hidden node *I,*

c_{ij} weight connecting the hidden node,

f transfer function,

K saturated hydraulic conductivity,

m number of hidden nodes,

N number of data points,

n argument of the transfer function,

n transfer function net input

Q flow rate,

Qs water content of the medium

u_j input node *j*

V fluid velocity,

w_{ij} weight connecting the input node *j* to the hidden node *I,*

wp sum of the weighted input,

x_i output from the hidden node,

x_j weighted sum of inputs into the hidden node *j* to the output node *i,*

y_i output from the output node *i,*

Y_t observed output value,

$\hat{Y}_t$ output of a feed-forward neural network.

Roman Characters

$\Delta h/L$ hydraulic radiant, the difference in hydraulic head per unit distance in the direction of flow (L)

σ_i output function of the hidden node

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Chapter 1

Introduction

1.1. Background

Discharges of urban stormwater could cause various unfortunate effects on urban areas and on receiving waters including flooding, sedimentation erosion, rise in temperature and species extension dissolved oxygen depletion nutrient enrichment and eutrophication, reduced biodiversity, toxicity and the associated impacts on beneficial water uses as mentioned by Marsalek in 1998. Those impacts were aggravated by the conventional drainage systems and end of pipe solutions which are known to be both expensive and inefficient (Chocat *et al.*, 2001) and have raised concerns. Consequently, this led to the introduction of stormwater management, representing a combined system of control and treatment strategies designed to eliminate such impacts either fully or partly (Marsalek and Chocate, 2002).

Stormwater management measures also referred to as sustainable urban drainage systems (SUDS) are usually being implemented as a form of treatment trains representing a sequence of SUDS. In the last 30 years most leading countries have practiced SUDS. Therefore, SUDS can be considered as a mature approach to a more sustainable stormwater management (Ellis, 1995).

Accordingly, as the use of more natural drainage arrangements is being recommended as much as possible, technical objectives of sustainable urban drainage have to prioritise certain issues regarding the implementation of SUDS in urban environments. These issues include maintenance of an effective public health barrier; avoidance of local or distant flooding; avoidance of local or distant degradation/pollution of the environment (water, soil and air); minimisation of the utilisation of natural resources (water, nutrients, energy, materials); and reliability in the long term and adaptability to future (as yet known) requirement (Butler and Parkinson, 1997) .

This list can be expanded to include the broader requirements of community affordability and social acceptability (Butler and Davies, 2004).

Flood detention devices are the most commonly used engineering approaches towards controlling water quality and quantity impacts of stormwater runoff. Design principles for a single detention facility at a given location are fully established (Yeh and Labadie, 1997).

In the past few decades the detention of urban stormwater in detention tanks and ponds is a commonly adapted strategy for stormwater quality improvement. These detention systems are normally utilised to multi function concerning the urban landscape design, fauna and flora conservation, passive recreation and stormwater pollutant control. The effectiveness of the detention systems in the removal of pollutants from stormwater is dependants upon a number of factors many of which influenced by the detention period of the pollutants. The characteristics of stormwater inflow in detention systems are highly stochastic in terms of the intermittent nature of inflow and alternating shapes of the inflow hydrographs and pollutographs. Therefore, these systems function over a broad range of hydrodynamic and pollutant loading conditions. Consequently, stormwater pollutant detention period and removal performance is expected to vary accordingly.

When designing a stormwater detention device, the influence of the system hydrology (i.e. inflow characteristics, preliminary storage conditions and outlet hydraulics) and the characteristics of the pollutograph on the long-term performance of such systems should be considered (Metzger *et al.*, 2008).

This thesis refers to a number of publications presented in various journals by the author and her colleagues at the University of Edinburgh. Each of the studies described in this thesis are summarised in the mentioned publications. The Glasgow sustainable urban drainage systems management project is presented in Appendix 1 (Scholz *et al.*, 2006). Assessing stormwater detention systems treating road runoff with an artificial neural network is presented in Appendix 2 (Kazemi Yazdi and Scholz, 2008). The Stormwater infiltration systems for road runoff contaminated with organic matter including dog faeces is presented in Appendix 3 (Nanbakhsh *et al.*, 2007, Scholz and Kazemi Yazdi, 2005) and the combined bio-infiltration, stormwater detention and infiltration system treating road runoff is presented in Appendix 4 (Scholz and Kazemi Yazdi, 2008).

1.2. The Current State of Knowledge

In April 2000, the Commission of European Communities established a Community Initiative concerning trans-European co-operation, known as INTERREG IIIB. The INTERREG IIIB initiative related to the whole of the European Union. One of the projects funded by this initiative was entitled Transformation of Rural and Urban Spatial Structure (TRUST). This project aimed to develop new approaches to both spatial planning and land use to meet the challenges of continuing urbanisation, along with reducing economic loss and reduction in biodiversity through the development of integral management methods. The theme of TRUST was based upon multi-functional water storage, integral surface water management, and public and stakeholder participation. Six different authorities and institutions throughout Europe contributed to this project. Glasgow City Council's contribution to the TRUST project is known as the 'Glasgow Surface Water Management Project'. The project proposed innovative urban drainage recommendations. The first study output was the 'The Glasgow Sustainable Urban Drainage System Management Project'. This thesis investigates in depth the process of providing a SUDS implementation guideline for the city of Glasgow. The thesis illustrates how different measures should be taken in account prior to design and development of SUDS. It also shows how SUDS can contribute to the overall catchment dynamics of cities such as Glasgow, ultimately relieving stress on the current predominantly combined sewer system.

On 17th June 2004 the Edinburgh and Lothians Structure Plan (ELSP) was approved by Scottish Ministers. This plan provides a long term planning vision for development and the environment in Edinburgh and the Lothians until the year 2015. It replaces the Lothian Structure Plan 1994, which received approval from the Secretary of State for Scotland on 4th July 1997. This plan had a time horizon of 2005 but established a strong long-term settlement strategy to last for more than the initially expected deadline. The recent plan builds on that strategy and extends the strategy to a new horizon of 2015(Lothian Councils, 2004).

The Edinburgh and Lothians Structure Plan 2015 states the following: "The Lothian Councils, in consultation with the Scottish Environmental Protection Agency, Scottish Water and development industry interests, will review the risk of flooding in the structure plan area and consider altering the plan if the review shows that strategic development allocations are affected. The potential for flooding inland and on the coast will be considered in every local plan. Development, individually and/or cumulatively, that may lead to a significant increase in the risk of flooding, or that may itself be at risk from flooding, should not be permitted. Development proposals for Greenfield and Brownfield sites should include sustainable drainage systems for the attenuation and treatment of surface water and to assist in reducing the risk of flooding unless local conditions prevent this approach." (Lothian Councils, 2004). As a result, with considerable number of developments projected for the Lothian region, there was an immediate need for a SUDS implementation guideline to be developed for the City of Edinburgh. The second study of this thesis examines the processes used to produce such a guideline. The guideline provides the decision makers and the parties involved in the regeneration plan with a comprehensive and easy to use support tool when considering SUDS implementation in the city. The tool includes measures that help to choose the optimal SUDS technique for each representative site that will eventually lead to control urban runoff and reduce the possibility of flooding by rivers draining from the adjoining areas.

Understanding the chatchment characteristics and dynamics are critical to keep our surface water sustainable for survival of humans and biological organisms. Biological data are complex and difficult to analyse when considered to use for revealing the system properties properly. Various variables of organisms and environmental factors are involved in a complex manner in different ecosystems, spanning toward trans-disciplinary holism (Park and Chon, 2007). Necessity of the integrative and adaptive models to cover the non-linearity in the system is envisaged in stormwater biological information processing. Conventional statistical multivariate methods are restricted because they are mostly applicable to linear data and have less flexibility in interpreting biological data. In recent years, techniques in biological informatics have been developed from an interdisciplinary framework of the learning methods in computational science and biology (Park and Chon, 2007). However, there was still a need for more complex computational models to be utilised in order to promote the use of advanced machine-learning techniques for clarifying the principles of information processing. Therefore, to meet this requirement, an artificial neural network model was developed in the third study of this thesis to predict the biological contamination of stormwater detention and infiltration devices.

To date, no data have been reported in the literature on stormwater detention/infiltration systems' ability to remove pathogenic bacteria and more specifically two of its indicator species, total coliform and intestinal enrerococci. Various studies have been conducted to investigate the ability of other forms of SUDS to remove pathogens (Karim *et al.*, 2003 and Quiñónez-Dìaz *et al.*, 2001) but there is no evidence of such studies being done regarding stormwater detention facilities. Therefore, a study was carried out during the course of this research to investigate the microbiological removal efficiency of stormwater detention/infiltration facilities, manually contaminated by faecal contamination (Study 4).

There are tendencies to improve existing urban drainage systems rather than to design and construct completely new ones (Verworn, 2002). Stormwater is being regarded as a source to be managed. This includes the criteria of source control, in which stormwater runoff is not only being stored but also treated (via filtration or infiltration) within these systems, at or close to its generation point (Hatt *et al.*, 2004). Therefore, there is a continuing demand for below ground detention tanks as the most proper solution in many situations (NSC Council, 2002). There is an urgent need to modify common stormwater detention systems to meet more stringent water quality guidelines (Butler and Parkinson, 1997; Scholz, 2006). Research was needed to focus on the implementation of sustainable filters within the current structures of detention systems.

In Study 5 of this thesis a modern approach towards the use of below ground detention systems is recommended. The system being recommended here is a combined filtration, detention and infiltration device. This combined system assists in the control of both the quantity of runoff, through onsite detention and the quality of runoff through filtering and bio-infiltration of runoff.

1.3. Aims and Objectives

The purpose of this thesis is to investigate the feasibility and application potentials of various SUDS techniques combined with their capability to improve the quality of stormwater runoff.

To achieve this purpose this thesis comprises a series of studies:

Study 1: aimed to come up with SUDS demonstration areas (case studies) that are representative for both different sustainable drainage techniques and different types of areas available for development and regeneration in Glasgow. The objectives were to identify variables that determine the suitability of a site for the implementation of SUDS, identify suitable SUDS sites within the city of Glasgow, classify qualitatively and quantitatively sites suitable for different SUDS technologies, outline both a general SUDS decision support key and matrix, identify representative SUDS technologies for representative sites that could be used for demonstration purposes, provide detailed design and management guidelines and a brief cost–benefit analysis for representative sites and representative SUDS techniques for information and education purposes, and assess the soil contamination and the associated impact on environmental health.

Study 2: continuous of the first study 1, this study aimed to identify SUDS areas that were representative for both different sustainable drainage techniques and different types of areas available for development, regeneration and retrofitting in Edinburgh. A method of site investigation was developed with the resulting site data being used to develop the decision support tool. The objectives were to, identify suitable SUDS sites within Edinburgh, identify variables that determine the suitability of a site for the implementation of SUDS, outline and develop a decision support tool for Suds implementation in Edinburgh derived from site variables and SUDS feasibility matrices, represent a method to identify the best SUDS solution for a site, identify SUDS techniques for representative sites that could be used for demonstration purposes.

Based on the lessons learned from the SUDS management projects in Edinburgh and Glasgow, a series of 3 individual studies was undertaken to improve the design and operation guidelines of stormwater detention and infiltration facilities and to achieve an optimised treatment performance all year round in cold climates. The study can be split into the following sub-studies:

Study 3: aimed to investigate, the ability of machine –learning techniques in order to predict the microbial populations in the stormwater runoff, the system's microbial removal performance and the effectiveness of applying artificial Neural Networks (ANN) to predict the outflow water quality of the experimental rigs.

Study 4: this study aimed to, propose the combined stormwater detention and infiltration facilities as a more efficient approach for urban runoff treatment, examine the system's microbial removal performance and to investigate the potential effectiveness of C. Auratus upon controlling algae in stormwater infiltration ponds.

Study 5: the final study in this thesis aimed to suggest optimal design and operational guidelines for belowground stormwater detention systems to sustain the highest level of performance possible under various environmental conditions in cold climates. This included the introduction of a bio-infiltration device which can be beneficial to the treatment performance of such systems.

1.4. Outline of Thesis Contents

The following outlines the contents of this thesis:

Chapter 1. Introduction and literature review.

In this chapter, a brief background, the aims and objectives of this thesis are described. A short background of SUDS and mainly detention and infiltration facilities are also illustrated in this chapter, including main components, flow characteristics and removal mechanisms of pollutants. The findings of recent research into stormwater detention and infiltration devices are also discussed.

Chapter 2. Site description.

This chapter presents a comprehensive description of the study sites. This thesis consists of five individual studies, the Glasgow sustainable urban drainage management project, the Edinburgh sustainable urban drainage management project, the assessment of stormwater detention systems treating road runoff with an artificial neural network, the assessment of stormwater infiltration systems for road runoff contaminated with organic matter including dog faeces, and the combined bio-filtration, stormwater detention and infiltration systems treating road runoff. Each of the corresponding study sites are fully described in chapter 2.

Chapter 3. Materials and methods.

This chapter focuses on four individual studies. First is a SUDS feasibility study of the Belvidere Hospital and the Celtic FC Stadium area in Glasgow, this study was a part of the "The Glasgow Sustainable Urban Drainage System Management Project". The second study is a more comprehensive SUDS feasibility study concerning the city of Edinburgh. The other three studies describe the experimental setup and operation methods applied for differently designed experimental stormwater treatment systems treating road runoff. The systems include an experimental stormwater detention (extended storage) systems based on the Atlantis Water Management Ltd. detention cells receiving concentrated runoff that has been primarily treated by filtration with different inert aggregates, a combined SUDS system consisting of an below ground stormwater detention tank and a planted and an unplanted stormwater infiltration ponds and combined sustainable urban drainage system consisting of a gravel filter and infiltration detention tank in series. Systems' design and compositions as well as environmental conditions of the experimental systems are also portrayed.

Moreover, the application of an Artificial Neural Networks system in prediction of microorganisms in an experimental stormwater detention system is investigated, the chapter then investigates the treatment mechanisms and potentials for water quality variables in the studied sites. A simple removal model is applied to evaluate the removal potentials of the system. The overall performance of each system concerning nutrients and microorganisms as well as other water quality variables is also statistically compared in order to evaluate the efficiency of the design components as well as the operation conditions. Eventually, the chapter investigates the application of the U.S. Environment Protection Agency program SWMM model in order to understand the hydrological characteristics of a combined sustainable urban drainage system consisting of a gravel filter and infiltration tank in series.

Chapter 4. Results and discussions.

This chapter outlines a general SUDS decision support key and matrix for the Belvidere Hospital and the Celtic FC Stadium area in Glasgow and the representative sites in Edinburgh ; identifies representative SUDS technologies for the mentioned representative sites that could be used for demonstration purposes, provides detailed design and management guidelines, and a brief cost–benefit analysis for representative sites and representative SUDS techniques for information and education purposes; and assesses the soil contamination and the associated impact on environmental health.

The chapter also investigates the impacts of influent water quality and design factors of detention/infiltration systems on the performance, the fate of manually introduced microbial contamination in forms of dog faeces to the stormwater infiltration ponds are discussed as well as the systems efficiency regarding total coliforms and intestinal enterococci populations' reductions.

Furthermore, in this chapter the findings of a developed artificial neural networks model to predict total coliforms, and intestinal enterococci colony forming units are discussed. The findings are then compared with the findings from multiple regression models developed for the studies systems receiving concentrated runoff contaminated by dog faeces;

Chapter 5. Conclusions.

The thesis is concluded giving recommendations for further research potentials regarding stormwater detention and infiltration devices in chapter 5.

1.5. Drainage Systems

Urbanisation with current speed rate has resulted in higher interaction between human activities and the natural water cycle and consequently the need to design drainage systems to remove excess surface water. Commonly, the drainage system in most urbanised areas is comprised of a totally artificial system of sewers: pipes and structures that collect and dispose of water aiming to prevent local flooding by conveying the water away as fast as possible. Alternatively, in some smaller areas or areas without mains drainage, the runoff is infiltrated into the ground. This happens where the extent of urbanisation tends to be limited (Butler and Davies, 2000).

There are two major types of interaction between human activities and the water cycle to be found in an urban area: (i) water abstraction and, (ii) the replacement of the natural soil surface with impermeable surfaces that divert rainfall away from neighbouring natural drainage system. Therefore, there are two types of waters which require drainage (Butler and Davies, 2004).

Wastewaters, water arising from supplies that support life, maintain the living standards, and fulfil industrial needs. These waters require proper drainage after their use, otherwise, resulting in watercourse pollution, and creation of health risks (Butler and Davies, 2004).

And stormwater, waters remaining from any type of precipitation fallen on developed areas. These types of waters also demand appropriate drainage in order to prevent inconvenience, damage, flooding and possible health risks. Stormwater may hold pollution originating from the catchment surface, atmosphere or rain itself (Butler and Davies, 2000).

The purpose of the urban drainage systems is to manage the two types of water and to lower the risks and problems violating the environment and the human life. Initially, after each precipitation, one part of the rain water deposited on urban surfaces, returns directly to the atmosphere via evaporation and transpiration by plants; another infiltrates into the surface and ultimately reaches groundwater, the remainder though, runs off from the surface. Surface characteristics and storm duration control the relative proportions (Krebs and Larsen, 1997).

Urbanisation encourages more and more new developments that cover the ground by artificial surfaces resulting in much lower infiltration rates and consequently increased surface runoff and of course dramatic increase in the total volume of water reaching the rivers, immediately or soon after the rain (CIRIA, 2000).

The runoff travels with a much faster speed over harder surfaces or through sewers than it travels over natural surfaces and along natural streams. Hence, a higher peak flow and steeper rising and falling limbs in the storm hydrograph occur (Fig, 1), increasing the probability of sudden flooding. The rapid runoff washes off the pollutants and sediment from the surface and scours them from the rivers with a much quicker rate. Moreover, the artificial environments are more likely to be polluted than the natural environments both at the catchment surface and in the air. In general, urban drainage produces higher and more peaks in river flow and increases the risk of water pollution. Additionally, urban drainage tends to have adverse impact on aquatic ecology due to the loss of habitats, increased pollutions, higher stress and low regime (increasing the events of high and low flows) (Butler and Davies, 2000).

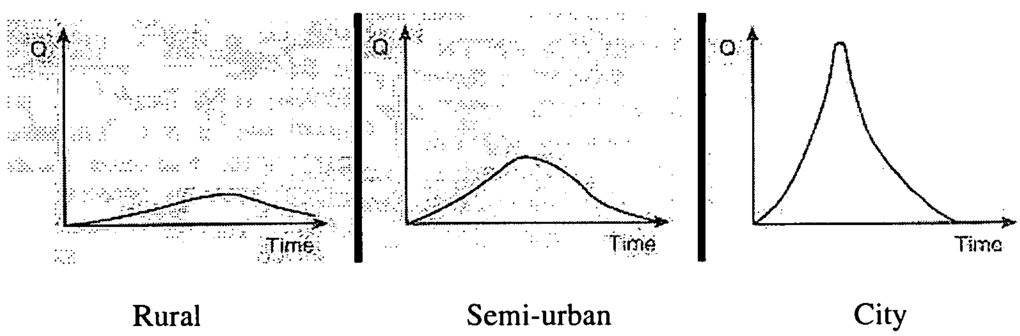


Fig 1. Effect of urbanisation on storm hydrograph, Q is discharge (Butler and Davies, 2000).

1.5.1. Problems with the Conventional Drainage Systems

In an urban infrastructure, urban drainage system seeks to avoid interference with the free movement of vehicles and pedestrians, material damage from storm flows, and risks to health and environment from precipitation (Silveira, 2001). When coupled with water supply facilities and sewage removal, urban drainage forms the basic structure of water management in urban areas. The conventional approach to the urban drainage emphasised on managing the quantity of storm runoff so as to avoid loss of life and damage to property (Sangare and Thibault, 1998).

It appears that the conventional solutions to urban drainage have reached the saturation point. The period of transition has arrived. A period in which the 19th century drainage model is being abandoned and rapidly replaced by more environmentally sustainably models (Silveira, 2001). The conventional drainage model was the consequence of storm drainage management practices which predominated the world for most of the 20th century. It was an idea which required rapid drainage of stormwater runoff without considering its effects on downstream waters (Silveira, 2001).

The conventional approach to urban drainage is still predominant in developing countries various reasons. Mainly, because the sustainable approaches are much more difficult and expensive to apply as they require collaborated action among greater parties, with multidisciplinary technical knowledge. The conventional approach on the other hand, considers local solutions derived by civil engineers only. Moreover, the outdated conventional approach is still being used due to its simplicity and because its required infrastructural works have a dramatically restricted behaviour and are simply designed (i.e. their function is rapid transport only) (Silveira *et al.*, 2001).

The conventional approach considers urban runoff as undesired water in developed areas which needed to be completely diverted from urban areas as quickly as possible. For many years, the combined sewer system was believed to be the most feasible and economic solution. Eventually, the established urban drainage system revealed significant weaknesses which derived the urban water decision makers to question the achievements of the conventional drainage system (Boller, 2004).

Major concerns include, peak hydraulic loads in sewers and treatment plants, sewer overflows and direct contamination of receiving waters with untreated sewage, temperature shocks in treatment plants and receiving waters, decrease of treatment plant performance, and additional marked loads of heavy metals and organic micro-pollutants from diffused sources in surface runoff (Boller, 1997).

Evidently, the pollution and contamination of watercourses and coastal waters from diffuse sources are the major problems both in rural and urban environments (D' Arcy *et al.*, 2000).

Urban runoff may contain heavy metals- primarily copper, iron and zinc- in toxic concentrations. Indicators of pathogenic microorganism such as faecal coliform bacteria also occur in urban runoff with relatively high concentrations (U.S. EPA, 1988). Clearly, conventional drainage systems are unable to properly control poor runoff quality. Importantly, the amenity aspects, like water resources, landscaping potentials, community facilities and provision of varied wildlife habitats are vastly ignored. Therefore, considering the debated issues there is an urgent need for more sustainable solutions water related problems in both terrestrial and aquatic environments.

1.6. Sustainable Drainage Systems

Sustainable urban drainage systems (SUDS) represent man made complex environmental systems which require proper support and maintenance to perform quantifiably and sustainably. These are management measures which are designed to provide environmental benefits without strict performance targets and full understanding of their long term operation and benefits (Krajewski *et al.*, 2000).

SUDS are dynamic environmental systems that mature over time and their performance may change (e.g. vegetation growth, species distribution and maturity, reduction of storage volumes/flow areas due to sediment deposition, clogging of the previous layers, storage of contaminated sediments susceptible to contaminant release and transfer of contaminants from sediment to the biota). Therefore, they cause secondary impacts on the environment which are not always well understood, or fully considered in the initial design. On that account, maintenance plays a crucial role to ensure the sustainability of SUDS and mitigation of secondary impacts. This includes both short term restorative measures and the long term preventative maintenance (i.e. rehabilitation of SUDS structures) (Bertrand-Krajewski *et al.*, 2000).

Sustainable solutions not only concern hydraulic criteria of stormwater but are also designed to consider its quality. These solutions can be developed by identifying the fluxes and control them by either source control measures or pollutant barriers including appropriate disposal of the accumulated waste (Boller, 2004).

The issues that sustainable urban drainage solutions should specifically address include, maintain an effective public health barrier and provide sufficient flood protection; avoid both local and more distant pollution of the environment (i.e. land, air and water); minimise the utilisation of natural resources; and be obtainable in the long term and adaptable to future requirements.

There are three strategies initially proposed to achieve progress towards sustainable urban drainage. These strategies being, minimising the inappropriate use of potable water as carriage medium in sewers; separately handling the industrial waste to enable the reuse of sewage sludge; and separately handling of stormwater to restore natural drainage patterns.

There are a few techniques that can be adopted when realising the mentioned strategies. These techniques comprise of domestic water conservation; recycling of grey water and rainwater in small scales; on-site stormwater infiltration or storage; utilisation of natural drainage pattern; and local sanitation technology (Butler and Parkinson, 1997).

Moreover, there are essential measures that need to be considered when master planning sustainable urban drainage solutions. These measures are dry weather performance for sewers (i.e. complete retention of sewage to the point of treatment and disposal, exclusion of erroneous inflows and infiltration, accommodation of growth and development, no significant accumulation of silt and debris); dry weather performance for watercourses (i.e. sufficient base flow to support life and maintain acceptable habitat, maintain acceptable amenity value, ability to discourage debris, litter and fly-tipping); wet weather performance (i.e. flood management, acceptable water quality, accommodation of climate change, growth and system deterioration) (Fleming and Slack, 2001).

Utilisation of stormwater as a resource is a major component of stormwater management. Meanwhile, the effects of pollution should also being considered.

Stormwater in open systems creates the basis for recreation and development of ecosystems with a diverse fauna and flora life. Ideally, SUDS should adopt the characteristics of water's behaviour in nature and adapt it to urban conditions and requirements (Åstebøl *et al.*, 2004).

There are a number of qualitative and quantitative conditions to be considered in an urban area. Considering that stormwater is vastly originated from the city and road surfaces therefore there are limitations in its potential usage. When designing an open water system it is essential to consider the requirements of a fauna and flora community. Otherwise measures must be implemented to reach defined quality objectives. Even though, water quantities are initially defined, flow variations can also be modified with relevant measures (Hvitved-Jacobsen and Yousef, 1991).

In recent years many manuals and textbooks have been published regarding the design and master planning of SUDS. The fact is, while the design of individual measures is well established in literature; it is difficult to select the best combination of measures which would result in meeting the project objectives (Barraud *et al.*, 1999).

1.6.1. SUDS Techniques

1.6.1.1. Filter Drains and Permeable Surfaces

Bio-filtration is proven to be an effective process and filtration systems have repeatedly performed well when treating urban runoff (DeBusk and Langston, 1997; Lloyd *et al.*, 2001). It is suggested that incorporating bio-filters into stormwater treatment systems' design leads to improved purification processes (Aldheimer and Bennerstedt, 2003).

Systems that comprise physical filtration have been categorised as methods of improving runoff water quality due to the association between runoff pollutants and particles (DeBusk and Langston, 1997). Filters have frequently confirmed high suspended solid removal efficiencies, in the region of 80 – 90% (Ellis and Crabtree, 1999; DeBusk and Langston, 1997; Hsieh and Davies, 2005, Lloyd *et al.*, 2001), which is considerably greater than removal efficiencies of conventional devices i.e. 10 – 30% for gully pots (Ellis and Crabtree, 1999). Even though, effective in the removal of most stormwater pollutants, traditional sand and gravel filters struggle to remove some soluble pollutants including phosphorus and metals (Brix *et al.*, 2001; Hsieh and Davies, 2005; Ray *et al.*, 2005).

Whilst some literature focus on the systems' failure other examine the potentials to enhance the existing systems. Organic materials combined with sand and gravel have been used in many cases aiming to tackle this problem, and have been found to be successful (Brix and Arias, 2005; DeBusk and Langston, 1997; Hsieh and Davies, 2005; Ray *et al.*, 2005) due to their ability to sorb soluble pollutants.

Although very popular but there are some problems associated with filter strips. Some literature challenge the frequently reported high reliability of filter strips claiming that these systems are prone to "high and early failure rates" (Ellis and Crabtree, 1999). Aldheimer and Bennerstedt (2003) state that filters are vulnerable to clogging, and therefore may cause street flooding.

Both filter drains and permeable pavements are devices containing a certain volume of permeable (or porous) material situated below ground and used as stormwater storage.

Permeable surfaces allow precipitation to infiltrate into the surface layers, hence, could be described as source control measures (CIRIA, 2000). Permeable surfaces are also classified as preventive measures, because they provide flow attenuation and stormwater treatment through filtration and biological degradation of pollutants (SUDSWP, 2000).

Various designs correspond permeable surfaces; for instance gravel surfaces (e.g. for areas with lighter traffic load), solid paving blocks containing holes or blocks with gaps in between (e.g. for rural areas) and continuous surfaces with an inherent system of voids (CIRIA, 2000).

Filter drains are linear devices, which receive precipitation via over-the-edge flow similar to swales. The porous filter material (e.g. gravel) provides capacity for storage. Similar to permeable surfaces, they offer attenuation and water quality improvement by filtration and some biological degradation (CIRIA, 2000; SUDSWP, 2000). Infiltration through the soil layer is often resulted in efficient biological degradation of pollutants (UBA, 2002).

1.6.1.2. Ponds, Basins and Wetlands

Ponds contain water in dry weather and are designed to hold more when it rains. They include balancing and attenuation ponds, flood storage reservoirs, lagoons, detention ponds and wetlands.

The structures can be combined to form a permanently wet area for wildlife which can be used to treat the runoff and prevent flooding. Ponds tend to be found towards the end of the surface water management train, so are used if source control cannot be fully implemented, if extended treatment of the runoff is required or if they are required for wildlife or landscape reasons (CIRIA, 2000).

The structure can be described to manage both quality and the quantity of the Water. On the one hand, ponds are able to control flow rates by storing flood water and releasing it slowly when the risk of flooding has pasted (a balancing pond). Ponds have to be designed to cope with both dry and wet weather as the stored water may change the water level. The amount of filtered water to the soil and ground where their conditions are appropriate can influence the quantity of the existing water in the pond (Barbosa and Hvitved-Jacobsen, 1999).

On the other hand, runoff can be treated within these structures by settlement of solids in still water. Vegetation cover in the water creates calm conditions and promotes settlement. Aquatic vegetation and sediment can absorb the particles in runoff and also biological activity can also influence the quality of water. In addition, these structures are excellent opportunities for the landscape designers. Ponds may store water for reuse and offer opportunities for the provision of wildlife habitats and improvement of the landscape. These schemes can become a part of public open space (Scholz, 2006).

Ponds are permanently wet, but the water level within them varies. Ponds are suitable for flows attenuating and pollution treatment. There are various types of stormwater ponds. Balancing ponds or flood storage reservoirs. These only store runoff until the flood peak has pasted resulting in a small treatment capacity. Lagoons that provide still conditions for settlement of solids, but offer no biological treatment. Retention ponds that have detention periods up to three weeks and their level of treatment is higher than extended detention basins. Wetlands with permanent water that flow slowly through the aquatic vegetation. Wetlands also have detention periods of up to two weeks, and are more efficient at treating pollutants is higher than retention ponds (CIRIA, 2000).

A constant base in flow is essential to prevent dryness within the ponds and wetland.

Ponds are relatively simple constructions. To avoid infiltration reduction and plant growth disturbance the ground should not be unusually compacted during construction. Before starting any operation the vegetation cover has to be allowed to be completely established. To shorten this process it is recommended to use pot-grown plants of local species and suitably prepared soil. There should be always an access to the basin or the pond for inspection and regular cutting of grass, the annual clearance of aquatic vegetation and where required the removal of silt (Scholz, 2006).

Both basins and ponds attenuate peak flows. In basins stormwater is mainly being treated by sedimentation of pollutants during the brief storage time. Whereas, in ponds and wetlands water is being retained for several weeks, therefore, all natural self-purification processes (e.g. sedimentation, filtration and microbial degradation) are worked together to improve water quality (SUDSWP, 2000).

Constructed wetlands are engineered man-made structures that are designed, built and operated to imitate the role of natural wetlands. They are created from a non-wetland ecosystem or a former terrestrial environment, largely for the purpose of pollutant removal from storm water. The constructed wetland treatment system is a less costly alternative for conventional storm water treatment using local resources and is an energy-efficient technology (Hua Sim *et al.*, 2008).

These devices employ wetland plants and micro-organisms, which are the active agents in the treatment processes. Most of the constructed wetland systems are shallow-water marshes dominated by emergent marsh plants such as cattails, bulrushes, rushes and reeds. Constructed wetlands offer numerous multiple-use values such as habitat creation, water quality improvement, flood prevention and control, and food and fibres production (also termed as constructed aquaculture wetlands). These systems can potentially tolerate variable volumes of water and altering contaminant levels (Hua Sim *et al.*, 2008).

The most important advantages of basins, ponds and wetlands are the potential of creating wildlife habitats within urban areas, improving the landscape, providing a focus for local people (e.g. recreation areas) and to bring back water into the urban environment (Scholz, 2006).

1.6.1.3. Infiltration Devices

Infiltration devices provide an area, for surface water runoff to infiltrate into the ground and therefore described as sustainable devices which take advantage of "natural attenuation processes" (Ellis, 2007). They also improve runoff quality by filtering out pollutants (Siriwardene *et al.,* 2007). These devices may be used as source control or serve a larger catchment. In some literature these devices are described as the 'most promising solution' to the issue of urban runoff (Ristenpart, 2003). Whereas, others claim that these techniques are prone to over 50% five year failure rates (Ellis and Crabtree, 1999; Ellis, 2000). In effect, in several occasions clogging has happened causing the failure of infiltration systems, some even when pre-treatment was performed (Taylor *et al.,* 2001; Bouwer, 2002).

However, infiltration devices are generally considered as valuable systems (Calabro and Viviani, 2006; Liang *et al.*, 2004; North Shore City Council, 2001) and are frequently featured in stormwater management practices (Siriwardene *et al.*, 2007).

In general, infiltration devices are only suitable in areas with unsusceptible groundwater (Ellis and Crabtree, 1999; Ellis, 2000; Marsalek and Chocat, 2002). To avoid early failure it is recommended to appropriately size and maintain these devices in order to associate with mean annual runoff volumes and mean suspended solid concentrations (Ellis and Crabtree, 1999).

These systems require suitable groundwater level and soil properties in relation to the quality and volume of the water being infiltrated. Soakaways, infiltration trenches and infiltration basins are some examples of infiltration devices; swales, filter drains and ponds can too serve as infiltration devices, depending on the soil permeability of the site (CIRIA, 2000).

The most common Infiltration devices are the infiltration trenches and soakaways. An infiltration trench is a linear excavation lined with a geotextile, backfilled with stone and could also be covered with turf. A soakaway is one of the below ground structures which can be stone filled formed with plastic mesh boxes, dry wall lined, or built with concrete ring units (Butler and Davies, 2004).After being diverted to these systems runoff can either infiltrate in to the soil or evaporate. By providing a larger contact surface area, the system creates storage and encourages infiltration. It is necessary that both of the systems are implemented at least in five meters distance with the foundations of buildings or on the roads. Any area that has pervious sub-soils such as sand, chalk, gravel and fissured rock is considered suitable for soakaway and trenches. When installed in land with gradients larger than 4% regular interval checking is required. The mentioned systems are suitable in areas with low water table that allows a free flow of the storm water in to the subsoil all over the year. In order to minimise the risk of polluting ground water the vertical distance between the base of these systems and the ground water level has to be more than one metre. These systems' ability to reduce the concentrations of pollutants from stormwater through physical filtration, absorption, and by chemical activities makes them a popular design choice (Butler and Davies, 2004).

For infiltration trenches draining motorway runoff the average annual removal efficiencies of 60-85% have been recorded for suspended solids, metals, PAHS, oil and COD (Colwill *et al.*, 1984). Other similar devices are filter drains. These systems are linear devices comprising of a porous or perforated pipe in a trench of filter material. Traditionally, they have been used beside roads to intercept and convey runoff but they can also be used as simple conveyance devices. Like permeable pavements they may or may not allow infiltration to the ground.

In all these devices water quality is improved through filtration, adsorption of particles, sedimentation and biological degradation of pollutants. Infiltration devices are easily integrated into the landscape, e.g. as playing fields or recreational areas (SUDSWP, 2000).

1.6.1.3.1. Darcy's Law

Infiltration into a homogeneous porous material obeys "Darcy's Law", a law formulated by Henri Darcy in 1856 (Ferguson, 1994). Equation (1) below defines this law:

$$Q=KA\ (\Delta h/L) \tag{1}$$

Where Q is the flow (cm^3/h), K is the saturated hydraulic conductivity (cm/h), A is the cross sectional area through the porous medium perpendicular to the flow (cm^2) and $\Delta h/L$ is the hydraulic radiant, the difference in hydraulic head per unit distance in the direction of flow (L)

Darcy's Equation (EQ 2) can be re-written by substituting the equivalency:

$$Q=qA \tag{2}$$

To derive:

$$q=k\ (\Delta/L) \tag{3}$$

Where q is the velocity of water through a unit cross section of the porous medium, called the Darcian Velocity (cm/h).

The velocity of fluid water through the pores of the medium is given in Equation (4):

$$V = q/Qs \tag{4}$$

Where, V is the fluid velocity (cm/h), Qs is the water content of the medium (equal to the mediums porosity minus the volume of trapped air in the medium's pores) (Ferguson, 1994).

Ferguson (1994) further explains this equation stating that the hydraulic gradient $\Delta h/L$ is the driving force that makes water move where there is a difference in the total head between two points in a soil mass. Water moves from the higher to the lower head, always moving in the direction of sleeper gradient Hydraulic conductivity k is high in materials with large continuous pores. In homogenous granular soils the highest conductivities are soils composed of a single large grain size; any gradation in grain size would fill the pores with smaller particles. In addition the soil is capable of developing structure- the aggregation of grains into larger particles or units. A soil with a high fraction of clay can be highly conductive if it has an aggregate structure or it can be nearly impermeable if the particles are kept dispersed and structure less.

The infiltration rate is described in 'stormwater Management' (Ferguson, 1994) as the flux of water into the soil in units of cm\h when the rate of delivery of water to the surface is smaller than the soils ability to take it in, water infiltrates as fast as it arrives. The infiltration rate rises when water is ponded over the surface (Ferguson, 1994), theorizes that the potential infiltration rate is controlled at the soil surface. However other workings have allowed for the possibility that hydraulic gradient and thus infiltration rate might be affected by conditions deep within the profile

(Ferguson, 1994).

1.6.1.4. Detention Devices

In 1986 the U.S. Environment Protection Agency described detention and retention facilities as “the most effective and reliable of the techniques”. Although great advances have been made since 1986 in the field of stormwater management but this statement is still valid in many cases. In recent years, detention facilities have received great attention with many researchers favouring them.

Detention tanks are considered to reduce flood damage and lower the size of downstream conveyance systems, therefore lowering costs (Stahre and Urbonas, 1990).

Parallel to the hydrological values of source control, detention tanks have also been found capable of reducing pollutant discharges (Jacopin *et al.*, 1999). Detention tanks like gravel filters have proved to be especially effective in reducing solid concentrations in stormwater (Calabro and Viviani, 2006). Detention tanks operate by intercepting the flow, detaining it and allowing sedimentation to occur, and are considered to be a "single continuum of treatment" (Wong *et al.,* 2001). They can be used to effectively reduce the impact of urbanisation on hydrological processes (Nascimento *et al.,* 1999).

There are various design guidelines available for detention tanks. Design guideline published by North Shore City Council (2002) state that the time of concentration of the downstream catchment should be considered when designing detention tanks. This guideline also implies that detention tanks should be designed to be capable of coping with a large range of storm sizes. Calabro and Viviani (2006) proposed that tank volumes of 30-50 cubic metres per hectare provide the best combination of efficiency and cost and that an increase in tank volume will not enhance efficiency proportionally. This report also suggested that the removal efficiencies of the offline tanks are likely to be greater than those of the online tanks.

However, the space requirement associated with the detention tanks is considered as a disadvantage (Aldheimer and Bennerstedt, 2003 and Stahre and Urbonas, 1990). This suggests that the land resources required may lead to unfeasible costs. The fact that the majority of runoff requiring treatment is developed on city centre roads weakens the applicability of the detention tanks for stormwater treatment where treatment measures are limited (Ray *et al.,* 2005).Therefore, below ground detention tanks is proved to be highly useful in this respect. Research shows that tanks consisting of modules are considered to have a faster infiltration and emptying rates when compared with tanks containing porous material (Liang *et al.*, 2004). Hence, application of tanks consisting of modules result in reduced excavation and land requirements and consequently reduced costs.

1.6.1.5. Other Common Techniques

1.6.1.5.1. Permeable Pavements

Permeable pavements are considered as a practical solution to the problem raised by increased stormwater runoff and decreased stream water quality associated with vehicles. Permeable pavement systems are usually consisting of a matrix of concrete blocks or a plastic web-type structure with voids filled with sand, gravel, or soil (Fig 2). The purpose of the voids is to allow stormwater to infiltrate through the pavement into the underlying soil. This can significantly influence the impacts of stormwater runoff caused by urban development (Brattebo and Booth, 2003).

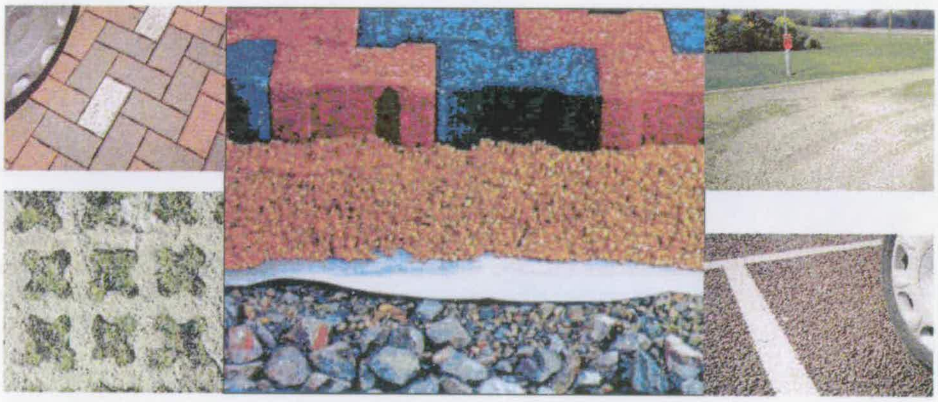


Fig 2. Examples of different variety in permeable paving – clockwise from bottom left: continuous-laid concrete with large voids, small impermeable blocks with voids in between, small impermeable blocks with voids in between cross-section, open-textured soil, continuous-laid porous asphalt (Martin *et al.*, 2000).

This technique is commonly used on parking lots and residential roads. One option is the permeable macadam which is very expensive and has a tendency to clog after 1-3 years and needs considerable maintenance effort. Also in extensive use are lattices of blocks with the infiltration surface beneath the load-bearing surface (Burkhard *et al.*, 2000).

1.6.1.5.2. Swales

Swales are grassed channels taking up runoff from roads or parking lots. The runoff gradually flows through the grass swale and infiltrates into the ground (Fig 3). There should be no stagnant water in a carefully designed swale (Burkhard *et al.*, 2000).

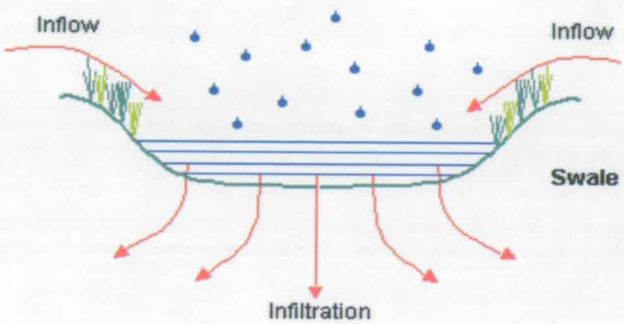


Fig 3. A cross section view of a normal swale (Martin *et al.*, 2000).

1.6.1.5.3. Constructed Wetlands

Constructed wetlands are one of the most cost-effective techniques in stormwater treatment plans. These systems are designed to degrade organic substances and nutrients from stormwater runoff (Rodgers and Dunn, 1992; Lakatos *et al.*, 1997), and can be utilised to remove metals from mining effluent and special industrial wastewaters (Crites *et al.*, 1997). Vegetation plays an important role in constructed wetland for the removal of pollutants (Brix, 1994). Plants take up nutrients, and adsorb/accumulate metals. *Phragmites australis*, and some *Cyperus* species are the most commonly used plant species in constructed wetlands (Crites *et al.*, 1997; Greenway and Woolley, 1999; Ayaz and Akca, 2001; Okurut *et al.*, 1999).

1.6.1.5.4. Bioretention

Bioretention areas function as soil and plant based filtration devices that eliminate pollutants from stormwater runoff through a number of physical, biological, and chemical treatment processes. Bio-retention is considered as one of the most sustainable approaches towards stormwater runoff treatment. This process is an important part of low impact development (LID), as the practice has the potential to reduce runoff volumes, minimize peak flows, recharge ground water, increase evapotranspiration, and reduce the mass of pollutants entering surface and ground waters. When compared with other "ultra-urban" SUDS, bio-retention proved to be considerably cost effective (Hunt *et al.*, 2008).

1.7. Pollutant Removal Mechanisms

There are a number of mechanisms which improve water quality within stormwater detention/infiltration systems (D'Arcy *et al.*, 2000), these mechanisms can be classified as, settling of suspended particulate matters; chemical transformation; filtration and chemical precipitation through contact of water with the substrate and litter; adsorption and ion exchange on the surface of plants, substrate, litter and sediments; pollutants and nutrients breakdown, transformation and uptake by microorganisms and plants; predation and natural die off of the pathogens (Lee and Scholz, 2006).

1.7.1. Suspended Solids Removal

Suspended solids are resulted from the degradation of macrophytes and the overflow contamination. In the case of infiltration systems SS predominantly settle at the surface of the system. Theses contaminants can also interact with the substrate and attach to the granules, causing a process called granular medium filtration (Tchobanoglous *et al.*, 2003).

1.7.2. Biochemical Processes

1.7.2.1. Nitrate and Nitrite in Water

Biochemical mechanisms contribute to the degradation of organic and inorganic matter in the water. The major components in need of removal from stormwater are nitrogen and phosphorous.

The conversion of ammonium ions to nitrate is called nitrification and is vital for the majority of plants including aquatic plants as they are able to take up nitrate but not ammonia or ammonium. The nitrification process is described in Equations (5) and (6) (O'Neill, 1998). Unfortunately nitrate is very soluble in water and easily leaked from soils. Thus, it is important to control nitrate concentrations in source controls (O'Neill, 1998).

$$4NH_4^+ + 6O_2 \rightarrow 4NO_2^- + 8H^+ + 4H_2O \quad (5)$$

$$4NO_2^- + 2O_2 \rightarrow 4NO_3^- \quad (6)$$

As nitrate is dissolved easily in water and its shown stability over a large range of environmental conditions, its main action in water is that it feeds plankton, aquatic plants, and algae and ultimately helps maintain the aquatic food chain. The bacteria quickly convert nitrite to nitrate. Humans and wildlife's health and safety are at risk when excessive concentrations of nitrate and nitrite are present in water.

Nitrate is the more harmful of the two, especially for humans because it is broken down in the intestines to become nitrite. A condition known as methemogtlobinemia syndrome is caused when methemoglobin is produced when nitrite comes in contact with haemoglobin in human blood, which negatively affects the ability of red blood cells in carrying oxygen. Infants are particularly in risk as they don't have the required enzyme to correct this condition. Fish are also at risk as the high concentrations of nitrate and nitrite can produce "brown blood disease". The blood in the fish turns a chocolate brown as nitrite enters through their gills. As this happens it negatively affects the blood in carrying sufficient amounts of oxygen, thus resulting in the suffocation of the fish even if the is an adequate supply of oxygen in the water. Eutrophication which has already been explained is also common when excessive amounts of nitrates are added to the water, when algae and aquatic plants are produced in huge quantities (O'Neill, 1998).

Therefore, stormwater treatment systems should encourage denitrification. The regeneration of dinitrogen from nitrate is raised under both aerobic and anaerobic conditions in the water. Under anaerobic conditions, organisms could make use of nitrate to replace dioxygen as an electron acceptor and as their source of respiratory energy (O'Neill, 1998). The process of denitrification is illustrated in Equation (7).

$$5CH_2O + 4NO_3^+ + 4H^+ \rightarrow 2N_2 + 5CO_2 + 7H_2O \tag{7}$$

1.7.2.2. Ammonia

Ammonia is an organic form of nitrogen which is the least stable form of nitrogen in water. It is converted to nitrate in waters containing oxygen without great difficulty and is also transformed to nitrogen gas in waters that that are low in oxygen. The two forms of ammonia in water include the ammonia ion (NH_4^+) and the dissolved, unionised ammonia gas (NH_3).

Total ammonia is the term given for the sum of ammonium and unionized ammonia. Temperature and pH of the water are the factors which the dominant form depends on. Equation (8) which can be seen below can be written showing the reaction between the two forms:

$$NH_3 + H_2O \leftrightarrow NH_4^+ + OH^- \tag{8}$$

When the pH changes the form of ammonia will also easily change. H^+ concentration decreases and OH^- concentrations increase when the pH increases, thus changing the above equation as the amount of aqueous NH_3 is increased.

At a pH value of less than 8.75, NH_4^+ will predominate.

At a pH value of 9.24, roughly half of aqueous NH_3 is transformed to NH_4^+.

At a pH value of above 9.24, NH_3 will predominate.

Research has shown that unionised ammonia (NH_3) is more toxic to aquatic organisms than the ammonium ion (NH_4^+). Human health is deteriorated by the toxic concentrations of ammonia, thus resulting in the loss of equilibrium, convulsions, coma, and death. The health of fish is also negatively affected as the ammonia concentrations change their structural development (O'Neill, 1998).

1.7.2.3. Nitrogen Removal

Nitrogen removal process in most SUDS involve, plant uptake, volatilisation, adsorption and nitrification/denitrification, the latest proved to play a major rule in water treatment performances.

Both aerobic and anaerobic environments are required for nitrification/denitrification processes to occur. The presence of the nitrifying bacteria is necessary for nitrification to happen these bacteria are sensitive organisms and react to a wide range of parameters including pH, dissolved oxygen and temperature (D'Arcy *et al.*, 2000).

The enzyme required for denitrification processes however, may be suppressed when dissolved oxygen is present. Therefore, nitrification/denitrification processes simultaneously only happen in soils containing both aerobic and anaerobic zones (Cooper *et al.*, 1996).

Bio-retention is an effective plant- and soil-based low impact treatment/infiltration facility that provides treatment to stormwater runoff. There are two concerns when considering bio-retention for nitrogen removal of stormwater runoff. The first is the uptake of nitrogen compounds during the time scale of storm events. Research shows that due to sorptive interactions with the soil media, ammonia is moderately removed from infiltrating stormwater (Davis *et al.*, 1998b). On the other hand, due to its anionic form, nitrate is nearly neglected by soil components, and consequently almost no nitrate is removed. The second nitrogen issue in bio-retention is on the long-term time scale. Organic and ammonia nitrogen are accumulative compounds and will accumulate in the bio-retention system. Therefore, considerations for their removal from the bio-retention device must be taken and optimized (Kim *et al.*, 2003).

1.7.2.4. Phosphorous Removal

Phosphorous is contributed to stormwater from sources like fertilizers on agricultural or residential cultivated land, natural organic material (e.g. leaf litter, grass clippings, unfertilized soils), laundering and commercial cleaning processes, treatment of boiler waters, biological processes instigated by sewage, food residues, material waste and rainfall, (APHA 1998; USEPA 1998). Sand filtration is proven to be a new mechanism to cost effectively remove phosphorus from stormwater runoff (Erickson *et al.*, 2007).

In most wetland systems phosphorous is immobilised through chemical precipitation with metals, substrate adsorption of P, plant and algal uptake, incorporation into organic matter and bacterial activities. There is a strong interaction between phosphorous and the wetland's soil and biota. This results in sustainable long term storage of P (Drizo *et al.*, 1997).

Chemical treatment methods for phosphorus removal involves precipitation by calcium, aluminium, or iron and surface adsorption to iron oxide or aluminium oxide, these are all a function of pH. Phosphorus precipitation is controlled by iron and aluminium when pH<6 and calcium when pH>6 (Stumm and Morgan, 1981). Phosphate adsorption is maximum in acidic conditions but ~50% of available phosphorus can be adsorbed to iron at pH 10 (Stumm and Morgan, 1981). American national secondary drinking water standards (USEPA 1988) suggest that pH values remain between 6.5 and 8.5.

Reddy and D'Angelo (1994) have summarised the phosphate retention mechanism by mineral soils. First, in acid soils, phosphorus is fixed as aluminium and ferric phosphates, if the activities of these cations are high. Second, in alkaline soils, phosphorus fixation is directed by the activities of calcium and magnesium. And third, phosphorus availability is highest in soils with somewhat acidic to neutral pH. Furthermore, besides biological uptake and depending on soil type, there are two other potential sinks for phosphorus. Firstly, in mineral soils dominated by iron oxides, phosphorus can be readily immobilized through sorption and precipitation by ferric oxyhydroxide, and formation of ferric phosphate in the oxidized zones at the soil-water interface; secondly, in calcareous systems, phosphorus released into the overlying water column can be precipitated as calcium mineral bound-phosphorus (Erickson *et al.*, 2007).

It is stated in the literature that time plays a major role in establishing the trends in the quantity of the P removed from wetland systems. In shorter periods the quantity of P removal by the three processes is substratum > macrophyte > biofilm, whereas, in longer periods it is macrophyte >substratum > biofilm. In addition, it is suggested that plant harvesting can increase the P removal rate by 10-20 % (Lantzke *et al.*, 1999).

1.7.3. Microbial Contamination

Urban stormwater holds significant amount of debris and pollutants, including litter, organic matter, sediments, nutrients, oils, heavy metals and micro-organisms (Davies and Bavor, 2000). Therefore, it has been documented as a major source of diffuse pollution to the aquatic environment (Yu and Nawang, 1993).

Untreated stormwater runoff will also contain human and animal faeces (Polprasert, C. 2007). Findings by Feachem *et al.*, 1983, (cited in Polprasert, 2007) indicate the quantity of faeces production in some European and North American cities to be between 100 and 200g wet weight per capita daily. Most adults produce between 1 and 1.3kg of urine, dependent on the amount of liquids they drink, and on the local climate (Polprasert, 2007). The solid matter in faeces is mostly organic but its carbon/nitrogen ratio is only between 6 and 10 which is lower than the optimum C/N ratio of 20-30 necessary for effective biological treatment (Polprasert, 2007). The amount and composition of animal wastes (faeces and urine) excreted per unit of time differ broadly also and rely on factors such as the total live weight of the animal, the animal species, size, age , food and water intake, climate and management practices etc (Polprasert, 2007). The faeces of domesticated animals (i.e. dogs) can be a major source of pollutants in stormwater, and also a potential source of Cryptosporidium and Giardia (Polprasert, 2007).

The occurrence of faecal microbial contamination in stormwater can be associated with sewer leakage and overflow, septic tank seepage and domestic animal faeces. Recent epidemiological studies indicated that there is a rising risk of adverse health associated with swimming in recreational waters contaminated with untreated or poorly treated urban stormwater (Haile *et al.*, 1999).

In the USA numerous studies have investigated the sources of microorganisms, within urban catchments (e.g. Bannerman *et al.*, 1993; Steuer *et al.*, 1997). Therefore, faecal contamination was found to be higher in runoff from residential areas than from commercial and industrial areas (Bannerman *et al.*, 1993). Suggesting that this faecal contamination was originated from wildlife and pets, as these animals were much more extensively inhabited in residential developments (McCarthy *et al.*, 2006).

A study at the American Watershed Management (1999) claims that microorganisms present in urban catchments are originated from several human and non-human sources. Human sources can be classified as sewer overflow structures, illegal sewer connections to stormwater systems and failing septic systems. Non-human sources as mentioned before are domestic animals, wildlife and livestock (McCarthy *et al.*, 2006). These sources were also identified by Schiff and Kinney (2001), but they suggested that leakage of the sewer systems and re-suspension of contaminated sediments can both be probable sources of microbiological contamination in stormwater systems. Furthermore, the sources of faecal contamination in an urban catchment in Florida were found to be wild animals, humans (i.e. human sewerage input) and dogs (Whitlock *et al.,* 2002).

The human originated sources of stormwater microbial contamination can be at the level of individual households, possibly associated with DIY ('do it yourself') enthusiasts and can happen in existing residential developments, or can be as a result of mistakes made during the construction of new developments (i.e. wrongly connected foul sewers from whole streets or blocks). The impacts of such mistakes is most clear during low flow conditions, as the foul flows, although intermittent, are largely independent of weather conditions (O'Keefe *et al.*, 2003).

In the UK, another cause of foul drainage reaching surface water sewers and watercourses is the outcome of a cost-saving practice adopted for post-war separately sewered housing, Meaning that two separated sewers are provided with common manholes; called dual manholes. Therefore, overflows of foul are facilitated into surface water at the manhole where the only thing separating the two drainage channels is a low weir. This weir could easily be overtopped each time the foul sewer gets blocked (O'Keefe *et al.*, 2003). In a dual manhole, the foul blockages are hard to notice in the first place, because water could still get into a watercourse through the surface water sewer. Therefore, this can result in gross pollution that will continue until noticed by the pollution control authorities or becomes the subject of public complaint about the pollution.

Studies have been carried out concerning the non-human originated sources of microbial contamination in stormwater. For example, in Melbourne the dog fouling load has been estimated to be the pollution equivalent of the untreated sewage from 90,000 people, a study in the Pipers Creek in Seattle also suggested that cats were most important. In the UK, the dogs' population is believed to be between 6.5 million and 7.4 million, producing nearly 1000 tonnes of faeces each day (O'Keefe *et al.*, 2003).

Each dog's daily faecal output is estimated between 100–200g (Keep Britain Tidy Web page, 19.03.2003). These statistics oppose the common myth that suggests such pollution sources are natural (i.e. considering the high density of pets such as cats and dogs in urban areas compared with equivalent wild species in natural habitats).
The effects of non-human sources of diffused urban pollution on water quality are most severe after a storm event when the pollutants are mobilised by rainfall. Some of the contaminating materials will be deposited on impermeable surfaces, some drained to surface water sewers and consequently into the watercourses, and some will be drained into combined sewers (i.e. in older parts of towns).

Wetlands and ponds provide a combination of physical, chemical and biological processes that contribute to the removal or transformation of pollutants including microorganisms in stormwater.

The removal of faecal indicator bacteria from wastewater by some SUDS is well recognised (Bavor *et al.*, 1987; Gersberg *et al.*, 1987; Perkins and Hunter 1999). Reported removal efficiencies for coliforms in constructed wetlands frequently exceed 90% (Kadlec and Knight 1996) the removal efficiencies were significantly higher in largely vegetated systems compared with non-vegetated systems (Gersberg *et al.*, 1987). The faecal streptococci removal efficiencies by wetlands generally exceed 80% (Kadlec and Knight 1996). This is believed to be associated with filtration, solar irradiation, sedimentation, aggregation, oxidation, antibiosis, predation and competition (Gersberg *et al.*, 1987).

1.7.4. Heavy Metal Contamination

Urban stormwater runoff, especially road runoff, contains massive amount of heavy metals that, unlike organic pollutants, cannot be degraded in the environment. The most important sources of heavy metals in stormwater runoff are building materials (e.g. Cu from roofs and Zn from galvanized steel), and traffic-related sources such as brake linings (Cu, Ni, Cr, Zn, Pb), tire wear (Zn), and autocatalysts (Pt, Pd, Rh). Due to short term (e.g. acute toxicity) and long-term (e.g. carcinogenity and reproducing damages) unpleasant effects of heavy metals in the aquatic environment, treatment of stormwater runoff containing heavy metals has become more and more important (Genc-Fuhrman *et al.*, 2007; Legret and Pagotto, 1999). Antifreeze salts used in the winter, are also a source for high zinc and cadmium levels as well as for sodium, calcium, and chlorine, resulting from a development of corrosion phenomena caused by the heavy metal mobilizing effects of de-icing solutions containing sodium chloride and calcium chloride (Bauske and Goetz, 1993). Another important source of pollutants is considered to be the road itself, as the components of asphalt, i.e. stone materials (whose content is approximately 95%) and bituminous binders (5%) release various contaminants., As well as a number of hydrocarbons, bitumen also contains trace metals including vanadium, iron, nickel, magnesium, and calcium. Nevertheless, metal content in stone material should not be neglected, considering its percent amount in asphalts (Lindgren, 1996).

Heavy metals, in particular Cu, Pb, Cd, and Zn, are the most widespread pollutants in urban and highway runoff and often are found to go beyond water quality standards (Cole *et al.,* 1984; Ellis *et al.,* 1987).

All the above mentioned pollutants can enter aquatic systems largely via runoff; hence contributing to water and soil contamination (Boxall and Maltby, 1995; Maltby, 1999; Perdikaki and Mason, 1999), the degree of such contamination is related to numerous factors, including traffic load, rainfall and size of receiving waters (Mangani *et al.*, 2004).

Heavy metals exist in stormwater runoff in soluble or particulate forms; they are most bioavailable when soluble either in ionic or weakly complexed form (Scholz, 2004). Metal bioavailability/removal is driven by chemical processes such as acid volatile sulphide formation and organic carbon binding/sorption in reduced sediments of constructed wetlands (Wood and Shelley, 1999).

Some of the other SUDS structures such as small detention ponds have been believed to be the most cost effective source control strategy for urban runoff (Finnemore and Lynard, 1982; Maestri and Lord, 1987). Findings from numerous studies suggest that detention ponds can be reasonably effective in reducing metal concentrations in urban runoff (Yousef *et al.,* 1990; Martin and Miller, 1987; Striegl, 1987; and Mesuere and Fish, 1989).

In detention ponds sedimentation of particulate metals is believed to be the major removal process, consequential to a long-term metal accumulation in the top 5 to 10 cm of pond sediments (Yousef *et al.,* 1990; Martin and Miller, 1987; Nightingale, 2007). Nevertheless, overall metal removal efficiencies differ significantly amongst the investigated detention pond systems. Numerous factors seem to be important in determining the overall performance. These factors include input metal speciation, storm duration and density and the pond's hydraulic retention time (Hvitved-Jacobsen *et al.,* 1987; Martin and Miller, 1987).

There are requirements to meet in order to achieve the desired water quality standards, the pond should be appropriately sized and storage detention time and contaminant removal characteristics should be carefully considered. The concentration heavy metals in ponds are mediated by sediments and aquatic plants as well as associated physicochemical conditions. Although, the hydrologic design of detention ponds is well established, but this design criteria (i.e. based on the peak flow control) may not offer the desired treatment for stormwater runoff (Middleton *et al.*, 2008).

1.8. Urban Runoff Characteristics

Pollution sources are commonly classified as point source and non-point source. Point source refers to the polluted effluent originated from a point or a particularly small area (e.g. domestic or industrial wastewater).The path and the quantity of point source pollutants are easy to measure or control. Non-point sources, on the other hand, occur during rainfall, and the pollutants are being discharged from a broad area therefore it cannot be considered as a point source (Choe *et al.*, 2002). Establishing an appropriate process to control non-point sources are considered to be complex procedures since the source and the path of effluent are uncertain, adding to the fact that the concentration is high during rainfall events (Choe *et al.*, 2002).

Runoff from urban areas shows a characteristic associated with pollutants that mirror human activities and the urban development of the catchment. This diffuse, non-point source pollutant load consists of litter, debris, and sediment as the more visually apparent components, and nutrients, coliforms, heavy metals, and toxic chemicals (e.g., polycyclic aromatic hydrocarbons, polychlorinated biphenyls, organochlorines) as a ''hidden'' components. As urban catchments are mainly impermeable, large surface flows are created during rainfall events. Structural drainage control devices then convey the urban runoff to a point of discharge into the receiving water body. Thus, the diffuse pollution generated and accumulated over a broad area is transformed into a point source of pollution prior its entry into the aquatic environment. Therefore, it is necessary to implement prevention measures to protect the waterways receiving urban runoff from different types of contamination (Davis and Birch, 2008).

The effluent of pollutants originating from a non-point source during storm events not only contains various types of pollutants but also holds a large pollutant load therefore; it exerts an enormous influence on receiving waters (Whipple and Hunter, 2007; Characklis and Wiesner, 1997).

The type of the land use in the catchment and the rainfall condition are playing a major role in determining the concentrations and the load of non-point source pollutants. Particularly, stormwater runoff generated in residential and industrial areas is well expected to contain hazardous materials (i.e. oil components, heavy metals and floating materials) (Choe *et al.*, 2002). Therefore, the characterisation of stormwater runoff pollutant is essential for a water quality management plan to urban stream (Lee and Bang, 2000).

1.9. Gully Pot Liquor

A gully pot is a small settling chamber or sump, provided along the kerb of roads to maintain sediments from road runoff before it enters the sewer system. Gully pots are widely used in urban drainage networks; it is reported that there are more than 17 million gully pots in England and Wales alone.

In wet seasons, pollutants deposited on road surfaces are washed off and drained into the pot, and then a number of physical processes occur. The received runoff and pollutants combined with the pot liquor, resulting in a considerable degree of turbulence, dilution and washout of dissolved and sediment-attached pollutants. More dense sediments (and attached pollutants) settle in the pot under the influence of gravity and form a layer of sludge on the invert, and those sediments already deposited could be re-eroded (Memon and Butler, 2002).

In dry seasons however, biochemical processes become active within the gully pot liquor and sludge. This is particularly dependant on the ambient temperature conditions. Oxygen-demanding pollutants, that were detained in the pot during previous storm events, begin to decompose resulting in oxygen deficiency within the pot liquor, consequential to pollutant transformation from one phase to another (e.g. metal form changes), COD reductions in gully pot liquor, ammonium transformation and the release of partially stabilised by-products from anaerobic digestion of sludge in pot invert (Lee, 2006). The biochemical processes are likely to increase pollutant (particularly dissolved pollutants) levels in the overlying liquor, which are eventually washed out to sewer during the following storm event (Memon and Butler, 2002).
Taking into account the scale of use and pollutant loads associated with gully pots, the manner in which they are managed could have considerable downstream implications. In particular, if the pots are connected to separate storm sewers, they will have a direct influence on the quality of runoff discharged to receiving waters. Most local authorities in the UK have gully cleaning programmes (Butler and Clark, 1995), but the objective of these is more concerned with uses such as the avoidance

of local ponding and flooding rather than water pollution control. The question remains, nevertheless, firstly whether existing cleaning programmes have a beneficial effect on runoff quality, and secondly whether cleaning practice or gully pot design could be enhanced in some way (Chen and Adams, 2007).

1.10. Modelling Storm Water Quality

Advances in urbanisation result in the degradation of receiving waters. In order to reduce the consequent damage upon aquatic ecosystems, the extent of the problem must be known. Unfortunately, sampling programs are not very cost effective to carry out. Moreover, planning level estimates are often required prior to the urbanisation of natural catchments. Therefore, the prediction of urban storm water quality at unmonitored catchments is required. The high variability associated with mean concentrations at single sites challenges the validity of applying simplistic representative estimates of site mean concentrations. In addition, mean concentrations variability is also observed between sites. This suggests the need for complex models capable of predicting mean concentrations variability at single sites and among multiple sites. The development of process based models is complicated. Essential calibration data may not be readily accessible, resulting in large inaccuracies when calibration parameters are estimated not using the actual data from the site of interest. Hybrid models are also restricted, often only crude approximations of reality. For instance, buildup-washoff models ignore potentially significant processes, including the rainout and washout of nitrogen compounds, pervious area erosion and the stream scour of sediments (May and Sivakumar, 2008).

Moreover, most buildup-washoff models imperfectly presume that all available accumulated pollutant is washed off during a given storm (Vaze and Chiew, 2002). This limitation is compounded when taking into consideration that pollutant accumulation data cannot be directly measured. Therefore, Huber (1992) stated that the use of literature values to predict buildup could cause model predictions to be more than an order of magnitude out. This has created the requirement for statistical models capable of predicting urban storm water quality. Two widespread, statistically based studies have been previously undertaken to predict urban storm water quality. First, the study by Driver and Tasker (1990) where the data from the Nationwide Urban Runoff Program (NURP) was used to construct multiple linear regression models, capable of predicting event mean concentrations (EMCs) at sites located throughout the United States. Second, was the study performed by Brezonik and Stadelmann (2002) also using multiple linear regression models to predict EMCs at watersheds in the Twin Cities metropolitan area, Minnesota, USA.

The use of logarithmically transformed data in each of these two studies allowed the simplistic representation of nonlinear relationships. Nonetheless, such relationships were limited to potentially over simplified power relationships. These relationships were believed to be rather crude approximations of the complex diversity of nonlinear relationships present in the environmental systems under study. Unfortunately, the enormous collection of complex, interrelated processes influencing urban storm water quality are difficult to define prior to model development. This has caused the demand for more complex models such as artificial neural networks (May and Sivakumar, 2008).

Artificial neural networks (ANN) are information processing structures inspired by the functioning of the human brain. They are consisting of a vast, interconnected structure of processing elements. The computational power of these processing elements is minimal when in isolation. Nevertheless, within large networks, the computational power is extremely large. The parallel distribution of information within the ANNs provides the capacity to model complicated, nonlinear, interrelated processes. This eventually allows ANNs to model environmental systems without prior specification of the algebraic relationships between variables (May and Sivakumar, 2008). This has led to the application of ANNs in many water resources applications (Holmberg *et al.*, 2006; Mazvimavi *et al.*, 2005; Riad *et al.*, 2004; Sarangi and Bhattacharya, 2005; Tayfur *et al.*, 2005). Regardless of its strong theoretical potential, ANN application is subject to a number of challenges. In particular, it is widely recognised that the generalisation of an ANN is dependent upon network topology and the selection of key network parameters, including the transfer function, the error function, learning rate, and momentum (Goethals *et al.*, 2007). A time consuming trial and error approach is frequently implemented to optimise ANN models.

Chapter 2

Site Description

2.1. Introduction

This chapter focuses on four individual studies. First is a SUDS feasibility study of the Belvidere Hospital and the Celtic FC Stadium area in Glasgow, this study was a part of the "The Glasgow Sustainable Urban Drainage System Management Project". Second, the "Edinburgh Sustainable Urban Drainage System Management Project" will be described in the chapter. The other three studies describe the experimental setup and operation methods applied for differently designed experimental stormwater treatment systems treating road runoff. The systems include an experimental stormwater detention (extended storage) systems based on the Alderburgh Ltd. detention cells receiving concentrated runoff that has been primarily treated by filtration with different inert aggregates, a combined SUDS system consisting of an below ground stormwater detention tank and a planted and an unplanted stormwater infiltration ponds and combined sustainable urban drainage system consisting of a gravel filter and infiltration detention tank in series. Systems' design and compositions as well as environmental conditions of the experimental systems are also portrayed.

Moreover, the application of an Artificial Neural Networks system in prediction of microorganisms in an experimental stormwater detention system is investigated, the chapter then investigates the treatment mechanisms and potentials for water quality variables in the studied sites. A simple removal model is applied to evaluate the removal potentials of the system. The overall performance of each system concerning nutrients and microorganisms as well as other water quality variables is also statistically compared in order to evaluate the efficiency of the design components as well as the operation conditions. Eventually, the chapter investigates the application of the U.S. Environment Protection Agency program SWMM model in order to understand the hydrological characteristics of a combined sustainable urban drainage system consisting of a gravel filter and infiltration tank in series.

2.2. The Glasgow Sustainable Urban Drainage System Management Project

2.2.1. Background to Case Studies

The Belvidere Hospital site is located to the south of London Road (major road into Glasgow), and is owned by Kier Homes. It is in a prime development area due to its proximity to the Glasgow City centre and amenities such as parks, shopping centres, Celtic Park, and the proposed national indoor sports arena. The southern border of the site is adjacent to the River Clyde.

The Celtic FC Stadium area is located to the north of London Road (see above), 2 km from the City Centre, in Glasgow's East End (also known as the Celtic Triangle). The area includes the Celtic FC Stadium (Celtic Park) to the east, visitor and coach car parking to the southwest, and housing (partly under construction) to the northeast. The west of the area is owned by Glasgow City Council.

2.2.2. Site Identification

Fig 4 is a map of Glasgow highlighting the spatial distribution of 46 areas (associated with 57 sites) that were identified as potentially suitable for the implementation of SUDS. Eight areas had the potential for more than one SUDS system, and were therefore subdivided into subareas (i.e., sites). Every effort has been made to investigate also areas currently represented only sparsely by discussing SUDS opportunities with planners employed by the Council.

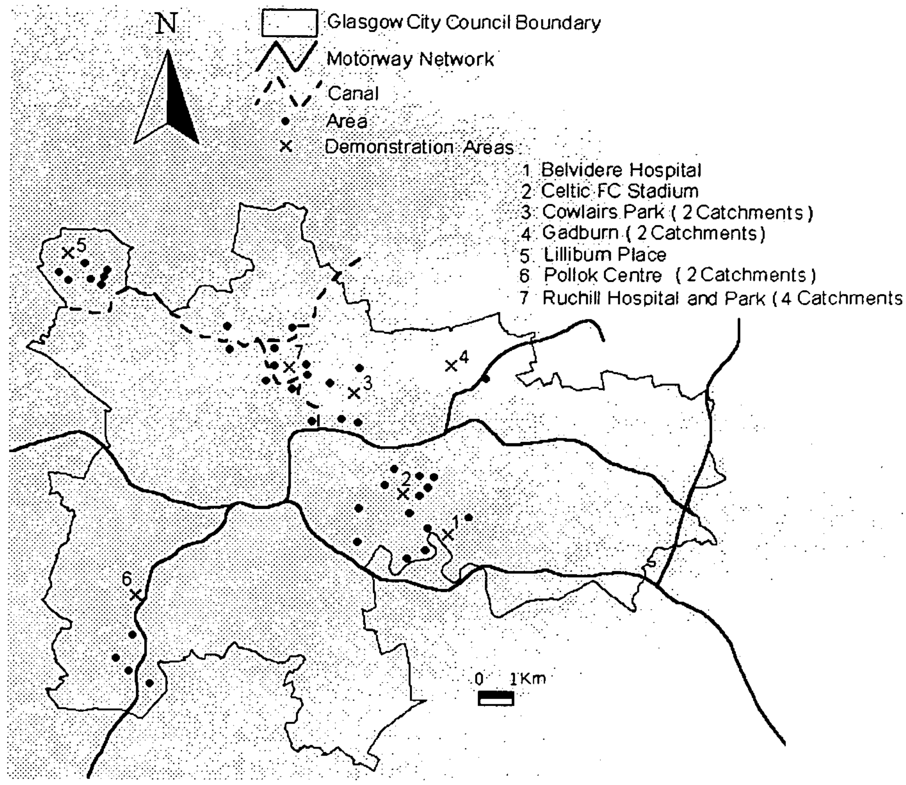


Fig 4. Indication of 46 potential areas comprising 57 sites for the implementation of sustainable urban drainage systems (SUDS). The SUDS demonstration areas have been highlighted.

2.2.3. Site Classification

Fifty-seven sites were hierarchically classified (nine levels) according to their public acceptability, land costs, water supply, drainage issues, site dimensions, slope, groundwater table depth, fragmentation of ownership and ecological value (Fig, 5). The classification was based on expert water-engineering understanding, rather than on statistical evaluation, and account for flexibility in selecting (numerical) thresholds (e.g., estimated land cost).

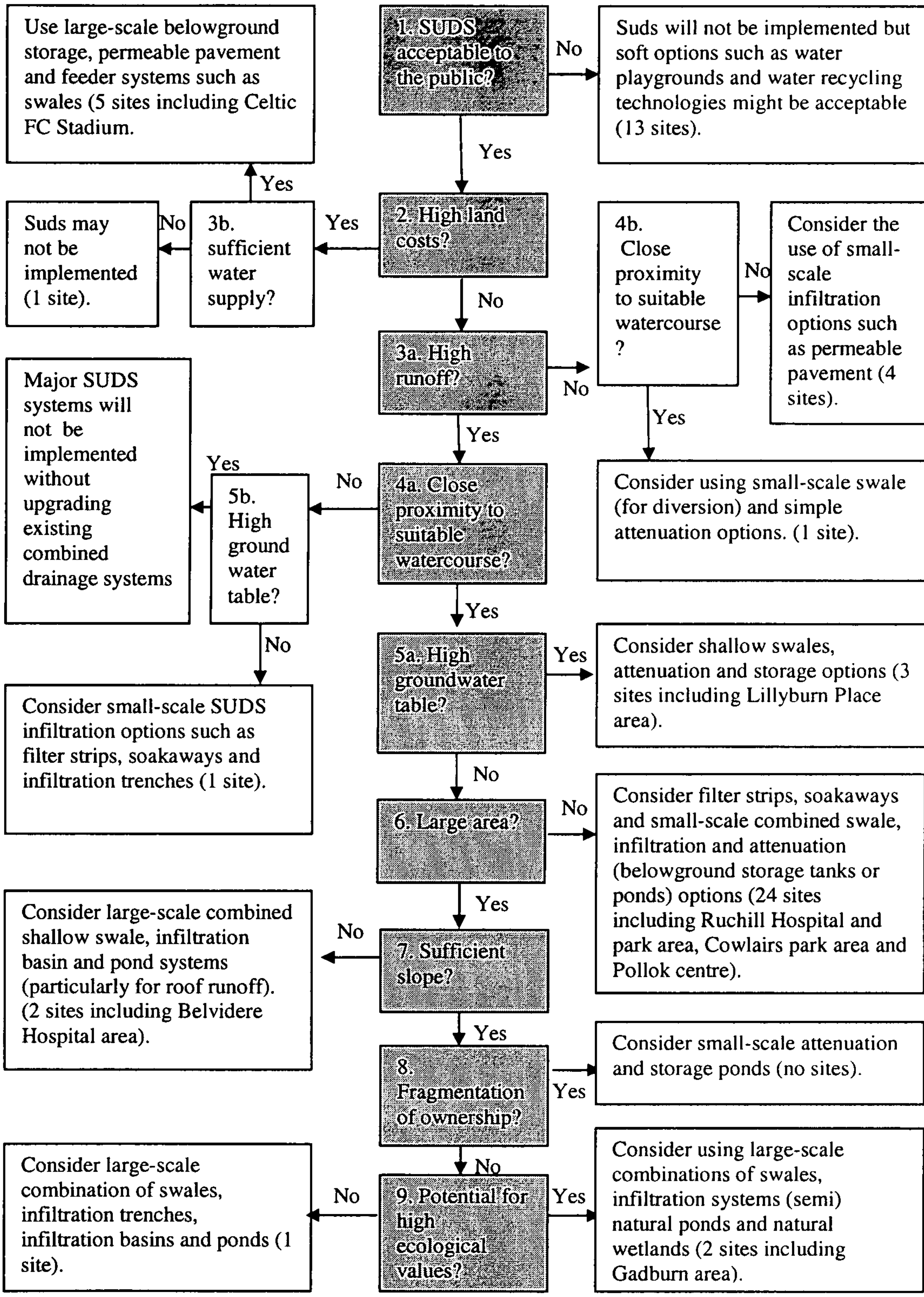


Fig 5. The sustainable urban drainage system (SUDS) decision support key and classification of 57 sites located in 46 areas available for regeneration and development

Moreover, this classification should be used as a general framework that supplements detailed frameworks and management guidelines dealing with specific regeneration issues such as leaching of metals (Kossen *et al.*, 2002).

2.3. The Edinburgh Sustainable Urban Drainage System Management Project

2.3.1. Background to Case Studies

A number of sites appropriate for the implementation of SUDS were identified within Edinburgh City. This was conducted by assessing the development and regeneration plans of Edinburgh City Council. The number of appropriate SUDS sites recognised from this study was considered insufficient for the development of an accurate SUDS decision support tool. Thus, site investigation was carried out to identify more potential SUDS sites. Fig 6 highlights the spatial distribution of 103 sites including seven demonstration sites.

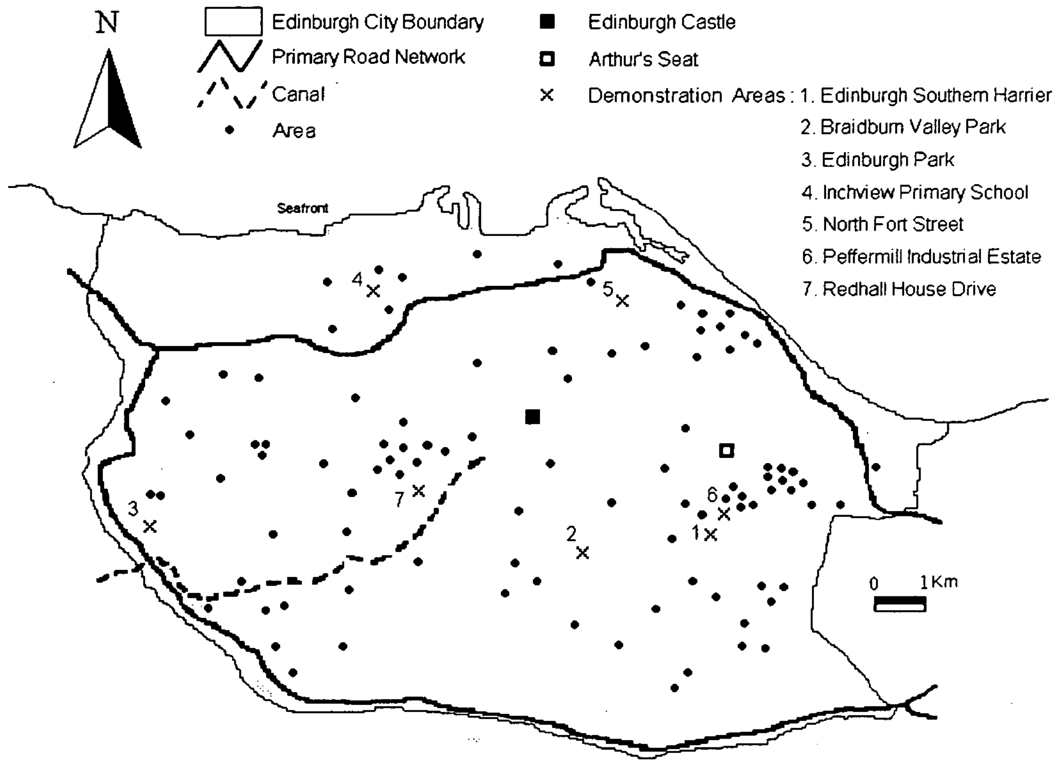


Fig 6. Spatial distribution of 103 potential SUDS sites including 7 demonstration sites.

The Edinburgh Southern Harrier site is located 5 km to the South East of Edinburgh City centre. The North Eastern boundary is adjacent to Old Dlakeith Road (A7). The site is only accessible via Ferniehill Drive (B701) located 200m further South East off the Old Dlakeith Road. The area was identified as a possible retrofitting site using recreational areas. The site consists of residential housing and tertiary roads with a public park located in the North being managed by the Edinburgh City Council. The park consists of a playground and an outdoor running track used by Edinburgh Southern Harriers Athletics Club. The residential area is located at a higher level to the South East of the park (between Fernieside Avenue and Ferniehill Drive) which can be drained to a SUDS feature located within the park.

The Braidburn Valley Park lays approximately 4 km South of Edinburgh City centre. It was identified as a suitable retrofit site using a recreational area. Braidburn Valley Park is a large public park with the Braid Burn running through it. Several opportunities were identified for SUDS implementation within the park, including the possibility to direct runoff from residential units along Greenbank Crescent (lying to the West of the park) and Comiston Road (A702) which is adjacent to the Eastern edge of the park. The drainage catchment could be extended to the East beyond Comiston Road to the edge of the Braid Hills. Construction of the residential area within this catchment started in 1905 and properties are of relatively high financial value.

New Edinburgh Limited (NEL) has been developing Edinburgh Park since 1990 and is a joint venture between Miller Group, a privately owned house building, property development and construction business, and CEC Holdings Ltd, part of the City of Edinburgh Council. The Site identified as suitable for SUDS awaits development and is currently scrubland, square in outline and is relatively flat. Edinburgh Park is an out of town business park near the South Gyle, 7 km West of Edinburgh City centre and adjacent to the Edinburgh City By-Pass and the M8 motorway to the West. The site boundaries are defined by Lochside Way to the North, Lochside Court to the East and the main Edinburgh-Glasgow Railway line to the South. The recently opened Edinburgh Park railway station borders the South East corner of the site.

The former Inchview Primary School (demolished) which is currently awaiting development. Edinburgh City Council has identified the site as suitable for affordable housing. Located 3 km North West of Edinburgh City centre, the site is enclosed by West Pilton Avenue along its Northern boundary and Ferry Road Avenue along its Southern edge. Existing housing make up the East and West boundaries. The surrounding area has traditionally been dominated by council housing, much of which is being redeveloped. Several housing sites exist within this area of Edinburgh as part of the wider regeneration program of Muirhouse and Pilton (City of Edinburgh, 2001).

North Fort Street is a small site located 3 km to the North East of Edinburgh City Centre, near Leith. The former school site is owned by Edinburgh City Council and it is proposed to develop the site for housing (City of Edinburgh, 1998). The South Western boundary of the site adjoins North Fort Street and all other boundaries are closed by high walls. The Leith area is currently undergoing considerable regeneration with several sites similar to North Fort Street existing.

Peffermill Industrial Estate is located approximately 4 km South East of Edinburgh city centre and around 4 km from the A1. Access is possible via Kings Haugh, adjacent to Peffermill Playing Fields on Peffermill Road (A6095). It is a popular development of small workshop and storage units with a number of national occupiers. The industrial estate is privately owned with 50% of the site already developed and the rest to undergo development in the near future. It was identified as a suitable SUDS site given the large amount of impermeable surfaces which would result from development and thus the increased surface water discharge. The Braid Burn is adjacent to the Northern boundary of the area, beyond which is Prestonfield Golf Course. Duddingston Road West provides an Eastern boundary and a goods railway lies adjacent to the Southern border. Access is possible through the existing industrial area situated on the Eastern section of the site. The site is part of the Edinburgh City Council's initiative for developing the South Eastern wedge of the city and the regeneration of Craigmillar area, the neighbourhood within which Peffermill industrial estate lies.

The Redhall House Drive site is located 5km to the South West of Edinburgh City centre. Redhall House mansion, which is located on the site, was built after 1755 and is now the property of Edinburgh City council. Within the grounds of this house is Graysmill School and Cairnpark School, which, as part of the Council's PPP project will be upgraded (City of Edinburgh, 2001). It is therefore possible to use retrofitting SUDS on this site to reduce surface water discharge. The site is located in mature woodland with the Water of Leith flowing 100 metres to the North. The surrounding residential area is of high financial value with Craiglockhart to the North East and Colinton to the South West.

2.4. Assessing Stormwater Detention Systems Treating Road Runoff with an Artificial Neural Network

2.4.1. Experimental system setup

Five mature detention systems (plastic crates wrapped in geotextile, and marketed as Matrix Geo-Cell, provided by Atlantis Water Management (Alderborough, Sladen Mill Industrial Complex, Littleborough, England, UK)), were located outdoors at The King's Buildings campus (The University of Edinburgh, Scotland, UK) to assess the system's performances during a period of more than one year (2005-04-01 to 2006-09-13). However, the rig was in operation since 2004-03-31.

Two plastic crates (total height, 1.7 m; length, 0.68 m; width, 0.41 m) on top of each other comprised one detention system. The tank volume below each filter was 0.08 m^3. The detention system filter volumes for all five systems were 0.24 m^3.

The bottom cell (almost 50% full at any time) was used for water storage and passive treatment only. The top cell was used as a coarse filter. Different arrangements of aggregates, and planting were used within the filtration zones of each detention system. Different packing order arrangements of aggregates, and plant roots were used in the systems (Table, 1) to test for the effects of gravel, sand, Ecosoil® (product based essentially on sand, and bark, and provided by Atlantis Water Management), block paving, and turf on the water treatment performance.

Table 1. Packing order of the stormwater detention systems

Height (mm)	System 1	System 2	System 3	System 4	System 5
861-930 (top)	Air (and common reed in summer)	Air	Block paving, and 6 mm gravel (within spaces)	Block paving, and turf (within spaces)	Air
791-860					Turf
751-790			6mm gravel	Sand, and Ecosoil®	Sand, and Ecosoil®
745-750			Geotextile	Geotextile	
711-744			Drainage cell	Drainage cell	
693-710					6mm gravel
687-692			Geotextile	Geotextile	
661-686			6mm gravel	6mm gravel	
451-660			20mm gravel	20mm gravel	20mm gravel
437-450			Sand	Sand	Sand
431-436			Geotextile	Geotextile	Geotextile
201-430	Water, and common reed		Air	Air	Air
0-200 (bottom)	Gravel (water, and roots within voids)	Water	Water	Water	Water

Systems 1, and 2 represented sand, and gravel filled constructed wetlands planted with Common Reed, *Phragmites australis* (Cav.) Trin. ex Steud), and a detention basin, respectively. Systems 3, 4, and 5 were similar to slow sand trickling filters.

Inflow water, polluted by road runoff, was collected by manual abstraction with a 2 L beaker from randomly selected gully pots on the campus. Temperature, and dissolved oxygen were measured onsite, and the corresponding water samples were subsequently transferred into the campus-based public health laboratory for further water quality analyses.

All detention systems were watered as slow as possible within 3 to 5 min approximately twice per week with 5 L gully pot liquor artificially contaminated by dog faeces (180 g), and drained by gravity afterwards to encourage air penetration through the filtration system (Gervin and Brix, 2001). The quantity of gully pot liquor used per system was approximately 3.6 × the mean annual rainfall volume (data obtained from The University of Edinburgh Weathercam Station in 2006) to simulate a 'worst case scenario'. The hydraulic residence times were in the order of one hour.

2.5. Stormwater Infiltration Systems for Road Runoff Contaminated with Organic Matter Including Dog Faeces

2.5.1. Study Site

The study site (Fig 7) was based on a combined silt trap (1), a below-ground detention tank (i.e. converted from a constructed wetland) (3 and 4), and one planted (6) and one unplanted (7) infiltration pond. The system was located in a remote area of The King's Buildings campus. The system was fed from the runoff of a nearby road (area of 730 m^2).

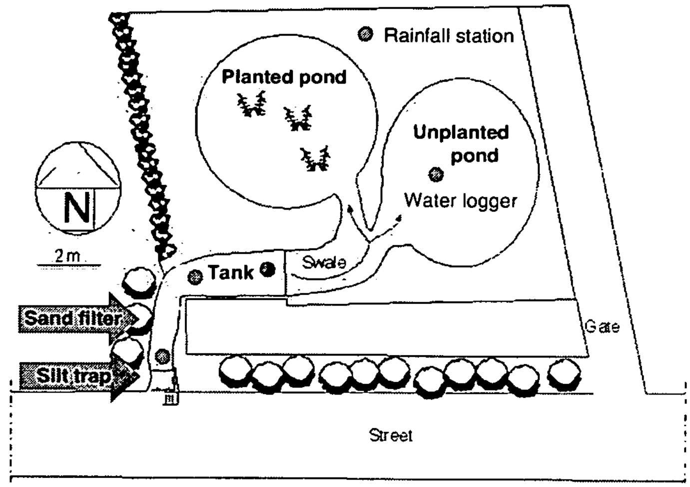


Fig 7. Case study site at The King's Buildings including a detention cell system (converted from a constructed wetland) (inflow (3) and outflow (4) provided by Atlantis Water Management.

From April 2003, a planted and an unplanted runoff demonstration pond as part of a SUDS at the King's Buildings campus of The University of Edinburgh were in operation (Fig, 7). The dominant macrophyte of the constructed wetland and planted pond were *Phragmites australis* (common reed) *and Typha latifolia* (broadleaf cattail) and Salix viminalis (whispering willow) respectively.

Precipitation from the nearby road in the campus was channelled to infiltration ponds but (filamentous) green algae began to grow - until *C. Auratus* (common goldfish) were introduced in April 2004. Twenty healthy *C. auratus* of approximately 180 g total weight were introduced into each pond. Both watercourses were covered with a plastic mesh to prevent animals such as *Ardea cinerea* (grey heron) and *Felis cattus* (cat) to prey on the fish.

From April 2005, approximately 400g/week of fresh dog excrements were added directly to the silt trap protecting the ponds predominantly from solid contaminants.

This was to represent the real-life conditions for SUDS systems in urban areas and since pets were not allowed at the University campus the faecal contamination required to be manually introduced. A comprehensive survey was carried out to determine the amount of dog faeces presented in the urban areas of the city of Edinburgh. 33 sites were identified within the city boundary and surveys were conducted for each site individually. The amount of dog droppings per m^2 was calculated based on factors like freshness of the contamination and the type of area.

In May 2005 the existing constructed wetland located between the silt trap and the ponds was converted into a belowground detention tank aiming to in order to represent belowground detention facilities in a combined SUDS structure and to prevent contamination from dissolved organic pollutants and potentially pathogenic organisms. The fish were monitored to determine whether *C. auratus* could cope with any additional nutrient (particularly nitrogen and phosphorus) load and total coliforms and intestinal enterococci (pathogenic bacteria) build-up. All water quality determinations were undertaken according to standard methods (Clesceri *et al.*, 1998).

Runoff, which was naturally contaminated with organic matter including leaves, entered a silt trap immediately at the beginning of the system where it was artificially contaminated with dog faeces twice a week. The runoff was then filtered through a combined sand and gravel filter (2), and then stored and passively treated in below-ground detention cells, before it overflowed into a swale (5), which conveyed the pretreated runoff into two parallel infiltration wetlands of which one was planted and the other one was unplanted

There were a number of designated sampling points within the silt trap and the storage tank. Water samples from the ponds were also taken immediately to the lab for further analytical examination. The water level and storm event frequency within the unplanted pond was recorded all five minutes via a digital water level logger.

2.6. Combined Bio-filtration, Stormwater Detention and Infiltration System Treating Road Runoff

2.6.1. Study Site

The combined system was constructed in March and April 2006, and has been in operation since May 2006. The experimental site is located in Edinburgh at The King's Buildings campus, The University of Edinburgh. The system is in essence a combination of a gravel filter, and combined detention and infiltration tank, operated in series (Fig, 8). The system has been designed to control and treat urban runoff from a small adjacent car park covered entirely by asphalt. The car park has an area of 640 m^2 and a slope of approximately 1%.

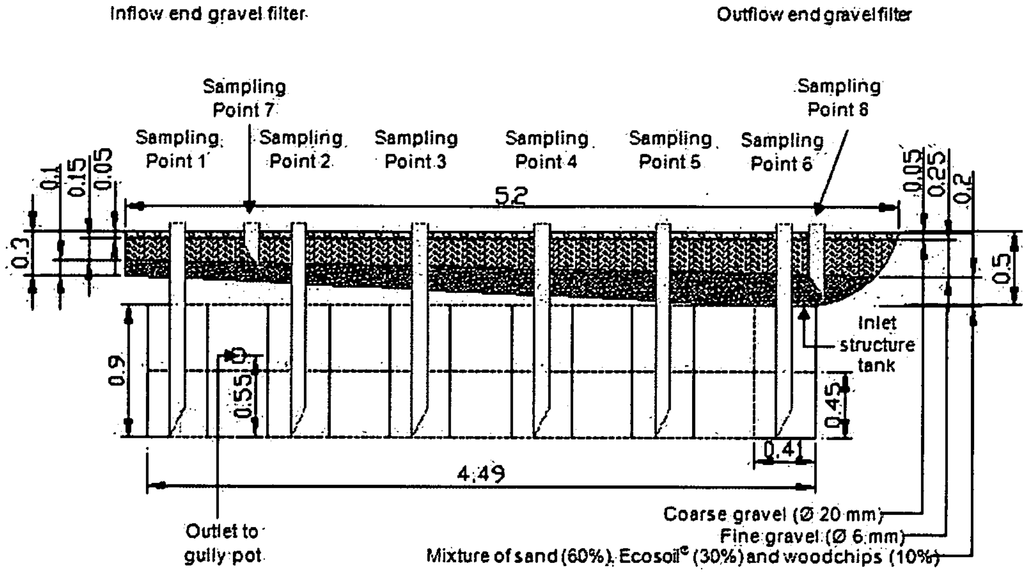


Fig 8. Cross-section view of the system.

From the car park entered the gravel filter via a kerbside inlet. The dimensions of the filter can be found in Table 2. The gravel filter is made up of three layers: the top layer is a gravel layer with a thickness of 50 mm and mean grain size of 20 mm, the middle layer is a gravel layer ranging in thickness from 150 m to 250 mm with a mean grain size of 6 mm, and the lower layer is a mixture of sand (60%), Ecosoil (30%; supplied by Alderburgh Ltd.) and woodchips (10%) with a thickness between 100 and 200 mm.

Table 2. Dimensions of the gravel filter

Gravel Filter Dimensions	
Length	4.50 m
Width at inlet	1.30 m
Width at outlet	1.40 m
Depth at inlet	0.30 m
Depth at outlet	0.50 m
Surface area	7.0 m^2
Total volume	4.4 m^3

A geotextile separates the middle from the lower layer. The bottom of the filter is separated from the underlying soil by a plastic liner. This ensures that no infiltration occurs from this section of the system. A row of willows was planted in the filter aiming to enhance nutrient removal, and helping the system to integrate into the natural environment, subsequently enhancing local urban aesthetics. The filter is slightly inclined. Therefore, the water accumulates in the lower end of the filter before entering the subsurface tank.

The stormwater tank consists of Matrix II tank modules (plastic mesh boxes supplied by Alderburgh Ltd.). In total 132 of these modules were used to construct the infiltration tank (11 modules long, 6 modules wide and 2 modules deep), giving the system a total void volume of 14.95 m^2. Dimensions of the modules and of the tank are provided in Table 3. An impermeable plastic liner separates the rows of modules within the tank, resulting in an increased length of the flow path, which enhances pollutant removal. Both the top and bottom of the tank were lined with a geotextile. There are two possibilities for the fate of the stormwater within the tank; it either is detained and infiltrated into the soil or it overflows into the sewer system via the outlet structure.

Table 3. Stormwater detention tank properties

Stormwater Tank Properties		
Module Properties		
Length	0.408	m
Width	0.685	m
Depth	0.450	m
Tank Properties		
Length	4.49	m
Width	4.11	m
Depth	0.90	m
Surface area	18.45	m^2
Volume	16.60	m^3
Void volume	14.94	m^3
No. of modules	132	

Overflowing occurred only on occasions when the water level in the tank was >0.55 m. When the water depth within the tank exceeded this threshold, treated runoff discharged through a plastic pipe (i.e. outlet) into a modified gully pot, and subsequently in the sewer.

The infiltration system was equipped with five aeration pipes and eight sampling wells permitting water to be abstracted from the system at various locations. Sampling points 3 to 8 provided access to water within the tank. Point 1 was located at the inlet to the filter and point 2 was situated at the filter outlet. The sampling wells comprised a perforated plastic pipe wrapped around with geotextile.

Chapter 3

Materials and Methods

3.1. The Glasgow Sustainable Urban Drainage System Management Project

3.1.1. SUDS Decision Support Key

Sustainable urban drainage should not cause any public health problems, avoid pollution of the natural environment, minimize the use of resources, operate in the long term, and be adaptable to change in requirements (Butler and Parkinson, 1997). Taking this statement into consideration, the following list of criteria for defining SUDS options and a corresponding summary matrix (Table, 4) has been proposed:

Runoff (low or high): the site has to be associated with a potential source of water (e.g., car park runoff) that results in sufficient runoff (to be defined on a case by case basis).

Catchment size (specified for individual SUDS options): the site needs to have sufficiently large dimensions (e.g., width ≥150 m and length ≥300 m).

Area suitable for SUDS (specified for individual SUDS options): the site has to be acceptable for development, regeneration, or retrofitting to Glasgow City Council, developers, and the wider public (e.g., Greenfield and Brownfield areas). The site should also be associated with a separate area to which water can drain (e.g., canal or river).

Serious soil contamination (yes or no): the site should not be associated with major soil contamination issues.

Land value (low, medium, high, or not applicable): the land costs should preferably be not too high (e.g., <£200/m^2) before development or regeneration work has commenced.

Fragmentation of ownership (yes or no): the site should preferably be owned by only a few individuals or organisations (e.g., <5 parties).

High groundwater level (yes, no or not applicable): the site should preferably be associated with a low groundwater table (e.g., groundwater level >2 m below ground level).

Sufficient channel slope (yes, no or not applicable): the site should have a sufficient slope (e.g., ≥ 1 in 50 m) to enable conveyance structures to function properly. However, the site should not be too steep to make three-dimensional SUDS features too expensive.

Potential of high ecological impact (yes, no or not applicable): the site should be of potentially high ecological impact in the future, but not during the planning phase (e.g., not a site of specific scientific interest [SSSI]).

Soil infiltration (low, high, or not applicable): The site should have sufficiently high soil infiltration, if filtration is considered to be desirable for the proposed SUDS structure.

The representative SUDS demonstration areas have been selected based on these criteria (see above and Table, 4).

Table 4. Sustainable urban drainage system (SUDS) decision support matrix.

	Run off	Catch-Ment Size (m^2)	Area of suita-bility for SUDS feature	Seri-ous conta-minati-on	Land value	Own er-ship frag-mented	High Ground water level	Suffi-cient hannel slope	Potent-ial of high ecolog ical impact	Soil infiltr ation
Wetlands	High	>50000	>5000	No	<2	No	N/A	N/A	Yes	N/A
Ponds	High	>15000	>50	No	<3	Yes	N/A	N/A	N/A	N/A
Lined ponds	High	>15000	>50	Yes	<3	Yes	N/A	N/A	N/A	High
Infiltra-tion basins	High	>15000	>50	No	<3	Yes	No	N/A	N/A	N/A
Swale	High	N/A	>200	No	<3	No	No	Yes	N/A	N/A
Shallow swale	High	N/A	>200	No	<3	No	Yes	Yes	N/A	High
Filter strip	High	>15000	>600	No	<3	Yes	No	Yes	N/A	High
Soak-away	Low	>3000	>200	No	<3	Yes	No	Yes	N/A	High
Infiltra-tion trench	Low	>3000	>50	No	<3	No	No	Yes	N/A	High
Permeable Pavem-ent	Low/ High	N/A	N/A	No	N/A	Yes	N/A	N/A	N/A	High
Below ground	Low/ High	N/A	>40	Yes	N/A	Yes	N/A	N/A	N/A	N/A

storage										
Supple-mentary water play ground	High	>200	>10	No	N/A	Yes	Yes	N/A	N/A	N/A

Land Value: 1=Low (<£100/m^2), 2=Medium ($\geq$ £100/m^2 and $\leq$£200/m^2), 3= High (>£200/m^2)

Only seven areas suitable for SUDS implementation are not represented by the selected demonstration areas (Fig, 5). However, it has to be emphasized that the selection is rather qualitative than quantitative considering that most selection criteria do not require a numerical assessment. It follows that the SUDS classification is similar to an expert system, and not to a statistically unbiased assessment that would not be suitable in this case anyway because of the lack of numerical information such as land value (e.g., recognizing also the future potential after regeneration).

3.1.2. Fieldwork activities

Soil was sampled twice: selected samples were initially taken at a few locations where major SUDS structures were likely to be implemented. Composite samples were taken at 10 cm depth intervals within trenches of up to approximately 55 cm depth. Further samples were taken at locations that are part of a proposed wider SUDS structure and that were located most closely to the nearest node of a randomly placed 50 × 50 m equally spaced sampling grid. Only one sample at 50 cm depth per sampling site was taken during a second expedition. If a sampling location was not acceptable (e.g., below tarmac or a house), an alternative representative sampling location was determined up to 5 m (if not stated otherwise below) away from the original location. However, if no sampling locations deemed to be appropriate, the location was not sampled and a "not accessible" entry was located on the map showing the sampling strategy and locations.

3.1.3. Analytical Work

The soil recording and pre-treatment before analysis was carried out in agreement with British Standards (British Standard Institute, 1999a, 2002). The determination of particle size distribution was also carried out according to British Standards (British Standard Institute, 1999b). Composite samples were collected and stored at -10°C prior to analysis. After thawing, approximately 2.5 g of each soil sample was weighed into a 100-mL digestion flask to which 21 mL of hydrochloric acid (strength of 37%, v/v) and 7 mL of nitric acid (strength of 69%, v/v) were added. The mixtures were then heated on a Kjeldahl digestion apparatus (Fisons, UK) for at least 2 h. After cooling, all solutions were filtered through a Whatman Number 541 hardened ashless filter paper into 100 -mL volumetric flasks. After rinsing the filter papers, solutions were made up to the mark with deionized water. The method was adapted from the section "Nitric Acid-Hydrochloric Acid Digestion" (American Public Health Association, 1995).

An Inductively Coupled Plasma Optical Emission Spectrometer (ICP-OES) was used for the analysis of metals and other heavy elements. Total concentrations of elements in filtered (Whatman 1.2-μm cellulose nitrate membrane filter) samples were determined by ICP-OES using a TJA IRIS instrument (ThermoElemental, USA).

Multi-element calibration standards with a wide range of concentrations were used and the emission intensity measured at appropriate wavelengths.

Concerning the analysis of major nutrients, 2 mL sulphuric acid (strength of 98%, v/v) and 1.5 mL hydrogen peroxide (strength of 30%, v/v) were used as an extraction media (Allen, 1974). Approximately 0.1 g of each dried sample and the associated digestion media were placed in a tube and heated at 320°C for 6 h. Aliquots were taken and digests were made up to 100 mL with distilled water.

For analysis of total nitrogen (Ntotal), the following procedure was adopted: Ammonium (present in the digest) reacts with hypochlorite ions generated by alkaline hydrolysis of sodium dichloroisocyanurate. The reaction forms monochloroamine, which reacts with salicylate ions in the presence of sodium nitroprusside to form a blue indephenol complex. This complex is measured colorimetrically at 660 nm using a Bran & Luebbe (Northampton, UK) autoanalyzer (model AAIII).

For analysis of total phosphorus (Ptotal), the following procedure was adopted: Ortho-phosphate (present in the digest) reacts with ammonium molybdate in the presence of sulphuric acid to form a phosphomolybdenum complex. Potassium antimonyl tartrate and ascorbic acid are used to reduce the complex, forming a blue color, which is proportional to the Ptotal concentration. Absorption was measured at 660 nm using a Bran & Luebbe autoanalyzer (model AAIII).

For the analysis of total potassium (Ktotal), the digest was analyzed by a flame atomic absorption spectrometer (Unicam 919, Cambridge, UK) at a wavelength of 766.5 nm and with a bandpass of 1.5 nm. Standards were prepared in 100-mL flasks using 2 mL concentrated sulphuric acid and 1.5 mL hydrogen peroxide (30% v/v) and made up to mark with deionized water. Caesium at a concentration of 100 mg/L was added to both standards anddigests to overcome ionisation.

Subsamples of 3 ±0.1 g of field moist soil were mixed with an excess of sodium sulphate (Analytical Grade, Fisher, UK) to make it "free flowing" and the resulting mixture extracted in 10 mL of HPLC grade dichloromethane (Fisher, UK) in an ultrasonic bath (Model XB14, Grant instruments, Cambridge, UK) for 10 minutes. After agitation, samples were filtered through 0.45m nylon syringe filters (Qm_X Laboratories Limited, Thaxted, UK).

The sample extracts were scanned for the presence of organic contaminants by HP 6980 gas chromatograph coupled to HP 6973 mass spectrometer. A 4 μL aliquot of each sample was injected in the splitless mode onto a 30 m HP5MS fused silica column directly coupled to the ion source of an HP 6973 mass spectrometer.

The mass spectrometer was run in the scanning mode with a mass range of 50 to 700. Identification of the peaks on the total ion chromatograms was made using libraries of preinstalled databases of reference spectra. An initial peak width and initial threshold values were set to identify significant peaks.

All metal and nutrient analyses were conducted at the Contaminated Land Assessment & Remediation Research Centre (CLARRC) at the University of Edinburgh.

3.1.4. Data Analysis and Software Used

The data analysis was carried out using Microsoft Excel, and statistical methods outlined elsewhere (Fowler and Cohen, 1998) were applied. ArcView (GIS and Mapping Software 9.x, ESRI) was used to draw design proposals.

3.1.5. Belvidere Hospital Area Case Study

The Belvidere (not Belvedere as usually read) Hospital area is located approximately Longitude 4°12´ West and Latitude 55°51´ north. The area has been cleared of all surface structures for new housing, with one remaining former hospital building, which is a Grade B Listed Building. However, parts of the area contain residual housing foundations below the current ground level. Nevertheless, the overall topography of the site is even.

Fig 9. Belvidere Hospital area: site photograph taken on 14 May 2004 (Scholz, *et al.*, 2006).

Future development of this site for housing will require the removal of all residual foundations and asphalted areas (Figs. 9-12). The main entrance driveway of the original hospital still exists with two large semi-vegetated areas (mainly rows of tall trees) flanking both sides. The remaining building on the area is situated to the west. To the south of the building is a steep embankment covered in dense woodland. The slope increases approximately from east to west, and is at its maximum 60°. At the base of this embankment (not within the area boundary marked by a 3 m high corrugated iron fence), runs a public walk and cycle path along the River Clyde. The height difference from the crest of the embankment down to the walkway is approximately 11 m. However, this area is likely to remain unaffected by any building and road construction works due to its potentially high ecological and amenity value.

Fig 10. Belvidere Hospital area: artist impressions of proposed site development (pencil drawing and computer animation) (Scholz, *et al.*, 2006).

A desk study concerning the Belvidere Hospital proved to be unrewarding, as there were no historical documents held by Glasgow City Council pertaining to this area. However, the area is known to have been a hospital for approximately 100 years, and during this time the hospital grounds were subjected to the cleaning of hospital equipment, and might be contaminated with diffuse hospital waste.

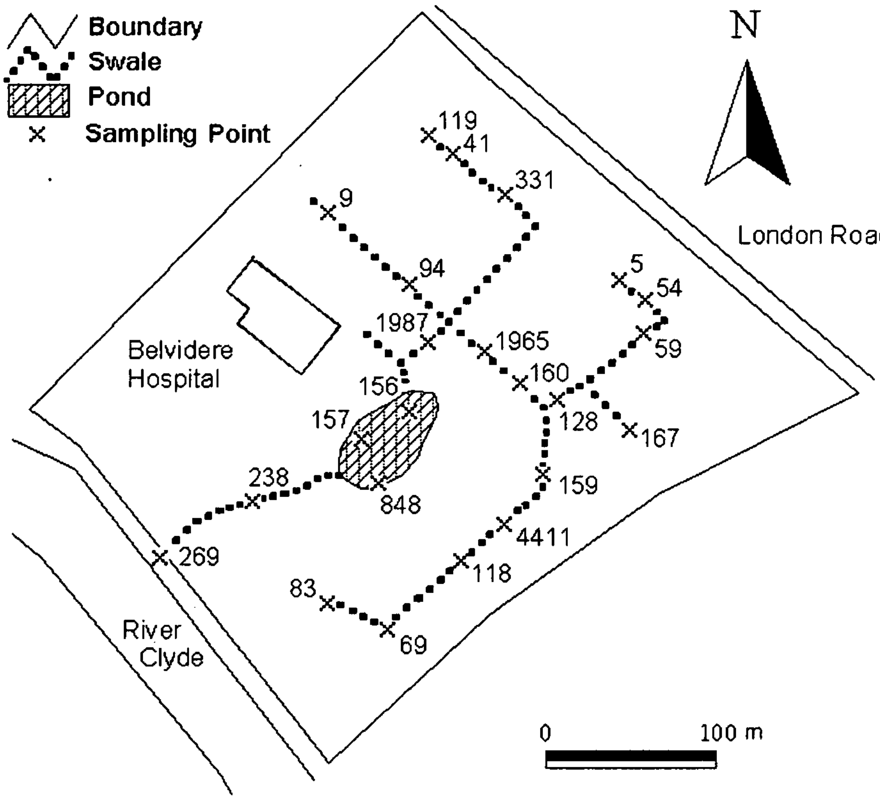


Fig 11. Belvidere Hospital area: spatial distribution of lead (mg/kg dry weight) at 50 cm depth on 5 July 2004.

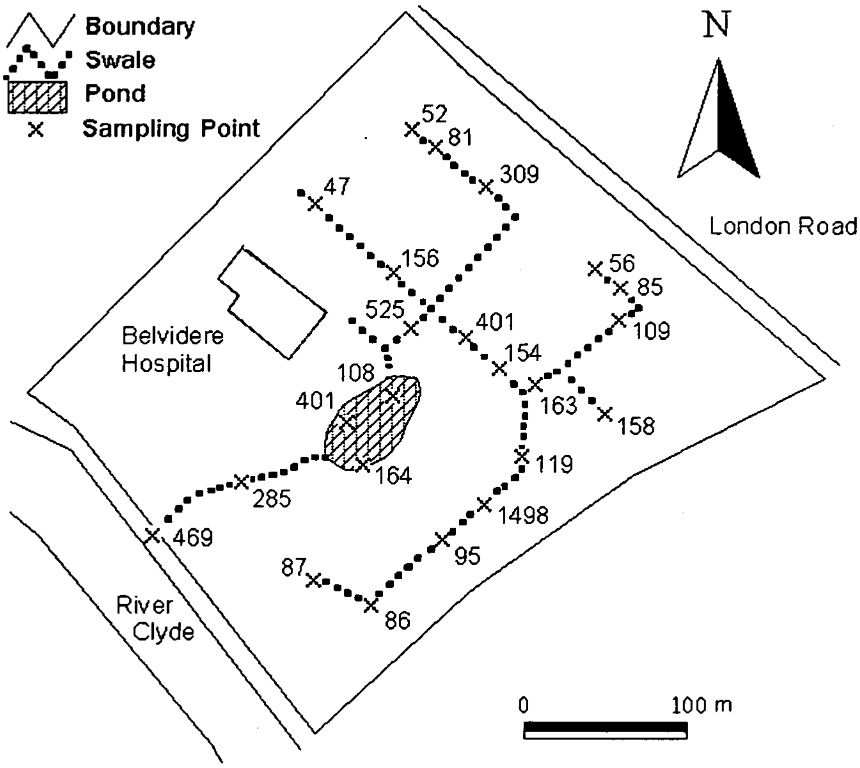


Fig 12. Belvidere Hospital area: spatial distribution of zinc (mg/kg dry weight) at 50 cm depth on 5 July 2004.

3.1.6 Celtic FC Stadium Area Case Study

The Celtic FC Stadium Area is approximately bordered by Janefield Street in the northeast, Stamford Street in the northwest, Dalriada Street in the southeast, and Lon don Road in the south. A major part of the area in the west is used as a car park. The Celtic stadium is located in the southeast of the demonstration area. The immediate Celtic FC Stadium Area is located approximately Longitude 4°13´ west and Latitude 55°51´ north. The Celtic Park stadium has capacity for approximately 60,000 spectators, and occupies a prime location in the heart of Glasgow's East End. On home match days, the stadium is generally filled to capacity (approximately 26 times per year between July and May), and this volume of visitors to the area clearly impacts on any integrated SUDS in the future.

Previous site investigations in this area show that the site is underlain by sandstones, siltstones, and mudstones, with seams of coal belonging to the Lower and Middle Coal Measures of the Carboniferous System. The natural superficial deposits are indicated on glacial maps to be alluvial clay and silt, partly overlain by made ground. The total thickness of superficial deposits is indicated to be between 20 and 30 m. The 1980 investigation of the areas as recorded in the Glasgow City Council's Geodatabank showed the general succession to be made ground with a thickness between 1.7 and 10.8 m and clay with sand bands with a thickness of at least 2 m (Glasgow City Council, 1995). The granular constituents of the made ground are in a generally medium dense state of compaction, but the cohesive constituents are generally in a soft or very soft state (Glasgow City Council, 1980).

The groundwater level in this area is at a depth between 7 and 8 m, based on borehole data. Seepage of water was recorded in a couple of boreholes, but the report indicates that groundwater did not gather during the time of boring in the remaining boreholes. However, it is possible that pockets of perched groundwater may occur anywhere in made ground (Glasgow City Council, 1980).

The northern parts of the site, the areas located between Janefield Street and Stamford Street, are currently under redevelopment. There is a coach parking located in the East of the area that is expected to be retained. There are a number of occupied and unoccupied low standard housing blocks in the West. An older housing estate will be demolished, and the site will subsequently be redeveloped. The East End regeneration route will be located to the west of the demonstration area. It follows that large belowground storage facilities would be required to attenuate highway runoff in the future.

3.2. The Edinburgh Sustainable Urban Drainage System Management Project

3.2.1. Site Classification with the SUDS Decision Support Key

The SUDS decision support key was developed during the Glasgow SUDS Management Project (Scholz, 2004) containing nine SUDS site characteristics. The characteristics are public acceptability, land costs, runoff quantity, drainage issues, groundwater table depth, site dimensions, slope, fragmentation of ownership and ecological value. Site classification was carried out during the site visits using the SUDS decision support key. The decision support key is based on expert water engineering understanding, rather than on mathematical and statistical evaluation, and account for flexibility in selecting (numerical) thresholds (e.g. estimated land cost) (Scholz, 2004).

The results from the decision support key gave an outline classification of SUDS sites into various SUDS classes where the most appropriate SUDS practice for implementation is identified.

The data analysis was undertaken using Microsoft Excel and statistical methods outlined elsewhere (Stroud, 1995; Kreyszig, 1999) was applied. AutoCAD was used for all design proposals.

3.2.2. Computer-based SUDS Decision Support Tool

A computer-based decision support tool (DST) was developed competent of identifying appropriate SUDS practices for all 103 sites studied in Edinburgh. This is comprehended by analysing all site classification data to achieve an output in terms of a SUDS practice. The outcome may either be a singular technique or a combination of two techniques effectively producing a SUDS treatment train.

3.2.2.1. Process

The data collected from all the 103 Edinburgh SUDS sites were trimmed in the DST database. Site data fields had to be formatted in order to produce singular numerical values to be used in the DST. Other data fields could be used directly as they already were in the form of simple 'Yes' or 'No' statements. The following were considered prior to the main data function:

All sites were analysed to identify the possible presence of a sewer, a flowing watercourse or both within them and whether they will be able to receive water discharges from a SUDS feature.

The land cost estimation was normalised to a numerical value between one and five with values relating to low, low-medium, medium, medium-high and high. A value of one therefore referred to low land value and five to a high land value.

The slope data for each site (or each sub-catchment within a site) determined from contour maps (Site Study) was used to calculate the gradient of the ground in degrees.

The surface area available for a potential SUDS feature within a site was calculated from dimensions established during site visits. If the site had more than one appropriate location for a SUDS feature, then the area of each location was calculated and the maximum, minimum and average area values were noted.

The total horizontal surface areas of the sites were calculated from planimeter readings. Each site had three planimeter readings and the average value was chosen. This value was then scaled using the relevant scale from the map used. The area calculation using a planimeter was accurate and therefore the computer decision DST overrides the dimension and area data determined by judgement. The impermeable areas of the sites were analysed using the horizontal surface area of the catchments and estimated future runoff. All sites were accredited particular runoff coefficients which approximate the future impermeable area in terms of the proposed use of the site, as shown in Table 5. The future impermeable area of retrofit sites was unlikely to change from the existing condition and therefore retrofit sites were given coefficients which represented the current use of the site. If recreational areas were to be used for retrofitting then the nature of the drainage catchment within which the recreational area lied was represented by an appropriate coefficient. Development and regeneration sites were given a coefficient based on the probable future use of the site. Sites consisted of different sources of runoff and were hence assigned altering runoff coefficients. The DST took the maximum coefficient value for each site. This was then multiplied by the horizontal catchment surface area, giving an impermeable area for each site.

Table 5. Runoff coefficients.

Runoff Sources	Coefficient
Car Park	0.90
Motorway	0.80
Primary Road	0.70
A-Road	0.50
B-Road	0.40
Tertiary Road	0.30
Institutional	0.70
Commercial	0.75
Industrial	0.80
High Density Housing	0.80
Medium Density Housing	0.60
Low Density Housing	0.40

A runoff value was then attributed to each site, ranging from one to five (one being the lowest), depending on the impermeable area size. This runoff value assessment was achieved by a number of operations. The impermeable area values were divided into a number of 'narrow' classes (of 5000, m^2 in size) to produce a frequency curve. Irregularly large or small values were removed from the curve for analysis purposes. This gave a new maximum and minimum impermeable area value. The average values of the remaining impermeable areas were then found. These values were now the maximum, minimum and the average impermeable area sizes. A margin was set at 20%.The maximum impermeable area had 20% of its value deducted from itself to give a threshold value. This value was the threshold for the highest runoff category. Any impermeable area above this threshold value was given a runoff value of five. An upper mean value was then calculated between the highest runoff threshold and the average value of all impermeable areas. Any impermeable area falling between this upper mean and the high runoff threshold was given a runoff value of four. Impermeable area values falling between the upper mean and the overall average impermeable area value received a runoff value of three. A lower mean value was calculated between the overall average impermeable area value and the minimum impermeable surface area. Sites falling between this lower mean and the average impermeable area were given a runoff value of two. Any site with a value less than this lower mean received a runoff value of one.

Runoff quality was analysed by assessment of its source. Roof runoff was commonly less contaminated than road runoff (Martin, 2000). The road runoff sources deemed to have a negative effect on water quality was defined as motorways, primary roads, A-roads and large car parks. Sites draining runoff from more than one of these sources were considered to have poor water quality and were given a score of three. Sites draining only one of the mentioned road runoff sources were considered to have average water quality and were given a score of two. If it was not possible to drain any of the mentioned road runoff sources to a SUDS feature, then the runoff was considered as containing little contaminants and was given a score of one.

3.2.2.2. Final Data Arrangement

The final data output to be used further in the DST was shown in the 'Outputs Sheet' of the DST. The characteristic variables of the following data fields were either produced by the DST or chosen directly from the obtained site data. They were employed in series in the DST to determine the appropriateness of a site for the implementation of SUDS.

The data fields were shown with the characteristic variables in brackets:

A) General Site Acceptability (Yes; No)

B) Final Runoff Value (1; 2; 3; 4; 5)

C) Scaled Catchment Area Size (m^2)

D) Maximum Area for suitable SUDS locations (m^2)

E) Serious Contamination (Yes; No)

F) Land Cost Estimation Value (1; 2; 3; 4; 5)

G) Number of Owners (value)

H) High Ground Water Level (Yes; No)

I) High Ground Infiltration (Yes; No)

J) Slope for SUDS transfer structures (value)

K) Runoff Quality Value (1 (good); 2; 3)

L) Potential for High Ecological Impact (Yes; No)

M) Drainage to Watercourse (Yes; No)

N) Drainage to Sewer Only (Yes; No)

O) Drainage to Sewer or Watercourse (Yes; No).

3.2.2.3. Decision Support Matrix for a Singular SUDS Technique

Table 6 compares the final data field outputs produced for each site with the decision support matrix.

Table 6. Single Sustainable Urban Drainage System (SUDS) decision support matrix.

	Final Runoff Values[A] 1-5	Min Scaled Catchment size (m^2)	Min Area Suitable for SUDS Feature (m^2)	Serious Contaminant Allowable (Y/N)	Land Cost Estimation Values[B] 1-5	Ownership Fragmented Value	High GWL (Y/N)	High Ground Infiltration (Y/N)
Wetland	≥3	65,000	1000	N	<3	<3	Y	NA
Basin / Pond	NA	25,000	100	N	<5	NA	Y	NA
Lined Basin Pond	NA	25,000	100	Y	<5	NA	NA	NA
Infiltration Basin	NA	25,000	100	N	<5	NA	N	Y
Wet Swale	<4	10,000	100	N	<4	<10	Y	NA
Dry Swale	<3	10,000	100	N	<4	<10	NA	NA
Lined Swale	<3	10,000	100	Y	<4	<10	NA	NA
Green roof	<3	NA	NA	NA	NA	<3	NA	NA
Infiltration trench	<4	10,000	100	N	<5	<10	N	Y
Soakaway	<2	NA	10	N	NA	NA	N	Y
Filter strip	<4	NA	300	N	<3	<4	NA	Y
Pervious pavement	NA	NA	NA	N	NA	NA	NA	NA
Belowground storage	<5	NA	NA	N	>2	NA	NA	NA
Water playground	<2	1,000	10	N	NA	NA	NA	NA

	Max Slope for SUDS Transfer Structures	Min Slope for SUDS Transfer Structures	Runoff Quality[C] 1-3	Potential for High Ecological Impact (Y/N)	Drainage to Watercourse Only (Y/N)	Drainage to Sewer Only (Y/N)	Drainage to Sewer or Watercourse (Y/N)
Wetland	NA	NA	<3	Y	Y	N	NA
Basin / Pond	NA	NA	≤3	NA	Y	N	NA
Lined Basin Pond	NA	NA	<3	NA	Y	N	NA
Infiltration Basin	NA	NA	<3	NA	NA	NA	Y
Wet Swale	10	0.5	<3	NA	NA	NA	Y
Dry Swale	10	0.5	≤3	NA	NA	NA	Y
Lined Swale	10	0.5	<3	NA	NA	NA	Y
Green roof	NA	NA	NA	NA	NA	NA	Y
Infiltration trench	NA	0.5	<3	NA	NA	NA	NA
Soakaway	NA	NA	<3	NA	NA	NA	NA
Filter strip	6	2	<3	NA	NA	NA	Y
Pervious pavement	NA	NA	NA	NA	NA	NA	NA
Belowground storage	NA	NA	<3	NA	NA	NA	Y
Water playground	NA	NA	<2	NA	NA	NA	Y

In the singular SUDS technique support matrix, the variables set were the essentially satisfying numerical parameters for a SUDS technique to be considered as feasible. Each site had its output fields checked in series by the DST to establish the most appropriate SUDS technique.

The outcome for each site was a list of 'Yes' or 'No' values stating the most possible SUDS techniques out of the twelve for that particular site.

3.2.2.4. Combined Singular SUDS Techniques

Possibility analyses for combining two SUDS techniques were determined using the outcomes of the single SUDS techniques analysis. In cases that two singular SUDS techniques were feasible for a particular site then the DST would combine the two to form a SUDS management train (Jefferies *at al.*, 1999). Seven basic SUDS management trains were considered in this analysis: dry swales and wetland, dry swales and basin / pond, lined swales and lined basin / pond, dry swales and infiltration basin, infiltration trench and infiltration basin, green roof and soakaway, and pervious pavement and belowground storage.

The DST confirmed whether the two chosen SUDS options in the combination are appropriate according to the single SUDS decision support matrix. A 'Yes' output would indicate the feasibility of particular combination and a 'No' if not.

3.2.2.5. Decision Support Matrix for Dual SUDS Techniques

Table 7 shows a dual SUDS technique decision support matrix. The table analyses the site output data to determine the most feasible SUDS combinations.

Table 7. Dual SUDS technique decision support matrix.

	Final Runoff Values[A] 1-5	Min Scaled Catchment size (m^2)	Min Area Suitable for SUDS Feature (m^2)	Serious Contaminant Allowable (Y/N)	Land Cost Estimation Values[B] 1-5	Ownership Fragmented Value	High GWL (Y/N)	High Ground Infiltration (Y/N)
Dry Swales & Wetland	≥3	70,000	1,250	N	<3	<3	Y	NA
Dry Swales & Basin / Pond	NA	30,000	125	N	<4	<10	Y	NA
Lined Swales & Lined Basin / Pond	<4	30,000	125	Y	<4	<10	NA	NA
Dry Swales & Infiltration Basin	NA	30,000	125	N	<4	<10	N	Y
Infiltration Trenches & Infiltration Basin	NA	30,000	125	N	<5	<10	N	Y
Green Roof & Soakaway	<3	NA	10	N	NA	<3	NA	Y
Pervious Pavement & belowground Storage	<5	NA	NA	N	>1	NA	NA	NA

	Max Slope for SUDS Transfer Structures	Min Slope for SUDS Transfer Structures	Runoff Quality[C] 1-3	Potential for High Ecological Impact (Y/N)	Drainage to Watercourse Only (Y/N)	Drainage to Sewer only (Y/N)	Drainage to Sewer or Watercourse (Y/N)
Dry Swales & Wetland	10	0.5	<3	Y	Y	N	NA
Dry Swales & Basin / Pond	10	0.5	≤3	NA	NA	NA	Y
Lined Swales & Lined Basin / Pond	10	0.5	<3	NA	NA	NA	Y
Dry Swales & Infiltration Basin	10	0.5	<3	NA	NA	NA	Y
Infiltration Trenches & Infiltration Basin	10	0.5	<3	NA	NA	NA	Y
Green Roof & Soakaway	NA	NA	NA	NA	NA	NA	NA
Pervious Pavement & belowground Storage	NA	NA	<3	NA	NA	NA	Y

The variables set in the dual SUDS technique decision support matrix were the essentially satisfying numerical parameters for a combined SUDS technique to be considered as feasible. Each site had its output fields checked in series by the model to determine the feasibility of the SUDS combination.

The outcome for each site was a list of 'Yes' or 'No' values stating the most possible SUDS techniques out of the seven for that particular site. Outcomes from both single and dual matrixes were then assessed to conclude the most suitable SUDS technique to be implemented.

3.2.2.6. Prevalence Rating Approach (PRAST)

A range of different appropriate SUDS techniques suitable for implementation was identifies for all sites. This included both singular and dual techniques. The Prevalence Rating Approach for SUDS Techniques (PRAST) was applied to determine the most suited technique to be implemented on a particular site. This was achieved by rating all the SUDS options on a scale by considering their particular attributes (Martin, 2000; Wilson *et al.*, 2004). Each site therefore had given an ultimate SUDS recommendation based on the prevailing SUDS option from the PRAST scale.

The following PRAST Scale was used in the 'Final SUDS Output' sheet of the DST and began with the most desirable SUDS options. The values appended with the letter 'B' are the alternating options for a particular technique and should not be thought of as the sole. The following order is in conjunction with the order used in the DST:

1) Dry swales and wetland

2A) Dry swales and basin / pond

2B) Lined swales and lined basin / pond

3) Dry swales and infiltration basin

4) Infiltration trench and infiltration basin

5) Wetland

6A) Basin / pond

6B) Lined basin pond

7) Infiltration Basin

8) Wet swales

9A) Dry swales

9B) Lined dry swales

10) Green roof and soakaway

11) Green roof

12) Infiltration trench

13) Soakaway

14) Filter strip

15) Pervious pavement and belowground storage

16) Pervious Pavement

17) Belowground storage

18) Water Playground.

3.2.2.7. Acceptance Warning

The final stage in the DST was to focus on public acceptance. The output can be seen in the 'Final Best SUDS Solution' sheet of the DST. Safety is one of the most important issues when considering the public acceptance of SUDS, primarily in relation to children (McKissock *et al*, 1999). Hence, any site whit a neighbouring a playground or a school closed to the proposed SUDS feature in the immediate vicinity received one of two cautionary flags. An orange cautionary flag indicates that some minor safety actions were required for the SUDS feature to be acceptable. A red warning flag required some major safety actions to be developed around the SUDS feature. Sites receiving a green flag indicate that no safety issues were identified and standard SUDS safety actions were most likely to be sufficient for the proposed SUDS features.

3.2.3. Fieldwork Activities

All 103 sites identified in Edinburgh were visited several times. No site or premises was considered if recognised as private property or required permission for entry. Sites were assessed by walkover study with site data being recorded on proforma sheets.

Soil samples were taken at locations where major SUDS features were likely to be implemented. Samples were taken at 10cm depth intervals within trenches to be dug up to 100 cm in depth. If a sampling location was not acceptable (e.g. below tarmac or a house), an alternative representative sampling location was determined up to 5m (if not stated otherwise) away from the original location. Soil samples were stored in sealed plastic bags prior to analysis to preserve moisture content.

3.2.4. Laboratory Analyses

The determination of particle size distribution was performed according to British Standards (British Standard Institute, 1999b). Soil samples were dried out for 24 hours prior to analyses. Standard laboratory safety measures were observed.

3.3. Assessing Stormwater Detention Systems Treating Road Runoff with an Artificial Neural Network

3.3.1. Data Set

The sampling of data was done simultaneously for all systems. However, the number of samples is sometimes different between inflow and outflow for the same variable because outliers and human error have been identified at a later stage during data analysis. Consequently, values identified as flawed have been removed from the data set. It follows that correct data that directly correspond to all removed entries were also removed during further analysis and modelling to obtain an overall data set that only contains matching pairs. All tested variables were $\log_{10}$-transformed to achieve normality for subsequent statistical tests if required.

3.3.2. Modelling

In the last few decades, artificial neural network (ANN) modeling approaches have been numerously applied in the area of water quality modeling, where they proved to be particularly successful in predictions based upon complex, inter-related, and often non-linear relationships between multiple parameters (Brion and Lingireddy, 2003). In their research, Sandhu and Finch (1996) indicate that ANN models have been more successful in estimating river salinity than other simulation and commonly used statistical models. However, there are difficulties involved with applying models for microbial water quality predictions; mostly as a result of complexities in environmental distribution; mobility and fate of microbes. Microbial contaminants are known to be non-conservative, unevenly distributed and their numbers and growth rates may change in the environment depending on the conditions they live in. The inter-relationship and interactions between microbial colonies in stormwater cause various modeling challenges that have been overcome for particular case studies by applications of ANN to multi-parameter data sets (Brion and Lingireddy, 2003).

Artificial neural networks are mathematical modelling tools that are applicable in the field of prediction, and forecasting in complex settings. They are relatively good tools for interpolation in the range of observed conditions, but can be very poor in prediction and forecasting, especially in case of overtraining (Scholz, 2006). Fundamentally, they operate through simulating, at a simplified layer, the activity of the human brain. The network fulfils this through a vast number of highly interconnected processing elements (called nodes in this paper), working in accord to solve specific problems, including forecasting, and pattern recognition. In an ANN, each node is connected to other neighbouring nodes with different coefficients or weights, which represent the relative influence of the varying node inputs to other nodes (Hamed *et al.,* 2004).

Each neuron in a network has a scalar bias b, the bias is similar to a weight except that it has a constant input of 1. The transfer function net input n in the ANN is also a scalar and is equal to the sum of the weighted input wp and the bias b. This sum is the argument of the transfer function f. A transfer function can be a step function or a sigmoid function, which takes the argument n and produces the output a. Both w and b are adjustable scalar parameters of the neuron. The main concern in ANN is the adjustability of such parameters so that the network would be able to reveal most desired and interesting behavioural patterns. A neuron with a single scalar input and a scalar bias b appears in Fig 13.

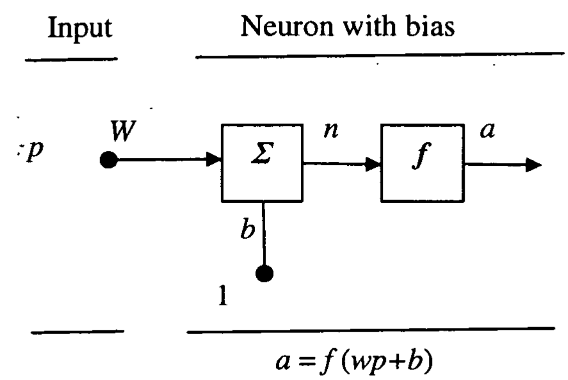


Fig 13. A neuron with a single scalar input and a scalar bias; *p* is the scalar input, *w* is the scalar weight, *wp* is the scalar product, *f* is the transfer function, which produces the scalar output, *a* is the scalar output, *b* is the scalar bias, and *n* is the transfer function net input.

Artificial neural networks vary in type. A basic example of a neural network is given in Fig 14, containing one input, one hidden, and one output layer; they are all connected without any feedback connections. The weighted sum of the inputs are transferred to the hidden nodes, where it is transformed using an output function (also called transfer or activation function). In return, the outputs of the hidden nodes perform as inputs to the output node where another transformation happens. Network outputs often have associated processing functions; these functions are used to transform user-provided target vectors for network use. Network outputs are then reverse-processed using the same function to produce output data with the same characteristics as the original user-provided targets. A typical processing function for the output of the hidden layer is the output function given in Equation (9).

$$x_i = \sigma_i\left(b_i + \sum_{j=1}^{n} w_{ij} u_j\right) \tag{9}$$

where x_i is the output from the hidden node, σ_i is the output function of the hidden node (usually the hyperbolic tangent *tanh*), b_i is the bias input to the hidden node *i*, *n* is the number of input nodes, w_{ij} is the weight connecting the input node *j* to the hidden node *i*, and u_j is the input node *j* (Sarle, 2002).

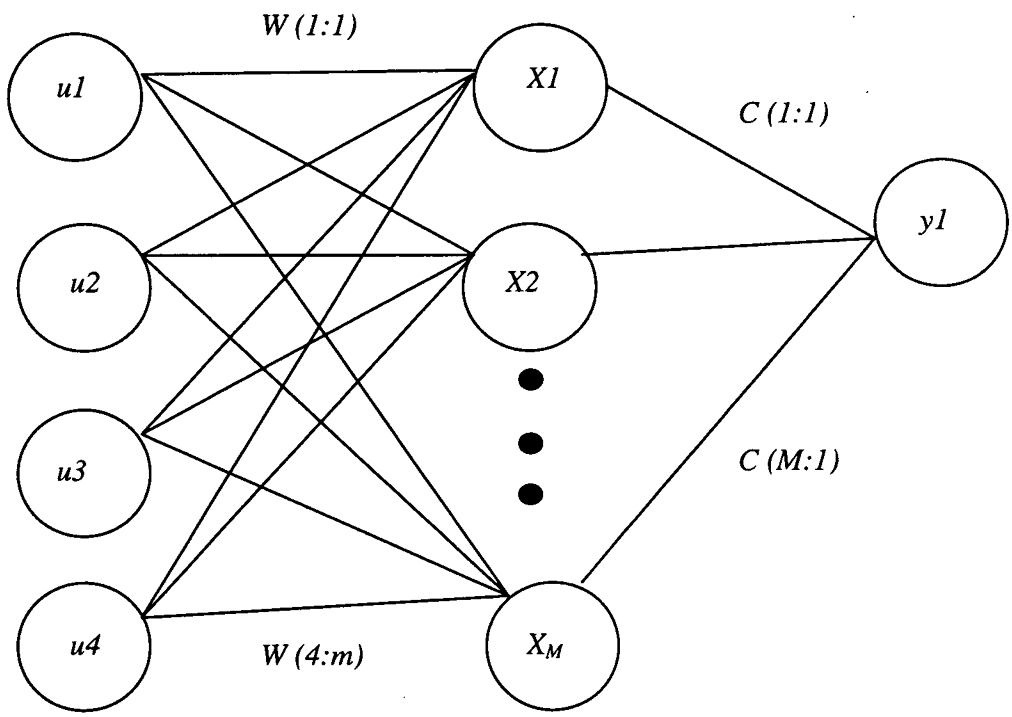


Fig 14. Neural network architecture (M=8 for intestinal enterococci and M=64 for total coliforms); *u* is the input *layer*, *x* is the hidden *layer*, *y* is the output node, *W* is the weight *matrix* connecting the input node to the hidden node, and *C* is the weight *matrix* connecting the hidden node to the output node.

A representation of the hidden node *i* is given in Fig 15 Moreover, the typical processing function for the output of the network can be expressed in Equation (10).

$$y_i = \sum_{j=1}^{m} c_{ij} x_j \tag{10}$$

Where y_i is the output from the output node i, m is the number of hidden nodes, c_{ij} is the weight connecting the hidden node, and x_j is the weighted sum of inputs into the hidden node j to the output node i.

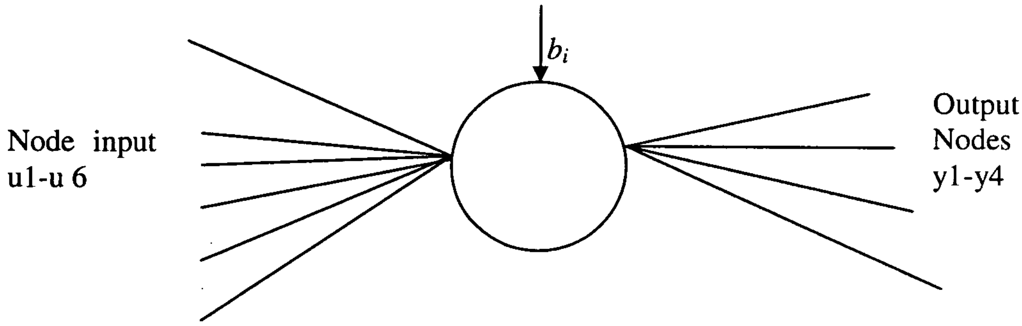


Fig 15. Schematic representation of the hidden node i; bi is the bias term, and σ is the output function of the hidden node (usually the hyperbolic tangent tanh).

During network training, the connection weights, and biases of the ANN are adapted through a continuous process of simulation. The primary training goal is to minimize an error function by searching for connection strengths, and biases that make the ANN produce outputs that are equal or close to the targets. Equation (11) expresses the mean square error (MSE) of the output values.

$$MSE = \sum_{t=1}^{N} \left(Y_t - \hat{Y}_t\right)^2 / N \tag{11}$$

Where MSE is the mean square error, N is the number of data points, Y_t is the observed output value, and $\hat{Y}_t$ is the output of a feed-forward neural network.

The minimization procedure consists in the optimization of a non-linear objective function. A number of optimization routines can be applied. Practically, the Levenberg-Marquardt routine is often used as it finds better optima for various problems than the other optimization methods (Sarle, 2002).

3.3.3. Development of the Artificial Neural Network Model

In this study, one of the most commonly used types of ANN was used: the feed-forward network, where the information is transmitted in a forward direction only. According to Tomenko *et al.,* (2007), feed forward ANN models were found one of the most efficient and robust tools in predicting constructed treatment wetland performance if compared to traditional models. For example, Neelakantan *et al.,* (2001) have developed a simple feed-forward back propagation ANN model, which was successful in predicting *Cryptosporidium* and *Giardia* populations with a number of other biological, chemical and physical variables in the Delaware River.

A multi-layer, feed-forward ANN usually contains one input, one output, and one hidden layer. Different numbers of hidden nods, and various output functions were tested during the model development. Although, at present, no specific standards exist for the selection of the number of hidden nods, there are various guidelines proposed in literature (Rogers and Dowla, 1994; Maier and Dandy, 1998). Six model architectures were applied for each set of input parameters. The number of applied hidden nods was 2^k, with k varying from 1 to 6. The optimum number of hidden nods was 8 for the prediction of intestinal enterococci colony forming units, and 64 for the prediction of total coliform colony forming units. The Levenberg-Marquardt optimization method was applied for all models. The MATLAB neural network tool box (version 5.3) was used.

The counts of total coliforms and intestinal enterococci per 100 ml in outflow samples collected from 2005-04-14 to 2006-09-15, ranged between 300 and 7100, and between 300 and 2010, respectively. The corresponding inflow counts were between 550 and 8420, and between 360 and 2130, respectively. Table 8 summarizes statistics for total coliforms, and intestinal enterococci. European legislation sets a mandatory water quality standard requiring that total coliforms, and faecal streptococci should not exceed 10,000 cfu/mL, and 2000 cfu/mL for 95% of the water samples, respectively.

Table 8. Summary statistics for total coliform, and intestinal enterococci counts for the entire dataset (2005-04-15 to 2006-09-13) comprising the inflow and outflows for systems 1 to 5.

Statistics	Total coliforms (n=61)						Intestinal enterococci (n=63)					
	Inflow	1	2	3	4	5	Inflow	1	2	3	4	5
Maximum	8420	5280	7100	6130	5530	3500	2130	1900	2010	1990	1950	1400
Minimum	550	320	390	280	300	300	360	300	300	300	300	300
Mean	3801	1807	2776	2489	1204	939	1140	793	966	923	878	668
Standard deviation	2742	1363	2169	1870	1079	748	506	382	473	454	419	272

All units are cfu/ml; n, number of samples.

A certain number of relevant inputs should exist to achieve a successful determination of the relationships amongst the input variables, and the model output. When utilizing equations for chemical, biological or physical processes in a model, the specifications of the processes determine the required input parameters. The selection of inputs is not determined in ANN; therefore, inputs can be selected on the basis of intuitive or empirical understanding of the processes. However, advanced systematic analytical techniques such as principal component analysis or sensitivity analysis can be used when selecting input parameters (Maier and Dandy, 1996; Zhang *et al.*, 1998).

When compared with multiple regression analyses, where a p value indicates the significance of a variable, and its suitability for inclusion in a model, ANN provide no standard statistical measure to determine the significance of an input variable. Consequently, the input variables (turbidity, pH, conductivity and dissolved oxygen) selected in this study were chosen on this basis and of the information gathered from previous literature.

The dataset comprised 60 observed data per parameter per system, and was divided into testing, validation, and training data sub-sets. The training set contained 65% of the entries for the entire dataset (i.e. 39 observations), whereas the validation, and testing sets consisted of 15% (9 observations), and 20% (12 observations) of the entire dataset, respectively. Fig 16 schematically indicates a series of steps that have been conducted during the model development process (Hamed *et al.,* 2004).

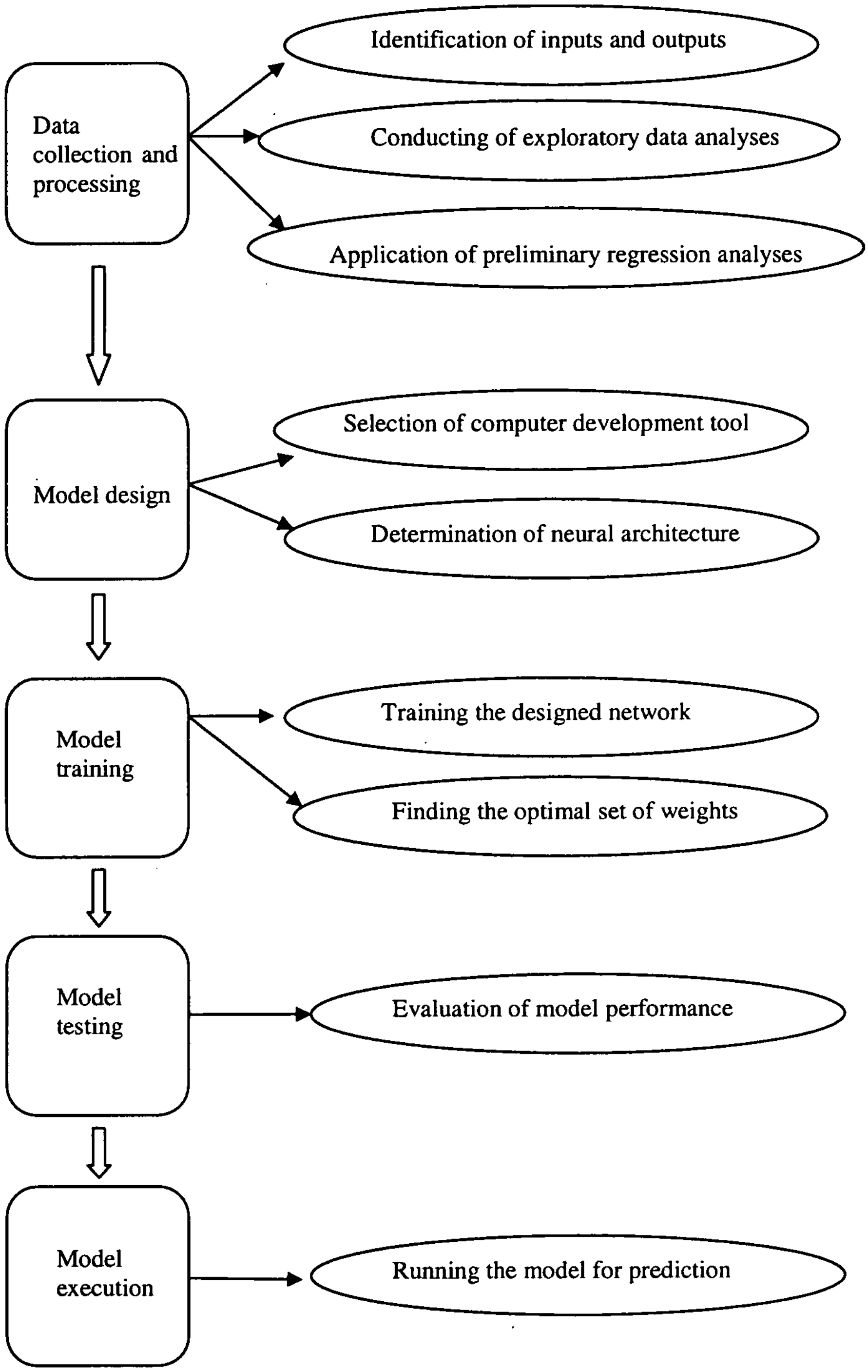


Fig 16. Steps of the model development process.

3.4. Stormwater Infiltration Systems for Road Runoff Contaminated with Organic Matter Including Dog Faeces

3.4.1. Sampling Procedure

A scoping survey was undertaken to assess the approximate amount of dog faeces contamination per square meter of urban pavement areas within the City of Edinburgh. Water quality monitoring was performed approximately three times per week (i.e. twice weekly for chemical and nutrient analysis, and once per week for microbiological tests).

Sampling took place at four locations: the inflow and outflow points of the detention tank, and the centres of the planted and unplanted ponds. The silt trap was dry unless there was a large rainfall event shortly before or during sampling.

3.4.2. Analytical Laboratory Works

All analytical procedures were performed according to the Standard Methods for the Examination of Water and Wastewater (APHA, 1998). Water samples were tested immediately for temperature, five-day at 20°C N-Allylthiourea biochemical oxygen demand (BOD), suspended solids (SS), total solids (TS), conductivity, turbidity, dissolved oxygen (DO), pH and the oxidationreduction potential (redox). The BOD (mg/L) concentrations were determined with the OxiTop IS 12-6 system

(Wissenschaftlich-Technische Werkstätten, Weilheim, Germany). A Whatman PHA 230 bench-top pH meter, a Hanna HI 9142 portable waterproof dissolved oxygen (DO) meter, a HACH 2100N turbidity meter and a Mettler Toledo MPC 227 conductivity meter were used to determine pH, DO (% and mg/L), turbidity (NTU) and conductivity (μS), respectively (Scholz, 2006). A Hanna HI 98201 ORP meter with a platinum tip electrode was used to measure the redox potential.

Nutrients were determined by automated precision colorimetry methods using a Palintest Photometer 5000 instrument. The nitrate test gave concentrations for nitrate (NO_3) and nitrite (NO_2). The ammonia test measured total ammonia (NH_4) concentrations. Phosphorus was tested in terms of total phosphate (PO_4) concentrations, using a method specific for low concentrations of phosphate in water (Scholz, 2006).

Concerning microbiological examinations, the spread plate method was used. Each water sample was diluted six times. For each dilution, a 100 μL sample was spread on three different types of agar: nutrient agar, Slanetz and Bartley agar, and MacConkey No. 3 agar (Atlas, 1995). The tests were replicated three times for verification purposes. The agars were prepared according to the manufacturers' instructions, autoclaved and then poured onto sterile Petri dishes. Once all (except for controls) Petri dished had been contaminated with the various sample dilutions, they were placed into the incubator for 48 hours at 25°C. The results of the plate counts are expressed as colony forming units (CFU). A 'valid' plate count contained between 30 and 300 CFU per plate (Atlas 1993, 1995). During sampling, the height of water was recorded manually at all sampling points, the air temperature was measured and rainfall was determined from a Snowdon rain gauge at the study site. The rain gauge comprised a measurement cylinder fed through a funnel with a diameter of 12.7 cm (Scholz, 2006). Samples were immediately tested after sampling for nutrients (ammonia, nitrate, nitrite and phosphate) and microbiological indicator organisms.

3.5. Combined Bio-filtration, Stormwater Detention and Infiltration System Treating Road Runoff

3.5.1. Sampling Procedure

Water was abstracted from the system at all sampling points (i.e. if water present) twice per week. An intense sampling period began on 16th October 2007 and ended on 14th March 2008. Sampling water could occasionally not be obtained from some points within the tank due to either low water levels or ponding above the sampling point opening. The water depth was estimated by inserting a measurement rod through the sampling wells to the bottom of each sampling pipe, and subsequently reading off the water level. This was undertaken for each point during all sampling occasions.

3.5.2. Analytical Laboratory Works

The air and water temperatures and the oxygen concentrations within the water samples were measured immediately on site using a WTW Oxi315i meter. The water samples were then transported to the campus-based laboratory for further analyses. Electrical conductivity, pH and total dissolved solids were measured using a portable Hanna 991300 meter. The turbidity was recorded using a Hach-Lange 2100 turbidity meter. The five day biochemical oxygen demand (BOD) was determined under the influence of a nitrification inhibitor with the OxiTop manometric measuring system manufactured by the Wissenschaftlich-Technische Werkstätten GmbH, Weilheim, Germany. Suspended solids concentrations were measured using Whatman glass microfiber filters with pore diameters of 200 μm. Firstly, these filters were weighted before use. Suspended solids were calculated using 500 mL of sample water, which was passed through the filter papers. The wet filters were then placed in an oven at 104°C for 48 h, left to cool and then reweighed. Total solids were determined using 100 mL of water, which was placed in glass beakers. The beakers were then dried in an oven at 104°C for 48 h, allowed to cool, and then reweighed. If not stated otherwise, all other variables including ammonia-nitrogen, nitrate-nitrogen and ortho-phosphate-phosphorus were determined according to American standard methods (APHA, 1995).

Precipitation data were provided by the School of GeoSciences, and were obtained from the University weather station on The King's Buildings campus. Precipitation was recorded at minutely, hourly and daily time steps. Data from the Blackford Hill climate station in Edinburgh were used for estimating evapotranspiration using a method described by Muller (1996).

3.5.3. Modelling with the SWMM

Hydraulic flows were modelled with the United States Environmental Protection Agency program stormwater Management Model (SWMM, Version 5.0; Rossman, 2005). The model was calibrated with water level data measured on site. Calibration was carried out by comparing the real water levels in the infiltration tank with the predictions made by the model. The calibration parameters were adjusted until a good fit was obtained. An earlier version of the model was successfully calibrated and verified by Fabritius (2007) using the parameters summarized in Table 9. These parameters were based on physical measurements whenever possible, or otherwise on estimations using values obtained from the literature.

Table 9. Parameters originally assigned to subcatchments

Properties	Unit	Car_Park	Gravel_Filter
General			
Area	m^2	640	7
Width	m	13	1
Slope	%	1.00	0.00
Outlet	-	Gravel_Filter	Detention_Tank
Impervious area	%	100	0.00
Manning's roughness coefficient (pervious area)	-	0.012	-
Depression storage (impervious area)	mm	2.5	-
Depression storage (pervious area)	mm	-	5.0
Impervious area with zero depression storage	%	0	0
Infiltration			
Maximum infiltration rate	mm/h	-	120
Minimum infiltration rate	mm/h	-	60
Decay constant	1/h	-	1
Drying time	Days	-	3
Maximum volume	mm	-	N/A
Groundwater			
Aquifer name	-	-	Gravel_Filter
Receiving node	-	-	Detention_Tank
Surface elevation	m	-	1.3
Groundwater flow coefficient	$m^{2.5}/(s.ha)$	-	63
Groundwater flow exponent	-	-	0.5
Surface water flow coefficient	Variable	-	0
Surface water flow exponent	-	-	0
Interaction coefficient	-	-	0
Fixed surface water depth	m	-	Variable
Threshold groundwater elevation	m	-	0.925

A continuous long-term simulation was undertaken. The system's response was modelled from representative sample data obtained between 16/10/07 (10:00) and 14/03/08 (12:00). Details of the time steps used for various aspects of the simulation are given in Table 10.

Table 10. SWMM time

steps.Aspect	Time Step	
Reporting	60	minutes
Dry weather runoff	30	minutes
Wet weather runoff	1	minutes
Time step for routing	60	second(s)

Model calibration was carried out through a trial and error approach with the aim of achieving an acceptable and realistic match between modelled and measured water levels in the tank. An evaluation of the root-mean-square values between observed and modelled water levels was used for comparison purposes. As the model had previously been calibrated for the period between June and October 2006 (Fabritius, 2007), some parameters changed only slightly. The parameters, which have changed considerably, are listed in Table 11.

Table 11. SWMM calibration parameters.

Parameter	Unit	Initial Value
Porosity	-	0.25
Conductivity	mm/h	40
Conductivity slope	mm/h	10
Tension slope	mm	15
Lower groundwater loss	mm/h	1
Water table elevation	m	0.95
Unsaturated zone moisture	-	0.1
Groundwater flow coefficient	$m^{2.5}/(s.ha)$	63
Groundwater flow exponent	-	0.5

The infiltration function was recalculated based on water levels obtained from the period between 16/10/07 and 14/3/08. Changes in water level within the tank were analysed for periods of dry weather during which no inflow to the tank occurred.

The SWMM was used to generate the data required to analyse the hydraulic efficiency of the system. Parameters used in the calculation of the hydraulic efficiency were lag time, reduction of peak flow and the benefit factor.

Chapter 4

Results and Discussion

4.1. The Glasgow Sustainable Urban Drainage System Management Project

4.1.1. SUDS and Soil Quality

Fig 4 shows the outline of the SUDS decision support key and the classification of all sites visited during the exploratory stage of this project. The key may be used in combination with Table 4 outlining a SUDS decision support matrix (as presented in Scholz *et al.*, 2006).

This matrix has been tested with the exploratory data set collected during the site visits, and Table 12 summarizes the outcome of the application of this tool. The findings for SUDS structures in Table 12 are based on the assumption that the soil contamination issues for all sites have been identified during the planning phase, and that contaminated soil will be removed wherever relevant soil contamination guidelines and/or the introduction of unlined SUDS structures require such a measure.

Table 12. Sustainable urban drainage system (SUDS) options based on the SUDS decision support matrix.

Area	Catchment	Wetland	Pond	Infiltration basin	Swale	Infiltration trench	Soakaway	Filter strip	Permeable pavement	Underground storage	Water playground
Belvidere Hospital	Entire area		XXX	X	XX	X			XX	X	XX
Celtic FC Stadium	Entire area					X	X		XX	XXX	
Cowlairs Park	North		XXX	X	XX	X	X	X	XX	X	XX
	South		XXX		XX		X	XX	XX	X	XX
Gadburn	North	X	XXX		XX	X			XX	X	XX
	South	XXX	XX		XX				XX	X	XX
Lillyburn Place	Entire area		X			XXX	X		X		X
Pollok Centre	West				XX	X			XX	XXX	
	East		X		XXX	XX	X		XX	X	X
Ruchill Hospital and Park	North-east				XX				XX	XXX	
	South-east				XX				XX	XXX	
	South		XXX		XX				XX	X	XX
	West		XXX		XX	X	X	X	XX	X	XX

X = possible option; XX = recommended option; XXX = predominant SUDS design feature.

Concerning nutrients and heavy metals, Table 13 summarizes the soil quality for the most important nutrients and metals at 10 cm depth within Glasgow. Table 13 allows the reader to compare the contamination for selected demonstration sites with the mean contamination for the whole of Glasgow. Moreover, Table 14 shows the major nutrients and heavy metals at different soil depths during the exploratory investigation.

Table 13. Major total nutrients and major heavy metals (mg/kg dry weight): comparison of soil quality at 10 cm depth during the exploratory investigation of 57 sites.

Area	Site	N	P	K	Al	Cr	Cu	Fe	Mn	Ni	Pb	Zn
Belvidere	North	1951	771	7601	13173	56	8	18100	1169	23	478	147
Hospital	South	351	391	5492	6688	94	10	29724	301	17	35	52
Celtic FC	North	1991	815	2290	7874	78	30	30783	528	12	103	135
stadium	East	724	615	3496	10695	112	88	29566	570	444	346	370
Cowlairs	North	1476	384	3817	4033	908	47	19053	526	29	59	92
Park	South	625	841	8963	15998	23	10	35441	476	33	46	71
Gadburn	South	2283	554	3522	9330	74	31	21312	339	27	85	85
Lillyburn Place	West	124	213	1607	4839	66	14	24883	374	19	51	45
Pollok Centre	West	524	290	9260	13156	147	72	28847	548	114	145	121
Ruchill	Northeast	1840	568	7813	3125	13	30	22065	416	27	170	123
Hospital	East	554	280	3243	2765	7	22	15809	483	17	374	217
and	Centre-east	2308	467	3393	3653	15	41	21629	283	23	451	231
Park	South	505	308	4315	9131	77	34	24688	594	25	1307	434
	Northwest	2412	716	3884	11507	78	37	23606	311	30	194	158
	Park	1663	575	7025	4515	21	33	33096	504	35	298	135
All sites for all areas		1612	605	4562	12538	96	72	27375	485	34	198	180

Table 14. Major total nutrients and major heavy metals (mg/kg dry weight): comparison of soil quality at different depths during the exploratory investigation.

Area	Depth (cm)	Count	N Mean	N SD	P Mean	P SD	Pb Mean	Pb SD	Zn Mean	Zn SD
Belvidere	10	2	731	813.7	423	241.5	145	222.4	69	53.2
Hospital	20	2	801	781.7	397	251.5	157	215.7	80	53.1
	30	2	933	716.4	450	248.2	222	200.8	110	60.4
	40	1	418	-	402	-	117	-	57	-
	50	1	453	-	316	-	78	-	77	-
Celtic FC	10	2	1357	896.2	715	141.3	225	171.8	253	166.3
Stadium	20	2	1160	626.5	666	299.6	239	28.3	281	20.2
	30	2	1373	471.3	1072	188.3	809	157.3	692	158.9
Cowlairs Park	10	2	1102	412.2	653	301.5	55	15.4	82	11.8
	20	2	1779	1066.8	621	281.1	89	63.5	104	36.5
	30	2	1786	1058.8	459	174.1	106	67.6	120	44.1
Gadburn	10	1	2283	-	554	-	85	-	85	-
	20	1	9937	-	919	-	236	-	141	-
	30	1	5156	-	944	-	187	-	86	-
	40	1	5916	-	1101	-	181	-	121	-
	50	1	3740	-	1026	-	369	-	437	-
Lillyburn Place	10	1	124	-	213	-	51	-	45	-
	20	1	156	-	581	-	41	-	97	-
	30	1	219	-	629	-	55	-	90	-
	40	1	213	-	761	-	85	-	91	-
Pollok Centre	10	1	534	-	290	-	145	-	121	
	20	1	688	-	262	-	153	-	91	
	30	1	381	-	178	-	14	-	95	
	40	1	336	-	254	-	145	-	91	
Ruchill	10	6	1547	836.3	486	168.4	466	425.5	216	115.2
Hospital	20	6	1330	1559.6	505	400.0	194	126.8	155	98.0
And	30	6	1372							

The individual contamination profiles can be compared with the average contamination profiles for Glasgow (Table, 14). Table 15 shows major nutrients and selected heavy metals for soil at a depth of 50 cm for all areas that would be occupied by SUDS structures. Contamination level variations at a depth relevant for conveyance structures such as swales are shown to give the reader an indication of the potential remediation work to be undertaken for unlined SUDS structures to avoid leaching out of nutrients and metals from the soil into the runoff (Table, 15).

Table 15. Major total nutrients and major heavy metals (mg/kg dry weight): comparison of soil quality at a depth of 50 cm for all areas that would be occupied by SUDS structures. Sampling sites have been chosen based on proximity to nodes on a 50 m × 50 m grid.

Area	Count	N		P		Pb		Zn	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD
Belvidere Hospital	23	816	331.1	532	220.1	505	1012.4	244	307.6
Celtic FC Stadium	8	1572	343.7	665	154.5	651	651.1	439	168.9
Cowlairs Park	31	733	348.9	453	234.4	107	57.4	98	57.8
Gadburn	22	2458	2748.0	596	339.5	124	114.1	146	119.0
Lillyburn Place	8	708	320.4	604	189.8	96	35.0	90	31.5
Ruchill Hospital and Park	33	815	364.2	500	721.8	262	252.7	138	119.5

SD=standard deviation.

Concerning organic contaminants, Fig 17 shows an example gas chromatograph result for a representative demonstration area (Belvidere Hospital). The largest peak observed was an artefact of the extraction procedure, and showed up in the method blank as well as all the samples.

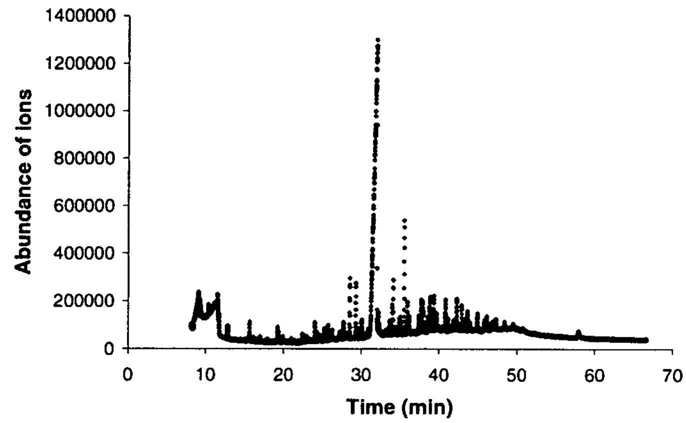


Fig 17. Belvidere Hospital area (proposed pond location): gas chromatograph findings associated with organic contamination of soil at 50 cm depth on 5 July 2004.

4.1.2. Case studies

A planimeter investigation has shown that the horizontal area of the Belvidere Hospital area available for the integration of SUDS techniques would be 94,000 m^2 (Figs, 11 and 12).

Fig 9 shows a photograph of the site for which a major SUDS feature is planned. In comparison, Fig 10 shows an artist impression of this site after regeneration.

The proposed SUDS design for the Belvidere Hospital area is shown in Figs 11 and 12. Fig 17 shows an example of a total ion chromatogram. Fig 18 shows a photograph of the site for which a major SUDS feature is planned.

Fig 18. Celtic FC Stadium area: site photograph taken on 14 May 2004.

The proposed SUDS design for the Celtic FC Stadium area is shown in Figs 19 and 20. A planimeter investigation has shown that the horizontal area of the Celtic FC Stadium area (excluding Celtic Park) available for the integration of SUDS techniques would be 58,500 m^2.

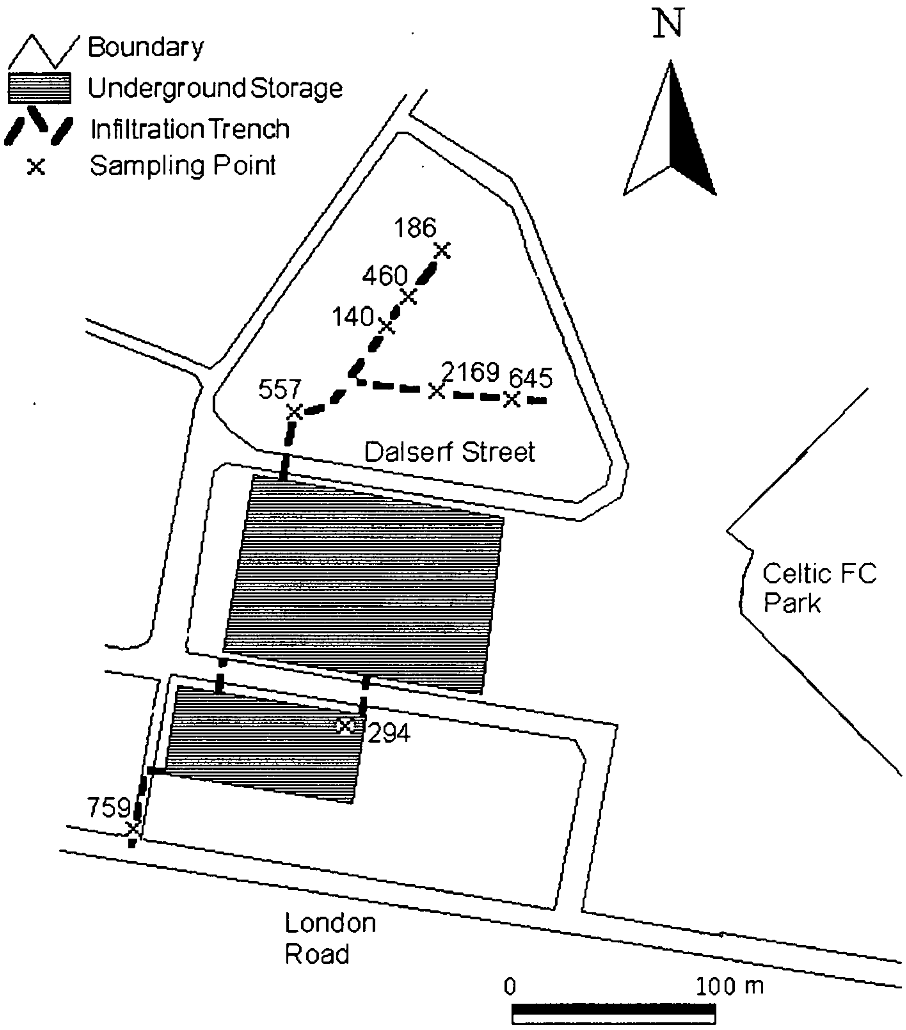


Fig 19. Celtic FC Stadium area: spatial distribution of lead (mg/kg dry weight) at 50 cm depth on 21 July 2004.

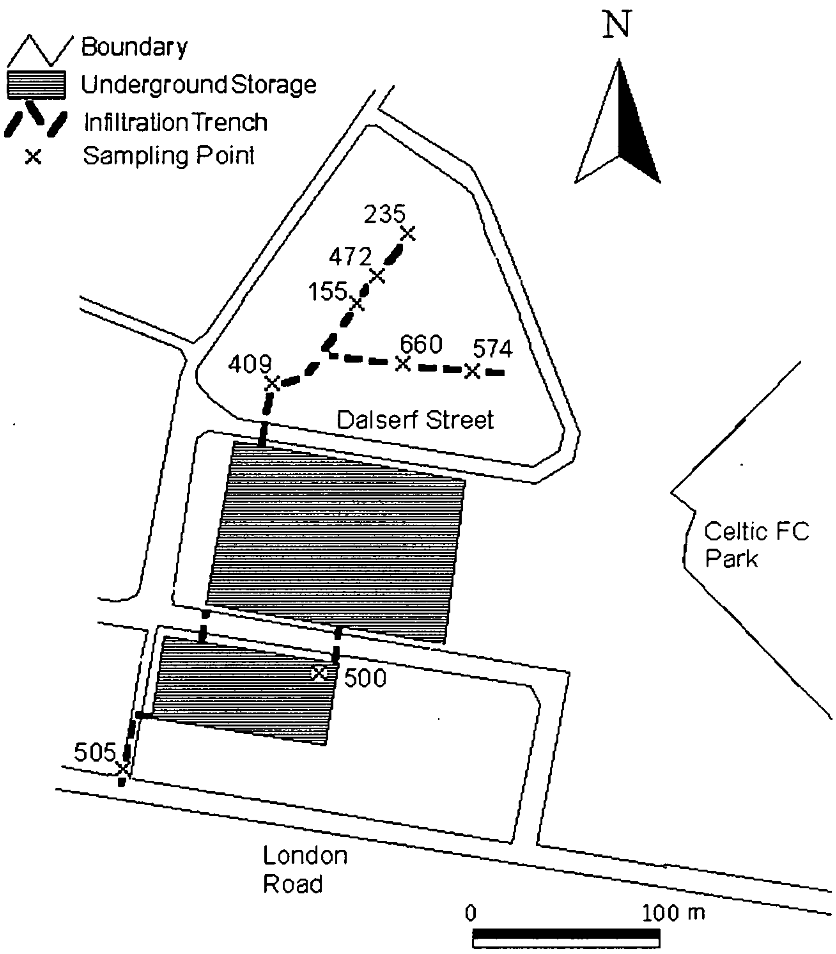


Fig 20. Celtic FC Stadium area: spatial distribution of zinc (mg/kg dry weight) at 50 cm depth on 21 July 2004.

4.1.3. Definitions for Proposed SUDS Techniques

The abbreviation SUDS is an acronym for Sustainable Urban Drainage System or also known as Best Management Practice (BMP) in the USA (Scholz, 2006). For the purpose of the case studies, a SUDS is defined as either an individual or a series of management structures and associated processes designed to drain surface runoff in a sustainable approach to predominantly alleviate capacities in existing conventional drainage systems (predominantly combined sewers in Glasgow) in an urban environment (SEPA, 2003; Butler and Davis, 2000; CIRIA, 2000 and Scholz, 2006).

The pond proposed for the Belvidere Hospital area is a depression structure that increases the duration of the flow hydrograph with a consequent reduction in peak flow, with the depression having a minimum depth of water present at all times, and an overflow outlet to the river. The pond can be used for attenuation, detention, retention, storage, infiltration, and recreational purposes (Guo, 2001 and Scholz, 2003, 2004). As the pond matures, it may become heavily vegetated, and could be classified as a wetland with the potential to enhance the ecological habitat (Scholz and Trepel, 2004 and Scholz, 2006).

The proposed network of swales at the Belvidere Hospital comprises grass-lined conveyance structures (approximately 5 m in width) designed to infiltrate but predominantly to transport runoff from the site, while controlling the flow and quality of the surface water. The swales convey water to a river via a pond. The contaminated soil will have to be removed to avoid the leaching of metals into the runoff (Scholz, 2006).

The proposed infiltration trenches in the Celtic FC Stadium area are linear drains (also known as French Drains). An infiltration trench consists of a trench filled with a permeable material and with a perforated pipe at designated depth to promote infiltration of surface runoff to the ground. Some of the infiltration trenches will also convey water; if their gradient is sufficiently steep (Scholz, 2006).

Belowground stormwater storage tanks have been proposed for the Celtic FC Stadium car parking areas. These subsurface structures are designed to accumulate surface runoff, and release it subsequently, as may be required to increase the flow hydrograph, if there is no risk of flooding. Moreover, the structure may contain aggregates or plastic boxes (e.g., Matrix Geo-Cell detention system promoted by Atlantis Water Management Ltd.) and can act also as a water recycler or infiltration device (Scholz, 2006).

4.1.4. Relevant Soil Contamination Guidance

The soil contaminants summarized in Tables 13 to 15 should be seen in context with soil contamination guidelines (Society of the Chemical Industry, 1979; Ministry of Housing, Spatial Planning and Environment, 2000 and Environment Agency, 2002). The guidelines specify thresholds for heavy metals such as chromium, copper, manganese, nickel, lead, and zinc.

Concerning chromium, the threshold for residential properties with and without plant uptake is 130 and 200 mg/kg dry weight, respectively. In contrast, the threshold for commercial and industrial land is 5,000 mg/kg dry weight (Environment Agency, 2002). In comparison, the Dutch intervention concentration is 380 mg/kg dry weight (Ministry of Housing, Spatial Planning and Environment, 2000) while the Kelly threshold (i.e., Kelly Indices Guidelines for Contaminated Soils; specifically developed for gasworks sites in London) is 200 mg/kg dry weight (Society of the

Chemical Industry, 1979). However, chromium is not a major concern for both selected case study areas.

Concerning copper, the Dutch intervention concentration is 190 mg/kg dry weight (Ministry of Housing, Spatial Planning and Environment, 2000) while the Kelly threshold is 200 mg/kg dry weight (Society of the Chemical Industry, 1979).

Concerning manganese, the Kelly threshold is 1,000 mg/kg dry weight (Society of the Chemical Industry, 1979). Nevertheless, neither copper nor manganese is a particular concern for both selected case study areas.

Concerning nickel, the threshold for residential properties with and without plant uptake is 50 and 75 mg/kg dry weight, respectively. In contrast, the threshold for commercial and industrial land is 5,000 mg/kg dry weight (Environment Agency, 2002). In comparison, the Dutch intervention concentration is 210 mg/kg dry weight (Ministry of Housing, Spatial Planning and Environment, 2000), while the Kelly threshold is 50 mg/kg dry weight. However, the latter concentration is correct for available nickel only (Society of the Chemical Industry, 1979). Except for the East of the Celtic FC Stadium area, nickel is not a problem for both case studies.

Concerning lead, the threshold for residential properties (with and without plant uptake) is 450 mg/kg dry weight. In contrast, the threshold for commercial and industrial land is 750 mg/kg dry weight (Environment Agency, 2002). In comparison, the Dutch intervention concentration is 530 mg/kg dry weight (Ministry of Housing, Spatial Planning and Environment, 2000) while the Kelly threshold is 1,000 mg/kg dry weight (Society of the Chemical Industry, 1979). Lead is a major problem for both case study areas. Depending on further ground investigations, it is, however, likely that large quantities of top soil need to be removed on both sides before new residential developments can be built.

Concerning zinc, the Dutch intervention concentration is 720 mg/kg dry weight (Ministry of Housing, Spatial Planning and Environment, 2000) while the Kelly threshold is 500 mg/kg dry weight (Society of the Chemical Industry, 1979). Zinc is potentially a major problem for both case study areas. It is likely that top soil in some parts of both selected demonstration sites needs to be removed before new residential developments can be built.

4.1.5. Cost–benefit Analysis

Pollution and runoff volume justifies the (potentially additional) costs of SUDS. Concerning both selected case studies, planning permission will only be given if the developers can demonstrate that no additional runoff will impact on the existing sewer system during storm event. It follows that either SUDS or a more traditional drainage solution in form of a detention system (e.g., large below ground storage tank) has to be considered during the planning phase.

Concerning the Belvidere Hospital area, the capital costs for both systems is likely to be similar (approximately £700k) as shown in previous studies in Scotland (Broad and Barbarito, 2004). A traditional solution would provide more space for housing while a SUDS solution has the additional benefit of enhancing the ecological value of a landscape and reduce environmental pollutants.

However, unless the SUDS can be integrated into the proportion of green space that is usually reserved for recreational purposes (i.e., approximately 10% of a new site), the traditional system is likely to be marginally less expensive. On the other hand, the maintenance costs of SUDS are usually lower (approximately by 30%) than for conventional systems (Butler and Davis, 2000; Broad and Barbarito, 2004).

Concerning the Celtic FC Stadium area, the proposed SUDS solution (i.e., predominantly belowground storage) is virtually the same as a traditional subsurface detention tank. Estimated capital costs are approximately £500k. Therefore, the maintenance will also be virtually identical.

Retrofitting of a detention system can easily be justified with flood prevention measures considering that this part of Glasgow is subject to frequent and regular devastating floods. The gained sewer storage space can subsequently justify the regeneration of neighboring estates where currently most flats are empty.

A detailed cost–benefit analysis comparing SUDS with traditional drainage systems or even comparing different SUDS treatment trains with each other is beyond the scope of this paper. Moreover, the planning phase has not progressed sufficiently to enable a calculation to be based on detailed information.

4.1.6. Belvidere Hospital Area Design Proposal

Hospital area is not yet finalized, as planning permission has yet to be sought by Kier Homes (former owner: National Health Trust). However, medium density residential properties are assumed to dominate a future landscape, and thus all SUDS recommendations have been made with regard to this assumption (Figs, 11 and 12). The main entrance to the Belvidere Hospital area is located approximately in the middle of the northeast face of the area adjacent to London Road. Two large vegetated areas flank the main driveway, which runs from the main entrance in a southwesterly direction. The driveway separates two fields suitable for housing: The first field to the northwest of the main driveway is overgrown with some small trees and shrubs. The dimensions of this site are approximately 150 ×150 m. The second field, which is located to the southwest of the main driveway, is also overgrown but contains residual asphalted car parking and building foundations throughout. The area is approximately 150 ×450 m in size. Both areas are mainly level apart from some depressions towards the southwest and south of the site (Figs, 11 and 12).

A central depression exists to the southeast of the remaining building. This area has therefore been identified as an ideal location for the implementation of a detention and attenuation pond and an associated outlet swale (or culvert) structure, and it is therefore recommended that no building construction work should be undertaken in this part of the Belvidere Hospital area. Moreover, the residual foundations in the centre of the Belvidere Hospital area appear to be at ground floor level with an existing basement level beneath in a depression. Excavation of these structures would form a suitable depression for a detention pond, which would provide a sufficient attenuation period for surface runoff.

The best engineering option recommended is to have essentially two interconnected networks of swales throughout both fields allowing suitable spacing for a medium-dense housing development. Also, it is recommended that the existing building's guttering should be redirected into a swale, which should be connected with the inlet structure of the detention pond that serves also the combined network of both swale systems (Figs, 11 and 12).

The detention pond area should include space for decorative embankment planting, seating areas, and a footpath circling the pond and woodland to create a high amenity value by providing interesting landscaping features to the local community (Figs, 10–12).

From this detention pond a further cascading swale, acting as a combined overflow and outlet structure should flow down the embankment through the existing glade of mature tress to the public river walkway (Figs, 11and 12). It is recognized that a swale may be difficult to construct due to the established vegetation, and therefore, a cascade of small ponds (interconnected with a culvert) or an open channel (lined with decorative brick) or even a subsurface pipe may be more suitable for some stretches pending on an outstanding ecological habitat assessment. A suitable provision should be made to allow the overflow to flow under or across the walkway by means of guttering into the River Clyde.

Transport structures such as feeder roads and car parks should be constructed from permeable or pervious pavement. A short culvert below the main driveway (connecting London Road with the former hospital building), which is expected to be retained, from the swale network in the north to the detention pond in the centre of the area should be considered (Figs, 11 and 12).

Contaminants such as manganese, lead (Fig, 11) and to a lesser extend zinc (Fig, 12) are present in high concentrations in the soil (Tables, 13–15). The soil in the center southwest of the area is heavily contaminated with lead and zinc and would require removal. However, lead in particular is very difficult to dissolve in water, and would not cause a problem for the outflow concentration of most SUDS structures (Scholz *et al.*, 2006).

The concentration of organic compounds found was low (estimated to be less than 1 mg/kg). Compounds found included polycyclic aromatic hydrocarbons (PAH). For example, pyrene, fluoranthene, chrysene, benz(a)- anthracene, benzo(k)fluoranthene, benz(a)pyrene, and benzo(g,h,i)perylene were found at very low concentrations at Belvidere Hospital (proposed pond location).

Other compounds found included aliphatic hydrocarbons such as tetracosane, eicosane, heptadecane, heptacosane, and nonadecane which are commonly constituents of diesel–type fuels. Phenol derivatives and carbolic acid related compounds were also found. These types of compounds were often used as cleaning agents and disinfectants in hospitals and schools (Fig, 17).

4.1.7. Celtic FC Stadium Area Design Proposal

The areas surrounding the Celtic Park stadium (Fig, 18) are currently under development and regeneration. Celtic FC was granted planning consent in 1994 for the redevelopment of the stadium to from an all-seated stadium with a capacity for 60,000 spectators. As part of this planning application, the club was required to provide 377 car-parking spaces within the curtilage of the stadium. This has been achieved to the satisfaction of Glasgow City Council.

In 1998, the club was granted planning consent for the formation of a temporary coach park on the site of a former bakery in the Camlachie to the West of the stadium. This consent allowed for the parking of 171 coaches, and was granted for a period of 3 years until June 2001. Renewal of this consent was granted in July 2001, for a further period of 3 years. This facility is used for coaches of home supporters. The catchment area excluding the stadium is about 58,500 m^2.

Considering the current state of the Celtic FC car park and its heavy use during match days, this area would be ideal for an integrated belowground storage system underneath the present car park. The suggested area in the West for the integration of belowground storage facilities is the site surrounded by Dalserf Street in the north and Barrowfield Street in the south. The storage area would be approximately 14,600 m^2. A further but smaller belowground storage area of 4,900 m2 could be located in the southeast of the main storage tank just south of Barrowfield Street (Figs, 19 and 20).

According to a recent site investigation and the current characteristics of the area in the north, the construction of a simple infiltration trench network with two branches seems to be feasible. The branches of the infiltration trench network should be located in the north and northeast, respectively. The land in the north is associated with the highest ground level in the study area. The infiltration trench network will transfer the water from the roofs and paved surfaces to the major belowground storage through an inlet in the northwest of the main storage tank (Figs, 19 and 20).

Infiltration trenches or culverts should connect the two storage tanks and transfer the runoff to the smaller storage tank and when required to the sewer system located on London Road. The overflow of the storage tank system is located in the southwest of the study area (Figs, 19 and 20).

Considering the current state of the Celtic FC car park in the west, renovation work is likely to be required within a couple of years. The implementation of the recommended SUDS would therefore be easily approachable. The area is contaminated with lead (Fig, 19) and zinc (Fig, 20) that might be linked to pollution from cars (Tables, 13–15). The soil requires removal, if used by residents in the future. However, particularly lead is very difficult to dissolve in water (Scholz *et al.*, 2006), and is unlikely to cause a problem for the outflow concentration of belowground SUDS structures if pH levels are high and conductivity values low (Scholz and Trepel, 2004; Scholz, 2006).

The overall concentration of organic compounds found was low (estimated to be less than 0.5 mg/kg). Compounds found included PAH. For example, pyrene, fluoranthene, chrysene, benz(a)anthracene, benzo(k)fluoranthene, benz(a)pyrene, and benzo(g,h,i)perylene were found at very low levels in the Celtic FC Stadium area (at the proposed network of swales). Other compounds found included aliphatic hydrocarbons such as tetracosane, eicosane, heptadecane, heptacosane, and nonadecane, which are commonly constituents of diesel–type fuels.

4.2. The Edinburgh Sustainable Urban Drainage System Management Project

4.2.1. Edinburgh Sites Data

The data collected for the 103 Edinburgh Sites is summarised in Table 16. 6% of the 103 sites identified in Edinburgh found to be contaminated were regeneration sites or deserted areas classified as development sites. Evidence of demolition activities observed on some sites in the form of rubble or machinery tracks. Sources of contamination included asbestos in the rubble and hydrocarbon dumping on the ground. No soil samples were collected for laboratory purposes during the site visits as SUDS recommendations were only sought form the collected data and not SUDS proposals. The percentage contamination was viewed low considering that 38% of the sites studied were classified as regeneration.

Table 16. A summary of the 103 Edinburgh sites representing the current situation.

Characteristic	Category				
Contamination	Yes: 6	No: 94			
Possible SUDS area (%)	≤1: 47	1<x≤5: 28	5<x≤10: 10	10<x≤50: 9	>50: 6
Land values	Low: 33	Low-Medium: 19	Medium: 35	Medium-High: 9	High: 4
Runoff Quantity	1: 39	2: 26	3: 25	4: 4	5: 6
Roads	Motorways: 0	Primary road: 12	A road: 18	B road: 3	Tertiary: 67
Drainage type	Sewer Only: 60	Watercourse: 40			
Runoff Quality	Good: 65	Average: 30	Poor: 5		
Groundwater	High: 27	Low: 65	Currently not determined: 8		
Soil infiltration	High: 65	Low: 27	Currently not determined: 8		
Current Impermeable Surface (%)	≤20: 53	20<x≤40: 14	40<x≤60: 7	60<x≤80: 8	>80: 18
Future Impermeable Surface (%)	≤20: 7	20<x≤40: 14	40<x≤60: 42	60<x≤80: 25	>80: 12
Catchment size (m^2)	≤50000: 45	50000<x≤100000: 30	100000<x≤200000: 13	200000<x≤300000: 9	x>300000: 3
Slope (x in 100m)	≤1: 21	1<x≤5: 53	5<x≤10: 20	10<x≤20: 4	x>20: 2
Ownership	Council only: 39	Private only: 35	Council / Private: 26		
Ecological Impact	Yes: 26	No: 74			
Acceptance Warning	Green Flag: 80	Orange Flag: 17	Red Flag: 3		
Site classification	Development: 21	Regeneration: 38	Retrofitting: 17	Retrofit with parks: 24	

[a]Higher classified roads take precedence over any lower classified roads and score the associated bin entry.

Retrofitting sites account for 17% of the 103 sites with a further 24% being retrofitted using recreational areas sites. Thus, 41% of the sites studied were considered as retrofit sites. This mirrored on the nature of Edinburgh as a city where most developments are taking place on the outskirts with regeneration projects being restricted to brownfield sites within the city and the waterfront (Lothian Councils, 2004). Therefore, the study was focused within the Edinburgh City boundaries; a high percentage of retrofit sites were acceptable.

Most of the sites investigated in Edinburgh were small in size. 45% of those had catchments less than 50,000 m^2 with 75% having catchment areas smaller than 100,000 m^2. Given the small size of these sites, a catchment wide program of SUDS implementation on numerous sites was needed to take place prior to benefits in terms of peak flow reduction in watercourses.

The land values were a reflection of the location of the SUDS sites. Retrofit sites were predominantly located in affluent areas and as a result received a land value classification of medium or higher. The lower value sites were generally regeneration sites located in areas of Edinburgh featuring in a wider city council redevelopment programme. These areas included Leith and areas of in the north of the city such as Muirhouse and Pilton (City of Edinburgh, 2001).

The ownerships of sites were relatively even with 39% of sites being owned by the council and 35% privately owned. Nevertheless, 26% of the sites were classified as being council or privately owned. This figure had occurred due to some retrofit site catchments taking runoff form council and private properties. The relative ownership varied on these retrofit sites and therefore the council/private classification was considered to be the simplest method of documentation.

There were no motorways in or around Edinburgh. The city by-pass was classified as a primary road or dual-carriageway and linked into the M8 motorway beyond the city boundaries.

4.2.2. SUDS Decision Support Tool

4.2.2.1. The process

Table 5 shows the runoff coefficients attributed to each site. The probable future use of the site is determined from its surroundings; i.e. if the site is surrounded by industrial buildings then it is given a coefficient of 0.8, if a site is surrounded by low density housing it is given a coefficient of 0.4. Engineering judgement was used to determine the values of these coefficients, based on the unit percentage area covered by the runoff sources.

If a site was surrounded by several different runoff sources, then the DST took the largest runoff coefficient identified for that site. The chosen coefficient was multiplied by the horizontal surface area of the site to give an impermeable area for the site. This impermeable area value was then used to determine a runoff value for the site.

Runoff values attributed to each site were ranging from one to five based on the calculated impermeable surface area. The threshold values for the impermeable areas are outlined in Table 17 and the threshold values of the numbered categories are shown on Figure 21.

Table 17. Impermeable area threshold values.

Runoff Values:	Value (m^2)
Runoff Value 1	0<x<19123
Runoff Value 2	19123<x<38058
Runoff Value 3	38058<x<89262
Runoff Value 4	89262<x<140467
Runoff Value 5	x>140467

The threshold values were not determined by creating five equal sized categories between the maximum and minimum impermeable. A frequency distribution curve (shown in Fig, 21) was drawn do identify the typical impermeable area sizes for the Edinburgh sites.

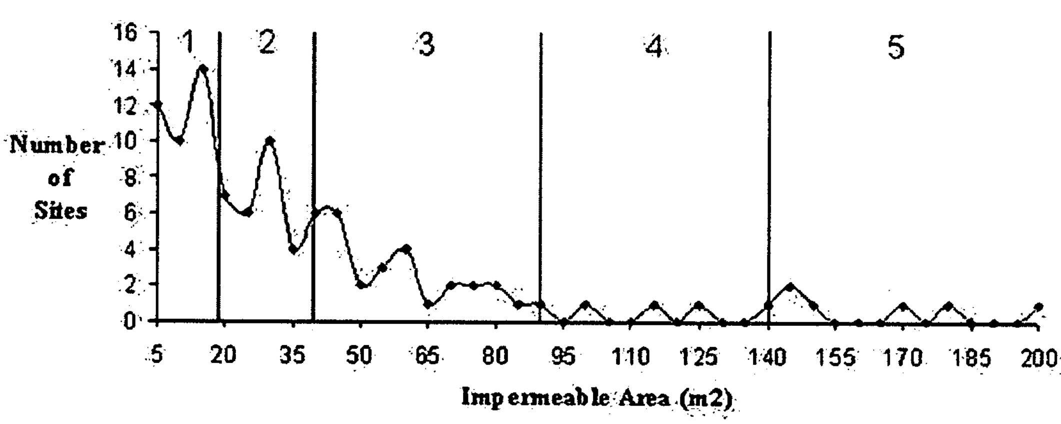


Fig 21. Runoff analysis frequency distribution.

This curve was not shaped as a standard or normal distribution curve but showed a larger distribution of smaller impermeable area values, which was representative of the 103 sites. Table 16 indicates that 75% of the catchments studied are less than 100,000m^2. This proved that most Edinburgh sites were have been assigned a runoff value of one or two had equal sized categories been used in the model.

The process outlined in Section 3.2.2.1 in Chapter 3 was therefore adopted as the preferred method for runoff classification. This method gave a fair distribution of impermeable area values into five categories and a better indication of the runoff quantities of the sites investigated. A margin of 20% was selected as five different runoff categories were to be defined. A fifth of the calculated maximum impermeable area value was subtracted from itself to give the highest threshold. This margin could be altered if the number of runoff categories changed. The other thresholds were calculated from this point.

4.2.2.2. Data Arrangement and Output

The DST arranged all data fields into a list as seen in Section 3.2.2.2. of Chapter 3. All data fields in this list had output characteristic variables defined for them from site information. The decision support tool then compared these characteristic variables with SUDS variable limits set in the matrices (Tables, 6 and 7). If all characteristic variables matched the variable limits set for each SUDS technique then an output of 'Yes' was given. This indicated that a particular SUDS technique was suited to a site. If any characteristic did not match the matrix variables then an output of 'No' was yielded, meaning that this particular technique was unsuitable for implementation on site.

4.2.2.3. Decision Support Tool Application

The established method of SUDS identification was applicable to all sites, including sites which existed outside Edinburgh. It was proposed to use Edinburgh as a template for the development of a decision support tool for the implementation of SUDS by utilising the various features of Edinburgh City. The detailed characteristic site data determined by proforma sheets and site studies allowed accurate definition of characteristic variables for each site. This accuracy was reflected by the decision support tool output where acceptable SUDS recommendations were made for all 103 Edinburgh sites.

4.2.3. SUDS Variables and Matrixes Analysis

4.2.3.1. Single SUDS Decision Support Matrix

4.2.3.1.1. Single SUDS Matrix Variables

The single SUDS decision support matrix (DSM) (Table, 6) had a set of characteristic variable limits which was established from design criteria (Nuttall *et al.*, 1998; Martin, 2000; Pratt *et al.*, 2001; Wilson *et al.*, 2004). The variables related to the maximum or minimum data field limits which specific site data needed to meet in order for a SUDS feature to be appropriate. The operation '>' specified that the characteristic site variable had to be greater than limit set in the matrix for the relative SUDS feature to be suitable. Similarly, '<' indicated that the characteristic site variable had to be smaller than the limit set in the matrix for the relative SUDS feature to be considered suitable.

Considering the matrix (Table, 6), a wetland received a final runoff value of $\geq$3. This value was set considering the water balance within the wetland. It is important that wetlands do not dry out (Wilson *et al.*, 2004) and therefore runoff quantities from sources classified as greater than or equal to three were believed to be sufficient for a permanent water level to exist. This runoff value for wetlands related to the minimum catchment area required (65,000 m^2) to provide such runoff quantities.

Most constructed wetlands for stormwater treatment were surface flow wetlands comprising of shallow water areas with general depth of 0.3 – 0.6 m (Wilson *et al.*, 2004). A time of 16-24 hours was also recommended for water treatment performance (Nuttall *et al.*, 1998). The volume of wetlands is equal to the retention time and the inflow volume of water (Martin, 2000). Therefore, where a catchment area of greater than 65,000m^2 (as specified in the matrix) drains to the wetland, the surface area of the wetland should be sufficiently large to afford the necessary volume. The minimum area required for wetland implementation was therefore set at 1000 m^2 in the matrix.

Any urban area greater than 1000 m^2 in size is likely to have several owners. For wetland implementation, the number of owners is set at <3. Fragmented ownership of property carries a variation of interests and would therefore have certain implications as to the possibility of implementing a wetland.

The process discussed for the identification of matrix variable limits for wetlands was used for all other SUDS features. Design criteria set for all SUDS techniques (Wilson *et al.*, 2004) employed by the DST was taken in account when setting up matrix variable limits.

A classification of not acceptable 'NA' is given when data fields were irrelevant to a specific SUDS feature. When design criteria for soakaways were considered data fields such as land cost, fragmentation of ownership, ground slope and ecological impact were less relevant. Soakaways are small below ground structures and consequently the critical variables were ground contamination, volume of runoff and groundwater level. The mentioned variables were defined in the matrix with the less relevant variables classified as 'NA'.

4.2.3.1.2. Single SUDS Decision Support Matrix Output

Table 18 illustrates the DST results employing the singular SUDS decision support matrix (Table, 6). One site ('Edmonstone' EDSUDS075) believed to be suited to wetlands held a combination of desired features including a large catchment area together with low land value, high water quality and a single owner. The mentioned factors satisfied the characteristic variables set in the single SUDS decision support matrix for wetlands. No other sites had the required characteristics and therefore no further wetlands were recommended. This mirrored on the nature of the 103

Edinburgh sites where smaller sites dominated and therefore other SUDS features such as soakaways were better suited.

Table 18. Single Sustainable Urban Drainage System (SUDS) decision support matrix.

SUDS Techniques:	Number of sites:	% of Sites:
Wetland	1	1%
Basin / Pond	8	8%
Lined Basin / Pond	1	1%
Infiltration Basin	34	33%
Wet Swales	9	9%
Dry Swales	19	18%
Lined Swales	3	3%
Green Roof	49	48%
Infiltration Trench	19	18%
Soakaways	22	21%
Filter Strips	3	3%
Pervious Pavement	97	94%
belowground Storage	38	37%
Water Playground	25	24%
SUDS Not Suitable	1	1%

Pervious pavements could be implemented on 97 of the sites (Table, 18). 6 sites were identified as being contaminated (Table, 16) and thus were considered unsuitable for pervious pavements. The single specified variable for pervious pavement in the single SUDS decision support matrix was ground contamination. This condition was included as allowing infiltration of water into contaminated ground might speed up the leakage and spread of contaminants (Pratt *et al.*, 2001). The DST identifies whether the SUDS feature could be implemented on the site according to the obtained site data. Retrofit site data might satisfy the matrix variables, but in reality pervious pavement could not be appropriate for some sites because the existing impermeable surfaces may not be changed due to planning or public acceptance issues. The DST was not developed with the intention of proposing SUDS features for particular sites. The purpose of the tool was to offer decision support for the user, with engineering and planning judgement being employed to establish the feasibility of the SUDS features.

A momentous number of sites (38) were suitable for belowground storage (Table, 18). Belowground storage systems could be implemented where other land use was proposed (e.g. car park) and surface SUDS features like ponds and swales were not considered suitable. This was shown in the single SUDS decision support matrix where land cost estimation was required to be higher than two (>2). It is reasonable that if an area of a site had an intention, then it would have a higher value. For that reason, the most appropriate SUDS feature to be integrated was belowground storage other matrix criteria of relevance was runoff quality and ground contamination. Subsurface chamber systems could recharge the groundwater. And then it could damage the surrounding soil if low quality runoff containing pollutants (e.g. hydrocarbons from large roads) was allowed to infiltrate. This was also the case for ground contamination where infiltrating water could take contaminants to the water table (Waltham, 2002).

If contamination was proven to exist on a site, then lined swales and lined ponds could be specified. Peffermill Industrial Estate was the only site believed to be appropriate for lined ponds (Tables, 18 and 19) with three out of the six contaminated sites being suited to lined swales. This indicated a clear trend where the DST found lined swales and lined ponds appropriate for contaminated sites.

Table 19. Edinburgh SUDS demonstration site data.

Classification Fields:	Edinburgh Southern Harrier	Braidburn Valley Park	Edinburgh Park	Inchview Primary School	North Fort Street	Peffermill Industrial Estate	Redhall House Drive
Area Reference Number (from decision support tool)	006	013	026	044	051	064	094
Site Classification	Retro / Rec	Retro / Rec	Dev	Reg	Reg	Dev	Reg
General Site Acceptability: (Y/N)	Y	Y	Y	Y	Y	Y	Y
Final Runoff Value: 1-5 (Min-Max)	3	5	4	1	1	2	2
Scaled Catchment Area Size: (m^2)	108,300	389,200	110,000	7,500	1,600	45,800	68,300
Max Area Suitable for SUDS Feature: (m^2)	200	150	2,500	100	100	400	40
Serious Contamination: (Y/N)	N	N	N	N	N	Y	N
Land Cost Estimation Value: 1-5 (Min-Max)	3	4	2	1	3	2	3
Number of Owners	50	100	1	1	1	1	1
High GWL: (Y/N)	N	N	Y	Y	Y	Y	N
High Ground Infiltration: (Y/N)	Y	Y	N	Y	Y	Y	Y
Slope for SUDS Transfer Structures	5.3	9.0	1.0	1.0	2.9	0.6	5.5
Runoff Quality Value: 1-3 (Good-Poor)	2	2	3	1	1	1	1
Potential for High Ecological Impact: (Y/N)	N	Y	Y	N	N	N	Y
Drainage to Watercourse: (Y/N)	N	Y	Y	N	N	Y	Y
Drainage to Sewer Only: (Y/N)	Y	N	N	Y	Y	N	N
Drainage to Sewer Or Watercourse: (Y/N)	Y	Y	Y	Y	Y	Y	Y

Retro / Rec: Retrofit with utilisation of designated park land
Dev: Development
Reg: Regeneration

49 sites were found to be appropriate for implementation of green roofs. This high number of suitable sites could be attributed to there being only three single SUDS

decision support matrix variables to satisfy. The runoff variable was set at less than three in relation to structural issues (BS EN 12056-3:2000) where there were limits to the area of roof which could safely house this SUDS feature. The number of owners should also be small as certain premises may have several different occupants with varying SUDS acceptances (McKissock *et al.*, 1999). Ultimately, green roofs could be drained to a sewer or watercourse.

Table 16 illustrates that 59% of the sites studied in Edinburgh were development and regeneration sites and therefore it could be verified that most of these sites were suitable for implementation of the green roofs. However, retrofit sites were unlikely to feature green roofs because of the structural and the financial implications caused by changing existing structures.

One site was found to be inappropriate for SUDS. No site data could match the criteria set in the single SUDS decision support matrix.

4.2.3.1.3. Combined Singular SUDS

The combined singular SUDS were developed to give an underlying principle for the development of SUDS drainage systems at a variety of scales. It was a hierarchy which was customised to suit the size and the complexity of the area being drained. A major SUDS site would have a range of integrated surface water drainage components. Retention ponds and wetlands were the main treatment and detention facilities whereas basins, swales and infiltration systems were the major forms of runoff control. Source controls like green roofs, soakaways and pervious pavement could also form combinations.

The DST functioned by checking the outputs for the individual SUDS types forming the combinations. If the outcome from the single SUDS decision support matrix signified that both techniques were suitable, then a combination featuring these types was also considered possible.

Table 20 illustrates the outcome of combining singular SUDS techniques. No sites were suitable for combination of dry swales with a wetland and no combination of dry swales with basin/pond was possible. This was due to the variables set in the single SUDS decision support matrix. The variables set for wetlands and basins/ponds contradicted the variables set for swales, meaning that when one technique of the combination was suitable for a particular site, the other technique was not. For instance, a wetland required a runoff value greater than or equal to three (≥ 3), while a swale required a runoff value of less than three (<3). Equally, the runoff value for ponds was classified as 'not applicable' while dry swales required a runoff value of less than three (<3). This discrepancy yielded no result from the DST.

Table 20. Combining singular SUDS techniques results.

SUDS Techniques:	Number of sites:	% of Sites:
Dry Swales & Wetland	0	0%
Dry Swales & Basin / Pond	0	0%
Lined Swales & Lined Basin / Pond	1	1%
Dry Swales & Infiltration Basin	9	9%
Infiltration Trenches & Infiltration Basin	15	15%
Green Roof & Soakaway	17	17%
Pervious Pavement & belowground Storage	38	37%
Combination SUDS Not Suitable	38	39%

The DST effectively required a 'Yes', 'Yes' output from the singular SUDS matrix for it to classify a combination as possible. This could explain the reason for which 38 of the sites investigated had no combination SUDS possible. One site was classified as being suitable for a lined pond by the single support matrix and this site featured again as the combination of lined swales with lined basin/pond ('Peffermill Industrial Estate' EDSUDS 064).

4.2.3.2. Dual SUDS Decision Support Matrix

4.2.3.2.1. Dual SUDS Matrix Variables

A dual SUDS matrix (Table, 7) was developed to facilitate the identification of appropriate SUDS combinations for the Edinburgh sites. The variables contained within the matrix took the same character as the single SUDS decision support matrix, i.e. '>' meaning higher than and '<' meaning smaller than the specified variable. Values for some data fields were changed to accommodate two SUDS types. The area necessary for SUDS features was increased since each combination could occupy a larger area of a site. The additional water loss probable from having a number of SUDS features on a site was accounted for by increasing the minimum catchment sizes for all combinations. Features like ponds and wetlands require permanent volumes of water which was justified by increasing the minimum catchment size (Wilson *et al.*, 2004).

The important difference in the dual SUDS decision support matrix was that all combinations had single variables assigned to them. As a result, no contradiction between the variables of the singular SUDS techniques forming the combination was possible. These variables were employed in the DST in the same way as the single SUDS decision support matrix to identify appropriate SUDS combinations for all sites.

4.2.3.2.2. Dual SUDS Decision Support Matrix Output

Table 21 demonstrates the DST results employing the dual SUDS decision support matrix (Table, 7). One site was found appropriate for a wetland and dry swale; the same site had also been identified by the single SUDS decision support matrix as being appropriate for wetlands ('Edmonstone' EDSUDS075). This site had adequately large characteristics for dry swales to be implemented in conjunction with wetland, which would then decrease the amount of synthetic drainage materials otherwise required ahead of the development of the site.

Table 21. Dual SUDS technique decision support matrix results.

SUDS Techniques:	Number of sites:	% of Sites:
Dry Swales & Wetland	1	1%
Dry Swales & Basin / Pond	9	9%
Lined Swales & Lined Basin / Pond	4	4%
Dry Swales & Infiltration Basin	16	16%
Infiltration Trenches & Infiltration Basin	16	16%
Green Roof & Soakaway	32	31%
Pervious Pavement & Underground Storage	38	37%
Dual SUDS Not Suitable	20	19%

40 sites were recommended for the green roof and soakaway combination. This related to the variables set in the dual SUDS matrix where a small area and high ground infiltration was requisite by the combination. Table 16 illustrates that 65% of the Edinburgh sites had high infiltration. Edinburgh was also dominated by small areas which were only appropriate for minor SUDS features. The green roofs and soakaway combination could be implemented in these sites with the only factors lessening their suitability for all sites being runoff (<3) and contamination.

4.2.3.2.3. Analysis of combining singular SUDS and dual SUDS Decision Support Matrix Outcomes

The results of combining singular SUDS techniques could be compared with the dual SUDS technique outcomes. Table 22 is a coupled version of Tables 20 and 21 for a simpler comparison.

Table 22. Analysis of combination SUDS and dual SUDS matrix results.

	Combining Singular SUDS		Dual SUDS Support Matrix		
SUDS Techniques:	Number of sites:	% of Sites:	Number of sites:	% of Sites:	Change +/-
Dry Swales & Wetland	0	0%	1	1%	1
Dry Swales & Basin / Pond	1	1%	9	9%	8
Lined Swales & Lined Basin / Pond	1	1%	4	4%	3
Dry Swales & Infiltration Basin	9	9%	16	16%	7
Infiltration Trenches & Infiltration Basin	15	15%	16	16%	1
Green Roof & Soakaway	17	17%	32	31%	15
Pervious Pavement & belowground Storage	38	37%	38	37%	0
SUDS Not Suitable	38	37%	20	19%	-18

The dual SUDS decision support matrix was introduced to improve on the outcomes realised from combining singular SUDS techniques. Table 22 shows positive changes with most combinations becoming more suitable using the dual SUDS decision support matrix.

A large change was observed for green roof and soakaways with 31% of sites suited by the dual SUDS matrix. This related in the combination having single variables in the dual matrix. The DST thus compared the characteristic variables of the sites with the matrix variables to reach a 'Yes' or 'No' output.

The combination of green roofs and soakaways as singular SUDS structures was only found suitable for 17% of the sites. This lower value occurred due to the variables for the two techniques being different in the single SUDS decision support matrix. In cases where the outcomes for the two techniques using the single SUDS decision support matrix are different, then a combination based on these outcomes is not possible.

4.2.3.3. Prevalence Rating Approach for SUDS Techniques (PRAST)

4.2.3.3.1. Development of PRAST

SUDS structures vary in terms of their runoff management and pollutant removal abilities. Development, regeneration and retrofitting sites have diverse land uses in which SUDS structures must attribute to control runoff and treat pollutants. Guidance is available for design and monitor the hydrological performance of SUDS structures (Martin, 2000; Wilson *et al.*, 2004).

Limited knowledge is available for developers as to the suitability of SUDS structures to particular functions and so the type of SUDS techniques that may be appropriate for various situations is identified by D'Arcy and Wild (2003). These suggestions are general and do not preclude a reasoned case being made for different SUDS solutions to suit individual local circumstances. Generally, SUDS suggestions are made for the following types of sites:

Industrial estates - local site controls such as swales or detention basins, with retention ponds or stormwater wetlands as regional (estate) facilities.

Housing - filter drains / swales / soakaways where possible, or extended detention basins with wetland base.

Roads - swales/filter drains, plus extended detention basins (larger ponds or wetlands may be appropriate for flood control purposes).

Therefore, the Prevalence Rating Approach for SUDS Techniques (PRAST) had been developed, where all individual and combination SUDS options were rated on a scale by their particular attributes. The primary rating shown in Table 23 was common for the 103 Edinburgh sites and represents the order of preference for implementation for all SUDS options. Every site investigated had some SUDS features appropriate for implementation and the PRAST scale was used by the DST to identify the most appropriate of the SUDS structures.

Table 23. Principal SUDS rating for all SUDS options.

Position	SUDS Technique	Primary SUDS Rating
1	Dry Swales & Wetland	100
2A	Dry Swales & Basin / Pond	95
2B	Lined Swales & Lined Basin / Pond	95
3	Dry Swales & Infiltration Basin	92
4	Infiltration Trenches & Infiltration Basin	85
5	Wetland	80
6A	Basin / Pond	75
6B	Lined Basin / Pond	75
7	Infiltration Basin	72
8	Wet Swales	60
9A	Dry Swales	55
9B	Lined Dry Swales	55
10	Green Roof & Soakaway	54
11	Green Roof	50
12	Infiltration Trench	45
13	Soakaways	40
14	Filter Strips	35
15	Pervious Pavement & belowground Storage	25
16	Pervious Pavement	15
17	Belowground Storage	10
18	Water Playground	5

The rating was established with consideration of all data fields investigated for each of the Edinburgh sites. Single or combined SUDS options were rated on their capability for runoff control and pollutant removal. Factors also affecting the rating of a SUDS option were; their ability to function in a variety of ground conditions, the cumulative impacts that they would have on a development, the impact that any discharge would have on receiving watercourses or CSO spill frequencies, and the chances for habitat and amenity enhancement. Dual SUDS structures were found to suit these factors best.

4.2.3.3.2. PRAST Outcomes

PRAST results for the Edinburgh sites are shown in Table 24 showing the percentage of sites suited to each SUDS option. Combination SUDS were rated highest on the PRAST scale with the dry swales and infiltration basins being the most appropriate option for 16 sites. Infiltration trenches as singular structures could be implemented on a further 18 sites with green roofs and soakaways being the optimum option for 17 sites. Green roofs as singular structures were suited to 11 sites. The SUDS features identified by the DST were the SUDS systems which were best suitable to the Edinburgh sites. The city was dominated by small areas appropriate for SUDS and the techniques chosen by PRAST could be implemented in small blocks of land to treat and control runoff. This compared to the work carried out in the city of Malmö Sweden, where green roofs were found to be the best SUDS option (Villarreal *et al.*, 2004).

Table 24. Decision support tool results using the Prevalence Rating Approach for SUDS Techniques (PRAST) for most feasible SUDS identification.

PRAST Number:	SUDS Techniques:	Number of sites	% of Sites
1	Dry Swales & Wetland	1	1%
2A	Dry Swales & Basin / Pond	8	8%
2B	Lined Swales & Lined Basin / Pond	4	4%
3	Dry Swales & Infiltration Basin	16	16%
4	Infiltration Trenches & Infiltration Basin	0	0%
5	Wetland	0	0%
6A	Basin / Pond	3	3%
6B	Lined Basin / Pond	0	0%
7	Infiltration Basin	18	17%
8	Wet Swales	3	3%
9A	Dry Swales	4	4%
9B	Lined Dry Swales	0	0%
10	Green Roof & Soakaway	17	17%
11	Green Roof	11	11%
12	Infiltration Trench	0	0%
13	Soakaways	4	4%
14	Filter Strips	0	0%
15	Pervious Pavement & Belowground Storage	9	9%
16	Pervious Pavement	5	5%
17	Belowground Storage	0	0%
18	Water Playground	0	0%
19	SUDS Not Suitable	0	0%

A large number of SUDS features (e.g. wetlands and belowground storage) were not suitable for any sites. This did not indicate that they could be implemented. The PRAST scale identified the most suitable SUDS structures for a site and a zero value only shows that the PRAST rating system had identified a better SUDS solution.

4.2.3.3.3. Civil Engineering and the Environmental Rating for PRAST

The PRAST system could be developed to take into consideration different fields of engineering by providing a weighting for all SUDS options in terms of the Primary rating (used for the Edinburgh sites), a civil engineering rating and an environmental rating. These ratings (shown in Table, 25) were derived by taking into account the benefits of each SUDS structure in terms of the context (primary, civil or environmental) within which it was being applied. For example, a wetland has a low rating in terms of civil engineering as it requires a large site area whereas it receives a high environmental rating due to the ecological benefits realised by its implementation.

Table 25. Civil engineering and environmental rating for PRAST.

SUDS Techniques:	Primary SUDS Rating	Civil Engineering Rating	Environmental Rating	Final Weighted Score
Dry Swales & Wetland	100	50	100	1000
Dry Swales & Basin / Pond	95	90	95	1120
Lined Swales & Lined Basin / Pond	95	90	95	1120
Dry Swales & Infiltration Basin	92	80	65	975
Infiltration Trenches & Infiltration Basin	85	100	60	1005
Wetland	80	45	90	850
Basin / Pond	75	60	85	870
Lined Basin / Pond	75	60	85	870
Infiltration Basin	72	75	70	870
Wet Swales	60	55	82	766
Dry Swales	55	58	80	747
Lined Dry Swales	55	58	80	747
Green Roof & Soakaway	54	60	73	729
Green Roof	50	57	68	682
Infiltration Trench	45	72	25	588
Soakaways	40	71	20	544
Filter Strips	35	10	55	380
Pervious Pavement & belowground Storage	25	45	10	335
Pervious Pavement	15	42	35	348
Belowground Storage	10	65	5	325
Water Playground	5	10	20	125

SUDS feature ratings established for all three fields could be multiplied by weighting factors as seen in Table 26, which favour any specified field. If SUDS structures were to be considered primarily in a civil engineering context then the civil engineering weighting should be set at '5' with other fields assuming the weightings of '4' and '3' as necessary.

Table 26. PRAST weighting factors.

Weighting:	
Primary SUDS Rating	5
Civil Eng Rating	4
Environmental Eng Rating	3

If the ratings specified in Table 25 were multiplied by the weightings specified in Table 26, then a scoring system would be used to develop a PRAST scale for the most appropriate SUDS options. If each SUDS option was given a rating out of 100 and the weighting for each field given as '3','4' and'5', then the greatest score possible for each SUDS feature is 1200. Table 27 shows the final weighted score realised from this process with a final position given to the SUDS structure. The original PRAST scale used in Table 23 was also noted.

Table 27. Results of analysis of methods of rating PRAST scale.

Original PRAST Number:	SUDS Techniques:	Final Weighted Score	Final Position
2A	Dry Swales & Basin / Pond	1120	1A
2B	Lined Swales & Lined Basin / Pond	1120	1B
4	Infiltration Trenches & Infiltration Basin	1005	2
1	Dry Swales & Wetland	1000	3
3	Dry Swales & Infiltration Basin	975	4
6A	Basin / Pond	870	5A
6B	Lined Basin / Pond	870	5B
7	Infiltration Basin	870	6
5	Wetland	850	7
8	Wet Swales	766	8
9A	Dry Swales	747	9A
9B	Lined Dry Swales	747	9B
10	Green Roof & Soakaway	729	10
11	Green Roof	682	11
12	Infiltration Trench	588	12
13	Soakaways	544	13
14	Filter Strips	380	14
16	Pervious Pavement	348	15
15	Pervious Pavement & Belowground Storage	335	16
17	Belowground Storage	325	17
18	Water Playground	125	18

The PRAST scale shown in Table 27 was not used by the DTS for identifying appropriate SUDS sites in Edinburgh. Nevertheless, this method might be used for establishing the most appropriate SUDS structure for implementation. It is valuable for use if a number of conflicting interests exist (such as civil engineering, environmental and city planning) during the process of identifying the most appropriate SUDS options.

4.2.3.4. Demonstration Sites

Seven demonstration sites were chosen as representatives for development, regeneration and retrofitting SUDS in Edinburgh. A geographical representation of development in Edinburgh was seen with regeneration sites being located in the north of the city and development sites located on the outskirts. Retrofit sites were located within the city boundary. Sites represented the majority results of the DST, however particularly interesting sites such as Peffermill Industrial Estate were also considered. The demonstration sites considered here are as the followings: Edinburgh Southern Harrier; Braidburn Valley Park; Edinburgh Park; Inchview Primary School; North Fort Street; Peffermill Industrial Estate and; Redhall House Drive.

The Peffermill Industrial Estate was chosen as the representative of all above and an will be discussed in more details.

4.2.3.5. Demonstration Site (Detailed Design)

Future building design plans for the Peffermill Industrial Estate area were not yet confirmed. Nevertheless, the development of small scale industrial units was planned and therefore all SUDS recommendations have been made with regard to this assumption (Fig, 22).

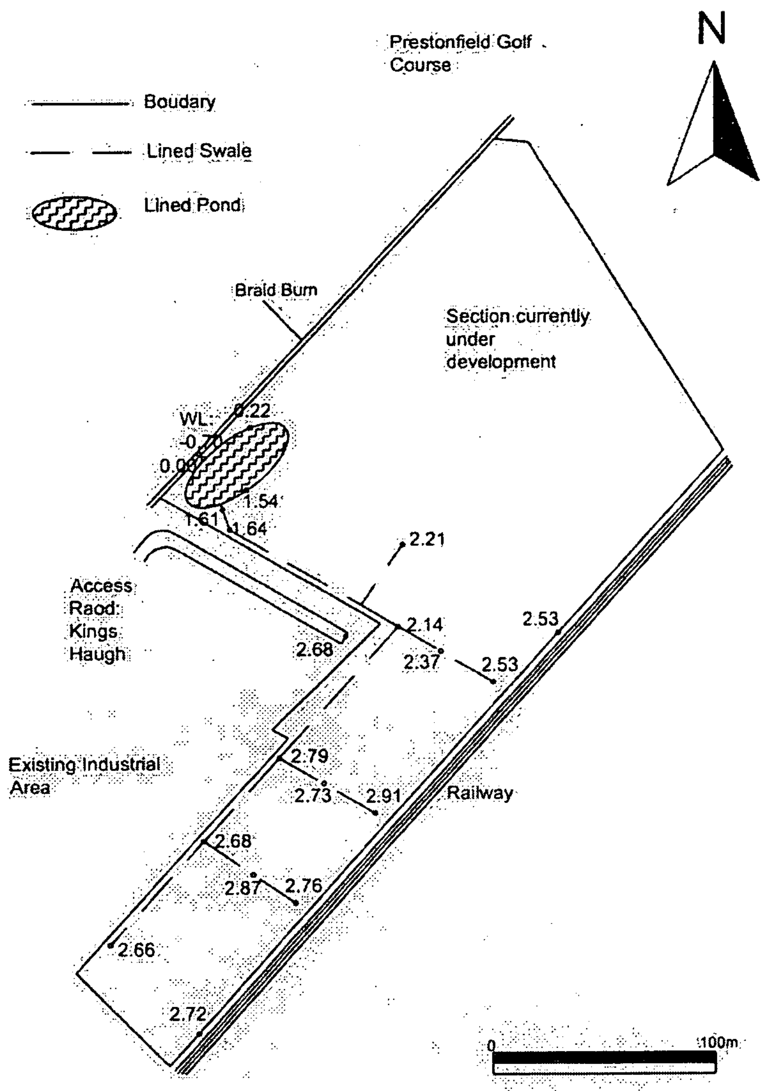


Fig 22. Peffermill Industrial Estate: proposed sustainable urban drainage system design drawing.

The main entrance to the Peffermill Industrial Estate site is through Kings Haugh, located adjacent the eastern boundary of Peffermill Playing Fields on Peffermill Road (A6095). The Braid Burn borders the North West edge of Kings Haugh, which runs from the main entrance of the industrial estate in a north easterly direction. Kings Haugh gives access to industrial units on its south eastern edge. Once passed the recently developed industrial units, it then alters direction towards the south east, giving access to the area proposed for development and potential SUDS site.

The south eastern area of the site, located between the already existing industrial units and a goods railway, is overgrown with some small trees and shrubs. The central area, adjacent to Kings Haugh, wass wasteland with small piles of rubble dumped randomly on the surface. Litter (plastic bags, vehicle tyres and paper etc) covered the entire site. Site clearance had recently taken place in the north east area of the site in preparation for development (as of April 2005) and consequently was not included within this site design.

4.2.3.5.1. SUDS Planning

The entire site was characterised by a moderate downward slope to the northeast. Sections of the site were level except for some small depressions on the site surface. The best engineering option was to have an inter-connected network of swales throughout the site allowing suitable spacing for small scale industrial units. The presence of a network of swales was essential due to the possible layout of the site after development. Recommended locations for the swale networks are shown in Fig 22. Swales are to be located within grass verges and culvert below access roads. It was also recommended that roof runoff from some of the existing industrial units was redirected into the swales. All swales should convey water to a detention basin/pond.

A large depression existed in the northwest angle of the site, adjoining the Braid Burn and Kings Haugh. This area had then been identified as an ideal location for the implementation of a detention or attenuation pond and an associated outlet structure to the Braid Burn (Fig, 23). The detention pond would allow runoff to be attenuated before it was diverted to the burn. It was therefore recommended that no building construction work should be undertaken in this part of the Peffermill Industrial Estate. The available surface area for a SUDS feature here was found to be approximately 400 m^2. The future impermeable area of the section studied (with green roofs implemented) was estimated to be 8000 m^2. As a result, a pond draining runoff from a rainstorm of 2cm would require a volume of 160 m^3. The pond system had to be sized in a way that it could solve the problems associated with eutrophication and oxygen depletion and ensure a permanent pool exist. A suitable depth would then be 1.5 m in the centre (Martin, 2000) with the total pond fitting into the available space of 400 m^2.

Fig 23. Peffermill Industrial Estate: photograph taken on 2nd May 2005 (Scholz *et al.*, 2006).

The detention pond area should include space for decorative planting and landscaping, a seating area and a footpath to create an area of high amenity value. A flowing swale acting as a combined overflow and outlet structure should run from the detention pond to the Braid Burn. If this was complex to construct, then an open channel lined with decorative brick might be more suitable.

Pervious pavements were found unsuitable for the site due to the likelihood of heavy goods transport. However, green roofs were identified by the DST as alternative suitable SUDS option. This would help to detain roof runoff while improving the appearance of the site, especially from Arthur's Seat.

4.2.3.5.2. Soil Analysis

The soil sieving analysis (Fig, 24) demonstrated that infiltration through the soil surface was expected to be slow due to the presence of the relatively fine particles. However, this was not relevant for the proposed SUDS features as they required a liner due to supposed ground contamination. This liner would isolate the swale and basin/pond from the ground and prevent infiltration, in effect making the SUDS devices useful for runoff detention only.

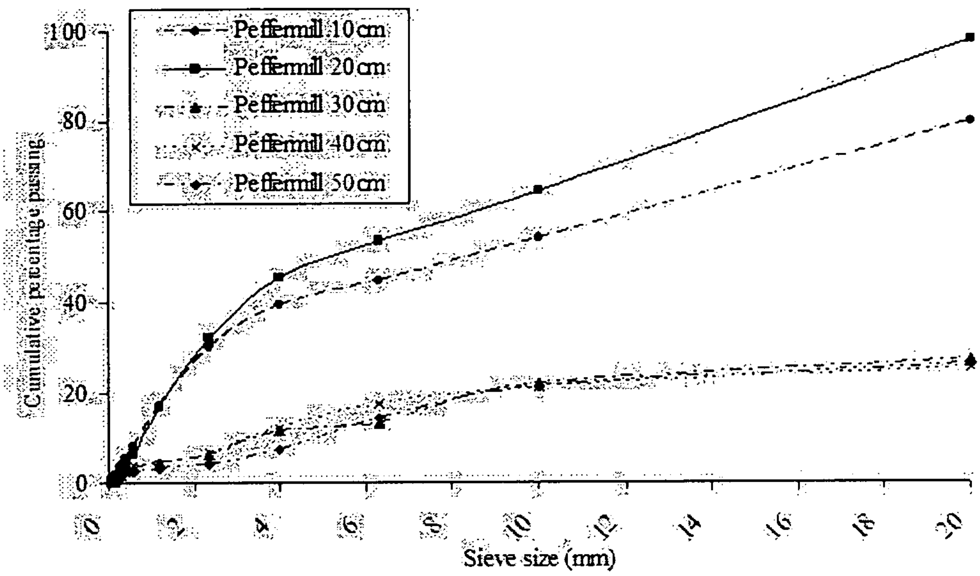


Fig 24. Peffermill Industrial Estate: particle size distribution curves for samples taken at 10cm intervals within a ditch of 0.5 m depth

At greater depths, the ground was found to contain larger material, in particular stones of 3cm in diameter or more. These were compressed together, forming a hard layer of ground beneath the surface layer of soil. These findings matched the site history where it was established that the site was formerly a brewery with the 'Innocent Railway' running close by (Scottish Library). The material sampled at depths greater than 20 cm were especially similar in nature to railway ballast and it was relatively possible that this was found. Some fine soil existed between the voids of the material with the percentage present decreasing with depth, as shown in Fig 24. The characteristics of the material did not change at greater depths and for that reason samples were only taken to a depth of 50 cm.

Considering the solid ground conditions and the presence of a large depression, it was therefore feasible to implement a lined detention basin/pond in this location.

4.3. Assessing Stormwater Detention Systems Treating Road Runoff with an Artificial Neural Network

4.3.1. Inflow and Outflow Water Quality

Table 28 summarises values representing the inflow water quality variables. Particularly during warmer seasons, values for five-days at 20°C biochemical oxygen demand (BOD) (nitrification inhibitor applied), suspended solids, ortho-phosphate-phosphorous and nitrate-nitrogen are above commonly accepted water quality standard thresholds (25, 35, 2 and 15 mg/l, respectively) for secondary wastewater treatment (ECC, 1991). This is partly due to the inflow water quality being representative of the 'worst case scenario', and the lack of precipitation between 2006-03-24, and 2006-09-13.

Table28. Inflow water quality.

Variables	No. of samples	Mean[f]	Mean[g]	Mean[h]	Standard deviation[f]	Standard deviation[g]	Standard deviation[h]
pH (-)	70	6.91	7.03	7.05	0.351	0.556	0.524
DO[a] (mg/l)	69	3.2	4.0	3.7	2.24	1.33	2.31
BOD[b] (mg/l)	67	23.0	49.5	54.1	17.05	50.75	318.72
SS[c] (mg/l)	70	99.9	52.2	68.3	106.60	48.21	191.67
TS[d] (mg/l)	69	588.6	523.9	1791.3	395.18	408.19	1427.38
TDS[e] (mg/l)	70	112.5	117.9	186.1	78.11	141.39	138.93
Conductivity (μS)	70	222.9	228.0	372.5	157.94	268.75	278.10
Turbidity (NTU)	69	37.6	55.3	111.4	15.01	35.58	125.49
Ortho-phosphate-phosphorus (mg/l)	69	1.6	3.3	22.8	1.95	3.97	15.55
Ammonia-nitrogen (mg/l)	68	1.4	0.7	1.8	1.63	0.60	1.33
Nitrate-nitrogen (mg/l)	68	0.2	1.4	1.0	0.10	3.45	1.06

[a]dissolved oxygen; [b]five-days at 20°C bochemical oxygen demand (nitrification inhibitor applied);[c]total suspended solids; [d]total solids; [e]total dissolved solids; [f]2005/03/23-2005/09/15; [g]2005/09/22-2006/03/16; [h]2006/03/24-2006/09/13.

Values for outflow water quality variables are shown in Table 29. Considerable improvements in the quality of the outflow have been observed, particularly if compared to the inflow values summarized in Table 28. This is the case during cold periods for variables such as suspended solids, BOD, and turbidity, where most values are considerably below water quality treatment standard thresholds (ECC, 1991).

Table 29. Outflow water quality.

Variables	No. of samples	Mean[f]	Mean[g]	Mean[h]	Standard deviation[f]	Standard deviation[g]	Standard deviation[h]
			System 1				
pH (-)	66	7.33	7.39	7.47	0.321	0.635	0.136
DO[a] (mg/l)	65	3.8	4.5	3.4	1.56	1.83	2.24
BOD[b] (mg/l)	64	2.3	5.4	28.2	2.34	7.98	13.52
SS[c] (mg/l)	65	8.3	11.1	31	12.37	15.10	31.66
TS[d] (mg/l)	65	569.6	530.6	675.2	240.25	867.15	666.83
TDS[e] (mg/l)	66	557.5	121.8	207.4	954.00	41.64	60.13
Conductivity (μS)	65	1154.4	247.1	415.0	1926.73	84.23	120.77
Turbidity (NTU)	65	4.9	4.1	7.6	3.76	3.13	4.05
Ortho-phosphate-phosphorus (mg/l)	65	1.4	3.4	14.3	1.64	2.87	2.88
Ammonia-nitrogen (mg/l)	66	0.3	0.6	0.8	0.87	1.56	0.98
Nitrate-nitrogen (mg/l)	63	0.6	1.1	0.1	0.65	1.24	0.07
			System 2				
pH (-)	70	7.24	7.38	7.58	0.330	0.632	0.279
DO[a] (mg/l)	69	3.8	4.7	7.4	1.10	1.58	6.36
BOD[b] (mg/l)	70	1.9	4.7	24.9	1.45	2.75	11.61
SS[c] (mg/l)	70	6.5	7.8	27.5	3.23	7.39	11.81
TS[d] (mg/l)	70	518.6	300.2	427.5	367.67	104.00	213.85
TDS[e] (mg/l)	70	381.7	136.8	351.7	286.85	59.66	297.71
Conductivity (μS)	70	770.7	272.8	703.4	565.51	120.93	595.42
Turbidity (NTU)	69	3.9	4.4	41.1	2.14	3.10	57.08
Ortho-phosphate-phosphorus(mg/l)	69	1.2	4.3	19.9	1.48	3.97	14.34
Ammonia-nitrogen (mg/l)	68	1.6	0.7	3.3	1.47	1.34	3.53
Nitrate-nitrogen (mg/l)	68	1.7	1.1	0.6	1.79	1.33	1.20
			System 3				
pH (-)	70	7.39	7.44	7.54	0.270	0.675	0.097
DO[a] (mg/l)	69	3.3	3.9	3.9	1.16	1.53	2.31
BOD[b] (mg/l)	67	1.7	4.1	9.1	1.17	4.40	8.61
SS[c] (mg/l)	70	6.7	11.0	13.5	7.15	13.39	5.98
TS[d] (mg/l)	70	483.5	326.1	422.8	248.14	167.41	157.68
TDS[e] (mg/l)	70	438.5	127.7	352.1	596.67	65.63	220.37
Conductivity (μS)	70	877.3	262.1	703.7	1193.15	130.67	439.59
Turbidity (NTU)	67	5.3	7.6	5.0	3.40	7.29	1.93
Ortho-phosphate-phosphorus(mg/l)	70	1.5	2.9	17.2	1.82	2.28	31.71
Ammonia-nitrogen (mg/l)	70	0.4	1.1	1.9	0.72	1.77	2.20
Nitrate-nitrogen (mg/l)	68	0.6	1.6	0.8	0.32	1.31	1.60
			System 4				
pH (-)	70	7.47	7.45	7.68	0.181	0.654	0.136
DO[a] (mg/l)	69	3.2	4.4	4.7	1.13	1.48	2.42
BOD[b] (mg/l)	68	1.9	3.6	5.7	0.93	4.01	5.50
SS[c] (mg/l)	70	8.2	13.7	14.3	14.20	19.21	18.50
TS[d] (mg/l)	70	478.5	308.3	381.9	228.23	147.42	141.76
TDS[e] (mg/l)	70	344.2	130.1	403.4	225.34	54.05	322.35

(Table 29 cont)

Conductivity (μS)	70	693.8	263.3	806.6	445.47	107.26	644.60
Turbidity (NTU)	69	7.3	6.5	2.8	7.74	5.80	2.27
Ortho-phosphate-phosphorus (mg/l)	68	1.6	3.6	15.2	1.79	2.57	19.21
Ammonia-nitrogen (mg/l)	68	0.5	1.3	0.2	0.71	1.87	0.06
Nitrate-nitrogen (mg/l)	68	0.8	1.7	0.8	0.27	1.31	0.99
			System 5				
pH (-)	70	7.41	7.53	7.61	0.202	0.530	0.133
DO[a] (mg/l)	69	3.8	4.9	4.8	1.20	1.67	2.05
BOD[b] (mg/l)	68	2.4	5.1	5.3	1.97	4.84	11.12
SS[c] (mg/l)	70	6.2	4.1	7.0	4.80	3.32	5.40
TS[d] (mg/l)	70	594.8	320.9	400.5	348.56	133.50	145.67
TDS[e] (mg/l)	70	391.1	125.8	422.6	185.78	54.10	364.38
Conductivity (μS)	70	779.5	252.9	845.1	374.88	106.61	728.89
Turbidity (NTU)	69	15.2	6.5	2.6	53.63	7.38	0.57
Ortho-phosphate-phosphorus (mg/l)	70	1.6	3.7	19.9	1.69	2.48	46.75
Ammonia-nitrogen (mg/l)	70	0.4	0.5	0.2	0.57	0.59	0.14
Nitrate-nitrogen (mg/l)	68	0.5	1.6	0.6	0.25	1.84	0.92

[a]dissolved oxygen; [b]five-days at 20°C (nitrification inhibitor applied) biochemical oxygen demand; [c]total suspended solids; [d]total solids; [e]total dissolved solids; [f]2005/03/23-2005/09/15; [g]2005/09/22-2006/03/16; [h]2006/03/24-2006/09/13.

4.3.2. Multiple Linear Regression Analyses

Table 30 shows how BOD, and SS can be predicted by applying a multiple linear regression analysis covering eighteen months of experimental data. Electrical conductivity, turbidity, pH, ortho-phosphate-phosphorous, nitrate-nitrogen, and ammonia-nitrogen were selected for the prediction because the determination of these variables is less costly and time-consuming. Furthermore, stepwise regression was also undertaken to help in the selection of the most appropriate variables for prediction. Furthermore, total coliforms, and intestinal Enterococci colony forming units did not exhibit a significant correlation ($p<0.05$) with any of the proposed predictors.

Table 30. Multiple linear regression analyses applied to predict the five-days at 20ºC (nitrification inhibitor applied) biochemical oxygen demand (BOD, mg/l), and suspended solids (SS, mg/l).

Sample	a	b	c	d	e	f	g	SEE[a]	R^{2b}
				BOD					
Inflow	0	1.21	31.1	9.19	0	0	-207.0	12.3	0.88
System 1	0	0	0	2.46	0.30	0	-1.6	5.3	0.81
System 2	0	0.35	0	0	-4.55	9.93	2.8	5.1	0.82
System 3	0	0	0	0	-1.78	1.36	4.7	3.5	0.53
System 4	0	0	-2.1	0	-0.99	0	20.2	2.8	0.43
				SS					
Inflow	0.190	0.56	0	0	0	NA	34.9	33.16	0.83
System 1	0	0.38	0	2.76	-0.94	NA	2.8	7.74	0.79
System 2	0.001	0.18	3.3	0.69	0	NA	-17.8	6.14	0.65
System 3	0	0	10.4	0	0	NA	-68.3	4.25	0.81
System 4	0	0	11.9	0	0	NA	-77.6	7.75	0.79
System 5	0.003	0	0	-1.94	0	NA	17.9	3.38	0.46

The multiple regression equation (Variable to be predicted = a× (electro conductivity, μs) + b × (turbidity, NTU) + c × (pH) + d × (orthophosphate-phosphorous, mg/l) + e × (nitrate-nitrogen, mg/l) + f × (ammonia-nitrogen, mg/l) + g) was fitted. [a]standard error of the estimate; [b]coefficient of determination. NA, not applicable.

As indicated in Table 30, the application of multiple linear regression analyses for the prediction of BOD was relatively successful when applied to samples from the inflow, and systems 1, and 2. This has been attributed to a high correlation between BOD, and most of the selected predictors. Moreover, as there has been no strong correlation between BOD, and other key water quality variables for system 5, a multiple regression analysis was not performed.

Standard errors of the estimates for suspended solids were higher than the corresponding ones for the BOD. The coefficients of determination (r^2) are relatively high for all systems with the exception of system 5. However, multiple regression analysis is not successful in predicting suspended solids if a considerable number of outliers are part of the corresponding dataset.

4.3.3. Analyses of Variance

A one way ANOVA was conducted to test if the systems performed similarly concerning stormwater treatment. The outcome of this analysis allows the design engineer to opt for a system that performs well and is cost-effective. For example, if there is no significant difference between the performances of two different systems for the most important key variables, the designer would be well advised to choose the less costly option.

There were significant differences in treatment performances concerning BOD, ammonia-nitrogen, total coliforms, suspended solids, and intestinal enterococci with F values (ratio of the mean variance between groups divided by the mean variance within groups) of 5.3, 8.0, 10.0, 3.6 and 4.1 respectively.

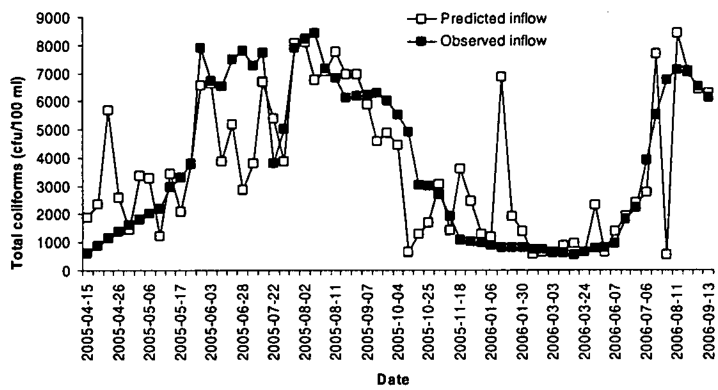
Furthermore, the ANOVA indicated that there were significant ($p<0.05$) differences between some of the water quality parameters in the inflow, and outflow for each system. Significant differences with respect to system 1 were found for total dissolved solids, turbidity, electrical conductivity, dissolved oxygen, ortho-phosphate-phosphorous, nitrate-nitrogen, intestinal enterococci counts, and total coliform counts with F values of 2085.2, 8.9, 2006.7, 3.4, 48.2, 69.9, 45.0 and 42.0, respectively. For system 2, there were significant differences found for turbidity, ortho-phosphate-phosphorous, and nitrate-nitrogen with F values of 35.5, 18.6, and 26.8, respectively. Concerning system 3, the ANOVA shows significant differences for BOD, total dissolved solids, dissolved oxygen, ortho-phosphate-phosphorus, ammonia-nitrogen, intestinal enterococci, and total coliforms with F values of 4.5, 10.7, 1.9, 20.2, 3.5, 225.5, and 7129.3, respectively. The results from system 4 showed significant differences for suspended solids, electric conductivity, ortho-phosphate-phosphorous, nitrate-nitrogen, intestinal enterococci counts, and total coliforms counts with F values of 6.0, 20.3, 12.6, 74.9, 26.6 and 126.3, respectively. Finally, an ANOVA for system 5 detected significant differences for turbidity, dissolved oxygen, ortho-phosphate-phosphorous, nitrate-nitrogen, intestinal enterococci counts, and total coliform counts with F values of 83.5, 3.7, 10.8, 301.7, 6.4 and 127.6, respectively.

4.3.4. Artificial Neural Network Modelling

The coefficient of determination (R^2) values for predicting total coliform counts for the inflow and outflows of systems 1 to 5 were 0.89, 0.94, 0.91, 0.98, 0.59 and 0.95, respectively. The corresponding R^2 values for predicting intestinal enterococci counts were 0.80, 0.63, 0.78, 0.73, 0.71 and 0.15, respectively. It follows that the models were able to successfully predict the total coliform, and intestinal enterococci colony forming unit counts with an exception for the prediction of intestinal enterococci in system 5.

Figs 25 and 26 show the observed, and predicted total coliform and intestinal enterococci counts in the inflow and the Outflows of all systems, respectively. The artificial neural networks successfully predicted total coliform, and intestinal enterococci counts for the inflow water. The models were very successful in predicting total coliform counts for all systems except for system 4. Concerning intestinal enterococci counts, the models were relatively successful. When predicting total coliform counts with the artificial neural network models for the inflow, and systems 1, 2, 3 and 5, one can undertake predictions confidently resulting in mean squared errors close to zero. In the case of intestinal enterococci counts, the inflow and systems 2, 3 and 4 had similar R^2 values.

a)



b)

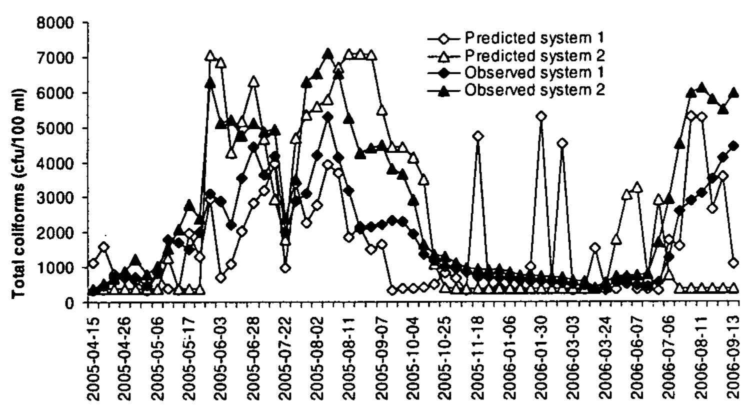
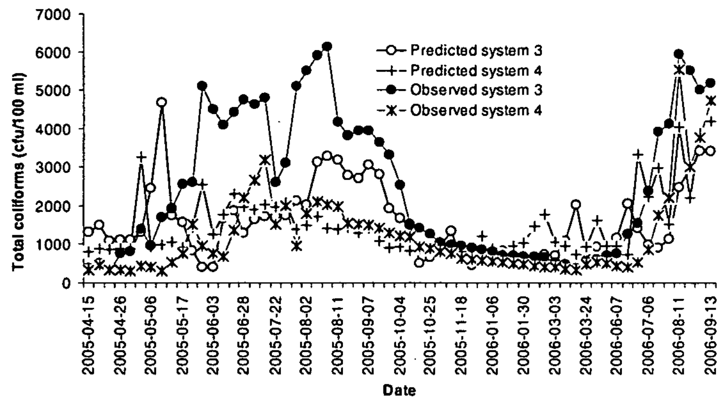


Fig 25. Observed, and predicted total coliform counts in (a) the inflow and the outflows of (b) systems 1, and 2, (c) systems 3, and 4, and (d) system 5.

c)



d)

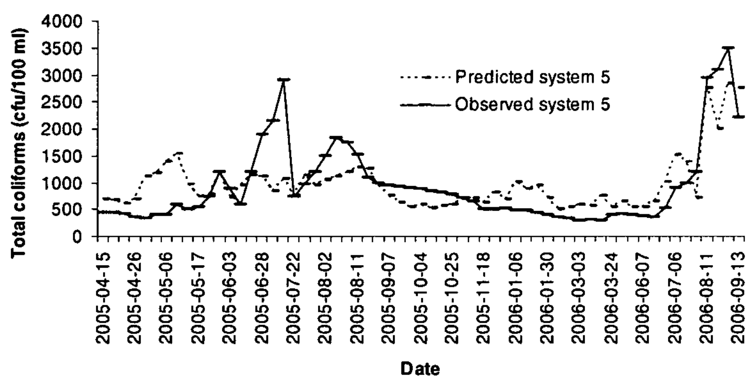
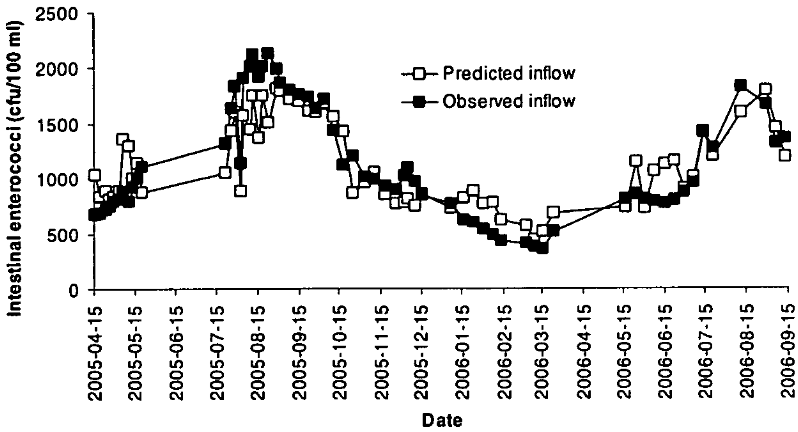


Fig 25 (continuation). Observed, and predicted total coliform counts in (a) the inflow and the outflows of (b) systems 1, and 2, (c) systems 3, and 4, and (d) system 5.

a)



b)

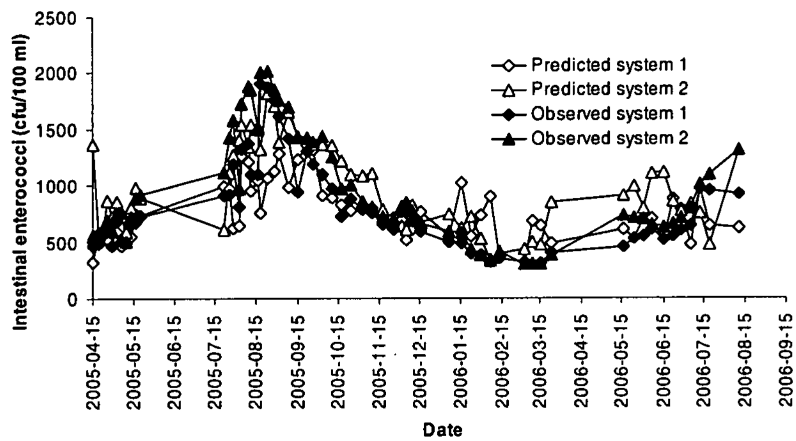
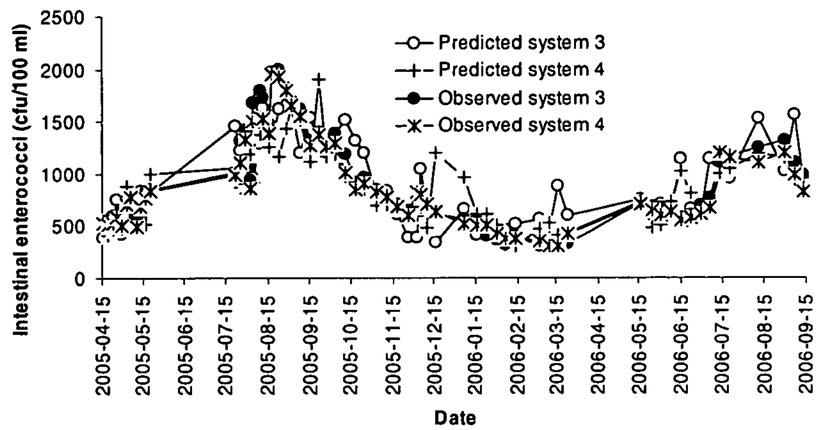


Fig 26. Observed, and predicted intestinal enterococci counts in (a) the inflow and the outflows of (b) systems 1, and 2, (c) systems 3, and 4, and (d) system 5.

c)



d)

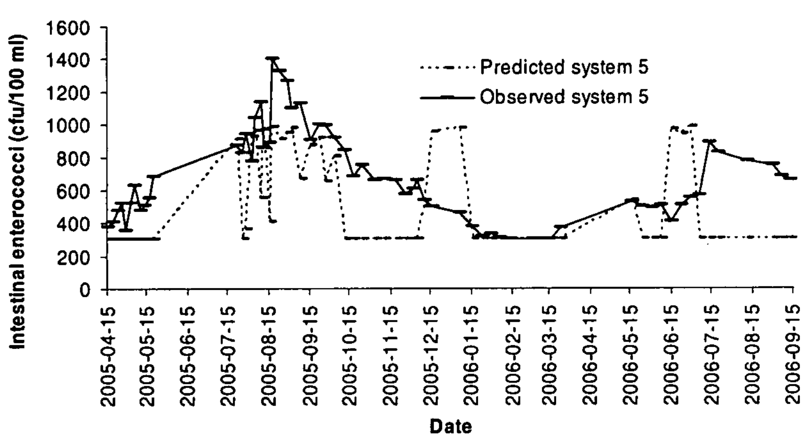


Fig 26 (continuation). Observed, and predicted intestinal enterococci counts in (a) the inflow and the outflows of (b) systems 1, and 2, (c) systems 3, and 4, and (d) system 5.

4.4. Stormwater Infiltration Systems For Road Runoff Contaminated with Organic Matter Including Dog Faeces

4.4.1. Water Quality Performance

The main objectives of the SUDS demonstration plant were to reduce BOD, SS and nutrient concentrations, and the number of total coliforms. Very few water samples were collected from the silt trap, which was frequently dry. Therefore, percentage reduction values for most water quality variables cannot be realistically determined. The inflow water to the detention tank was therefore assumed to be similar to the real inflow water. However, this approximation underestimates the treatment performance of the system, because considerable treatment of runoff takes place in the silt trap and the gravel ditch leading to the detention tank. Tables 31 and 32 summarise the overall water quality of the systems. The results from the nutrient analysis of the detention tank and the infiltration ponds are shown in Tables 33 and 34 respectively.

Table 31. Water quality variations of the detention tank (20/3/2005-23/1/2006).

Variable	Unit	Count		Standard Deviation		Mean Spring[8]		Mean Summer[9]		Mean Autumn[10]		Mean Winter[11]	
		In	Out	In	Out	In	Out	In	Out	In	Out	In	Out
BOD[1]	mg l^{-1}	37	35	12.3	11.7	21.0	9.5	29.0	7.9	41.0	16.3	23.5	8.3
SS[2]	mg l^{-1}	39	39	71.5	61.6	72.7	77.9	127.3	72.4	17.0	10.2	3.9	6.1
TS[3]	mg l^{-1}	39	39	427	104	194	108	391	104	452	195	256	80
Turb[4]	NTU	41	41	16.0	9.6	11.5	7.6	15.3	10.0	31.6	12.3	10.8	9.1
DO[5]	mg l^{-1}	43	43	1.4	0.8	2.2	3.6	1.1	2.6	1.3	2.4	0.7	2.8
pH	-	44	44	0.4	0.4	6.9	6.8	7.3	7.2	7.5	7.2	7.5	7.2
Con[6]	µS	42	42	141	64	199	102	261	128	444	146	338	93
Tem[7] (air)	°C	41	41	5.8	5.8	14.4	14.4	19.7	19.7	11.2	11.2	6.9	6.9
Tem[7]	°C	44	44	2.9	3.0	10.5	9.5	12.9	12.3	9.8	9.5	7.7	6.9

[1]five-day at 20°C N-Allylthiourea biochemical oxygen demand; [2]suspended solids; [3]total solids; [4]turbidity; [5]dissolved oxygen; [6]conductivity; [7]temperature; [8]21/03/05-21/06/05; [9]22/06/05-21/09/05; [10]22/09/05-21/12/05; [11]22/12/05-21/03/06.

Table 32. Water quality variations of the planted (PP) and unplanted (UP) infiltration ponds (20/3/2005-23/1/2006).

Variable	Unit	Count		Standard Deviation		Mean Spring[8]		Mean Summer[9]		Mean Autumn[10]		Mean Winter[11]	
		PP	UP	PP	UP	PP	UP	PP	UP	PP	UP	PP	UP
BOD[1]	mg l^{-1}	34	36	11.9	10.1	15.1	13.4	21.0	24.4	21.8	6.8	9.3	20.0
SS[2]	mg l^{-1}	40	39	161	177	150	170	50	161	28	9.0	7.0	11.0
TS[3]	mg l^{-1}	40	39	276	188	273	165	364	350	281	142	332	113
Turb[4]	NTU	43	42	65.2	81.7	16.3	24.0	37.7	68.8	14.2	9.4	4.3	2.6
DO[5]	mg l^{-1}	43	45	1.8	2.0	4.7	5.3	2.8	4.6	1.3	2.4	1.6	2.3
pH	-	46	45	0.4	0.5	6.9	7.4	7.1	7.9	7.4	7.5	7.5	7.5
Con[6]	µS	44	43	48	43	273	167	266	206	263	166	337	140
Tem[7] (air)	°C	47	47	5.8	5.8	14.4	14.4	19.7	19.7	11.2	11.2	6.9	6.9
Tem[7]	°C	48	49	4.6	4.9	10.8	11.8	13.6	14.8	10.3	9.2	5.7	5.5

[1]five-day at 20°C N-Allylthiourea biochemical oxygen demand; [2]suspended solids; [3]total solids; [4]turbidity; [5]dissolved oxygen; [6]conductivity; [7]temperature; 821/03/05-21/06/05; 922/06/05 -21/09/05; 1022/09/05-21/12/05; 1122/12/05 -21/03/06.

The SS concentrations in all parts of the system occasionally exceeded the UK threshold of 20 mg/L for secondary treated wastewater; particularly in summer when water levels were low and biomass production high. In general, the systems show better treatment performances for BOD than for SS (Tables, 31 and 32). The results indicate that all systems perform better during colder seasons such as winter; this can partly be expressed by algal blooms during warmer seasons within the infiltration ponds and the faster rate of debris decay in summer. The combination of high conductivity, BOD and SS values indicates high decomposition rates.

4.4.1.1. Nutrient Removal Performance

Results for orthophosphate- phosphorus analysis indicate significant reductions towards the end of the system operation. Findings show relatively high ammonia concentrations in the inflow to the detention tank. However, ammonia was considerably reduced inside the detention tank during a period of extended storage (Table, 33).

Table 33. Nutrient concentrations in the detention tank (20/3/2005-23/1/2006).

Variable	Unit	Count		Standard Deviation		Mean Spring[1]		Mean Summer[2]		Mean Autumn[3]		Mean Winter[4]	
		In	Out	In	Out	In	Out	In	Out	In	Out	In	Out
O-phosphate-phosphorus	mg l^{-1}	47	45	4.5	2.1	1.4	1.2	1.7	1.7	6.5	2.6	12.3	4.9
Ammonia-N	mg l^{-1}	47	44	22.1	0.5	4.2	0.7	9.1	0.9	35.9	0.2	18.8	0.3
Nitrate-N	mg l^{-1}	47	45	3.1	0.2	0.14	0.30	0.21	0.41	2.44	0.40	0.28	0.50

[1]21/03/05-21/06/05; [2]22/06/05-21/09/05; [3]22/09/05-21/12/05; [4]22/12/05-21/03/05.

Ammonia data points for the infiltration ponds were generally even lower, indicating removal due to adsorption onto soil particles and productive nitrifying bacteria (Table, 34).

Table 34. Nutrient concentrations of the planted (PP) and unplanted (UP) infiltration ponds (20/3/2005-23/1/2006).

Variable	Unit	Count		Standard Deviation		Mean Spring[1]		Mean Summer[2]		Mean Autumn[3]		Mean Winter[4]	
		PP	UP	PP	UP	PP	UP	PP	UP	PP	UP	PP	UP
O-phosphate-phosphorus	mg l^{-1}	48	49	0.6	0.6	0.74	0.44	1.10	0.86	0.46	1.10	0.60	0.61
Ammonia-N	mg l^{-1}	48	49	2.5	8.1	0.32	1.01	0.87	3.33	2.02	1.63	0.11	0.24
Nitrate-N	mg l^{-1}	48	49	0.1	0.1	0.11	0.17	0.21	0.20	0.14	0.17	0.12	0.05

[1]21/03/05-21/06/05; [2]22/06/05-21/09/05; [3]22/09/05 -21/12/05; [4]22/12/05 -21/03/05.

The occasional increase in nitrate concentrations towards the outflow of the detention tank is caused by high microbial nitrification within the tank. Nitrate levels were reduced even further within the ponds. Nitrifying bacteria are also responsible for the reduction of nitrate within the ponds. Micro-organisms converted organic nitrate into inorganic nitrate, which has subsequently been taken up by plants (Tables, 33 and 34).

4.4.1.2. Microbiological Performance

The findings from the microbiological study of the detention tank and the infiltration ponds are shown in Tables 35 and 36, respectively.

Table 35. Microbiological determinations concerning the detention tank (20/3/2005-23/1/2006).

Medium	Unit	Count		Standard Deviation		Mean Spring[4]		Mean Summer[5]		Mean Autumn[6]		Mean Winter[7]	
		In	Out	In	Out	In	Out	In	Out	In	Out	In	Out
NA[1]	CFUml^{-1}	46	46	2.0E+07	1.1E+08	1.2E+07	5.0E+07	6.1E+06	2.1E+07	2.1E+06	6.4E+07	1.0E+05	3.0E+05
S &B[2]	CFUml^{-1}	24	24	275	306	996	1129	701	716	360	445	305	400
MC3[3]	CFUml^{-1}	24	24	4762	1035	6547	1631	751	501	440	325	395	420

[1]nutrient agar; [2]Slanetz and Bartley medium; [3]MacConkey No. 3 medium; [4]21/03/05-21/06/05; [5]22/06/05-21/09/05; [6]22/09/05-21/12/05; [7]22/12/05-21.

The results from microbiological plate count tests (Tables, 35 and 36) indicate high faecal contamination. Relatively high microbial activity amongst heterotrophic bacteria within the system is apparent, if compared to data (not shown) before contamination with dog faeces.

Table 36. Microbiological determinations concerning the planted (PP) and unplanted (UP) infiltration ponds (20/3/2005-23/1/2006).

Medium	Unit	Count		Standard Deviation		Mean Spring[4]		Mean Summer[5]		Mean Autumn[6]		Mean Winter[7]	
		PP	UP	PP	UP	PP	UP	PP	UP	PP	UP	PP	UP
NA[1]	CFUml^{-1}	46	46	4.7E+07	2.4E+07	3.0E+07	1.4E+07	3.4E+07	9.2E+07	8.4E+07	5.0E+07	7.2E+07	1.1E+07
S &B[2]	CFUml^{-1}	24	24	179	93	456	412	631	499	370	350	325	305
MC3[3]	CFUml^{-1}	24	24	834	869	1122	1369	418	529	325	305	315	325

[1]nutrient agar; [2]Slanetz and Bartley medium; [3]MacConkey No. 3 medium; [4]21/03/05-21/06/05; [5]22/06/05-21/09/05; [6]22/09/05-21/12/05; [7]22/12/05 -21/03/05.

The lower concentration of heterotrophic bacteria during cold winter months could be explained with the accelerated growth rate of these organisms at temperatures above 10°C (upper threshold at 45°C). A comparison of data gathered from the microbial plate count tests using Slanetz and Bartley media with the 200 CFU/mL EU standard for intestinal enterococci in bathing waters indicated the frequent failure of the system in reducing the concentration of intestinal enterococci below this guideline. However, bathing in SUDS pond is prohibited anyway. Regarding total coliform numbers, the SUDS performance was satisfactory during colder seasons (<500 CFU/mL), but has shown signs of failure during spring and summer.

4.4.1.3. Overall Performance

An analysis of variance indicated that there were significant differences between most sampling points with regard to their treatment performances ($P < 0.05$); exceptions were found for temperature, total solids, nitrate and the bacteria counts on nutrient agar. Nevertheless, the planted and unplanted infiltration ponds performed similar, and a planted pond has no major advantage over an unplanted pond from a water quality point of view. However, a planted pond has obvious ecological and recreational benefits.

4.4.2. Active Control of Algae with C. Auratus

Algae began to grow in the infiltration ponds until *C. auratus* were introduced in April 2004. The result was a pleasant and clean SUDS during the second year of operation despite fears of water quality deterioration voiced elsewhere (Richardson and Whoriskey, 1992) (Tables 37 and 38).

Table 37. Summary statistics: water quality of the planted pond receiving the outflow from the constructed wetland before (01/04/03-31/03/04) and after (01/04/04-31/03/05) the introduction of *C. auratus* (common goldfish).

Variable	Unit	Sampling number		Mean		Standard deviation	
Temperature	°C	56	94	8.7	11.6	4.60	4.90
BOD[a]	mg/l	36	43	15.5	19.3	18.91	14.25
Suspended solids	mg/l	47	93	58.7	24.7	116.61	55.45
Ammonia-N	mg/l	34	71	0.3	0.1	0.58	0.21
Nitrate-N	mg/l	28	69	0.7	0.4	2.25	0.84
Phosphate-P	mg/l	33	72	0.18	0.25	0.149	0.238
Conductivity	μS	58	93	310.5	246.9	116.86	83.21
Turbidity	NTU	58	94	18.4	14.2	20.02	29.84
Dissolved oxygen	mg/l	52	94	6.1	3.5	7.01	1.54
pH	-	57	94	7.2	7.2	0.24	0.25

[a]five-day at 20°C biochemical oxygen demand.

Table 38. Summary statistics: water quality of the unplanted pond receiving the outflow from the constructed wetland before (01/04/03-31/03/04) and after (01/04/04-31/03/05) the introduction of *C. auratus* (common goldfish).

Variable	Unit	Sampling number		Mean		Standard deviation	
Temperature	°C	56	94	8.7	11.6	4.60	4.90
BOD[a]	mg/l	36	43	15.5	19.3	18.91	14.25
Suspended solids	mg/l	47	93	58.7	24.7	116.61	55.45
Ammonia-N	mg/l	34	71	0.3	0.1	0.58	0.21
Nitrate-N	mg/l	28	69	0.7	0.4	2.25	0.84
Phosphate-P	mg/l	33	72	0.18	0.25	0.149	0.238
Conductivity	μS	58	93	310.5	246.9	116.86	83.21
Turbidity	NTU	58	94	18.4	14.2	20.02	29.84
Dissolved oxygen	mg/l	52	94	6.1	3.5	7.01	1.54
pH	-	57	94	7.2	7.2	0.24	0.25

[a]five-day at 20°C biochemical oxygen demand.

Carassius auratus (similar to *Cyprinus carpio* or also known as common carp) is classified as herbivores with wild specimens predominantly feeding on plants. This particularly applies to closed pond systems. Therefore, *C. auratus* could be used to control aquatic weeds and potentially algae in ponds (Zheng *et al.*, 2005, Richardson and Whoriskey, 1992 and Gouveia, and Rema, 2005).

Concerning the field experiment, relatively high numbers of filamentous green algae (Chlorophyta) were counted in pond samples taken on 29 March 2004. The dominant algae present was *Odeogonium capillare* that is cosmopolitan in freshwater. *Odeogonium capillare* can form mats in small ponds, and is often mistaken for the more common *Cladophora glomerata* (blanket weed) (Zheng *et al.*, 2005).

Carassius auratus was introduced to control predominantly filamentous green algae and to increase public acceptance of SUDS. Concerning samples of algae taken on 4 October 2004, both the unplanted and planted ponds were less dominated by *Odeogonium capillare* in comparison to estimations on 29 March 2004. Moreover, the unplanted pond developed a greater diversity of filamentous green algae if compared to the planted pond. This may be due to the absence of macrophytes that would compete with algae for nutrients (particularly phosphorus). Moreover, large macrophytes (located in the planted pond) provide shade leading to a reduction of sunlight penetrating the water, and subsequently reducing the growth of algae (CIRIA, 2000 and Zheng *et al.*, 2005).

Nevertheless, the estimated algal biomass was considerably higher (at least one order of magnitude) in the planted if compared to the unplanted pond on 28 April 2005.

This can be explained with the obvious observation that algae are the dominant (virtually only) plant food source in the unplanted pond.

4.4.3. Integration of SUDS into Urban Planning

Flood protection management and the recreational value of urban landscapes can be improved at the same time by integrating SUDS (in contrast to conventional drainage) into the urban planning and development processes. Recreational activities may include watching ornamental fish such as *C. auratus* and birds, walking, fishing, boating, holding picnics and teaching children about aquatic ecology [Butler and Davies, 2000 and Scholz *et al.*, 2005).

The confidence of town planners towards SUDS and public acceptance of infiltration ponds can both be increased by correct dimensioning of sustainable systems (Zheng *et al.*, 2005) to avoid flooding, enhance water pollution control by using a robust pre-treatment train (e.g., silt trap, constructed wetland and swale) (D'Arcy and Frost, 2001) and control algae by biological (e.g., *C. auratus*) and not chemical (e.g., copper sulphate) means (Scholz, 2004 and Zheng *et al.*, 2005). Moreover, stormwater can be reused for watering gardens and flushing toilets as part of an urban water resources protection program (Butler and Davies, 2000, Scholz, 2004 and Zheng *et al.*, 2005).

4.4.4. Urban Water Hygiene

The issue of urban water hygiene requires consideration. Runoff water could sweep some animal faeces into SUDS. Particularly dog faeces being carried in by floodwaters are a problem in urban environments despite local government efforts to encourage dog owners to scoop up droppings (Mckie, 2005).

Preliminary findings indicated that the additional nutrient load was very small in comparison to the background load (e.g., leaves and soil), and that no accumulation of bacteria in the system was detectable. After this experiment, *C. auratus* would be introduced to SUDS sites within cities such as Glasgow and Edinburgh.

4.5. Combined Bio-filtration, Stormwater Detention and Infiltration System Treating Road Runoff

4.5.1. Water Balance

By summing up the flows calculated with the SWMM for the period between 16/10/07 and 14/03/08, it was possible to assess the water balance within the catchment. The percentage figures do not total 100%. This was a result of the SWWM continuity errors and the 2.1% of water remaining in the tank on 14/03/08. Evaporation from the gravel filter accounted for a large volume of the water leaving the catchment.

Long hydraulic lag times were associated with the presence of the gravel filter and ponding, which often occurred over long periods of time, leading to increased evaporation. Of the water entering the detention tank, approximately 50% of the treated runoff was infiltrated and 50% overflowed into the sewer system. The water balance of the catchment is summarized in Fig 27.

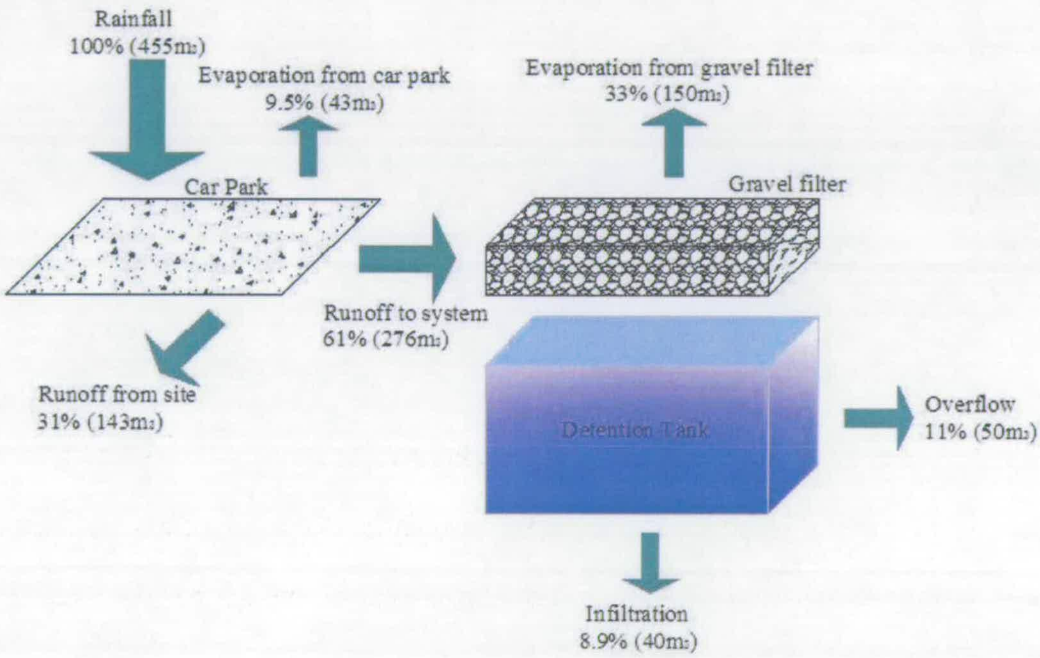


Fig 27. Water balance of the catchment.

4.5.2. System Hydrograph

Hydrographs were plotted for storms of varying intensity and duration to give an overview of the system's response to different rainfall events. In the case of the longest observed rainfall event during the period of this study, the storm event lasted for 208 hours in total. However, the event comprised a number of storms taking place in quick succession. Mean intensities ranged between 0.2 and 3.64 mm h^{-1} and durations between 1 and 14 h were recorded. A high volume of rainfall fell at the beginning of the event, and was matched quickly with an equal volume of runoff.

The gravel filter retained the runoff for a considerable length of time and subsequently released it as slow and steady outflow to the tank. The outflow from the tank mimicked the outflow from the filter, with peaks occurring almost simultaneously throughout the storm event. The tank retained 58% of the rainfall volume, thus decreasing the volume of runoff from the site significantly. The advantage of the system can clearly be seen from Figure 4, as large peaks of runoff were evened out and water exited the catchment to the sewer system consistently at a low flow.

4.5.3. Water Treatment Performance

All filter removal rates were found to be high ranging from 66% for nitrate-nitrogen to 96% for total solids. On the contrary, with the exception of the biochemical oxygen demand and suspended solids removal rates in the tank, all other parameters had negative removal efficiencies. Ortho-phosphate-phosphorus concentrations considerably increased in the tank; removal rates of -208% were noted. Despite the generally poor performance of the tank, the proficiency of the filter ensured that removal rates for the system as a whole were all positive. The lowest system removal rate was recorded for total dissolved solids (28%) and the highest for biochemical oxygen demand (98%). Despite of problems on the hydraulic side, the filter was obviously providing a valuable function with respect to water quality improvements.

Chapter 5

Conclusion

5.1. Concluding Remarks

A summary of the concluding remarks is given below:

In the first study it was concluded that a combination of infiltration trenches or swales with ponds or underground storage were the most likely SUDS options for the majority of the demonstration areas. Soil contamination issues were considered when selecting SUDS because heavy metals such as lead and zinc can cause environmental health problems.

In the second study a decision support model was developed to present urban developers with a novel realistic tool to assess the suitability of different SUDS techniques for a particular site with and without applying their own judgement. The SUDS model (http://www.see.ed.ac.uk/research/IIE/research/environ/uw12.html) can be applied for other urban sites with similar characteristics to those in Glasgow and Edinburgh.

In the third study a multiple regression analyses showed a relatively successful prediction of the biochemical oxygen demand, and total suspended solids for most systems but due to a relatively weak correlation between the predictors, and both microbial indicators, multiple regression analyses were not applied for the prediction of intestinal enterococci, and total coliform colony forming units. However, artificial neural network models predicted microbial counts relatively well for most detention systems.

The forth study an analysis of variance indicated that the systems were significantly different in terms of most of their treatment performance variables. Findings also show that the introduction of *C. auratus* to the planted and the unplanted infiltration ponds resulted in the improvement of most water quality variables despite of a deterioration of almost all common inflow water quality variables based on an annual comparison.

In the fifth study the assessment of the system's hydrological efficiency specified mean lag times of 1.84 h and 10.6 h for the gravel filter and the entire system, respectively. Mean flow volume reductions of 70% and mean peak flow reductions of 90% were reached compared to conventional drainage. The system showed potentials in removal efficiencies for biochemical oxygen demand (77%), suspended solids (83%), nitrate-nitrogen (32%) and ortho-phosphate-phosphorus (47%). The most important removal processes were believed to be biological degradation (predominantly within the gravel ditch), sedimentation and infiltration.

5.2. The Glasgow Sustainable Urban Drainage System Management Project

A survey of 57 sites within 46 areas of Glasgow shows that it is feasible to implement different SUDS techniques throughout Glasgow. The likely contribution of future SUDS to the overall catchment dynamic of representative demonstration areas has been assessed. The preliminary designs will help to understand the challenges of catchment management and diffuse pollution. The implementation of SUDS will help to relief the local sewer system, and subsequently allows for more regeneration activities to take place.

Characteristics that determine the suitability of a site for the implementation of SUDS have been identified. Representative areas and sites that are suitable for different representative SUDS techniques have been identified qualitatively and quantitatively. A SUDS decision support key and matrix that are adaptable to different cities have been proposed. The matrix can be used as part of a decision support model in the future. Seven entirely different SUDS demonstration areas that are representative for both different sustainable drainage techniques and different types of areas available for development, regeneration, and retrofitting of SUDS within Glasgow have been identified. Design and management guidelines for demonstration sites that should be constructed to inform and educate the public, developers, and politicians have been proposed. Belowground storage tanks and ponds linked with swales and infiltration trenches have been identified as the most useful sustainable drainage techniques for large sites within Glasgow.

Concerning the case studies, the proposed drainage system for the Belvidere Hospital area is dominated by a network of swales draining into a large attenuation and detention pond. The runoff will ultimately drain from the pond into a nearby river. In comparison, the drainage of the Celtic FC Stadium area is dominated by two large belowground detention tanks beneath car parks. The runoff, after a considerable lag period, will ultimately drain into the sewer after the risk of flooding has gone down.

Furthermore, a brief cost–benefit analysis has shown that the capital costs for the proposed SUDS solutions are likely to be similar to the costs for a comparable traditional drainage system. However, a SUDS solution would be preferable, if it could be integrated into the area reserved for green space.

The soils for both selected case studies were contaminated predominantly with lead and zinc. Moreover, hot spots of nickel contamination were detected in the east of the Celtic FC Stadium area. In comparison, organic contamination was insignificant.

The application of the SUDS decision support matrix has identified that unlined SUDS structures such as swales can only be implemented if the risk of runoff being contaminated by metal leaching is eliminated. Large quantities of top soil therefore require removal before construction work on residential properties can commence to avoid environmental and water pollution as well as potential health problems for the residents.

5.3. The Edinburgh Sustainable Urban Drainage System Management Project

A survey of 103 sites in Edinburgh showed that it was feasible to implement diverse SUDS techniques throughout Edinburgh. Characteristics that determine the suitability of a site for the implementation of SUDS were generally identified. A practical SUDS decision support tool was developed that would provide the practitioner a novel tool to assess the suitability of diverse SUDS techniques for a particular site with and without applying their own expertise.

A general singular SUDS feasibility matrix was outlined. Single SUDS techniques suitable to be joined to form a combination were identified. A feasibility matrix for these combinations where two different SUDS techniques are combined was drawn. A rating system which identifies the best SUDS solution for a site was developed. The best SUDS solutions for the Edinburgh were identified.

Seven completely different SUDS demonstration sites/areas that were representative for both different SUDS techniques and different types of areas available for development, regeneration and retrofitting within Edinburgh were shortlisted. The representative areas suitable for SUDS were qualitatively and quantitatively defined. A detailed design was provided for one site (Peffermill Industrial Estate). The drainage of the Peffermill Industrial Estate should be dominated by a network of swales draining into a large detention pond/basin. The runoff would eventually drain into a close by river. By establishing an integrated SUDS system on the Peffermill site, a solution was provided which met the landscape and technical criteria set in the Edinburgh and Lothians Structure Plan. The solution had clear advantages compared to a conventional system in that surface runoff was detained resulting in a reduction in the peak flow of the Braid Burn.

The probable contribution of future SUDS to the overall catchment dynamic of the area was assessed. The initial design would help to understand the challenges of holistic catchment management and diffuse pollution. Open water surfaces would have a positive effect on the appearance and outdoor well-being of the site.

Design and management guidelines for SUDS sites were identified. It was revealed that instead of routing urban runoff directly into a piped system, SUDS could provide alternative approaches which employ the natural drainage patterns of a catchment and on-site infiltration into the soil. The study indicated that where infiltration was not possible, the development of natural drainage patterns would offer a range of opportunities for conservation, recreation and amenity, as well as providing basic flood and pollution control.

A wide range of SUDS techniques could be implemented for most new and redeveloped sites in Edinburgh to lessen environmental impact form surface water drainage. Where retrofitting sites with SUDS were considered, results showed that there were numerous opportunities to install SUDS structures, and that doing so, could have positive effects on reducing stormwater runoff. Green roofs, infiltration basins, dry swales in conjunction with infiltration basins and green roofs linked with soakaways were identified as the most practical sustainable drainage techniques for sites within Edinburgh.

5.4. Assessing Stormwater Detention Systems Treating Road Runoff with an Artificial Neural Network

An analysis of variance showed significant differences between different experimental system performances in treating concentrated road runoff. Systems containing turf showed better biochemical oxygen demand, and suspended solids removal performances in comparison to less complex systems without turf. However, the assessment was unclear with respect to microbiological indicator variables.

Multiple regression analyses indicated a relatively successful prediction of the biochemical oxygen demand but unsuccessful predictions of both total coliform, and intestinal enterococci counts. However, artificial neural network models predicted both total coliform, and intestinal enterococci counts relatively well.

The artificial neural networks successfully predicted total coliform, and intestinal enterococci counts for the inflow water. The models were highly successful in predicting microbial counts for most systems. Predictions resulted in mean squared errors close to zero.

The results of this study show that the artificial neural network models developed for the prediction of the total coliform counts, and the intestinal enterococci counts have performances consistent with other findings reported in the literature. However, the relatively low R^2 values reported for some systems, and more specifically for predicting intestinal entercocci counts in the densely planted system five indicate the difficulty in identifying the necessary explanatory variables to characterize a large percentage of the variability observed in the microbial dataset. In cases where water quality standards are observed for total coliform and intestinal enterococci counts, artificial neural networks provide a good modeling technique to predict a potential violation.

The model could be applied outside the experimental setup for similar problems. The main condition is that the boundary conditions are comparable. Otherwise, the model would require retraining.

5.5. Stormwater Infiltration Systems for Road Runoff Contaminated with Organic Matter Including Dog Faeces

During summer, the five-day at 20°C biochemical oxygen demand concentrations ccasionally exceeded the UK and US thresholds of 20 and 30 mg/l, respectively, for secondary treated wastewater. The suspended solids concentrations were frequently above the UK threshold of 30 mg/l, for secondary treated wastewater. The SUDS system showed signs of higher microbiological contamination during warmer periods of the year.

The ortho-phosphate phosphorus and ammonia-nitrogen concentrations reduced towards the end of the system operation, and nitrate-nitrogen concentrations were significantly lower in the infiltration ponds than in the detention tank.

An analysis of variance indicated that there were significant differences between most parts of the systems in terms of their treatment performance. However, the presence of macrophytes did not make a significant difference in enhancing the water quality. The infiltration ponds performed relatively well in reducing the amount of organic pollution and artificial microbial contamination. Further research is required to improve the SS reductions after system set-up and to enhance the treatment of phosphate within the detention tank.

5.6. Combined Bio-filtration, Stormwater Detention and Infiltration System Treating Road Runoff

The system's hydraulic efficiency during the representative sample period between 16/10/07 and 14/03/08 varied considerably, depending on the extremity of the rainfall events. For low and moderate storms, the system coped well resulting in a mean benefit factor of 73%. The mean peak flow reduction was even higher (80%), demonstrating the system's efficiency at reducing runoff from the site.

Out of the 68 events resulting in runoff, only 36 were retained within the system. Mean lag times of 3.61 and 7.98 h were observed for the filter and infiltration system, respectively. Since 2006, lag times for the filter doubled despite of more precipitation falling in the selected observation period in 2007 and 2008. This was most likely a result of clogging of the filter, which was accounted for in the Stormwater Management Model by the significant decrease in hydraulic conductivity. The infiltration rate in the detention tank decreased since 2007, probably as a consequence of sediment build-up at the base of the tank and partial clogging of the geotextile.

During extreme events, the system did not perform as well. For example, between January and March 2008, rainfall was much higher than would usually be expected, with numerous storms of both high volume and long duration. By modelling the system with the Stormwater Management Model, it was determined that runoff must be leaving the site by some other means other than either infiltration or tank overflow. During the observation period, approximately 143 m^3 of water was thought to have left the site unaccounted for. Pooling has been observed at the gravel filter inlet on many occasions. The filter appeared to be frequently clogged at the inlet end, and thus as a result did not appear to be processing the runoff as intended. It is probable that the water was ponded at the inlet and subsequently overflowed the kerbstone, leading to infiltration via the surrounding meadows.

Thus it can be concluded that the system performed well for low and moderate rainfall events. However, the system struggled to cope with extreme rainfall events primarily because of the gravel filter. More frequent maintenance of the gravel filter could result in much better efficiency during stronger storms.

Removal rates for both inlet and outlet sampling points of the filtration trench were found to be high. On the contrary, except for biochemical oxygen demand and suspended solids, all sampling points in the tank showed negative removal rates. Although the detention tanks showed evidence of insufficient performance regarding pollutant removal, the very high performance of the filter raised the entire system's removal rates as they were all positive.

5.7. Recommendations for Future Research

The decision support tool showed satisfactory results for the Glasgow and Edinburgh sites. Nonetheless, there are several improvements or alterations possible for data fields. It would also be beneficial to include additional data fields which would inevitably increase the ability of the tool to identify suitable SUDS techniques for various sites (e.g. inclusion of public acceptance measure and more detailed ground water and contamination level data)

The findings of this research have significant implications on the future design, operation monitoring and management of stormwater detention and infiltration systems for urban runoff treatment since the experimental rigs investigated in this study were highly efficient and their performances were stable in a cold climate. Therefore, costs can be reduced by choosing alternatives on filter material, aquatic plants and water sampling.

The physical, chemical and microbiological pollutant removal mechanisms in stormwater detention and infiltration systems were identified during the course of this research. The contribution of vegetation cover (willows) and filter media to the detention/infiltration systems was also assessed. However, a comprehensive assessment of filter media and the flow patterns in such systems would be beneficial. In addition, a more detailed investigation on the effect of vegetation on the overall pollutant removal in stormwater detention/ infiltration systems is strongly recommended.

Further analyses on microbial contamination are required to identify the fate of different types of microbes in stormwater detention/infiltration systems.

An investigation for speciation and distribution of heavy metals in the sediments of belowground stormwater detention tanks may also enhance the understanding of the mobility and retention mechanism of such metals in stormwater detention/infiltration systems. As a result, heavy metals retention capacity of such systems can be assessed.

Further detailed research on the health of *C. auratus* is required to investigate whether *C. auratus* is capable of coping with the additional nutrient load and if there is a possibility for potentially dangerous build up of *E. coli* from excrements in urban runoff.

The maintenance-related problems of stormwater detention systems seem to be neglected in the literature and therefore further studies are suggested.

5.7.1. Applied Research Recommendations

Animal dropping in urban areas seems to be the major source of stormwater biological contamination. Therefore, more in depth research is required to investigate the magnitude and effects of such contaminations on the environment in the long-term. This research initiated a novel approach towards this problem but there is still a need for comprehensive studies on the fate of different types of microbial populations in stormwater runoff treated with detention and infiltration facilities.

A brief survey was conducted during this research to determine the approximate amount of dog faeces existing per square meter in Edinburgh city. Therefore, as there is yet no evidence of such surveys in the literature, it is recommended that a more detailed investigation should be undertaken to address this issue.

The effect of vegetation on a stormwater bio-infiltration device was explored during this research for the first time. While the results showed signs of significant improvements in water quality of the system, it is recommended that similar studies should be considered to investigate the removal efficiency potentials of different plant species on stormwater runoff.

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Appendix

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Appendix 1

The Glasgow Sustainable Urban Drainage System Management Project: Case Studies (Belvidere Hospital and Celtic FC Stadium Areas)

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Source of work: The Glasgow SUDS Management Project

Running title: **Glasgow: sustainable urban drainage systems**

Abstract

'The Glasgow Sustainable Urban Drainage System (SUDS) Management Project' satisfies the first phase of the 'Glasgow Surface Water Management Project'. This is Glasgow City Council's contribution to the Transformation of Rural and Urban Spatial Structure (TRUST) project, one of the European Union's (EU) inter-regional (INTERREG IIIB) funded research projects. The remit of this EU project comprises also other representative regions in Europe. The project shows also how SUDS can contribute to the overall catchment dynamics of cities such as Glasgow, ultimately relieving stress on the current predominantly combined sewer system. Fifty-seven sites within 46 areas of Glasgow were identified for investigation. A detailed soil chemistry analysis, a preliminary SUDS feasibility assessment and a desk study relating to historical planning issues that may be relevant for subsequent future development and regeneration options were undertaken. Detailed design and management guidelines were then drafted for selected representative demonstration areas (Belvidere Hospital and Celtic FC Stadium Areas) of high public and property developers interest, and education value. A combination of infiltration trenches or swales with ponds or underground storage were the most likely SUDS options for the majority of the demonstration areas. Soil contamination issues were considered when selecting SUDS because heavy metals such as lead and zinc can cause environmental health problems.

Key words: Glasgow; sustainable urban drainage system; Brownfield; attenuation; pond; underground storage tank; soil contamination; heavy metal

Introduction

Water Framework Directive and SUDS

The European Union's (EU) Water Framework Directive (Council of European Communities, 2000), which came into force on 23 October 2000, requires all inland and coastal waters to reach 'good status' by 2015. The Directive sets a framework that should provide substantial benefits for the long-term water quality management of waters. The implementation of sustainable urban drainage systems (SUDS) based on current guidelines (CIRIA, 2000; Jefferies *et al.*, 1999; McKissok *et al.*, 1999) in Glasgow can help preventing flooding from watercourses and sewer systems, and combined sewer overflows to spill untreated sewage into receiving watercourses such as rivers and canals during storms (DEFRA, 2000; Scholz, 2004). Furthermore, SUDS can help to reduce the impact of diffuse pollution on urban watercourses by promoting passive treatment (D'Arcy and Frost, 2001). However, metals may leach out of the soil of SUDS embankments during winter causing water quality problems (Scholz, 2004).

Transformation of Rural and Urban Spatial Structure (TRUST) project applied in Glasgow

In April 2000, the Commission of European Communities established a Community Initiative concerning trans-European co-operation, known as INTERREG IIIB. The INTERREG IIIB initiative relates to the whole of the European Union. One of the projects funded by this initiative is entitled Transformation of Rural and Urban Spatial Structure (TRUST). This project aims to develop new approaches to both spatial planning and land use to meet the challenges of continuing urbanisation, along with reducing economic loss and reduction in biodiversity through the development of integral management methods.

The theme of TRUST is based upon multi-functional water storage, integral surface water management, and public and stakeholder participation. Six different authorities and institutions throughout Europe are currently contributing to this project. They are in alphabetical order British Waterways (Watford, UK), Gewestelijke Ontwikkelingsmaat-schappij (Brugge, Belgium), Glasgow City Council (Glasgow, UK), Hoogheemraadschap van Scieland (Rotterdam, The Netherlands), Provincie Noord Holland (Haarlam, The Netherlands) and University of Osnabrück (Osnabrück, Germany).

Glasgow City Council's contribution to the TRUST project is known as the 'Glasgow Surface Water Management Project'. The project proposes innovative urban drainage recommendations. The first study output is the 'The Glasgow Sustainable Urban Drainage System Management Project', led by Dr Scholz and executed by the authors of this paper, and funded by The Royal Academy of Engineering and Glasgow City Council.

Background to case studies

The Belvidere Hospital site is located to the South of London Road (major road into Glasgow), and is owned by Kier Homes. It is in a prime development area due to its proximity to the Glasgow City centre and amenities such as parks, shopping centres, Celtic Park and the proposed national indoor sports arena. The southern border of the site is adjacent to the River Clyde.

The Celtic FC Stadium area is located to the North of London Road (see above), 2 km from the City Centre, in Glasgow's East End (also know as the Celtic Triangle). The area includes the Celtic FC Stadium (Celtic Park) to the East, visitor and coach car parking to the Southwest, and housing (partly under construction) to the Northeast. The West of the area is owned by Glasgow City Council.

Rationale, aims and objectives

The feasibility to implement different SUDS throughout Glasgow and their potential contribution to the overall catchment dynamics has been studied. The data should help in understanding the challenges of holistic catchment management, diffuse pollution, and the 'linking scales' in catchment management.

This project aims to come up with SUDS demonstration areas (case studies) that are representative for both different sustainable drainage techniques and different types of areas available for development and regeneration.

The objectives are to:

(i) Identify variables that determine the suitability of a site for the implementation of SUDS;

(ii) Identify suitable SUDS sites within Glasgow;

(iii) Classify qualitatively and quantitatively sites suitable for different SUDS technologies;

(iv) Outline both a general SUDS decision support key and matrix;

(v) Identify representative SUDS technologies for representative sites that could be used for demonstration purposes;

(vi) Provide detailed design and management guidelines, and a brief cost-benefit analysis for representative sites and representative SUDS techniques for information and education purposes.

(vii) Assess the soil contamination and the associated impact on environmental health.

Methodology and Experimental Protocols

Glasgow

Figure 1 is a map of Glasgow highlighting the spatial distribution of 46 areas (associated with 57 sites) that were identified as potentially suitable for the implementation of SUDS. Eight areas had the potential for more than one SUDS system and were therefore sub-divided into sub-areas (i.e. sites). Every effort has been made to investigate also areas currently represented only sparsely by discussing SUDS opportunities with planners employed by the Council.

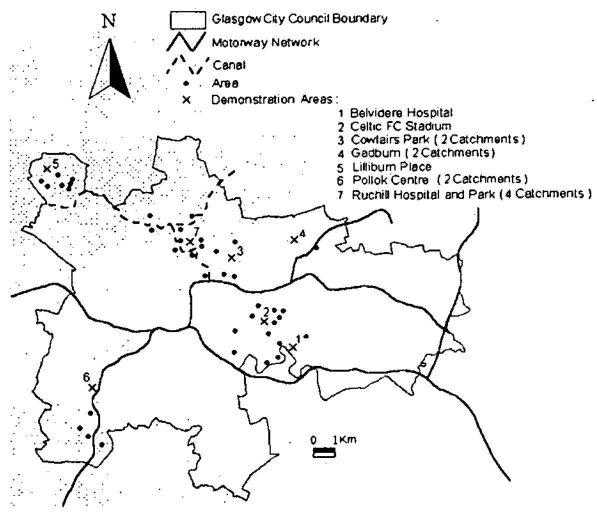

Figure 1. Indication of 46 potential areas comprising 57 sites for the implementation of sustainable urban drainage systems (SUDS). The SUDS demonstration areas have been highlighted.

Site classification

Fifty-seven sites were hierarchically classified (nine levels) according to their public acceptability, land costs, water supply, drainage issues, site dimensions, slope, groundwater table depth, fragmentation of ownership and ecological value. The classification was based on expert water-engineering understanding, rather than on statistical evaluation, and account for flexibility in selecting (numerical) thresholds (e.g. estimated land cost). Moreover, this classification should be used as a general framework that supplements detailed frameworks and management guidelines dealing with specific regeneration issues such as leaching of metals (Kossen *et al.*, 2002).

SUDS decision support key

Sustainable urban drainage should not cause any public health problems, avoid pollution of the natural environment, minimise the use of resources, operate in the long-term and be adaptable to change in requirements (Butler and Parkinson, 1997). Taking this statement into consideration, the following list of criteria for defining SUDS options and a corresponding summary matrix (Table 1) have been proposed:

(1) Runoff (low or high). The site has to be associated with a potential source of water (e.g., car park runoff) that results in sufficient runoff (to be defined on a case by case basis).

(2) Catchment size (specified for individual SUDS options). The site needs to have sufficiently large dimensions (e.g., width ≥150 m and length ≥300 m).

(3) Area suitable for SUDS (specified for individual SUDS options). The site has to be acceptable for development, regeneration or retrofitting to Glasgow City Council, developers and the wider public (e.g., Greenfield and Brownfield areas). The site should also be associated with a separate area to which water can drain (e.g., canal or river).

(4) Serious soil contamination (yes or no). The site should not be associated with major soil contamination issues.

(5) Land value (low, medium, high or not applicable). The land costs should preferably be not too high (e.g., <£200/m^2) before development or regeneration work has commenced.

(6) Fragmentation of ownership (yes or no). The site should preferably be owned by only a few individuals or organisations (e.g., < 5 parties).

(7) High groundwater level (yes, no or not applicable). The site should preferably be associated with a low groundwater table (e.g., groundwater level > 2 m below ground level).

(8) Sufficient channel slope (yes, no or not applicable). The site should have a sufficient slope (e.g., ≥1 in 50 m) to enable conveyance structures to function properly. However, the site should not be too steep to make three-dimensional SUDS features too expensive.

(9) Potential of high ecological impact (yes, no or not applicable). The site should be of potentially high ecological impact in the future, but not during the planning phase (e.g., not a SSSI site).

(10) Soil infiltration (low, high or not applicable). The site should have sufficiently high soil infiltration, if filtration is considered to be desirable for the proposed SUDS structure.

The representative SUDS demonstration areas have been selected based on these criteria (see above and Table 1). Only seven areas suitable for SUDS implementation are not represented by the selected demonstration areas. However, it has to be emphasised that the selection is rather qualitative than quantitative considering that most selection criteria do not require a numerical assessment. It follows that the SUDS classification is similar to an expert system, and not to a statistically unbiased assessment that would not be suitable in this case anyway because of the lack of numerical information such as land value (e.g., recognising also the future potential after regeneration).

Fieldwork activities

Soil was sampled twice: selected samples were initially taken at a few locations where major SUDS structures were likely to be implemented. Composite samples were taken at 10 cm depth intervals within trenches of up to approximately 55 cm depth.

Further samples were taken at locations that are part of a proposed wider SUDS structure and that were located most closely to the nearest node of a randomly placed 50 m × 50 m equally spaced sampling grid. Only one sample at 50 cm depth per sampling site was taken during a second expedition.

If a sampling location was not acceptable (e.g., below tarmac or a house), an alternative representative sampling location was determined up to 5 m (if not stated otherwise below) away from the original location. However, if no sampling locations deemed to be appropriate, the location was not sampled and a 'not accessible' entry was located on the map showing the sampling strategy and locations.

Analytical work

The soil recording and pre-treatment before analysis was carried out in agreement with British Standards (British Standard Institute, 1999a, 2002). The determination of particle size distribution was also carried out according to British Standards (British Standard Institute, 1999b).

Composite samples were collected and stored at -10°C prior to analysis. After thawing, approximately 2.5 g of each soil sample was weighed into a 100 ml digestion flask to which 21 ml of hydrochloric acid (strength of 37%, v/v) and 7 ml of nitric acid (strength of 69%, v/v) were added. The mixtures were then heated on a Kjeldahl digestion apparatus (Fisons, UK) for at least 2 h. After cooling, all solutions were filtered through a Whatman Number 541 hardened ashless filter paper into 100 ml volumetric flasks. After rinsing the filter papers, solutions were made up to the mark with deionised water. The method was adapted from the section 'Nitric Acid-Hydrochloric Acid Digestion' (American Public Health Organisation, 1995).

An Inductively Coupled Plasma Optical Emission Spectrometer (ICP-OES) was used for the analysis of metals and other heavy elements. Total concentrations of elements in filtered (Whatman 1.2 μm cellulose nitrate membrane filter) samples were determined by ICP-OES using a TJA IRIS instrument (ThermoElemental, USA). Multi-element calibration standards with a wide range of concentrations were used and the emission intensity measured at appropriate wavelengths.

Concerning the analysis of major nutrients, 2 ml sulphuric acid (strength of 98%, v/v) and 1.5 ml hydrogen peroxide (strength of 30%, v/v) were used as an extraction media (Allen, 1974). Approximately 0.1 g of each dried sample and the associated digestion media were placed in a tube and heated at 320°C for 6 h. Aliquots were taken and digests were made up to 100 ml with distilled water.

For analysis of total nitrogen (Ntotal), the following reactions and procedure was adopted: Ammonium (present in the digest) reacts with hypochlorite ions generated by alkaline hydrolysis of sodium dichloroisocyanurate. The reaction forms monochloroamine, which reacts with salicylate ions in the presence of sodium nitroprusside to form a blue indephenol complex. This complex is measured colorimetrically at 660 nm using a Bran & Luebbe autoanalyser (model AAIII).

For analysis of total phosphorus (Ptotal), the following reactions and procedure was adopted: Ortho-phosphate (present in the digest) reacts with ammonium molybdate in the presence of sulphuric acid to form a phosphomolybdenum complex. Potassium antimonyl tartrate and ascorbic acid are used to reduce the complex, forming a blue colour, which is proportional to the Ptotal concentration. Absorption was measured at 660 nm using a Bran & Luebbe autoanalyser (model AAIII).

For the analysis of total potassium (Ktotal), the digest was analysed by a flame atomic absorption spectrometer (Unicam 919, Cambridge, UK) at a wavelength of 766.5 nm and with a bandpass of 1.5 nm. Standards were prepared in 100 ml flasks using 2 ml concentrated sulphuric acid and 1.5 ml hydrogen peroxide (30% v/v) and made up to mark with de-ionised water. Caesium at a concentration of 100 mg/l was added to both standards and digests to overcome ionisation.

Sub-samples of 3 ± 0.1 g of field moist soil were mixed with an excess of sodium sulphate (Analytical Grade, Fisher, UK) to make it 'free flowing' and the resulting mixture extracted in 10 ml of HPLC grade dichloromethane (Fisher, UK) in an ultrasonic bath (Model XB14, Grant instruments, Cambridge, UK) for 10 minutes. After agitation, samples were filtered through 0.45 um nylon syringe filters (Qm$_X$ Laboratories Limited, Thaxted, UK).

The sample extracts were scanned for the presence of organic contaminants by HP 6980 gas chromatograph coupled to HP 6973 mass spectrometer. A 4 μl aliquot of each sample was injected in the splitless mode onto a 30 m HP5MS fused silica column directly coupled to the ion source of an HP 6973 mass spectrometer.

The mass spectrometer was run in the scanning mode with a mass range of 50 to 700. Identification of the peaks on the total ion chromatograms was made using libraries of pre-installed databases of reference spectra. An initial peak width and initial threshold values were set to identify significant peaks.

Data analysis and software used

The data analysis was carried out using Microsoft Excel, and statistical methods outlined elsewhere (Fowler and Cohen, 1998) were applied. ArcView was used to draw design proposals.

Belvidere Hospital area case study

The Belvidere (not Belvedere as usually read) Hospital area is located approximately Longitude 4°12' West and Latitude 55°51' North. The area has been cleared of all surface structures for new housing, with one remaining former hospital building, which is a Grade B Listed Building. However, parts of the area contain residual housing foundations below the current ground level. Nevertheless, the overall topography of the site is even. Future development of this site for housing will require the removal of all residual foundations and asphalted areas (Figs. 2 to 5).

Figure 2. Belvidere Hospital area: site photograph taken on 14 May 2004.

Figure 3. Belvidere Hospital area: artist impressions of proposed site development (pencil drawing and computer animation).

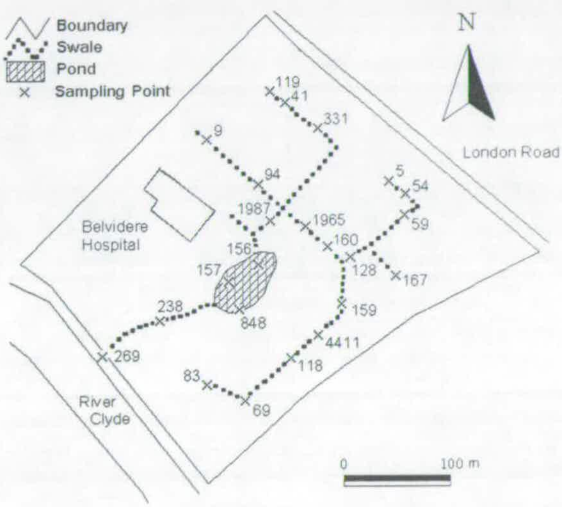

Figure 4. Belvidere Hospital area: spatial distribution of lead (mg/kg dry weight) at 50 cm depth on 5 July 2004.

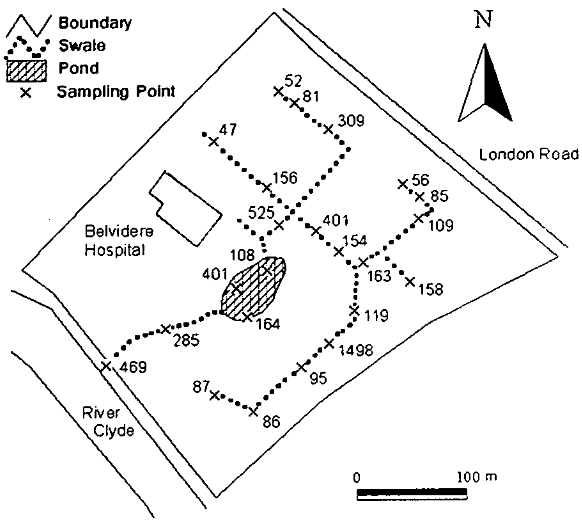

Figure 5. Belvidere Hospital area: spatial distribution of zinc (mg/kg dry weight) at 50 cm depth on 5 July 2004.

The main entrance driveway of the original hospital still exists with two large semi-vegetated areas (mainly rows of tall trees) flanking both sides. The remaining building on the area is situated to the West. To the South of the building is a steep embankment covered in dense woodland. The slope increases approximately from East to West, and is at its maximum 60°. At the base of this embankment (not within the area boundary marked by a 3 m high corrugated iron fence), runs a public walk and cycle path along the River Clyde. The height difference from the crest of the embankment down to the walkway is approximately 11 m. However, this area is likely to remain unaffected by any building and road construction works due its potentially high ecological and amenity value.

A desk study concerning the Belvidere Hospital proved to be unrewarding as there were no historical documents held by Glasgow City Council pertaining to this area. However, the area is known to have been a hospital for approximately 100 years, and during this time the hospital grounds were subjected to the cleaning of hospital equipment and might be contaminated with diffuse hospital waste.

Celtic FC Stadium area case study

The Celtic FC Stadium Area is approximately bordered by Janefield Street in the Northeast, Stamford Street in the Northwest, Dalriada Street in the Southeast and London Road in the South. A major part of the area in the West is used as a car park. The Celtic stadium is located in the Southeast of the demonstration area.

The immediate Celtic FC Stadium Area is located approximately Longitude 4°13' West and Latitude 55°51' North. The Celtic Park stadium has capacity for approximately 60,000 spectators and occupies a prime location in the heart of Glasgow's East End. On home match days, the stadium is generally filled to capacity (approximately 26 times per year between July and May) and this volume of visitors to the area clearly impacts on any integrated SUDS in the future.

Previous site investigations in this area show that the site is underlain by sandstones, siltstones and mudstones with seams of coal belonging to the Lower and Middle Coal Measures of the Carboniferous System. The natural superficial deposits are indicated on glacial maps to be alluvial clay and silt, partly overlain by made ground. The total thickness of superficial deposits is indicated to be between 20 and 30 m.

The 1980 investigation of the areas as recorded in the Glasgow City Council's Geodatabank showed the general succession to be made ground with a thickness between 1.7 and 10.8 m and clay with sand bands with a thickness of at least 2 m (Glasgow City Council, 1995). The granular constituents of the made ground are in a generally medium dense state of compaction but the cohesive constituents are generally in a soft or very soft state (Glasgow City Council, 1980).

The ground water level in this area is at a depth between 7 and 8 m based on borehole data. Seepage of water was recorded in a couple of boreholes but the report indicates that groundwater did not gather during the time of boring in the remaining boreholes. However, it is possible that pockets of perched groundwater may occur anywhere in made ground (Glasgow City Council, 1980).

The northern parts of the site, the areas located between Janefield Street and Stamford Street are currently under redevelopment. There is a coach parking located in the East of the area that is expected to be retained. There are a number of occupied and unoccupied low standard housing blocks in the West. An older housing estate will be demolished, and the site will subsequently be redeveloped.

The East End regeneration route will be located to the West of the demonstration area. It follows that large underground storage facilities would be required to attenuate highway runoff in the future.

Results

SUDS and soil quality

The outline of the SUDS decision support key and the classification of all sites visited during the exploratory stage of this project was conducted. The key may be used in combination with Table 1 outlining a SUDS decision support matrix.

Table 1. **Sustainable urban drainage system (SUDS) decision support matrix.**

	Runoff	*Catchment size (m^2)*	*Area of suitability for SUDS feature*	*Serious contamination*	*Land value*	*Owner-ship fragmented*	*High ground-water level*	*Sufficient channel slope*	*Potential of high ecological impact*	*Soil infiltration*
Wetlands	High	>50000	>5000	No	<2	No	N/A	N/A	Yes	N/A
Ponds	High	>15000	>50	No	<3	Yes	N/A	N/A	N/A	N/A
Lined ponds	High	>15000	>50	Yes	<3	Yes	N/A	N/A	N/A	N/A
Infiltration basin	High	>15000	>50	No	<3	Yes	No	N/A	N/A	High
Swale	High	N/A	>200	No	<3	No	No	Yes	N/A	N/A
Shallow swale	High	N/A	>200	No	<3	No	Yes	Yes	N/A	N/A
Filter strip	High	>15000	>600	No	<3	Yes	No	Yes	N/A	High
Soak-away	Low	>3000	>200	No	<3	Yes	No	Yes	N/A	High
Infiltration trench	Low	>3000	>50	No	<3	No	No	Yes	N/A	High
Permeable pavement	Low/ High	N/A	N/A	No	N/A	Yes	N/A	N/A	N/A	High
Under-ground storage	Low/ High	N/A	>40	Yes	N/A	Yes	N/A	N/A	N/A	N/A
Supplementary water playground	Low	>200	>10	No	N/A	Yes	Yes	N/A	N/A	N/A

Land value: 1 = low (<£100/m²), 2 = medium (≥£100/m² and ≤ £200/m²), 3 = high (>£200/m²); N/A = not applicable.

This matrix has been tested with the exploratory data set collected during the site visits, and Table 2 summarises the outcome of the application of this tool. The findings for SUDS structures in Table 2 are based on the assumption that the soil contamination issues for all sites have been identified during the planning phase, and that contaminated soil will be removed wherever relevant soil contamination guidelines and/or the introduction of unlined SUDS structures require such a measure.

Table 2. **Sustainable urban drainage system (SUDS) options based on the SUDS decision support matrix (Table 1).**

Area	*Catch-ment*	*Wet-land*	*Pond*	*Infiltra-tion basin*	*Swale*	*Infiltra-tion trench*	*Soak-away*	*Filter strip*	*Perme-able pave-ment*	*Under-ground storage*
Belvidere Hospital	Entire area		XXX	X	XX	X			XX	X
Celtic FC Stadium	Entire area					X	X		XX	XXX
Cowlairs	North		XXX	X	XX	X	X	X	XX	X
Park	South		XXX		XX		X	XX	XX	X
Gadburn	North	X	XXX		XX	X			XX	X
	South	XXX	XX		XX				XX	X

Lillyburn Place	Entire area	X			XXX	X		X	
Pollok	West			XX	X			XX	XXX
Centre	East	X		XXX	XX	X		XX	X
Ruchill Hospital	North-east			XX				XX	XXX
and Park	South-east			XX				XX	XXX
	South	XXX		XX				XX	X
	West	XXX		XX	X	X	X	XX	X

X = possible option; XX = recommended option; XXX = predominant SUDS design feature.

Concerning nutrients and heavy metals, Table 3 summarises the soil quality for the most important nutrients and metals at 10 cm depth within Glasgow. Table 3 allows the reader to compare the contamination for selected demonstration sites with the average contamination for the whole of Glasgow.

Table 3. **Major total nutrients and major heavy metals (mg/kg dry weight): comparison of soil quality at 10 cm depth during the exploratory investigation of 57 sites.**

Area	*Site*	*N*	*P*	*K*	*Al*	*Cr*	*Cu*	*Fe*	*Mn*	*Ni*	*Pb*
Belvidere	North	1951	771	7601	13173	56	8	18100	1169	23	478
Hospital	South	351	391	5492	6688	94	10	29724	301	17	35
Celtic FC	North	1991	815	2290	7874	78	30	30783	528	12	103
stadium	East	724	615	3496	10695	112	88	29566	570	444	346
Cowlairs	North	1476	384	3817	4033	908	47	19053	526	29	59
Park	South	625	841	8963	15998	23	10	35441	476	33	46

Gadburn	South	2283	554	3522	9330	74	31	21312	339	27	85
Lillyburn Place	West	124	213	1607	4839	66	14	24883	374	19	51
Pollok Centre	West	524	290	9260	13156	147	72	28847	548	114	145
Ruchill	North east	1840	568	7813	3125	13	30	22065	416	27	170
Hospital	East	554	280	3243	2765	7	22	15809	483	17	374
and	Centre-east	2308	467	3393	3653	15	41	21629	283	23	451
Park	South	505	308	4315	9131	77	34	24688	594	25	1307
	North west	2412	716	3884	11507	78	37	23606	311	30	194
	Park	1663	575	7025	4515	21	33	33096	504	35	298
All sites areas	for all	1612	605	4562	12538	96	72	27375	485	34	198

Moreover, Table 4 shows the major nutrients and heavy metals at different soil depths during the exploratory investigation. The individual contamination profiles can be compared with the average contamination profiles for Glasgow (Table 4).

Table 4. **Major total nutrients and major heavy metals (mg/kg dry weight): comparison of soil quality at different depths during the exploratory investigation.**

Area	*Depth*	*Count*	*N*		*P*		*Pb*		*Zn*	
	(cm)		*Mean*	*SD*	*Mean*	*SD*	*Mean*	*SD*	*Mean*	*SD*
Belvidere Hospital	10	2	731	813.7	423	241.5	145	222.4	69	53.2
	20	2	801	781.7	397	251.5	157	215.7	80	53.1
	30	2	933	716.4	450	248.2	222	200.8	110	60.4
	40	1	418	-	402	-	117	-	57	-
	50	1	453	-	316	-	78	-	77	-
Celtic FC Stadium	10	2	1357	896.2	715	141.3	225	171.8	253	166.3
	20	2	1160	626.5	666	299.6	239	28.3	281	20.2
	30	2	1373	471.3	1072	188.3	809	157.3	692	158.9
Cowlairs Park	10	2	1102	412.2	653	301.5	55	15.4	82	11.8
	20	2	1779	1066.8	621	281.1	89	63.5	104	36.5
	30	2	1786	1058.8	459	174.1	106	67.6	120	44.1
Gadburn	10	1	2283	-	554	-	85	-	85	-
	20	1	9937	-	919	-	236	-	141	-
	30	1	5156	-	944	-	187	-	86	-
	40	1	5916	-	1101	-	181	-	121	-
	50	1	3740	-	1026	-	369	-	437	-

Lillyburn Place	10	1	124	-	213	-	51	-	45	-
	20	1	156	-	581	-	41	-	97	-
	30	1	219	-	629	-	55	-	90	-
	40	1	213	-	761	-	85	-	91	-
Pollok Centre	10	1	534	-	290	-	145	-	121	
	20	1	688	-	262	-	153	-	91	
	30	1	381	-	178	-	14	-	95	
	40	1	336	-	254	-	145	-	91	
Ruchill	10	6	1547	836.3	486	168.4	466	425.5	216	115.2
Hospital	20	6	1330	1559.6	505	400.0	194	126.8	155	98.0
And	30	6	1372	1387.6	480	279.1	294	252.8	194	132.7
Park	40	3	1146	444.0	309	58.5	105	47.7	51	5.3
	50	3	1062	611.3	301	46.3	128	27.6	64	14.5
All sites for	10	40	1612	1164.4	605	255.9	198	218.4	180	157.1
all area	20	41	1583	1721.2	529	240.4	191	192.0	181	137.0
	30	36	1444	1376.8	510	249.2	192	243.4	165	152.0
	40	21	1802	1793.9	588	291.8	138	109.9	156	161.0

50	17	2227	3157.8	617	428.8	203	179.2	186	220.3

SD=standard deviation.

Table 5 shows major nutrients and selected heavy metals for soil at a depth of 50 cm for all areas that would be occupied by SUDS structures. Contamination level variations at a depth relevant for conveyance structures such as swales are shown to give the reader an indication of the potential remediation work to be undertaken for unlined SUDS structures to avoid leaching out of nutrients and metals from the soil into the runoff (Table 5).

Table 5. **Major total nutrients and major heavy metals (mg/kg dry weight): comparison of soil quality at a depth of 50 cm for all areas that would be occupied by SUDS structures. Sampling sites have been chosen based on proximity to nodes on a 50 m × 50 m grid.**

Area	*Count*	*N*		*P*		*Pb*		*Zn*	
		Mean	*SD*	*Mean*	*SD*	*Mean*	*SD*	*Mean*	*SD*
Belvidere Hospital	23	816	331.1	532	220.1	505	1012.4	244	307.6
Celtic FC Stadium	8	1572	343.7	665	154.5	651	651.1	439	168.9
Cowlairs Park	31	733	348.9	453	234.4	107	57.4	98	57.8
Gadburn	22	2458	2748.0	596	339.5	124	114.1	146	119.0
Lillyburn Place	8	708	320.4	604	189.8	96	35.0	90	31.5
Ruchill Hospital and Park	33	815	364.2	500	721.8	262	252.7	138	119.5

SD=standard deviation.

Concerning organic contaminants, Fig. 6 shows an example gas chromatograph result for a representative demonstration area (Belvidere Hospital). The largest peak observed was an artefact of the extraction procedure and showed up in the method blank as well as all the samples.

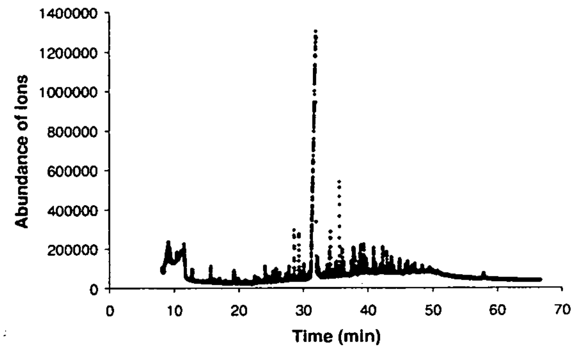

Figure 6. Belvidere Hospital area (proposed pond location): gas chromatograph findings associated with organic contamination of soil at 50 cm depth on 5 July 2004.

Case studies

A planimeter investigation has shown that the horizontal area of the Belvidere Hospital area available for the integration of SUDS techniques would be 94,000 m^2 (Figs. 4 and 5).

Figure2 shows a photograph of the site for which a major SUDS feature is planned. In comparison, Fig. 3 shows an artist impression of this site after regeneration. The proposed SUDS design for the Belvidere Hospital area is shown in Figs. 4 and 5.

Figure 6 shows an example of a total ion chromatogram.

Figure 7 shows a photograph of the site for which a major SUDS feature is planned.

Figure 7. Celtic FC Stadium area: site photograph taken on 14 May 2004.

The proposed SUDS design for the Celtic FC Stadium area is shown in Figs. 8 and 9. A planimeter investigation has shown that the horizontal area of the Celtic FC Stadium area (excluding Celtic Park) available for the integration of SUDS techniques would be 58,500 m^2.

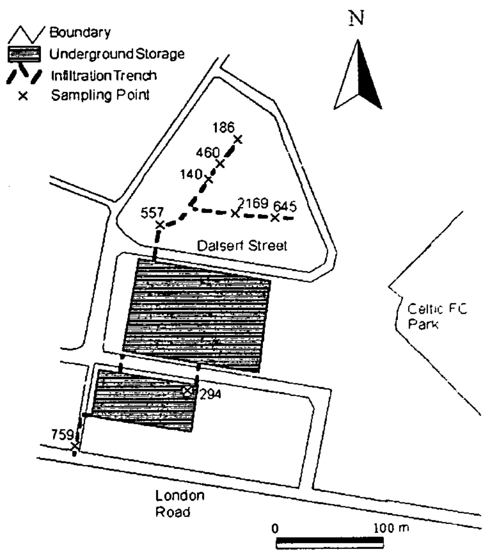

Figure 8. Celtic FC Stadium area: spatial distribution of lead (mg/kg dry weight) at 50 cm depth on 21 July 2004.

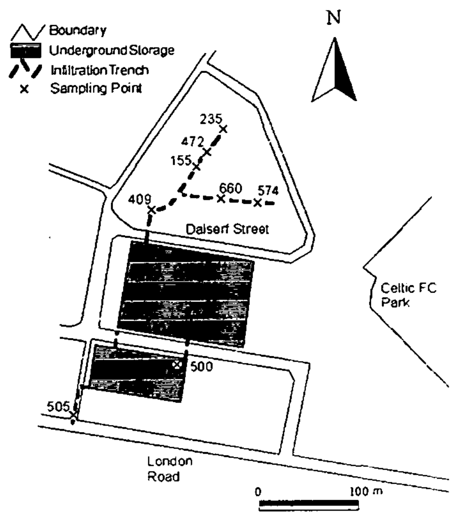

Figure 9. Celtic FC Stadium area: spatial distribution of zinc (mg/kg dry weight) at 50 cm depth on 21 July 2004.

Discussion

Definitions for proposed SUDS techniques

The abbreviation SUDS is an acronym for Sustainable Urban Drainage System or also known as Best Management Practice (BMP) in the USA. For the purpose of the case studies, a SUDS is defined as either an individual or a series of management structures and associated processes designed to drain surface runoff in a sustainable approach to predominantly alleviate capacities in existing conventional drainage systems (predominantly combined sewers in Glasgow) in an urban environment (CIRIA, 2000; Butler and Davis, 2000; SEPA, 1999).

The pond proposed for the Belvidere Hospital area is a depression structure that increases the duration of the flow hydrograph with a consequent reduction in peak flow, with the depression having a minimum depth of water present at all times, and an overflow outlet to the river. The pond can be used for attenuation, detention, retention, storage, infiltration and recreational purposes (Guo, 2001; Scholz, 2003, Scholz, 2004). As the pond matures, it may become heavily vegetated, and could be classified as a wetland with the potential to enhance the ecological habitat (Scholz and Trepel, 2004).

The proposed network of swales at the Belvidere Hospital comprises grass-lined conveyance structures (approximately 5 m in width) designed to infiltrate but predominantly to transport runoff from the site, while controlling the flow and quality of the surface water. The swales convey water to a river via a pond. The contaminated soil will have to be removed to avoid the leaching of metals into the runoff.

The proposed infiltration trenches in the Celtic FC Stadium area are linear drains (also known as French Drains). An infiltration trench consists of a trench filled with a permeable material and with a perforated pipe at designated depth to promote infiltration of surface runoff to the ground. Some of the infiltration trenches will also convey water, if their gradient is sufficiently steep.

Underground stormwater storage tanks have been proposed for the Celtic FC Stadium car parking areas. These sub-surface structures are designed to accumulate surface runoff, and release it subsequently, as may be required to increase the flow hydrograph, if there is no risk of flooding. Moreover, the structure may contain aggregates or plastic boxes (e.g., Matrix Geo-Cell detention system promoted by Atlantis Water Management Ltd.) and can act also as a water recycler or infiltration device.

Relevant soil contamination guidance

The soil contaminants summarised in Tables 2 to 4 should be seen in context with soil contamination guidelines (Environment Agency, 2002; Ministry of Housing, Spatial Planning and Environment, 2000; Society of the Chemical Industry, 1979). The guidelines specify thresholds for heavy metals such as chromium, copper, manganese, nickel, lead and zinc.

Concerning chromium, the threshold for residential properties with and without plant uptake is 130 and 200 mg/kg dry weight, respectively. In contrast, the threshold for commercial and industrial land is 5000 mg/kg dry weight (Environment Agency, 2002). In comparison, the Dutch intervention concentration is 380 mg/kg dry weight (Ministry of Housing, Spatial Planning and Environment, 2000) while the Kelly threshold (i.e. Kelly Indices Guidelines for Contaminated Soils; specifically developed for gasworks sites in London) is 200 mg/kg dry weight (Society of the Chemical Industry, 1979). However, chromium is not a major concern for both selected case study areas.

Concerning copper, the Dutch intervention concentration is 190 mg/kg dry weight (Ministry of Housing, Spatial Planning and Environment, 2000) while the Kelly threshold is 200 mg/kg dry weight (Society of the Chemical Industry, 1979). Concerning manganese, the Kelly threshold is 1000 mg/kg dry weight (Society of the Chemical Industry, 1979). Nevertheless, neither copper nor manganese are a particular concern for both selected case study areas.

Concerning nickel, the threshold for residential properties with and without plant uptake is 50 and 75 mg/kg dry weight, respectively. In contrast, the threshold for commercial and industrial land is 5000 mg/kg dry weight (Environment Agency, 2002). In comparison, the Dutch intervention concentration is 210 mg/kg dry weight (Ministry of Housing, Spatial Planning and Environment, 2000) while the Kelly threshold is 50 mg/kg dry weight. However, the latter concentration is correct for available nickel only (Society of the Chemical Industry, 1979). Except for the East of the Celtic FC Stadium area, nickel is not a problem for both case studies.

Concerning lead, the threshold for residential properties (with and without plant uptake) is 450 mg/kg dry weight. In contrast, the threshold for commercial and industrial land is 750 mg/kg dry weight (Environment Agency, 2002). In comparison, the Dutch intervention concentration is 530 mg/kg dry weight (Ministry of Housing, Spatial Planning and Environment, 2000) while the Kelly threshold is 1000 mg/kg dry weight (Society of the Chemical Industry, 1979). Lead is a major problem for both case study areas. Depending on further ground investigations, it is, however, likely that large quantities of top soil need to me removed on both sides before new residential developments can be built.

Concerning zinc, the Dutch intervention concentration is 720 mg/kg dry weight (Ministry of Housing, Spatial Planning and Environment, 2000) while the Kelly threshold is 500 mg/kg dry weight (Society of the Chemical Industry, 1979). Zinc is potentially a major problem for both case study areas. It is likely that top soil in some parts of both selected demonstration sites needs to me removed before new residential developments can be built.

Cost-benefit analysis

It has often been questioned if the reduction of pollution and runoff volume justifies the (potentially additional) costs of SUDS. Concerning both selected case studies, planning permission will only be given if the developers can demonstrate that no additional runoff will impact on the existing sewer system during storm event. It follows that either SUDS or a more traditional drainage solution in form of a detention system (e.g., large below ground storage tank) has to be considered during the planning phase.

Concerning the Belvidere Hospital area, the capital costs for both systems is likely to be similar (approximately £700k) as shown in previous studies in Scotland (Broad and Barbarito, 2004). A traditional solution would provide more space for housing while a SUDS solution has the additional benefit of enhancing the ecological value of a landscape and reduce environmental pollutants. However, unless the SUDS can be integrated into the proportion of green space that is usually reserved for recreational purposes (i.e. approximately 10% of a new site), the traditional system is likely to be marginally less expensive. On the other hand, the maintenance costs of SUDS are usually lower (approximately by 30%) than for conventional systems (Broad and Barbarito, 2004; Butler and Davis, 2000).

Concerning the Celtic FC Stadium area, the proposed SUDS solution (i.e. predominantly underground storage) is virtually the same as a traditional sub-surface detention tank. Estimated capital costs are approximately £500k. Therefore, the maintenance will also be virtually identical. Retrofitting of a detention system can easily be justified with flood prevention measures considering that this part of

Glasgow is subject to frequent and regular devastating floods. The gained sewer storage space can subsequently justify the regeneration of neighbouring estates where currently most flats are empty.

A detailed cost-benefit analysis comparing SUDS with traditional drainage systems or even comparing different SUDS treatment trains with eachother is beyond the scope of this paper. Moreover, the planning phase has not progressed sufficiently to enable a calculation to be based on detailed information.

Belvidere Hospital area design proposal

Future building design plans for the Belvidere Hospital area are not yet finalised as planning permission has yet to be sought by Kier Homes (former owner: National Health Trust). However, medium density residential properties are assumed to dominate a future landscape, and thus all SUDS recommendations have been made with regard to this assumption (Figs. 4 and 5).

The main entrance to the Belvidere Hospital area is located approximately in the middle of the Northeast face of the area adjacent to London Road. Two large vegetated areas flank the main driveway, which runs from the main entrance in a south-westerly direction. The driveway separates two fields suitable for housing: The first field to the Northwest of the main driveway is overgrown with some small trees and shrubs. The dimensions of this site are approximately 150 m x 150 m. The second field, which is located to the Southwest of the main driveway, is also overgrown but contains residual asphalted car parking and building foundations throughout. The area is approximately 150 m x 450 m in size. Both areas are mainly level apart from some depressions towards the Southwest and South of the site (Figs. 4 and 5).

A central depression exists to the Southeast of the remaining building. This area has therefore been identified as an ideal location for the implementation of a detention and attenuation pond and an associated outlet swale (or culvert) structure, and it is therefore recommended that no building construction work should be undertaken in this part of the Belvidere Hospital area. Moreover, the residual foundations in the centre of the Belvidere Hospital area appear to be at ground floor level with an existing basement level beneath in a depression. Excavation of these structures would form a suitable depression for a detention pond, which would provide a sufficient attenuation period for surface runoff.

The best engineering option recommended is to have essentially two inter-connected networks of swales throughout both fields allowing suitable spacing for a medium-dense housing development. Also, it is recommended that the existing building's guttering should be redirected into a swale, which should be connected with the inlet structure of the detention pond that serves also the combined network of both swale systems (Figs 4 and 5).

The detention pond area should include space for decorative embankment planting, seating areas and a footpath circling the pond and woodland to create a high amenity value by providing interesting landscaping features to the local community (Figs. 3 to 5).

From this detention pond a further cascading swale, acting as a combined overflow and outlet structure should flow down the embankment through the existing glade of mature tress to the public river walkway (Figs. 4 and 5). It is recognised that a swale may be difficult to construct due to the established vegetation, and therefore a cascade of small ponds (inter-connected with a culvert) or an open channel lined with decorative brick) or even a sub-surface pipe may be more suitable for some stretches pending on an outstanding ecological habitat assessment. A suitable provision should be made to allow the overflow to flow under or across the walkway by means of guttering into the River Clyde.

Transport structures such as feeder roads and car parks should be constructed from permeable or pervious pavement. A short culvert below the main driveway (connecting London Road with the former hospital building), which is expected to be retained, from the swale network in the North to the detention pond in the centre of the area should be considered (Figs. 4 and 5).

Contaminants such as manganese, lead (Fig. 4) and to a lesser extend zinc (Fig. 5) are present in high concentrations in the soil (Tables 3 to 5). The soil in the centre-southwest of the area is heavily contaminated with lead and zinc and would require removal. However, lead in particular is very difficult to dissolve in water and would not cause a problem for the outflow concentration of most SUDS structures (Scholz *et al.*, 2002).

The concentration of organic compounds found was low (estimated to be less than 1 mg/kg). Compounds found included polycyclic aromatic hydrocarbons (PAH). For example, pyrene, fluoranthene, chrysene, benz(a)anthracene, benzo(k)fluoranthene, benz(a)pyrene, benzo(g,h,i)perylene were found at very low levels at Belvedere Hospital (proposed pond location). Other compounds found included aliphatic hydrocarbons such as tetracosane, eicosane, heptadecane, heptacosane and nonadecane which are commonly constituents of diesel–type fuels. Phenol derivatives and carbolic acid related compounds were also found. These types of compounds were often used as cleaning agents and disinfectants in hospitals and schools (Fig. 6).

Celtic FC Stadium area design proposal

The areas surrounding the Celtic Park stadium (Fig. 7) are currently under development and regeneration. Celtic FC was granted planning consent in 1994 for the redevelopment of the stadium to from an all-seated stadium with a capacity for 60,000 spectators. As part of this planning application, the club was required to provide 377 car-parking spaces within the cartilage of the stadium. This has been achieved to the satisfaction of the Glasgow City Council.

In 1998, the club was granted planning consent for the formation of a temporary coach park on the site of a former bakery in the Camlachie to the West of the stadium. This consent allowed for the parking of 171 coaches, and was granted for a period of three years until June 2001. Renewal of this consent was granted in July 2001 for a further period of three years. This facility is used for coaches of home supporters. The catchment area excluding the stadium is about 58,500 m^2.

Considering the current state of the Celtic FC car park and its heavy use during match days, this area would be ideal for an integrated underground storage system underneath the present car park. The suggested area in the West for the integration of underground storage facilities is the site surrounded by Dalserf Street in the North and Barrowfield Street in the South. The storage area would be approximately 14,600 m^2. A further but smaller underground storage area of 4,900 m^2 could be located in the Southeast of the main storage tank just South of Barrowfield Street (Figs. 8 and 9).

According to a recent site investigation and the current characteristics of the area in the North, the construction of a simple infiltration trench network with two branches seems to be feasible. The branches of the infiltration trench network should be located in the North and Northeast, respectively. The land in the North is associated with the highest ground level in the study area. The infiltration trench network will transfer the water from the roofs and paved surfaces to the major underground storage through an inlet in the Northwest of the main storage tank (Figs. 8 and 9).

Infiltration trenches or culverts should connect the two storage tanks and transfer the runoff to the smaller storage tank and when required to the sewer system located on London Road. The overflow of the storage tank system is located in the Southwest of the study area (Figs. 8 and 9).

Considering the current state of the Celtic FC car park in the West, renovation work is likely to be required within a couple of years. The implementation of the recommended SUDS would therefore be easily approachable.

The area is contaminated with lead (Fig. 8) and zinc (Fig. 9) that might be linked to pollution from cars (Tables 2 to 4). The soil requires removal, if used by residents in the future. However, lead in particular is very difficult to dissolve in water (Scholz *et al.*, 2002) and is unlikely to cause a problem for the outflow concentration of underground SUDS structures if pH levels are high and conductivity values low (Scholz *et al.*, 2004).

The overall concentration of organic compounds found was low (estimated to be less than 0.5 mg/kg). Compounds found included polycyclic aromatic hydrocarbons

(PAH). For example, pyrene, fluoranthene, chrysene, benz(a)anthracene, benzo(k)fluoranthene, benz(a)pyrene, benzo(g,h,i)perylene were found at very low levels in the Celtic FC Stadium area (at the proposed network of swales). Other compounds found included aliphatic hydrocarbons such as tetracosane, eicosane, heptadecane, heptacosane and nonadecane which are commonly constituents of diesel–type fuels.

Summary and Conclusions

A survey of 57 sites within 46 areas of Glasgow shows that it is feasible to implement different SUDS techniques throughout Glasgow. The likely contribution of future SUDS to the overall catchment dynamic of representative demonstration areas has been assessed. The preliminary designs will help to understand the challenges of holistic catchment management, diffuse pollution, and the linking scales in catchment management. The implementation of SUDS will help to relief the local sewer system, and subsequently allows for more regeneration activities to take place.

Characteristics that determine the suitability of a site for the implementation of SUDS have been identified. Representative areas and sites that are suitable for different representative SUDS techniques have been identified qualitatively and quantitatively. A SUDS decision support key and matrix that are adaptable to different cities have been proposed. The matrix can be used as part of a decision support model in the future.

Seven entirely different SUDS demonstration areas that are representative for both different sustainable drainage techniques and different types of areas available for development, regeneration and retrofitting of SUDS within Glasgow have been identified. Design and management guidelines for demonstration sites that should be constructed to inform and educate the public, developers and politicians have been proposed. Underground storage tanks and ponds linked with swales and infiltration trenches have been identified as the most useful sustainable drainage techniques for large sites within Glasgow.

Concerning the case studies, the proposed drainage system for the Belvidere Hospital area is dominated by a network of swales draining into a large attenuation and detention pond. The runoff will ultimately drain from the pond into a nearby river. In comparison, the drainage of the Celtic FC Stadium area is dominated by two large underground detention tanks beneath car parks. The runoff, after a considerable lag period, will ultimately drain into the sewer after the risk of flooding has gone down.

Furthermore, a brief cost-benefit analysis has shown that the capital costs for the proposed SUDS solutions are likely to be similar to the costs for a comparable traditional drainage system. However, a SUDS solution would be preferable, if it could be integrated into the area reserved for green space.

The soils for both selected case studies are contaminated predominantly with lead and zinc. Moreover, hot spots of nickel contamination were detected in the East of the Celtic FC Stadium area. In comparison, organic contamination was insignificant. The application of the SUDS decision support matrix has identified that unlined SUDS structures such as swales can only be implemented if the risk of runoff being contaminated by metal leaching is eliminated. Large quantities of top soil require therefore removal before construction work on residential properties can commence to avoid environmental and water pollution as well as potential health problems for the residents.

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Appendix 2

Assessing storm water detention systems treating road runoff with an artificial neural network

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Abstract

This paper examines whether multiple regression analysis and neural network models can be applied successfully for the indirect prediction of the runoff treatment performance with water quality indicator variables in an experimental storm water detention system rig. Five mature experimental storm water detention systems with different designs treating concentrated gully pot liquor were assessed in this study. The systems were located on The King's Buildings campus at The University of Edinburgh and were monitored for a period of eighteen months. Multiple regression analyses indicated a relatively successful prediction of the biochemical oxygen

demand, and total suspended solids for most systems but due to a relatively weak correlation between the predictors, and both microbial indicators, multiple regression analyses were not applied for the prediction of intestinal enterococci, and total coliform colony forming units. However, artificial neural network models predicted microbial counts relatively well for most detention systems.

Keywords

artificial neural network; best management practice; biochemical oxygen demand; intestinal enterococci; multiple regression analyses; runoff; total coliforms; urban area

INTRODUCTION

Background

Storm water runoff is usually collected in gully pots that can be viewed as simple physical, chemical, and biological reactors. They are particularly effective in retaining suspended solids (Bulc and Slak 2003). Conventionally, gully pot liquor is extracted on virtually random occasions from road drains and transported (often over long distances) for disposal at sewage treatment works (Butler *et al.* 1995; Memon and Butler 2002).

Storm water management strategies generally involve controlling nonpoint source pollution by implementing best management practices (BMP) (Olding *et al.* 2004; Wu *et al.* 2006). Runoff pollution has been characterized, in magnitude and in concentration of pollutants, by intermittent and impulse-type discharges into receiving waters, causing shock-loading problems for the ecosystems of these water bodies (Wu and Ahlert 1978).

There are several source control methods to reduce the microbial contamination in runoff. Storm water detention systems treat runoff, for example, from parking lots, and roads locally and are more environmentally sustainable in comparison to traditional drainage technologies. This can reduce the costs of construction, transport, and treatment significantly. Moreover, other studies suggest that treated runoff can be used for irrigation purposes (Scholz 2006).

Below ground storm water detention systems are defined as sub-surface structures designed to accumulate surface runoff, and where water is released from, as may be required to increase the flow hydrograph. The structure may contain aggregates with a high void ratio or empty plastic crates, and act also as a water recycler or infiltration device (Butler and Parkinson 1997).

Since 1980, below ground storm water detention systems are specifically designed to reduce storm water flow. The surface water is being captured through infiltration. The filtered storm water is detained below the ground within a detention tank (Butler and Parkinson 1997).

Under normal circumstances, the runoff is treated by filtration prior to infiltration or discharge to the sewer or watercourse via a discharge control valve. The application of these systems reduces runoff in case of minor storms as well as encourages groundwater recharge, and pollution reduction. These detention systems can frequently be found in new developments (Scholz 2006).

There is an urgent need to modify common storm water detention systems to meet more stringent water quality guidelines (Butler and Parkinson 1997; Scholz 2006). Research should focus on the implementation of sustainable filters within the current structures of detention systems.

Microbial contamination

Faecal pollution within storm water runoff can cause significant health risks as a result of the presence of various infectious microorganisms such as *Escherichia coli*. It is well understood that dog fouling is the major source of faecal contamination in urban runoff. The UK's dog population is reported to be between 6.5, and 7.4 million, producing nearly 1000 tons of faeces per day. Additionally, daily faecal output per dog is estimated between 100, and 200 g (O'Keefe *et al.* 2005).

Bacterial indicator organisms have been frequently used to assess the presence of faecal contamination, and consequently pathogens in drinking, and bathing waters (NRC 2004). Total coliforms, and *Enterococcus* are the most commonly used indicators (NRC 2004), due to their relative ease of application, and low determination costs.

Modeling microbial water quality can be a useful approach for watershed managers, environmental regulators, and others involved in the evaluation, and protection of ecological habitats, and public health. Artificial neural networks (ANN) can be used to derive relationships between gathered data to predict microbial populations and other water quality parameters (Lee and Scholz 2006).

The microbial population in storm water runoff is controlled by different variables including temperature, and the availability of suspended solids, and nutrients. Studies show that *Enterococci* preferentially attach to particles with diameters from 10 μm to 30 μm, while total coliforms have a broader distribution (Jeng *et al.* 2005).

Modelling approaches

An artificial neural network can simply be described as an artificial computational copy of a brain (Iyengar and Kashyap 1989; Mohanty *et al.* 2002; Lee and Scholz 2006). The networks work by attempting to mimic the way in which human brains operate (Zurada 1992). Mathematically, an ANN is a nonlinear function comprising parameters that can be trained by an optimization procedure so that the ANN output becomes similar to the measured output on a known dataset (Scholz 2006). This ability to replicate non-linear relationships makes ANN suitable for modeling environmental systems (Maier and Dandy 1998). Recently, ANN models have been used in many water resources applications, such as, water quality forecasting, and the prediction of chemical, and microbiological dosage in water treatment plants (Maier and Dandy 2000; Lee and Scholz 2006).

Aim and objectives

The aim of this study is to show how different storm water detention systems cope with a 'worst case pollution scenario'. The research objectives are to:

- assess the general inflow, and outflow water quality;
- evaluate the water treatment efficiencies of different experimental storm water detention systems receiving concentrated runoff contaminated by dog faeces;
- develop multiple regression models for each system.
- undertake analyses of variance (ANOVA) to compare inflows and outflows,

and the performances of all systems; and

- predict total coliforms, and intestinal enterococci colony forming units by developing an ANN for each system and each variable.

METHODOLOGY

Experimental system setup

Five mature detention systems (plastic crates wrapped in geotextile, and marketed as Matrix Geo-Cell, provided by Atlantis Water Management (Alderborough, **Sladen Mill Industrial Complex, Littleborough,** England, UK)), were located outdoors at The King's Buildings campus (The University of Edinburgh, Scotland, UK) to assess the system's performances during a period of more than one year (2005-04-01 to 2006-09-13). However, the rig was in operation since 2004-03-31.

Two plastic crates (total height, 1.7 m; length, 0.68 m; width, 0.41 m) on top of each other comprised one detention system. The tank volume below each filter was 0.08 m^3. The detention system filter volumes for all five systems were 0.24 m^3.

The bottom cell (almost 50% full at any time) was used for water storage, and passive treatment only. The top cell was used as a coarse filter. Different arrangements of aggregates, and planting were used within the filtration zones of each detention system. Different packing order arrangements of aggregates, and plant roots were used in the systems (Table 1) to test for the effects of gravel, sand, Ecosoil® (product based essentially on sand, and bark, and provided by Atlantis Water Management), block paving, and turf on the water treatment performance.

Height (mm)	System 1	System 2	System 3	System 4	System 5
861-930 (top)	Air (and common reed in summer)	Air	Block paving, and 6 mm gravel (within spaces)	Block paving, and turf (within spaces)	Air
791-860					Turf
751-790			6mm gravel	Sand, and Ecosoil®	Sand, and Ecosoil®
745-750			Geotextile	Geotextile	
711-744			Drainage cell	Drainage cell	
693-710					6mm gravel
687-692			Geotextile	Geotextile	
661-686			6mm gravel	6mm gravel	
451-660			20mm gravel	20mm gravel	20mm gravel
437-450			Sand	Sand	Sand
431-436			Geotextile	Geotextile	Geotextile
201-430	Water, and common reed		Air	Air	Air
0-200 (bottom)	Gravel (water, and roots within voids)	Water	Water	Water	Water

TABLE 1. PACKING ORDER OF THE STORM WATER DETENTION SYSTEMS.

Systems 1, and 2 represented sand, and gravel filled constructed wetlands planted with Common Reed, *Phragmites australis* (Cav.) Trin. ex Steud), and a detention basin, respectively. Systems 3, 4, and 5 were similar to slow sand trickling filters.

Inflow water, polluted by road runoff, was collected by manual abstraction with a 2 l beaker from randomly selected gully pots on the campus. Temperature, and dissolved oxygen were measured onsite, and the corresponding water samples were subsequently transferred into the campus-based public health laboratory for further water quality analyses.

All detention systems were watered as slow as possible within 3 to 5 min approximately twice per week with 5 l gully pot liquor artificially contaminated by dog faeces (180 g), and drained by gravity afterwards to encourage air penetration through the filtration system (Gervin and Brix 2001). The quantity of gully pot liquor used per system was approximately 3.6 × the mean annual rainfall volume (data obtained from The University of Edinburgh Weathercam Station in 2006) to simulate a 'worst case scenario'. The hydraulic residence times were in the order of one hour.

Data set

The sampling of data was done simultaneously for all systems. However, the number of samples is sometimes different between inflow and outflow for the same variable because outliers and human error have been identified at a later stage during data analysis. Consequently, values identified as flawed have been removed from the data set. It follows that correct data that directly correspond to all removed entries were also removed during further analysis and modelling to obtain an overall data set that only contains matching pairs. All tested variables were $\log_{10}$-transformed to achieve normality for subsequent statistical tests if required.

Modeling

In the last few decades, artificial neural network (ANN) modeling approaches have been numerously applied in the area of water quality modeling, where they proved to be particularly successful in predictions based upon complex, inter-related, and often non-linear relationships between multiple parameters (Brion and Lingireddy 2003). In their research, Sandhu and Finch (1996) indicate that ANN models have been more successful in estimating river salinity than other simulation and commonly used statistical models. However, there are difficulties involved with applying models for microbial water quality predictions; mostly as a result of complexities in environmental distribution; mobility and fate of microbes. Microbial contaminants are known to be non-conservative, unevenly distributed and their numbers and growth rates may change in the environment depending on the conditions they live in. The inter-relationship and interactions between microbial colonies in storm water cause various modeling challenges that have been overcome for particular case studies by applications of ANN to multi-parameter data sets (Brion and Lingireddy 2003).

Artificial neural networks are mathematical modeling tools that are applicable in the field of prediction, and forecasting in complex settings. They are relatively good tools for interpolation in the range of observed conditions, but can be very poor in prediction and forecasting, especially in case of overtraining (Scholz 2006). Fundamentally, they operate through simulating, at a simplified layer, the activity of the human brain. The network fulfils this through a vast number of highly interconnected processing elements (called nodes in this paper), working in accord to solve specific problems, including forecasting, and pattern recognition. In an ANN, each node is connected to other neighbouring nodes with different coefficients or weights, which represent the relative influence of the varying node inputs to other nodes (Hamed *et al* 2004).

Each neuron in a network has a scalar bias b, the bias is similar to a weight except that it has a constant input of 1. The transfer function net input n in the ANN is also a scalar and is equal to the sum of the weighted input wp and the bias b. This sum is the argument of the transfer function f. A transfer function can be a step function or a sigmoid function, which takes the argument n and produces the output a. Both w and b are adjustable scalar parameters of the neuron. The main concern in ANN is the adjustability of such parameters so that the network would be able to reveal most desired and interesting behavioural patterns. A neuron with a single scalar input and a scalar bias b appears in Fig 1.

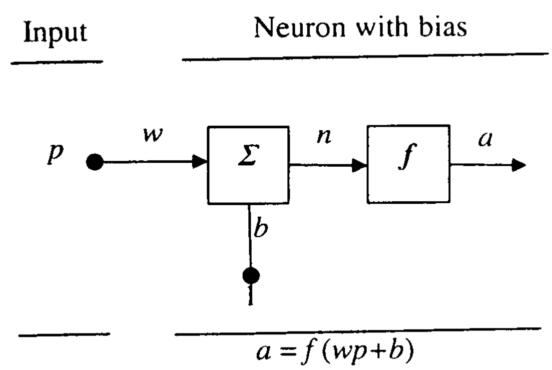


Fig. 1. A neuron with a single scalar input and a scalar bias; p is the scalar input, w is the scalar weight, wp is the scalar product, f is the transfer function, which produces the scalar output, a is the scalar output, b is the scalar bias, and n is the transfer function net input.

Artificial neural networks vary in type. A basic example of a neural network is given in Fig. 2, containing one input, one hidden, and one output layer; they are all connected without any feedback connections. The weighted sum of the inputs are transferred to the hidden nodes, where it is transformed using an output function (also called transfer or activation function). In return, the outputs of the hidden nodes perform as inputs to the output node where another transformation happens. Network outputs often have associated processing functions; these functions are used to transform user-provided target vectors for network use. Network outputs are then reverse-processed using the same function to produce output data with the same characteristics as the original user-provided targets. A typical processing function for the output of the hidden layer is the output function given in Equation 1.

$$x_i = \sigma_i\left(b_i + \sum_{j=1}^{n} w_{ij} u_j\right) \tag{1}$$

where x_i is the output from the hidden node, σ_i is the output function of the hidden node (usually the hyperbolic tangent *tanh*), b_i is the bias input to the hidden node *i*, *n* is the number of input nodes, w_{ij} is the weight connecting the input node *j* to the hidden node *i*, and u_j is the input node *j* (Sarle 2002).

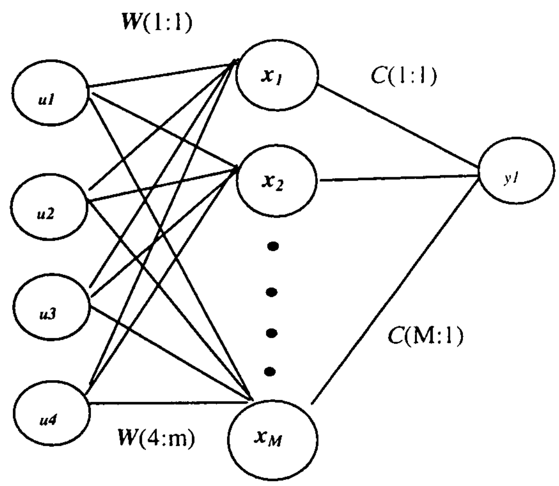


FIG. 2. NEURAL NETWORK ARCHITECTURE (M=8 FOR INTESTINAL ENTEROCOCCI AND M=64 FOR TOTAL COLIFORMS); *U* IS THE INPUT *LAYER*, *X* IS THE HIDDEN *LAYER*, *Y* IS THE OUTPUT NODE, *W* IS THE WEIGHT *MATRIX* CONNECTING THE INPUT NODE TO THE HIDDEN NODE, AND *C* IS THE WEIGHT *MATRIX* CONNECTING THE HIDDEN NODE TO THE OUTPUT NODE.

A representation of the hidden node *i* is given in Fig. 3. Moreover, the typical processing function for the output of the network can be expressed in Equation 2.

$$y_i = \sum_{j=1}^{m} c_{ij} x_j \tag{2}$$

where y_i is the output from the output node *i*, *m* is the number of hidden nodes, c_{ij} is the weight connecting the hidden node, and x_j is the weighted sum of inputs into the hidden node *j* to the output node *i*.

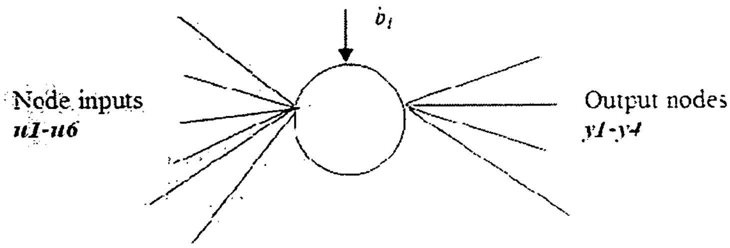


FIG. 3. SCHEMATIC REPRESENTATION OF THE HIDDEN NODE *l*; *B_l* IS THE BIAS TERM, AND Σ IS THE OUTPUT FUNCTION OF THE HIDDEN NODE (USUALLY THE HYPERBOLIC TANGENT *TANH*).

During network training, the connection weights, and biases of the ANN are adapted through a continuous process of simulation. The primary training goal is to minimize an error function by searching for connection strengths, and biases that make the ANN produce outputs that are equal or close to the targets. Equation 3 expresses the mean square error (MSE) of the output values.

$$MSE = \sum_{t=1}^{N} \left(Y_t - \hat{Y}_t\right)^2 / N \tag{3}$$

where *MSE* is the mean square error, *N* is the number of data points, Y_t is the observed output value, and $\hat{Y}_t$ is the output of a feed-forward neural network.

The minimization procedure consists in the optimization of a non-linear objective function. A number of optimization routines can be applied. Practically, the Levenberg-Marquardt routine is often used as it finds better optima for various problems than the other optimization methods (Sarle 2002).

Development of the artificial neural network model

In this study, one of the most commonly used type of ANN was used: the feed-forward network, where the information is transmitted in a forward direction only. According to Tomenko *et al.* (2007), feed forward ANN models were found one of the most efficient and robust tools in predicting constructed treatment wetland performance if compared to traditional models. For example, Neelakantan *et al.* (2001) have developed a simple feed-forward back propagation ANN model, which was successful in predicting *Cryptosporidium* and *Giardia* populations with a number of other biological, chemical and physical variables in the Delaware River.

A multi-layer, feed-forward ANN usually contains one input, one output, and one hidden layer. Different numbers of hidden nods, and various output functions were tested during the model development. Although, at present, no specific standards exist for the selection of the number of hidden nods, there are various guidelines proposed in literature (Rogers and Dowla 1994; Maier and Dandy 1998). Six model architectures were applied for each set of input parameters. The number of applied hidden nods was 2^k, with k varying from 1 to 6. The optimum number of hidden nods was 8 for the prediction of intestinal enterococci colony forming units, and 64 for the prediction of total coliform colony forming units. The Levenberg-Marquardt optimization method was applied for all models. The MATLAB neural network tool box (version 5.3) was used.

The counts of total coliforms and intestinal enterococci per 100 ml in outflow samples collected from 2005-04-14 to 2006-09-15, ranged between 300 and 7100, and between 300 and 2010, respectively. The corresponding inflow counts were between 550 and 8420, and between 360 and 2130, respectively. Table 2 summarizes statistics for total coliforms, and intestinal enterococci. European legislation sets a mandatory water quality standard requiring that total coliforms, and faecal streptococci should not exceed 10,000 cfu/ml, and 2000 cfu/ml for 95% of the water samples, respectively.

Statistics	Intestinal enterococci (n=63)					
	Inflow	1	2	3	4	5
Max	2130	2130	2130	2130	2130	2130
Min	360	360	360	360	360	360
Mean	1140	1140	1140	1140	1140	1140
Std	506	506	506	506	506	506
Statistics	**Total coliforms (n=61)**					
	Inflow	1	2	3	4	5
Max	8420	5280	7100	6130	5530	3500
Min	550	320	390	280	300	300
Mean	3801	1807	2776	2489	1204	939
Std	2742	1363	2169	1870	1079	748

All units are cfu/100ml; n, number of samples.
Std: Standard deviation

TABLE 2. SUMMARY STATISTICS FOR TOTAL COLIFORM, AND INTESTINAL ENTEROCOCCI COUNTS FOR THE ENTIRE DATASET (2005-04-15 TO 2006-09-13) COMPRISING THE INFLOW AND OUTFLOWS FOR SYSTEMS 1 TO 5.

A certain number of relevant inputs should exist to achieve a successful determination of the relationships amongst the input variables, and the model output. When utilizing equations for chemical, biological or physical processes in a model, the specifications of the processes determine the required input parameters. The selection of inputs is not determined in ANN; therefore, inputs can be selected on the basis of intuitive or empirical understanding of the processes. However, advanced systematic analytical techniques such as principal component analysis or sensitivity analysis can be used when selecting input parameters (Maier and Dandy 1996; Zhang *et al.* 1998).

When compared with multiple regression analyses, where a *p* value indicates the significance of a variable, and its suitability for inclusion in a model, ANN provide no standard statistical measure to determine the significance of an input variable. Consequently, the input variables (turbidity, pH, conductivity and dissolved oxygen) selected in this study were chosen on this basis and of the information gathered from previous literature.

The dataset comprised 60 observed data per parameter per system, and was divided into testing, validation, and training data sub-sets. The training set contained 65% of the entries for the entire dataset (i.e. 39 observations), whereas the validation, and testing sets consisted of 15% (9 observations), and 20% (12 observations) of the entire dataset, respectively. Figure 4 schematically indicates a series of steps that have been conducted during the model development process (Hamed *et al.* 2004).

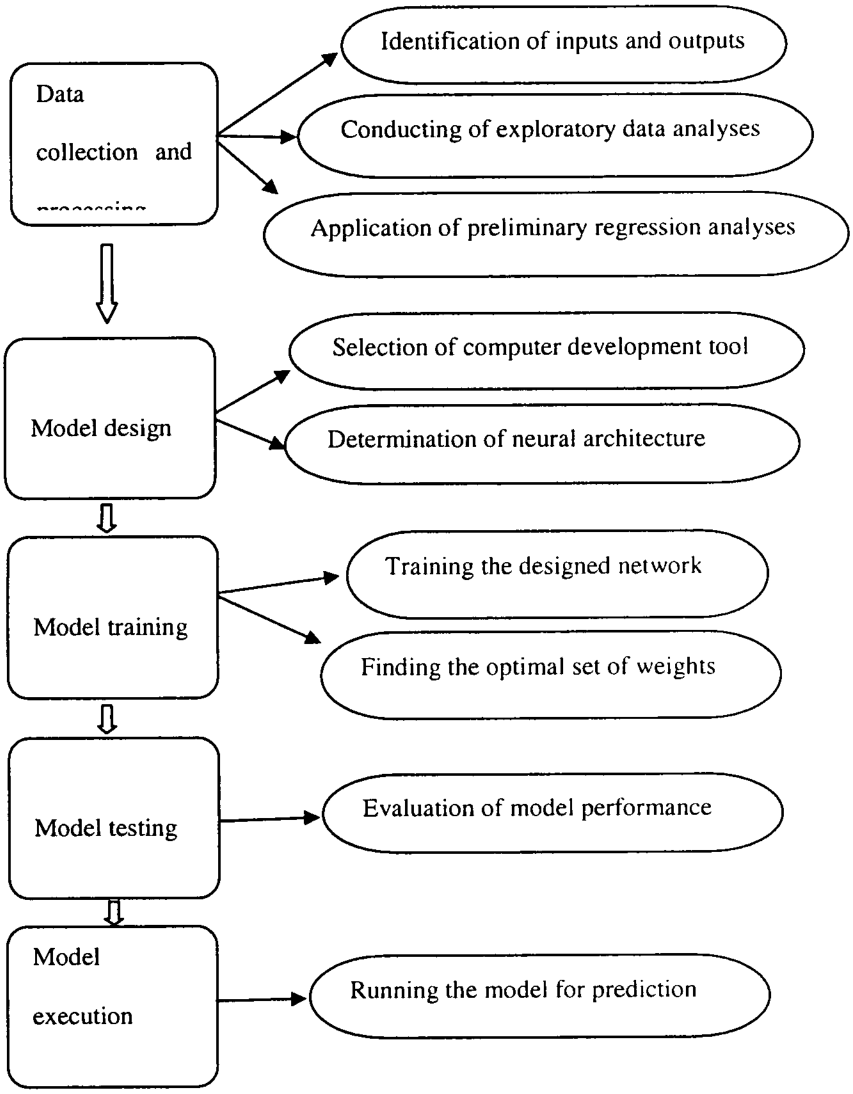


FIG. 4. STEPS OF THE MODEL DEVELOPMENT PROCESS.

RESULTS AND DISCUSION

Inflow and outflow water quality

Table 3 summarizes values representing the inflow water quality variables. Particularly during warmer seasons, values for five-days at 20°C biochemical oxygen demand (BOD) (nitrification inhibitor applied), suspended solids, ortho-phosphate-phosphorous and nitrate-nitrogen are above commonly accepted water quality standard thresholds (25, 35, 2 and 15 mg/l, respectively) for secondary wastewater treatment (ECC 1991). This is partly due to the inflow water quality being representative of the 'worst case scenario', and the lack of precipitation between 2006-03-24, and 2006-09-13.

Variables	No. of samples	Mean[f]	Mean[g]	Mean[h]	Standard deviation[f]	Standard deviation[g]	Standard deviation[h]
pH (-)	70	6.91	7.03	7.05	0.351	0.556	0.524
DO[a] (mg/l)	69	3.2	4.0	3.7	2.24	1.33	2.31
BOD[b] (mg/l)	67	23.0	49.5	54.1	17.05	50.75	318.72
SS[c] (mg/l)	70	99.9	52.2	68.3	106.60	48.21	191.67
TS[d] (mg/l)	69	588.6	523.9	1791.3	395.18	408.19	1427.38
TDS[e] (mg/l)	70	112.5	117.9	186.1	78.11	141.39	138.93
Conductivity (μS)	70	222.9	228.0	372.5	157.94	268.75	278.10
Turbidity (NTU)	69	37.6	55.3	111.4	15.01	35.58	125.49
Ortho-phosphate-phosphorus (mg/l)	69	1.6	3.3	22.8	1.95	3.97	15.55
Ammonia-nitrogen (mg/l)	68	1.4	0.7	1.8	1.63	0.60	1.33
Nitrate-nitrogen (mg/l)	68	0.2	1.4	1.0	0.10	3.45	1.06

[a]dissolved oxygen; [b]five-days at 20°C bochemical oxygen demand (nitrification inhibitor applied);[c]total suspended solids; [d]total solids; [e]total dissolved solids; [f]2005/03/23-2005/09/15; [g]2005/09/22-2006/03/16; [h]2006/03/24-2006/09/13.

TABLE 3. INFLOW WATER QUALITY.

Values for outflow water quality variables are shown in Table 4. Considerable improvements in the quality of the outflow have been observed, particularly if compared to the inflow values summarized in Table 3. This is the case during cold periods for variables such as suspended solids, BOD, and turbidity, where most values are considerably below water quality treatment standard thresholds (ECC 1991).

Variables	No. of samples	Mean[f]	Mean[g]	Mean[h]	Standard deviation[f]	Standard deviation[g]	Standard deviation[h]
System 1							
pH (-)	66	7.33	7.39	7.47	0.321	0.635	0.136
DO[a] (mg/l)	65	3.8	4.5	3.4	1.56	1.83	2.24
BOD[b] (mg/l)	64	2.3	5.4	28.2	2.34	7.98	13.52
SS[c] (mg/l)	65	8.3	11.1	31	12.37	15.10	31.66
TS[d] (mg/l)	65	569.6	530.6	675.2	240.25	867.15	666.83
TDS[e] (mg/l)	66	557.5	121.8	207.4	954.00	41.64	60.13
Conductivity (μS)	65	1154.4	247.1	415.0	1926.73	84.23	120.77
Turbidity (NTU)	65	4.9	4.1	7.6	3.76	3.13	4.05
Ortho-phosphate-phosphorus (mg/l)	65	1.4	3.4	14.3	1.64	2.87	2.88
Ammonia-nitrogen (mg/l)	66	0.3	0.6	0.8	0.87	1.56	0.98
Nitrate-nitrogen (mg/l)	63	0.6	1.1	0.1	0.65	1.24	0.07
System 2							
pH (-)	70	7.24	7.38	7.58	0.330	0.632	0.279
DO[a] (mg/l)	69	3.8	4.7	7.4	1.10	1.58	6.36
BOD[b] (mg/l)	70	1.9	4.7	24.9	1.45	2.75	11.61
SS[c] (mg/l)	70	6.5	7.8	27.5	3.23	7.39	11.81
TS[d] (mg/l)	70	518.6	300.2	427.5	367.67	104.00	213.85
TDS[e] (mg/l)	70	381.7	136.8	351.7	286.85	59.66	297.71
Conductivity (μS)	70	770.7	272.8	703.4	565.51	120.93	595.42
Turbidity (NTU)	69	3.9	4.4	41.1	2.14	3.10	57.08
Ortho-phosphate-phosphorus(mg/l)	69	1.2	4.3	19.9	1.48	3.97	14.34
Ammonia-nitrogen (mg/l)	68	1.6	0.7	3.3	1.47	1.34	3.53
Nitrate-nitrogen (mg/l)	68	1.7	1.1	0.6	1.79	1.33	1.20
System 3							
pH (-)	70	7.39	7.44	7.54	0.270	0.675	0.097
DO[a] (mg/l)	69	3.3	3.9	3.9	1.16	1.53	2.31
BOD[b] (mg/l)	67	1.7	4.1	9.1	1.17	4.40	8.61
SS[c] (mg/l)	70	6.7	11.0	13.5	7.15	13.39	5.98
TS[d] (mg/l)	70	483.5	326.1	422.8	248.14	167.41	157.68
TDS[e] (mg/l)	70	438.5	127.7	352.1	596.67	65.63	220.37
Conductivity (μS)	70	877.3	262.1	703.7	1193.15	130.67	439.59
Turbidity	67	5.3	7.6	5.0	3.40	7.29	1.93

(NTU)							
Ortho-phosphate-phosphorus(mg/l)	70	1.5	2.9	17.2	1.82	2.28	31.71
Ammonia-nitrogen (mg/l)	70	0.4	1.1	1.9	0.72	1.77	2.20
Nitrate-nitrogen (mg/l)	68	0.6	1.6	0.8	0.32	1.31	1.60
				System 4			
pH (-)	70	7.47	7.45	7.68	0.181	0.654	0.136
DO[a] (mg/l)	69	3.2	4.4	4.7	1.13	1.48	2.42
BOD[b] (mg/l)	68	1.9	3.6	5.7	0.93	4.01	5.50
SS[c] (mg/l)	70	8.2	13.7	14.3	14.20	19.21	18.50
TS[d] (mg/l)	70	478.5	308.3	381.9	228.23	147.42	141.76
TDS[e] (mg/l)	70	344.2	130.1	403.4	225.34	54.05	322.35
Conductivity (μS)	70	693.8	263.3	806.6	445.47	107.26	644.60
Turbidity (NTU)	69	7.3	6.5	2.8	7.74	5.80	2.27
Ortho-phosphate-phosphorus (mg/l)	68	1.6	3.6	15.2	1.79	2.57	19.21
Ammonia-nitrogen (mg/l)	68	0.5	1.3	0.2	0.71	1.87	0.06
Nitrate-nitrogen (mg/l)	68	0.8	1.7	0.8	0.27	1.31	0.99
				System 5			
pH (-)	70	7.41	7.53	7.61	0.202	0.530	0.133
DO[a] (mg/l)	69	3.8	4.9	4.8	1.20	1.67	2.05
BOD[b] (mg/l)	68	2.4	5.1	5.3	1.97	4.84	11.12
SS[c] (mg/l)	70	6.2	4.1	7.0	4.80	3.32	5.40
TS[d] (mg/l)	70	594.8	320.9	400.5	348.56	133.50	145.67
TDS[e] (mg/l)	70	391.1	125.8	422.6	185.78	54.10	364.38
Conductivity (μS)	70	779.5	252.9	845.1	374.88	106.61	728.89
Turbidity (NTU)	69	15.2	6.5	2.6	53.63	7.38	0.57
Ortho-phosphate-phosphorus (mg/l)	70	1.6	3.7	19.9	1.69	2.48	46.75
Ammonia-nitrogen (mg/l)	70	0.4	0.5	0.2	0.57	0.59	0.14
Nitrate-nitrogen (mg/l)	68	0.5	1.6	0.6	0.25	1.84	0.92

[a]dissolved oxygen; [b]five-days at 20°C (nitrification inhibitor applied) biochemical oxygen demand; [c]total suspended solids; [d]total solids; [e]total dissolved solids; [f]2005/03/23-2005/09/15; [g]2005/09/22-2006/03/16; [h]2006/03/24-2006/09/13.

TABLE 4. OUTFLOW WATER QUALITY.

Multiple linear regression analyses

Table 5 shows how BOD, and SS can be predicted by applying a multiple linear regression analysis covering eighteen months of experimental data. Electrical conductivity, turbidity, pH, ortho-phosphate-phosphorous, nitrate-nitrogen, and ammonia-nitrogen were selected for the prediction because the determination of these variables is less costly and time-consuming. Furthermore, stepwise regression was also undertaken to help in the selection of the most appropriate variables for prediction. Furthermore, total coliforms, and intestinal Enterococci colony forming units did not exhibit a significant correlation ($p<0.05$) with any of the proposed predictors.

Sample	a	b	c	d	e	f	g	SEE[a]	R^{2b}
				BOD					
Inflow	0	1.21	31.1	9.19	0	0	-207.0	12.3	0.88
System 1	0	0	0	2.46	0.30	0	-1.6	5.3	0.81
System 2	0	0.35	0	0	-4.55	9.93	2.8	5.1	0.82
System 3	0	0	0	0	-1.78	1.36	4.7	3.5	0.53
System 4	0	0	-2.1	0	-0.99	0	20.2	2.8	0.43
				SS					
Inflow	0.190	0.56	0	0	0	NA	34.9	33.16	0.83
System 1	0	0.38	0	2.76	-0.94	NA	2.8	7.74	0.79
System 2	0.001	0.18	3.3	0.69	0	NA	-17.8	6.14	0.65
System 3	0	0	10.4	0	0	NA	-68.3	4.25	0.81
System 4	0	0	11.9	0	0	NA	-77.6	7.75	0.79
System 5	0.003	0	0	-1.94	0	NA	17.9	3.38	0.46

The multiple regression equation (*Variable to be predicted = a× (electro conductivity, µs) + b × (turbidity, NTU) + c ×(pH) + d ×(orthophosphate-phosphorous, mg/l) + e ×(nitrate-nitrogen, mg/l) + f × (ammonia-nitrogen, mg/l) + g*) was fitted. [a]standard error of the estimate; [b]coefficient of determination. NA, not applicable.

TABLE 5. MULTIPLE LINEAR REGRESSION ANALYSES APPLIED TO PREDICT THE FIVE-DAYS AT 20°C (NITRIFICATION INHIBITOR APPLIED) BIOCHEMICAL OXYGEN DEMAND (BOD, MG/L), AND SUSPENDED SOLIDS (SS, MG/L).

As indicated in Table 5, the application of multiple linear regression analyses for the prediction of BOD was relatively successful when applied to samples from the inflow, and systems 1, and 2. This has been attributed to a high correlation between BOD, and most of the selected predictors. Moreover, as there has been no strong correlation between BOD, and other key water quality variables for system 5, a multiple regression analysis was not performed.

Standard errors of the estimates for suspended solids were higher than the corresponding ones for the BOD. The coefficients of determination (r^2) are relatively high for all systems with the exception of system 5. However, multiple regression analysis is not successful in predicting suspended solids if a considerable number of outliers are part of the corresponding dataset.

Analyses of variance

An one way ANOVA was conducted to test if the systems performed similarly concerning storm water treatment. The outcome of this analysis allows the design engineer to opt for a system that performs well and is cost-effective. For example, if there is no significant difference between the performances of two different systems for the most important key variables, the designer would be well advised to choose the less costly option.

There were significant differences in treatment performances concerning BOD, ammonia-nitrogen, total coliforms, suspended solids, and intestinal enterococci with F values (ratio of the mean variance between groups divided by the mean variance within groups) of 5.3, 8.0, 10.0, 3.6, and 4.1 respectively.

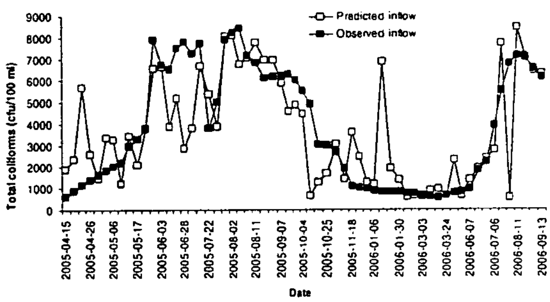
Furthermore, the ANOVA indicated that there were significant ($p<0.05$) differences between some of the water quality parameters in the inflow, and outflow for each system. Significant differences with respect to system 1 were found for total dissolved solids, turbidity, electrical conductivity, dissolved oxygen, ortho-phosphate-phosphorous, nitrate-nitrogen, intestinal enterococci counts, and total coliform counts with F values of 2085.2, 8.9, 2006.7, 3.4, 48.2, 69.9, 45.0, and 42.0, respectively. For system 2, there were significant differences found for turbidity, ortho-phosphate-phosphorous, and nitrate-nitrogen with F values of 35.5, 18.6, and 26.8, respectively. Concerning system 3, the ANOVA shows significant differences for BOD, total dissolved solids, dissolved oxygen, ortho-phosphate-phosphorus, ammonia-nitrogen, intestinal enterococci, and total coliforms with F values of 4.5, 10.7, 1.9, 20.2, 3.5, 225.5, and 7129.3, respectively. The results from system 4 showed significant differences for suspended solids, electric conductivity, ortho-phosphate-phosphorous, nitrate-nitrogen, intestinal enterococci counts, and total coliforms counts with F values of 6.0, 20.3, 12.6, 74.9, 26.6, and 126.3, respectively. Finally, an ANOVA for system 5 detected significant differences for turbidity, dissolved oxygen, ortho-phosphate-phosphorous, nitrate-nitrogen, intestinal enterococci counts, and total coliform counts with F values of 83.5, 3.7, 10.8, 301.7, 6.4, and 127.6, respectively.

Artificial neural network modelling

The coefficient of determination (R^2) values for predicting total coliform counts for the inflow and outflows of systems 1 to 5 were 0.89, 0.94, 0.91, 0.98, 0.59 and 0.95, respectively. The corresponding R^2 values for predicting intestinal enterococci counts were 0.80, 0.63, 0.78, 0.73, 0.71 and 0.15, respectively. It follows that the models were able to successfully predict the total coliform, and intestinal enterococci colony forming unit counts with an exception for the prediction of intestinal enterococci in system 5.

Figures 5 and 6 show the observed, and predicted total coliform and intestinal enterococci counts in the inflow and the Outflows of all systems, respectively. The artificial neural networks successfully predicted total coliform, and intestinal enterococci counts for the inflow water. The models were very successful in predicting total coliform counts for all systems except for system 4. Concerning intestinal enterococci counts, the models were relatively successful. When predicting total coliform counts with the artificial neural network models for the inflow, and systems 1, 2, 3, and 5, one can undertake predictions confidently resulting in mean squared errors close to zero. In the case of intestinal enterococci counts, the inflow and systems 2, 3 and 4 had similar R^2 values.

a)



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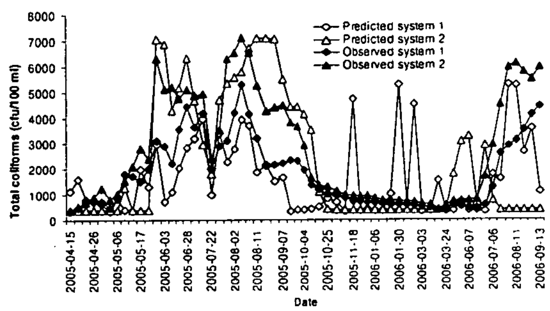

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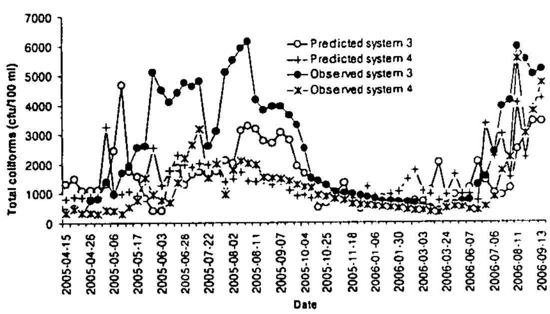

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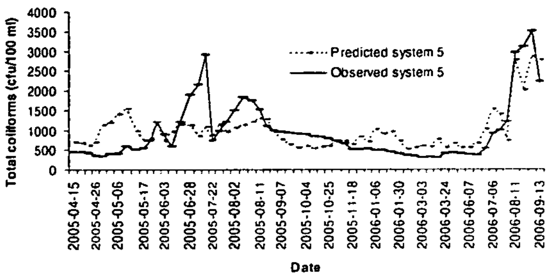
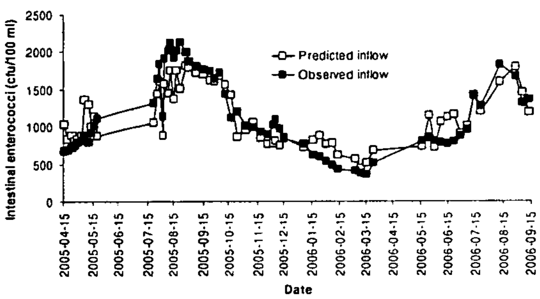
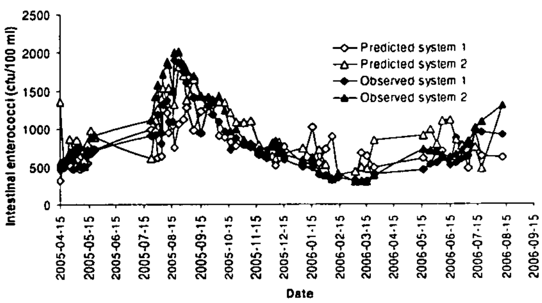


FIG. 5. OBSERVED, AND PREDICTED TOTAL COLIFORM COUNTS IN (A) THE INFLOW AND THE OUTFLOWS OF (B) SYSTEMS 1, AND 2, (C) SYSTEMS 3, AND 4, AND (D) SYSTEM 5.

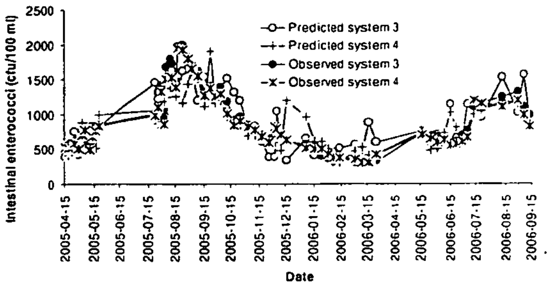
a)



b)



c)



d)

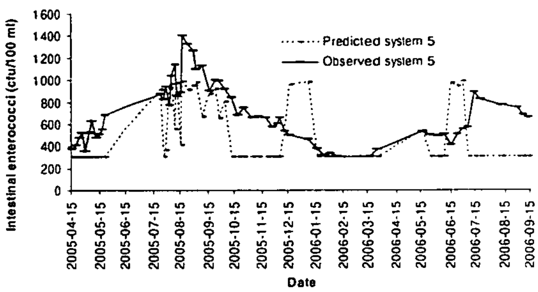


FIG. 6. OBSERVED, AND PREDICTED INTESTINAL ENTEROCOCCI COUNTS IN (A) THE INFLOW AND THE OUTFLOWS OF (B) SYSTEMS 1, AND 2, (C) SYSTEMS 3, AND 4, AND (D) SYSTEM 5.

CONCLUSION

(1) An analysis of variance showed significant differences between different experimental system performances in treating concentrated road runoff. Systems containing turf showed better biochemical oxygen demand, and suspended solids removal performances in comparison to less complex systems without turf. However, the assessment was unclear with respect to microbiological indicator variables.

(2) Multiple regression analyses indicated a relatively successful prediction of the biochemical oxygen demand but unsuccessful predictions of both total coliform,

and intestinal enterococci counts. However, artificial neural network models predicted both total coliform, and intestinal enterococci counts relatively well.

(3) The artificial neural networks successfully predicted total coliform, and intestinal enterococci counts for the inflow water (R^2 values of 89%, and 80%, respectively). The models were highly successful in predicting microbial counts for most systems. Predictions resulted in mean squared errors close to zero.

(4) The results of this study show that the artificial neural network models developed for the prediction of the total coliform counts, and the intestinal enterococci counts have performances consistent with other findings reported in the literature. However, the relatively low R^2 values reported for some systems, and more specifically for predicting intestinal entercocci counts in the densely planted system 5 indicate the difficulty in identifying the necessary explanatory variables to characterize a large percentage of the variability observed in the microbial dataset. In cases where the water quality standards are observed for total coliform and intestinal enterococci counts, artificial neural networks provide a good modeling technique to predict a potential violation.

(5) The model could be applied outside the experimental setup for similar problems. The main condition is that the boundary conditions are comparable. Otherwise, the model would require retraining.

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Appendix 3

How common goldfish could save cities from flooding

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ABSTRACT

Recently, there has been wide national (various British newspapers) and even international (German radio) public interest in Scottish experiments introducing *Carassius auratus* (common goldfish) into sustainable urban drainage systems (SUDS) applied to combat flooding. Moreover, dog faeces are added to these systems to simulate contaminated urban runoff. The purpose of this novel and timely research study is to increase public acceptance of zero discharge infiltration ponds, and to control algal growth with *C. auratus*. Findings show that *C. auratus* are improving most water quality variables after their introduction to a planted and an unplanted infiltration pond despite of a deterioration of virtually all common inflow water quality variables based on an annual comparison. Public interest is high because the study captures the imagination of the urban population facing re-occurring flooding problems in autumn in low-lying areas, and the nuisance of dog excrements despite of new regulations to scoop up droppings.

Keywords: Algae; Dog; Faeces; Goldfish; Pond; Sustainable Urban Drainage Systems.

1. Introduction

Conventional storm water systems are designed to dispose of surface water runoff as quickly as possible. This results in 'end of pipe' solutions that often involve the provision of large interceptor and relief sewers, huge storage tanks at downstream locations and centralized wastewater treatment facilities. These traditional civil engineering solutions often lead to flooding and environmental pollution due to combined sewer overflows during storm events [1,2].

In contrast, sustainable urban drainage systems (SUDS) such as combined attenuation and infiltration systems can be applied as cost-effective local 'source control' drainage solutions; e.g., delaying storm runoff leads to a reduction of the peak flow [3]. It is often possible to divert all storm runoff for infiltration or storage and subsequent water reuse. As runoff from roads is a major contributor to the quantity of surface water requiring disposal, this is a particularly beneficial approach where suitable ground conditions prevail [4]. Furthermore, infiltration of storm runoff can reduce the concentration of diffuse pollutants such as dog faeces and leaves, thereby improving the water quality of surface water runoff [5].

Cities have now found a new ally in their battle against the scourge of flooding: *Carassius auratus* (Linnaeus, 1758), usually known as common goldfish [4,6,7]. Artificial ponds as part of SUDS should be used to hold storm runoff waters in cities - and *C. auratus* helps to increase public acceptance, and to keep them clean. Plans are being drawn up for the world's first 'goldfish-friendly' housing estates in Glasgow and Edinburgh; e.g., two attenuation ponds in the Ruchill Hospital and Park area [8]. As a result, hundreds of fish could soon be nibbling unsightly, and sometimes smelly mats of algae and helping to keep estates aesthetically pleasing and good smelling. Every time a new housing estate is build in the city, flood problems may arise due to a lack of existing sewer system capacity. One solution is to create ponds and small lakes that will attenuate storm runoff water and/or allow it to slowly filter back into the ground [1,8,9].

The aim of this short and rapid communication is to show that SUDS can be kept clean, healthy and pleasing to look at with the help of *C. auratus*. The impact of *C. auratus* on the water quality of ponds and the associated inflow water quality on *C. auratus* will be assessed.

2. Materials and methods

Since 1 April 2003, a planted and an unplanted runoff demonstration pond as part of SUDS at the King's Buildings campus of The University of Edinburgh are in operation (Figs. 1 and 2). The dominant macrophyte of the constructed wetland and planted pond were *Phragmites australis* (common reed) *and Typha latifolia* (broadleaf cattail), respectively.

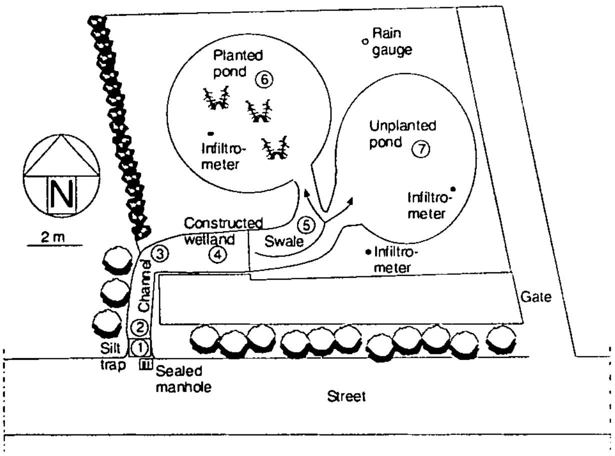

Figure 1. Runoff flows from the road (partly shown) into the silt trap (1), then via the gravel ditch (2) into the constructed wetland (3 and 4) and finally via the swale (5) into the infiltration ponds (6 and 7).

Precipitation from a road of the campus was channelled to infiltration ponds but (filamentous) green algae began to grow - until *C. auratus* (Fig. 3) were introduced on 1 April 2004. Twenty healthy *C. auratus* of approximately 180 g total weight were introduced into each pond. Both watercourses were covered with a plastic mesh (Fig. 2) to prevent animals such as *Ardea cinerea* (grey heron) and *Felis cattus* (cat) to prey on *C. auratus* [4].

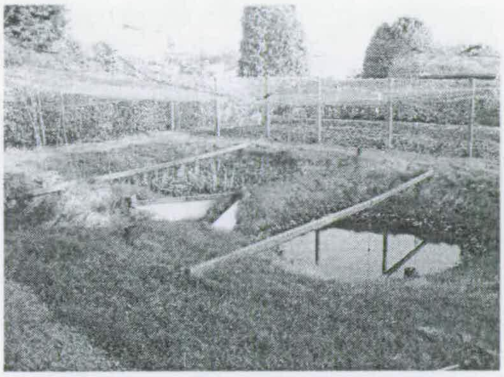

Figure 2. Case study site (picture taken by M. Scholz on 12 April 2005)

Since 1 April 2005, approximately 400g/week of fresh dog excrements are currently added directly to the silt trap protecting the ponds predominantly from solid contaminants. A constructed wetland located between the silt trap and the ponds should prevent contamination from dissolved organic pollutants and potentially pathogenic organisms, but it is an open research question if *C. auratus* can cope with any additional nutrient (particularly nitrogen and phosphorus) load and total coliforms including *Escherichia coli* (pathogenic bacteria) build-up. All water quality determinations were undertaken according to standard methods [10]. Further research on *C. auratus* is ongoing (see below and [7]).

3. Results and discussion

3.1. Water quality assessment and management

The water qualities of the constructed wetland inflow, constructed wetland outflow, unplanted infiltration pond and planted infiltration pond (Figs. 1 and 2) are shown in Tables 1, 2, 3 and 4, respectively.

Table 1. Summary statistics: water quality of the inflow to the constructed wetland (Fig. 1) before (01/04/03-31/03/04) and after (01/04/04-31/03/05) the introduction of *C. auratus* (common goldfish)

Variable	Unit	Sampling number		Mean		Standard deviation	
Temperature	°C	54	94	9.9	12.1	3.15	3.92
BOD[a]	mg/l	36	43	10.6	17.7	14.59	14.20
Suspended solids	mg/l	47	93	152.4	1174.0	245.45	1458.11
Ammonia-N	mg/l	32	73	0.5	1.1	0.77	2.54
Nitrate-N	mg/l	28	70	1.8	1.7	3.16	7.55
Phosphate-P	mg/l	32	73	0.09	0.68	0.085	1.144
Conductivity	μS	56	93	246.2	209.7	200.54	149.44
Turbidity	NTU	56	94	105.8	695.4	167.58	839.42
Dissolved oxygen	mg/l	18	94	4.6	3.3	1.59	1.15
pH	-	56	94	7.0	7.1	0.76	0.21

[a]five-day at 20°C biochemical oxygen demand.

Table 2. Summary statistics: water quality of the outflow to the constructed wetland (Fig. 1) before (01/04/03-31/03/04) and after (01/04/04-31/03/05) the introduction of *C. auratus* (common goldfish)

Variable	Unit	Sampling number		Mean		Standard deviation	
Temperature	°C	56	94	9.7	12.0	3.14	4.01
BOD[a]	mg/l	36	43	6.0	14.2	8.01	17.17
Suspended solids	mg/l	42	93	100.7	366.9	117.19	582.08
Ammonia-N	mg/l	30	73	0.2	0.4	0.15	0.43
Nitrate-N	mg/l	26	70	2.1	2.6	1.33	5.46
Phosphate-P	mg/l	31	73	0.08	0.30	0.040	0.496
Conductivity	μS	58	93	171.5	124.4	98.05	58.15
Turbidity	NTU	58	94	68.5	184.9	126.90	268.66
Dissolved oxygen	mg/l	52	94	6.0	3.8	8.67	1.15
pH	-	58	94	7.0	7.0	0.69	0.34

[a]five-day at 20°C biochemical oxygen demand.

Table 3. Summary statistics: water quality of the planted pond (Figs. 1 and 2) receiving the outflow from the constructed wetland before (01/04/03-31/03/04) and after (01/04/04-31/03/05) the introduction of *C. auratus* (common goldfish)

Variable	Unit	Sampling number		Mean		Standard deviation	
Temperature	°C	56	94	8.7	11.6	4.60	4.90
BOD[a]	mg/l	36	43	15.5	19.3	18.91	14.25
Suspended solids	mg/l	47	93	58.7	24.7	116.61	55.45
Ammonia-N	mg/l	34	71	0.3	0.1	0.58	0.21
Nitrate-N	mg/l	28	69	0.7	0.4	2.25	0.84
Phosphate-P	mg/l	33	72	0.18	0.25	0.149	0.238
Conductivity	μS	58	93	310.5	246.9	116.86	83.21
Turbidity	NTU	58	94	18.4	14.2	20.02	29.84
Dissolved oxygen	mg/l	52	94	6.1	3.5	7.01	1.54
pH	-	57	94	7.2	7.2	0.24	0.25

[a]five-day at 20°C biochemical oxygen demand.

Table 4. Summary statistics: water quality of the unplanted pond (Figs. 1 and 2) receiving the outflow from the constructed wetland before (01/04/03-31/03/04) and after (01/04/04-31/03/05) the introduction of *C. auratus* (common goldfish)

Variable	Unit	Sampling number		Mean		Standard deviation	
Temperature	°C	56	94	8.6	11.6	4.74	5.35
BOD[a]	mg/l	33	43	18.1	19.2	24.80	14.38
Suspended solids	mg/l	44	93	25.6	21.7	52.03	39.57
Ammonia-N	mg/l	33	73	0.6	0.8	1.49	2.25
Nitrate-N	mg/l	30	71	0.8	1.3	2.27	5.91
Phosphate-P	mg/l	35	73	0.24	0.35	0.458	0.551
Conductivity	μS	58	93	220.6	193.4	139.62	88.04
Turbidity	NTU	58	94	12.6	19.1	19.45	36.92
Dissolved oxygen	mg/l	52	94	6.4	4.2	10.80	1.75
pH	-	58	94	7.2	7.3	0.46	0.29

[a]five-day at 20°C biochemical oxygen demand.

After two years of operation, the water quality of the unplanted infiltration pond (Tables 3 and 4) was acceptable for disposal and recycling according to discussions particularly on potential five-day at 20°C biochemical oxygen demand and suspended solids threshold concentrations (e.g., 20 and 30 mg/l, respectively) [2-4]. However, water quality monitoring is currently not required for closed drainage systems (zero discharge) in Scotland [4].

Suspended solids and turbidity measurements are high in the constructed wetland inflow due to high loads of organic material such as decomposing leaves (Table 1). These concentrations are reduced due to treatment within the constructed wetland (Tables 1 and 2). Nevertheless, nutrient concentrations were sufficiently high to cause an algal bloom. Nitrate-nitrogen actually increased in the constructed wetland due to nitrification of ammonia-N. Mats of algae swimming partly on top of a watercourse are usually considered unpleasant in their appearance by the public [1,4,11].

3.2. Active control of algae with C. auratus

Algae began to grow in the infiltration ponds until *C. auratus* (Fig. 3) were introduced on 1 April 2004. The result was a pleasant and clean SUDS during the second year of operation despite fears of water quality deterioration voiced elsewhere [11] (Tables 3 and 4).

Figure 3. Miklas Scholz presenting a *C. auratus* (common goldfish) to The Observer (picture taken by M. MacLeod on 21 April 2005 [7])

Carassius auratus (similar to *Cyprinus carpio* or also known as common carp) is classified as herbivores with wild specimens predominantly feeding on plants. This particularly applies to closed pond systems (Figs. 1 and 2). Therefore, *C. auratus* could be used to control aquatic weeds and potentially algae in ponds [4,11-13].

Concerning the field experiment, relatively high numbers of filamentous green algae (Chlorophyta) were counted in pond samples taken on 29 March 2004. The dominant alga present was *Odeogonium capillare* that is cosmopolitan in freshwater. *Odeogonium capillare* can form mats in small ponds, and is often mistaken for the more common *Cladophora glomerata* (blanket weed) [4].

Carassius auratus was introduced to control predominantly filamentous green algae and to increase public acceptance of SUDS. Concerning samples of algae taken on 4 October 2004, both the unplanted and planted ponds were less dominated by *Odeogonium capillare* in comparison to estimations on 29 March 2004. Moreover, the unplanted pond developed a greater diversity of filamentous green algae if compared to the planted pond. This may be due to the absence of macrophytes that would compete with algae for nutrients (particularly phosphorus). Moreover, large macrophytes (located in the planted pond; Fig. 1) provide shade leading to a reduction of sunlight penetrating the water, and subsequently reducing the growth of algae [1,4].

Nevertheless, the estimated algal biomass was considerably higher (at least one order of magnitude) in the planted if compared to the unplanted pond on 28 April 2005. This can be explained with the obvious observation that algae are the dominant (virtually only) plant food source in the unplanted pond.

3.3. Integration of SUDS into urban planning

Flood protection management and the recreational value of urban landscapes can be improved at the same time by integrating SUDS (in contrast to conventional drainage) into the urban planning and development processes. Recreational activities may include watching ornamental fish such as *C. auratus* (Fig. 3) and birds, walking, fishing, boating, holding picnics and teaching children about aquatic ecology [2,8]. The confidence of town planners towards SUDS and public acceptance of infiltration ponds can both be increased by correct dimensioning of sustainable systems [4] to avoid flooding, enhance water pollution control by using a robust pre-treatment train (e.g., silt trap, constructed wetland and swale; Fig. 1) [5] and control algae by biological (e.g., *C. auratus*) and not chemical (e.g., copper sulphate) means [3,4]. Moreover, storm water can be reused for watering gardens and flushing toilets as part of an urban water resources protection program [2,3,4].

3.4. Urban water hygiene

The issue of urban water hygiene requires consideration. Runoff water could sweep some animal faeces into SUDS. Particularly dog faeces being carried in by floodwaters are a problem in urban environments despite local government efforts to encourage dog owners to scoop up droppings [7].

Can *C. auratus* cope with the additional nutrient load? Could there be a potentially dangerous build up of *E. coli* from excrements? Further detailed research on the health of *C. auratus* is therefore in progress.

Preliminary findings indicate that the additional nutrient load is very small in comparison to the background load (e.g., leaves and soil), and that no accumulation of bacteria in the system is detectable. After this experiment, *C. auratus* will be introduced to SUDS sites within cities such as Glasgow and Edinburgh. Construction work in the Ruchill Hospital estate will commence in autumn 2005.

4. Conclusion

The research has attracted wide national and even international public interest; e.g., The Observer, sundayherald, THE SUNDAY POST, Daily Mail and Deutschlandfunk. The public relate well to their urban environment and common goldfishes (*Carassius auratus*), which are often used as pets in aquariums and garden ponds. Moreover, the rather unappetising character of experiments with dog droppings interests the public – usually a taboo for both the 'dog-loving' public (an obvious but very 'human' contradiction), and even the scientific and engineering community (first scientific study according the Web of Knowledge and Scopus).

Findings show that the introduction of *C. auratus* to closed (zero discharge) systems such as infiltration ponds do improve most water quality variables (e.g., reductions of algae suspended solids, nitrate-N and turbidity), and should lead to an increase in public acceptance of SUDS. Moreover, the water qualities of the infiltration ponds were acceptable for water reuse (below likely future thresholds for biochemical oxygen demand and suspended solids) after the set-up period of the SUDS and the introduction of *C. auratus*. A bloom of filamentous green algae dominated by *Odeogonium capillare* during the springs of 2003 and 2004 were observed.

However, *Carassius auratus* decimated the algae particularly well in the unplanted pond, where no other (plant) food source was abundant.

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Appendix 4

Combined storm water treatment, detention and infiltration system treating road runoff

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ABSTRACT

Storm water detention devices collect runoff from impermeable catchments. They provide flow attenuation as well as storage capacity, and rely on natural self-purification processes such as sedimentation, filtration and microbial degradation. The aim was to assess the performance of an experimental combined planted gravel filter, storm water detention and infiltration tank system treating runoff from a car park and its access road. Flows were modelled with the US EPA Storm Water

Management Model. An overall water balance of the system was compiled, demonstrating that 50% of the rainfall volume escaped the system as evaporation, whereas, of the remaining 50%, approximately two thirds were infiltrated and one third was discharged into the sewer system. These findings illustrated the importance of evaporation in source control, and showed that infiltration can be applied successfully even on man-made urban soils with low permeability. The assessment of the system's hydrological efficiency indicated mean lag times of 1.84 h and 10.6 h for the gravel filter and the entire system, respectively. Mean flow volume reductions of 70% and mean peak flow reductions of 90% were achieved compared to conventional drainage. The assessment of the pollutant removal efficiency resulted in promising removal efficiencies for biochemical oxygen demand (77%), suspended solids (83%), nitrate-nitrogen (32%) and ortho-phosphate-phosphorus (47%). The most important removal processes were identified as biological degradation (predominantly within the gravel ditch), sedimentation and infiltration.

Keywords Attenuation; below ground storm water detention tank; bio-filtration trench; infiltration; lag period; ortho-phosphate-phosphorus; road and car park runoff; US EPA Storm Water Management Model; water quality; willow

1. Introduction

1.1. Background

Urban non-point source pollution has a significant contribution to water quality degradation (Brezonik and Stadelmann, 2002). Urban pollutants enter aquatic systems mainly via runoff and therefore contributing to water and soil contamination (Mangani et al., 2005) The mechanisms promoted in the removal of storm water pollutants include physical, chemical and biological processes. Because of the intermittent nature of storm water inflow, physical processes associated with detention for sedimentation and filtration (either through vegetated systems or through an infiltration medium) are the main mechanisms by which storm water contaminants are first intercepted, further chemical and biological processes then can influence the transformation of these contaminants (Wong *et al.*, 2006).

Urban storm water runoff discharged through sewer systems into streams may cause flush spills of water and pollutants in the receiving waters (Giller *et al.*, 1996). The quantitative and qualitative impacts of storm water runoff have become a main concern in urban development design. Traditionally, methods utilizing above ground structures such as storm water management ponds were used to address the quantitative impacts of increased peak flows and increased runoff volumes, through storm water detention, retention and/or attenuation. However, insufficient space, high land values, topography, maintenance, aesthetics and liability issues are some of the reasons to motivate designers to consider below ground detention systems more frequently (Finlay, 2000).

Conventional below ground detention tanks are traditionally used to reduce increased peak storm water flows from small developments. These systems limit peak flow from short duration storms by providing temporary storage for runoff and releasing it through an outlet structure, either manually or at the expense of increased flow duration (NSC Council, 2002).

Traditional below ground detention tanks require frequent inspection and maintenance to keep their optimum performances, and some design improvements are recommended to improve their reliability. When inspected, most tanks already in service are found to be blocked (in a recent survey in Sydney, 90% of tanks were blocked (Mitchell, 2005) or disabled by property owners (by manipulating the outlet structures to reduce the blockage problems) (NSC Council, 2002). However, the growing concern for more environmentally sustainable practices have led to a more precise assessment of existing systems and design principles and to the development of numerous alternative approaches (Verworn, 2002).

There are tendencies to improve existing urban drainage systems rather than to design and construct completely new ones (Verworn, 2002). Storm water is being regarded as a source to be managed. This includes the criteria of source control, in which storm water runoff is not only being stored but also treated (via filtration or infiltration) within these systems, at or close to its generation point (Hatt *et al.*, 2004). Therefore, there is a continuing demand for below ground detention tanks as the most proper solution in many situations (NSC Council, 2002).

Detention systems are subsurface structures specifically designed to accumulate surface runoff, and to release it, as may be required to increase the flow hydrograph. The structure can be modified to contain aggregates with a high void ratio, and act also as a water recycler or infiltration device. The surface water is being captured and then filtered through a combined gravel and sand filter. The filtered storm water is detained below ground within a detention tank (Butler and Parkinson, 1997).

In detention systems, runoff is treated by filtration prior to infiltration or discharge to the sewer or watercourse via a discharge control valve. The application of these systems reduces runoff in case of minor storms as well as encourages groundwater recharge, and pollution reduction. These detention systems can frequently be found in new developments (Scholz, 2006).

There is an urgent need to modify common storm water detention systems to meet more stringent water quality guidelines (Butler and Parkinson, 1997; Scholz, 2006). Research is needed to focus on the implementation of sustainable filters within the current structures of detention systems.

1.2. Rationale, aim and objectives

In this study, the authors recommend a modern approach towards the use of below ground detention systems. The system being recommended here is a combined filtration, detention and infiltration device. This combined system assists in the control of both the quantity of runoff, through onsite detention and the quality of runoff through filtering and infiltration of runoff.

This research study was aimed to assess the performance of a novel sustainable (urban) drainage system (SUDS) comprising a vegetated gravel-filled bio-filtration trench, and a below ground storm water detention and infiltration system. The objectives were to assess the water balance, hydraulic flows, physical, chemical and biological processes, and the overall water treatment performance.

2. Site and methodology

2.1. Water balance

An overall water balance of the system was compiled, which demonstrates that approximately 50% of the precipitation volume escaped the system as evaporation, while of the remaining 50%, approximately two thirds were infiltrated and one third was discharged directly to the sewer system via an emergency overflow system during heavy storm events. These findings illustrate the significance of evaporation in source control. The experiment also indicates good infiltration even on urban man-made compacted soils with perceived low permeability (Butler and Davies, 2006).
The evaporation rate of the gravel filter was relatively higher than the one of the car park. This effect is due to different surface properties: runoff from the car park takes only a short time, as the surface is impermeable. Moreover, only an estimated 1.5 mm of depression storage water is available for evaporation. Runoff does not infiltrate into the soil below the sealed gravel filter. Therefore, a free water surface is maintained for a relatively long period of time, permitting proportionally more

evaporation from the gravel filter than from the car park and road. It follows that the system does fulfil the purpose of attenuating the runoff from the catchment area.

The assessment of the system's hydrological efficiency yielded mean lag times of 1.84 h for the gravel filter and 10.6 h for the entire system. Mean flow volume and mean peak flow reductions of approximately 70 and 90%, respectively, were achieved.

An examination of the minimum, mean and maximum values for rainfall volume, duration and intensity showed strongly skewed distributions; i.e. the majority of the rainfall events were weak and storm events were rare. It follows that a longer measurement period would have been advantageous to test for the worst case rainfall scenario.

Considering the physical properties of the soil, a curve fitting exercise in US EPA Storm Water Management Model (SWMM) showed a good approximation of the infiltration behavior. The maximum infiltration rate observed by curve fitting in SWMM was around 1.3 mm/h which corresponds to a saturated groundwater flow velocity of 3.6×10^{-7} m/s.

2.2. Modelling

The key objects and parameters used for the calibration of the SWMM are summarized in Tables 1 and 2, respectively. Figure 1 shows the final set-up of the model in SWMM. The catchment area is represented by the sub-catchment Car_Park in the model. The gravel filter is represented by another sub-catchment (Gravel_Filter), which is linked to an aquifer object of the same name (not shown on the map) to model subsurface flow. Stormwater drains from the aquifer into the storage unit named Detention_Tank.

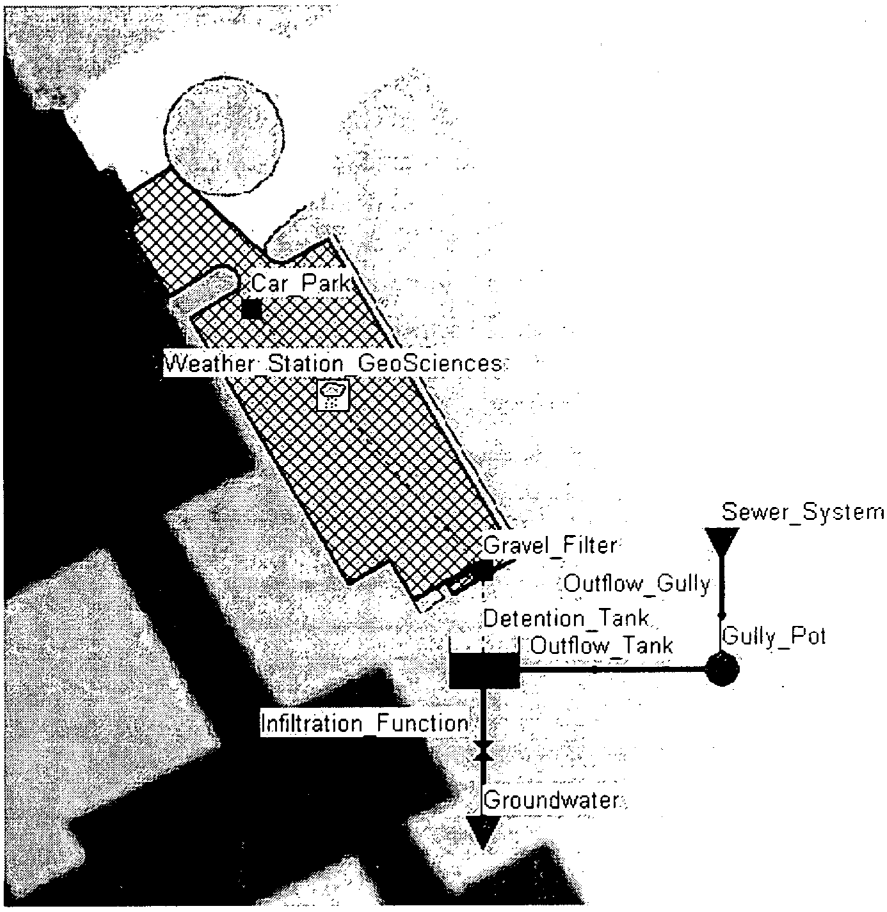


Fig1. Set-up of the Storm Water Management Model with the original object names.

Object name	Object type	Description
Car park	(Sub-)catchment	Input of precipitation data; output of losses (evaporation and infiltration) and surface runoff (routed to gravel filter)
Gravel filter A	(Sub-)catchment	Input of precipitation data and surface runoff from car park; output of losses, runoff and groundwater flow
Gravel filter B	Simulated 'aquifer'	'Aquifer' beneath the sub-catchment gravel filter; input of simulated groundwater flow; outflow to storage unit
Detention tank	Storage unit	Storage of inflow; two outflows are associated with the tank: infiltration function and outflow tank
Infiltration function	Outlet	Flow regulator simulating infiltration with a head-discharge relationship
Outflow tank	Conduit	Link simulating the connection between overflow tank and gully pot system
Groundwater	Outfall	Terminal node for infiltration into soil

Table 1 Essential objects used for the calibration of the Storm Water Management Model.

OBJECT AND PARAMETER	UNIT	INITIAL VALUE	RANGE OF VARIATION
TIME STEP			
RUNOFF (WET WEATHER)	MIN	10	1-15
ROUTING	MIN	0.5	0.2-1.0
EVAPORATION	%	100	70-100
CAR PARK			
WIDTH OF OVERLAND FLOW PATH	M	13	10-15
SLOPE	%	1.0	0.5-2.0
DEPRESSION STORAGE (IMPERMEABLE AREA)	MM	2.5	1.5-5.0
GRAVEL FILTER A			
MINIMUM INFILTRATION RATE	MM/H	60	30-100
MAXIMUM INFILTRATION RATE	MM/H	120	80-140
DECAY CONSTANT	1/H	1.0	0.1-5.0
GRAVEL FILTER B			
CONDUCTIVITY	MM/H	60	30-100
CONDUCTIVITY SLOPE	MM/H	10	1-50
TENSION SLOPE	MM	15	5-50

Table 2. Essential parameters for key objects used for the calibration of the Storm Water Management Model.

2.3. Site description and system layout

The SUDS investigated in this study is situated on The King's Buildings Campus at The University of Edinburgh. It was constructed in March and April 2006, and has been in operation since May 2006. The system comprises a filtration unit planted with willows (combined sand and gravel filter) and a subsurface detention and infiltration unit (storm water tank or soakaway). The system is designed as a source control device aiming to detain and treat storm water runoff from a small adjacent car park. The car park and part of the roundabout next to it form the catchment of the system. The catchment has an area of 640 m^2 and the mean slope is approximately 1.0%. The entire area is covered by asphalt.

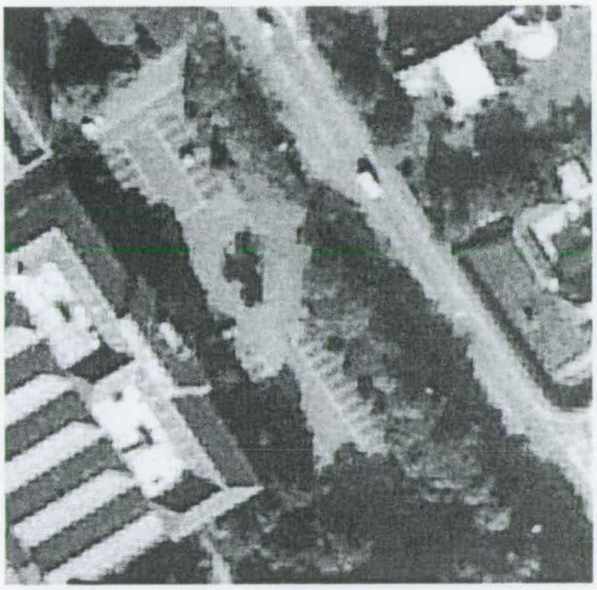
Runoff from the car park enters the slightly inclined gravel filter through a curbside inlet. At the lower end of the gravel filter, storm water accumulates and is then transferred to the detention tank via an opening covered with geotextile. Storm water is then either detained in the tank, allowing pollutant degradation and infiltration into the soil; or discharged to the sewer system through an outlet structure connected to the adjacent modified gully pot. Direct discharge to the sewer system occurred occasionally when the water level in the tank exceeded 0.55 m (the maximum height allowed in the tank).

The filter drain is formed like half a truncated cone or frustum. It is 4.5 m long, 1.3 m wide at the inlet and 1.4 m wide at the outlet. The depth is 0.3 m at the inlet and 0.5 m at the outlet. The total volume of the filter drain is 4.4 m^3 with a surface area of 7.0 m^2.

The filter comprises three layers; the uppermost layer consists of gravel with a mean grain size of 20 mm and has a thickness of 0.05 m; the middle layer consists of gravel with a mean grain size of 6 mm and has a thickness between 0.15 and 0.25 m; the lowest layer consists of a mixture of sand (60% by volume), EcoSoil® produced by Alderburgh® Limited (30% by volume) and woodchips (10% by volume), and has a thickness between 0.10 and 0.20 m.

The middle layer and the lowest layer are separated by geotextile. The lowest layer is isolated with a plastic liner against the surrounding soil to prevent seepage into or out of the filter drain. The filter was planted with willows to encourage nutrient removal as well as to increase the aesthetic appeal, thus making the system more pleasing to the public. Figure 2 shows an aerial photograph of the system's location and a view on the structure during a storm event.

(a)



(b)

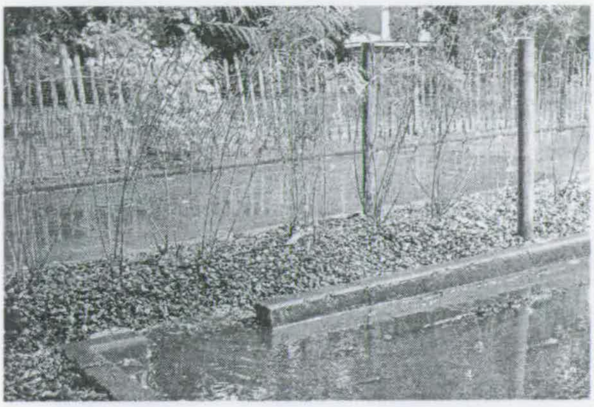


Figure 2. Storm water detention pilot plant site (a) from above (centre of picture; south-west to the street; Cities Revealed, copyright by The GeoInformation Group); and (b) during a heavy storm event in spring 2007.

The tank consists of plastic modules and is covered on top and at the bottom with a geotextile. The sides are isolated with plastic liner. The individual rows of cells are separated by plastic liners to achieve a longer flow path through the tank. The Matrix® II tank modules were supplied by Alderburgh® Limited. They have a void ratio of 90% according to manufacturer's specifications. The tank is made up of 132 modules and has a surface area of 18.45 m^2 and a total void volume of 14.94 m^3. The dimensions of the single modules and the tank are shown in Table 3.

Dimensions of Single Module		Dimensions of Tank	
Length (m)	0.408	Length (m)	4.49
Width (m)	0.685	Width (m)	4.11
Depth (m)	0.450	Depth (m)	0.90
Number of Modules used		Surface Area (m^2)	18.45
For length of tank	11	Volume (m^3)	16.60
For width of tank	6	Void Volume (m^3)	14.94
For depth of tank	2	Number of modules	132

Table 3. Dimensions of a single module and of the entire tank.

Sampling points are located at the inlet (number 1) and at the outlet (number 2) of the gravel filter and along one side of the tank (numbers 3 to 8). They consist of plastic pipes, fitted with geotextile to prevent solids entering the tank and covered on top with a plastic cap. The tank is also equipped with six aeration pipes (holes drilled in the plastic), which are located between the sampling pipes.

Samples were taken approximately twice per week during the period between June 2006 and November 2007. Water samples were collected from the six sampling points in the tank and – if there was sufficient water present – from the two sampling points in the filter drain. The water depth within the tank, air and water temperature and the oxygen content of the water samples were measured directly on-site. All other parameters were measured in the laboratory. Water depth was determined by lowering a rod down the sampling points and reading off the water level.

For dissolved oxygen and temperature measurements, a Hanna HI 9142 portable waterproof dissolved oxygen meter was used between June and August 2006. From

September 2006, a WTW oxygen meter was applied. For the measurement of pH, electrical conductivity (EC) and total dissolved solids (TDS), a Hanna HI 991300 portable meter was used. For the measurement of turbidity, a Hach-Lange 2100 turbidimeter was used. The oxidation-reduction potential (ORP) was determined with a Hanna HI 98201 pocket-sized redox meter. Total solids (TS) were measured by weighing water samples in glass beakers (200 ml), drying them for 48 h at a temperature of approximately 105°C in an oven and subsequently weighing the dry beakers. Suspended solids (SS) were measured by filtering water samples (between 50 and 200 ml, depending on the amount of solids present) with Whatman glass microfiber filters with a pore diameter of 200 μm. The amount of SS was measured by weighing the filters, drying them for 48 h at a temperature of approximately 105°C, and weighing the dried filters. After taking them out of the oven, beakers and filters were left to cool for 15 minutes before weighing. The biochemical oxygen demand after 5 days (BOD_5) was determined under the influence of a nitrification inhibitor with the OxiTop manometric measuring system manufactured by the Wissenschaftlich-Technische Werkstätten GmbH (WTW).

Some datasets from the storm water tank (i.e. sampling point numbers 3 to 8) were not complete due to sampling problems (e.g. water depth not sufficient for sampling) or due to measurement problems. For datasets with one or two missing values, the missing values were interpolated by fitting a polynomial curve to the measured values, applying the software package Microsoft Excel. The software calculates the best fit of the curve to the data by using the least squares method. The polynomial best fit curve was chosen, because the datasets did not show logarithmic, potential or exponential behaviour, but rather irregular or even random fluctuations. Datasets with three or more missing values were not used for analysis. Precipitation data were obtained from the weather station at The King's Buildings Campus (The University of Edinburgh).

2.4. Hydraulic flow modelling

Hydraulic flows were modelled with the US EPA program Storm Water Management Model (SWMM, Version 5.0; Rossman, 2005). The model was calibrated with water level data measured on site. Calibration of the model was carried out by comparing the real water level in the tank with the predictions made by the model. The calibration parameters were adjusted until a good fit was obtained. Calibration parameters can be subdivided into traditional (e.g. Manning's n, depression storage and infiltration parameters) and non-traditional parameters (e.g. impermeable area, width and slope) according to Liong *et al.* (1991).

Values for traditional variables are usually used for calibration, while values obtained from non-traditional variables are considered to be fixed. However, even for non-traditional parameters, a certain error margin needs to be considered due to measurement errors. Therefore, some non-traditional parameters are sometimes included in the calibration process (Temprano *et al.*, 2006).

In this study, the non-traditional parameters were used for calibration, where the measurement error was deemed significant. For these cases, calibration allowed for the values to be determined more accurately, which resulted in model improvements. For example, the groundwater flow coefficient and the inlet offset of the detention tank outflow could not be determined with satisfactory accuracy: therefore they were included among the calibration parameters. Table 2 shows the parameters used for calibration of the model, the initial values chosen and the range of variation allowed. The SWMM allows for the modelling of 'evapotranspiration' losses for standing water on sub-catchment surfaces, subsurface water in groundwater aquifers and water held in storage units (Rossman, 2005). Therefore, actual evapotranspiration (free water-surface values) had to be calculated from the potential evapotranspiration. Evaporation from a free water-surface was estimated to be between 70 and 100% of the potential evapotranspiration. Hence, this range of values was used for modelling with SWMM in this study.

3. Results and discussion

3.1. Water balance

An overall water balance of the system was calculated, demonstrating that approximately 50% of the precipitation volume left the system as evaporation, while of the remaining 50%, approximately two thirds were infiltrated and one third was discharged directly to the sewer system via an emergency overflow system during heavy storm events. This indicates the significance of evaporation in source control. Good attenuation and infiltration was achieved even on urban man-made compacted soils with low permeability (Figure 3).

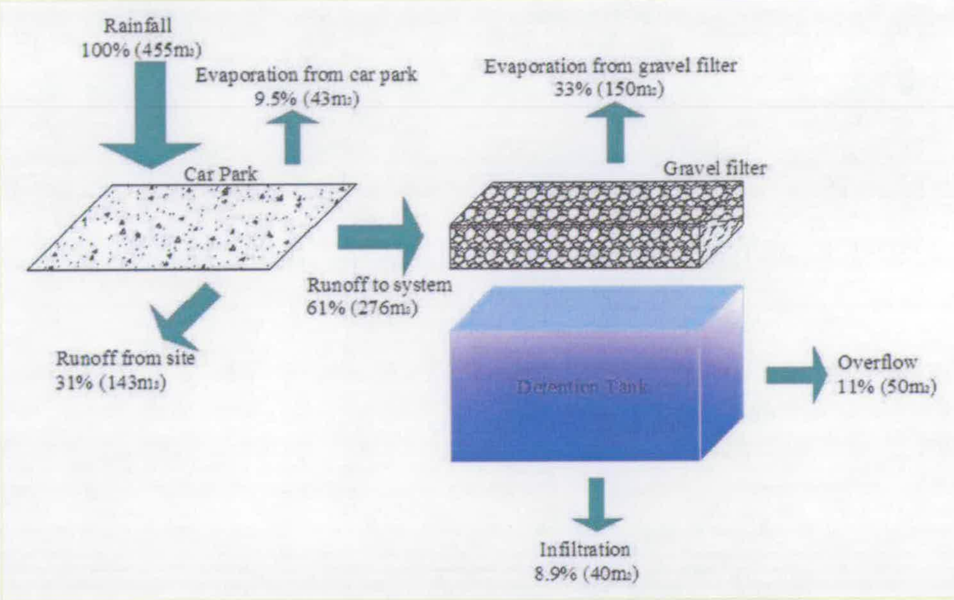


Fig 3.. Water balance of the catchment.

The evaporation rate was higher for the gravel filter than for the car park. This is due to different surface properties. Runoff from the car park takes only a short time, as the surface is impermeable. A free water surface is maintained within the gravel ditch for a relatively long period of time. This results in more evaporation from the gravel filter than from the car park and road.

The assessment of the system's hydrological efficiency gave mean lag times of 1.84 h for the gravel filter and 10.6 h for the entire system. A mean flow volume of approximately 70% and mean peak flow reductions of 90% were obtained. These values compare well with conventional drainage systems (Butler and Davies, 2006; Scholz, 2006).

Strongly skewed rainfall distributions were plotted . The majority of the rainfall events were weak, and storm events were rare. A longer measurement period would have been advantageous to test for the worst case rainfall scenarios.

Considering the physical properties of the soil, a curve fitting with the SWMM indicated a good approximation of the infiltration behavior. The maximum infiltration rate observed by curve fitting in SWMM was 1.3 mm/h which corresponds to a saturated groundwater flow velocity of 3.6×10^{-7} m/s, indicating loamy and clayey soil properties (Freeze and Cherry, 1979).

3.2. Modelling

The water within the storage unit can 'flow out' either via the outlet named Infiltration Function, which defines outflow by a head-discharge relationship, or via the conduit to the gully pot named Outflow_Pipe. This conduit is linked to the 'junction' Gully_Pot, which represents the gully pot located at the outlet of the storm water tank. Water is transferred from the gully pot to the sewer system via another conduit named Outflow_Gully. Eventually, storm water leaves the system via one of the 'outlets' that serve as downstream boundary conditions, named Ground-water for the infiltration pathway and Sewer_System for the tank overflow pathway.

Precipitation data were collected from the local rain gage Weather_Station_GeoSciences, which is associated with the subcatchments Car_Park and Gravel_Filter. There were no significant amounts of precipitation entering the tank directly due to the fact that it was constructed partly below the gravel filter and partly below a footpath covered with flagstones.

Evaporation data were provided in the form of monthly mean values for free water-surface evaporation. The values were calculated based on potential evapotranspiration values given by Müller (1996). The SWMM automatically associated the values with all sub-catchments. For the tank, evaporation was estimated to be a fraction of the evaporation rate. Approximately 50% of the precipitation left the system as evaporation. The flow volume was reduced by 32% as a result of evaporation from the car park. These findings compared well with values published by SNIFFER (2004) for conventionally drained surface areas in Scotland.

The tank was represented as a storage unit, as this approach gave considerably better results in comparison to a representation as a (sub-)catchment. The representation of the tank as (sub-)catchment resulted in an offset of the modelled water levels in the tank compared to the levels observed in reality, which could not be reduced sufficiently by calibration. This was due to the initial offset (SWMM assumes a zero water level in tank) and the resulting infiltration modelling problems (SWMM assumes unrealistic water levels leading to flawed infiltration and evaporation patterns).

After calibration of the SWMM, a satisfactory fit of the numerical model and with 'real' experimental data was achieved. It was therefore suggested that the model represented the actual flow regime in the SUDS sufficiently accurately.

3.3. Water treatment performance

An assessment of the pollutant removal efficiency indicated very good concentration reductions for suspended solids (SS, 83%), five days at 20°C N-Allylthiourea biochemical oxygen demand (BOD, 77%), and ortho-phosphate-phosphorus (PO_4-P, 47%). However, the results shown in Table 4 include relatively variable values gathered during the first three months of operation (i.e. immediately after construction) when the system was potentially overloaded by traces from construction materials including sand.

Variable	Statistics	Gravel ditch		Detention tank					
		No.1	No.2	No.3	No.4	No.5	No.6	No.7	No.8
BOD	Mean	36.2	17.5	5.8	6.9	7.8	8.6	10.5	8.2
	SD	18.5	10.0	12.4	13.4	9.5	12.1	11.3	11.4
	SN	41	37	60	59	60	60	60	60
SS	Mean	150.4	146.7	42.0	41.9	86.7	132.1	131.8	26.2
	SD	124.3	112.1	42.4	36.5	83.3	162.6	151.2	15.8
	SN	24	14	67	67	62	62	68	68
NH_4-N	Mean	0.2	0.16	0.21	0.23	0.26	0.25	0.2	0.3
	SD	0.24	0.13	0.20	0.27	0.25	0.21	0.24	0.30
	SN	26	17	62	62	62	62	62	61
NO_3-N	Mean	0.69	0.33	0.46	0.51	0.51	0.50	0.48	0.47
	SD	1.54	0.50	0.79	0.89	0.71	0.59	0.89	0.94
	SN	24	17	61	61	61	61	61	61
PO_4-P	Mean	0.92	0.37	0.49	0.44	0.41	0.49	0.42	0.49
	SD	0.80	0.32	0.63	0.47	0.48	0.66	0.37	0.59
	SN	26	17	60	60	60	61	61	61
DO	Mean	3.5	5.2	6.0	5.1	4.9	4.7	4.8	5.0

	SD	2.33	2.44	2.83	2.91	2.78	2.72	2.74	2.75
	SN	29	20	70	70	65	63	70	71
Redox	Mean	127	112	109	103	91	93	96	102
	SD	105.4	105.2	104.0	107.4	118.6	118.4	102.7	100.5
	SN	27	19	60	61	56	56	62	62
pH	Mean	6.08	6.16	6.63	6.59	6.69	6.54	6.35	6.40
	SD	0.429	0.514	0.427	0.460	0.501	0.558	0.555	0.461
	SN	34	23	74	74	68	68	75	75
Cond	Mean	299	91	165	176	215	183	136	138
	SD	177.3	75.0	62.0	63.3	84.7	79.8	91.3	69.6
	SN	32	20	73	73	68	68	74	74
Temp	Mean	13.2	12.1	12.5	12.4	12.1	12.0	12.8	13.1
	SD	2.92	3.66	2.49	2.59	2.82	2.90	2.68	2.95
	SN	28	20	74	74	74	74	74	74

BOD, five days at 20°C N-allythiourea biochemical oxygen demand (mg/l); SS, suspended solids (mg/l); NH_4-N, ammonia-nitrogen (mg/l); NO_3-N, nitrate-nitrogen (mg/l); PO_4-P, ortho-phosphate-phosphorus (mg/l); DO, dissolved oxygen (mg/l); redox, redox potential (mV); pH (-); Cond, conductivity (μS); temp, temperature (°C); SD, standard deviation; SN, sample number.

Table 4. Water quality of the sustainable drainage system.

The mean nutrient concentrations within the inflow and outflow of the system are shown in Figure 4. However, reductions were relatively low for nitrate-nitrogen (32%) and frequently negative for ammonia-nitrogen. This could be due to the overload with organic material from neighbouring mature trees during autumn.

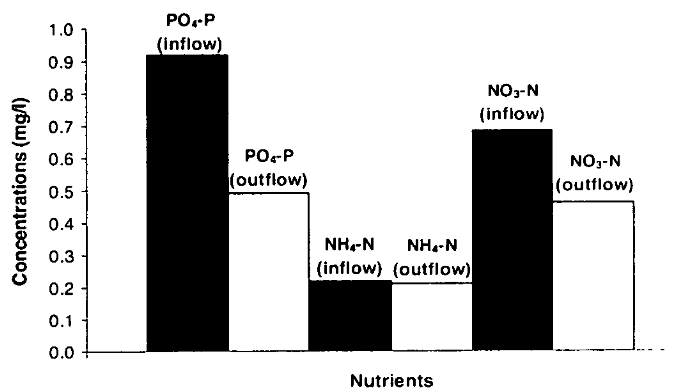


Fig. 4 Comparison of the mean nutrient concentrations between the inflow and outflow of the combined detention and infiltration system.

The results from an analysis of variance, which was conducted to assess the treatment potential of the system, show a significant difference between the inflow and outflow water quality of the bio-filter for most variables including conductivity ($p=0.002$), dissolved oxygen ($p=0.015$), BOD ($p=0.004$), pH ($p= 0.006$) and PO_4-P ($p=0.003$).

Negative removal efficiencies were sometimes calculated for BOD, SS and PO_4-P. The highest increase in load was observed for PO_4-P (times 2.5). During initial sampling, the outflow from the gravel filter had frequently a 'reddish' colour, which indicates the washout of fine particles associated with sand, leading to increased SS concentrations (Table 4). The increase of BOD and PO_4-P was due to the washout of organic matter, partly originating from decomposing plant matter (planted willows).

leaves (nearby trees), Ecosoil® and woodchips. However, the washouts observed during the initial start-up phase are not representative for long-term operations.

Concerning the filter trench, positive removal rates were observed for NH_4-N and NO_3-N (24% and 38% respectively). There are a number of possible causes for the removal including adsorption, uptake by the active microbial biomass and uptake by the willows. Concerning the detention tank, however, negative removal rates were observed for NH_4-N and NO_3-N, while the load of all other parameters decreased. The load reduction was very high for SS, BOD and PO_4-P with removal efficiencies of 92, 93 and 76%, respectively. Sedimentation and biological degradation are to be considered as the possible processes resulting in the reduction of the load of pollutants in the detention tank (Scholz, 2006). Furthermore, some micro-organism growth occurred in the tank (similar to the deeper zones of a natural pond), resulting in the breakdown of substrate such as BOD and PO_4-P.

4. Conclusions

The observed removal mechanisms were assessed by an in-depth analysis of single storm events. The initial washout of fines, biological degradation within the bio-filter, sedimentation and infiltration were the most important processes within the system. The system showed good hydraulic and water quality treatment performances. Less than 20% of the runoff reached the conventional drainage system (i.e. sewer).

The combination of biomass, aggregates and detention tank resulted in considerable enhancement of the water quality in the study site. Generally, the system was particularly successful in removing biochemical oxygen demand and suspended solids. The oxygen demand and particle removal were similar to semi-natural wetland systems. However, further studies are recommended to assess the effect of different choices of filter media on the overall performance of such systems and to assess how to improve the nitrogen removal performance.

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