An Investigation into the Quantification and Mitigation of Urban Diffuse Pollution

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List of Acronyms and Abbreviations

AMP	Asset Management Plan
BMPs	Best Management Practices
BOD	Biochemical Oxygen Demand
BWD	Bathing Water Directive
CIRIA	Construction Industry Research and Information Association
COD	Chemical Oxygen Demand
CSO	Combined Sewer Overflow (Outfall)
DCLG	Department for Communities and Local Government
DD	Downstream Defender
DFRA	Department of Environment, Food and Rural Affairs
DETR	Department of Environment, Transport and the Regions
DWD	Drinking Water Directive
A	Environmental Agency (England and Wales)
EEA	European Environment Agency
E.*coli*	Escherichia coli
EC	European Commission
EQS	Environmental Quality Standards
EQSD	The Environmental Quality Standards Directive
EU	European Union
FIO	Faecal Indicator Organism (faecal bacteria indicator)
FWMA	Flood and Water Management Act
HDVS	Hydro Dynamic Vortex Separator
GR	Grid Reference
IE	Intestinal Enterococci
IPPC	Integrated Pollution Prevention and Control
KMO	Kaiser-Meyer-Olkin measure of sampling adequacy
MAC	Maximum Allowable Concentrations
MS	European Union Member State
Ofwat	The Water Services Regulation Authority
ONS	Office of National Statistics
PAHs	Polyaromatic Hydrocarbons
PCA	Principle Component Analysis
PS	Priority Substances
PSD	Particle Size Distribution
PTS	Proprietary Treatment Systems (and Products)
RBWD	Revised Bathing Water Directive
RSPB	Royal Society for the Protection of Birds
SAGIS	Source Apportionment Geographical Information System
SEPA	Scottish Environmental Protection Agency
X4	Storm X4
SuDS	Sustainable Urban Drainage (and Systems)
SWD	Surface Water Drains
TOC	Total Organic Carbon
TSS	Total Suspended Solids
UWWTD	Urban Waste Water Treatment Directive
WFD	Water Framework Directive
WQV	Water Quality Variables
WWTWs	Waste Water Treatment Works

Abstract

The overall aim of this work is to extend the knowledge of Urban Diffuse Pollution (UDP) and to assess the effectiveness of available remediation solutions in a confined urban context, where disruption from retrofitting proprietary treatment systems and products (PTS) to existing infrastructure causes major difficulties. The research presented addresses the quantification and remediation of UDP across a study area of the River Douglas in Wigan in the North of England.

This study has involved an extensive programme of water quality sampling which has allowed micro-level changes in pollutant concentration to be observed as a result of weather event based urban runoff and enabling diffuse pollutants to be quantified. Following this the study goes on to suggest solutions to remediate sources of UDP exploring the use of vortex separation as a tool to treat polluting surface water drains. It provides performance data on several available PTS some of which were installed in outfalls into the study area in the River Douglas. These systems were monitored and further water quality data has allowed the quantification of the effectiveness in pollutant removal of different PTS.

The study identifies the significant contribution of urban areas to diffuse pollution of river water and shows that similar studies need to become widespread if the problem of UDP is to be effectively addressed. Based on the significant contribution to knowledge in terms of the new water quality data generated both in relation to river water quality and treatment products and systems, a series of practical recommendations are proposed in relation to the identification and remediation of urban diffuse pollution.

Chapter 1 - Introduction

1 Chapter 1 – Introduction

The North West of England has a significant industrial heritage, and the region led the world during the industrial revolution (Burton, 2003). However, the effluents generated by industrial growth left a legacy of some of the most highly polluted rivers in Europe (Wood and Handley, 1999). Major improvements were made with the introduction of basic waste water treatment (Rosenthal, 2014), but it was not until the introduction of the Rivers (Prevention of Pollution) Act of 1961, followed by the Control of Pollution Act in 1974, that considerably more progress was made to improve the water quality of the region's rivers.

As a consequence of this legislation, by the mid 1990's, pollution from point discharges, such as factory effluents and waste water treatment works (WWTWs), were under much tighter control. As a result of these improvements the Environment Agency's (EA) 2007 General Water Quality Assessment showed, that over two-thirds of all rivers in the UK were of good or very good biological and chemical quality (Environment Agency, 2007). However, several papers (Davies and Neal, 2007; Rothwell *et al.*, 2010a) demonstrate that, while improvements have been made in the water quality of the rivers in the North West and subsequent downstream waters over the last 25 years, mobilisation of pollutants along the rivers continues to occur from point and diffuse sources. The removal of major point source pollution has allowed identification of intermittent background pollution and the concept of 'diffuse pollution'[1] has become more prominent. The EA also suggest that 87 per cent of UK rivers, and half of all UK lakes are at risk from diffuse pollution; and it has become the limiting factor preventing further water quality improvements (Environment Agency, 2007).

Diffuse Pollution in urban areas has also come under increasing scrutiny, with the EA reporting in 2005 that one in seven urban rivers were of poor or bad chemical and biological quality. Primarily the pollution from urban areas can be attributed to two main sources:

1. Contamination from Combined Sewer Overflows (CSOs), leaky sewers, cross connections and WWTWs discharges.

2. Surface wash off from highways, industrial, residential and other land uses associated with urban areas.

[1] Diffuse Pollution is defined as "Pollution from widespread activities with no one discrete source, e.g. acid rain, pesticides, urban run-off, etc" (European Environment Agency, 2012).

There a wide range of pollutants observed from these sources, but typically they include heavy metals, suspended solids, nutrients, hydrocarbons as well as faecal contamination, and a range of other contaminants specific to the land use of the drained area.

The introduction of three new pieces of legislation in the UK in the last decade has presented a clear and pressing case to address urban diffuse pollution (UDP) within UK towns and cities, this legislation is:-

- the Water Framework Directive (WFD)
- the Flood and Water Management Act (FWMA)
- the Revised Bathing Water Directive (RBWD)

To explore this further, a proposal to identify and treat UDP was developed. The EA were approached and they concurred that there was a clear need to undertake such a study. With input from the EA, and through analysis of historical EA sample data the River Douglas which flows through the town of Wigan, was identified as a suitable study area.

Wigan was identified as a suitable study area for several reasons. Firstly, the upper course of the river was failing to meet several WFD water quality criteria, and secondly, the river's catchment has a wide mixture of land use including a substantial urban area as well as agricultural pasture and woodland. The Douglas is a tributary of the river Ribble, which discharges into the North Sea on the Fylde coast. Kay et.al, (2005) identified that half of the sewerage inputs to the Ribble basin are associated with the relatively small Douglas sub-catchment. The mixture of pollution inputs in Wigan are representative of the typical problems faced in many post-industrial towns and cities across the UK. Therefore it was felt that data and findings produced from the study would be applicable and relevant to address UDP in other similar waterbodies.

1.1 Knowledge Gap

With the increasing importance of addressing diffuse pollution evident, it is crucial that research is undertaken to investigate both its causes and solutions to mitigate its effects. Before the introduction of the WFD there was little or no legislation which sought to address the problems posed by UDP, therefore a fresh approach by regulatory authorities to river water pollution and treatment is required. There are several hurdles to overcome when investigating UDP and how to quantify and treat it. UDP is difficult to assess due to the episodic nature of polluting events, and the diffuse nature of sources are difficult to identify specifically. Existing estimates of UDP are very likely to be inaccurate when considering that the monitoring systems for urban water courses currently in place fail to take account of this episodic nature of pollution (J. Bryan Ellis and Mitchell, 2006).

Through an extensive literature review into the subject area surrounding the problem of diffuse pollution (Chapter 2), it is clear that it is a multidisciplinary problem requiring clear understanding of several different knowledge areas which include river chemistry and ecology, water sampling methodology, the physical mechanics of surface water wash off and sediment transport, as well as that of legislation and policy in relation to water quality and flooding. Therefore the review has collated and analysed a wide series of papers from multiple fields to produce a thorough and holistic view of the subject area. The review has identified a series of gaps in existing knowledge and highlighted a series of key problems with current approaches to the subject. These include:-

- Difficulties around identifying the contribution of diffuse sources;
- Unsuitable existing sampling regimes to identify diffuse pollution;
- A lack of data on the water quality performance of SuDS and PTS;
- A lack of consensus on the significance of the first flush effect;
- Significant barriers to the retrofit of SuDS and PTS in urban environments.

These problems are covered in detail in the literature review and summarised in section 2.6 and they indicate that there is a clear and pressing need to identify the sources and mitigate the impacts of UDP in UK Rivers; however given the complex nature of the overall problem this presents significant challenges. They also highlight a shortage of data available on the performance and effectiveness of different treatment solutions in the field.

1.2 Project Objectives

When considering these issues (section 1.1), it is clear there is a need to increase knowledge and understanding of UDP and its treatment. This PhD project aims to fill this gap in knowledge by setting and investigating the following objectives:-

1 To identify a suitable riverine study area and develop a micro level sampling regime for it, informed through consultation with the EA and site investigation.

2 To use collected sample data to observe pollutants in the river channel, identify those subject to the greatest fluctuation during storm events. This sample data will be used to indicate those which are most significant in terms of diffuse sources, as well as identifying any other trends that are apparent.

3 Using collected sample data, and other available information, to identify a series of locations for the installation of suitable SuDS or PTS in order to provide mitigation of the pollutants observed in the river monitoring.

4 Undertake monitoring of installed mitigation measures to assess their performance in respect to water quality improvement. The collected data will then be used to estimate the volume of pollutants discharged from each of the monitored sources.

1.3 Original Research

To provide answers to these questions it was necessary to undertake an original study. Although, as it will be shown in the literature review, previous studies have addressed some aspects of the overall problem of UDP, this is the first study to attempt an holistic approach addressing all the variables in real time trials of Hydrodynamic Vortex Seperators (HDVSs) in the field. This is certainly the case from a UK perspective. Specifically the novel aspects of the work completed in this thesis can be summarised as follows:-

- Use of a micro level sampling regime (multiple sample points across a small catchment) to monitor water pollution fluctuation accurately over short distances.
- Clear quantification of the contribution of diffuse pollution during a series of recorded rainfall events
- Completing real time monitoring of two off the shelf HDVS retrofitted to an existing drainage infrastructure using automatically triggered auto sampling units.

Through this new approach it was anticipated that insights would be gained into the suitability of existing sample regimes and water quality data to accurately quantify the contribution of diffuse pollution and to identify where improvements are needed in the existing approach.

1.4 Methodology Summary

To address these objectives the work completed for this PhD has been undertaken in several stages, as set out below:-

- Project Background and Development
- River Water Quality Sampling and Assessment
  - Develop and Complete a Sample Regime
  - Analysis of River Sample Data
- Treatment System Monitoring and Sampling
  - Selection of Treatment Systems and Locations
  - Monitoring of Treatment Systems
  - Analysis of Monitoring Data

Preliminary analytical work consisted of developing the research question as well as the project objectives and a literature review (Chapter 2). This work focused on identifying gaps in the literature, as well as developing a viable project that would yield sufficient new primary data. During this stage the EA were successfully approached to collaborate on the project.

The river water quality sampling commenced with selection of the River Douglas as a study area and the development of a holistic and extensive sample regime to be applied at the micro level. The purpose of the sampling programme was to identify the key pollutants in the river and to highlight those which contribute most significantly as diffuse pollutants. Samples were collected at 25 locations over a 4 month period between July and October 2012. Water samples were tested at EA laboratories for a wide spectrum of pollutants comprising 37 different variables, including nutrients, faecal bacteria indicators (FIOs), polyaromatic hydrocarbons (PAHs) and heavy metals. Supplementary rainfall and discharge data was used to further support analysis.

Following the characterisation of river pollutants, the study went on to investigate the different methods available to remediate them. Off-the-shelf proprietary water treatment (PTS) systems and products were utilised over SuDS due to their space saving properties, in the context of retrofitting treatment capacity to the confined urban study area. Informed by the river sampling results and literature on different PTS, seven different products were selected and paired with existing drainage assets (such as surface water drains (SWD) and road drainage assets) where PTS could be installed to provide treatment of storm runoff and discharges. Drainage Assets were selected through a mixture of analysis of river sample data, the use of the local water company asset register and other practical considerations.

Construction drawings were created for the construction design management process and the EA appointed a contractor to install the PTS. Budgetary limitations meant that only 4 of the 7 originally planned sites and their associated PTS were progressed to completion. With construction work completed the PTS were monitored to assess their performance. Automated sampling equipment was utilised to undertake storm monitoring at some sites whereas others were monitored by observing the accumulated weight of captured pollutants on an annual basis. The exact method of monitoring differed between products and more specific details for each are given in the methodology Chapter. The results obtained from monitoring enabled the removal efficiency of each product system to be calculated.

Parts of the work that were undertaken for this PhD were completed in partnership with the EA. Principally the EA provided a project management role, funding for all stages of the project as well as access to computer systems and laboratory resources. All data collection and analysis supporting this thesis was completed primarily by the student with some technical instrumentation and sample methodology support being supplied by the EA. The student also provided significant input to the construction design management process, including background research into which PTS were suitable to utilise, selection of sites to install them, development of all AutoCAD designs (under supervision), contractor appointment and liaising on site with the contractor on site during in product installation.

1.5 PhD Structure

This document provides a review of the literature in the subject area, it explains the methodology used, gives details of results collected and their analysis and makes a series of proposals for both river water management and further research work. The work completed is divided into seven chapters, as follows:-

Chapter 1 provides an overview of the project and explains the knowledge gap that exists around the effect of the urban land cover on river water quality. It also states the project objectives, summarises the methodology and gives an overview of the report structure.

Chapter 2 goes on to make a critical review of relevant literature concerned with diffuse pollution and its relevance to this study. It is divided into four main sub-sections:-

- Legislative background
- Catchment Management and Land Use
- UDP
- Urban Storm water treatment

Chapter 3 describes the methodology undertaken. It gives full details about the type of sampling; testing and analysis that was undertaken to identify pollution sources in the River Douglas. A detailed explanation of the range of mitigation methods identified to treat pollutants follows, including the criteria for selecting sites, drainage assets and the individual products that were used. The necessary design work completed which allowed the contractor to install products is also described as well as the methods used to monitor each installation.

Chapter 4 provides a full analysis and discussion of the results. This is divided into two parts to reflect the two phases of the project work. The first explains the results generated from the river sampling and the second contains the results from PTS monitoring. It also gives full details of the analysis completed to assess the UDP in the river Douglas. Secondly it presents the results from monitoring completed to assess the effectiveness of the different PTS products selected for application in the project.

Finally, Chapter 5 summarises the findings of this project and draws conclusions from the analysis of the data collected. It gives recommendations for the improvement and control of diffuse pollution in UK Rivers and makes suggestions for further research work.

Chapter 2 – Literature Review

2 Chapter 2 – Literature Review

In the introduction gaps in current knowledge and understanding around diffuse pollution were identified. In particular it was observed that the current regulatory framework which seeks to quantify and mitigate diffuse pollution in an urban context certainly fails to take account of the episodic nature of pollution from diffuse sources. This indicates the need for further research to better understand the contribution of diffuse pollution to rivers and to investigate methods and systems that are available to provide mitigation of this pollution. The work undertaken on this project therefore spans several different disciplines and subject areas that relate to this problem which constitutes a large body of relevant literature. To deal with this effectively the Chapter is divided into four main topics, which are:-

1. Legislative background to Urban Diffuse Pollution;
2. Catchment Management and Land Use;
3. Urban Diffuse Pollution;
4. Urban Storm Water Management.

2.1 Legislative Background to Urban Diffuse Pollution

Legislation in the UK has helped increase awareness and action over the problem of surface water runoff polluting rivers and lakes. After defining UDP this section explains the laws and mechanisms put in place by three key pieces of legislation: the Water Framework Directive, The Revised Bathing Water Directive and The Flood and Water Management Act. The issue of UDP is complex and understanding the nature of the problem is key to addressing it. In the absence of a statutory definition in England and Wales B. J. D'Arcy *et al.* (2000) defines it as: “Pollution arising from land-use activities (urban and rural) that are dispersed across a catchment, or sub-catchment, and do not arise as a process effluent, municipal sewerage effluent, or an effluent discharge from farm buildings”

D’Arcy’s report on the Environmental and Economic Impacts of Diffuse Pollution in the UK, also provides a useful insight for regulators observing that diffuse pollution sources are “scattered, discrete or dispersed inputs of contaminants which are collectively significant, but which regulatory agencies either could not or would not wish to try and control with discharge consents” (B. J. D'Arcy *et al.*, 2000). The Scottish Environmental Protection Agency (SEPA) has defined diffuse pollution as:

"….the release of potential pollutants from a range of activities that individually may have no effect on the water environment, but at the scale of a catchment can have a significant impact (i.e. reduction in water quality, decrease in wildlife, etc.)…." (SEPA, 2012a).

Applying these concepts in the urban environment, diffuse pollution may be more accurately defined as:-

"the contamination of rainfall and subsequently runoff, by interaction with deposited materials on urban surfaces and areas, which individually may not be significant but collectively lead to contamination and subsequent degradation the aquatic environment of receiving waters".

This is the working definition used in this project.

As urban diffuse pollution is a complex and multidisciplinary problem there is currently no single piece of legislation which addresses it. Holistic environmental management is increasingly a key objective of government policy, with decision making taking account of ecological, social and economic values (Jakeman and Letcher, 2003). However although integrated sustainable catchment management of which addressing urban diffuse pollution is a crucial part, it is still not being achieved in the UK (Macleod *et al.*, 2007). Schemes to address flood risk and water quality are completed independently of each other with one rarely taking much more than superficial consideration of the other. Nevertheless, there is still significant existing legislation which obligates UK authorities to take account of diffuse pollution both in urban and rural situations. The following paragraphs provide a brief historical review of development of UK legislation affecting water quality before focusing on the three of the most important current pieces of legislation, which are: The Water Framework Directive (WFD, 2000/60/EC), The Revised Bathing Water Directive (RBWD 2006/7/EC) and The Flood and Water Management Act (FWMA, 2010/DEFRA).

Historically the main driver behind improving water quality and preventing pollution was the protection of human health from infectious diseases such as cholera. Even before John Snow demonstrated in 1854 that the prevailing theory that disease and fever were spread miasmically was incorrect and established germ theory, legislation to protect public health was being introduced (Rosenthal, 2014). Had reforms such as the Public Health Act (1848) been based on germ rather than the miasmic theory much better results could have been achieved. In reality the 1848 Act resulted in degradation in river quality as it saw the introduction of a large scale water carriage system and piped sewerage discharges. As there was no reliable treatment available and no demand from agriculture to use the resulting liquefied effluent the only other option was for disposal to river courses (Rosenthal, 2014).

The construction of reservoirs throughout the 19th Century provided major improvements to water supplies; however it was only in the latter part of the 20th century that it can be considered environmental concern over river water quality became a mainstream consideration. This is reflected in the legislation introduced during this period; particularly the Rivers (Prevention of Pollution) Act of 1951which made it an offense to cause pollution to a river, with new discharges having to achieve certain standards. However existing discharges were exempt and it wasn't until the act was updated in 1961 that this was rectified. In 1963 the Water Resources Act was passed creating 'River Authorities' who became responsible for enforcing law in relation to river pollution, water resources, land drainage, fisheries and water space recreation. The Act also regulated the abstraction and impounding of water resources on a regional basis. This was the beginning of river basin management (Porter, 1978).

The scope of legislation was increased and consolidated throughout this period eventually spread over 20 separate acts, and much of the substantive law remained unchanged until the 1990's legislation regarding water quality and pollution. The 1991 Water Resources Act consolidated water resources legislation but water quality improvement objectives were also a key feature with several sections directly concerned with this. Section 82.4 required the introduction by the secretary of state of a classification system for the water bodies and for the EA to enforce the water quality standards set in the act and ensure that controlled waters have their quality objectives maintained. Section 85 of the 1991 Water Resources Act placed a statutory requirement for each discharge of sewerage or trade effluent made directly into surface water to have a discharge consent obtained from the EA. It also made polluting controlled water an offence (Department of Environment Transport and the Regions, 1991).

Several other sections of the act outline preventative measures and precautions against pollution entering controlled rivers, including the creation of water protection zones free from pesticides and other potential pollutants as well as the creation of Nitrate sensitive areas with the goal of reducing the volume of Nitrate reaching groundwater sources. Codes of good practice for agriculture promoted river water quality improvement through encouragement of good farming practices. Another key target of this act was to protect areas where Nitrate levels were likely to breach the 50mg/l limit set by the earlier EC Drinking Water Directive (80/778/EEC) and this target highlights the importance of the link between river water quality and drinking water quality (Department of Environment Transport and the Regions, 1991). In many parts of the UK river water abstraction for drinking water supply is an essential source for water companies, so protection of supplies against contamination from diffuse sources is important. The 1991 Water Resources Act was introduced shortly after the 1989 Water Act which saw the privatisation of the water supply and treatment section of the ten publicly owned regional water authorities, with regulatory responsibility being transferred to the National Rivers Authority which was soon to be renamed the Environment Agency by the 1995 Environment Act.

Up to this point legislation was focused towards control of point discharges, and although the importance of diffuse inputs from agriculture was beginning to be recognised, the contribution of diffuse inputs from urban sources was still not really considered. In addition the objectives outlined in the 1991 Water Resources Act were 'use-led' (Helmer and Hespanhol, 1997) and, whilst the act identifies the importance of diffuse agricultural pollutants such as nutrients and pesticides, the appreciation of the importance of management at catchment level, was still absent. A focus on just one portion of the watershed has limited effectiveness and the weakness of this approach has been documented (Born and Sonzogni, 1995). It is evident in many cases, that management of water resources has been focused on physical control of water and the associated economic implications, leading to ecological issues becoming subservient (Jakeman and Letcher, 2003).

The introduction of the WFD in 2000 (2000/60/EC) characterised the next stage of environmental policy evolution, emphasising that protection of public health and protection of the environment are synergistic, requiring a mixture of approaches beyond simple ''end-of-pipe'' solutions to encompass preventative and integrated management. The concept that restoration of the natural conditions of a river catchment can have a beneficial impact upon multiple management objectives has steadily gained momentum and become more accepted since the introduction of the WFD. This has seen the introduction of national legislation to implement the WFD, designed to more sensitively manage catchment areas through control of diffuse urban and farming pollution inputs.

2.1.1 Water Framework Directive

The introduction in June 2000 of the WFD was the culmination of three and half years of institutional negotiation and debate. Setting a common approach and goals for the management of water in 27 countries (15 member states (MS) and 12 pre-accession countries, which have since become MS) (WFD; 2000/60/EC); its introduction indicates an understanding that previous less integrated and disconnected management was not delivering required environmental improvements (Kallis and Butler, 2001). It gives clear evidence that managing rivers at a catchment level has moved into mainstream thinking and that when catchment management activity is conducted correctly it can achieve multiple benefits (European Commission, 2000). Provisions set in earlier directives are integrated within the WFD, allowing them to be gradually repealed (Macleod *et al.*, 2007).

The WFD also introduces new standards, criteria, institutions and processes which shift management of Europe's rivers towards an integrated ecosystem based approach (Kallis and Butler, 2001). While the WFD does not define or make specific reference to UDP, Article II specifies that identification and quantification of diffuse sources needs to be undertaken, with Articles IV and VII requiring the development of a program of measures to be laid out in a River Basin Management Plan (European Commission, 2000).

2.1.1.1 Responsibilities of Member States under the WFD

The WFD introduces a number of important changes, especially with respect to institutions and planning processes. Activities need to be co-ordinated at a geographical/administrative level of the "river basin district". Basins must be designated along with competent authorities who are responsible for them. A River Basin Management Plan must be produced by authorities every 6 years and cover:-

1. A description and maps of the catchment;
2. Identification and mapping of protected areas;
3. Identification and mapping monitoring networks;
4. Identification and assessment of significant pressure on the aquatic environment (including, estimation of point and diffuse pollution, summary of land use, estimation of abstractions);
5. Economic analysis of the cost of water;
6. A summary of the measures taken to achieve goals and comply with exiting legislation and the direction.

The directive requires monitoring of progress, with a progress report at the end of each 6 year period detailing the implementation of improvement measures and the recording of achievements and goals along with detailed mapped data from monitoring. Plans are required to be reviewed and revised as necessary for the following 6 year period (European Commission, 2000).

2.1.1.2 Implementation of the WFD in Member States

As well as the appointment of competent authorities and the development of River Basin Management Plan, the primary duty of member states (MS) is to comply with the environmental objectives laid out in articles IV, V and VI of the directive. The WFD classifies water bodies as surface waters, ground waters and protected areas. The obligations of MS for each of these defined water bodies varies slightly, but largely this means they must ensure that there is no further deterioration of water quality and implement a number of measures intended to reduce and phase out pollutants. In addition states must establish a register of all protected areas.

Each River Basin Management Plan has a series of basic measures required to conform to the directive; firstly this means implementation of the other relevant legislation for water protection such as the Integrated Pollution Prevention and Control Directive (IPPC, 2008/1/EC), Urban Waste Water Treatment Directive (UWWTD, 91/271/EC) and the Drinking Water Directive (DWD, 98/83/EC). If fulfilment of the terms of these various acts fails to ensure receiving waters achieve ‘good’ status as defined by the directive then further supplementary measures may be required.

These supplementary measures include additional pollution control measures consisting of emission limit values and recipient quality standards with the more stringent being applied to any point source not covered by the IPPC directive. At the time of publication there are no enforceable regulations covering diffuse discharges or sources. However in the UK a range of best management practice advice and associated schemes have been developed to help mitigate the contribution from diffuse sources to pollution of receiving waters. One of these schemes is the Catchment Sensitive Farming Delivery Initiative; a national policy which aims to address diffuse water pollution from agriculture in rivers, groundwater and other water sources. It is being run by the EA in coordination with Natural England and with funding being provided by DEFRA, the scheme works in conjunction with the Capital Grants Scheme which provides funding for improvements under the Catchment Sensitive Farming Delivery Initiative (CSF Evidence Team, 2011; Natural England, 2011).

In Scotland the Sustainable Land Management Incentive Scheme is a programme run by Scottish Water which aims to mitigate the effects of diffuse pollution by offering financial incentives to land owners and farmers and to provide advice and technical support in respect of farm diffuse pollution management, installation of a biobeds (soil and straw lined pits to collect pesticide washings), stock fencing and livestock watering. The aim of this scheme is to improve river quality and thus reduce water treatment costs (Morris, 2013).

The Sustainable Catchment Management Programme is an example of one scheme in England run by United Utilities and the RSPB. Its key objectives are improvements to water quality in respect of colour and sediment load, mitigation of downstream flooding issues and enhancement of biodiversity. This has been achieved by a series of measures which included large scale blocking of moorland drains (grips), re-vegetation of bare and exposed peat and degraded blanket bog and introduction of more sustainable management of grazing (Anderson, 2010).

In urban areas there are no schemes offering financial incentives to reduce pollutant runoff. Also there are no examples of action taken at a wider scale with efforts only being directed at a local level and primarily being concerned with flood mitigation rather than water quality. This probably reflects the greater number of vested interests in urban areas and the associated difficulty in achieving consensus between land owners. However there are examples of note, such as the work completed by the Council in Lambeth in 2011-12 where green roofs, highway soakaways and filter trenches as well as rain gardens in communal green spaces have been constructed, although again this was aimed at reducing flooding rather than surface water quality (Stovin *et al.*, 2013).

2.1.1.3 The EQS Classification System

The EQS Directive (section 2.1.4.3) is utilised by the EA to classify UK surface and coastal waters under the WFD. There is no specific method of determining sample locations for routine EQS sampling, where catchments are divided into separate waterbodies and sample location selected to be the most representative for the whole of each waterbody. This is typically located at the lowest extent of the waterbody, but in some large waterbodies additional sample points are added in tributaries. However there is no specific requirement that governs the distance between sample locations where operational, surveillance and investigational monitoring is undertaken by the EA.

Operational and surveillance sample locations are fixed, and see the collection of all biological (collected triennially), hydro morphological and physio-chemical (collected annually) elements. Investigational monitoring is used to collect further samples in different locations when water quality concerns are observed, or to monitor the impact of pollution incidents. As stated in section 2.1.4.3, the requirements of the EQS and Priority Substances (PS) Directives are transposed into UK law by the ‘River Basin Districts Typology, Standards and Groundwater threshold values (Water Framework Directive) (England and Wales) Directions 2010’. Parts 3-6 of this document contain limits for the various substances listed within the directives. For each of the different parts, samples intervals are: monthly for part 3 and 5 substances and quarterly for part 4 and 6 substances. An automated scheduling system programs sampling runs, so samples can be taken on any day of the week within working hours, although sampling is spread evenly across the year to account for seasonal variation.

The EA operational instruction 034_08 “Routine environmental monitoring in rivers supporting information for chemistry” covers much of this detail (Anon, 2014). As mentioned above the vast majority of sample locations are fixed but the document states that for diffuse inputs, an appropriate sampling network will be determined by the EA national office, although no further detail on how this is to be done is provided. From the guidance offered in this document the sample point located within a defined ‘waterbody’ is considered to be representative of its water quality.

The actual process of classifying a waterbody under the WFD has several stages. The physio-chemical quality elements for surface water are covered by parts three to six of the ‘River Basin Districts Typology, Standards and Groundwater threshold values Directions 2010’. Using collected samples, defined water bodies can then be classified to a certain ‘status’, which are: high, good, moderate, poor or bad for quality elements under ‘part 3’ of the WFD. For most other elements listed under parts 4, 5 and 6 a pass/fail criteria against an annual average or 95th percentile value is defined. To classify quality elements in ‘part 3’ a further classification of a site is required, based on the altitude of a site and the average alkalinity (in $CaCO3$). For this sites are split into seven types, with sites of greater elevation and lower alkalinity having more stringent standards. For some quality elements listed under 'part 5' standards are also expressed as maximum allowable concentrations (MACs) that are permissible in single samples. This details the method of river site classification; there are further steps to complete for lake, coastal and transitional site classification.

2.1.2 The Revised Bathing Water Directive

The Revised Bathing Water Directive (RBWD 2006/7/EC) came into force on the 24th of March 2006, to supersede the previous directive (BWD 76/160/EC CEC, 1976). The RBWD is a daughter directive of the WFD, and Annex VI of the WFD includes areas covered by the RBWD which require competent authorities in MS to limit concentrations of indicator bacteria in bathing waters, through a combination of point and diffuse source control.

The RBWD has several goals, but the main ones of interest here are to:

- Deliver a more scientific based approach to health and environmental protection as well as environmental management.
- Deliver more accurate and timely information to citizens about the quality of bathing waters.
- Integrate bathing water protection with other EU measures which seek to protect and improve other water bodies and sources, through progression from simple sampling and monitoring schemes to a more integrated management programme for bathing waters.

Although introduced in 2006 the RBWD will not be fully enforced until 2015 due to the transitional period to allow member states time to implement new requirements. The RBWD significantly simplifies the classification of bathing waters cutting the 19 water quality parameters specified in the previous BWD, to an assessment of just two FIOs, i.e., Escherichia coli (E.coli) and Intestinal Enterococci (IE). These parameters allow the classification of the quality of bathing waters, through the monitoring of bacterial levels in collected water samples. Another purpose of the transitional period of the directive is to allow member states to build-up data sets for all bathing water sites, as assessments require a comparable data set which covers a consecutive 4 year period for both FIO parameters.

Water bodies controlled by the directive will be classified into 4 different groups using the required four year data set, these are:

- Excellent (approximately twice as stringent as the current Excellent Guideline standard);
- Good (similar to the current good guideline);
- Sufficient (approximately twice as stringent as the current mandatory standard)
- Poor, for waters which do not comply with the Directive's standards.

The classification of each bathing water site is then calculated taking the 95 and 90 percentile of each data set. All MS should ensure that all bathing waters attain a minimum quality threshold of "sufficient", at the latest by the end of the 2015 season. For sites classified as "poor", MS should take measures such as the provision of information to the public, banning of bathing or advising against it and the implementation of suitable corrective measures.

After the collection of the initial 4 year data set in the transitional period MS must continue to monitor bathing waters on an annual basis during the bathing water season, which varies between MS (in the UK the bathing water season is between May and September). A minimum of four water samples should be provided in a bathing water season (subject to short seasons or special geographic constraints) and intervals between samples should be no greater than one month apart.

Member states should also produce 'profiles' for bathing water sites which give a description of the site, details of the potential threats and impacts to water quality such as sources of pollution and the location of water quality monitoring points. Profiles are meant to provide information to citizens and to function as a management tool for responsible authorities. Profiles had to be produced by 2011.

The EC and the EEA publishes an annual summary report on bathing water quality, based on the reports that all MS submit at the start of each bathing season. The EU wide report is produced in both paper and electronic formats whereas reports from individual MS are only required to be available electronically. Member States are also required under the RBWD to ensure that during the bathing water season timely information is disseminated to the public in relation to bathing water quality. Specifically this should include easily identifiable notices and signs advising against, or banning bathing, as necessary. From June 2008 interactive maps on the water information system for Europe giving detailed information on bathing water quality in individual bathing areas became available at the Water Europe website: www.water.europa.eu (European Commission, 2006).

The vast majority of bathing water sites are coastal, so the RBWD is not primarily concerned with inland river water quality. However due to the often significant volumes of faecal contamination released via rivers into estuaries, it is important to address faecal contamination in rivers, in order to ensure there is no degrading effect in estuarine bathing waters. A good example of this is detailed by both Kay *et al.* (2005) and Wither *et al.* (2005) who identified significant bacterial contamination from the River Douglas into the Ribble estuary and although it accounted for only 8% of the total discharge volume, it contributed over 60% of the Faecal Coliform load during both base and high flows.

This contribution from catchments indicates a clear driver for improved reduction of the contribution from urban sources, such as, outdated waste water treatment works, leaky CSOs and polluted surface water outfalls. Rural and poorly managed agricultural sources also need to be tackled, such as, private package sewerage treatment works, incorrect live stocking in riparian zones, livestock sheds which drain directly to rivers, etc. This clearly indicates a need for greater integration between different policy mechanisms and the science base that supports them to help achieve effective sustainable catchment management (Macleod *et al.*, 2007).

2.1.3 Flood and Water Management Act

In 2010 the FWMA legislation was introduced primarily as a response to the 2007 review by Sir Michael Pitt into the widespread flooding in the UK of that year, which was largely caused by surface water runoff inundating poorly maintained drainage networks and systems (M. Pitt, 2007). The FWMA is concerned with improving the management of flood risk in the UK, providing improved protection for consumer’s water supplies as well as protection of surface water drainage discharges. It also gives the EA the lead responsibility for development of a national flood and coastal risk management strategy whilst also giving local authorities the duty as Lead Local Flood Authorities to co-ordinate local flood risk management (Coulthard and Frostick, 2010; Department for Environment Food and Rural Affairs, 2010).

While not primarily concerned with achieving improvements in urban water quality the act does include a requirement for sustainable drainage of surface water in developments which require planning permission, or which have drainage implications. This requirement is outlined in schedule 3 of the act, the introduction of which has remained subject to continual delays. This also removes the automatic right, which was established by the water industry act, to connect to public sewers and it was intended to move the responsibility for granting permission for connection to local authorities SuDS approval boards who were to assess proposed SuDS against a new national standard which has yet to be published. These standards will detail the design, construction, operation and on-going maintenance of SuDS and consider runoff destinations, peak flow rates, and anticipated runoff volumes from developments as well as water quality considerations (Flood and Water Management Act, 2010).

Some details are still not known but it is likely that developments and projects not requiring planning consent and covering less than $100m^2$ will be excluded from the requirements. The provisions were initially proposed to be phased in over 3 years and at first to cover only major developments. Once fully implemented this schedule of the FWMA has the potential to provide a significant river water quality benefit, though not only through the provision of treatment capacity of SuDS such as filter strips, swales and infiltration trenches etc., but also the removal of surface water loading of foul sewers reducing their spill frequency into rivers. However implementation of schedule 3 has been subject to on-going consultations by the Department for Communities and Local Government (DCLG). Originally set to be introduced in January 2014, it was announced in early January 2015 that it is to be implemented from 6th April 2015, when responsibility for its implementation is to be handed to local authority planning departments with the creation of SuDS Approval Boards scrapped (DCLG, 2014).

2.1.4 Other Legislation

The other overlapping regulations which are also important when considering diffuse pollution and river water quality are explained below.

2.1.4.1 The Urban Waste Water Treatment Directive

The Urban Waste Water Treatment Directive (91/271/EC) addresses the collection, treatment and discharge of urban waste water and the treatment and discharge of waste water from certain industrial sectors. It seeks to protect the aquatic environment from negative impacts cause by the discharge of urban waste water. Annex II of the directive requires MS to produce a regularly updated list of both sensitive and less sensitive areas which receive treated waters. The degree to which urban waste water is required to be treated varies depending upon the sensitivity of the receiving water.

The directive was required to be implemented by December, 2005 when MS should have ensured that all discharges of urban waste water into non sensitive waters (those with bacterial population equivalent concentrations between 2000 and 15000 colony-forming units (cfu)) and for sensitive receiving waters (concentrations between 2000 and 10000 cfu) must have a connection to a treatment system. Monitoring of WWTWs and discharges is the responsibility of each MS and competent national authorities are required to publish a situation report every 2 years (UWWTD, 91/271/EEC).

The UWWTD in conjunction with the RBWD are the drivers for the on-going improvement of CSOs and other urban outfalls contaminated with wastewater from cross connections and other sources of FIO contamination. The introduction of the UWWTD led to a major upgrade programme for the 6000 estimated CSOs identified in the UK. This improvement is achieved through the identification of Unsatisfactory Intermittent Discharges by the EA and the subsequent improvement or replacement under the asset management plan of the water company in ownership of the asset (Myerscough and Digman, 2008).

Although the directive was introduced over 15 years ago, the issue of sewerage waste water discharge, primarily from CSOs, during storm events remains an on-going issue. As recently as October 2012 the Court of Justice of the European Union determined that the UK had breached the UWWDT, as a result of storm water overflows. The court concluded that the UK had not demonstrated through the use of best practice and while not incurring excessive costs, that measures to prevent the illegal discharges (CSOs) were technically impossible (Stovin *et al.*, 2013). The directive was also a driver for the introduction of biological water quality assessments alongside chemical assessments, and assessments of macro invertebrates and diatoms became a requirement of a full water quality assessment (Kelly, 2002).

2.1.4.2 Integrated Pollution Prevention and Control Directive

Integrated Pollution Prevention and Control (IPPC) Directive (96/61/EEC) requires MS to have in place a system to issue operational permits to certain types of industrial installations, based on Best Available Techniques which are defined in the directive. The IPPC directive requirements are included in the WFD and smaller industrial installations not directly covered by the IPPC directive would be included under the WFD and in turn make Best Available Techniques applicable to those installations. In respect of urban diffuse pollution the IPPC directive is of relevance in relation to the control of pollution to land and air which then has the potential to reduce pollutant deposition on urban surfaces and soils and in turn reduce the overall pollutant load available to be washed into water courses (Boymanns, 2002).

2.1.4.3 The Environmental Quality Standards Directive

The Environmental Quality Standards Directive (EQSD) (2008/105/EC), also known as the Priority Substance Directive, was introduced in 2008 to amend the WFD, and repeal a series of other directives, primarily the Dangerous Substances Directives (67/548/EEC), which also had a series of attached directives covering concentrations of specific hazardous substances. The directive established limits on concentrations of 33 priority substances and 8 other pollutants in surface waters, and these limits have been subsequently included in Annex II and X of the WFD. It obligated MS to establish an inventory of emission, discharges and losses of these substances. This was transposed into domestic legislation through the introduction of the 'River Basin Districts Typology, Standards and Groundwater threshold values (Water Framework Directive) (England and Wales) Directions 2010'. This document provides the basis for the chemical classification of surface water bodies, with parts 3-6 covering the list of priority substances within the directives giving the limits set to achieve a certain 'status'.

2.1.4.4 Drinking Water Directive

The Drinking Water Directive (DWD) (98/83/EC) was first implemented in 1980 and was revised in 1998. It contains 48 parameters which drinking water is required to achieve to be fit for human consumption. These include limits for heavy metals, pesticides and Nitrates which all affect water quality adversely and their control has had major implications for agricultural policy. While its primary goal is protection of the consumer, the limits set for pollutants are important for the maintenance and protection of good quality sources from which drinking water supplies can be abstracted. In some cases the management of urban runoff can be an important part of protecting an urban source, so the limits set in the DWD are directly referenced in other legislation such as the Pesticides Marketing Directive in respect of the limit value set in the DWD.

This sub-section has exposed the legislative context that exists in the UK which provides the primary incentive and drive for improvements to fresh and coastal water environments. It identifies the importance of addressing issues at a catchment level. The following section gives further detail on this and explores the relationship between land use and water quality.

2.2 Catchment Management and Land Use

This section explores what good Catchment Management is and how it can not only benefit river water quality but also play a role in decentralised flood control. It also looks at the principle land use in the UK, how various land use practices can contribute to the degradation of water quality of runoff and how these effects can be mitigated through the use of best management practice (BMPs) and control measures. As discussed in the previous section policy and legislation affecting water quality is increasingly seeking to deal with issues at the scale of the whole catchment. It is therefore important to view management activity in a holistic way, and understand the relationship between management and land use change. The achievement of multiple benefits through modern BMPs is now a common theme. The key principles to consider when manipulating catchment characteristics to deliver benefits and better services to populations and the environment are explored in this section.

2.2.1 What is Catchment Management?

Catchment Management like UDP lacks a statutory definition and so the term is used by various organisations to imply slightly different things and, as with any emerging concept its definition varies depending on the perspective of the writer. Other terms used including catchment management planning and sustainable catchment management, although they largely refer to the same concept. SEPA for example refers to Catchment Management Planning as:-

"…… the process of bringing together stakeholders to develop actions that conserve and enhance the ecological quality of the river and its environments. Catchment management planning embraces the principles of ecosystem services. It recognises that rivers are integral to land use management and support a range of diverse activities and service...." (SEPA, 2012b).

However for the purposes of this project a more comprehensive definition is used:-

"The management of catchment areas in such a way as to provide a range of environmental, social and economic benefits, through an integrated and sustainable approach delivered by all the affected stakeholders working in collaboration".

Moving to a management system that conforms to this definition presents significant challenges, especially when the pressures of increased population growth, dwindling resources, demand for increased agricultural output, growing levels of urbanisation and climate change are considered. There is a need to deliver more for less and to tackle issues in a holistic way that demonstrate impacts at the whole catchment scale (Wilkinson *et al.*, 2014).This is further complicated by the wide range of different organisations and stakeholder groups who have interests in catchment management, including water companies, local authorities, government bodies (e.g. EA, DEFRA, Natural England, etc.), local communities, wild life and river trusts, etc. Considering the broad and diverse nature of stakeholders it is important to ensure high levels of integration in order to ensure delivery of multiple benefits (Mostert, 1999).

Provision of 'good quality' water is an important function of catchment management (Macleod *et al.*, 2007), but it is also important to appreciate the effects of the various types of land use and that the management of land has a clear bearing upon potential improvements or risk of degradation of expected environmental outputs, such as increased biodiversity or lower flood risk. Methodologies for mitigating the degradation of water quality and flood risk share many objectives and consideration of this would allow delivery of joint benefits, for example improved source control of rainwater not only has a positive effect on flood control through water retention, but it also prevents runoff conveying pollutants into receiving waters, which it would otherwise do. This key concept is a feature of the 'blueprint to safeguard Europe's water resources', however its application in practice is a rarity (European Commission, 2012).

This lack of integration could be the cause for the limited progress of urban and agricultural pollution policy measures in improving river water and ecological quality across Europe, Australia and the US (McGonigle *et al.*, 2012; OECD, 2012). For example in England and Wales there has been limited success in further improvements to river water quality; the percentage of total river length exceeding 0.1mg/l of Phosphate dropped by 10% between 1990 and 1995, whereas to achieve a further 10% reduction it took until 2009 by which time 50% of total river length was still failing to achieve this standard. In respect of Nitrate levels, these have fluctuated since 1990, however the percentage of total river length failing to achieve less than 30mg/l concentration fell by just 7% in the ten years between 1990 and 2009, with almost 30% of total river length still failing this standard (DEFRA, 2009). Figures such as these suggest that while improvements are being made using current methods progress is slow. Holistic catchment management is fundamental to the ultimate success of delivering water quality improvements and appreciating the direct link between land use management and its relationship with the quality and quantity of runoff.

2.2.2 Land Use and Diffuse Pollution

Land use and diffuse pollution are inherently linked. There is considerable evidence of a causal link between various types of land use and degradation of receiving waters. To understand this problem fully it is important to appreciate the difference between land use and land cover. The Food and Agricultural Organisation of the United Nations defines land use as "the arrangements, activities and inputs people undertake in a certain land cover type to produce, change or maintain it" as opposed to land cover which is defined as "the observed (bio)physical cover on the earth's surface" (Gregorio and Jansen, 2005). For example much of the UK's moorland would be classified as grassland in respect of land cover; however it would be classified as agricultural in terms of land use.

Three different land use activities comprise over 98% of the UK's land use: agriculture (75%), forestry (13%); and, urban areas (10%) (Khan *et al.*, 2012). The first two classes are therefore very important in understanding the overall diffuse pollution load entering the UK's rivers. There are a series of parallels that can be drawn with respect to how the agricultural and forestry sectors have approached the issue of how to tackle diffuse pollution. For example the linking of financial incentives to changes in land management practices in order to encourage landowners to reduce diffuse pollution from their activities. Also important is the need for coordinated action between all affected stakeholders across the catchment. There is much common ground in best management practice to learn from these two sectors in relation to the situation experienced in urban areas. However as the primary focus of this study is urban diffuse pollution, further exploration of the literature on diffuse pollution from agriculture and forestry activity is detailed in Appendix I. The following section explores the contribution of urban land use to river pollution where attention is drawn to the key lessons from the other land two use sectors.

2.3 Urban Diffuse Pollution

This section identifies the changing nature of urban environments, and how the increase in impermeable surfaces has led to problems with quality and quantity aspects of water management in urban environments. The exploration of urban diffuse pollution that follows includes identification of the main pollutants in urban areas, their sources and pathways into watercourses, the main factors that determine pollution intensity in rivers as well as the barriers to assessment. It concludes by identifying the potential of best management practices to improve urban river water quality.

Urban landscapes are artificial environments and the hydrology of catchments is known to be significantly impacted by the level of urbanisation within them. The natural processes of the water cycle are disruptively influenced by the impermeable land surfaces that are typically associated with urbanisation. Urban land is considered to be that which is affected or adapted by humans, i.e., buildings, transport and other infrastructure. This also includes quarries, industrial facilities as well as areas that are not built upon but are associated with these activities, such as land fill areas and urban green spaces. Between 2000 and 2010 urban land cover increased by 141,000 to a total of 2,748,000 hectares and, although urban and artificial land use makes up only about 10% of the total land area in the UK, due to the heavily modified nature of the urban landscape the effect on water quality can be significant (Khan *et al.*, 2012).

To accurately quantify the impact of an urban area on a catchment it is necessary to take into consideration a wide range of factors that affect runoff (Beven, 2012). Whilst the urbanised area of a catchment is generally small, its effect on rainfall-runoff relationships can be significant, depending on the percentage of impermeable and permeable surfaces within an urban area and the associated network of natural and manmade drainage systems. Urban drainage infrastructure conveys runoff quickly to water courses, increasing the volume and reducing the quality of discharges (Butler and Davis, 2011).

2.3.1 Sources of Pollutants

Urban areas are strongly associated with a range of pollutants from both anthropogenic and natural sources. Principally there are 3 main sources of pollutants:-

- Atmospheric deposition
- Erosion and corrosion of surfaces
- Sewerage spillage and contamination

Atmospheric deposition occurs as particulates in the air settle back to the earth. Emissions from industry, such as from iron, steel and cement production, from energy production, particularly coal fired power stations or from waste incineration and transport activities contribute to this. Particulates also occur naturally from sources such as forest fires and volcanic activity. Depending on the particle size of dust and other matter, once airborne, they can be deposited onto surfaces such as buildings and roads or onto plant surfaces through either dry or wet atmospheric deposition. Heavier material will fall back to earth independently but finer material will remain suspended rising into higher strata of the atmosphere. This material will often be intercepted by falling precipitation and is often dissolved within rain droplets (Göbel *et al.*, 2007; Malmquist and Svensson, 1983). For example heavier material in dust and other small particles containing heavy metals such as copper arising from activities such as quarrying, will settle back to ground and will eventually be washed from surfaces by rainfall. Conversely gaseous emissions, containing pollutants such as sulphates from industrial chimney stacks will remain in the upper atmosphere and will be intercepted and dissolved into rainfall as it falls through the atmosphere.

As well as deposited materials, the erosion and corrosion of material from building roofs and other surfaces are also sources of pollutants. Buildings throughout urbanised areas vary extensively in age, materials, size and aspect. This not only affects the levels of potentially corrosive acidic rainfall deposited on a building, it also influences the amount of corrosion and erosion that it suffers as a result of weathering (Townsend, 2002). Roofs and guttering may be particularly susceptible to erosion and corrosion as traditionally metals such as lead and copper have been used as flashing for roofs as well as for guttering and down pipes. The different materials used on buildings result in variations in the type and concentrations of pollutants found in runoff (Chang *et al.*, 2004; Lye, 2009; Van Metre and Mahler, 2003).

Through combustion vehicular traffic produces a range of pollutants such as heavy metals and Polyaromatic hydrocarbons (PAHs). The polluting effect of vehicles on urban water bodies has been well documented (Legret and Pagotto, 1999; Napier *et al.*, 2008; Rule *et al.*, 2006). In addition, a range of other pollutants are deposited through the use of vehicles, including materials arising from road surface abrasion, tyre abrasion, brake pad wear, leaking of liquids (i.e. fuel, gear oil, grease, brake fluid, antifreeze, etc.) and materials generated by the wear of engine parts (Crabtree *et al.*, 2006; Sansalone and Buchberger, 1997).

As well as inorganic materials which result primarily from anthropogenic activity, organic pollutants are also found in urban runoff, including detritus material from plants and excrement deposited on urban surfaces from birds and domestic pets such as dogs (Göbel *et al.*, 2007).

Another important source of pollution to urban rivers is as a result of discharge from CSOs during rainfall and continuous emissions from WWTWs. While these are not strictly diffuse sources they remain an integral part of the urban water quality problem. The use of CSOs means that during peak storm flows raw untreated sewage is transmitted into river systems and this obviously has a significant negative impact on river ecosystems (Myerscough and Digman, 2008; Weyrauch *et al.*, 2010). The incorporation of CSOs into sewer systems has been prevalent in much of the UK due to the fact that much of the existing sewerage network system is combined (approximately 70%). The costs associated with expanding the capacity of systems to convey peak storm discharges to WWTWs, to thus avoid flooding and improve water quality are prohibitive (Butler and Davis, 2011).

Typically the majority of urban pollutants are from anthropogenic sources, such as the results of combustion, construction, transportation or any activity which results in the emission or production of particulate matter which is subsequently deposited on urban surfaces. These deposited pollutants are then transported via a number of different pathways into water environments.

2.3.2 Pollutant Pathways to Waterbodies

Historically urban areas have not been designed to retain precipitation. Thus collected precipitation is conveyed through guttering, pipes, channels and underground drains into existing natural channels, principally streams and rivers. Pollutants deposited on urban surfaces and those that are corroded and eroded from buildings and infrastructure are washed and flushed by water into the drainage systems where they are transported to receiving waters. Knowledge of pollutant build-up and kinetic wash off is limited (Egodawatta *et al.*, 2009; Goonetilleke *et al.*, 2005), primarily because it is difficult to measure and quantify.

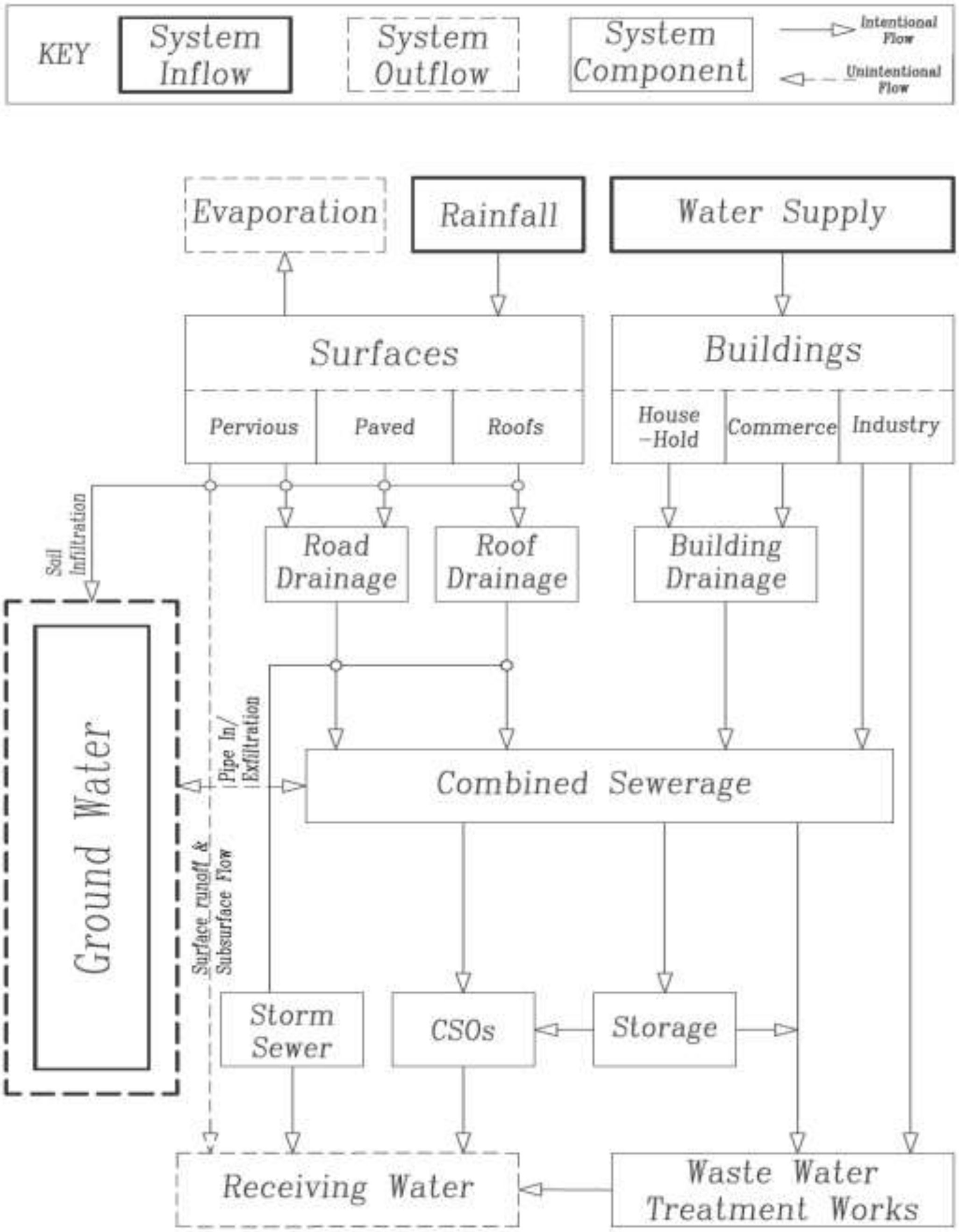


Figure 1 - Urban Water System: Hybrid System (Butler and Davis, 2011).

Figure 1 shows a diagram from Butler and Davis (2011) ‘Urban Drainage’ which provides an excellent summary of the different elements and interactions involved in a typical urban drainage system. The diagram shows a hybrid system which contains elements of both combined and separate sewerage and this represents the situation in much of the UK. This is due to the relative age of UK sewerage systems. Newer systems usually at the suburban periphery are totally separate systems, whereas central urban areas are older and are usually combined systems (Butler and Davis, 2011).

This situation leads to a series of potential pollutant sources within urban areas, with pollutants not only from surface runoff, but also from flooding of sewers and discharge of water via tripping of CSOs. Water and runoff passing along the pathways between components in the drainage system collect deposited pollutants and transport them towards receiving waters. As storm flows recede some of the particulate matter is deposited within pipes. Subsequent storm events with greater discharges occurring over a shorter time period result in scouring of this settled material which in turn leads to higher pollution peaks (Mulliss *et al.*, 1996).

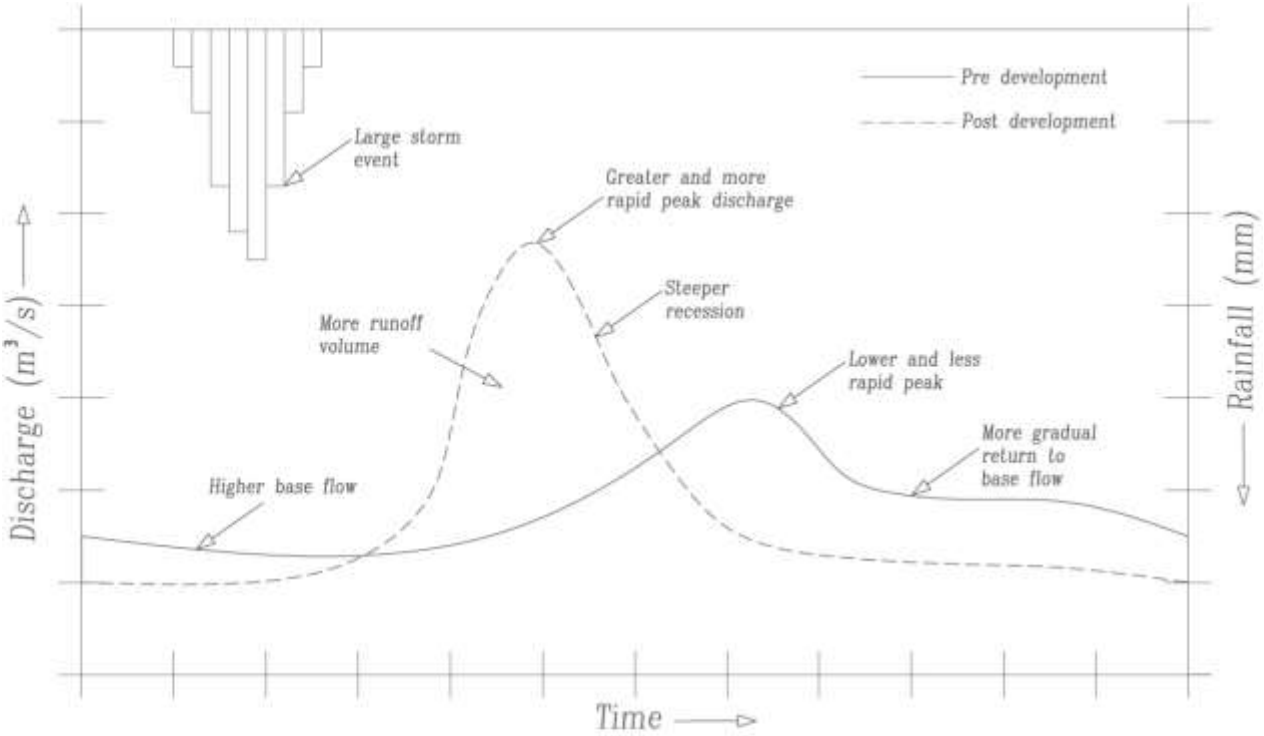


Figure 2 - Effect of development on river discharge (CIRIA. *et al.*, 2007).

With the installation of artificial drainage systems, the conveyance of runoff to streams and rivers is modified with piped drainage systems moving water much more rapidly into river systems. Greater overland flow and runoff is generated as a result of lower infiltration rates during rainfall, due to the removal of vegetated soils and their replacement with impermeable surfaces, preventing natural storage and attenuation of the subsurface. This results in reduced lag times on hydrographs as larger volumes of runoff enter rivers in a shorter period of time causing much higher flood peaks, as illustrated in Figure 2. This also leads to increased loading of pollutants in runoff and reduced infiltration times also leading to reductions in the recharge of ground water levels (Mulliss *et al.*, 1996; Wheater and Evans, 2009). Through a combination of landscape modification and rainfall patterns, storm water is the primary mechanism that transports non-point pollutants to receiving waters (Goonetilleke *et al.*, 2005). In an extensive study which monitored event loads and mean concentrations for 343 events Brezonik and Stadelmann (2002) concluded that rainfall amount and intensity as well as the characteristics of the drained area were the most important variables affecting event load, indicating the importance of urban land change on pollutant loads.

2.3.3 Combined Sewer Overflows

As has been previously stated (section 2.3.1) CSOs are a feature of most existing combined sewerage systems. When a combined sewer reaches its hydraulic capacity, a release mechanism is required in order to prevent sewer flooding. As this often results in domestic or commercial premises being inundated with dilute sewerage, it is often the lesser of two evils to discharge raw sewerage of varying concentration in to natural waterbodies. Overflows can be located anywhere on sewerage networks, e.g. on a branch sewer remote from a WWTWs, at a sewage pumping station; or on an inlet sewer to the WWTWs.

Discharge to a water course or other receiving water has always been the last option and to reduce this to a minimum many CSOs have emergency storage so wherever possible to avert the contamination of watercourses. Historically CSOs were designed to discharge once more than six times the dry weather flow (DWF) of the system was experienced, generally this meant that it was larger storm events which resulted in CSO discharge as the system was not designed to contain stormwater runoff. It was considered that during large storm events due to the dilution of foul water by storm water, that the negative impacts to watercourses were acceptable. New research in the 1970s (covered in section 2.3.4.1) demonstrated the harmful effects to watercourses of any foul water discharge and therefore a new standard for CSO was proposed known as ‘Formula A Flow’ (Butler and Davis, 2011).

This is calculated based on the type of sewer and, for a fully combined system, the following formula is used:-

Formula A Flow (1)

$$\text{Formula A} = \text{DWF} + 1306 \text{ x P} + 2\text{E}$$

Where:

Dry Weather Flow (DWF) = P x G + I + E
P = Population served
G = Water consumption/head/day (typically 150 litres)
I = Infiltration (varies between different catchments and geological areas)
E = Trade Effluent Flow (litres)

This formula varies between different sewer system configurations, for example, in a fully separate system 3 times the DWF is added to the 3 times the population served rather than 1306 for a combined system. This is primarily as the capacity of a fully separate sewer is considerably more predictable than a combined one as there are no large fluctuations due to surface water inflow to consider, this also means that generally the system capacity is not required to be as significant as volumes are smaller. Typically Formula A flow equates to approximately seven times the value of the DWF. For any CSO emitting to a receiving water that is particularly sensitive or that cannot provide sufficient dilution of its discharges, then storm tank or sewer capacity may need to be increased to accommodate greater than Formula A flow to reduce to occurrence and size of spills.

In the UK, the progress made by most of the major water companies with detailed sewer modelling of the networks they manage, it is now preferable for CSO design to be based on outputs from these models linked or combined with models predicting the potential impact of discharges on the water quality of receiving waters.

There are two principal methods available to reduce the spill frequency from CSOs. These are; to provide increased storage volume within the sewer network, or to reduce the volume of surface water entering combined systems through better source control therefore freeing up the capacity in the existing system. The reduction in surface water runoff, ‘source control’ is expanded on significantly in further sections of this literature review (section 2.4.1). While there is significant evidence to suggest that source control is more cost beneficial (Stovin et al., 2013), currently most major water companies in the UK are still pressing ahead with significant infrastructure investment within their asset management plans, to provide increased storage and capacity for sewer networks. For example, over the period from 2010 to 2015 United Utilities has invested over £200 million to reduce sewer flooding and prevent CSO discharge in central Manchester alone.

2.3.4 Pollutants

Through surface wash off, flushing and overflow of drainage and sewerage systems, pollutants are conveyed into aquatic environments. However the physical, chemical and biological impact will be dependent on the type and quantity of pollutants within these discharges. A wide variety of pollutants are typically associated with urban areas, but the following section focuses on the 3 main groups of pollutants that attract the majority of attention within the research literature. They are:-

- Nutrients and Faecal Contamination
- Heavy Metals
- Suspended Solids

2.3.4.1 Nutrients and Faecal Contamination

Nutrients have been long established as one of the most important groups of polluting chemicals to surface waters and the two main chemicals synonymous with this are Phosphorous and Nitrogen. These chemicals occur in different forms in riverine and lacustrine environments, their most commonly dissolved forms being Orthophosphate for Phosphorous and Nitrate, Nitrite and ammonium for Nitrogen.

A study in the U.S. by Puckett (1995) found that in the majority of the streams studied, non-point sources were the dominant source of Nitrogen, but less important in respect to Phosphorous. A similar experience was observed in the UK by Zhang *et al.* (2014) who reported on the cross sector contributions of total phosphorus, total Nitrogen and sediments. In the case of Nitrogen, inputs were dominated by agriculture, with approximately 80% contributed by the sector, whereas collectively urban sources (i.e. CSOs, storm tanks and diffuse urban sources) were estimated to contribute only 3.5% (when excluding WWTWs discharges). In terms of total Phosphorous the contributions were different, with WWTWs being identified as the main contributor at just under 50% and collectively urban sources (as above) contributing 17%. However diffuse urban sources only contributed 2% to this 17% indicating that the contribution of urban surface wash off to nutrient pollution is fairly insignificant in comparison with wastewater sources such as WWTWs and CSOs.

This view is corroborated by several other studies (Davies and Neal, 2007; Neal and Heathwaite, 2005; Rothwell *et al.*, 2010a) showing the greater importance of agricultural sources (see detail on agricultural diffuse pollution in Appendix I) with respect to Nitrogen inputs and the greater significance of wastewater point sources (i.e. WWTWs, CSOs and package sewerage treatment works) in respect to Phosphorous pollution. Rothwell *et al.* (2010a) reported that 44% of sample sites included in the EA's general quality assessment scheme in the North West had an average Orthophosphate concentrations greater than 0.12 mg/l (i.e. the WFD EQS for good quality), with many sites being characterised by proximity to wastewater point sources. This also highlights the importance of future urban expansion into greenbelt areas and growth of rural towns as a result of population increase and how this is likely to contribute to nutrient pollution loads (Jarvie *et al.*, 2006) without enhanced control measures.

In terms of the actual physical impact on receiving waters, the process of eutrophication is the primary negative impact associated with excessive nutrient inputs. Several definitions of varying complexity are given for the process in the literature and legislation; typically eutrophication is explained simply as the 'process of nutrient enrichment'. However such definitions do not cover the details and complexity of the chemical interactions and resulting undesirable effects. It is important to note that Nitrogen and Phosphorous do not contribute equally to the eutrophication process and it is generally the availability of Phosphorous and not Nitrogen which is the limiting factor of the process (B. J. D'Arcy *et al.*, 2000). This is further corroborated by the fact that losses from Orthophosphate are observed downstream of discharges as it is readily absorbed into the water column (Davies and Neal, 2007). Therefore catchment scale monitoring, where samples are taken at wide geographic distances from each other, may well fail to observe the full scale of Phosphate pollution to receiving waters.

Eutrophication has several negative impacts on an aquatic environment. Primarily it reduces biodiversity as the influx of nutrients allows the proliferation and dominance of nutrient tolerant plants and algal species. The dominance of these species disrupts the structure of the aquatic ecology as more sensitive species of greater conservation value are displaced. Eutrophic waters also typically experience oxygen depletion resulting in the death of invertebrates and fish. Eutrophication can also lead to adverse impacts on a wide range of water uses such as potable water supplies, irrigation etc., and undesirable aesthetic effects causing discolouration, sludge and foam formation on water surfaces (Hilton *et al.*, 2006; Smith *et al.*, 1999).

As the primary source of nutrient pollution in urban environments are sewerage sources there is a direct relationship between nutrient pollution and faecal contamination. FIOs are an important indicator of water quality. Not only are they a pollutant in their own right, being dangerous to human health, they are also a common indicator of sewerage contamination to water course, which not only raises concerns in relation to nutrient pollution but to other pollutants that are associated with foul sewerage discharges.

The cells of both *Escherichia coli* (*E.coli*) and *Intestinal Enterococci* (*IE*) are both used as FIOs for the detection of sewerage contamination of rivers and other fresh waters. They are commonly found in the lower intestine of warm blooded endotherms and have a limited ability to survive outside the body for long periods of time (McCarthy *et al.*, 2012; Noble *et al.*, 2004). This means they are well suited as indicator organisms to test water samples for evidence of faecal contamination (Masters *et al.*, 2011). These FIOs are the primary bacteria used by the EU's RBWD for the classification of bathing waters (European Commission, 2006).

The survival of a microorganism, such as E.coli or IE, in aquatic environments is dependent on the bacterium's ability to tolerate conditions that are alien to it. The tolerance of E.coli to biological and physio-chemical factors has been well documented, primarily in laboratory studies (Flint, 1987; Noble *et al.*, 2004; Whitman *et al.*, 2004). Understanding of both the mortality rates of bacterial indicators and the effect of dilution are crucial to understanding the findings of any study into FIOs.

Several factors are key to determining the mortality rate of FIOs in riverine and estuarine environments (Flint, 1987; Hood and Ness, 1982; Menon *et al.*, 2003; Servais *et al.*, 2007). The primary biological parameters include competition with native micro flora, virus induced lysis, autolysis and nutrient depletion, physio-chemical parameters such as insolation intensity, temperature fluctuation and stress due to osmotic shock when released into sea water. Of all these factors insolation has been identified as the most important (Burkhardt Iii *et al.*, 2000; Sinton *et al.*, 1999; Whitman *et al.*, 2004), however these studies on the effect of insolation on faecal bacteria have mainly been conducted in marine waters. Less information is available on the survival rates of FIOs in river water, particularly on how bacterial contamination changes with distance downstream from the sources of contamination (McCarthy *et al.*, 2012).

This is a crucial issue because evaluating faecal contamination at a catchment level dictates the frequency at which samples need to be taken to ensure sources of contamination are identified. As with nutrient pollution, in order to reduce microbial contamination, it is important to understand the primary sources of faecal bacterial inputs within an urban catchment. Sewerage inputs to urban surface waters can come from a variety of sources such as effluent from WWTWs, CSOs, small package Sewage Treatment Works and domestic cross connections (Mulliss *et al.*, 1996; Weyrauch *et al.*, 2010; Withers *et al.*, 2011). It is also important to have a reliable and comprehensive data set showing the contribution of pollutant levels from CSOs and other sources of microbial pollution which are induced by storm events and rainfall. However such data are not common and are sparse in the relevant literature (Langeveld *et al.*, 2012; R. Pitt *et al.*, 1993).

In summary, the risk of eutrophication is more dependent on the discharge of Phosphate than Nitrogen. Several authors (Puckett, 1995; Rothwell *et al.*, 2010a; Zhang *et al.*, 2014) have shown that the contribution of urban sources, when factoring-in sewerage sources, is more significant than in the case of agricultural land use thus it is crucial for urban sources to be addressed in the context of achieving the WFD goal of 'good' water quality status by 2015.
As Phosphorous and faecal contamination is typically discharged from similar sources, addressing them collectively could be undertaken using a common approach, and provide benefit in respect to compliance of both WFD and RBWD targets.

2.3.4.2 Heavy Metals

Multiple studies (Mulliss *et al.*, 1996; Neal *et al.*, 2000; Rothwell *et al.*, 2010a; Rowland *et al.*, 2011) identify that in respect to urban pollution, as a general group heavy metals are important. Generated from a wide range of sources they are found in runoff in both particulate and dissolved form and where they occur in excess they are toxic to aquatic life. Typically studies focus on a relatively small number of metals, i.e. those that are listed as priority substances under the WFD or they have been historically identified within a catchment as problematic. The following metals have been identified as being of the most concern in respect to urban water pollution:-

- Lead (Pb)
- Chromium (Cr)
- Aluminium (Al)
- Arsenic (As)
- Cadmium (Cd)
- Zinc (Zn)
- Nickel (Ni)
- Copper (Cu)
- Iron (Fe)

There are two main ways that the impact of heavy metal pollution on receiving water can be measured. This is in terms of the severity on the immediate and long term toxicity to aquatic fauna. High concentrations of heavy metals are associated with storm flows and discharges can 'shock' aquatic environments as polluted water with several times the concentration of receiving waters flows in (Goonetilleke *et al.*, 2005). Short term effects can also be caused by discharges from urban surface runoff. Several studies (Crabtree *et al.*, 2006; Jun Ho Lee and Bang, 2000; Mulliss *et al.*, 1996; Rule *et al.*, 2006) have identified high heavy metal concentrations experienced in urban surface water outfalls during rainfall. Both Mulliss *et al.* (1996) and Weyrauch *et al.* (2010) identified the high contribution of intermittent sources such as surface water outfalls in terms of total annual pollutant load and that they can be greater in comparison with consistent sources such as WWTWs. This demonstrates that discharges from surface water outfalls are highly concentrated in terms of heavy metal pollutant load delivered over a short time period.

Heavy metals can also cause long term impacts as they can bio-accumulate within the food chain and this has been well established in terrestrial fauna (Heikens *et al.*, 2001). However there has been little research into this affect in aquatic flora (Goodyear and McNeill, 1999) and as flora absorb small amounts of metals, concentrations are compounded as this moves up the food chain leading to potential toxic effects in fish and aquatic mammals which consume a large amount of insects and plants containing small amounts of heavy metals. Goodyear and McNeill (1999) identified strong correlations between environmentally available heavy metals and the concentrations found in macro-invertebrates.

Two studies have undertaken substantial reviews of stormwater pollutant concentrations, and specifically in terms of metal concentrations reported that values fluctuate significantly from fractions of a microgram per litre (µg/l) of water to several thousand (Göbel *et al.*, 2007; Ingvertsen *et al.*, 2011). These reviews found that Copper, Zinc, and Lead exhibited the highest median concentrations, varying from 15 to 2600, 103 to 6000, and 10 to 344 µg/l, respectively, whereas Cadmium, Chromium, and Nickel range from 0.7 to 4.2, 4 to 15, and 4 to 45 µg/l respectively. It should be recognised, that larger concentrations of one metal compared with another may not necessarily reflect greater impact on receiving waters.

2.3.4.3 Suspended Solids

Suspended Solids is a general term applied to the solid material which is carried in suspension by the flow of a river and the dominant contribution to suspended solid loads in rivers is from diffuse sources over that from sewers or industrial effluents (B. J. D'Arcy *et al.*, 2000). The specific composition of the solid column of a river discharge is dependent on several factors, land use, characteristics of the catchment (i.e. soil type, geology, topography, vegetation cover and local climate) (Helmer and Hespanhol, 1997), discharge volume and velocity (Shaw, 2011) and also important is the modification of runoff as it is transported to receiving waters.

Typically the solids within a river channel primarily consist of fine to medium grained soil particles, plant detritus, particulate Phosphorous, Nitrogen, carbon, silica, heavy metals and pesticides. This is in the form of eroded and wash off particulate matter light enough to be carried by the rivers flow, usually composed of particle sizes of less than 0.062mm and also river bed sediments, particles generally greater than 0.062 but dependant on the quantity and size of different materials on the river bed (Knighton, 1998). Ultimately the size of suspended particles is a result of the strength and velocity of river discharge.

Knowledge of the particle size of the solid element of the river pollution is essential as it has been shown that heavy metal and PAH concentrations in water samples demonstrate a strong association with that of total suspended solids (Horowitz, 1995). This is important as solids have been used as a surrogate indicator of the presence of other pollutants as they act as a mobile substrate for pollutants such as metals and PAHs (Goonetilleke *et al.*, 2005). There is further debate in the literature in respect to which particle size fraction carries the greatest amount of pollutants and this remains a contentious issue, with strong implications for management decisions and design of treatment systems. J.-Y. Kim and Sansalone (2008) found that between 65-99% of particulate matter in effluent flow was smaller than 75µm. This can be attributed to the fact that smaller and less dense particles have a greater likelihood of being mobilised by storm flows (Ingvertsen *et al.*, 2011). Natural particle variation as well as differences in sampling and analytical methodology may be responsible for discrepancies between Particle Size Distribution (PSD) studies (Li *et al.*, 2005).

Irrespective of particle size suspended solids have the potential to have negative impacts on aquatic environments in a number of ways, and again the specifics are dependent on catchment characteristics. Typically they can effect aquatic flora and fauna through increased turbidity reducing light penetration and in turn photosynthesis, eutrophication when carrying nutrient particulates, modification of habitats and blocking the feeding mechanisms of filter feeders and gills of other aquatic organisms (Gray, 2010). Pollutants which are bound and absorbed within solids such as heavy metals, pesticides, nutrients and faecal pathogens are dissolved in river water and ingested by flora and fauna. This can also affect chemical aspects of waters affecting reactions, solubility and interactions with river bed and other dissolved material. They also cause siltation and sedimentation of waterways, rivers and reservoirs adversely affecting their economic activity (B. J. D'Arcy *et al.*, 2000).

2.3.5 The 'First Flush' Effect

The 'first flush' phenomenon is another contentious issue in urban drainage and various studies have argued its significance. It relates to the initial portion of runoff at the beginning of a storm event and suggests that this segment of runoff is more highly concentrated with pollutants than the runoff discharged during the mid or latter stage of the storm or rainfall event.

There is conflict in research over the significance of the 'first flush' effect, with some studies showing an important and distinctive contribution (Barco *et al.*, 2008; J. H. Lee *et al.*, 2002), whereas others have found it less significant or struggled to identify it at all (Hall and Ellis, 1985; Saget *et al.*, 1996). Despite this, the effect is commonly reported and defined in qualitative terms, however an increase in pollutant concentrations in the early stages of a storm event alone cannot be considered to be adequate evidence (Goonetilleke *et al.*, 2005). A likely reason for these variations in experience is due to the multiple definitions that are used to characterise the phenomenon. Generally studies use a curve of the cumulative fraction of total pollutant mass in comparison with the fraction of total cumulative runoff; where the pollutant mass exceeds total runoff for a chosen percentage of time the 'first flush' is considered to be demonstrated. However this percentage varies between studies and when the different conditions and catchments used in studies is factored-in, it then makes comparison between results very difficult (Deletic, 1998).

The physical location that sampling is conducted is also important, Deletic (1998) concluded that strong 'first flush' effects observed at the end of drainage systems were more likely due to transformations and transport of material from within the system, rather than an influx of material into them. To further complicate matters both J. H. Lee *et al.* (2002) and Barco *et al.* (2008) found that there were variations between the strength of the effect for different pollutants. From this it can be concluded that the effect is highly complex and the interaction of a wide range of variables affects the extent to which it is observed. Data which accurately characterises the scale and importance of this phenomenon is important, as it has significant economic implications for management decisions and the design of storm water treatment systems, e.g., provision of structural retention measures such as basins or storm tanks designed to capture the first few centimetres of runoff (Deletic, 1998).

2.3.6 Importance of Over Abstraction and Low Discharge Periods

During low base flow periods the dilution of pollutants becomes much lower, meaning that they are likely to become more concentrated, making up a greater portion of a river's discharge. In these conditions pollution sources can have a disproportionate effect on receiving waters with concentrations of nutrient chemicals having a more negative impact on the ecology of a river system. Low river levels are largely due to low rainfall, but in the UK over abstraction can also lead to low discharge conditions. This is a particular problem in southern parts of England where a larger proportion of water supplies are from river abstraction. In a review in 2009 Cave (2009) identified that one third of catchments are over-abstracted or over-licensed in England, whereas this is only 16% in Wales. This is an issue for surface waters in summer months where inputs from rainfall are low, but this can also affect ground waters, e.g. from boreholes, all year round.

When considering abstraction in the UK it is also important to consider water availability in comparison with population density. Whilst it makes up just over half of the UK land mass, England contains a much greater population density with just under 84% of the population (ONS, 2012), while also receiving much less rainfall than either Scotland or Wales. Figure 3 shows how much less rainfall England receives in comparison with Scotland and Wales (Met Office, 2014).

Also evident is the considerably low rainfall experienced during 2010 and 11; particularly in 2011 when the unusually low levels of rainfall did not recharge ground water and reservoir levels in large parts of England, which resulted in the drought in early 2012. This resulted in water restrictions in the early part of the year and the declaration of "Drought Zones" in over seventeen English counties. During such periods the potential for greater negative impacts from unregulated urban discharges is important when considering regulated discharge consents, which are based on the receiving water ability to assimilate discharged pollutants without detriment to it.

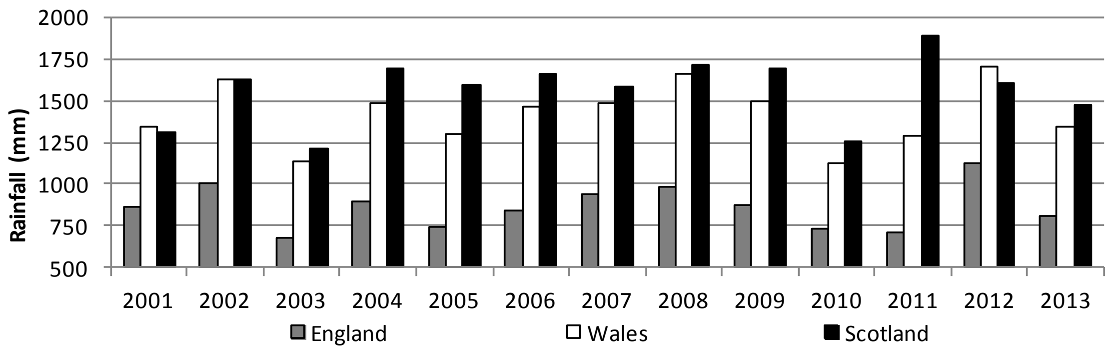


Figure 3 - Average Rainfall in England, Wales and Scotland (2001-13) (Met Office, 2014).

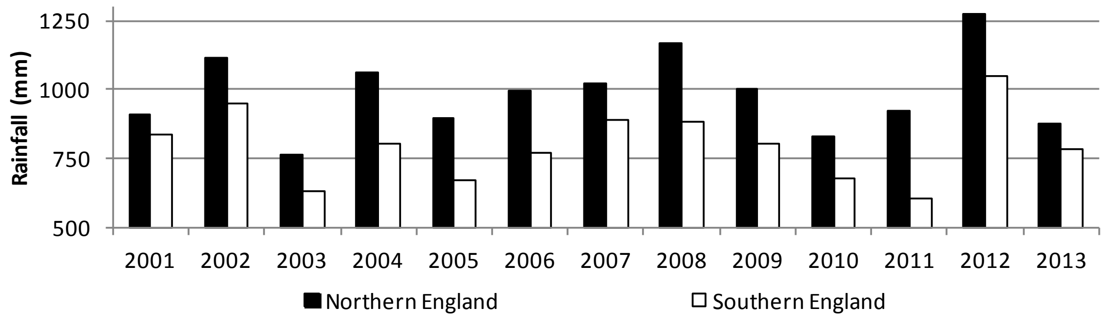


Figure 4 - Average Rainfall Northern and Southern Britain (2001-13) (Met Office, 2014)

When further examining the available rainfall data for England, there is a clear north-south divide, with the south consistently receiving less rain that the north as shown in Figure 4. Factor in greater population densities and abstraction rates in the south of England and low flows become a significant concern in relation to their ability to dilute urban pollutants.

2.3.7 Development and Use of Best Management Practices

As has been established, the key driver behind urban diffuse pollution is rainfall, which makes this pollution episodic in nature in that it occurs as a result of and during rainfall. The existing EQS system was established to observe long term trends in water quality and identify consistent pollutant inputs. The exploration of the literature above indicates that the EQS system is inappropriate for the identification of diffuse pollutants, where pollutants emit from multiple different locations and concentrations fluctuate significantly during wet weather periods. Sampling undertaken to assess compliance with EQS is taken across a wide spatial areas, further reducing its potential to identify locations where pollutants are entering surface waters, which in the context of an urban river may be nucleated into a short stretch of the river's course (J. B. Ellis *et al.*, 2002).

BMPs in respect to land use are a critical part of ensuring improvement in and maintenance of reductions in pollutants and runoff water quality. Existing BMPs for forestry, in the form of 'the UK Forestry Standard the governments' approach to sustainable forest management', and for agriculture through the delivery of agri-environment schemes such as the countryside stewardship scheme and the catchment sensitive farming initiative are explored in Appendix I.

A range of BMPs are available for urban environments and most take the form of guidance on the use of SuDS. Much of this guidance in England and Wales comes from the Construction Industry Research and Information Association (CIRIA) mainly in the form of the SuDS Manual (C697) (Woods-Ballard *et al.*, 2007), but also in other guidance notes and documents, such as, 'The Water Sensitive Urban Design in the UK' guide (Morgan *et al.*, 2013) as well as the range of fact sheets available on the CIRIA website: http://www.susdrain.org/. These documents offer a range of advice in relation to better management of surface runoff in urban areas, such as increasing the proportion of permeable areas and thus increasing infiltration and reducing runoff.

The main difference between current BMPs for urban environments in England and Wales is the lack of any incentive mechanism, either statutory or financial, to encourage the take up of urban BMPs with respect to water quality (B. D'Arcy and Frost, 2001). As outlined in section 2.1.3, there have been continued delays to the introduction of mandatory requirements for SuDS for new developments. Considering the long turnover period of UK building stock (70-90 years) it would take a considerable and unacceptable time to address the issue of urban water pollution through the application of SuDS and BMPs only to new developments.

The mechanism used by the forestry commission would be a good example to emulate in an urban context as this would enable much of the existing guidance and advice to have a far greater amount of influence in improving urban runoff quality in the short term. It would be challenging to enforce due to the larger number of landowners, asset owners, local authorities, water companies, energy companies, residents and a range of other interested parties in urban areas compared to forested areas. However the wide range of groups with a vested interest would also allow the financial burden of implementation of BMPs to be spread more widely, thus reducing costs to individuals and organisations.

This section (2.3) has highlighted the most important sources, transport methods and pollutants that are associated with urban areas. It also highlights a series of contentious issues around the presence of the 'first flush' effect and the importance of particle size in the design of WWTWs such as SuDS and PTS. It emphasises the importance of mitigation and treatment of urban pollution in the context of meeting water quality objectives set in the legislation covered in the section 2.1. Using this knowledge the section 2.4 explores the principles and design of WWTWs, such as SuDS, as well as the availability and capabilities of off-the- shelf PTS.

2.4 Urban Storm Water Treatment

The previous section identified the importance of the urban stormwater problem and how it contributes on a wider scale to river pollution. It also introduced the key components around urban water pollution, as well as some concepts around storm water that are influential in the design of treatment solution methods and systems. This section identifies the key principles of stormwater management and treatment and briefly describes the most commonly utilised systems in the UK. Finally the section explores the physical methods of stormwater treatment and management, some of which have previously been mentioned. Principally these systems take two main forms, which are Proprietary Treatment Systems (PTS) and Sustainable Urban Drainage Systems (SuDS).

In the UK it is impossible to talk about urban storm water treatment without also discussing SuDS and certainly in the UK the two terms are synonymous. SuDS are also commonly referred to as structural BMPs outside the UK; other terms such as Green Infrastructure are also included under the umbrella of SuDS. It may be helpful at this point to clarify that 'SuDS' as a generic term is often used to refer to not only the physical systems or actions that are put in place, but also the principles behind the systems. The term includes a broad and complex range of different drainage solutions to a range of potential pollution sources made by different organisations (DEFRA, 2004). A more accurate way to think about it is that SuDS are surface water drainage systems that are developed in accordance with the principles of sustainable development (CIRIA. *et al.*, 2007). As will be explored further below the principles of good stormwater management and the principles behind sustainable drainage are largely similar.

To further clarify this, it is useful to begin by examining the definition of SuDS and as with several other terms and concepts defined in this review, the term SuDS means a range of different things to a range of different organisations and stakeholders, however the one used here is that used in the 600 page CIRIA compendium on the subject, the 'SuDS Manual':

"An approach to surface water management that combines a sequence of management practices and control structures designed to drain surface water in a more sustainable fashion than some conventional techniques."(CIRIA. et al., 2007).

This can be further elaborated by considering that “Sustainable Drainage” means managing rainwater (including snow and other precipitation) with the aim of;-

(a) reducing damage from flooding,

(b) improving water quality,

(c) protecting and improving the environment,

(d) protecting health and safety, and

(e) ensuring the stability and durability of drainage systems.”

(Department for Environment Food and Rural Affairs, 2010).

Although different organisations with different priorities see sustainable drainage differently, to fully maximise the benefits of sustainable drainage to stormwater management it is important to understand the principles upon which it is based.

2.4.1 Principles of Sustainable Drainage and Stormwater management

The key objectives of sustainable drainage are to reduce and mitigate the impacts of an urban development on the volume and condition of runoff, whilst also providing other benefits to biodiversity and amenity (Morgan *et al.*, 2013) as conceptualised in Figure 5.

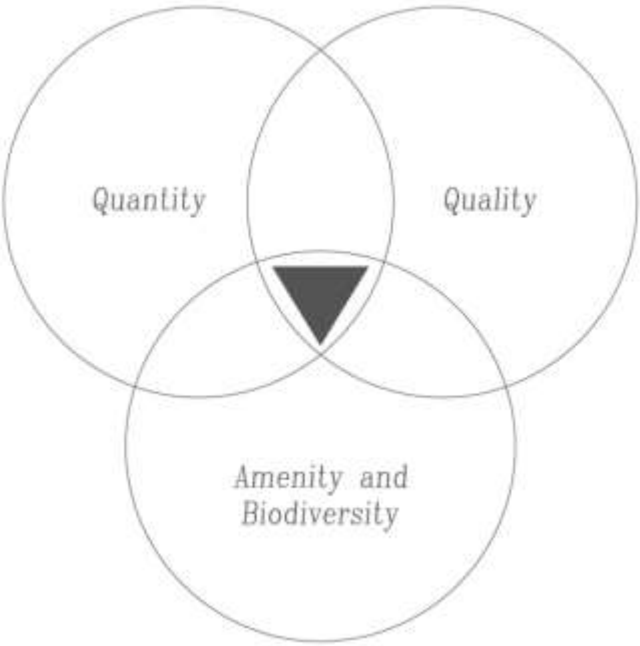


Figure 5 - Sustainable Drainage Objectives (CIRIA. et al., 2007)

In a perfect solution all three objectives would receive equal emphasis; however this is rarely possible in practice due to the variation in site conditions and the agendas of stakeholders. Emulation of the natural drainage conditions of a site prior to development is the philosophy that underpins sustainable drainage (DEFRA and EA, 2009).

However achieving this objective within the context of a highly urbanised area is not always possible. In this case there are a series of principles and measures applied in an integrated sequence which when correctly implemented incrementally reduce pollution and volumes of runoff (CIRIA. *et al.*, 2007). This process is termed a 'treatment train' or 'management train' and the principles should be applied in the following order:-

1. Runoff Prevention – well-designed individual properties and premises should minimise available pollutants and runoff. For example sweeping of impermeable areas removing pollutants and use of rainwater harvesting or storage for reuse on site.
2. Source Control – runoff that cannot be stored should be controlled at or very close to its source (where precipitation falls). For example runoff should be minimised through use of soakaways, or other means of infiltration.
3. Site Control – management of runoff collectively at a local site or area, for example collection of runoff from nearby building roofs or car parks and conveyance of it to a local area where is put into a control measure such as detention basins or larger infiltration measures.
4. Regional control – larger scale collection of runoff conveyed from multiple sites, for example using a large detention basin, balancing pond or wetland.

Figure 6 shows the management train graphically, where the principle aim is to treat water and try to remove the need for it to accumulate through conveyance in a piped drainage system however it is recognised that this isn't possible at all sites. It is not necessary for runoff to pass through all stages in the train, it could move directly to site control but the principle is to deal with runoff locally and return it to the natural drainage network as near to the source as possible. Good design and use of measures at a local level can remove the need for more substantial control measures further down the train. Typically the further down the process train runoff progresses, the greater the cost and energy consumption used in control measures. For example water collected in larger basins may well require additional pumping and treatment prior to its reintroduction into the local water courses, resulting in the need for additional measures such as diversion to WWTWs or treatment using a proprietary system (CIRIA. *et al.*, 2007).

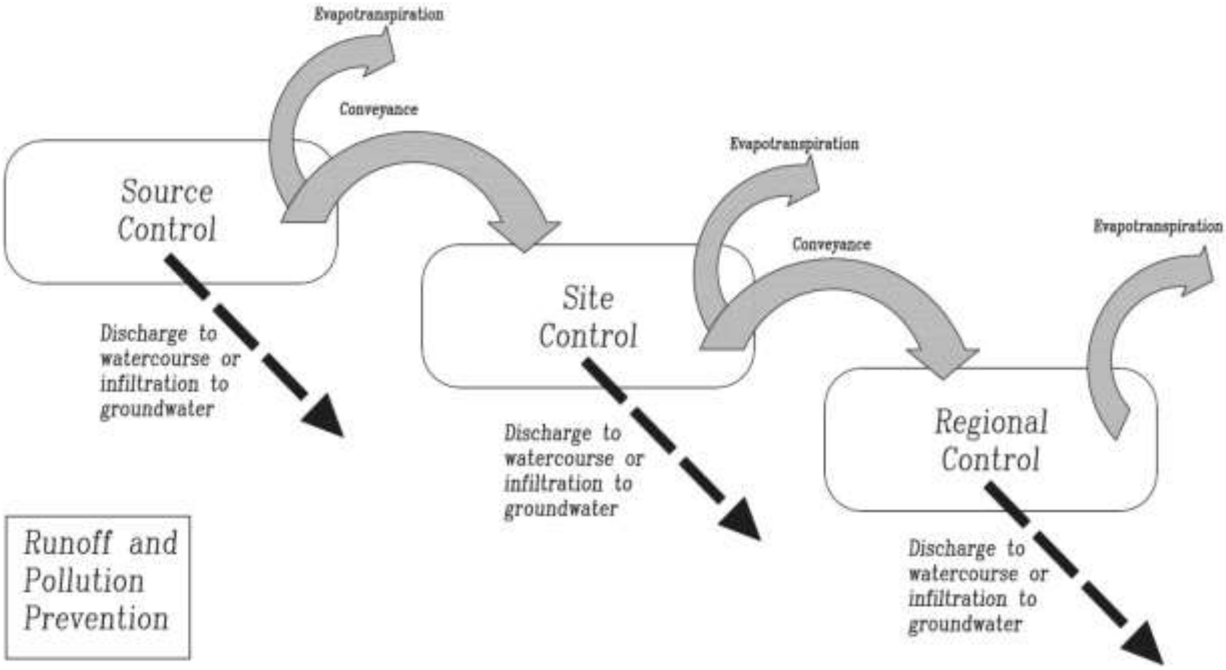


Figure 6 - SuDS Management Train (CIRIA. *et al.*, 2007).

The management of urban runoff and the design of control measures require active decisions to select between different options. This is often dependant on the risks associated with different actions, e.g. risks of area flooding will be balanced with the cost of protecting the area from different levels of floods. The management train concept promotes division of areas with different drainage characteristics and land uses into sub catchments each with its own drainage strategy. Dealing with water at a local level not only reduces quantities that have to be dealt with at one point, but also reduces the need to convey water off site via a piped system. During sub division of catchments into smaller areas it is important to retain a perspective on the effect this has on the whole catchment management and hydrological cycle (DEFRA and EA, 2009).

As well as the principles embodied in the management train there are a range of other concepts and actions that are important to the overall concept of good storm water management and again the fundamental principle is to revert drainage conditions, where ever and as near as possible, back to the natural drainage conditions of the site prior to development. These concepts are further divided in to two groups. The first is the higher level mechanisms and physical actions whereas the second is the individual processes which occur within different structural measures such as SuDS and PTS. These higher level mechanisms involve:

1. Infiltration – This is the soaking of water into soil and the subsurface, which is the most preferential process to emulate as it restores or retains the natural hydraulic process that would be prevalent if the site was undeveloped. Infiltration rates vary with different soil types and composition, condition and volume of rainfall. Where ground water supplies are sensitive to pollution, generally infiltration of significant volumes of urban runoff is undesirable as it can carry dissolved pollutants into ground waters.
2. Detention/attenuation – This is the process of capturing and calming of surface waters before they are discharged to receiving waters. Typically achieved through the use of dry basin/ subsurface tanks, permanent pond/wetlands with a restricted outlet.
3. Water harvesting/ reuse – Capture and retention of site runoff for the purposes of reuse for lower grade activities, such as toilet flushing, irrigation of gardens or car washing. The benefit of such systems to water quality improvement and mitigation of flood risk will depend on their scale. It also needs to be ensured that some storage capacity is available prior to storm events to avoid the discharge of additional runoff due to bypass of storage during rainfall events.
4. Conveyance – Relating to the transport of runoff from one location to another through a range of different mediums such as open channels, piped drainage and trenches. An important part of managing flows allowing the connection of different stages of the management train together.

These represent a series of physical movements of water which when correctly managed result in preferential outcomes such as mitigation of flooding and within which a combination of physical, chemical and biological processes result in water quality improvements (CIRIA. *et al.*, 2007; S. Wilson *et al.*, 2004). The physical, chemical and biological processes within structural measures include:

1. Sedimentation/ Settlement – The solid fraction of runoff is known to contain significant levels of pollution, as pollutants bind with sediment particles. By lowering the flow velocity of a body of water sediments and particles fall out of suspension (as previously discussed in section 2.3.4.3), the quantities of solids and sediments that can be carried by water is largely subject to velocity and particle size (Knighton, 1998). During system design care should be taken to ensure minimal re-suspension of solid material by

subsequent re-inundation of the system. This relies on simple gravitational settlement requiring longer retention times in order to remove smaller particulates from suspension.

2. Vortex/Cyclonic Separation – This functions on the principles of sedimentation and settlement but utilises the high velocities to direct water laterally, in a centrifugal motion, allowing sedimentation to function over a shorter distance and time. It relies on enhanced gravitational settlement, through the use of a rotating flow field (Faram *et al.*, 2003).
3. Filtration and biofiltration – Extremely fine sediments will take significant time periods to fall out of suspension and filtering them maybe more appropriate. As waters flow or percolate through aggregates, sediments or geotextile layers in the constructs of systems this finer fraction of material will be removed from flows. This is usually used when higher grade water quality is required, however they are prone to become clogged and blocked with larger sediments if it has not been previously removed.
4. Adsorption – As water percolates through soils pollutants become bound to soil particles, this process is complex and is dependent on chemical compositions of soils/ media and runoff. Several process however are prevalent, adsorption of pollutants as they bind to soil/ aggregate surface, attraction between clay minerals and cations, solutes in runoff are absorbed or chemisorbed into soils/aggregates and organic matter. Acidity of runoff is important as it affects the rate at which pollutants are absorbed by soils or substrates. Once saturation occurs pollutants will cease to be absorbed by a section of soils. Specially designed media can also utilise this process, such as sponges or substrates which bind or capture specific pollutants such as hydrocarbons or phosphates. Plants can absorb pollutants from soils and can further concentrate nutrients and heavy metals as previously discussed (section 2.3.4.2)
5. Biodegradation – Break down of water pollutants through contact with biological treatment, such as passing runoff through microbial communities which consume chemicals, nutrients oils and greases, removing them from runoff. The effectiveness depends on the maintenance of suitable conditions within soils or media (correct temperature, pH, oxygen supply, suitability of media and materials for colonisation).
6. Volatilisation – As chemical reactions occur within them, soils and pollutants are subject to changes in temperature and pressure so that some pollutants will be converted to vapours and gases. As this occurs they are transferred to the air in the soil and ultimately back to the general atmosphere. This process is primarily concerned with organic compounds such as pesticides and sections of petroleum pollution.

Control measures essential to good storm water treatment utilise these processes to effect water quality improvement and can take many different forms, however as mentioned for the purposes of this project they have been divided into two key groups, the first of these is SuDS (S. Wilson *et al.*, 2004).

2.4.2 Sustainable Urban Drainage Systems

As already identified in the literature, SuDS are synonymous with good storm water management in urban areas. There is a wealth of material on the subject, and selecting that which is most important and critical is difficult. With the principles of good storm water management established, this section briefly describes the main components used. As discussed in section 2.4.1, SuDS are designed along a management train, which prioritises minimisation of runoff or by dealing with rainfall as close as possible to where it falls. Ultimately however there are three key physical actions that to a greater or lesser extent all SuDS function on, which are:

- detention and storage of water,
- facilitation of infiltration of water into soils and the subsurface, or
- conveyance of water to another site.

Systems utilise more than one of these functional components as summarised in Table 1.

Table 1 - Summary of SuDS Components

Component	Description and Summary of performance.
Permeable pavements	Performance depending very much on the location and runoff conditions experienced, they typically utilise a mixture of sediment stratification and geotextiles. When functioning correctly they deliver good removal of all forms of pollution. A good quality geotextile is important for effective hydrocarbon interception and treatment.
Wetlands	Dependent on size and retention time wetlands can produce good pollutant removal, as they have a permanent pool of water they are less efficient at removing oils and hydrophobic pollutants, which in infiltration features are broken down in the soil subsurface. Their effectiveness in pollution control depends on water retention time and their available storage volume governs their effectiveness as flood control measures. Seasonal vegetation can result in nutrient collection and subsequent release; however this can be controlled through good maintenance.
Basins/ Infiltration Basins	The effectiveness of basins is very dependent on their design and the hydraulic conditions of rainfall events treated. They are able to achieve high levels of infiltration, but nutrient reduction is generally minimal due to a short retention time, although this is subject to the levels of infiltration they achieve. Measurement of pollution levels in the sediments of basins has shown that infiltration basins are effective at capturing pollutants present as suspended solids. This also affects the risk of transfer of pollutants to ground waters as dissolved pollutants are not always removed.
Filter strips	Primarily functioning as sediment filtration devices, the composition of sediment layers, vegetation, slope length and uniformity are the most important factors in their performance. They can be difficult to use in urban areas due to their space requirements but they are popular for treatment of stormwater or highway runoff at a site level.
Green roofs	The runoff experience from green roofs is very much dependant on the materials used in their construction. Bedding material of the roofs can retain hydrocarbons which can be washed out in heavy rainfall; typically runoff produced contains fine organic particles which can be easily treated using a soakaway or infiltration feature.
Rainwater Harvesting Systems	Collection of rainwater from roofs and storage for reuse for lower grade functions on properties such as outdoor uses like car washing or for watering gardens, indoor uses such as toilet flushing. Runoff quality is very much dependent on the level of atmospheric deposition on roofs and the materials used in roof construction. Here metals can lead to contamination through corrosion. Popular materials in the UK such as roofing slates and tiles produce low levels of runoff contamination.

Table 1 - Summary of SuDS Components (continued)

Component	Description and Summary of performance.
Ponds	Functioning in a similar way to wetlands, the performance experienced is fairly comparable. Short circuiting and stratification can result in anaerobic effects in sediments in the bottom of the pond, which will limit how effectively a pond is able to perform. Due to the reduced levels of vegetation compared with wetlands they require reduced maintenance. Performance in terms of treating nutrients is strongly correlated with the retention time of water in the pond, so large ponds with longer retention periods can maximise water quality improvements.
Swales	Common in many countries especially the USA, they deliver good performance in respect to reducing runoff volumes and providing treatment capacity. Pollutant removal is a subject to design, composition of and physical characteristics of materials used in construction, it has been demonstrated that up to 80% of runoff volume can be mitigated. This means that while concentrations remain in runoff, significant amounts of pollutants are removed. Water quality performance then is primarily governed by reduction in flow volume with hydraulic performance being their most beneficial effect. Under drained swales, which utilise further subsurface drainage will result in further improvements in water quality improvement. Operational performance is maintained provided standard maintenance is completed
Infiltration trenches and soakaways	Typically used for effluents with low sediment loads such as roof drainage, they can be utilised for road runoff but there is a risk of clogging due to sedimentation which discourages their use for this purpose. Due to the pollutants associated with rainfall and road runoff soils and sediments below soakaways can contain hazardous levels of pollution however this is typically contain within the top 300mm of material and can be removed and disposed of through appropriate maintenance.
Filter trenches	Functioning in much the same way as trickle filters they produce good improvements in water quality, this is chiefly due to their low treatment capacity and greater retention time as they are typically fed with a detention feature. With good nutrient retention (Nitrate 30% removal, Phosphate 60% removal) solids are also removed well and up to 80% of metals. As with infiltration trenches and soakaways infiltrations can experience pollution of the soils in the subsurface as they are retained from water infiltration. Trenches are then filled with aggregate of graded size to filter water as it percolates down, the efficiency depends on the composition of the aggregate and the flow the drain experiences. Stones and larger material in trenches can become blocked and clogged with sediment and sustaining good performance in terms of pollutant removal is subject to an appropriate maintenance schedule.

Compiled using DEFRA and EA, 2009; Jones and Macdonald, 2007; Pratt, 2004; Wallingford and Bray, 2004; S. Wilson et al., 2004; Woods-Ballard et al., 2007

From Table 1 it can be seen that there are suitable SuDS for almost all applications and through the use of different components in sequence significant water quality improvement can be achieved and, as a by-product of this treatment, systems retain larger volumes of water which has a subsequent flood control benefit. However the converse of this is that systems at a local or regional scale often require significant space. This may be suitable for new suburban developments but in most modern urban environments space is not a commodity that is usually in great supply. Therefor in existing “tight” urban situations there is a need for systems that can provide similar treatment capacity but which do not require as significant a foot print. In this respect PTS’s can offer a distinct advantage over SuDS.

2.4.3 Proprietary Treatment Systems

Proprietary Treatment Systems, as with SuDS, function along many of the principles of sustainable drainage; however there are some exceptions to this. Generally PTS do not provide significant flood control benefit as they are primarily concerned with water treatment and pollutant removal rather than providing any infiltration or retention of runoff. Typically the systems described here are designed to be installed within conventional piped drainage systems and can be readily retrofitted to existing systems.

In contrast to SuDS, PTS’s are typically constructed off site and delivered for installation as sealed systems, encapsulated within an outer skin with a defined inlet and outlet. Pollutants are removed and stored within the system and removed by regular maintenance. Systems such as these are less prevalent in the UK but are now well established in other countries such as the USA. Table 2 provides a summary description of main systems available.

Table 2 - Summary of PTS

Treatment System	Description and Summary of performance.
(Hydro Dynamic) Vortex Separators	In the context of proprietary water treatment a HDVS is a chamber (new or existing) placed to capture the flow of a piped or channel drainage system. Within the chamber the water is forced down and is then directed into another internal chamber in a centrifugal upward motion. During this process suspended solids fall out of the flow into a collection sump at the base of the chamber while water is directed upwards and out of the chamber. Oils and other floatable pollutants rise to the surface of water and are collected by a floating screen which sits between the outer and internal chamber walls rising and falling with the water level. Their removal efficiency varies depending on discharge rates and the particle size of the suspended solids passing through the system and so they are typically built to a specific size to deal with a target discharge rate. Maintenance through emptying the lower sump where solids gather and the removal of floatables and collected oils is essential to good performance, the required frequency depending on the quality of the effluent treated and the capacity of the system.
Filter systems	Filter system have a number of similarities to HDVS in the way they function; again they comprise a chamber into which piped drainage is directed. Water enters the chamber and is directed downwards and is then passed through filters on either side of the chamber. A screen prevents larger material from passing into the filter (which settles in the chamber bottom), flow distributing media directs flow evenly across the filter media as the water level rises through the chamber. Treated water that has passed through the filter media then passes out of the chamber. A number of different selections are available for filter media allowing the removal of a number of different pollutants. Again appropriate maintenance is essential as filters need to be replaced at a set periods depending on the quality of treated effluent.
Sorption/ Absorption systems	Sorption and Absorption systems act to remove specific pollutants from effluents such as nutrients and hydrocarbons; again they are generally contained within an outer container with effluents passed through a media, substrates or bacterial cultures contained within. Systems such as these are designed to remove pollutants that are present in solution and can be come easily clogged or damaged by sediment if this is not removed from effluent prior to through flow. Typically these systems are at the end of a series of treatment devices providing the final treatment prior to discharge. Effectiveness again varies depending on flow rates, concentrations of pollutants in effluents and maintenance intervals
Infiltration Systems	As with SuDS infiltration features these systems use water percolation through a series of aggregates or soils to remove pollutants. However unlike SuDS, PTS features do not all offer further infiltration to ground water. A system may consist of a concrete chamber filled with a graded aggregate or media, which filters runoff as it passes through. The surface of the chamber is left open to receive water and can often be planted with vegetation. An outlet at the base of the chamber directs discharge back into a conventional piped drainage system. Other systems contain both a perforated floor and piped outlet to allow discharge to conventional drainage only when high flows are experienced. Systems such as these can be retrofitted to existing urban infrastructure such as road gully pots where road drainage is forced through a media prior to its discharge. Linear drains that typically drain car parks or other impermeable open spaces can also be retrofitted with media in this way to offer pollutant removal and some water retention.

Compiled using Butler and Davis, 2011; Faram et al., 2004; Langeveld et al., 2012; M. Wilson et al., 2009

2.4.4 Benefits

The overall environmental benefits of SuDS and PTS already mentioned in the previous sections of this review are primarily water quality improvement and flood control and to this can be added associated economic and social benefits. The specifics of improvement to both water quality and flood control will be a subject of the specific site conditions, the quality (in terms of pollutant concentrations) and the quantity of water that a certain system deal with.

2.4.4.1 Water Quality Improvement

The potential for SuDS to improve water quality, i.e., to remove a particular pollutant of concern, or such systems treatment efficiency in general has in the past received much less attention than their role as flood control devices (Scholes *et al.*, 2008). However with the rise to prominence of water quality issues through the implementation of legislation such as the WFD (section 2.1.1), the quality of system effluents has become a much more important factor. The quantification and homogenisation of benefits from both SuDS and PTS is very difficult and it is unclear in the available literature due to the lack of comparable data.

Quantifying the performance of SuDS in terms of water quality is problematic, due to difficulties monitoring water flow in and out of the system, unquantifiable water losses through evaporation as well as take-up through plants and infiltration. In systems where water is retained or where a permanent water feature is present (e.g. pond or wetland) it is also very difficult to compare water inflow to water outflow as water may be retained in systems for days or weeks. In relation to infiltration, removal of water from a discharge cannot be considered to result in a corresponding rate of pollutant removal, as infiltrated water may well retain some pollutants and enter ground water or be discharged elsewhere when it meets an impermeable rock layer and runs along it. Bressy *et al.* (2014) compared a typical piped drainage catchment to three others that were served by a range of SuDS measures. The results showed that a reduction in pollutants correlated well to the reduction of mass discharges through loss to infiltration, but there was variation in results between sites, with some sites demonstrating greater drops in contaminant mass than water volume. However, the study concluded that no purifying effect was identified.

Other studies have focused on quantifying this 'purifying effect'. Laboratory tests (Charlesworth *et al.*, 2012) have shown that microbial communities in soils and composts can efficiently break down pollutants such a oils. Another study by SNIFFER (Scotland and

Northern Ireland Forum for Environmental Research) in 2008 found that there was an effective level of pollutant attenuation in soil based systems, with the vast majority of heavy metals, PAHs and petroleum hydrocarbons being retained in the top 10 cm of soil structure. This indicated that there was a low risk of pollution to ground water from highway runoff passing through a soil based SuDS.(Macualay Science & Consulting Ltd, 2008).

Conversely, PTS demonstrate a benefit in comparison with SuDS, in that they do not retain water for significant periods and loss of water to infiltration is reduced or not present. This facilitates monitoring and quantification of removal efficiency for a specific storm event. Several studies have examined the removal efficiency of different PTS in relation to a range of pollutants, the approach of these studies varies, some presenting data from scaled down systems monitored within a laboratory setting (Alkhaddar *et al.*, 2001; Phipps *et al.*, 2008).

Others have conducted controlled field tests, managing some of the key variables such as incoming discharge and pollutant load using a uniform material such as a silica-sand mixture to assess TSS removal with a range of different particle sizes. This approach was utilised by M. Wilson *et al.* (2009) in a study comparing six different hydrodynamic separators to treat stormwater. The results showed high variability for the removal efficiency of the devices which was dependant on influent discharge/ device height and diameter and the particle size of pollutants. The devices monitored were more effective at removing larger sized particles than smaller ones and the study concluded than none of the devices tested removed much of the smaller silt or clay particles. However, as Li *et al.* (2005) found that between 35-70% of the mass of solids in stormwater runoff from highways was under 100 µm this indicates that the efficiency of vortex separators to treat such road runoff is likely to fluctuate significantly. Cho and Sansalone (2013) undertook a similar study to observe the wash-out of a HDVS system, again with the focus being on the analysis of particle size rather than the overall efficiency of the system.

A smaller number of studies have conducted full scale field testing on systems retrofitted to existing infrastructure. Langeveld et al. (2012) tested three different filtration systems, including a Lamella and Sand and Soil Filter. This study monitored these measures for a 2 year period recording between 50-150 events at each of the filters. From the resultant data it was observed that the efficiency of filters fluctuated between 40-70% for TSS removal and 21-93% for heavy metals. The study attributed this range to the variety of flow rates (and associated pollutant loads) observed over the different intensities of events experienced in the study. However the results indicated that alone the filters were insufficient to suitably treat water to a quality where it could be discharged, because mean site concentrations exceeded the maximum allowable concentrations for effluent to receiving waters.

Overall these studies demonstrate the current “state of the art” in conducting monitoring, but also that there is a deficiency in the number studies where monitoring is conducted on systems retrofitted to existing infrastructure. In these circumstances conditions may often be very different from simulated lab or controlled field test, in terms of the hydraulic loading and pollutant volumes. Within a UK context this is certainly the case and the number of publish studies in relation to PTS performance is very small in comparison to studies assessing SuDS performance where the overwhelmingly the emphasis is placed on flood control benefit rather than monitoring of water quality output.

2.4.4.2 Flood Control

Existing long term catchment management strategies in the UK have exacerbated flooding problems (Swan, 2010). As with water quality improvements the flood control benefit of different systems varies significantly depending on the design of the system, the size and characteristics of the connected catchment area and the magnitude and frequency of rain and storm events that a catchment experiences. Benefit is either derived through the infiltration of water reducing total volumes or lowering peak flows through detention. As with water quality studies there is little comparable data available in the literature due to the high variation between installed systems and different site conditions.

Osborn *et al.* (2000) analysed rainfall changes between 1961–1995 from 110 UK weather stations on a seasonal basis. Their observations identified that over this period winter rainfall distributions had moved from a situation where there were higher contributions from lower and medium rainfall events to one where greater contributions came from heavy rainfall events at most locations in the country. Summer trends were the reverse of those in the winter with a decrease in the importance of heavy events and greater contribution coming from low and medium events. Trends in spring and autumn from some regions show the same behaviour as the overall winter trend whereas others show the opposite. These observations of trends in rainfall, demonstrate the need to ensure adequate retention and infiltration provision.

A study by Villarreal *et al.* (2004) reported the results of installation of a series of control measures (green roofs, storm water ponds and open channels) into an inner city suburb in Malmo Sweden. The experience from this study was that the storage volume of the system was the most significant factor in controlling storm event flows with ponds eliminating the vast majority of storm flow to the combined sewer system. As with water quality improvement Bressy *et al.* (2014) also observed significant reductions in discharge rates from catchments containing SuDS measures over the catchment containing only traditional drainage systems, with two out of three SuDS catchments monitored showing much less variation in discharge that that of the reference catchment with no SuDS features.

In Australia and the US PTS which incorporate high flows and no retention are being increasingly used and while they offer water quality improvements they offer little in the way of water retention or infiltration to contribute to flood control or pre development flow regimes (Burns *et al.*, 2012). From this it can be observed that it is important for PTS to be used in conjunction with SuDS to provide additional water quality improvements whilst flood control is also achieved.

2.4.5 Challenges and Barriers

There are a range of challenges and barriers to wider implementation of sustainable drainage including legal and regulatory problems with regard to which organisations have the responsibility and authority to require or implement structural measures (Stovin *et al.*, 2013). Modern cities are labyrinth's of small parcels of land owned by a multitude of different stakeholders and other interested parties, and this land ownership pattern in urban areas compounds the conflicts of interest over a whole catchment of a few square miles (Martin *et al.*, 2007). Gaining consensus from this multitude of different groups is a significant barrier when considering retrofit of SuDS or PTS. The existing piped drainage systems also represent a substantial investment made over many decades with established practice developed for well over a century. In theory more sustainable surface water management should be regulated by the planning and land use system, utilising building regulations and advice from the EA and other consulted parties, however in practice this existing system is not dealing with such issues effectively (White and Howe, 2004).

2.4.5.1 Financial Mechanisms and Incentives

The provision of a financial incentive or mechanism for structural measures is essential for greater uptake of urban BMPs has already been discussed (section 2.3.7). This has also been identified in other countries, such as Australia, as a major barrier to the provision of greater levels of structural BMPs in urban areas, as well as adoption of existing sustainable systems (Burns *et al.*, 2012). Evidence suggests that effluent and end user charges have the greatest potential for helping to pay for environmental improvement (Helmer and Hespanhol, 1997), however in relation to diffuse pollution this may not be practicable. J. B. Ellis (2013) suggested the introduction of an impermeable area tax to incentivise removal of such surfaces.

Other possibilities in the UK could be to re-direct portions of infrastructure budgets, such as the water company asset management plan funding. The asset management plan represents significant investment, for the larger companies amounting to several billion pounds, so that redirection of even small parts of this funding would allow significant investment in sustainable drainage practices. The service incentive mechanism operated by Ofwat (the water industry regulator) incentivises water companies to provide improved customer service by linking the score they achieve through the service incentive mechanism to the price they are allowed to charge customers. A similar mechanism is being trailed for reducing abstraction from sensitive waters (Fenn, 2012), and a comparable mechanism could be proposed to encourage water companies to reduce the volume of surface water runoff .

To minimise initial costs in the assessment of likely effectiveness of different control measures, it is important that water quality impact assessment should comprise simple desk study based methods, without the need for detailed hydraulic modelling (J. B. Ellis *et al.*, 2012). Although benefits are increasingly recognised the conflict between those who will stand to benefit the most (the broader public) and those who are expect to pay (local authorities and water utility customers) means greater implementation of structural measures remains a challenge (Stovin *et al.*, 2013). Ultimately, finding the mix of incentives and rewards is a difficult balancing act, meaning that contributions from all affected stakeholders will be required.

2.4.5.2 *Space constraints and Retrofitting*

Retrofitting of existing urban environments with structural measures that res-establish flow regimes back to those of pre-development conditions is very problematic. As already identified, the amount of land control measures require is a major issue in the redevelopment of existing built-up areas, meaning that a pragmatic mix of different measures from both SuDS and PTS will need to be adopted as well as retention of substantial parts of the conventional drainage system. This also means that preventative methods and source control can be the most feasible options due to the lack of land available for larger control measures (Jones and Macdonald, 2007). Perception is an important factor in retrofitting, for example the loss of courtyard areas outside properties was raised as a concern by some residents (Villarreal *et al.,* 2004). In some cases standing water is considered to be a hazard, and for some landowners and members of the public, it is preferential to convey stormwater quickly and safely to sewers, with the nearest watercourse being the most common final destination (White and Howe, 2004).

The emphasis placed on sustainable drainage within urban infill and regeneration schemes by local planning authorities is a key factor affecting uptake of sustainable drainage. A report commissioned by Thames Water as part of the consultation on the Tideway Tunnel identified that it would be feasible to disconnect 30-35% of existing impermeable land cover from surface and combined sewers in the inner London area (Ashley *et al.,* 2010). However the report also found insurmountable barriers around acquisition of land, as well as adoption and maintenance issues which would prevent the introduction of source control measures on a larger scale. J. B. Ellis (2013) suggests a range of examples where local authorities could feasibly insist upon the introduction of sustainable drainage with minimum cost or disruption:

- During the replacement or upgrade of existing impermeable surfaces or during utility replacement works.
- During the refurbishment of existing buildings or where infill schemes are proposed
- The offering of incentives to property owners to disconnect their roof or drive ways through introduction of an impermeable area tax
- Where drainage improvements are being undertaken anyway particularly on retail/ commercial premises or where combined sewer systems are running close to capacity.

In respect to retrofitting PTS there is little practical advice readily available, however as already explored they do offer a number of advantages over SuDS. Firstly, the smaller foot print means that even on particularly constrained sites PTS can be utilised, and the ability to be installed into existing underground piped systems or into surface channels or drainage infrastructure.

2.4.5.3 Maintenance and Adoption

Maintenance of systems is essential to good hydraulic and water quality performance, as previously identified (sections 2.4.2/3), however the requirement of maintenance is a disincentive for many stakeholders. Traditional piped drainage is the responsibility of water companies and in some cases local authorities, where maintenance is infrequent and tends to happen only where problems are identified.

Currently, to ensure sewerage undertakers will adopt infrastructure a developer must design and construct drainage in accordance with the adoption guidelines, which is the case for both foul and surface water (White and Howe, 2004). Thus in this situation it is implausible to expect land owners and other stakeholder to provide structural measures to manage surface water runoff and then take responsibility for them when currently they can pass responsibility for new piped systems to water companies for the insignificant cost of water rates.

Activity	Typical tasks	Indicative frequency
Routine/regular maintenance	Monthly (for normal care of SuDS)	• litter picking • grass cutting • inspection of inlets, outlets and control structures.
Occasional maintenance	Annually (dependent on the design)	• silt control around components • vegetation management around components • suction sweeping of permeable paving • silt removal from catchpits, soakaways and cellular storage.
Remedial maintenance	As required (tasks to repair problems due to damage or vandalism)	• inlet/outlet repair • erosion repairs • reinstatement of edgings • reinstatement following pollution • removal of silt build up.

Table 3 - Summary of System Maintenance (CIRIA, 2014).

The costs of maintenance of both SuDS and PTS vary and, as with the benefit gained from a system, costs fluctuate depending upon specific site conditions. The (Susdrain website) offers a useful summary of typical maintenance tasks shown in Table 3. In summary it can be seen that typical maintenance tasks are insignificant and would not present large costs and such actions could be easily adopted within the normal operating procedures used by most commercial and industrial facilities. More substantial maintenance would have greater costs but would only need to be completed at infrequent intervals. The often short-term view taken on drainage considerations means structural sustainable drainage measures are often rejected as being too expensive and disruptive (Swan, 2010) and due to these uncertainties their implementation on a larger scale has been slow.

As identified earlier in this report (section 2.1.3) the introduction of schedule 3 of the FWMA (2010) has been delayed and this is due to the difficulties in establishing which stakeholders should take responsibility for the long term maintenance of SuDS once they become a requirement for new developments. The adoption of existing retrofitted systems can be completed using existing legislation such as section 106 of the Town and Country Planning Act 1990 or section 38 of the Highways Act 1980 (CIRIA. *et al.*, 2007).

This section has laid out the principles behind sustainable stormwater management and summarised the main structural methods of surface water management in an urban context. It has also covered the benefits of their use and the barriers to their more widespread uptake.

2.5 Studies relevant to Urban Diffuse Pollution

This section explores a selection of the wide range of studies from around the world which have investigated various aspects of UDP. There is a mixture of physical data collection, new analysis of existing data and modelling, together with reviews of existing literature and policy. Much of the work into UDP builds on the existing knowledge around water management.Tsihrintzis and Hamid (1997) undertook an early review of the literature around the management and modelling urban stormwater runoff, identifying several areas in this field where further research was needed. While this paper is now fairly old, the recommendations it made are still relevant and supported by much of the more recent literature cited within this review of the literature. Tsihrintzis and Hamid (1997) made four primary recommendations to develop understanding, which are:

- Characterise the effects of urban runoff sediments of varying particle sizes on receiving waters, specifically exploring the varying physical properties of sediments arising from typical land uses;
- Assess the effectiveness of different structural BMPs (SuDS) in field conditions to control specific pollutants and report results in a consistent and comparable format;
- Conduct studies within small experimental watersheds of one predominant land use with a small receiving body which can be easily accessed to monitor changes;
- Use data from these studies to improve effectiveness and accuracy of models, whilst also improving selection of input parameters.

These objectives that are only outlined here are still very important and relevant in the field today, and while it was focused on runoff from car parks, a more recent extensive review by Revitt *et al.* (2014) into suitable treatment methods for this runoff gave similar conclusions, including :-

- There is still a need for greater levels of data collection on pollutant runoff concentrations:
- More data on performance of stormwater SuDS/BMPs and proprietary products is needed;
- Further study of importance of finer particulate sizes (<75 microns) in transport of surface water pollutants.

Other desk based studies have focused on the need to accelerate the uptake of SuDS measures. For example J. B. Ellis *et al.* (2012) developed a simple methodology to assess surface water runoff quality following SuDS treatment. The authors argue that, in order to encourage a greater uptake of sustainable drainage practices, there is a need to develop a simple and transparent impact assessment methodology to assess surface water runoff quality. The developed tool produces an overall site pollution index using three indices:

- An impermeable runoff factor based on a GIS land use layer to estimate runoff;
- A pollution index based on a large volume of existing stormwater quality data from existing studies which is cross referenced with the WFD EQS's;
- A pollution mitigation index, this assigns a value to individual SuDS features based on their ability to remove/ retain various pollutants from runoff.

The tool essentially assesses areas of land and based on their use, and predicts typical runoff and pollutant volume. It then uses values assigned to SuDS components to suggest the most appropriate components (swales, ponds, etc.) to manage the sites stormwater runoff. While such a tool provides a good basis for selection of SuDS features for any particular site, it cannot account for site specific details which often preclude the use of certain features. It also fails to account for other criteria also important to the overall design such as cost and existing site topography. There are several other papers which have developed similar approaches with Scholes *et al.* (2008) developing a similar tool for a comparative assessment of the potential for SuDS to remove different pollutants. J. B. Ellis (2013) also produced a review of the potential for the retrofit of SuDS and other green infrastructure in the UK identifying the opportunities and associated benefits.

As well as novel approaches such as this it is also important to consider how other sectors have addressed diffuse pollution, particularly agriculture as all catchments will be affected by this land use. Novotny (1999) provided an important global outlook in respect of diffuse pollution from agriculture, identifying how changes in farming practices, such as intensification and increased use of inorganic fertilisers and chemicals, have resulted in significant degradation of receiving waters around the world. This paper identifies the need for structural BMPs (SuDS equivalent) to be adopted on a large scale to help to reduce pollutant runoff from farm land.

A further point raised by the paper is that in developing countries diffuse pollution is not considered to be a priority with respect to water pollution. In such countries point source pollution, which in most developed nations is already under tighter regulation and better control, is still a considerably greater source of pollution and damage to receiving waters. A more detailed review of diffuse pollution from agriculture and what aspects of it need to be applied in an urban context is given in Appendix I.

2.5.1 Catchment Focused Studies

Examination of research into the overall environmental benefits of SuDS and PTS shows that work tends to be conducted at two contrasting scales: either the whole catchment scale where a wider area is monitored or at a site scale where a group or single system is monitored more rigorously. Work monitoring catchments provides an important part of our understanding of UDP because it allows a greater insight into how existing drainage and water systems interact and helps identify the most significant challenges with respect to water quality in different areas.

Bressy et al. (2014) compared a typical piped drainage catchment to three others that were served by a range of SuDS measures all located in Northern France. A detailed investigation of the catchments was undertaken, along with monitoring of rainfall and discharge. Water and soils were also sampled. Results showed that a reduction in pollutants correlated well to the reduction of mass discharges through loss to infiltration, but there was variation in results between sites, with some sites demonstrating greater reductions in contaminant mass than in water volume.

This study concluded that rather than focusing only on large events which lead to the greatest risk of flood damage there is an increased opportunity to reduce mass discharges and subsequently pollutant discharge from smaller more frequent rainfall events, through increased levels of infiltration. However it cannot be assumed that just because water is lost to infiltration that all pollutants are captured in soils and surface layers. Depending on the particle size of pollutants and whether they are held in solution, there is risk of contamination of ground water in sensitive areas. Indeed the study does identify that the reduction in pollutants is explained primarily through a reduction in runoff volumes, and does not demonstrate that results reveal any kind of “purifying effect” in the classical meaning (i.e. lower concentrations).

It is also important to appreciate that the varying climatic conditions between countries means that actions seen as desirable in one country maybe inappropriate in another. Burns *et al.* (2012) detail a study from Melbourne where a comparison of the hydrologic effects of two alternative conventional approaches to urban stormwater management was made. These were a drainage-efficiency (no retention or treatment) focused and a pollutant-load-reduction (under drained bio filtration system) focused approach. The various disadvantages of these approaches were identified, and their hydrological outcomes contrasted with a more progressive method which focused on restoring a natural flow regime (combined rainwater harvesting and vegetated infiltration system). This study identified that both the conventional approaches failed to sufficiently retain storm water and disrupted river flows and highlights the need for water retention features to be used in conjunction with pollution control measures. Whist important, these findings are less relevant to those countries with a higher average rainfall, where stormwater runoff will form a smaller proportion of river base flow.

Other studies focus on the potential of the existing drainage system to contribute to pollutant loading of receiving waters. For example G. Kim *et al.* (2007) who completed a study in Daejeon City in South Korea. This examined the volumes and quality of the effluent discharging from CSOs in the city. The study monitored a single CSO discharge from a catchment area of 136 ha, over 5 rainfall events. Only 3 of these events were of sufficient size to cause CSO discharge which was sampled using auto sampling equipment, with further manual samples being collected during very high discharges. Results observed show that only an average of 10mm of rainfall was required for CSO discharge to occur, resulting in very poor quality discharge with high levels of solids, organics and nutrients observed in samples. It also indicated that by attenuating the equivalent of 5mm of rainfall the pollutant loading from the CSO to receiving water could be reduced by 80%.

While this study provides useful data into the quality of discharge effluent of CSOs in the region, it is likely that due to the limited number of storm events and monitoring points used that further data collection from other locations and from a greater number of storm events may result in an a different conclusion.

Some studies have investigated supplementing long term best practice approaches with sort term interventionist actions in order to quickly deliver water quality benefits. Özkundakci et al. (2010) reports the results of a 5 year project to restore Lake Okaro in New Zealand, which had become eutrophic, to return it to more oligotrophic conditions. To achieve this reversal a 2.3 ha purpose made wetland was construction and protection of riparian margins through measures such as livestock exclusion, fencing and planting of native plant species along the stream banks and lake margins. These measures to reduce external loading of Phosphorous was complimented by application of alum and modified zeolite chemicals to absorb Phosphate from the lake, to reduce internal loading.

As a result of these measures the total phosphorus concentration in the lake decreased by 41%. This is a good example and represents a more intensive approach addressing not only the external diffuse nutrient pollution inputs of a water body but treatment of that water body to address the existing pollution issues. The use of geoengineering techniques such as the dosing of the lake with Alum and zeolite is less common than the control of external loading with riparian management, etc. However the use of such a technique is less applicable to rivers due to the retention time of water. Also the study does not give specific details in relation to the application of the zeolite. Was it applied in a solid form meaning that subsequently it needed to be recovered once sufficient time had pass to allow absorption of phosphates?

A number of studies have also investigated the ‘first flush’ phenomenon, which is closely linked to understanding the importance of particle size of pollutants. J. H. Lee *et al.* (2002) undertook a study of 13 separate urban catchments in Chongju in South Korea, with the aim of observing the ‘first flush’ phenomenon. In this study the ‘first flush’ was defined as being “the occurrence of high concentrations of pollutants in the early stages of a storm event”. Thirty eight separate storm events were monitored and based on the ratio of runoff volume to pollutant mass over the duration of events it was considered that a ‘first flush’ was observed. The strength of the effect varied for different variables over different catchments with a variety of predominant land uses, for example TSS loading was high from residential areas and conversely COD was lower from catchments with greater industrial coverage. The ratio was also found to be influenced by the percentage of impervious surface area within a watershed and was observed to be more pronounced in smaller watersheds.

While the overall number of storm events monitored in the study was large, events were split over 13 separate watersheds meaning that only between 2 to 3 events were recorded at each. As observed in section 2.3.5 it also needs to be considered that there is no consensus in the literature in relation to a clear definition nor standard method to assess the strength of the 'first flush' (Deletic, 1998). J. H. Lee *et al.* (2002) further observed that through the use of different methods of analysis the strength of the observed 'first flush' varied.

It is important to consider the importance of particle size when considering the rate of nutrient wash off in urban areas. (Miguntanna et al., 2013) conducted a study in Southport Australia, which examined the importance of nutrient wash-off with respect stormwater runoff. Using small uniform 3m^2 plots located in three different urban districts where the land use was predominately residential, industrial and commercial respectively. The available pollutant load at each site was determined by vacuuming a plot of equivalent size at each site so it could be factored into results. Six plots at each site were then subject to simulations of 6 different rainfall intensities of differing duration. Samples of runoff from plots were collected and tested for a range of Nitrogen and Phosphorous indicators as well as examined for particle size.

It is not clear from the paper how samples were collected, i.e. manually or using automation. It is also unclear what the antecedent period was prior to the rainfall simulation as this would likely have affected results. It was found that Nitrogen and Phosphorous displayed different behaviour in response to the simulated rainfall, it was the quantity of Nitrogen wash-off was limited to qualities in the initial pollutant load whereas wash-off quantities of Phosphorous was limited by the availability of runoff to transport it. Nitrogen was detected in higher dissolved quantities that Phosphorous, indicating that was more readily removed by lower intensity rainfall. In particulate form Nitrogen was predominately seen in the small fractions less than 150 microns, whereas Phosphorous was observed in similar quantities at particles sizes smaller and larger than 150 microns.

Chiew and McMahon (1999) used a straight forward modelling approach to estimate runoff and diffuse pollution loads for urban catchments in Australia. The study found that the key variable for estimating annual runoff was the fraction of effective impervious area within the catchment. Using a simple runoff rainfall plot, the angle of the slope of a best fit line gave a good approximation of the impervious area. As illustrated in the sample plot from the study, included as Figure 7.

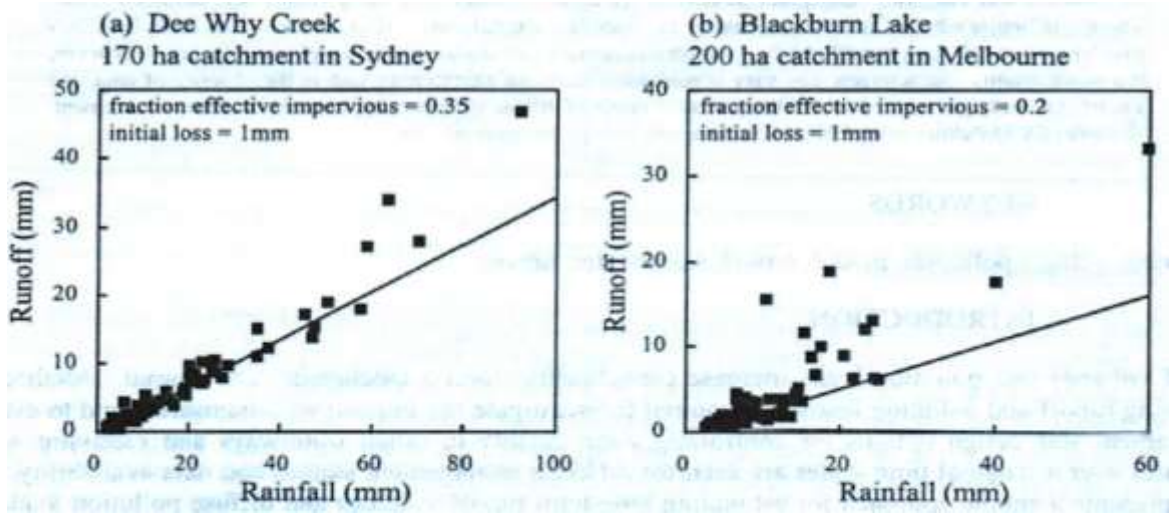


Figure 7 - Sample plot from Chiew and McMahon (1999)

To calculate the annual pollutant volumes the Event Mean Concentration (EMC) of known storm events was multiplied by runoff. In the absence of primary data on EMC's for the studies' catchments researchers used data from published literature. Using these inputs it was found the resulting model could predict with reasonable accuracy the expected long term runoff from a catchment. However in the absence of local data on water quality the estimations of pollutant load were less accurate. Liu *et al.* (2013) who conducted a similar study into the influence of rainfall and catchment characteristics on stormwater quality, reported similar findings. They found that complex response of urban storm water quality to rainfall may not be adequately represented by the limited number of factors used in modelling to make accurate site specific predictions.

When considering modelling work such as this it is essential to have accurate rainfall data in relation to studies of catchments but also to appreciate long term rainfall trends. Osborn *et al.* (2000) analysed rainfall changes between 1961–1995 from 110 UK weather stations on a seasonal basis. Their observations identified that over this period, for most locations in the country, winter rainfall distributions had moved from a situation where there were higher contributions from lower and medium rainfall events to one where greater contributions came from heavy rainfall events. Conversely summer trends were the opposite, with a decrease in the importance of heavy events and greater contribution coming from low and medium events. Trends in spring and autumn from some regions show the same behaviour as the overall winter trend whereas others show the opposite. These observations of trends in rainfall demonstrate the need to ensure adequate retention and infiltration provision.

2.5.2 SuDS Focused Studies

Studies addressing a large catchment scale are subject to much greater error due to the very large number of variables affecting results, and so it is also important to undertake studies at a smaller scale where the range of variables can be controlled more easily. Individual SuDS have been demonstrated to improve the water quality of runoff, i.e., to remove a particular pollutant of concern, but in the past this aspect of SuDS has received much less attention than their role as flood control devices (Scholes et al., 2008). However with the rise to prominence of water quality issues through the implementation of legislation such as the WFD (section 2.1.1), the quality of system effluents has become a much more important factor.

Villarreal et al. (2004) reported the results of installation of a series of control measures (green roofs, storm water ponds and open channels) which were opportunistically retrofitted during the renovation of an existing development located in an inner city suburb in Malmö Sweden. The main aim of the work was to reduce flood risk and improve the water quality of receiving waters by preventing CSO spillage, however no flow monitoring of the site was completed and researchers relied upon demonstrating the benefits of the scheme through modelling using synthetic hydrographs and flow routing routines. This design approach was favoured as it was less invasive than the works that would have been required for the implementation of a conventional separate sewerage system.

The experience from this study was that the storage volume of the system was the most significant factor in controlling storm event flows with ponds eliminating the vast majority of storm flow to the combined sewer system, but it is unclear if this was a direct observation or a conclusion drawn from the conducted modelling. The chief benefit in terms of water quality improvement reported was the potential reductions in surface water runoff, through the increased attenuation provided by ponds. This improved the capacity of the combined system to which the area eventually discharged, which in turn gave greater protection against spillages from CSOs. No direct water quality analysis of runoff leaving the system to the sewer was undertaken, making it difficult to assess the capacity of the system to deliver pollution control.

Haycock and Muscutt (1995) completed a review paper on the importance of buffer strips in controlling diffuse pollution from agricultural runoff. This identified that buffer strips can have an important role to play in respect of pollutant removal from runoff providing various benefits including enhanced biodiversity and amenity as well as flood control and better river bank stability (where farm land is adjacent to river banks). The paper presents a summary of the specified sizes of buffer strips in the literature, listing large dimensions of 100's of metres whereas in respect to water quality treatment more recent guidance on the use of buffer strips in SuDS specifies that widths of between 5-7 meters are sufficient to achieve treatment (Lampe, 2004). While larger strips provide further benefit and are more appropriate in rural areas, in urban areas where land availability is reduced the proximity of strips to busy road and car parks may limit their value as amenity space.

Often laboratory testing can be helpful in developing guidance figures to consider in respect of SuDS. A series of laboratory tests undertaken by Charlesworth et al. (2012) investigated if the use of green waste or food composts could be used as a replacement for normal topsoil in SuDS features such as swales. Using leaching columns, an investigation of microbiological development within the composts and normal topsoil was undertaken, with oils and dusts collected from street sweepings added, to simulate pollutant loads entering a SuDS system and rainfall simulated at 15mm per hour. In this experiment application was made at single or bi-fortnightly intervals, which simulates spillages or intense rainfall where large amounts of pollutants are washed into a system; however this does not simulate as effectively lower intensity storms where pollutants may enter a system more gradually. Also pollutants such as some heavy metals enter a SuDS system already in solution, so again this would not be simulated with spate application of pollutants and rainfall used in this experiment.

It was found that many of the differences in performance between mixed waste and green waste compost were insignificant, although both performed better in terms of pollutant retention than standard top soils. The superior levels of microbial communities in compost means they can more efficiently break down organic pollutants such as oils than normal top soils. This demonstrated that there may be good potential to use composts in SuDS using material that may be considered waste and avoiding the need for disposal to landfill. The study identifies that these laboratory findings need to be corroborated through field testing.

2.5.3 PTS Focused Studies

The principles of PTS are outlined in section 2.4.3, and as identified they are normally specifically designed to address a single function, i.e. water quality or flood control. The studies into PTS that seek to affect water quality benefit are primarily focused on two main areas: firstly, demonstrating the removal potential of systems on an event basis and secondly, examining the importance of particle size and how effectively systems retain different fractions of particle size.

Several studies have examined the removal efficiency of different PTS in relation to a range of pollutants, using a variety of different approaches. Some, such as (Phipps *et al.*, 2008) undertook testing on a scaled down system monitored within a laboratory setting. The researchers constructed a model of a commercially available HDVS and the study examined the potential of the device to capture and store TSS in a lower sump which was physically connected to the main chamber but hydraulically disconnected preventing re-entrainment of particular matter from subsequent events. Tests were conducted to assess the residence time of the system by injecting a dye mixture and monitoring water for colour change at several points in the system under various rates of discharge. Monitoring of TSS removal efficiency was also undertaken using granulated active carbon sediment, this mixture was introduced to the system at the same point as the dye. A fine mesh filter bag sited 1m downstream of the outlet pipe was used to trap material not removed by the system.

It is unclear if the material gathered in the sump was quantified. This is important as this volume of sediment, not the wash out material at the end of the system, should be considered the 'removed pollutant volume' as material not captured and stored in this section of the system is subject to re-suspension and wash out by successive events. The study found that the volume of material removed by the HDVS correlated well with the rate of discharge through the system, with efficiency reducing with increased discharge rates reflecting the lower retention time.

Other approaches have under taken field tests under controlled conditions. In St Pauls, Minneapolis USA (M. Wilson *et al.*, 2009) completed such a test on 6 different HDVS devices. The HDVS monitored were located in six different locations and catchment conditions, however all variables were controlled to remove the effect of different sites on results. Each device was tested 3 times at 4 different rates of discharge; controlled using a discharge from a fire hydrant with rates being set between 15-100% of the HDVS treatable discharge rate. As all devices monitored were located in active storm water systems proprietary bungs were used to seal off existing flow paths into devices. Between 10-15kg of a sand silica mixture with a known particle variation in terms of its particle size fractions, was input into the system at a constant concentration of 200mg/l. Input was between the influent from the hydrant and inlet to respective devices. Following event simulation, 15-20 minutes was allowed for particles to settle and following this the system was dewatered and settled solids from the sump section were removed before being dried, weighed and the different fractions of particle sizes measured.

The results showed high variability for the removal efficiency of the devices which was dependant on influent discharge/device height and diameter and the particle size of pollutants. The devices monitored were more effective at removing larger sized particles than smaller ones and the study concluded than none of the devices tested removed much of the smaller silt or clay particles. While the key focus of the study was to assess the different removal rates of particle size it is considered that the simulation of events by the study was not in line with those experienced in real world conditions. For example a consistent discharge rate and pollutant loading was utilised where in reality through piped storm water system these variables fluctuate rapidly. Therefore it is likely that results give a more favourable value for removal efficiencies and volumes of smaller particle sizes captured.

A smaller number of studies have conducted full scale field testing on systems retrofitted to existing infrastructure. (Langeveld *et al.* (2012)) tested three different filtration systems, including a Lamella and Sand and Soil Filter. This study monitored these measures for a 2 year period recording between 50-150 events at each of the filters. While the exact locations of monitoring equipment in respect to flow measurement and sample collection varied slightly between filters due to differences in their design, the configuration of monitoring was the same at each site. Discharge was monitored by a flow meter at the piped inlet to each system with samples taken upstream and downstream of the respective treatment component.

From the resultant data it was observed that the efficiency of filters fluctuated between 40-70% for TSS removal and 21-93% for heavy metals. The study attributed this range to the variety of flow rates (and associated pollutant loads) observed over the different intensities of events experienced in the study. However the results indicated that alone the filters were insufficient to suitably treat water to a quality where it could be discharged, because mean site concentrations exceeded the maximum allowable concentrations for effluent to receiving waters. There is a notable difference between the function of monitored filters, the lamella filter was an online feature in that all discharge passed through it up to its maximum treatment rate, whereas the soil and sand filters were offline with effluent being pumped from the main drain into the system.

These findings with respect to particle size are confirmed by (Cho and Sansalone (2013)) who undertook a study to observe the wash-out of a HDVS system, with the focus being on the analysis of particle size rather than the overall efficiency of the system. They demonstrated that the ability of HDVS to retain TSS reduced with smaller particle sizes. Based on these studies it seems evident that the suitability of HDVS to treat storm water flows is very much dependant on which fraction of particulate matter is most significant in respect to the volume of pollutants it carries to receiving waters. Li *et al.* (2005) found that between 35-70% of the mass of solids in storm water runoff from highways was less than 100 µm which indicates that the efficiency of vortex separators to treat such road runoff is likely to fluctuate significantly.

Overall these studies demonstrate the current “state of the art” shown in the literature around the understanding of UDP and its amelioration, but also that there is a deficiency in the number studies where monitoring is conducted on systems retrofitted to existing infrastructure. In these circumstances conditions may often be very different from simulated laboratory or controlled field tests, in terms of the hydraulic loading and pollutant volumes. Within a UK context this is certainly the case and the number of published studies in relation to PTS performance is very small in comparison to studies assessing SuDS performance where the overwhelming emphasis is placed on flood control benefits rather than monitoring of water quality output. This is contrary to the historic SuDS philosophy which gives equal weighting to flood control, water quality and biodiversity/amenity.

2.6 Literature Review - Summary

This review has addressed the research literature with direct relevance to urban diffuse pollution, and confirms it as a multidisciplinary problem. It requires a clear understanding of several different areas including, river chemistry and ecology, water sampling methodology, the physical mechanics of surface water wash off and sediment transport as well as knowledge of the legislation and policy in relation to water quality and flooding. Therefore this review has collated and analysed a wide series of research studies from related sectors to produce a thorough and holistic view of the subject area. The review has identified a series of gaps in existing knowledge and highlighted the problems with current approaches to the subject. They include:-

- Inadequacies of existing sampling regimes to identify diffuse pollution,
    - Covered in sections 2.1.1.3 and 2.3.7
- Difficulties around identifying the contribution of diffuse sources,
    - Covered in section 2.3
- A lack of data on the water quality performance of SuDS and PTS,
    - Covered in section 2.5 and 2.4.4
- A lack of consensus on the significance of the first flush effect,
    - Covered in section 2.5 and 2.3.5
- Significant barriers to the retrofit of SuDS and PTS in urban environments.
    - Covered in section 2.4.5

The existing EQS method of sampling (explained in section 2.1.1.3) is unsuitable for the identification of the contribution of diffuse pollution to river water quality considering that:-

- the highly variable and unpredictable nature of pollution sources,
- that sources are not easily traced,
- toxic components are not well defined and
- there are no EQS values for the build-up of contaminates within sediments.

Existing legislation still fails to take account of the episodic nature of the problems surrounding urban diffuse pollution sources and the fact that current sampling regimes will often fail to identify the variable nature of the degradation in water quality from these sources (J. B. Ellis *et al.*, 2002). This also questions the outcomes of many of the studies based on this existing water quality data which is routinely collected by the EA for the purpose of assessing water-bodies in relation to WFD compliance. This points towards the need for sampling regimes with much greater density of sample location and frequency and especially conducted during storm events so that more accurate identification of the contribution of diffuse pollutants can be made.

This review has also identified the paucity of data available on the water quality performance of different SuDS and PTS. Considerably more emphasis and associated funding is placed on flood risk management which contrasts with comparatively little attention or priority given to the water quality risks of impermeable surface runoff (J. B. Ellis and Revitt, 2010). The lack of consensus on the significance of the first flush in terms of its relationship with the volume of pollution delivered to a river system from a storm event is another key finding from the review. The narrowness of the current data sets indicates the need to collate much more data on the treatment of storm water discharges while also considering the importance of the ‘first flush’ effect which collectively would provide further insights to this problem. In addition the difficulties surrounding the retrofit of SuDS and PTS to treat existing untreated storm water discharges needs to be considered in this context.

Chapter 3 – Methodology

Chapter 3 – Methodology Phase 1

3.1 Materials and Methods

This Chapter outlines the experimental methodology for this study which was undertaken in stages as shown below:

- Project Background and Development
- Phase 1 - River Water Quality Sampling and Assessment
    - Develop and Complete Sampling Regime
    - Analysis of River Sample Data

- Phase 2 - Treatment System Monitoring and Sampling
    - Selection of Treatment Systems and Locations
    - Monitoring of Treatment Systems
    - Analysis of Monitoring Data

Each section describes a number of different processes which were integral to the completion of the overall project. Figure 8 is a flow chart showing the whole process chronologically. Each task is dealt with separately in the methodology, an explanation is given about tasks and why the approach selected was used. How each task was undertaken is then detailed, and finally the predicted outcomes of each objective are explored. The project background and development of the sampling regime are described in this Chapter (phase 1) and the analysis of river sample data follows in Chapter 4. Selection of treatment systems, their locations and how they were monitored is covered in the second part of this Chapter (phase 2) and the second part of Chapter 4 presents the analysis of the results of monitoring the effectiveness of the treatment systems after they were installed.

Typically researchers utilise either quantitative or qualitative research work, however some researchers have combined one or more research methods in the same study. This project has undertaken the collection and collation of both qualitative and quantitative data. Originally developed through the study of natural phenomena in natural sciences, quantitative methods are now also widely used in social sciences. They include activities such as surveys, laboratory experiments, formal methods and numerical methods such as mathematical modelling. By contrast, qualitative research methods were developed, to enable researchers to study social and cultural phenomena in the social sciences (Myers, 1997).

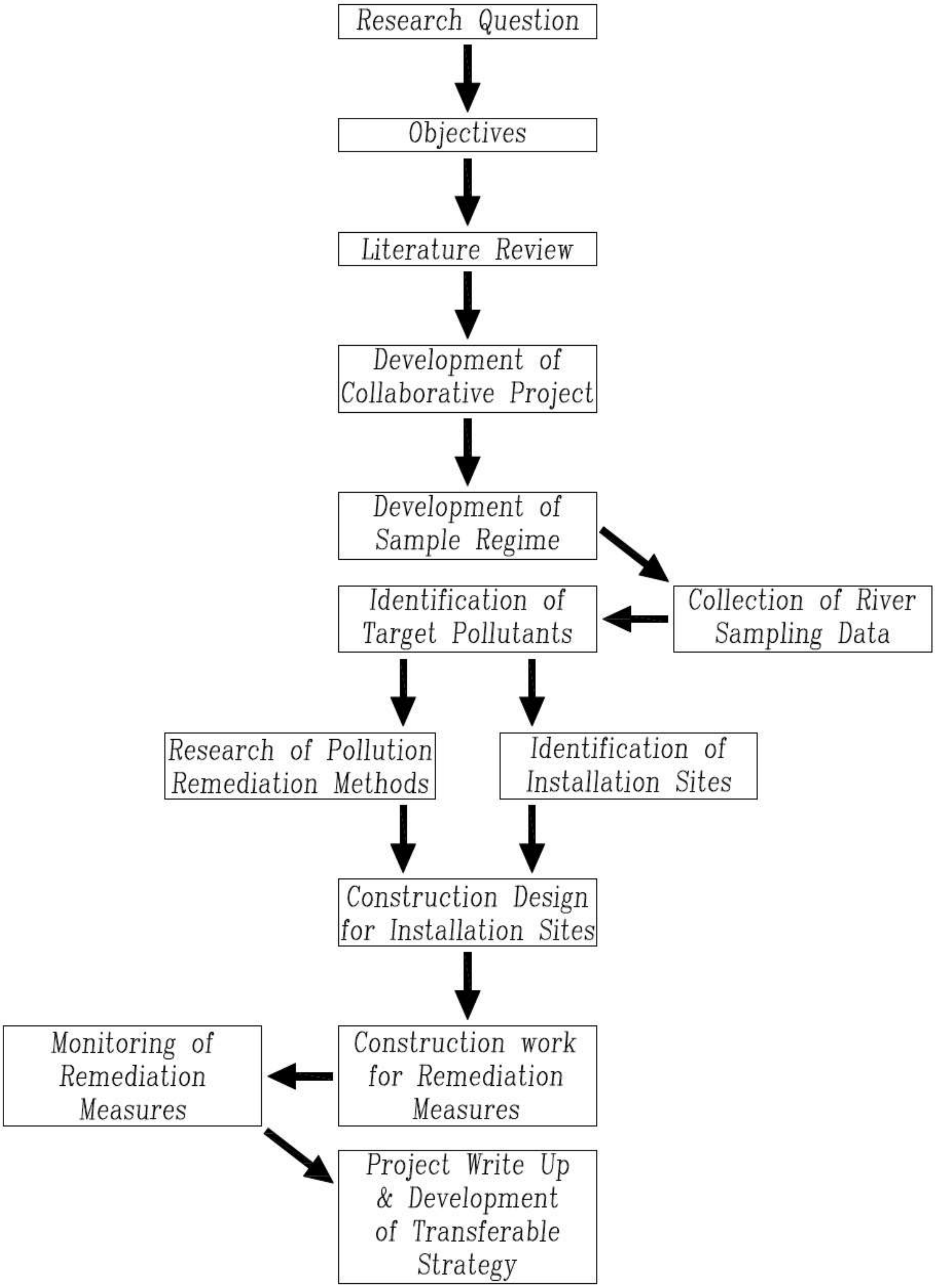


Figure 8 - Research Methodology Flow Chart

3.2 Project Background and Development

From the outset it was clear from the literature, that there were a number of areas where new research and data collection was needed to further develop the understanding surrounding the problem of river water quality, specifically around diffuse pollution. The focus and scope of this work were developed jointly by University of Salford and the EA. Initially the interest was wide, encompassing catchment management and diffuse pollution, however this was gradually refined to focus more specifically on UDP and limited to the following:-

- The selection of Wigan as a suitable and representative study area
- Development of a sampling programme to quantify the contribution of urban diffuse pollutants to the Wigan urban catchment.
- Collection of samples to develop a detailed data set.
- Analysis of sample data to quantify the contribution of urban areas to river pollution, and where possible identify:
    - specific pollutants i.e. heavy metals, PAHs, etc.;
    - pollutant sources, i.e. polluting surface water drains (SWD) contributing areas, road gullies, etc.;
- Investigate the potential of mitigating identified pollution sources.

The extent of the work undertaken required the support of the EA to aid fieldwork, supply sampling equipment and to undertake laboratory analysis. The project's scope was to later increase, as it became clear that it was feasible to install a series of structural mitigation measures. To do this, approval was sought from the EA for additional funding for the purchase of PTS, SuDS and any other measures used, as well as the associated design and construction work required for installation.

3.3 Identification of Study Area

The primary goal of this work is to identify and quantify UDP, but also, to explore the effectiveness of different methods to treat sources of UDP. While selecting a suitable study area in which to conduct this investigation, background data and information was collated to enable the most informed decision on an appropriate area. Due to the limitation of resources, only a section of a river catchment could be considered as a study area. The feasibility of primary data collection and the work required to implement remediation of pollution sources also had to be considered, as well as the potential benefits from identification and remediation of pollution sources.

The EA national water quality data base is used to classify water bodies on a chemical basis. Locations within the North West failing chemically were identified, and considered as potential study sites. The failure of coastal RBWD sites is also important, as river FIO inputs can contribute to this. Historical studies of water quality monitoring in the region, also highlighted catchment areas where on-going water quality issues remain unresolved. The availability of supporting data sets for potential study areas was also considered. The EA maintains a national rain gauge and hydrometry station network, so access and proximity to these data streams was also desirable in the selected study area.

As well as examining existing data sets and literature, there were also a number of practical considerations and criteria that the selected area had to display in order to make it feasible to undertake work. These were:-

- An urban environment which was representative of a typical UK town, so that outcomes could be applied and be relevant to other similar areas.
- A catchment size small enough so it that could be studied using available resources, it was desired to have a high density of sample points which would not be possible over a large study area.
- To contain a significant portion of urban land cover, as it was primarily urban diffuse pollution that was the focus of this study.
- To contain a river network, where urban runoff is significant enough to allow the contribution of surface wash off to be observed on the river network.
- To be located within the North West so it was readily accessible. A purely practical consideration to facilitate the collection of samples.

Using these criteria, the upper River Douglas catchment was identified. The catchment drains an extensive area, where intensive agriculture and horticulture dominate the land use. Wigan and Skelmersdale urban areas also constitute a significant section of the catchment, and the study site was centred around this area, focusing on a 16 kilometre section of the river Douglas directly up and downstream of the Wigan urban area. Figure 9 shows the extent of the study area. Draining a dense urban area, the section of the river flowing through Wigan has a range of pollution issues, illustrated by its failing WFD status. The good local knowledge of EA staff was helpful in conducting the river monitoring, and in identifying existing known pollution issues.

3.4 River Sampling Regime

The primary goal of sampling was to identify the pollutants which are prevalent in the river under different conditions. Other studies examining riverine water quality (Davies and Neal, 2007; Neal *et al.*, 2011; Stapleton *et al.*, 2008) that used a sampling regime to collect primary water quality data, have overwhelmingly used macro level sampling, or they have used existing EA water quality data.

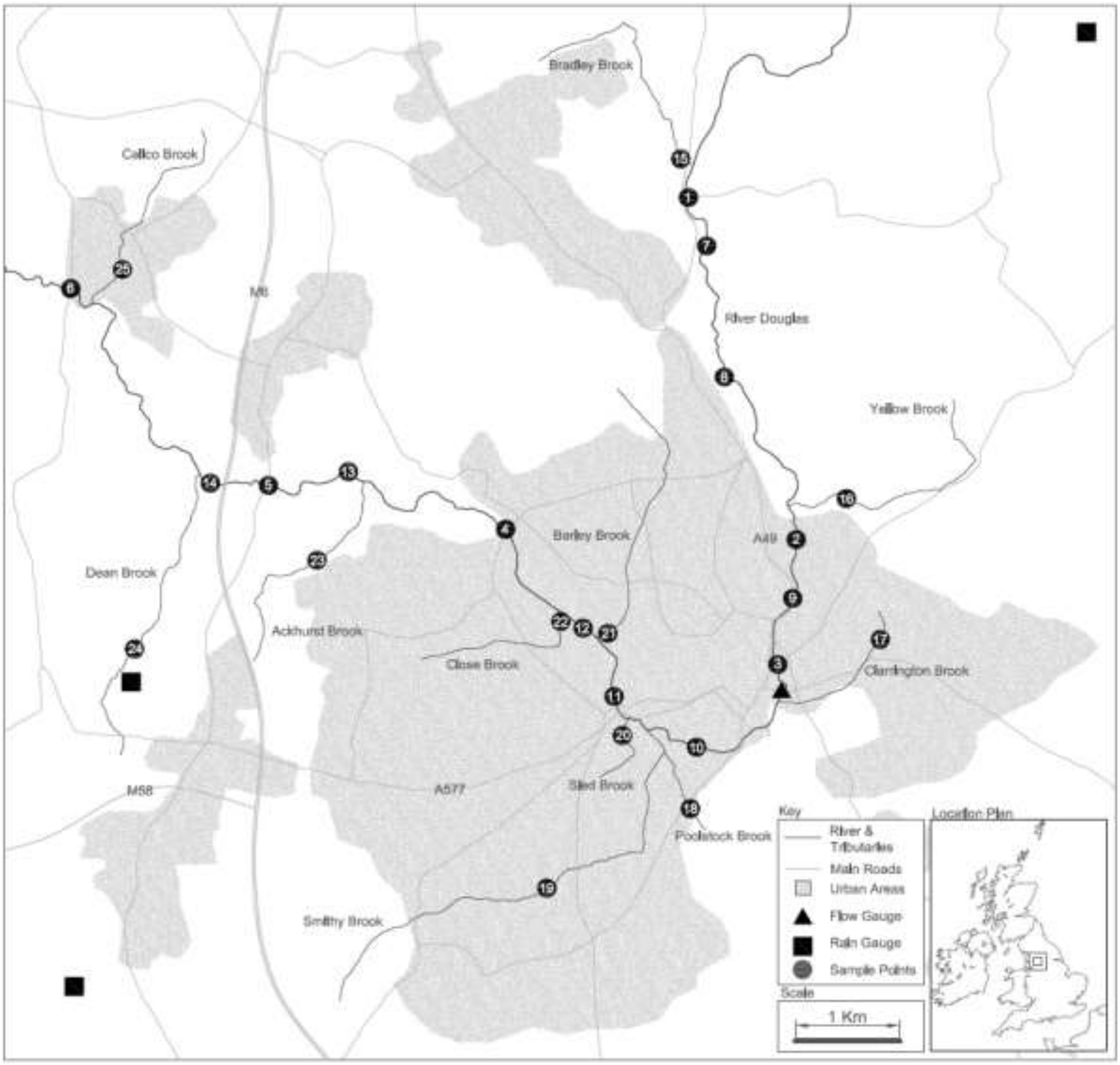


Figure 9 - Study Area and Final Sample Locations

The land use for the whole catchment was identified from the 2007 Land Cover Map (CEH, 2007) Figure 10. The overall area of the catchment considered in the study, from the farthest downstream sample location, was 190 km^2, also shown on Figure 10.

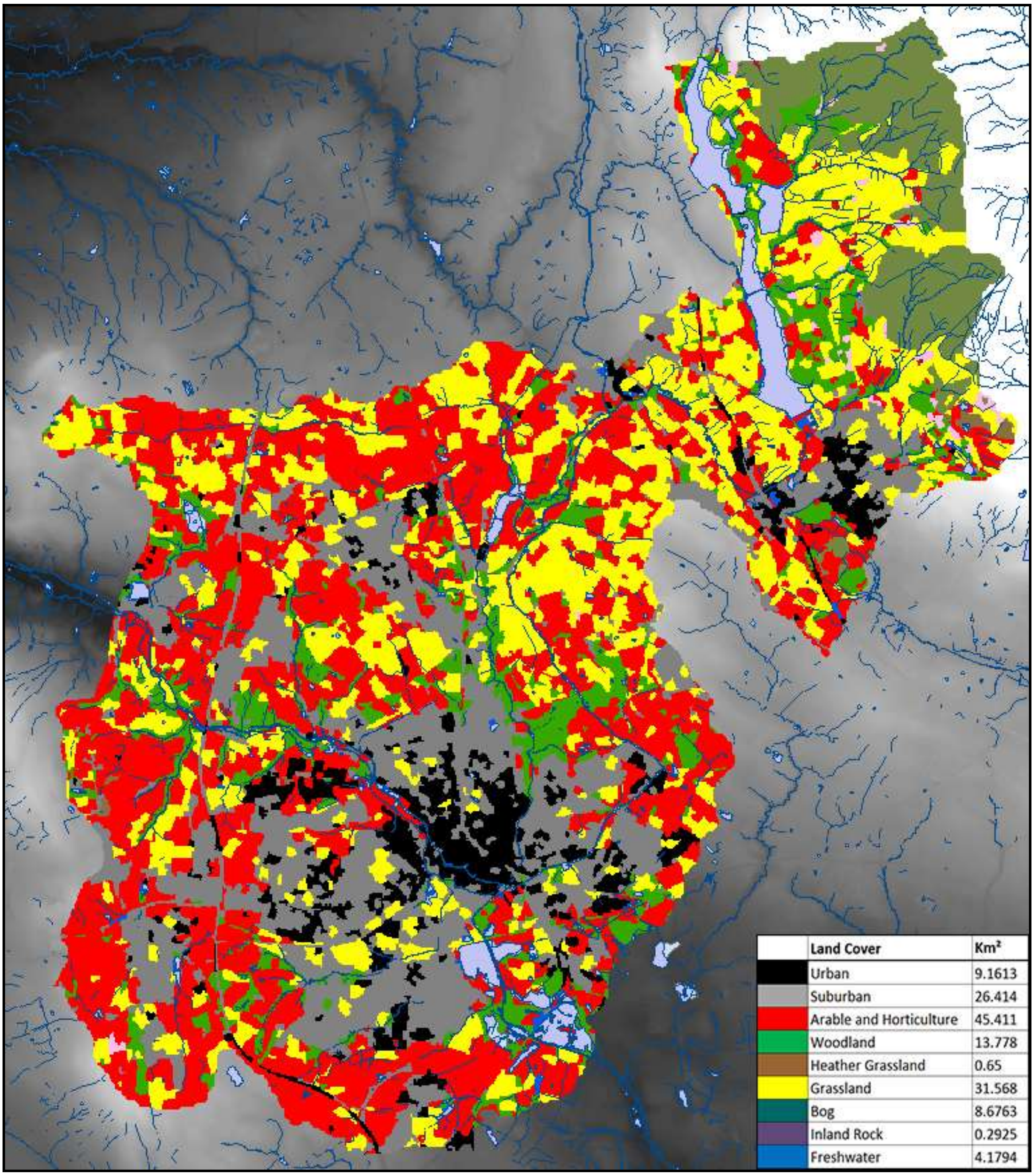


Figure 10 - Land use across study catchment (CEH.,2007)

The selection of sampling locations was influenced by several criteria. Firstly, it was preferred to use existing EA sampling locations, as this facilitated a comparison with historical data. Unfortunately there was only a single existing sample location in the study area, meaning multiple new locations needed to be selected. Secondly, new sample point locations were limited by the physical access to the river and its tributaries, particularly in central Wigan, where there are several culverted sections which made gaining safe access difficult.

Initial sample locations were identified to provide a holistic coverage of all sections of the river course passing through the Wigan urban area, including all major tributaries. Suitable siting was not always possible, due to both the culverted nature of some tributaries and/or lack of an accessible location where samples could be easily taken. A high density of sample locations was desired to observe the downstream fluctuation of pollutants in response to different conditions. Using these considerations, along with local knowledge and the ordnance survey data of the area, a list of potential sample locations was generated, and subsequently site visits were carried out to confirm their suitability.

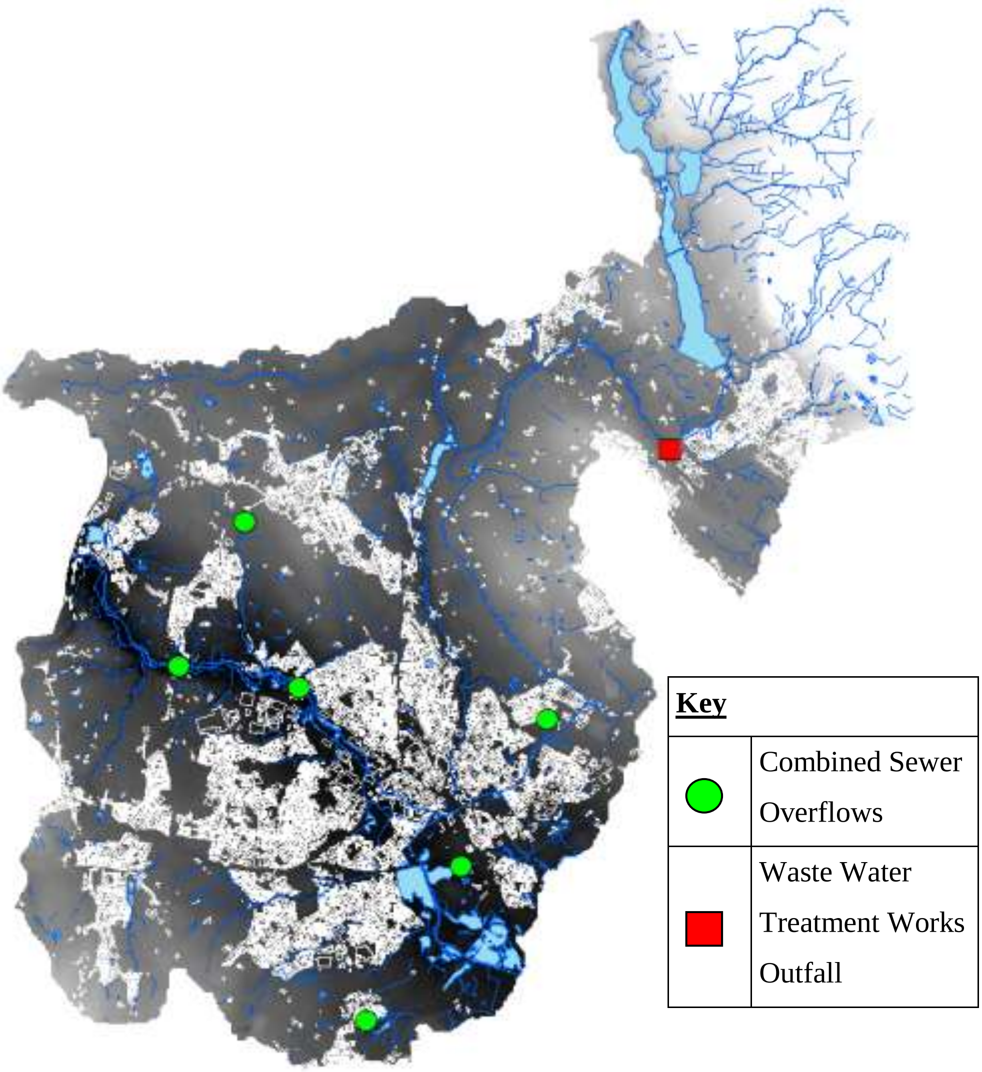


Figure 11 - Map of known CSO and WWTWs Discharge locations

Existing CSO and WWTWs discharge locations were also identified, however as the primary interest was to monitor pollutant load generated from surface water runoff sample points were not sited to seek to observe the effect of CSO/WWTWs discharge on river water quality. A map of known CSO locations was obtained from the EA and is included as Figure 11. Further details of the adjacent Horwich WWTWs are included as Appendix VIII. Further CSO details were also sought from UU but it was not possible for them to be supplied. They indicated that most CSOs were built to an “asset standard” but this would be dependent on when they were installed these will have varied.

It was desirable, where possible, to locate sample points in close proximity to existing SWD. However, this was not always possible, due to the limited access to the river banks. Despite these constraints 25 site locations were identified which are shown on Figure 8 (a full list of sample sites giving locations details are included as Appendix II). Finally, in addition to the points above, several other practical considerations also came into effect to determine the suitability of sample locations. These were:-

- Can the site be easily accessed on foot?
- Is there appropriate parking close by? (samples were decanted at the vehicle)
- Can samples be safely collected?
- Is it possible to take a sample from the centre of the river channel?
- Geographical proximity – the total number locations that can feasibly be visited by a single researcher within a day.
- What is the land use of the area surrounding the sample location?

Using these criteria, the most suitable sample points were concentrated in the lower portion of the catchment area, where the land use surrounding the Douglas’s course was primarily urban and suburban, as it was the contribution of runoff from these areas that was of most interest.

	Metals Dissolved		**Halogenated Solvents**
1.	Lead	10.	Dichloroethane
2.	Cadmium	11.	Dichloroethane
3.	Zinc	12.	Benzene
4.	Chromium	13.	Trichloromethane (Chloroform)
5.	Nickel		
6	Aluminium		
7.	Copper		
8.	Iron		
9.	Arsenic		
	Polyaromatic Hydrocarbons		**Faecal Indicator Bacteria**
14.	Benzo (b)Fluoranthene	20.	Escherichia Coli
15.	Benzo (k) Fluoranthene	21.	Intestinal Enterococci (Pres)
16.	Fluoranthene	22.	Intestinal Enterococci (Conf)
17.	Indeno (1, 2, 3-cd) pyrene	23.	Total Coliforms (TC)
18.	Benzo (a) Pyrene	24.	Faecal Coliforms (FC)
19.	Anthracene		
	Physico Chemical Parameters		
25.	pH	32.	Orthophosphate (reactive as P)
26.	Temperature	33.	Oxygen, Dissolved (% Saturation)
27.	Conductivity (at 25 C)	34.	Oxygen, Dissolved (as O2)
28.	Ammonia (un-ionised as N)	35.	Biological Oxygen Demand (BOD)
29.	Nitrate (as N)	36.	Chemical Oxygen Demand (COD)
30.	Nitrite (as N)	37.	Total Suspended Solids (TSS)
31.	Alkalinity (pH 4.5 as CaCO3)	38.	Total Organic Carbon (TOC)

Figure 12 - Proposed WQV for Testing

The physical operation of sampling followed the EA's standard method of sample collection, as described in EA document "Chemical and microbiological sampling of water - Operational instruction 19_09" (Anon, 2010). Specifically this involves dropping a tethered, clean, stainless steel can into the river either from a bridge or using a telescopic sampling pole, and drawing a water sample. This process is repeated three times at each site. When a sample is collected, the first two samples are discarded preventing cross contamination with the previous location sampled. The third sample is decanted into 4 separate sterilised bottles, before being transferred into a cool box for transport to the local EA depot, where it was stored in a fridge before transfer to the EA Laboratory. All laboratory testing was conducted by the EA National Lab Service and is completed within 24 hours of samples being collected from the river. Spot samples were collected from the centre of the river and considered to be representative of the channel cross sectional water quality at the time and location they were collected.

The frequency at which samples were collected considered the episodic nature of diffuse pollution. Ten separate sets of samples were collected and samples were taken in sequential order from upstream to downstream. It was attempted with each sample collection to capture different conditions in the river. For example, some sample sets were collected during base or low discharge conditions whereas with other sets a specific attempt was made to collect samples throughout the duration of a storm event. Details of the total numbers of samples collected, and the dates of sample collection are covered in Chapter 4 (section 4.4).

Finally, the water quality variables (WQV) for which samples were tested were identified through a literature search for typical urban pollutants, as well as historical sample data indicating pollutants that had been previously observed in high concentrations. Figure 12 displays the range of WQV which were identified through this process, including bacteria which are associated with faecal contamination to rivers, and several other chemicals which are identified as priority control substances under the WFD. All the variables selected are important water quality indicators, and most have EQS limits set under the WFD/RBWD.

3.5 Analysis of River Sample Data

Following the completion of the sampling programme, collected data was analysed to identify patterns and trends in the data set. Due to the wide range of WQV collected, it was important to explore relationships between them. Several different methods of analysis and statistical tests were used, which were selected by examining previous studies in the literature and through the use of existing knowledge. SPSS, MS Excel and Grapher 8 were all used to conduct statistical analysis, and generate graphs and figures. All of these software packages were readily available, and have been commonly used for similar types of analysis.

To establish links between parameters statistically, a principle component analysis (PCA) was completed on the data set. PCA brings out the most significant parameters from a large data set, rendering data reduction with minimum loss of original information (Vega *et al.*, 1998). It aims to exclude redundant information from the original raw data set, by obtaining a small number of variables that makes it more comprehensible and furthers the analysis. In addition, it provides insights into the degree of correlation between variables. It is noted however that PCA is sensitive to outliers, missing data and poor linear correlation between variables can result in inadequately assigned variables (Sârbu and Pop, 2005).

Publically available discharge and rainfall data sets were correlated with the analysis of collected sample data. Using a flow duration curve generated from discharge data, the rivers base flow was characterised, and samples were split into those taken at low and high discharges, based on the discharge at the time they were collected. A T-test was used to identify if there were any statistically significant differences between high and low groups. The split data was also used to produce the mean and the ranges of values for each variable, at each site, which were subsequently plotted as a box plot. By plotting samples collected during storm events against discharge and rainfall data, the change in sample concentration across events was observed.

Finally, collected water quality data was used to authenticate outputs from the EA's Source Apportionment Geographical Information System (SAGIS) model. This model uses the EA's water quality monitoring database to predict the quality of water along all water courses in the UK. Outputs from the model were compared against the concentrations observed in the water samples collected, to determine the accuracy with which the model was predicting values.

Methodology – Phase 2

3.6 Selection of Treatment Systems and Locations

The completion of the analysis on the river sampling data marked the end of the first phase of this study. Using the knowledge and understanding gained from the analysis of sample data, the next stage of the project was to try to address, where possible, the levels of diffuse pollutants observed in the river. This section of the methodology explains the rationale behind the selection of potential sites for installation of mitigation measures on the River Douglas, the selection of suitable measures, their design and installation and the methods used to analyse their effectiveness in mitigating the effects of UDP. It also explains why stakeholder engagement is important in relation to the work around treatment systems, and how the requirements of various stakeholders were addressed. It covers in detail the required design work for the installations, and how this was subsequently carried out. Finally, it explains in more detail the specific methods used in the monitoring and quantification of pollutant removal for each of the products/ systems.

After some initial investigation of costs and works required to treat even small discharges, it was obvious the funding made available by the EA was insufficient to affect significant change to the whole study area. Therefore it was decided to select a small number of unregulated surface water discharges, and to retrofit them using either SuDS or PTS to provide water treatment, and to subsequently monitor them to quantify the benefit they provided. The river sampling study was completed by the EA and the University of Salford working in partnership. The work required to design the PTS and complete the subsequent construction works to install them, necessitated the development of relationships and engagement with a range of different stakeholders.

The primary function considered for structural measures was the removal of pollutants from surface runoff, while this may result in some flood control benefit this was a secondary consideration in product selection. Pathways for transport of pollutants to water courses were explored in the literature review (section 2.3.2). Interception of pollutants along these pathways was considered to be an effective method of reducing the volume of pollutants ultimately discharged into watercourses and receiving waters. The methods of achieving this have also been summarised in the literature review (section 2.4.2 and 2.4.3). Figure 13 visualises the effect a system should have on pollutant concentration in comparison with discharge. A series of key factors were considered when selecting methods and systems of pollution remediation.

- Cost of product/ system/ works;
- Required space (m^2);
- Treatment capacity (m^3);
- Main treatment process, i.e. infiltration, settlement, retention, etc.;
- Retention time;
- Method of monitoring.

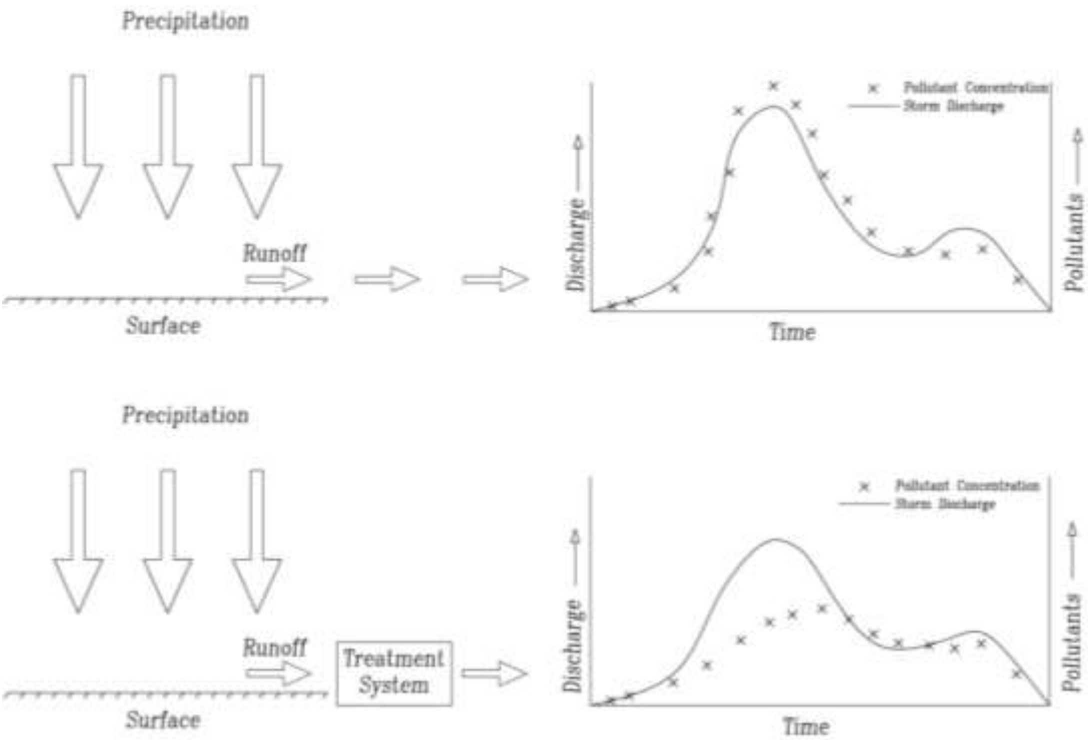


Figure 13 - Treatment system effect pollutant concentration during storm event

The limitations of some sites precluded the possibility of using certain systems or techniques, and similarly the size or nature of some systems and techniques made it impossible for them to be utilised in certain locations. This meant that, compromises had to be made in due to practical considerations, to ensure that it would eventually be possible to install and monitor products. Initially a range of different measures were considered for installation at several sites in the vicinity of areas of poor water quality, identified through the river sampling.

As has been established in the introduction, the primary focus of this work was to investigate water quality and how it could be improved. When considering the methods available to provide treatment this was the primary consideration. As discussed in the literature review the different techniques; namely SuDS (section 2.4.2) and PTS each (section 2.4.3) have a series of separate advantages and disadvantages. SuDS can deliver water quality improvement but they give equal focus to flood control and amenity/ biodiversity provision. In this study there was a need to monitor and quantify the effectiveness and performance of measures. As identified in the SuDS section of the literature review, monitoring their water quality performance is problematic due to the long retention times of detention features and the water loss through infiltration. Therefore it was concluded that from a monitoring perspective, PTS offered a definite advantage in this study as systems have a defined inlet and outlet where samples could be taken and compared.

Therefore it was decided that PTS were more suitable to use on the project, considering the confined nature of the urban environment and the difficulties with monitoring of SuDS. Using existing knowledge and stakeholder engagement a range of suitable PTS were identified. They were subsequently paired with sites where their installation would provide water quality benefit. Secondly, to allow the installation of PTS by a contractor at identified sites, construction design drawings were produced (section 3.10 contains the details of the design work completed for each PTS used).

In all samples collected in elevated discharge TSS and heavy metals were observed in collected river samples (section 4.4). The different methods of mitigating these pollutants were investigated, and a list of seven different suitable products generated. Separate sites were identified where their installation was proposed. Initially it was planned to progress all seven products to site, however as the scope of work became defined for each location, and more comprehensive quotes were received from the civil engineering contractor, increased costs necessitated a reduction in the number of product/sites to ensure the extent of the site works did not push the project over budget.

To find tangible locations where mitigation measures could be installed, and where the associated engineering works could be undertaken, the drainage asset register of United Utilities (UU) was used to identify the existing surface water system. A large quantity of Wigan's surface water drainage requirement is served by a number of separate surface water drains. Using the UU register, 15 untreated surface water discharges were identified in Wigan that discharged into the sampled study area. Those which discharged in the vicinity of the two central Wigan sample locations were investigated first, as this was where average pollution concentrations were observed in the river monitoring. The EA had recently produced a catchment walkover report for the area. This report identified all untreated discharges for the whole of the North West region and the project study area was completely covered. The report did not highlight any further polluted discharges, where the installation of PTS was feasible or which had not already been identified by this study.

Unfortunately no suitable locations could be identified, due to the fact that most discharges located in central Wigan were below buildings, roads etc. This precluded utilising such sites due to the unacceptable levels of disturbance that would be caused. Ultimately only four sites proved suitable, and these were progressed to completion, where a PTS was installed, monitoring carried out and results generated. The four sites and the details of the PTSs installed are listed in Table 4.

Table 4 - Project Sites and PTS

Site	Grid Ref	PTS Installed	Name	Supplied by
Cherry Gardens	SD 58274 07542	Hydrodynamic Vortex Separator	Downstream Defender	Hydro International
Coppull Lane	SD 58639 06608	Hydrodynamic Vortex Separator with additional filtration	Storm X4	Poly pipe Civils
Little Wigan Theatre	SD 58449 05803	Oleophilic Polymer Sponge	Passive Skimmers	Smart Sponge Products Ltd
Scott Lane	SD 55903 06535	Oleophilic Polymer Sponge	Passive Skimmers	Smart Sponge Products Ltd

3.7 Monitoring of Treatment Systems

Monitoring of systems needed to address one key goal, to determine their benefit in terms of the efficiency with which they removed pollutants from the effluent they were installed to treat. To understand how effective a certain system is in terms of its benefit to water quality and the volume of pollutant it can remove from an effluent, two key parameters need to be measured (indicated in Figure 13). These are, the volume of discharge passing through the system during a storm event, and the volume of pollutants entering and subsequently exiting a system. Through a comparison of these parameters the quantity of pollutant removed and retained by the system can be calculated, thus determining its efficiency.

As systems did not function in the same way it was not possible to use an identical method or equipment to monitoring each one. In practice monitoring was comprised of one of two different methods. These were either, the use of automated sampling equipment to take a series of samples of effluent entering and exiting a system, or the measurement of the volume of pollutant a PTS removed over a certain time period. Section 3.10 covers each installed product in more detail, and gives a detailed explanation of how monitoring was conducted at each site with different products and systems. As with the river sampling the monitoring of PTS was event based. For the purposes of this study 'an event' refers to, an increase and subsequent reduction, following rainfall, in discharge through a monitored system resulting in an increase in pollutants concentration. To quantify this type of episodic pollution, the target was to monitor as many events as possible for each system. This approach was very much dependent on weather conditions experienced during the period available for this phase of monitoring. Therefore it was proposed where possible to capture a minimum of five events at each installed system.

3.8 Stakeholder Engagement

The term 'stakeholder engagement' has been described as '*an active initiative to bring together groups of stakeholders, usually in response to a specific exercise or need. From the perspective of those consulted, this engagement gives them the opportunity to make their needs and requirements known to the consulting body. From the perspective of the consulting body to engage stakeholders means or should mean not just taking on board views but being prepared to take notice of them'*(adapted from Stewart (2009).

As was identified in the literature review (section 2.4.5) the complex nature of the urban environments is a significant barrier to the retrofit and wider use of more sustainable drainage systems and products (Martin et al., 2007). Within a very small urban area there are many differing views and expectations from dozens of different parties, all with a legitimate interest over the best method to adopt to address a certain problem within the river, or to achieve a certain goal in improvement of the water quality. Therefore, any project to retrofit such systems requires significant engagement with affected stakeholders to ensure success.

Considering the views of stakeholders is important, since without their cooperation it would be impossible to complete certain tasks. However sometimes accommodating these views can have cost implications, as compromises and additional elements may have to be considered and incorporated into the site design. Due to the complex nature of the project, and the often conflicting perspectives of different stakeholders, careful and considered engagement was required. This ensured the successful cooperation of interested parties, to ultimately allow all the worked planned to be undertaken, in terms of installation of PTS and subsequent monitoring. Specifically for this project a range of different stakeholders were involved including:-

- Local Authority – Wigan Borough Council,
- Water Company – United Utilities,
- Manufacturers of the PTSs utilised,
    - (Hydro International, Poly Pipe Civils and Source Control)
- Civil engineering contractor – William Pye,
- Consulting engineer – Tim Booth Associates,
- Construction Design and Management consultants – Black and Veech,
- Instrumentation Company – Environmental Monitoring Solutions.

Interactions will all these different groups and organisations were necessary as they provided services and permissions all of which were required to allow the completion of this study.

3.9 Pre Site Works

With a range of products paired with feasible sites, the next stage was to ensure land and asset owners would provide permission for works to go ahead. Out of the four sites, three (Cherry Gardens, Coppull Lane and Little Wigan Theatre) were owned by Wigan Borough Council (WBC) which made gaining permission for works more simple, as the Council has a standard applications process for this. The fourth site (Scott Lane) was located between two commercial garages, and the permission of both garage managers to undertake works was obtained.

Prior to the commencement of works, permission from the owner of the assets (drains, access manholes, etc.) had also to be obtained. All assets were owned by UU and so a legal agreement was negotiated between the EA and UU legal teams, which placed liability for all future maintenance and responsibility on the EA. This highlights a key issue that arose during the project, i.e., that neither UU, WBC nor the EA have any enthusiasm to take over the new assets and more crucially their on-going maintenance, irrespective of the long term benefit they would provide. In regards to the long-term effectiveness of PTS assets this is likely to be problematic, because suitable cleaning and maintenance of systems is required for them to continue functioning effectively. This institutional and organisation issue was identified in the literature review as an existing barrier to the use of both SuDS and PTS (section 2.4.5.3)

Other pre-site works included the collation of pre-construction information, a services search, to identify possibly affected services. An unexploded ordnance survey was also completed. Due to the nature of the construction work required to install the products, it was necessary to produce construction drawings for three of the four sites. The fourth site, where passive skimmers were installed to road gullies in the vicinity of Little Wigan Theatre, was a straightforward process and did not require any design work. Once the designs were finalised, contractor appointment was completed through the EA's standard tendering process. A range of contractors were invited to tender for the work, and standard criteria were used to select the best bid. The following section describes each product utilised, detailing the design criteria that needed to be addressed, a summary of the construction work required for installation, and then a detailed explanation of how monitoring was completed.

3.10 Design, Installation and Monitoring

Construction designs were needed for works at Cherry Gardens, Coppull Lane and Scott Lane. Design work was completed by the researcher under supervision at Tim Booth Engineering consultancy. As it was desired to actively monitor each installation to determine its water quality benefit, additional features were required at some sites which would not normally be necessary at unmonitored sites. As products were not identical, the method to monitor them and determine how effectively each was operating differed slightly. The HDVS type PTS's deliver more active treatment and function primarily during the episodic pollution generated by rainfall. Whereas the hydrocarbon sponges functioned passively, meaning that a different method was used to monitor them.

3.10.1 Downstream Defender

Design work to facilitate the installation of the DD at Cherry Gardens was undertaken first. Figure 14 shows the long section of the design, and illustrates the system design and construction (a full set of drawings is included as Appendix IV). As a part of the legal agreement with the UU, the chamber had to be located offline of the original drain so that it could be reinstated in the future. Water passing into the system was diverted by a chamber into the DD unit, where settlement of particulates occurred. Water was then discharged out of the unit and back into the main drain through a second downstream chamber. These chambers doubled as accommodation for the required sampling equipment to monitor the system. The design also allowed the utilisation of a high flow bypass in the event of a blockage of the inflow to the DD unit.

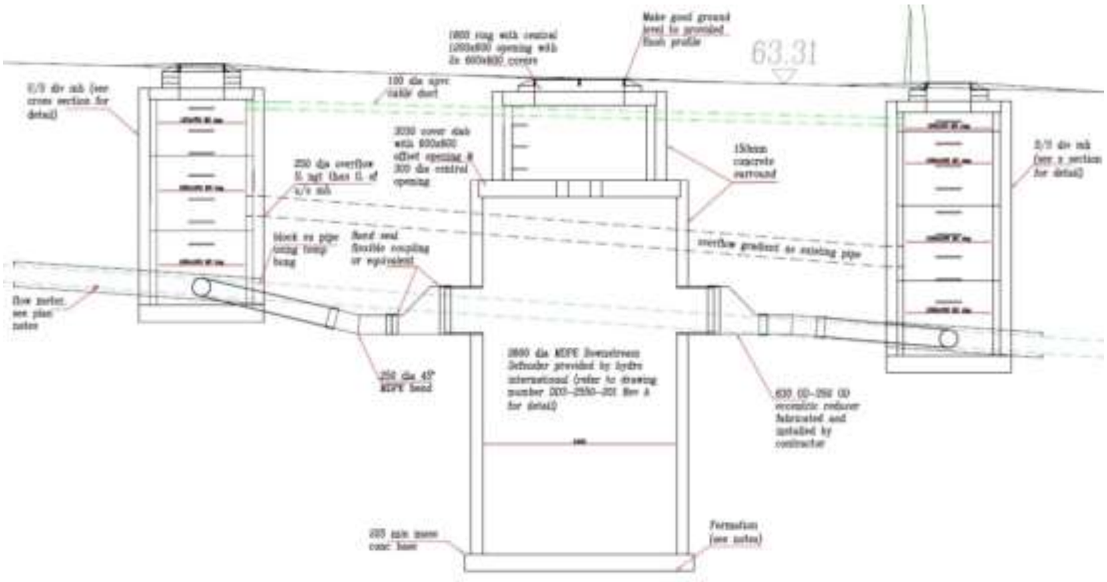


Figure 14 - DD Long Section Detail

The manufacturer of the DD does not provide predicted removal efficiencies for the device in sales literature; they specify that the performance of the device varies depending on the site specifics, which include:

- the volume of discharge passing through the system;
- the flow treatment capacity of the unit installed;
- the particle size distribution within the TSS entrained in discharge;
- land uses of the catchment area served by the storm water drain.

The size of the DD installed in this project was a 2.55 diameter meter unit, which has a maximum treatable flow rate of 110 l/s with a maximum through flow of 600 l/s. A technical drawing of the DD is included in Appendix IV.

A series of photographs taken during the installation of the DD are included as figures 15-17.

Figure 15 - DD Sump placement

Figure 16 - DD being lowered in to place

Figure 17 - Upstream Diversion chamber construction

Figure 15 shows the sump section of the DD being lowered into place, the top section visible to the right was then placed on top, and the two sections fixed were together as shown in Figure 16. Figure 17 shows the upstream diversion chamber being constructed. Construction work was completed on the DD site by the end of September and the monitoring of this product was undertaken between October 2013 and March 2014.

Several key pieces of equipment were used in monitoring; these were two ISCO 6712 auto sampling units and an ISCO 750 flow module and sensor. One sampler was placed in the diversion manhole upstream of the DD chamber, and the second sampler was placed in the downstream diversion chamber. This arrangement is shown in Figure 24. Figures 18 and 19 show the sampler units, and Figures 20 and 21 show the location of the flow sensor and the positioning of the end of the sample line. Sample units were suspended in the sample chamber using a frame hanging on the lip of the manhole frame. The external battery required to power the units was placed above, as can be seen in Figures 18, 22 and 23.

Figure 18 - ISCO 6712 Sample Unit

Figure 19 - Sampler with Cover Removed

Figure 20 - (left) View up the pipe showing the placing of the flow sensor

Figure 21 - (above) View showing fixing of the sample line in the chamber

To remove samplers from the manholes a custom made platform and winch was used, which is shown in Figures 22 and 23. This allowed the sampler to be easily removed from the sample chamber, to check collected samples and download the flow sensor data log stored on the internal memory of the upstream sample unit.

Figure 22 - Sampler winched out of sample chamber

Figure 23 - Sampler and Winch

The sample units were programmable and could take between 12 and 24 samples at chosen intervals. The total number of samples collected per event varied, and depended on the duration of the storm event. The sample programme was triggered by a signal from the flow sensor which detected increases in pipe level and velocity. The samplers were connected together by a communication cable, and the downstream sampler (which acted as slave to the upstream unit) would be triggered after the upstream sample program commenced and mirrored its program. This configuration of the samplers and flow sensor allowed the collection of a set of comparable samples from both up and downstream of the DD.

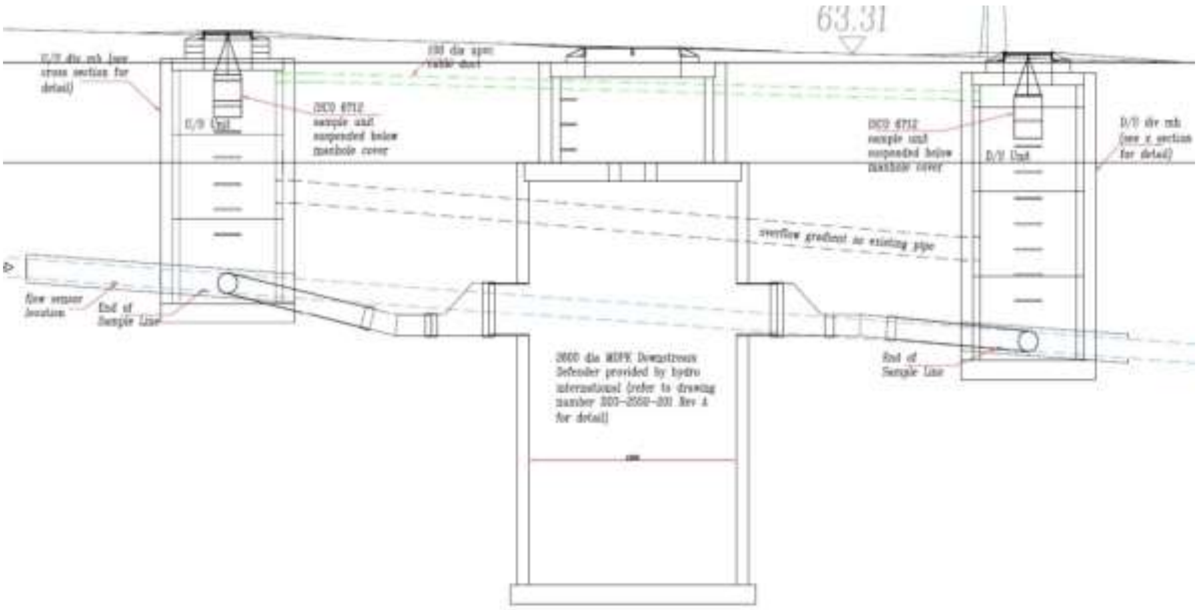


Figure 24 - Long sections showing location of sampling components

Figure 25 - The carousel units separated from the auto sampler

As with the river samples, chemical testing of water samples was completed at the Environment Agency labs. Samples were transported to the EA depot in the carousel (shown in Figure 25), which was detached from the bottom of the sampler. Here they were decanted into standard bottles (shown in Figures 26 and 27) which were then boxed and transported via courier to the lab.

Figure 26 - Decanting of sampler bottles

Figure 27 - Decanted Samples

3.10.2 Storm X4 Heavy Traffic

The physical arrangement of the X4 was very similar to the DD and they functioned in a similar way. This meant that there were a number of similarities in terms of the design and construction. The same offline configuration was used to satisfy the legal agreement with UU, and to allow the original drain to be reinstated after the end of the monitoring period if so desired. Figure 26 shows the layout drawing of the X4 chamber (a full set of the drawings are included as Appendix V). As mentioned, the arrangement of the system is similar to that used for the DD chamber, in terms of the locations of the diversion chambers and the siting of the main unit offline.

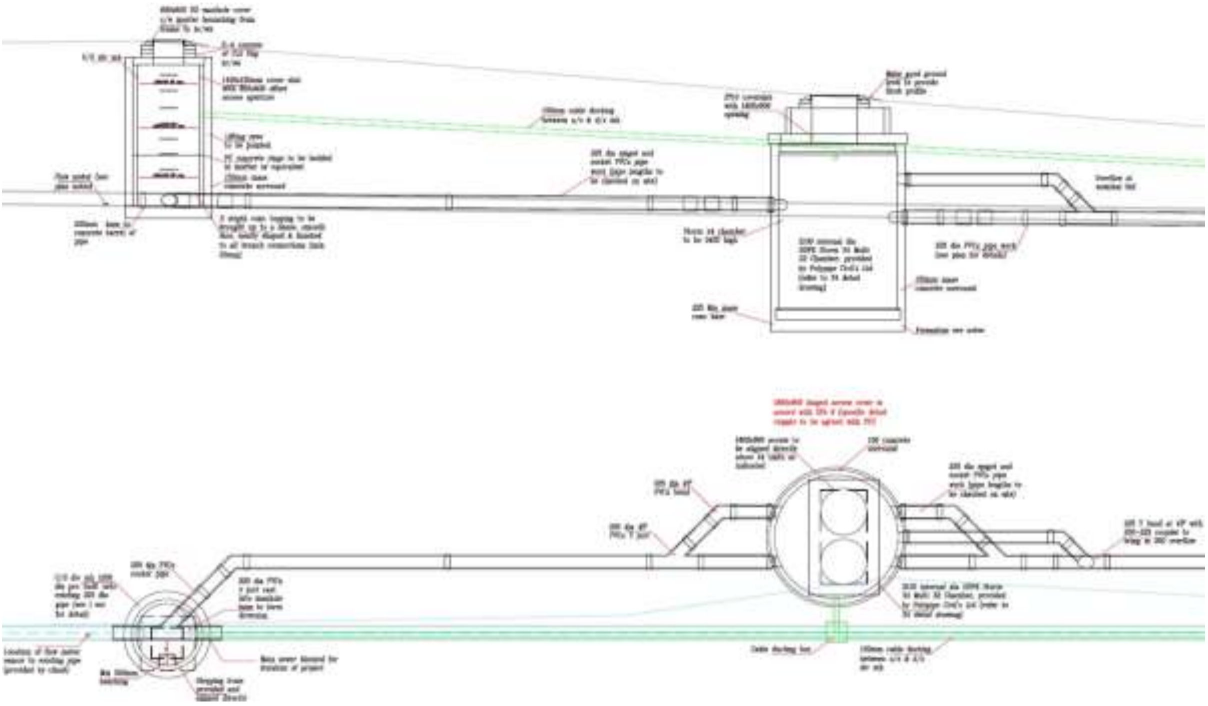


Figure 28 - X4 Construction Drawings

The key difference between the DD and the X4 is the additional filtering capabilities of the X4 unit. However, this limits the discharge that the unit can treat to 14 l/s. To increase its treatment capacity the product used an “X4 heavy traffic”, which is essentially two separate X4 units contained within a larger outer chamber. This means that the unit as a whole incorporates 2 separate inlets, and the two units also had a shared internal bypass to accommodate discharges which were greater than the unit’s treatment capacity. All this can be seen on the plan view of the design in Figure 28. Figure 29 shows the chamber housing the two X4 units being lowered into place. Figure 30 shows the chamber in situ. The excavation for the upstream diversion manhole can be seen behind it.

Figure 29 - X4 chamber installation

Figure 30 - X4 chamber (foreground) and upstream diversion manhole (background)

The pipe gradient at the Coppull Lane site was very gradual, and to ensure the system had enough fall to function correctly, the up and down stream diversion manholes had to be spaced much further apart than the DD at Cherry Gardens. There was only approximately 400mm of fall in the drain across the whole site, and there was a 250mm difference between the in and outlets of the X4 chamber to allow it to function. This left only 150mm available to accommodate the required pipe gradient to ensure water flow through the system. While the DD site experienced a water base flow meaning there was always discharge through the system, the X4 site did not experience discharge except during rainfall. As a result of this, the system acted like a sink and a small amount of water was always present in the upstream manhole of the system.

Due to the similar design and construction, for all intents and purposes, the monitoring conducted on the X4 product was made using the method previously described (section 3.10.1), as samples were again collected in up and downstream sampling chambers. There was however one important difference between the sites in terms of sampling. Due to the inlet to the upstream sampling chamber being permanently full, the available dopler flow meter could not detect the level change which was required to trigger the sampling equipment (as described in the previous section). To overcome this problem, the flow meter was relocated to the downstream manhole allowing triggering to be achieved. Other than this change, the X4 and DD sampling were completed in the same fashion. The same sample units and flow sensor were used, and the method of sample collection, decantation and transport were as described in the previous section (3.10.1).

As with the DD the X4 was a prefabricated PTS, it was delivered to site fabricated with both X4 units enclosed within an outer chamber. The fabrication of the unit is completed within a factory setting; individual components of the system are assembled prior to the constraints of site. As the system is made of two separate units and requires two separate inlets the main 225 mm pipe at the Coppull Lane site had to be bifurcated to pass flow through both units. A custom made Y section of pipe was used for this purpose to ensure that as far as possible flow was divided equally between the two units. Invariably there will be times that flow is not equally distributed between the two units however it was considered that with the appropriate maintenance the risk of one unit becoming blocked over the other was minimal.

The manufacturer of the X4 provides stated 'aims' for the device which they expect it to achieve with respect to average annual loads of nutrients, heavy metals and oils. These details are provided along with a technical drawing in Appendix V. As with the DD specific removal efficiencies are not provided for similar reasons. The X4 heavy traffic unit was installed on site for this project has a maximum treatable flow rate of 28 l/s and a maximum through flow of 92 l/s.

3.10.3 Auto-Sampler Monitoring Methodology

The process of sampling both the DD and X4 with auto samplers has been outlined in section 3.10.1. There were a number of factors which complicated this monitoring process (particularly in respect of the DD), and the subsequent analysis of the collected data. Normally surface water drains (SWD) only contain discharge during rainfall, meaning they are empty during dry weather, making it easy to differentiate periods of discharge induced by storms. However between the original investigation and the construction of the DD, it was found that the selected SWD had a constant base discharge passing through it. This fluctuated between approximately 5 to 8 l/s, and is thought to be caused either by a small brook which has been diverted into the drain, or through a water main leak. This flow was observed to be clear, with little or no TSS material visible in samples taken, indicating that the latter of these explanations was more plausible. This meant that differentiating between base flow flux, and the start of storm events was sometimes difficult, as the hydrograph was altered by the addition of this base flow.

It also meant triggering the sampling equipment during storm events was more difficult. For example, a flow sensor was used to trigger the sampling programme, by setting a threshold value in respect to either level or velocity. Due to the fluctuation in the base flow, this needed to be set above the typical highest values it produced (with respect to level or velocity); otherwise the samplers would be triggered by the base flow and not as the result of a storm event. As a consequence of this, the beginning of some events were missed, because, if the base flow was low the samplers would not commence collecting samples, until the storm had raised discharge enough to pass the trigger level.

The frequency at which samples were collected was generally set at 5 minute intervals for TSS and 10 minutes intervals for metals. In one event a shorter sample interval (2 minute) was used to determine if there was a significant change in sample concentration, when they were taken at shorter frequencies. However there was no appreciable additional fluctuation in pollutant concentration during this shorter interval, so for all following events, 5 minute intervals were used. Thus samples were considered to be representative of the 5 minute period, during which they were collected.

An important part of analysis has been to accurately compare the volume of pollutants entering the system against that exiting it, but due to their significant cost it was not possible to purchase two separate flow sensors to allow the monitoring of discharge both upstream and downstream of the DD unit. The significant internal volume of the DD (approximately 15 m^3) caused a 'lag' effect on the discharge as it passed through the system. For example, when discharge increased as a result of a storm event, the change passes through the upstream sample chamber, being recorded by the flow sensor; it would then take a period of time for it to pass through the connecting pipe into and through the DD chamber until it was observed in the downstream sample chamber.

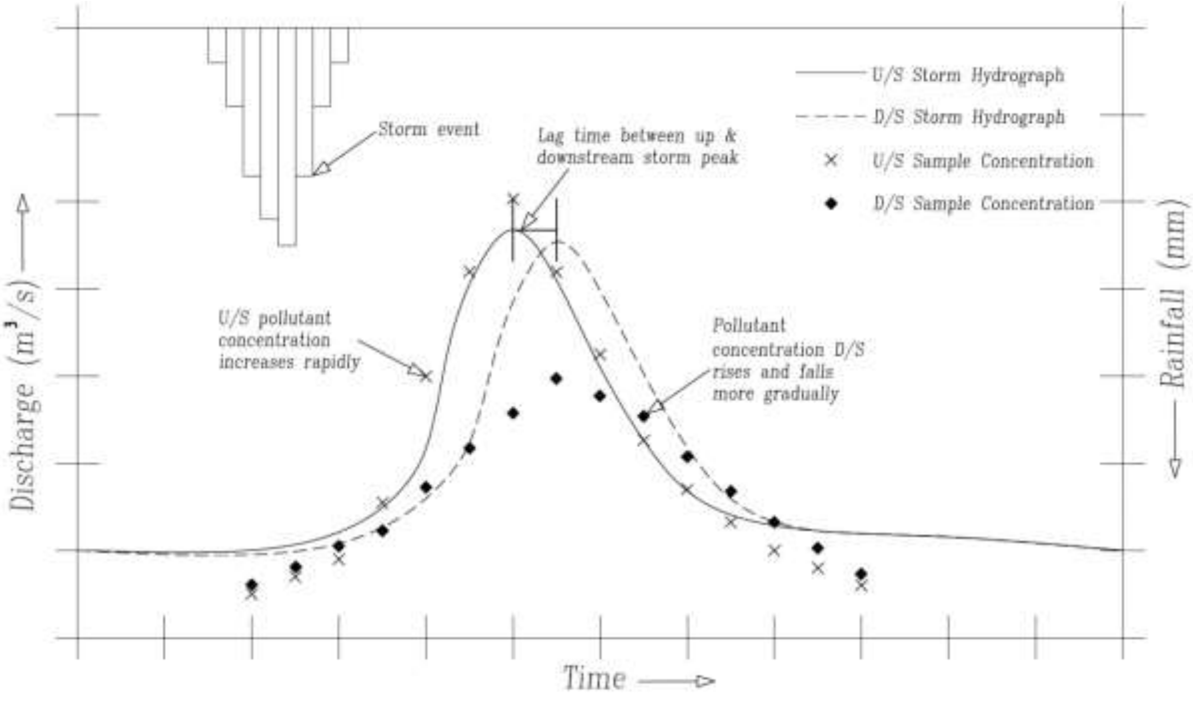


Figure 31 – Demonstration of lag time on sample concentration

As the samplers were set to take simultaneous samples, it was necessary due to lag for the downstream sample results to be offset in terms of the time they were collected, when comparing them with the log of the discharge, collected in the upstream sample chamber. The lag time also varied as discharge fluctuated, being larger at low discharges and smaller at high discharges. If two flow meters had been available, the calculation of the volume of pollutants upstream and downstream would have been completed using a log of discharge from each manhole. However as only one flow meter was available, the effect of the lag time between sample chambers has to be considered. This effect is summarised in Figure 31.

Initially it was proposed to take flow weighted composite samples, however due to the problems encountered with triggering the flow meter discussed above this was not possible. As only one flow meter was available the only method available to trigger the downstream sampler was to mirror the upstream sample program to the downstream sampler. If flow weighted sampling has been utilised this would have resulted in the downstream sampler forming composite samples based on flow data at a different location as the time lag effect of transition through the DD chamber meant that the downstream flow rate was different to the upstream. While the method adopted may be considered less representative, with the potential for greater error, it was considered that time weighted sampling was more suitable to deal with the constraint posed by the absence of a downstream flow meter. As the number of samples collected for each event was at least 12 the relatively high density of samples can be considered to reduce the level of error significantly.

The manufacturer (Hydro International) has monitored a similar device under laboratory conditions to ascertain the lag time of discharge passing through the DD chamber. This work determined that the lag time of water between the upstream and downstream of a DD system was equal to 0.61% of the mean residence time[2] of water within the system. Lag was then, defined as:-

Lag Time (2)

$$LT = NMRT \; X \; \frac{v}{Q}$$

Where:

LT = lag time

NMRT = Normalised Mean Residence Time

v = Volume

Q = Discharge

[2] Mean residence time is the time taken to replace the entire volume of a system at a certain rate of discharge.

Application of Equation 2, allowed the water samples collected downstream to be offset backwards based on the discharge at the time a sample was taken and compared against the actual representative discharge from flow sensor located in the upstream sample chamber. This allowed a more accurate comparison with the equivalent upstream discharge.

Level and velocity were logged at 2 minute intervals by the flow sensor, but it was often found during low flow periods that the sensor struggled to detect velocity accurately and this is partly why water level was selected as a more accurate trigger mechanism. This also meant that for some intervals, the sensor recorded a value of zero for the rate of velocity and so, to allow the calculation of discharge in these periods, a second equation has been used to provide missing velocity readings.

To calculate this equation, velocity readings were taken from the whole of the monitoring period. Readings for each increase in level were then divided and sorted, with missing velocity values removed, and the average of the remaining values for each level interval plotted on a common scatter plot. A polynomial line was applied to the plot, and the equation governing this line was used to calculate missing velocity values in the analysis and is shown Equation 3. The plot is also depicted in Figure 32.

Velocity (3)

$$\mathrm{V} = 1911.4 \times \mathrm{L}^2 - 325.84 \times \mathrm{L} + 14.387$$

Where:

V = Velocity

L = Level

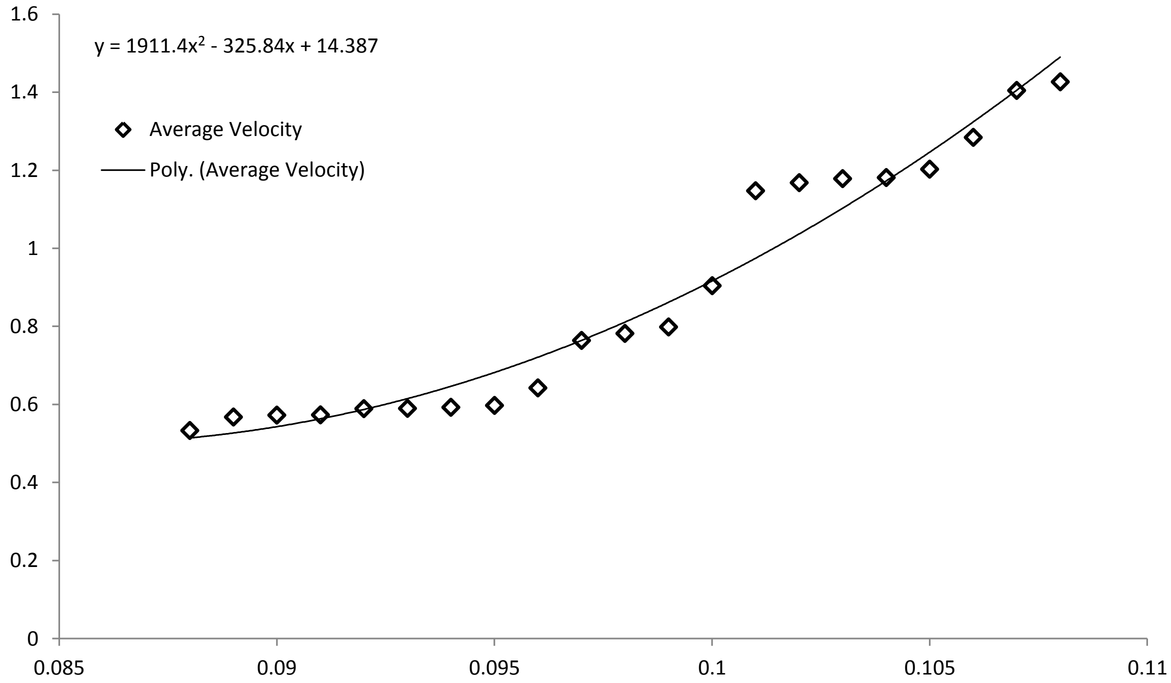


Figure 32 - Average Velocity against Level

3.10.4 Smart Sponge Passive Skimmers - Cage

Smart Sponge is a proprietary blend of polymers which is hydrophobic and oleophilic, allowing water to pass through its structure while hydrocarbons are chemically bonded into its chemical matrix, and sediments are filtered and retained within the product structure. The structure of the product maximises the effectiveness of these polymers, by forming them into an extremely porous structure that allows effective, long-lasting absorption, without clogging or channelling. The absorption of hydrocarbons and free phase products is permanent, and the saturated product cannot be washed, pressed or leached out of the sponge structure. Free Phase is used to describe hydrocarbon contamination which is found occurring as a floating layer on the surface of a body of water, or other surface, as it is less dense than the water. This is also commonly referred to as Light Non-Aqueous Phase Liquid, Common contaminates within this category are kerosene, diesel and petrol.

Figure 33 - Clean Smart Sponge Skimmer

This product is available in a number of different forms for application in different stormwater treatment situations. It is particularly versatile, and can be easily retrofitted with little disruption. Therefore it is particularly attractive for application in inner city urban environments where the retrofit of other products is not possible due to the requirements of more significant and disruptive civil engineering works. The specific product used for this project was a passive skimmer. This consisted of a net bag containing 12 tubular shaped pieces of smart sponge material; as shown in Figure 33.These products were then utilised differently at two separate sites. At Scott Lane, 40 bags were enclosed within a cage and placed into a discharge. A further 40 bags were utilised in a series of separate locations centred around Little Wigan Theatre where individual skimmers were installed into road gullies. At both sites skimmers were left in situ for 15 months before being removed. Skimmers were installed initially at Scott Lane, and were located between two commercial garages where an unmarked 600mm drain discharged into a small basin. The discharge was undercutting banks and causing the basins slopes to fail (The site is shown in Figures 35 and 36). From this basin, the drain flowed through a small culvert, which passed under the A49 before discharging into a small pond, which in turn flowed into the main river channel. After a site visit, there was clear visual evidence that this discharge contained pollutants.

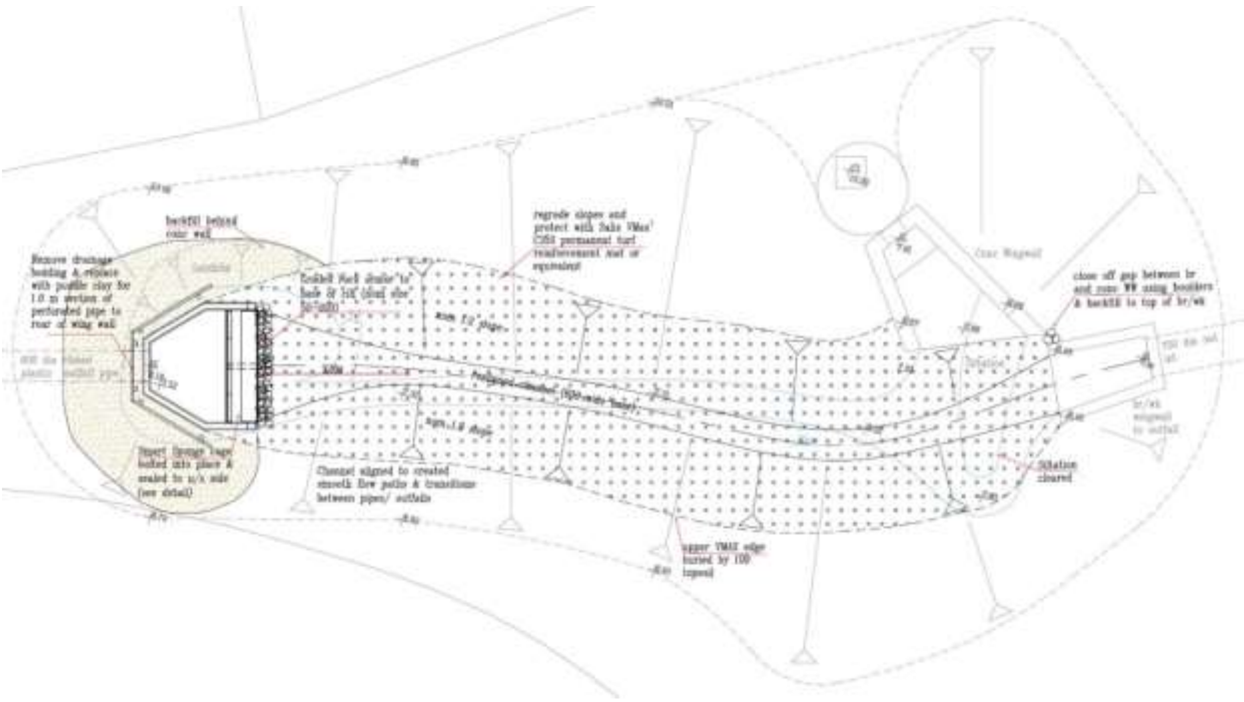


Figure 34 - Scott Lane design, plan view

The design completed for this site enlarged the drainage channel, and re-profiled the position of the drainage channel to prevent further erosion to the basin slopes. The re landscaped area was also protected with a seeded geotextile to encourage vegetation growth and prevent erosion. Where the drain itself discharged, the end of the 600mm pipe was enclosed with a new wing wall structure providing erosion control and preventing undercutting of the drain. To the mouth of this wing wall a concrete block was cast and on top of this a cage was fixed which contained 40 passive skimmer packs. The pool created behind the concrete block functioned as a settling pool to remove heavy debris. The discharge then passed over the block flowing through the cage containing the smart sponge. This allowed the removal of hydrocarbons and sediment from the water column discharging from the drain. This design is shown in Figure 34 (A full set of the drawings is included in Appendix VI).

Figure 35 - The discharge at Scott Lane

Figure 36 - View down the site away from the discharge

The original site is shown in Figures 35 and 36. Figure 35 shows the discharge prior to any works. Pollution is visually evident and the erosion of the bank to the right of the picture is clear. Figure 36 shows a view down the site, prior to works heavily over grown with the direction of flow undercutting the bank.

Figure 37 - View of the new wing wall

Figure 37 shows the new wing wall in-situ supporting the 600mm discharge; the wooden panelling is the formwork for the concrete block which was cast on site. New seeded matting to encourage re-vegetation and provide erosion control can be seen lining the slopes. Figure 38 shows the completed site from the outlet to where it passes through the culvert beneath the road prior to discharging to the river. The cage holding smart sponge can be seen fixed above the block.

Figure 39 shows the new arrangement in operation. Water is held back behind the cage before passing through the sponge as it is discharged, allowing interception of hydrocarbons and sediments. The rock armour below the wing wall provided additional erosion control by preventing any undercutting of the structure.

Figure 38 - View from site outlet

Figure 39 - Smart sponge in operation

3.10.5 Smart Sponge Passive Skimmers - Road Gullies

Due to the simple nature of installing the passive skimmers to road gullies no design work was required. Figures 40 and 41 show the installation of skimmers, and how they were fixed to the lids of road gullies. As is evident from Figure 41 each, skimmer bag has a fabric loop which in conjunction with a length of rope was used to attach bags to the metal cover of the gully. All gullies were cleaned prior to skimmers being installed, so the majority of pollutants absorbed by each skimmer and which accumulated in the gullies should have done so during the time that skimmers were in situ.

Figure 40 - Cleaning of the road gully prior to skimmer installation

Figure 41 - Fixing of skimmer to manhole lid

As previously mentioned (section 3.10.4/5), at both sites skimmers were left in situ for just over a year. Once removed, to quantify the volume of sediment and hydrocarbons captured by the skimmers, the following procedure was undertaken. Bags were dried in a drying oven at 50ºC for 12 hours, before being weighed using two separate scales (Kern CBX and Kern KB) ensuring any discrepancy between scales could be observed, as shown in Figure 42and45).

Figure 42 - Weighing of individual passive skimmer prior to cleaning

Once weighing of the skimmers was completed, each bag was then vigorously washed removing the sediment and material captured within the product. Most bags were heavily silted as is illustrated in Figures 42 and 43. The manufacturer stated, that even vigorous action on used the Smart Sponge would not wash out absorbed hydrocarbons, and therefore, a jet wash was used to clean bags and Smart Sponge pieces, Figure 44.

Figure 43 - Piece of SS prior to washing

Figure 44 - A jet wash was used to clean Smart Sponge pieces

Skimmers were then re-dried in the drying oven for a further 12 hours at 50ºC, before each bag was weighed again (using the same two scales). Skimmer bags and associated Smart Sponge pieces were weighted separately, shown Figure 45.

Figure 45 - Cleaned smart sponge pieces being weighed

By recording the weights of skimmers both prior to and after cleaning, and then comparing these weights with the known average weight of a clean product, the various fractions of removed pollutants (sediments and hydrocarbons) could be calculated. This process was duplicated for all bags that were recovered from the Scott Lane and Little Wigan Theatre sites.

This Chapter has covered in detail the difficulties and barriers that can be experienced during the process of gaining the required permissions from land and asset owners to allow remediation works to go ahead. It also describes the process of selecting, designing and installing the range of proprietary systems that were selected. Finally the methods used to monitor these products at their point of application have been explained. The following penultimate Chapter lays out the results of this monitoring and places the volumes of pollutants removed in the context of the estimated pollutant input across the whole study area. Where possible comparisons between products are drawn in terms of the volume of pollutants they removed.

3.11 Analysis of Treatment System Monitoring Data

The monitoring of individual treatment products yielded a second data set requiring analysis, which is outlined in the second part of Chapter 4. As described in the previous section, PTS utilised for the project varied between sites necessitating a different monitoring method. This in turn yielded differing results from each site. Where data allowed two main pieces of analysis were completed for each system. These were:-

- The presentation of one of the events monitored, displaying the discharge of the event, the concentrations of pollutants monitored in samples upstream of the PTS and the corresponding pollutant concentration of samples downstream. This allowed a visual representation of the effect of the installed PTS on pollutant concentration of samples during a storm ‘event’.
- A summary of all other events monitored, displaying the total volume of each event monitored, and the measured volume of pollutant entering and exiting the PTS. This allowed the benefit of PTS in terms of the actual physical volume of different pollutants removed to be quantified.

Unlike the river water quality analysis it was not necessary to undertake any statistical tests, and all necessary calculations, graphs and figures were completed in MS Excel. Using the log of discharge data collected by the flow sensor attached to the sample units, in comparison with the data from the event based sampling, the overall volume of pollutants removed by systems per event was calculated. The intensity of events was calculated by examining the volume of discharge against the time it was delivered at monitoring sites

During the sampling period, it was not possible to capture all rainfall events, due to factors such as; equipment malfunction and/or a marginal event failing to trigger monitoring equipment, or just missing events due to unpredictability of the weather. Using the log from the flow sensor installed at monitoring sites, the total number of events during the entire period that equipment was installed was observed. In turn, using the average volume of pollutants removed for monitored events, an estimate of pollutant discharge could then be applied to unmonitored events. Using this information, a further estimate could be made about the total discharge of pollutants over the monitoring period. The known costs of treating each discharge can then be applied to the whole study area, (based on the number of known SWD) to give a cost for treating all known discharges. The potential benefit of installing the systems used to all outfalls in the study area can also be calculated using figures on the volumes of pollutants removed.

3.12 Summary

This Chapter details the methodological approach used in this study. It details the stages in the development of the project, firstly to monitor, analyse and understand the pollutants in the river, followed by the selection, installation, monitoring and analysis of treatment systems (PTS). The next Chapter presents the results of both the phases of data collection covered in this methodology; this is again divided into two sections to reflect the two separate bodies of data.

Chapter 4 – Results and Discussion

4 Chapter 4 – Results and Discussion

Phase 1 - River Monitoring Results and Analysis

The methodology (section 3.2) outlines the sample programme that was completed in the main river. The samples and subsequently generated data from this programme is presented in this Chapter. As well as the analysis of collected sample data this Chapter includes analysis of supporting discharge and rainfall data sets. The presentation and analysis of these results is split in to 5 sub-sections:

Section 4.1 - The central Wigan discharge data,
Section 4.2 - Long term and seasonal analysis of rainfall data,
Section 4.3 - Principle Component Analysis (PCA) of water quality data,
Section 4.4 - Water sampling,
Section 4.5 - Comparison between the study data and the EA's Source Apportionment Geographical Information System (SAGIS).
Section 4.6 - A Summary of Findings

4.1 Analysis of Central Wigan Discharge Data

To put the discharge conditions experienced during the sampling effort in context with the long term flow conditions of the Douglas, discharge data, from the central Wigan Gauging Station, was analysed for the period in which sampling was undertaken and compared to historical data. 34 years of Daily Mean Discharge data are available for the station. However it should be noted that the station equipment has been upgraded during this period, so data has been collated together to form 3 separate records corresponding to the periods 1978-1994, 1994-1999 and 2000-2012. The records for the two older stations are less complete than the 2000-12 records, so this last section of data is used for analysis – see Figure 46.

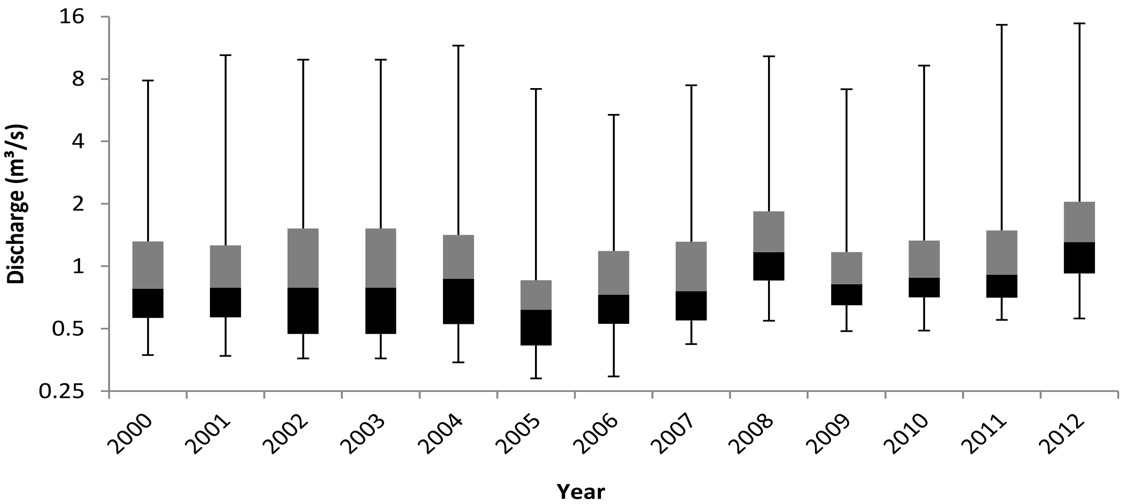


Figure 46 - Boxplot of annual discharge at central Wigan gauging station

The peak daily mean discharge recorded by the central Wigan Douglas gauging station (GR, SD 58523 06049, shown on Figure 9, section 3.4) during the project study period was 14.8m³/s. This was the highest daily mean discharge that was recorded in 2012, greater than in both 2011 and 2010, which had a maximum daily mean discharge of 14.6m³/s and 9.3m³/s respectively. It should be noted that the Wigan Flood Alleviation Scheme was completed upstream of where the gauging station is located on 01/03/2011. Designed to hold back discharges above approximately 20m³/s the scheme also includes a trash screen which could lead to lower discharges being restricted in the event this becomes blocked. 2012 saw greater mean discharges than the period 2000-2011 and the sampling conducted under the project saw the full range of these conditions in terms of fluctuations in discharge (shown in Figure 46).

Since 2003 the gauging station has also collected discharge values at 15 minute intervals and this much higher density data set provides a greater appreciation of the range of discharges experienced by the station. To ascertain the discharge conditions using this data set, a flow duration curve was generated using the data from 2003-12. This is shown as the solid black line in Figure 47 and it can be observed that discharges exceeding 2m³/s only occur for 13-15% of the collection period.

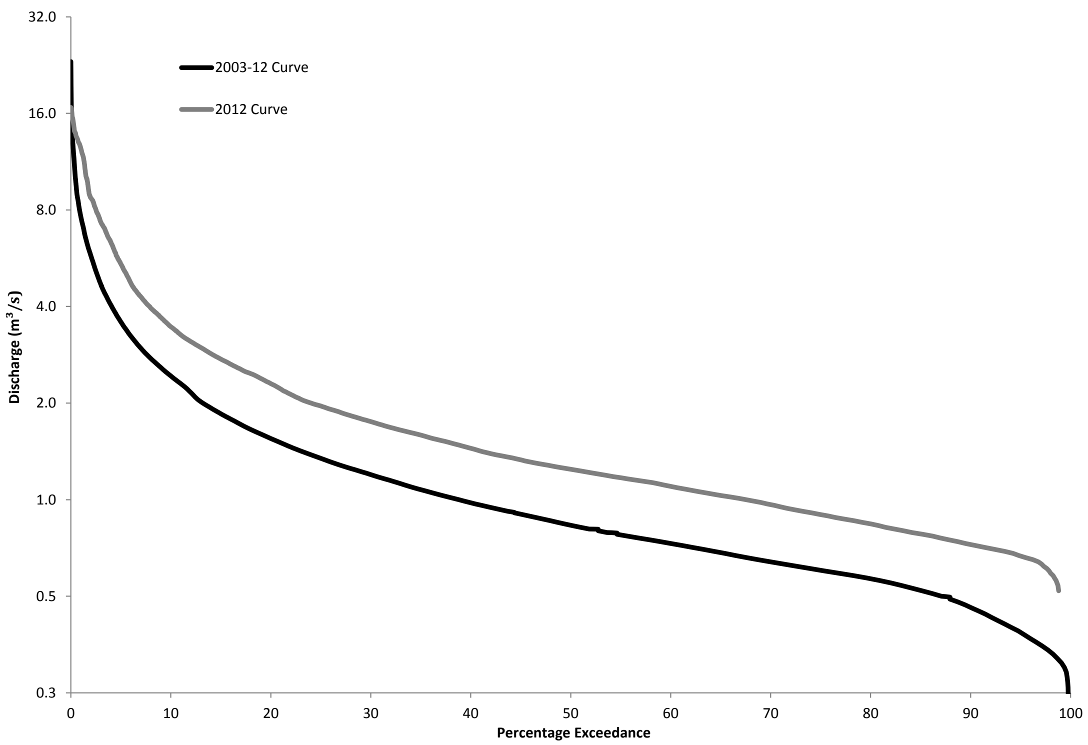


Figure 47 - River Douglas Flow Duration Curve

To provide a clear comparison between discharges experienced during the sampling period, a further curve was calculated using 15 minute interval data for 2012. This is shown as the lighter grey line in Figure 47. It can be observed that the discharges during 2012 were on average 0.2-0.3m^3/s higher than the long term discharge curve. Base discharge was taken at the 90th percentile. For the longer term data set this was 0.5m^3/s, however when considering the elevated discharge during 2012 values of greater than 0.8m^3/s were taken to be above base discharge. In sections 4.4.1-6 when comparing the discharge data with the river sampling results, it should be noted that most of the sampling points were in a different geographical location from the central Wigan gauging station. For those sample locations close to central Wigan the readings are indicative, but for those sample locations which are further up and downstream, discharge values from the central gauging station will not be as representative.

4.2 Long Term and Seasonal Analysis of Rainfall in Wigan Study Area

In order to understand the rainfall conditions experienced during the study period, the rainfall for the duration of the sampling period was analysed. Data was obtained from the EA's national rain gauge network, from three gauges located in the study area; these are at Lower Rivington (SD 63141 12113) Upholland Dean Wood (SD 52501 05900) and Billinge Hill (SD 52263 01791) (location of gauges are shown in Figure 9, section 3.4). Data from these gauges is available in two forms at each site, either as daily total rainfall values, or in the form of totals per 15 minute intervals.

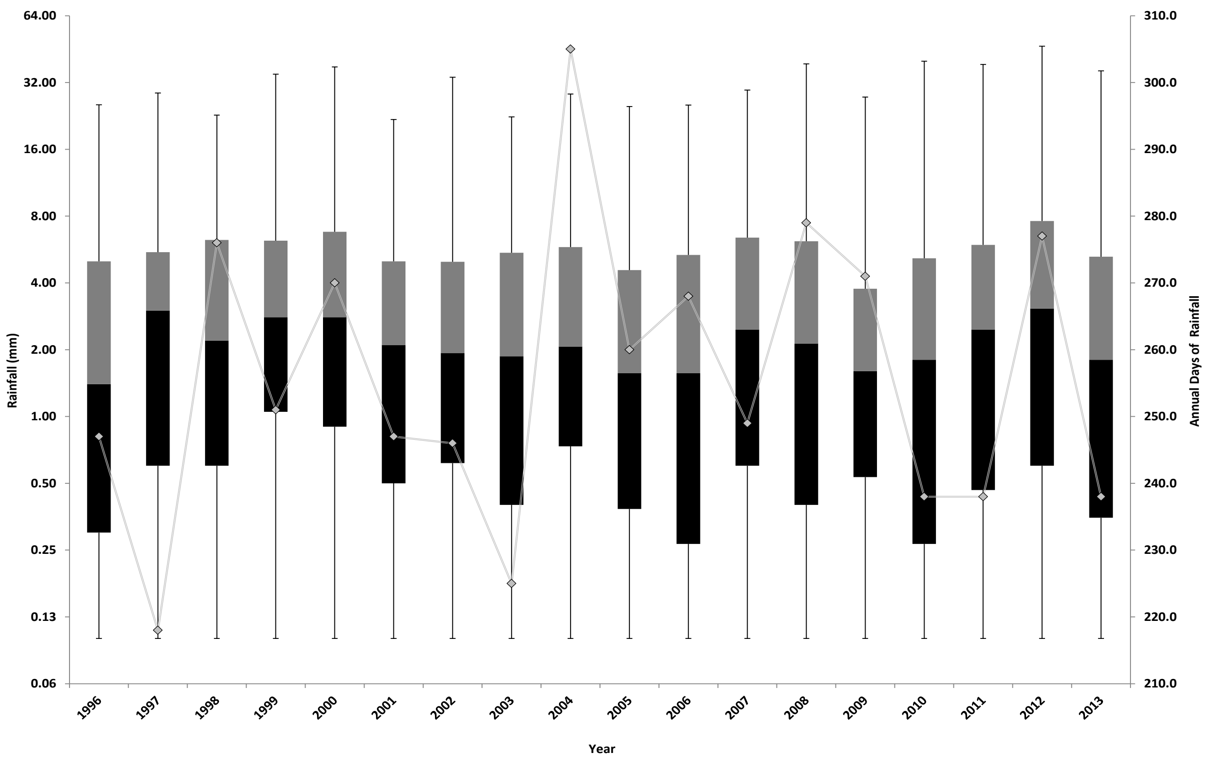


Figure 48 - Wigan Annual Average Rainfall 1996-13

Records are available from 1996 to the present for Lower Rivington and Billinge but only from 2002 for Upholland Dean Wood. Using the daily total values from Lower Rivington and Billinge for 1996 to 2013 and from all three gauges for the period 2002 to 2013 a range of average values for each year was generated. From this average data, the minimum, 25th, 50th and 75th percentiles as well maximum values were calculated for each year and this was plotted as a box plot (Figure 48). Only days when rainfall was recorded are included in the analysis, so this value is plotted via a line on the secondary axis. Due to the wide range of values a log scale was used, it can be seen that that average annual rainfall values fluctuate significantly from year to year. Between 2009 and 2012 average rainfall increased until 2012, which from data available is wettest year on record, with a total annual rainfall of 1438mm and it also contained the highest daily total rainfall on record at 46.6mm. This is also complimented by national figures which indicate the same tread.

In addition to the daily total rainfall, the 15 minute data set was also used to analyse long term trends in rainfall. Similar to the equivalent discharge data set, this is more detailed and gives a much more accurate measure of the range of rainfall experienced. The total annual volume was calculated and compared to the duration of time that rainfall occurred, so determining the average ‘intensity’ of rainfall. Figure 49 shows these values plotted for the period 1996 to 2013 and this confirms 2012 as the wettest year on record. However it also shows 2012 with the highest total duration of annual rainfall recorded, (for period 1996-2013) with over 36 days of total rainfall across the whole year. So while 2012 was a particularly wet year the average rainfall intensity was lower than, for example, 2004 or 2008 where comparably more rain fell over short periods of time.

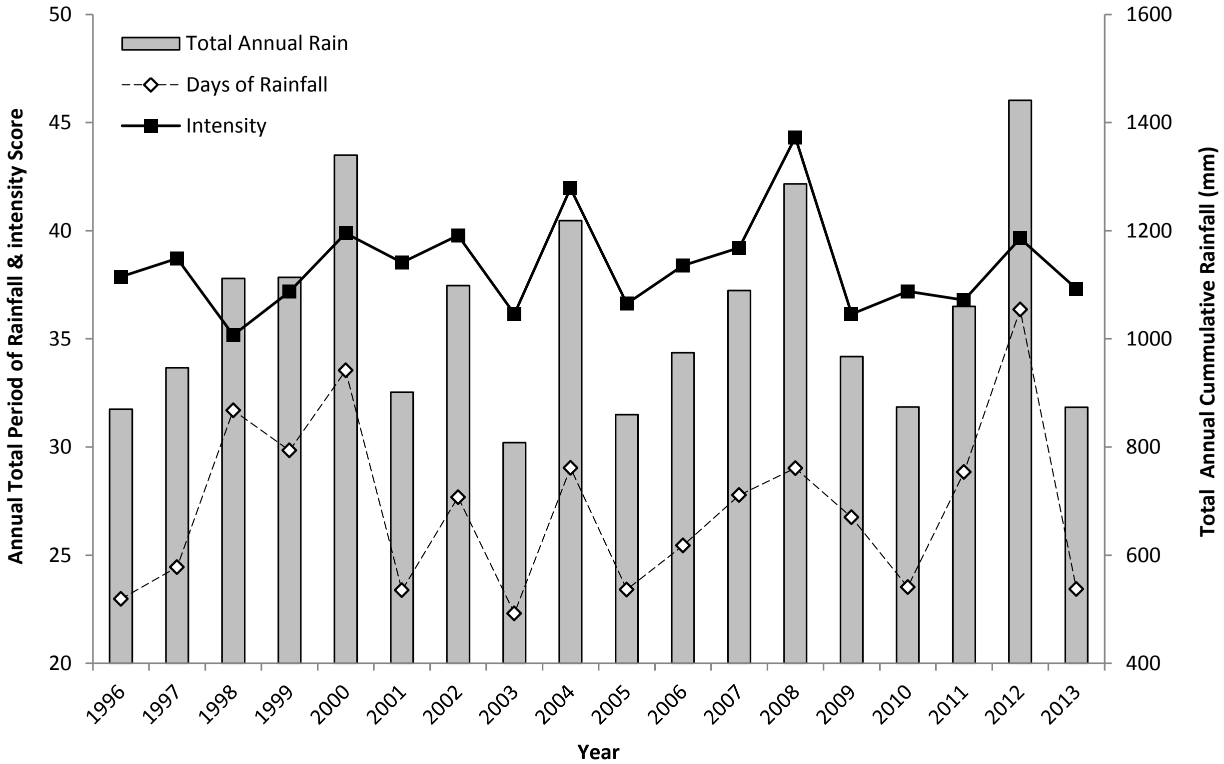


Figure 49 - Average Annual Rainfall Volume, Duration and Intensity (1996-2013)

As well as the long term trend, the seasonal variation in rainfall volume, duration and intensity was observed. Figure 50 indicates that on average July, August and September experienced a comparatively larger volume of rainfall, over a much shorter duration resulting in a much greater intensity of rainfall during these months. In contrast, on average, October, November and December are the wettest months with an increase of between 15-20mm in rainfall. However they also experience the longest duration over which rainfall is experienced resulting in a lower intensity of rainfall. It is important to observe that the data collection period of this study was in 2012, the wettest year on record and during July, August and September which are typically the months which receive the most intense rainfall. There are two effects this could have on river sampling results. Firstly the increased rainfall will result in increased volumes of runoff and its associated pollutants meaning higher concentrations of pollutants in samples. Secondly, and in contrast, after prolonged wet weather runoff will remove the majority of pollutants built up during antecedent periods and in combination with increased river discharge resulting in dilution of pollutants, observed overall concentrations may be reduced.

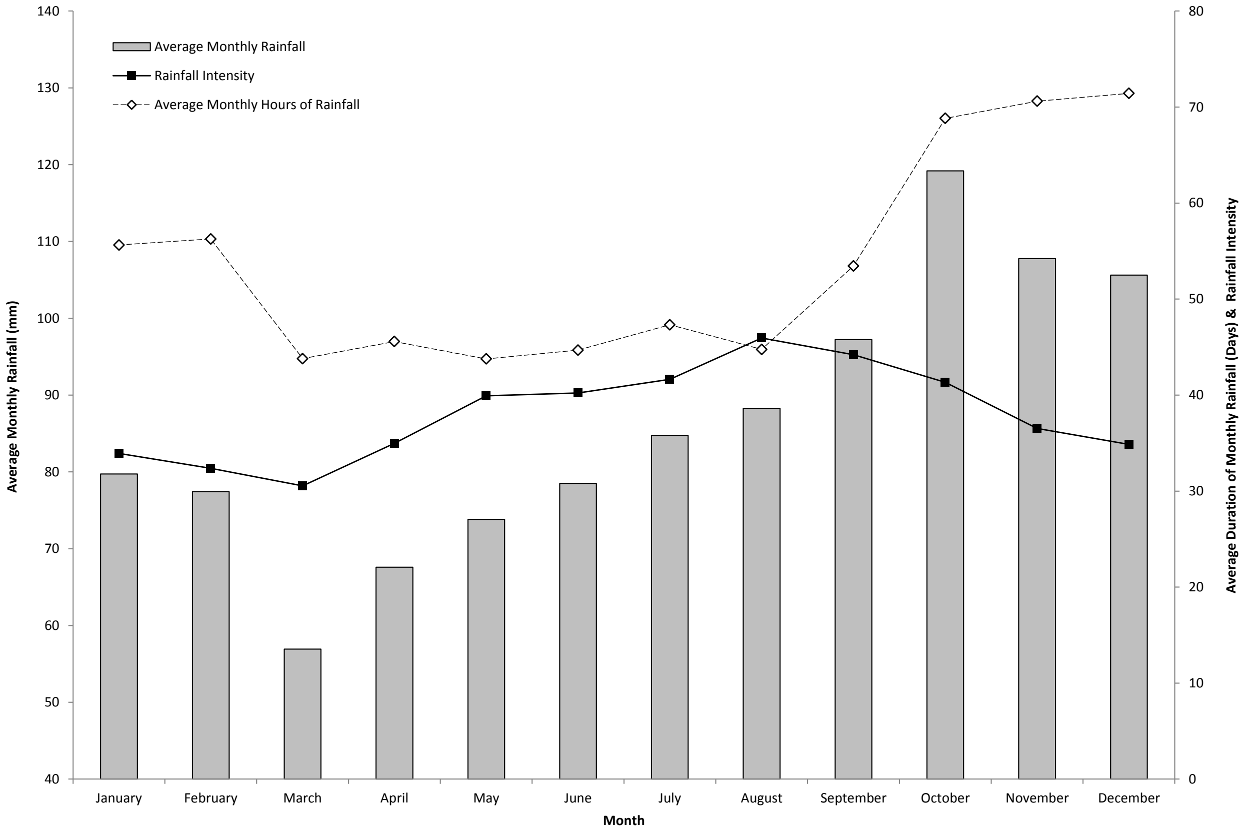


Figure 50 - Average Monthly Rainfall Volume, Duration and Intensity (1996-2013)

Figure 51 shows the same values displayed in Figure 50 using the same data but only that from the 2012 period. This shows that the average monthly rainfall throughout the year fluctuated significantly. July experienced more than a 70% increase in rainfall over the average expected. However the duration of rainfall also increased by more than 70% so the intensity of rainfall was fairly typical. August however experienced the usual rainfall volume in only approximately 85% of the usual duration, resulting in very intense rainfall during that month. September experienced over twice the typical rainfall but the duration it was experienced over was also much higher resulting in only slightly increased rainfall intensity. The conditions then in which samples were collected saw periods of unusually wet and intense rainfall.

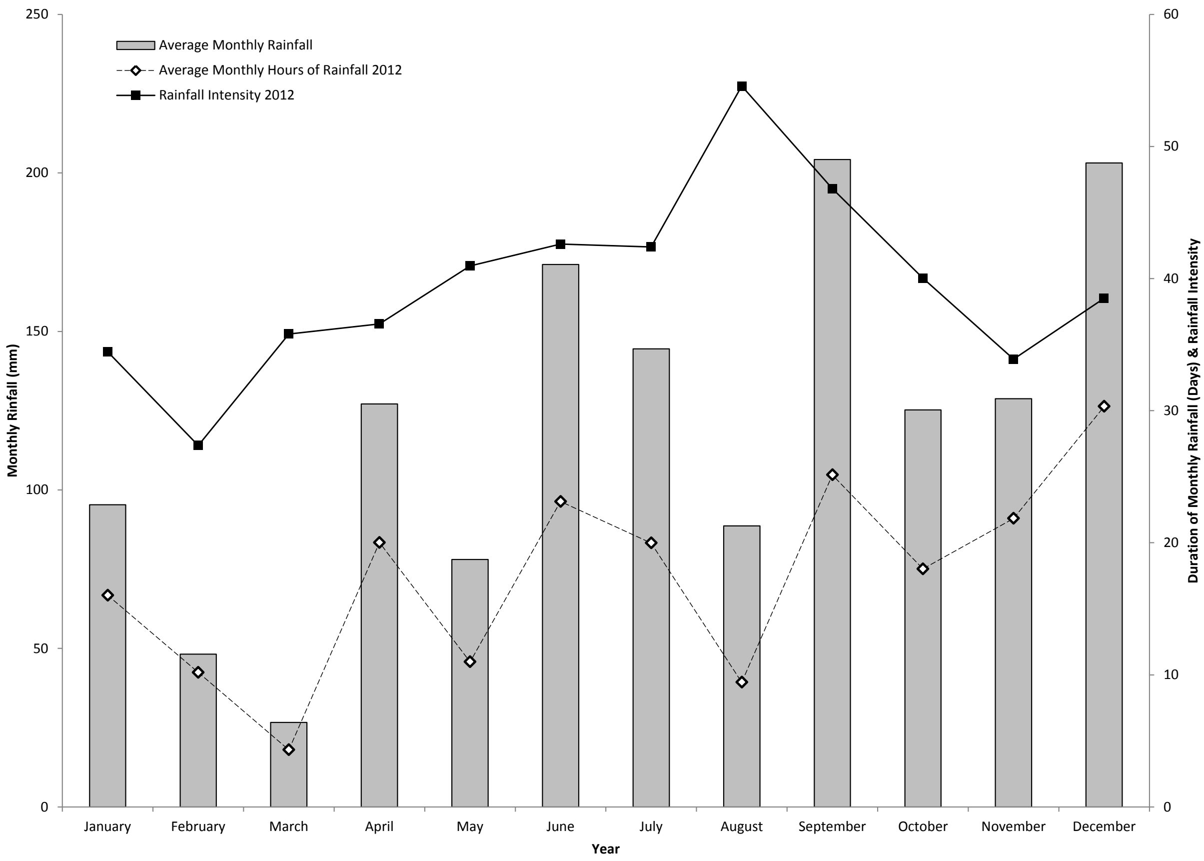


Figure 51 - Average Monthly Rainfall Volume, Duration and Intensity in 2012

4.3 Analysis of River Sampling Results

This section presents the results of samples collected between, the 19th of July and the 31st of October 2012, across the study area at the points indicated on Figure 9 (displayed in section 3.4). Following the relationships identified in the literature review and the linking of WQV (WQV) through PCA, the list of WQV identified in the methodology (Figure 12) have been divided into a series of groups which are explored in six separate sub-sections. These are:-

- Nutrients,
- Faecal Indicator Organisms (FIOs),
- Heavy Metals and TSS,
- Polyaromatic Hydrocarbons (PAHs),
- Halogenated Solvents,
- Other Indicators.

These groups represent a selection from the original 38 WQV. As stated in the methodology (section 3.4) not all have been analysed in detail. Each of the six groups contains a series of different sub-analyses for each WQV. These are:-

Effect of Discharge on Pollutant and Comparison with Regulatory Standards

To ascertain the effect of discharge on sample concentration, several pieces of analysis were completed. Firstly a T-test was used to establish a statistical difference between samples taken at high and low discharges. For this the samples were divided into two groups, namely those taken at high and at low discharge. To make this subdivision the base flow was characterised by producing a flow duration curve generated from the discharge data (section 4.1) and this established that river base flows lie between 0.5-0.8m^3/s. This was further correlated with rainfall values, i.e., when samples taken above these discharge levels or when rainfall was observed within prior to sampling, then samples were considered to be taken under high discharge conditions. Conversely those taken at or below base flow or without prior rainfall were considered to be taken under low discharge. Using these two groups, an F-test (which is used to identify equal or unequal variance between samples) was used, this allowed the appropriate T-Test (assuming equal or unequal variance) to be completed, determining if there was a statistically significant difference between groups.

Secondly a comparison was made between any relevant standards for the variables covered by the section. For example the technique for calculating averages using the EQS classification method (detailed in section 2.1.1.3) was used to calculate comparable values to compare with standards for the WFD. Finally using the data split into high and low groups for each site, the range, 25th, 50th (mean) and 75th percentile were calculated for sample concentration under high and low discharge conditions for each site. This was then plotted as a box plot.

Pollutant Fluctuation during Storm Events

Using the available 15 minute interval data for rainfall and discharge, several examples of different sized storm events were plotted against the concentrations observed in collected samples. Examples of low, medium and high discharge events were selected for analysis and characterised by the duration of time the river remained above its typical base discharge (classified by the flow duration curve in section 4.1)

Downstream Pollutant Fluctuation

Using all samples collected for each variable, averages were generated for each site. This was displayed as a simple graph with sample locations plotted at representative distances to each other allowing observation of the downstream trend in pollutant concentration.

Table 5 - Number of Samples Collected at Sites

Main Channel Sites	**N***	**Tributary Sites**	**N***
Red Rock Bridge	10	Bradley Brook	10
Chorley Road PS	10	Yellow Brook	10
Downstream of Yellow Brook	10	Clarrington Brook	4
Coppull Lane	9	Poolstock Brook	4
Downstream of Great Acre CSO	9	Smithy Brook	10
Upstream of Scholes Weir	10	Sled Brook	4
Swan Meadow Lane	9	Barley Brook	10
A49 Road Bridge	10	Close Brook	10
Pemberton Screens	9	Ackhurst Brook	8
Scott Lane Bridge / Martland Bridge	9	Dean Brook	9
Downstream of Ackhurst Brook	9	Calico Brook	9
Gathurst Bridge	9		
Downstream M6	9		
Appley Bridge	9		
Total	134		84

Not all of these pieces of analysis have been completed on all variables because when not elevated as a result of high discharge or observed in high concentrations, the contribution of a variable to diffuse pollution was considered to be less important and therefore was not analysed further. Overall 222 water samples were collected under this study. This consists of 134 collected at 14 separate sites along the main river and 84 collected from 11 tributaries. Table 6 shows the number of samples collected at each site.

4.3.1 Nutrient Concentrations at Sample Sites

As identified in the literature review (section 2.3.4.1) two nutrients are considered to contribute most significantly to river pollution. These are Phosphorous (reported here as Orthophosphate) and Nitrogen (reported here as Ammonia). As explained several different pieces of analysis have been conducted on each of these pollutants.

4.3.1.1 Effect of discharge on Nutrients and Comparison with Regulatory Standards

The classification of different water quality parameters is covered in the literature review (section 2.1.3). Using this method all of the sample sites used on the project were classified as either type 5 (Ammonia), or type 4n (for Phosphate). Once this was determined the values for a site to achieve 'good' status are listed in corresponding tables which vary between different 'types' of site. Table 7 shows the relevant values for sites to achieve 'good' status for both Orthophosphate and Ammonia. The Table also contains mean values for samples recorded at high and low discharges and a T-Test value determining if there is a statistically significant difference between these values.

Table 6 - Orthophosphate and Ammonia T-Test Values

Determinant	HD Mean	LD Mean	T-Test P Value	WFD EQS (Good)
Orthophosphate (mg/l)	0.194	0.176	0.0924	0.12 (mg/l)
Ammonia (as N) (mg/l)	0.213	0.070	6.869×10^{-23}	1.1 (mg/l)

Table 7 shows that Orthophosphate values taken during different discharge conditions do not demonstrate a statistically significant difference from each other, although neither mean value recorded is below the 0.12 mg/l limit required for the river to be classified as 'good' quality. This indicates that surface wash off as a result of increased rainfall is not the most significant factor contributing to observed concentrations of Orthophosphate. The fact that the mean values do not change significantly as a result of increased discharge also indicates that a consistent point discharge is a likely cause of observed Orthophosphate levels.

Ammonia demonstrates the opposite of that observed for Orthophosphate. The high and low mean values do display a statistically significant difference from each other. However Ammonia also displays the opposite of Orthophosphate in the respect that both mean values are considerably less than the concentration required to achieve good status under the WFD. As mean values fluctuate with increased rainfall and discharge this indicates that intermittent diffuse sources are contributing to Ammonia pollution. However concentrations observed are insubstantial with the mean high flow being less than a quarter of the standard for a site to achieve good status.

4.3.1.2 Downstream Nutrient Fluctuation

Figure 53 shows a graph of the fluctuation of average Orthophosphate concentrations across the study area, as can be seen from the graph, there is a strong downward trend in average sample concentrations. Other studies (Davies and Neal, 2007) have observed losses from Orthophosphate downstream of discharges, as it is readily absorbed into the water column. It is clear from Figure 53 that there is a discharge or pollutant source located upstream of the study area, which is leading to high Orthophosphate concentrations in samples.

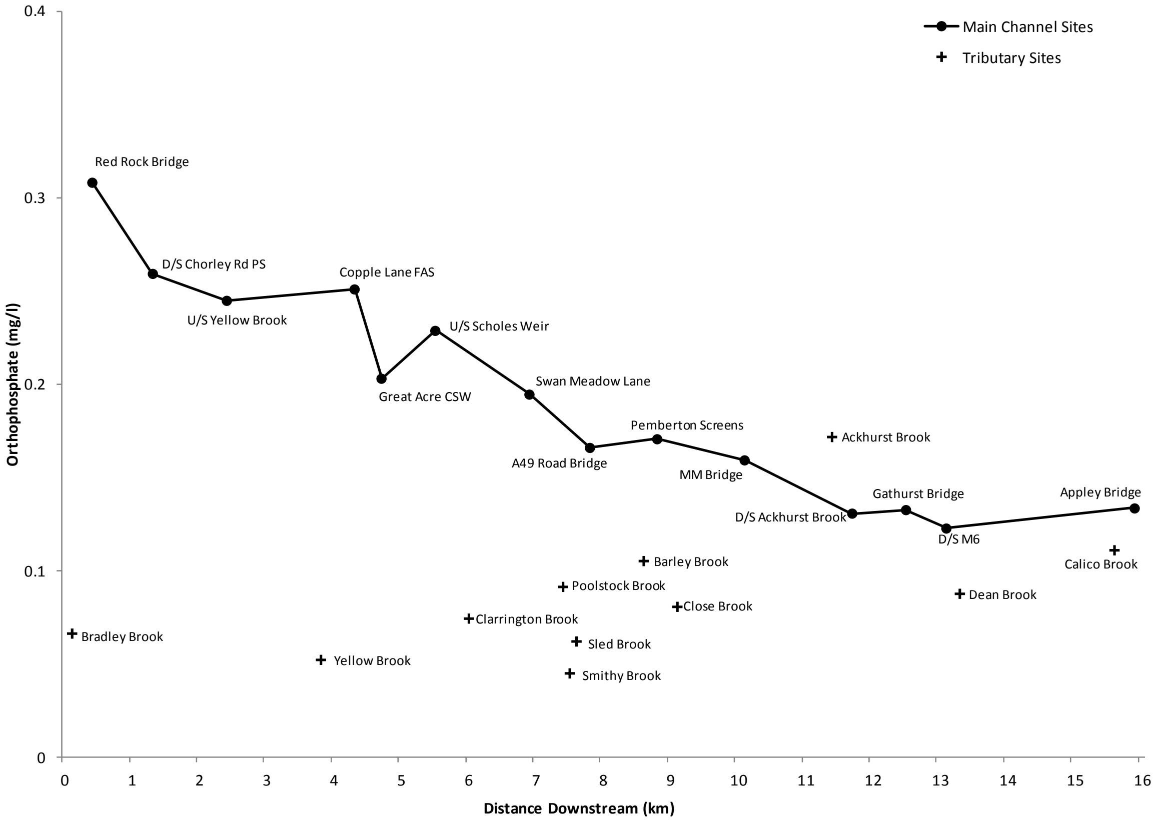


Figure 52 - Average Orthophosphate Change Downstream

The contribution from tributaries is not significant in comparison with the input above the study area, with all but Ackhurst Brook having concentrations lower than all sample locations in the main river. Contributions from the Ackhurst and Calico Brooks are the most significant, both of which are located downstream of Wigan.

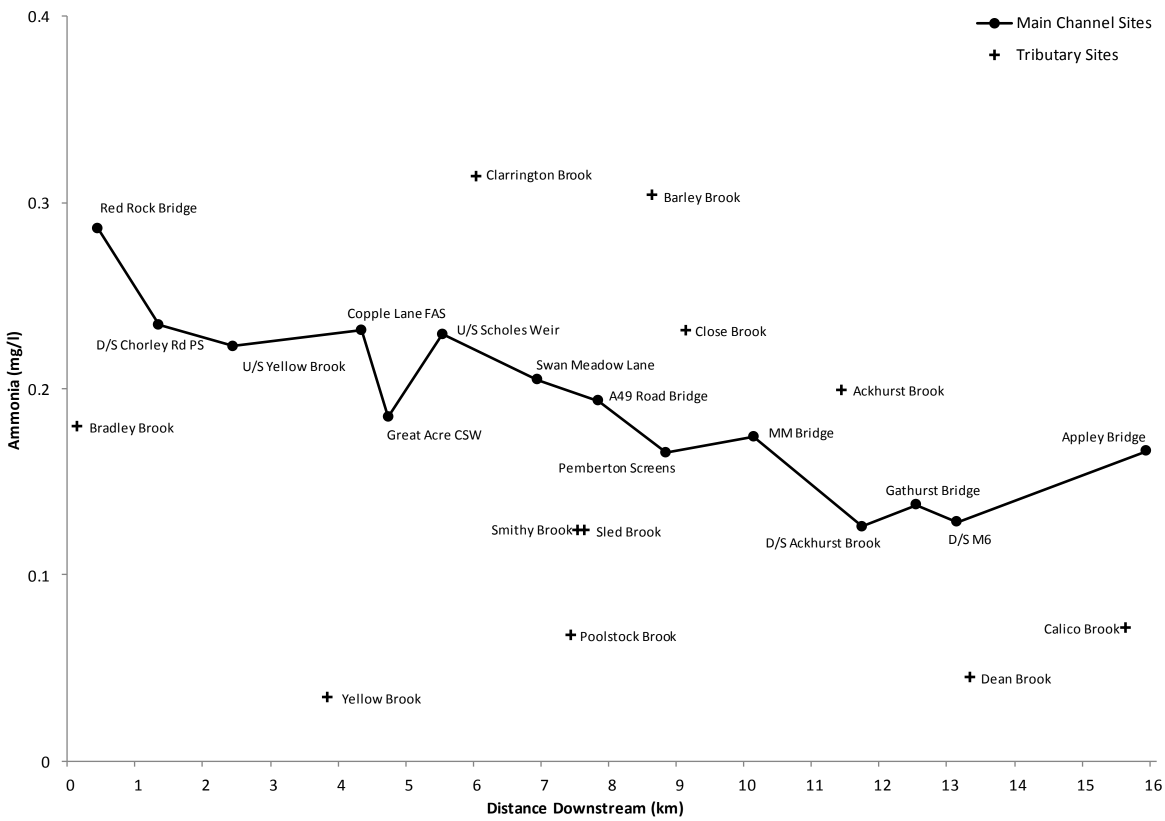


Figure 53 - Average Ammonia Change Downstream

Figure 54 shows the same graphical information for Ammonia. Again average values show a reduction with distance downstream and with average concentrations showing a strong correlation with Orthophosphate indicting they are coming from similar sources. Unlike Orthophosphate, however several tributary sites recorded higher concentrations of Ammonia than those observed in the main channel. This shows pollutant inputs to these tributaries is likely to be a result of domestic cross connections or other sewerage contamination. These additional inputs could explain why main channel concentrations do not fall as quickly as the values for Orthophosphate. Both Phosphate and Nitrogen pollution are very important with respect to river pollution (section 2.3.4.1). However from the analysis of available sample data it is clear that over the area studied nutrient pollution is dominated by a consistent source from above the study area. This makes it difficult to observe the contribution of sources with less significant and intermittent inputs such as urban wash off because main channel concentrations are consistently high. Orthophosphate concentrations do not show a significant response to rainfall and high discharge, and Ammonia concentrations are consistently low, therefore further analysis of these parameters has not be conducted.

When considering the average values of both nutrients at the highest upstream sample point (Red Rock) they are consistently high, and fall on average by approximately 50% over the 16km length of the study area. Based on the known location of Horwich WWTWs approximately 8.5km above the study area (indicated on Figure 11 in the methodology) it is considered that the consistently observed nutrient inputs are as a result of the discharge from this works. The works takes a daily water sample for analysis with respect to nutrients; the collected samples from this location covering the same period in which samples were collected in the project are indicated in Table 8

Table 7 – Average nutrient concentrations in Horwich WWTWs discharge

	Ammonia (mg/l)	Phosphorus (mg/l)
No of Samples taken over Study Period	46	45
Mean Values	0.1510	1.4887
Maximum Values	0.7100	3.5400
Minimum Values	0.0600	0.4700

Average Phosphate values are higher than the average concentrations seen at the upstream end of the study area, indicating that the WWTWs is the most likely cause for the consistently high Phosphate values. Also evident is the rapid reduction in average concentrations, with Phosphate falling from an average of 1.49mg/l at the WWTWs discharge, to 0.343mg/l in high and 0.284mg/l in low discharge conditions at the Red Rock sample point. This is as a result of dilution and losses to the watercourse environment. Conversely Ammonia concentrations in the River were greater at the sample site in high discharge, indicating that the WWTWs was not the main source of Ammonia seen in samples. Again there was a sharp reduction in concentrations from the WWTWs outfall to the upstream sample sites, with average values reducing from 0.1510mg/l to 0.363mg/l in high and 0.046mg/l in low discharge. Further information on Horwich WWTWs is included as Appendix VIII.

4.3.2 FIO Concentrations at Sample Sites

FIO pollution is explored in the literature review (section 2.3.4.1) which identifies that faecal bacteria are not only a pollutant in their own right but a useful indicator of sewerage contamination and its associated negative effects on a river water quality. Crowther et al. (2002) observed when analysing FIO concentrations that the distribution of values demonstrated a more normal distribution when transformed to Log10. Therefore all statistical analysis of FIOs was completed using Log10 transformed values, except when comparing with untransformed values in legislation or literature.

4.3.2.1 Effect of discharge on FIOs and Comparison with Regulatory Standards

As with the nutrient parameters, *E.coli* and IE results were sub-divided using the same method into two groups of samples, namely those which were taken at high and low discharges. These groups were again compared using a T-Test, the results of which are displayed in Table 8. Both FIOs demonstrate a statistically significant difference between mean values by returning low P values. There was also a notable difference between the two faecal indicators, with *E.coli* being present in high flows between two to five times, and in lower flows two to six times greater concentrations than IE. To avoid duplication of results, further analysis is completed on *E.coli* alone as it always displays values that are between two and six times higher than IE.

Table 8 - FIO T-Test Values

Determinant	HD Mean	LD Mean	T-Test P Value
Escherichia coli (*E.coli*)	4.101	3.370	1.186×10^{-11}
Intestinal Enterococci (IE)	3.628	2.777	1.930×10^{-14}

The literature review (section 2.1.2) also identified that for the purposes of classification under the RBWD the two primary FIOs used are *E.coli* and IE and Table 9 shows the limits set in the RBWD for both these FIOs. Table 9 also gives comparable values calculated from collected samples using the method of calculation stated in the RBWD. The 90 and 95 percentile values were generated from the whole data set as well as the values recorded for low discharges. Even when comparing values collected during low discharge periods, contamination is still approximately 3 times higher than the standard set as a satisfactory level under the RBWD legislation.

As the data collected for this study are intended to explore riverine bacterial contamination in response to increased rainfall, a direct comparison to the standards set in the RBWD should be made with caution. Other factors mitigating against a direct comparison are a shorter sample period to that required to make a full assessment. In the RBWD this is a full bathing season, and not all samples collected in this study were recorded within the British bathing water season (May-September).

Table 9 - Comparison of RBWD and Wigan project values

	Parameter	Excellent Quality*	Good Quality+	Sufficient Quality+	Wigan Values			
					All Values*	Low Discharge Values*	All Values+	Low Discharge Values+
1	Intestinal enterococci (cfu/100 ml)	200	400	330	24677	5708	14405	3472
2	Escherichia coli (cfu/100 ml)	500	1000	900	83437	21676	48880	13270
* Based upon a 95 percentile evaluation / + Based upon a 90 percentile evaluation.								

The values in the RBWD are designed to be used for long term analysis so when looking at FIO concentrations it is also desirable to have a standard to compare values with on an event by event basis. For this purpose 'Investigation and Rectification of Drainage Misconnections' states that "*E.coli* counts higher than 2,000 are recommended to be used as a baseline for further investigation on watercourses" (Water UK and Environment Agency, 2009). This guide is designed to be used to assess Surface Water Outfalls polluting rivers, rather than grading river samples, which should have lower levels of coliforms due to the diluting effect of the main channel. The document also contains a 'Bacteriological Assessment Guide' the purpose of which is to gauge the possible harmful effects from observed concentrations as outlined in Table 10. This shows the level of coliforms and the corresponding description of the level of risk. A coliform reading of over 10,000 indicates 'inadequately treated sewerage levels'. Therefore samples returning values above this limit were considered more significant.

Table 10 - Bacteriological Assessment Guide

Range		Description
1	500	Background levels – watercourse
500	999	Low or intermittent contamination
1000	9999	Evidence of sewage contamination
10000	99999	Inadequately treated sewage levels
100000	1000000	Untreated sewage and health risk potential

Due to the high rainfall during the study period at some sites, it was possible only to take one or two low discharge samples and, as such, it was not possible to calculate a range of values to compare with samples taken in high discharge conditions. Nevertheless these findings corroborate those of Wither and Kay (Kay *et al.*, 2008; Wither *et al.*, 2005) who also found a strong correlation between increased discharge and an increase in faecal coliform concentration. When comparing concentrations observed in the collected samples against those given in Table 11, it can be seen that at all sites there was evidence of inadequately treated sewage and at some sites clearly untreated sewage was in the main river channel which could present a serious potential health risk to people coming into contact with river water. The wide range of concentrations in samples at sites indicates there are multiple sources of FIOs.

Following on from this comparison with the RBWD, further comparison between samples taken in low and high discharge conditions was completed. As explained in section 4.4, high and low values were plotted on a box plot for FIOs, this is shown in Figure 55. It can be seen that ten out of fourteen main channel sites, display a higher interquartile range in samples taken during high discharge conditions. At the remaining four sites, while the interquartile range during elevated discharge is still greater, samples with higher maximum concentrations were recorded in low rather than high discharge. Sites located further upstream show a more pronounced difference between high and low values than those located further downstream.

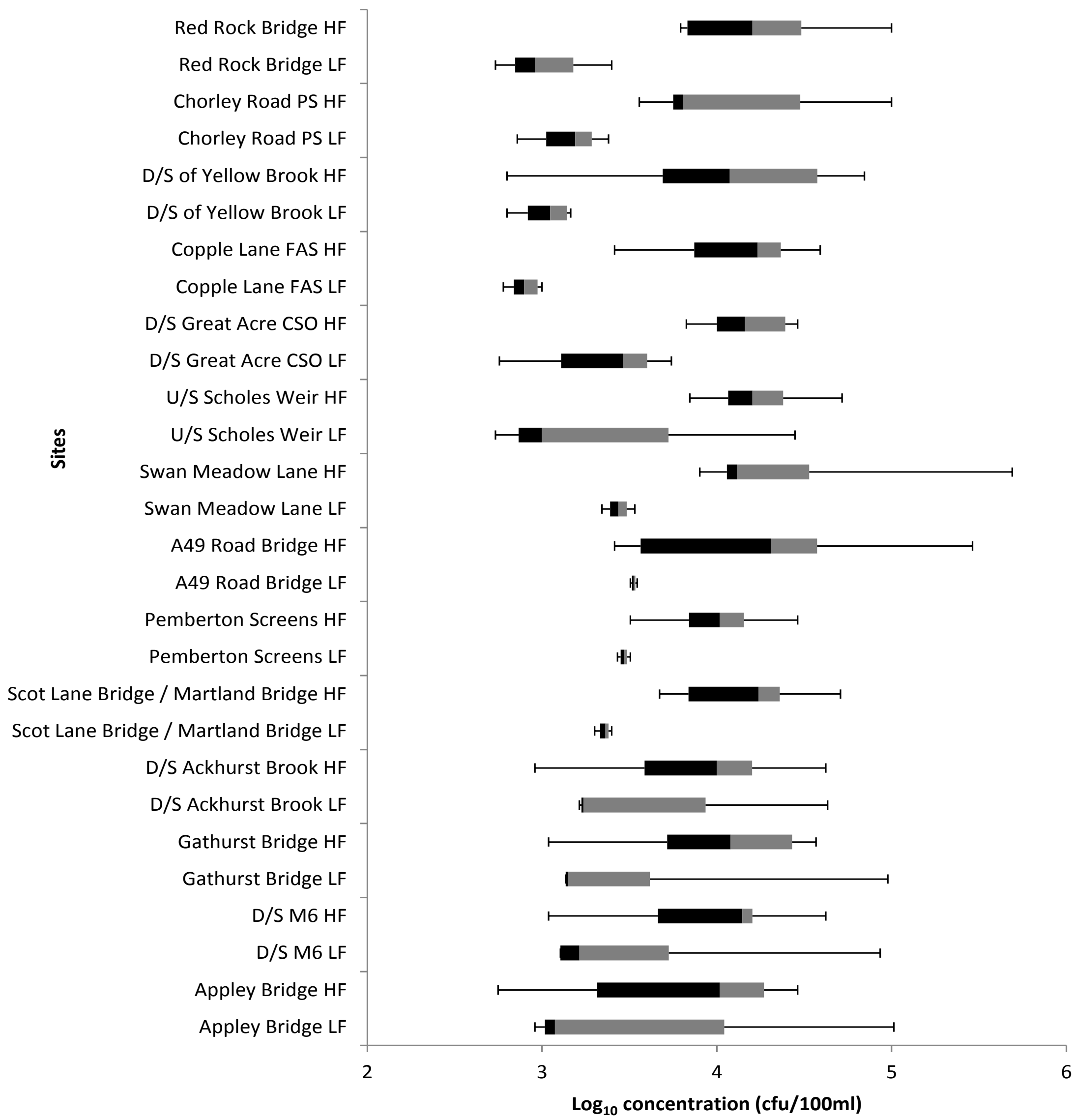


Figure 54 - Range of *E.coli* concentrations at sample sites

4.3.2.2 FIO Fluctuation during Storm Events

As well as the overall trend between discharge and FIOs, the variability of FIOs across rainfall events of varying intensities has also been briefly explored. To directly observe the response of FIOs to the sampled events recorded, *E.coli* values were plotted against the 15 minute data logs of the discharge (from the central Wigan gauging station analysed in section 4.1) and the rainfall (from available rain gauges analysed in section 4.2). As stated in the methodology (section 3.2) sampling was responsive in that samples were taken during or after forecast storm events. Ideally samples should be taken at uniform intervals across the duration of the whole storm, however with unreliable information about the size or extent of storms samples are more evenly distributed across some events than others. The following three Figures, 56-58, show three separate storm events of different scales, which were captured by the sampling.

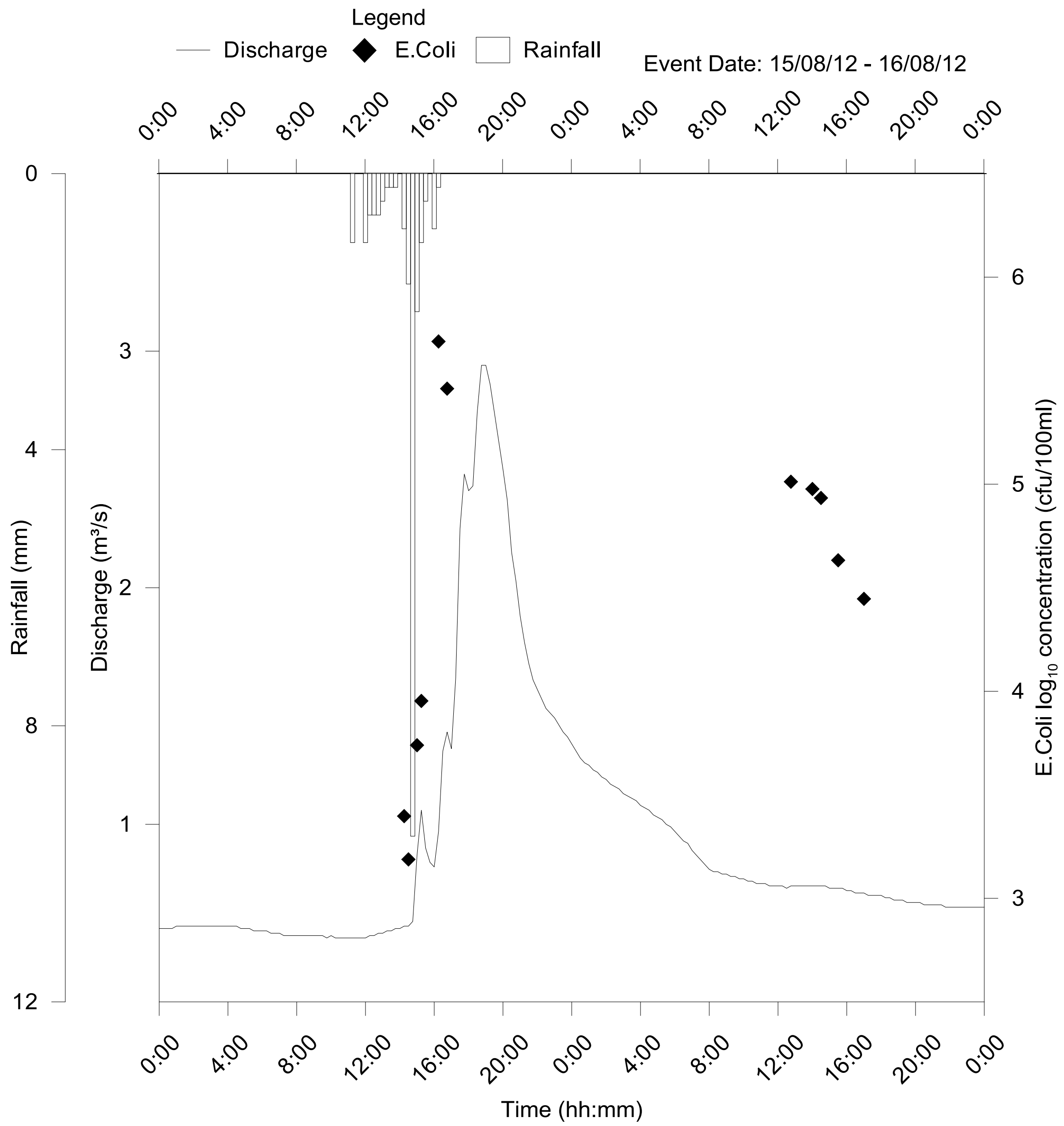


Figure 55 - *E. coli* Concentrations (Low Event)

Figure 56 displays samples collected over a high intensity event with a short duration, where the river's discharge increased only by a small amount over its base flow. It can be seen that a considerable amount of rain fell during a 4 hour period, containing one particularly intense 15 minute period when over 8mm fell. As a result the river discharge increased very rapidly from $0.57m^3/s$ to almost $3m^3/s$. Correspondingly the concentration of *E.coli* in spot samples collected during the event were the highest recorded from all the sampling completed in the project. Samples taken shortly before the river reached peak discharge returned readings of 5.69 (log_{10} cfu) and 5.49 (log_{10} cfu). Following the event's peak discharge, the river flow took approximately 12 hours to return to the normal base flow level. however in samples taken the following day *E.coli* counts, while significantly lower in comparison to the previous day, were still very high, with a further three samples being recorded around 5 (log_{10} cfu). These results indicate that sewerage contamination continued to enter the channel in significant volumes for 12-16 hours after it was first recorded.

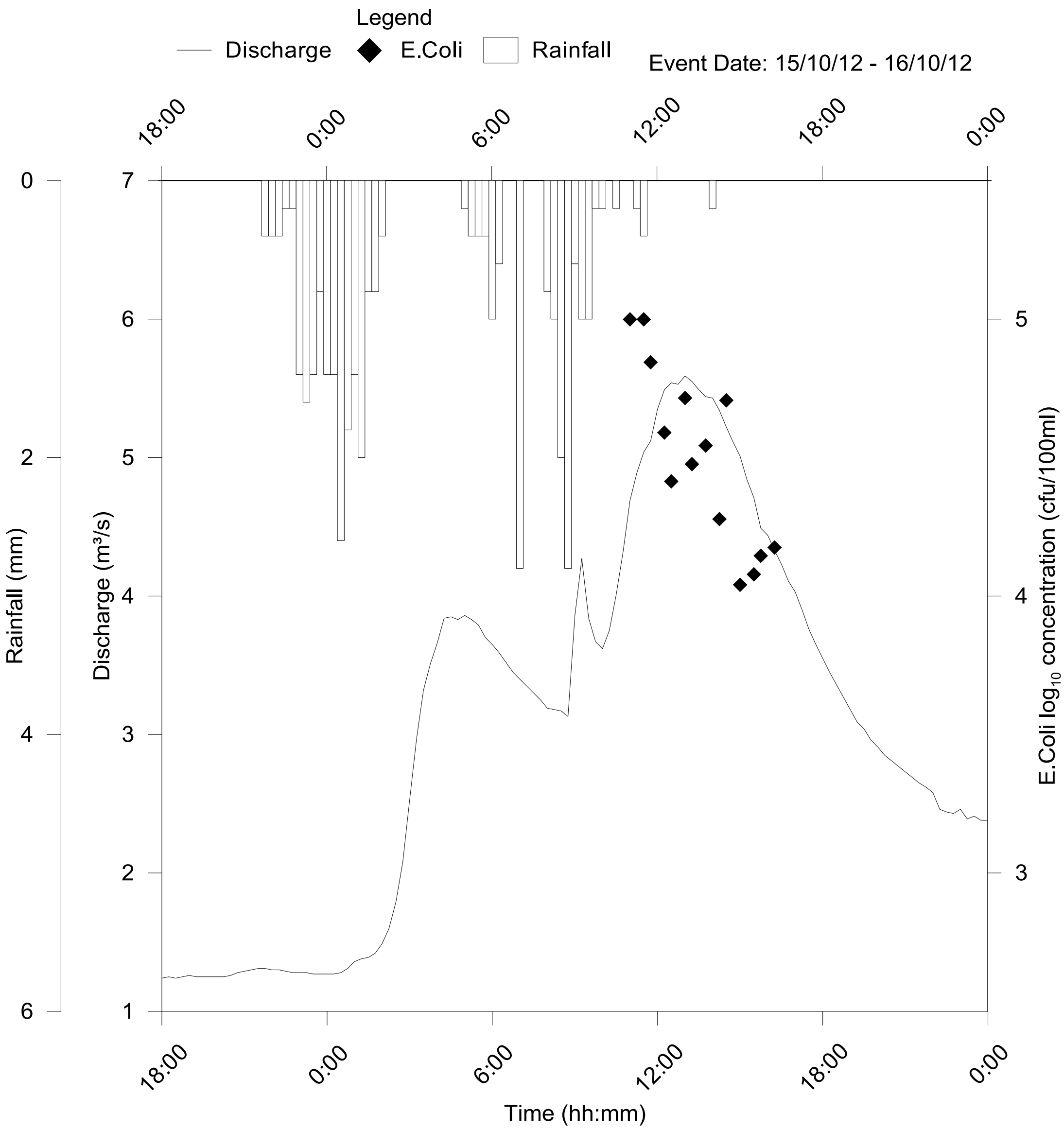


Figure 56 - *E. coli* Concentrations (Medium Event)

Figure 57 shows the results from samples collected during a more significant event in terms of the rivers increase above base flow. The whole event consists of three separate discharge peaks, each following a period of more sustained rainfall of moderate intensity. While rainfall was much less intense that the event recorded in Figure 23, it was more sustained resulting in a prolonged period of higher discharge. Samples started to be collected shortly before the final peak of the event which reached $5.59m^3/s$ at 13:00. Observed concentrations of *E. coli* in samples were initially very high, with 2 samples recorded at 5 $\log_{10}$ cfu/100ml. Subsequent samples collected throughout the day fluctuated, but generally fell as discharge decreased during the afternoon. Sample concentrations begin to fall prior to the final peak of the event so it is likely that previous peaks saw higher concentrations and the higher discharge of the final peak had a diluting effect on samples collected then.

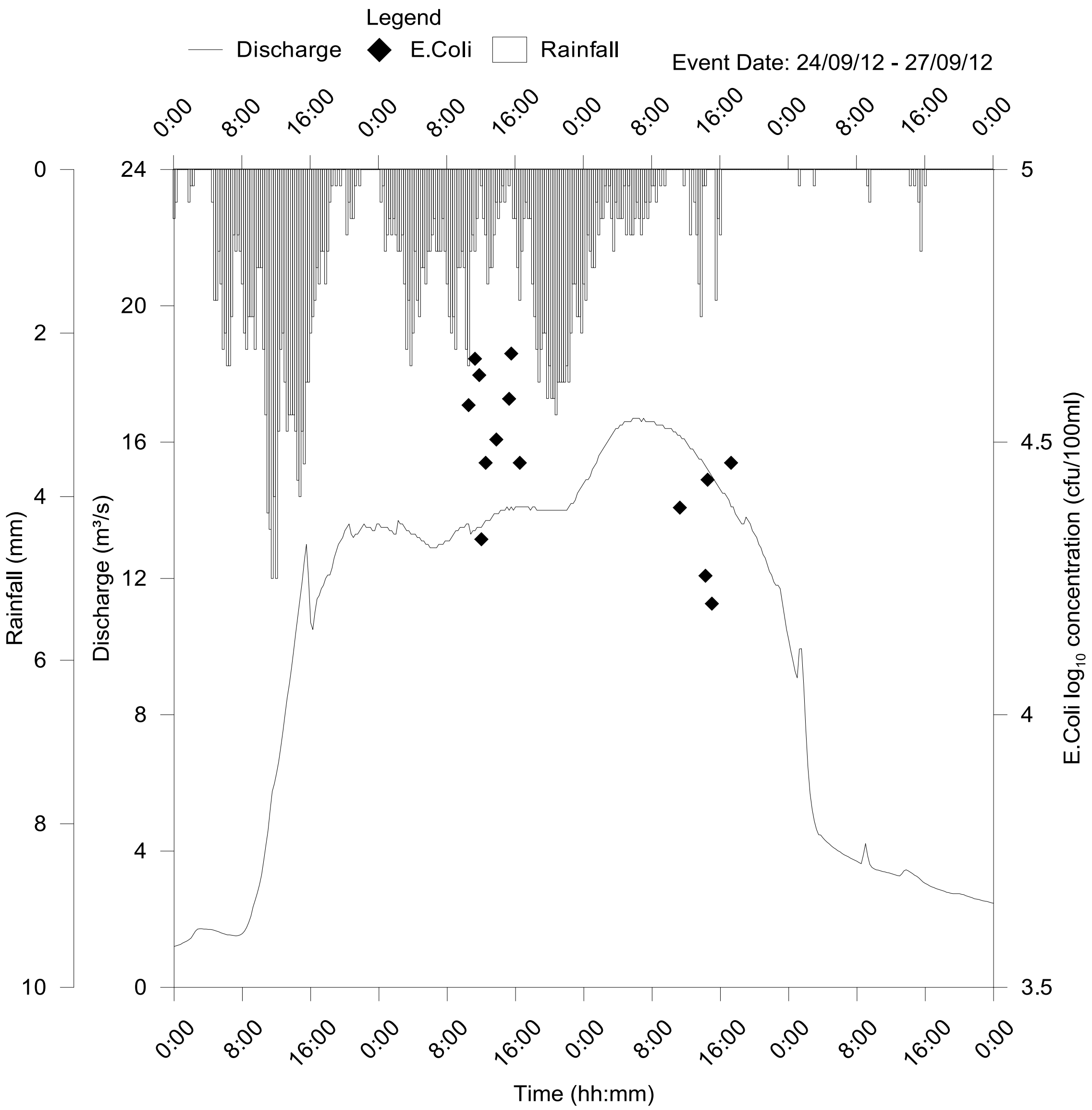


Figure 57 - *E. coli* Concentrations (High Event)

Figure 58 displays the results collected during a large event with a significant duration occurring over 3.5 days. Two of these days made up the highest daily mean discharge observed during the whole study duration. Rainfall across this period was almost uninterrupted for 60 hours, and was of medium to high intensity for prolonged periods. As would be expected, *E.coli* concentrations in all collected samples were elevated above typical base flow concentrations, and remained so during both periods over which samples were taken. However in comparison to samples observed in the events displayed in Figures 57 and 58 sample concentrations were reduced. An explanation for this could be due to the very large volume of extra water in the river levels of E. coli in samples was diluted resulting in comparatively lower concentrations, or that bacterial concentrations had peaked prior to the commencement of sampling.

From this analysis it can also be observed that more rapid increases in discharge (driven by intense rainfall) result in higher *E.coli* cfu in samples, as shown in the results from the second event record described in Figure 57, where *E.coli* was much lower than was recorded in the event shown in Figure 56, even though discharge was much higher. An explanation may be apparent when examining the rapidity of discharge increase, with bigger increases over longer periods resulting in lower *E.coli* counts in samples. Comparatively smaller discharge increases which occur much more rapidly as a result of high intensity rainfall events cause drains and sewers and other FIO sources to take considerable volumes of water in a short period of time, leading to a higher likelihood of discharge of sewerage from CSO valves and bypass of treatment at WWTWs.

4.3.2.3 Downstream FIO Fluctuation

To observe fluctuation across the study area, values for *E.coli* are plotted in Figure 59 using the same method used for Orthophosphate and Ammonia analysis. Samples taken in the main river are shown as a continuous line, with locations where tributaries join shown as separate points. Distance downstream is indicated on the x axis, with sample sites shown at geographically representative distances to each other. Figure 59 shows two separate averages for *E.coli*, with the solid black line denoting average values calculated using all the values collected. It can be seen that total average concentrations fall steadily from Red Rock Bridge to Coppull Lane before starting to increase; thereafter there is a significant increase in the average concentration in the vicinity of Swan Meadow Lane and the A49 road bridge sample sites. It then drops sharply between this point and the Pemberton Screens sample site. It is clear that there is a source causing significant FIO input in the vicinity of Swan Meadow and the A49, which is causing average FIO concentrations in the river to rise from 4.252 ($\log_{10}$ cfu/100ml) to almost 5.

When looking at the results from separate weeks, there were some exceptionally high values recorded during week 5 at each of these central locations. One of the most likely explanations for these very high concentrations is due to a spill from a CSO. There are 8 CSO discharges located within the study area and available spill data from these discharges was cross referenced with sample dates. However there were no known spills that coincided with the week 5 sampling (or any other weeks sampling) On the other hand considering almost all WQV were elevated a CSO spill not recorded by spill monitoring is the most plausible reason for these observations.

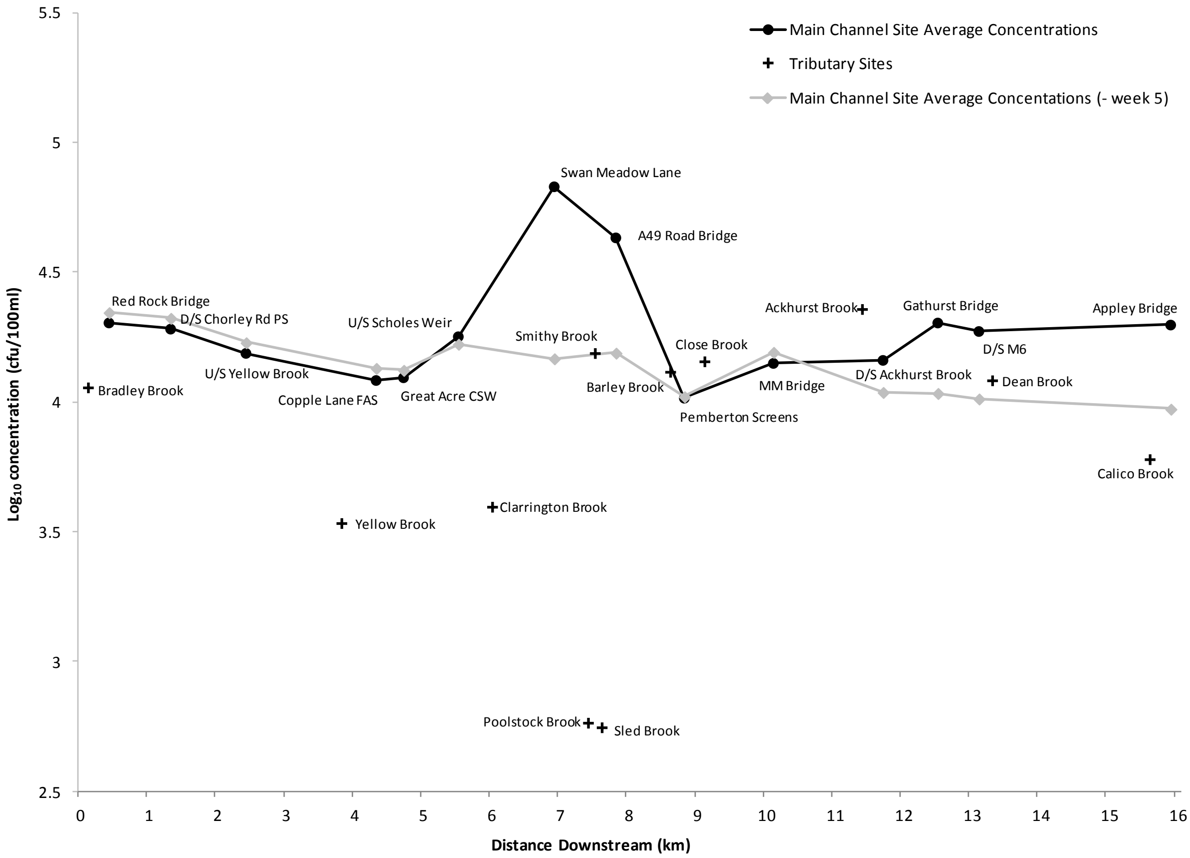


Figure 58 - Average *E.coli* Change Downstream

In order to clarify the overall trend of *E.coli* values the results from week five were removed from the average calculation and a separate average was calculated and the light grey line plotted in Figure 59 denotes the new average. It can be seen that removal of week five data alters the graph considerably. Total average values are dominated by the significant increase in central Wigan, whereas, with these results removed an overall downward trend across the study area can be observed. The higher average values at the top of the study area at Red Rock Bridge can most likely be attributed to the same sources that caused the elevated values of Orthophosphate and Ammonia described in the previous section.

With the high values recorded in week five removed average concentrations at Swan Meadow Lane and the A49 Road Bridge sites are considerably lower. There is still a further fall at Pemberton Screens and a subsequent rise at Martland Mill Bridge. Similarly to Orthophosphate results, the tributary sites also demonstrated low values for *E.coli*, with the exception of Ackhurst Brook, all sites recorded lower average values than main channel sites.

With respect to FIO pollution the most likely sources are from intermittent discharges such as CSOs and domestic cross connections, contributing to the high concentrations seen in central Wigan. On the other hand the more consistent discharges originate from above the catchment most likely from Horwich WWTWs or a package Sewage Treatment Works. As was observed with Orthophosphate and Ammonia there is a gradual downward trend in concentrations (when discounting the high values from week five). The literature review identified that there is less information available on the survival rates of FIOs in river water (section 2.3.4.1), particularly on how bacterial contamination changes with distance downstream from the sources of contamination.

The observations emerging from analysis of Figure 59 indicate a fairly consistent reduction in cfu counts. This is likely to be as a result of spatial and temporal differences causing high mortality rates for microorganisms. On average there was a reduction from 4.345 to 3.973 ($\log_{10}$ cfu/100ml) over a 16 km stretch of river, and this has important implications when conducting FIO studies in riverine environments. For example when sampling is conducted over large geographical area as in whole catchment studies, sample points spaced at wide distances could miss the true scale of local faecal contamination to a riverine environment. Without sampling the considerable number of individual discharges, it is difficult to attribute the pollutants observed in the river to specific discharge locations.

4.3.3 Metals and TSS

The literature review (section 2.3.4.2) identified that metals and suspended solid material are important, in respect of the polluting effects they can have on water courses and bodies. Collected water samples were tested for a total of 9 different metals, a full list of which is given in the methodology (section 3.4). As mentioned in the previous section if results did not show fluctuations or significantly high concentration then those variables have not been fully analysed. For example in this section lead has not be analysed in detail, as from over 130 readings there were only 2 occasions where concentrations increased over the minimum detectable levels. Similarly with nickel, while sample concentrations fluctuated between sites and conditions, they were typically low at around 3µg/l. For six of the nine metals analysed similar observations were made, and therefore further analysis on these metals was not completed.

4.3.3.1 Effect of discharge on Metals, Solids and Comparison with Regulatory Standards

The analysis in this section is focussed on the remaining three metals and TSS, which were observed to have more significant fluctuations and concentrations. As with the nutrient and FIO sections, a series of different pieces of analysis have been completed, commencing with the T-Test results and a comparison with EQS values set in the WFD, as shown in Table 12.

Table 11 - TSS and Metal T-Test Values

Determinant	HD Mean	LD Mean	T-Test P Value	WFD Value (Good)
Zinc (µg/l)	9.65	6.37	2.309×10^{-6}	75 (µg/l) Total
Aluminium (µg/l)	26.33	11.33	2.945×10^{-20}	N/A
Copper (µg/l)	4.32	2.51	2.169×10^{-28}	10 (µg/l)
TSS (mg/l)	35.79	7.90	2.762×10^{-16}	N/A

As can be seen from Table 12 the mean values recorded in high and low discharges for all variables display a statistically significant difference. Only EQS values for zinc and copper are given in the WFD. In relation to Copper, both high and low mean values were well below the EQS for a water body to achieve "good" status. The level set for zinc is for a total value whereas the results collected in the study are dissolved values making it difficult to make a direct comparison. However, as part of the PTS sampling (reported later in this Chapter), 58 samples were tested for both dissolved and total zinc, a ratio between the average of these samples was found to be 4:15. Using this, the dissolved averages in the river sampling equate to 33.7 for the high discharge mean and 22.3 for the low discharge mean. While this is not based on a large number of samples it does give an indication that the equivalent mean total zinc values in river sampling would be lower than the 75µg/l EQS value.

The relationship between the high and low flow values are also shown visually in box plots (Figures 60-63). From these graphs the fluctuation between different metals and solids at different sites can be observed. The method used to generate these graphs is the same as that used for *E.coli*, i.e., with variables split into high and low discharge groups for each site and then mean values and ranges calculated.

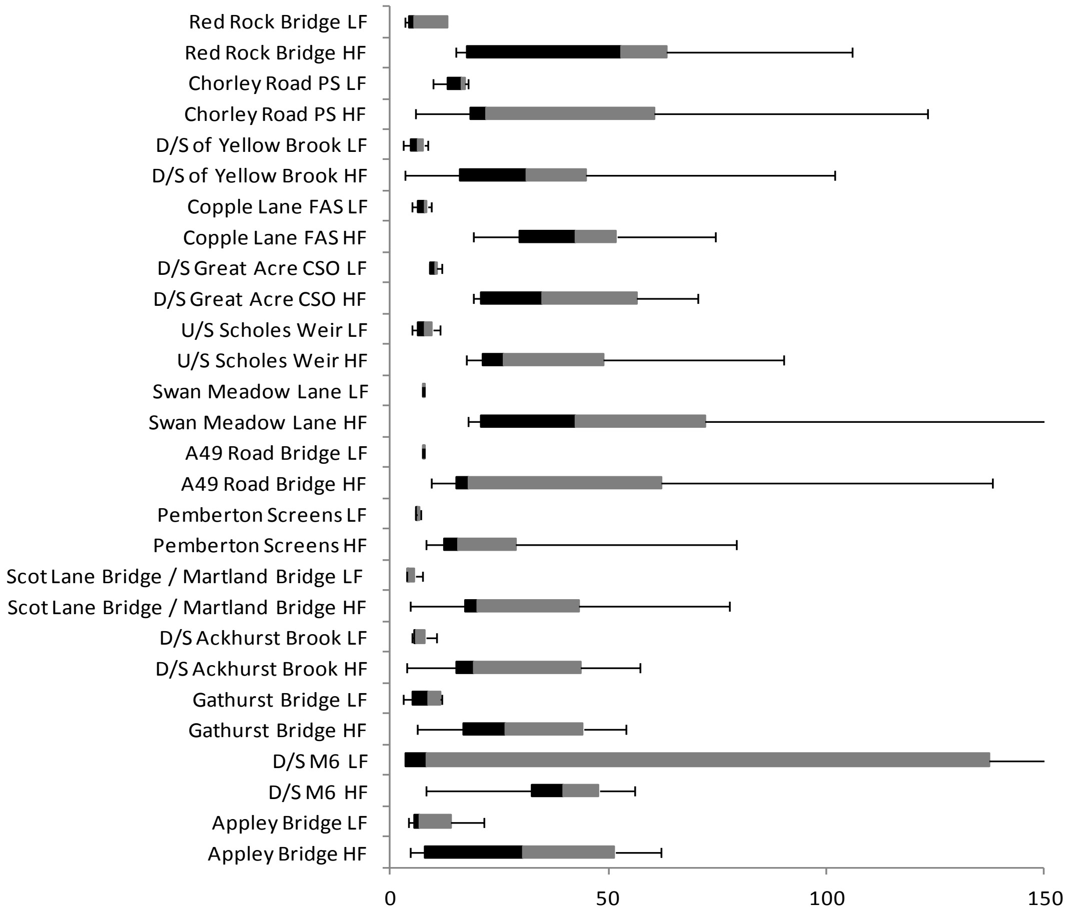


Figure 59 - Range of TSS concentrations at sample sites

Figure 60 shows the variation observed in TSS concentrations. It shows that all sites display higher concentrations in high rather than low discharges with one exception. This is at the site downstream from where the M6 motorway crosses over the river here on one occasion a very high reading of over 500mg/l, resulted in a higher mean and range at low discharge rather than at high discharge. As with all other variables analysed the Swan Meadow site also has a significantly elevated maximum value, this was recorded during week five's sampling, likely as a result of the storm event which covered the sample collection period.

Figure 61 shows the same information for zinc. At eleven out of fourteen sites the range of samples collected in increased discharge conditions were higher than the range of samples collected in low discharge conditions. In many cases samples collected at low discharge conditions return concentrations which were only at or below the minimum detectable level so mean values and ranges could not be calculated. One sample at the Swan Meadow Lane site recorded a concentration of 116µg/l. As this was over twice the concentration of any other sample it is not shown on the graph to allow the majority of values to be observed more easily. Samples collected further upstream are typically higher and there is a slight reduction in mean values across the study area.

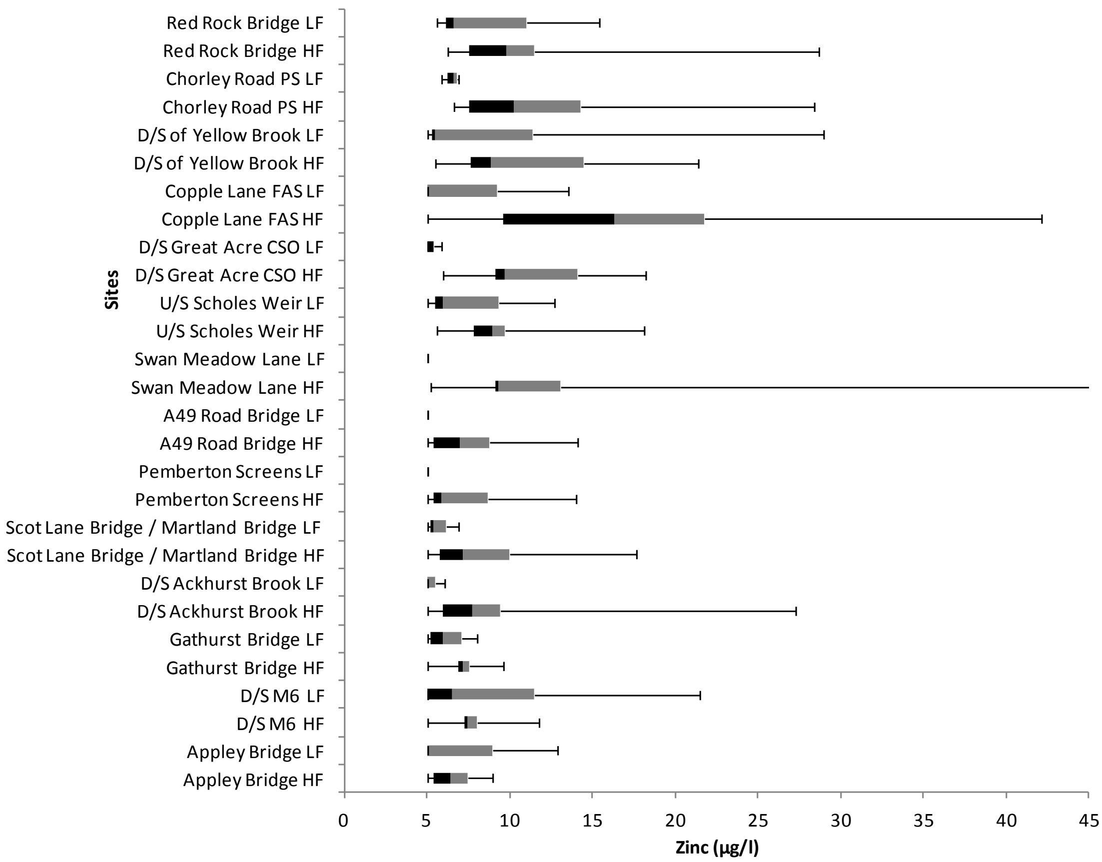


Figure 60 - Range of Zinc concentrations at sample sites

The same information is shown for Aluminium (Figure 62) and Copper (Figure 63) and for both of these metals no sites demonstrate higher mean values and ranges in low rather than high discharges and there is more distinction between values. Observable in both figures is the greater variance of metal concentrations in low discharges at sites further downstream.

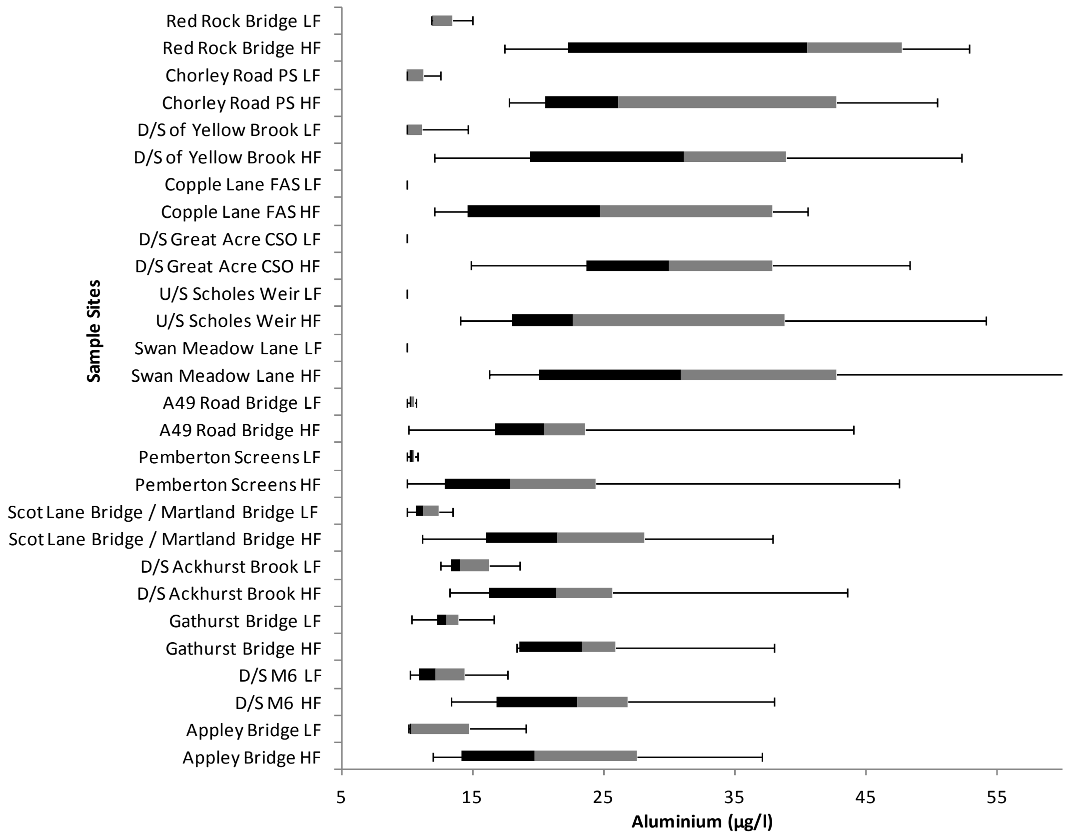


Figure 61 - Range of Aluminium concentrations at sample sites

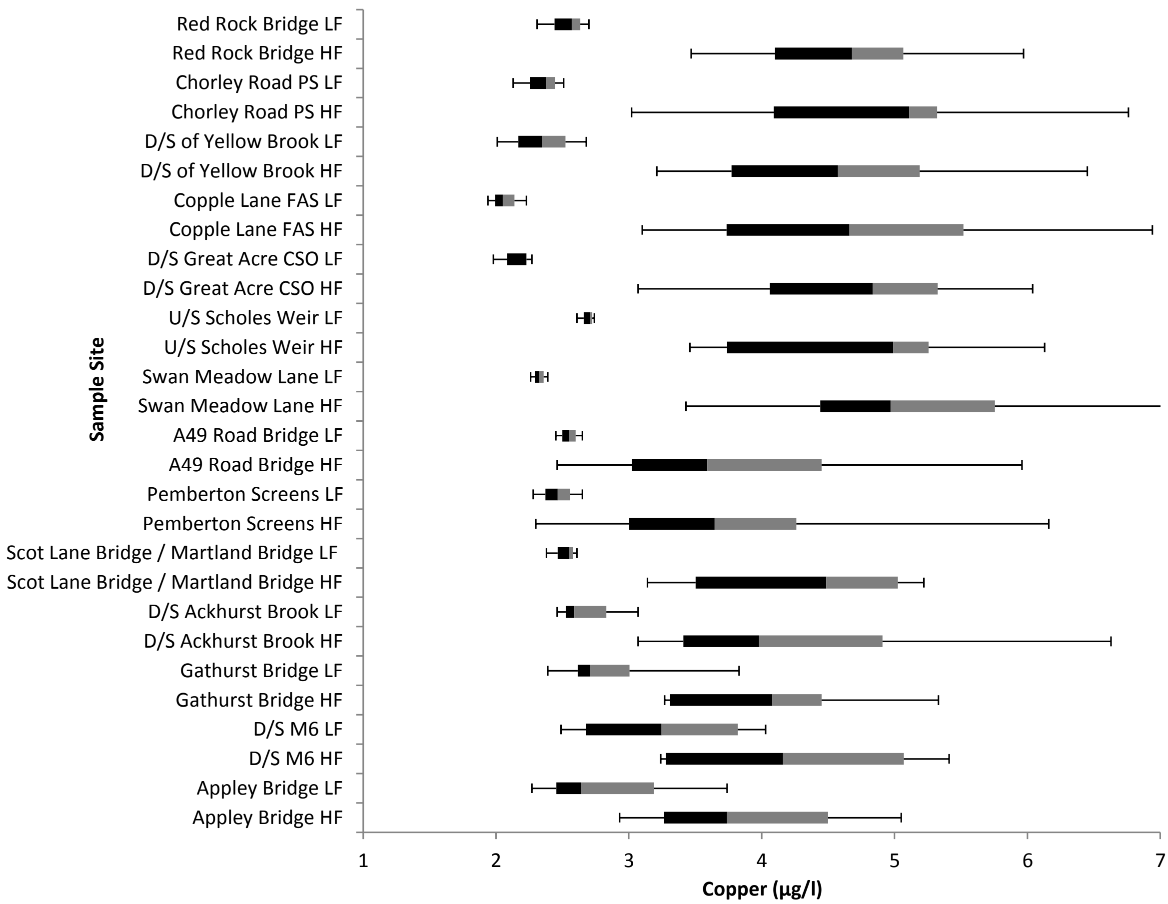


Figure 62 - Range of Copper concentrations at sample sites

4.3.3.2 Heavy Metal and TSS Fluctuation during Storm Events

In order to aid comparison the following six Figures, numbers 64-69 show the same three separate storm events that were displayed for *E.coli*. Discharge and rainfall are plotted against metal and solid concentrations to observe the effect of different storm event sample concentrations. As metals and solids are reported in different units they are shown on separate figures. As can be observed from all the figures generally metal concentrations occur in the same ratios, in that Aluminium was highest followed by Zinc and Copper which were observed at lower levels.

The first event is shown in Figures 64 and 65, as was observed in the FIO section. During the low event there was a significant period of rainfall which fell in a very short period resulting in a very steeply rising hydrograph where discharge increases from $0.5m^3/s$ to almost $3m^3/s$ over a 4 hour period. A series of samples were collected during the storm on the evening of the 15/06/12 followed by several more collected the following day as the discharge was still reducing.

Figure 64 shows all three metals and Figure 65 shows TSS concentrations. Initially before the discharge increased, concentrations of metals and solids were low. As discharge increased rapidly following heavy rainfall, two high solid values were recorded in the final two samples taken on the 15th. This was only observed in one of the final samples with respect to metal concentrations, Aluminium being at over 1000µg/l, Zinc at over 100µg/l and Copper at almost 20µg/l respectively. As these values were considerably higher than the concentrations in other samples they are not shown on the graph to facilitate clarity of the other observations. All other metals measured were also observed at higher than usual concentrations, including Lead, Chromium, and Cadmium, which were not normally detected in project samples at anything above trace levels. Concentrations in samples taken the following day, as discharge was returning towards base flow, were all at normal levels for the metal concentrations. For solids samples taken, they were also back to normal levels except for one which was at similar concentrations to the high readings from the day before. It is unclear as to why this sample was recorded at significantly elevated levels since there was no rainfall or discharge increase to justify it.

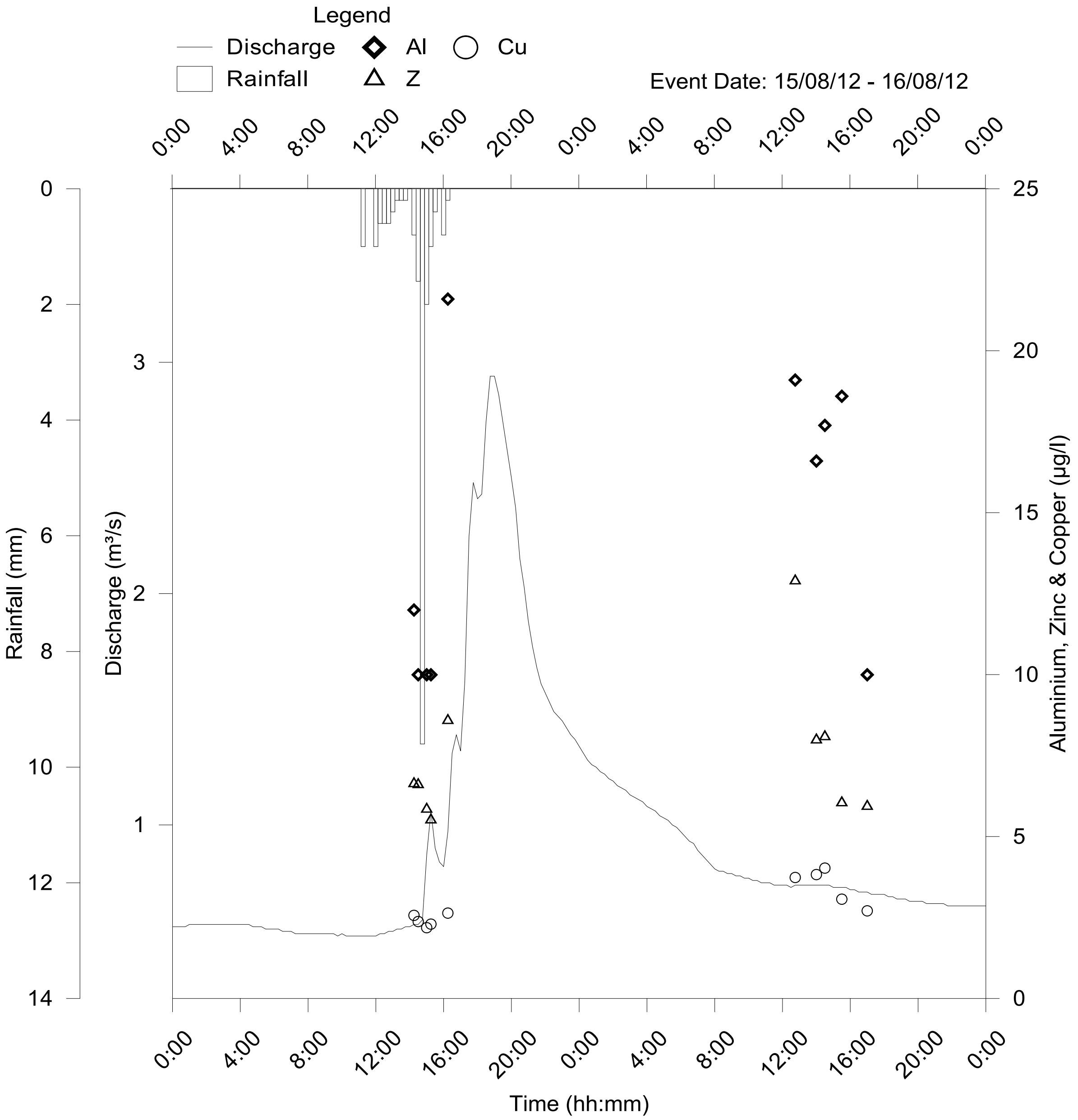


Figure 63 - Heavy Metal Concentrations (Low Event)

Figures 66 and 67 each display the same results for a medium level event and, as described in the previous section, it consists of 3 separate peaks. When sampling commenced, discharge was already elevated and was approaching the final peak of the storm and subsequently the sample concentrations were elevated. Aluminium and Zinc were at their highest concentrations in the first samples to be taken and concentrations then fell through the duration of the day, even before the peak discharge occurred. Copper started at a lower concentration, increased slightly then fell as discharge reduced. With the exception of the significantly elevated value for all three metals during the low event shown previously in Figures 64 and 65, average concentrations in this medium event were shown to be more consistently high. In contrast to the low event however, levels of other metals such a Lead, Cadmium, and Chromium were not observed above trace concentrations.

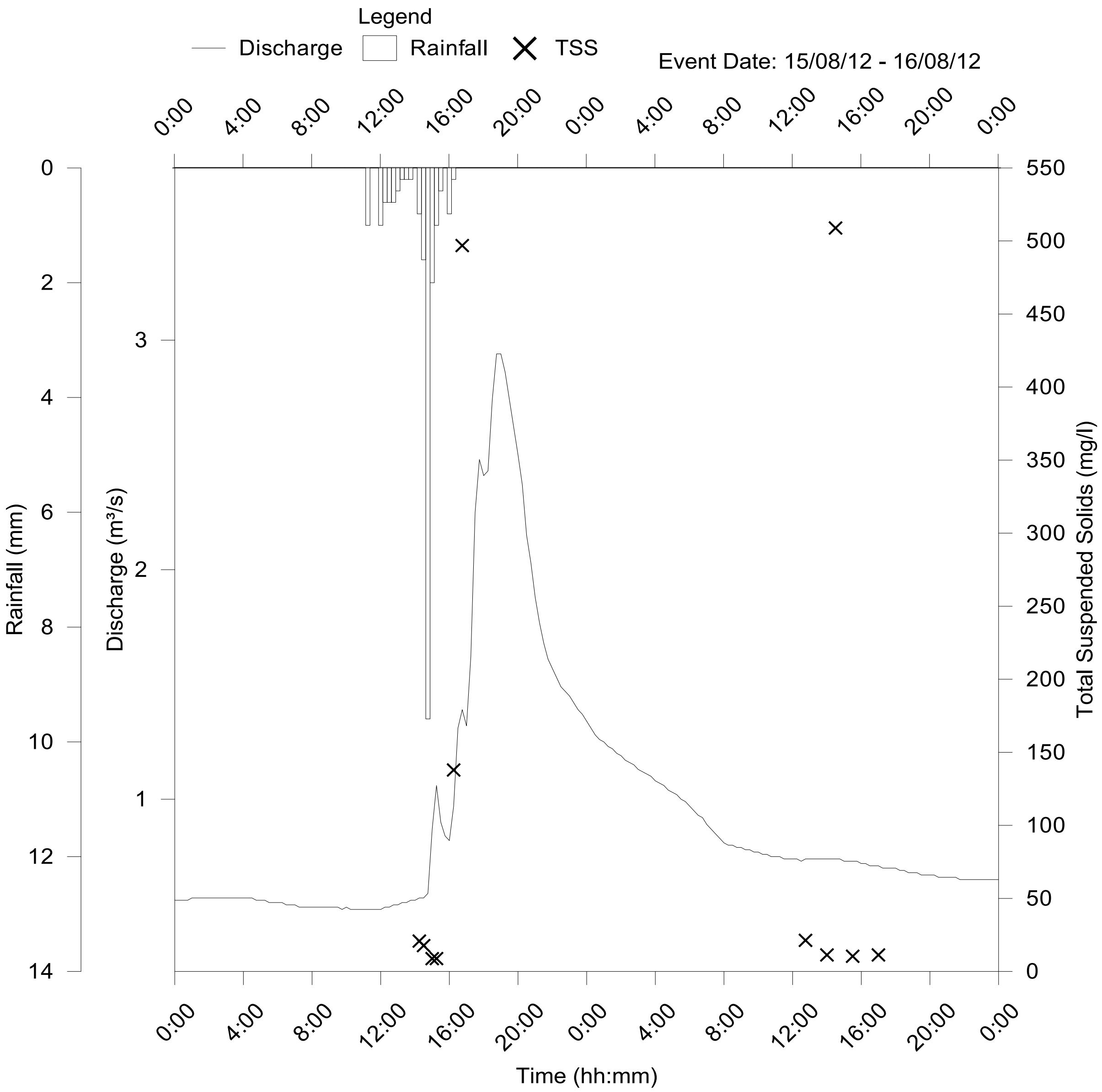


Figure 64 - TSS Concentrations (Low Event)

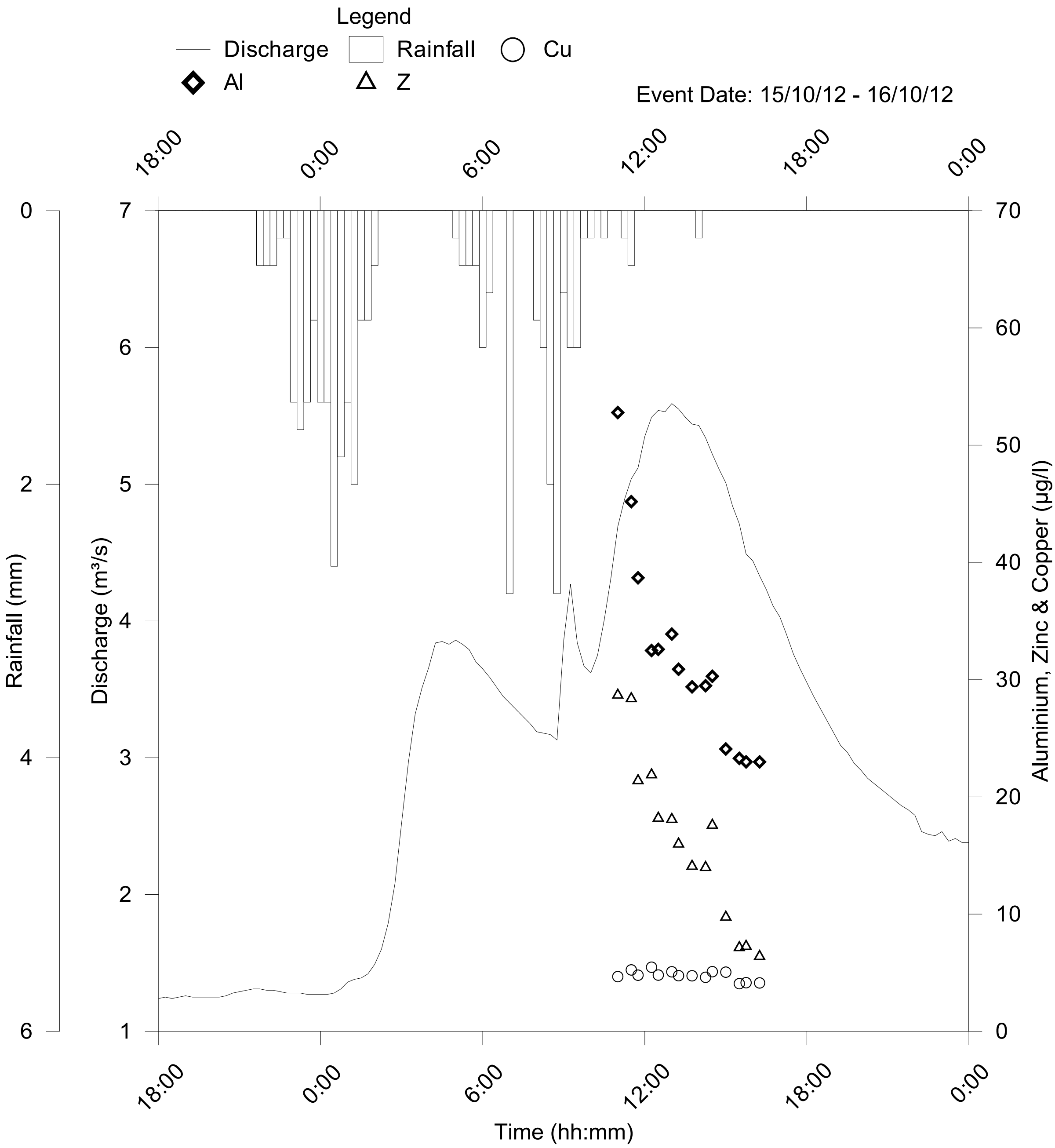


Figure 65 - Heavy Metal Concentrations (Medium Event)

Results for TSS for the medium event are displayed in Figure 66 and 67, as with metals, TSS showed a strong correlation with discharge. Initially high, the TSS increased further before falling in subsequent samples. Initially sample concentrations fell rapidly before increasing and then falling again across the rest of the study area. Similar to metals, TSS sample concentrations were not as extremely high as some of those taken during the low flow event were but were more consistently elevated.

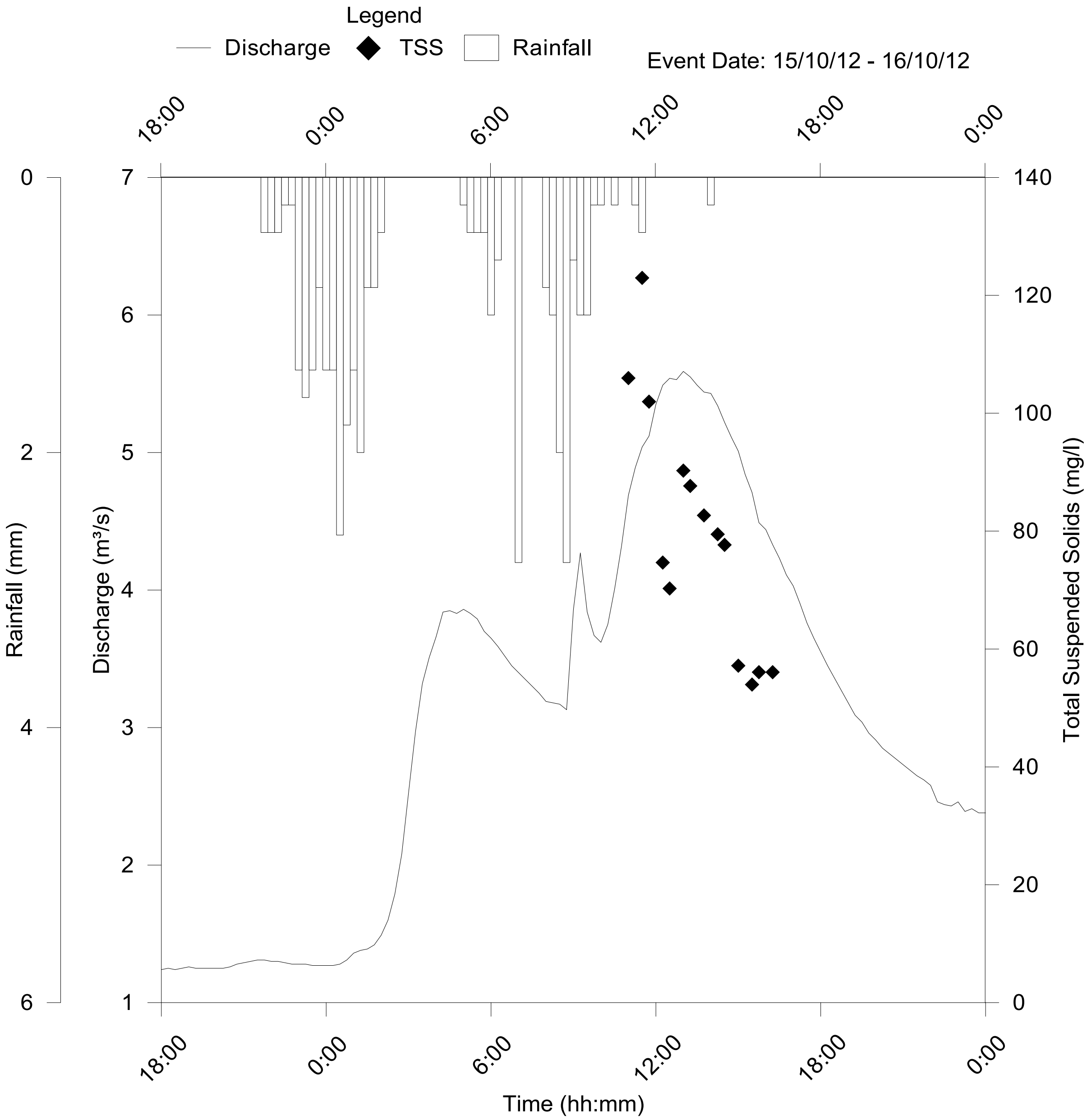


Figure 66 - TSS Concentrations (Medium Event)

Figures 68 and 69 shows the results obtained from a high event occurring over 3.5 days, with discharge over this period being the highest experienced during the whole of the study sampling period. A series of samples were collected over the event, and discharge in the channel remained over $12m^3/s$ for over 2 days. During the event all the typically detected dissolved metals that were sampled during this period were elevated. Aluminium, Zinc and Copper were higher than normal, small levels of Chromium were also detected at half of all main channel sites, whereas in normal base flow conditions typical concentrations were below detectable level. Only Lead and Cadmium were not detected. After 08:00 on the 27th the most significant rainfall had finished and the discharge in river had peaked.

On the second day of sampling, with reduced rainfall and discharge, sample concentrations were generally lower than those from the first day when rainfall was heavier. This was also observed in the concentrations of solids which are shown in Figure 67, which were consistently elevated above those experienced in low discharge. However sample concentrations were lower than those observed during both the previously described events. In the high event TSS was between 40-80mg/l whereas in the medium event it was between 50-120mg/l and in the low event it was between 20-500mg/l.

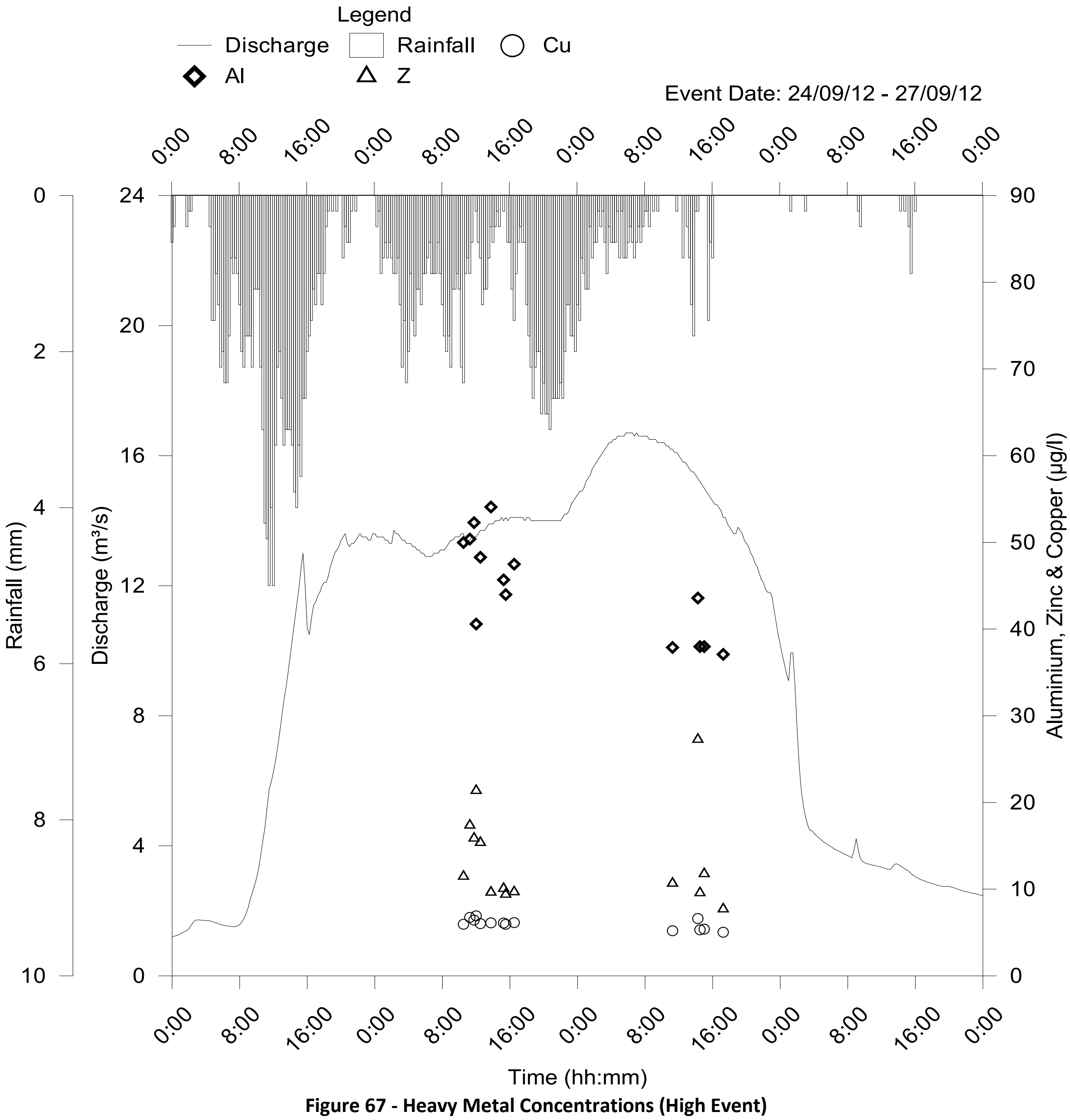


Figure 67 - Heavy Metal Concentrations (High Event)

The metals observed in the channel are likely to originate from road gully’s, which would have been thoroughly flushed with the significant prolonged wet weather for the duration of the sample period. On the second day of sampling it is likely that much of the material that had accumulated in road gullies prior to the rainfall had been washed into the river and transported downstream. Also much of the particulates which usually collect on urban surfaces would have been removed leaving reduced amounts of metallic material available for wash off and transport into the river.

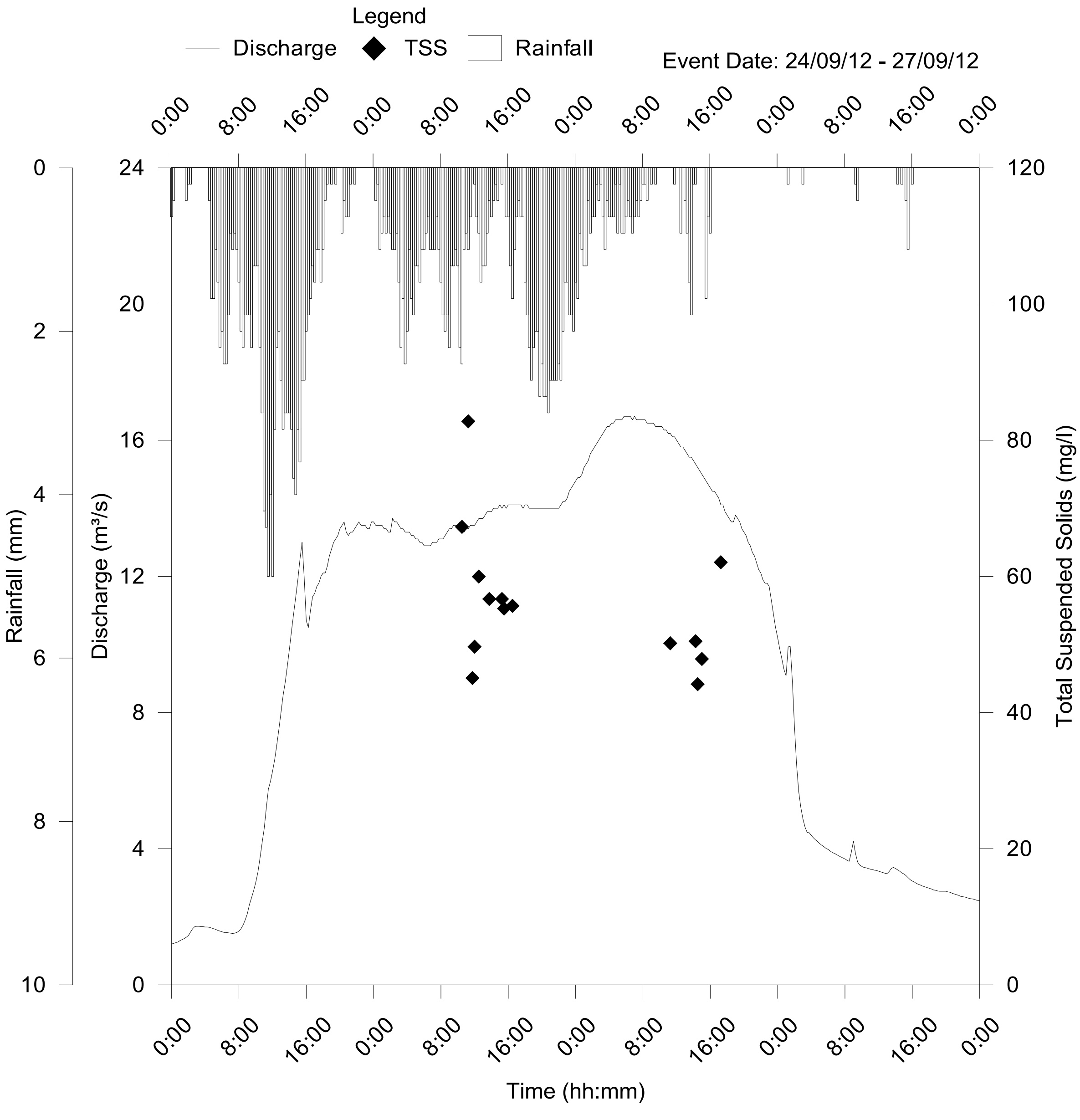


Figure 68 - TSS Concentrations (High Event)

Similar to the observations on *E.coli* concentrations in response to the same events it can be ascertained from these results that it is not necessarily the size (in terms of time period and volume of discharge) of the individual event that is the most important factor affecting sample concentration. For example, as with *E.coli*, the highest concentrations of both metals and solids were observed during the low flow event when the rainfall experienced was very intense. Whereas high *E.coli* levels are likely due to inundation of WWTWs and CSOs, with metals and solids important contributions are received from surface wash off and flushing of stormwater drains where sediments have accumulated from previous events and antecedent periods. The sampling undertaken in this study supports the view that during short periods of intense rainfall, high water velocities and volumes suspend and carry greater quantities of particulates into the river resulting in higher concentrations.

4.3.3.3 Downstream Metal and TSS Fluctuation

The average concentrations of metals and TSS have been plotted and are shown in Figures 70-73. Figure 70 shows the results for TSS and, as has been discussed throughout this section, the contribution from a small number of samples with exceptionally high concentrations has resulted in un-characteristically high average values at some sites. For TSS these are not only at the Swan Meadow Lane sampling site, as with *E.coli*, but also at the site downstream of where the M6 crosses the river. The average values of TSS have been plotted on Figure 70 and are indicated with the solid black line. The light grey line is the same data with the exceptional values omitted. The removal of these outlier results demonstrates how an individual event can lead to significant volumes of pollutants being released into watercourses. Concentrations in tributaries were generally lower than those found in the main channel with the notable exceptions of Clarrington and Close Brooks where average solid levels were above 35 mg/l.

All metal levels were also significantly elevated at the Swan Meadow site, and consequently average values for the three metals analysed we also significantly elevated. To avoid replication all three metals have been plotted with the exceptional values removed. Figure 71 shows results for Zinc. Average concentrations initially increase between Red Rock Bridge and Coppull Lane, which had the highest average of any main channel site. Overall levels then fall across the study area, but fluctuate between the remaining sites. All tributary sites reported lower mean values than those identified in the main channel, except Clarington brook which had the highest average zinc level of all sites, indicting a pollutant source contributing to zinc concentrations in this tributary. For much of its course Clarrington Brook is enclosed in a culvert, and the identification of locations where pollutants could be entering the channel proved difficult, as this precludes further visual inspection without employing specialist equipment.

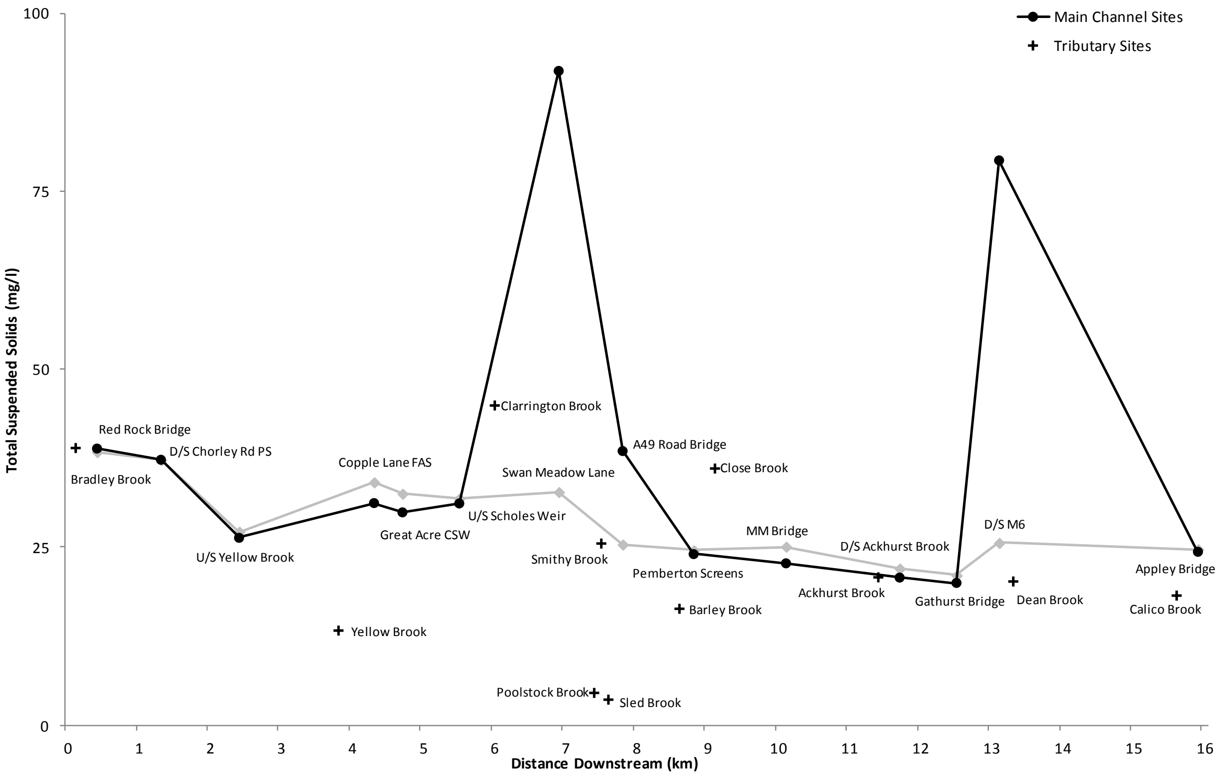


Figure 69 - Average TSS Change Downstream

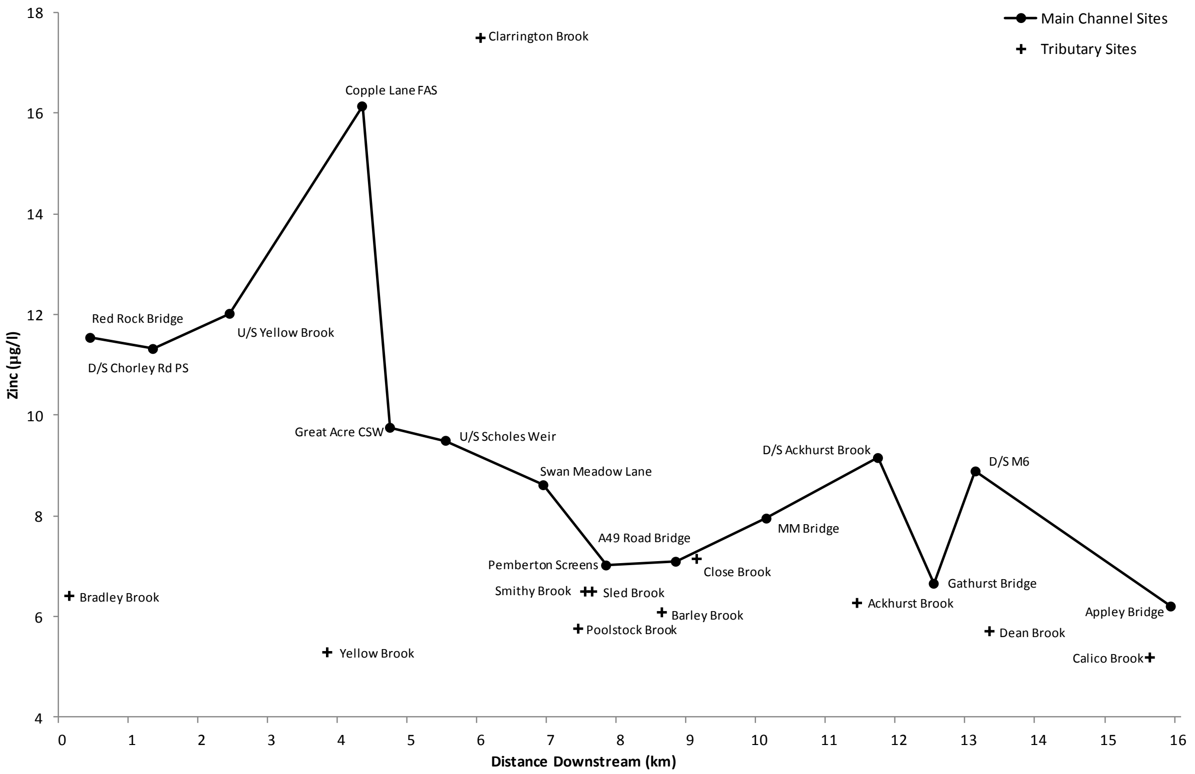


Figure 70 - Average Zinc Change Downstream

Figure 72 displays results for Aluminium. Overall there is a fall in Aluminium concentrations at sample sites from upstream to downstream, but there are some fluctuations. Levels fall from Red Rock to Coppull Lane, then rise slightly between here and the Great Acre CSW site, before falling gradually between this point and Appley Bridge, the last site in the study area. Again, as with Zinc, concentrations of Aluminium were highest at the Clarrington Brook sample site, further indicating that there is a source of metal pollution entering its course. Also notable is the levels in Ackhurst and Calico Brooks, which both have average concentrations higher than all main channel sites except Red Rock.

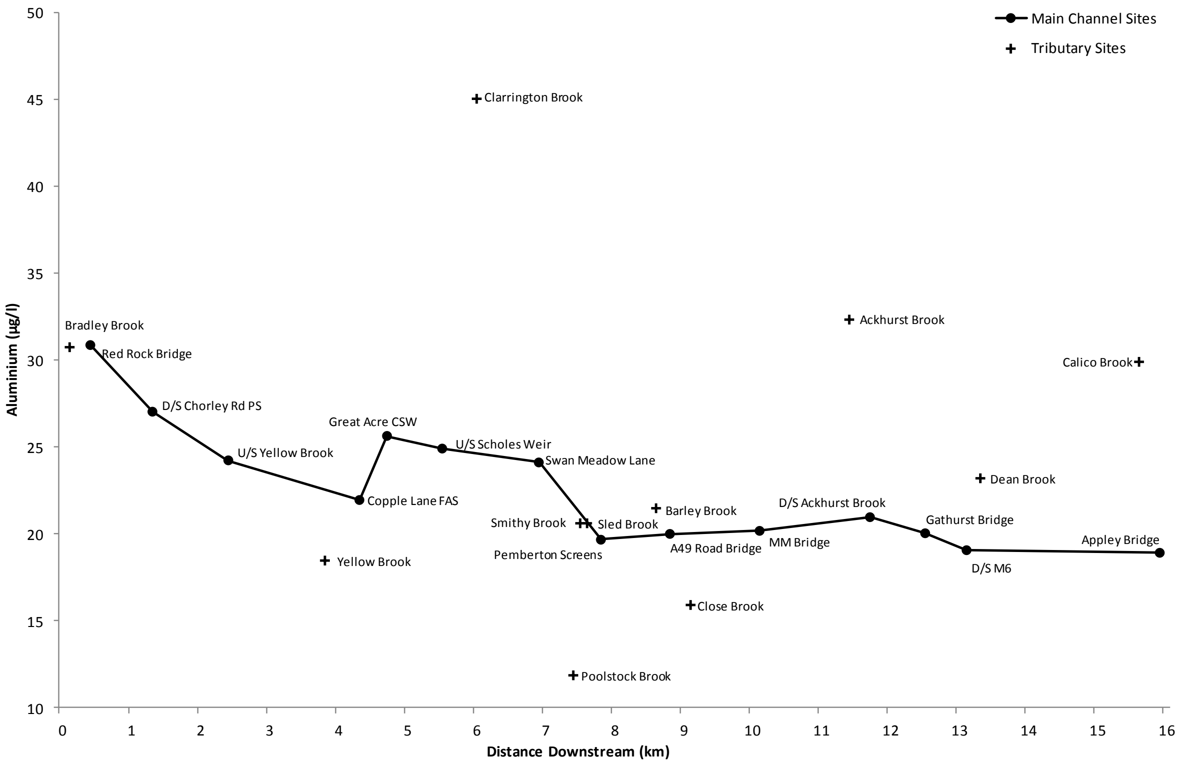


Figure 71 - Average Aluminium Change Downstream

Figure 73 shows the results for Copper, which were generally observed to be low concentrations, typically between 3-5µg/l. The levels fluctuate across the study area but demonstrate a very slight downward trend. As with Aluminium, Ackhurst and Calico Brooks produced higher average concentrations than were observed in the main channel. However conversely average concentrations in Clarrington Brook, were lower than the average values found in the main channel.

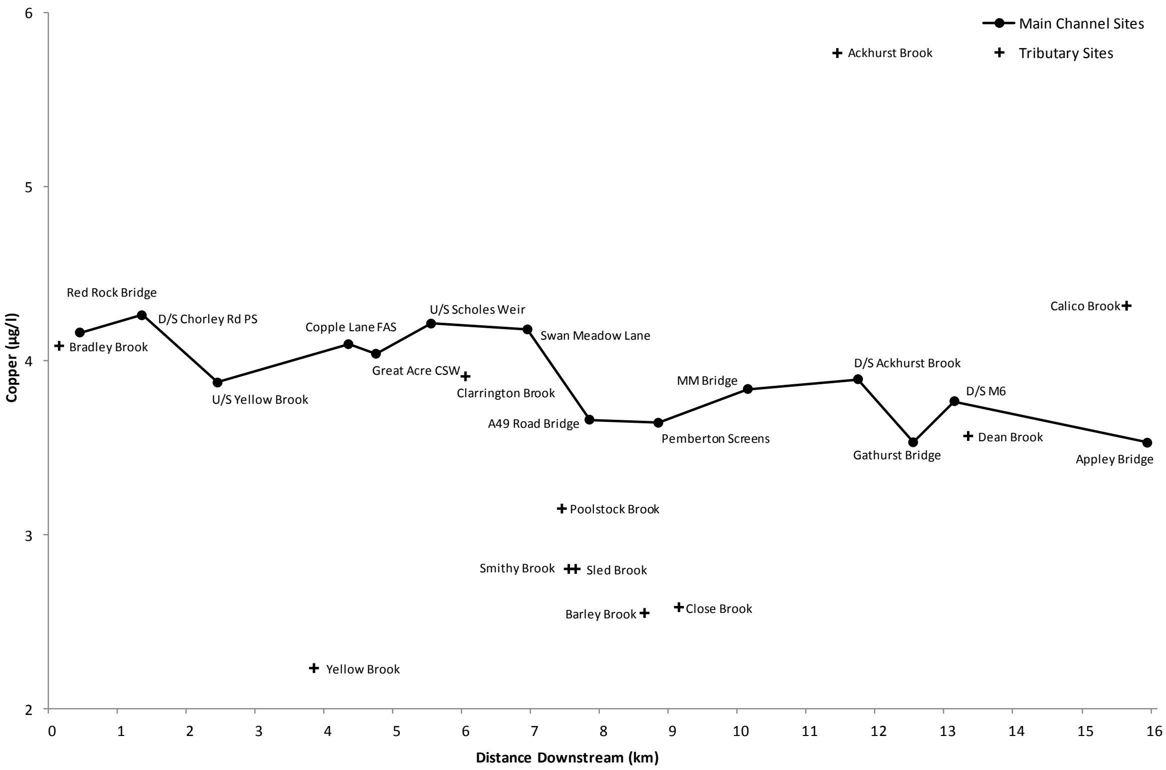


Figure 72 - Average Copper Change Downstream

4.3.4 Polyaromatic Hydrocarbons

In line with other variables PAHs have been subject to a series of different sections of analysis depending on whether they were observed in significant concentrations or if concentrations varied between sites or in response to fluctuations in discharge or rainfall. PAHs are generated from incomplete combustion and they can arise from a multitude of sources in an urban environment. They arise primarily from vehicle exhausts, but other sources such as atmospheric deposition are also important. They are also generated from land management such as burning of vegetation as well as fires that occur in urban environments (Vane et al., 2013). PAHs are hydrophobic and are not soluble in water; however they can readily become bound to particulate matter which means that when channels are turbid and heavily loaded with solids high concentrations can be experienced. Nearly all PAHs are carcinogenic although their potency varies.

Table 12 - Summary of PAH Detections in the Study

Chemical	**Range detected**	**EQS/MAC**
Benzo(b)fluoranthene	0.01 – 0.0498µg/l	0.03µg/l (annual average) (sum of benzo(b)fluoranthene and benzo(k)fluoranthene)
Benzo(k)fluoranthene	0.01µg/l	0.03µg/l (annual average)
Fluoranthene	0.01 – 0.0824µg/l	0.1µg/l (annual average) 1µg/l (MAC)
Indeno(123cd)pyrene	0.01 – 0.04µg/l	0.002µg/l (annual average)
Anthracene	0.01µg/l	0.1µg/l (annual average) 0.4µg/l (MAC)
Benzo(a)pyrene	0.01 – 0.04µg/l	0.05µg/l (annual average) 0.1µg/l (MAC)
Benzene	0.1µg/l	10µg/l (annual average) 50µg/l (MAC)

Table 13 summarises the ranges of PAHs values observed in water samples collected. Concentrations of PAHs in water samples were detected very intermittently and where they were observed they were almost all uniformly low. Whereas concentrations of metals and solids were seen consistently in samples and can be seen to fluctuate in response to discharge and location, allowing analysis to be completed. This was not the case with PAHs. Their concentrations were generally below detectable levels (0.01µg/l) meaning more detailed analysis was difficult. As was observed with other pollutants, concentrations increased during elevated discharge. However, irrespective of location or conditions, frequently PAHs sample concentrations were still only detected at trace levels and where they were slightly elevated a maximum concentration of 0.51µg/l was detected.

A series of EQS values MAC values (as seen in Table 13) are available in the WFD for PAHs. Many of the EQSs are expressed as Annual Averages which is not directly comparable to concentrations seen over a shorter sample period, but nevertheless they have been used to provide a comparison with values observed in water samples. MAC's have also been compared where available and these are preferable as they can be compared against single samples. For many of these substances the concentrations were only detected at trace levels and well below the available EQS. Some recorded values for indeno(1,2,3cd)pyrene, benzo-b-fluroanthene and benzo-k-fluoranthene concentrations are above annual average but were only detected intermittently and did not exceed MAC values.

4.3.5 Halogenated Solvents

Solvents have a number of industrial uses, for example in paints and adhesives, but as they are also commonly used as degreasing agents it is this use that most likely leads to river water contamination. As with PAHs solvents were detected only intermittently and at low concentrations. Table 14 shows a summary of their detected concentrations and, where available, a comparison with EQS values set in the WFD.

Table 13 - Summary of HS Detections

Chemical	Range of concentrations detected	EQS/ MAC
1,1-dichloroethane	0.1µg/l	No EQS – see following aquatic toxicity data in Table 15
1,2-dichloroethane	0.1µg/l	10µg/l (annual average)
Chloroform	0.1 – 0.51µg/l	2.5µg/l (annual average)

Only one substance, dichloroethane, has no EQS in the WFD, therefore, to put the measured levels of this compound into context it is compared to the concentration available from aquatic toxicity data. This is limited and the only toxicity data that were found for this compound is for one species of fish, shown in Table 15. This suggests that this compound is of low acute toxicity to the species tested. Due to the limited data, the EQS for 1,2-dichloroethane can be used as an indicator because it is a structurally similar compound.

Table 14 - Toxicology Data for Dichloroethane

Test Species	Test Duration	Effect Concentration (mg/l)	Source
Pimephales Promelas (fathead minnow)	1 day	500	Ecotox
	1 day	100	Ecotox

As neither PAHs nor solvents were detected in any significance, more detailed analysis of levels of these chemicals has not been undertaken.

4.3.6 Biochemical Oxygen Demand, Chemical Oxygen Demand and Total Organic Carbon

This final section of the river sample results presents a series of standard tests that are routinely completed on water samples to identify several different properties. These are COD, BOD and Total Organic Carbon (TOC). The quantity of organic carbon in water and wastewater consists of a variety of organic compounds in differing states of oxidation. These compounds can be further oxidised by biochemical or chemical processes and these fractions expressed appropriately as BOD or COD. TOC is a more direct and convenient expression of the total organic content of a sample. TOC is independent of the oxidation state of organic matter, and organically bound elements such as Nitrogen and hydrogen. Other in-organics that can contribute to oxygen demand are not measured by the test (Eaton *et al.*, 1995).

BOD is a test which quantifies the oxygen within a water samples that would be consumed if all the organic matter within a sample was oxidised by bacteria and protozoa. This test involves subtracting the dissolved oxygen content of a water sample which has been incubated at room temperature for 5 days from the dissolved oxygen of the sample at the time it was taken from a water course. The incubation should be kept in complete darkness to ensure no further oxygen is produced by photosynthesis. The bacteria will metabolize normally consuming dissolved oxygen in the process and the remaining figure is the amount of oxygen demand of microbes during the 5 day period.

COD measures oxygen demand of different properties of a sample in that it measures the oxygen equivalent of organic and inorganic matter content. The test utilises the process of oxidising organic and inorganic compounds using a strong oxidising agent under acidic conditions to indirectly measure the quantity of such compounds within a water sample. To determine the quantity of TOC, molecules must be broken down into single carbon units and converted to a singular molecular form that can be quantitatively measured. Organic carbon is then converted to carbon dioxide through the use of heat and oxygen, ultraviolet irradiation, chemical oxidation or a combination of these processes which is subsequently measured indicating TOC quantity (Eaton *et al.*, 1995). Samples were split, using the same method as previously described in this Chapter, into those samples collected in high and low discharge conditions and a T-test was used to compare mean values. The results are shown in Table 16, which also shows the available EQS value for BOD from the WFD.

Table 15 - COD, BOD and TOC T-Test Values

Determinant	HD Mean	LD Mean	T-Test P Value	WFD Value (Good)
COD (mg/l)	39.67	20.49	2.307×10^{-13}	N/A
BOD (mg/l)	4.34	1.43	2.786×10^{-15}	5 mg/l
TOC (mg/l)	11.30	6.26	6.870×10^{-23}	N/A

All three tests display a statistically significant difference between high and low discharge samples. Utilising the same format as for previous variables, Figures 74-76 present box plots showing the range of each variable under high and low discharge conditions at each sampled site.

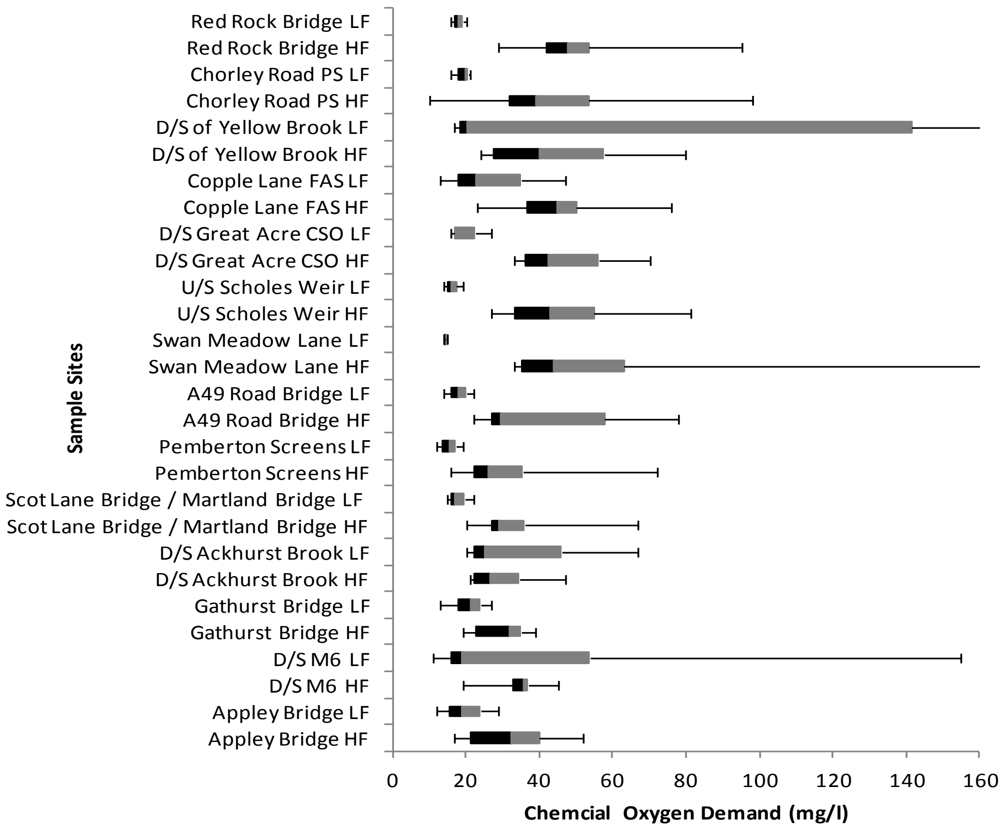


Figure 73 - Range of COD concentrations at sample sites

As with other variables analysed, the overall trend with all three of these tests was that, means and ranges were typically higher in high discharge conditions and lower in low discharge conditions, as indicated in the T-Test results. There are a small number of exceptional values which are worth attention. For COD (Figure 74) high values recorded at the sample site downstream of Yellow Brook and the M6 sample site result in high means and ranges in low rather than high flow.

In respect of BOD (Figure 75) the three furthest downstream sites all recorded greater maximum values in low rather than high discharge, whereas all other sites upstream of Gathurst Bridge displayed the reverse behaviour. As with the previously described water quality parameters, some extreme high values were recorded and the scales of figures have been adjusted not to show these values, so the more subtle differences between the majority of samples can be more easily observed. In the case of sample sites downstream of Yellow Brook and the M6 the reason for the very highly elevated averages in low discharge conditions was due to the collection of a single sample that was so much greater than all other collected samples that it obscured the typical sample concentration of this site. This was observed in several variables so is unlikely to be due to erroneous testing.

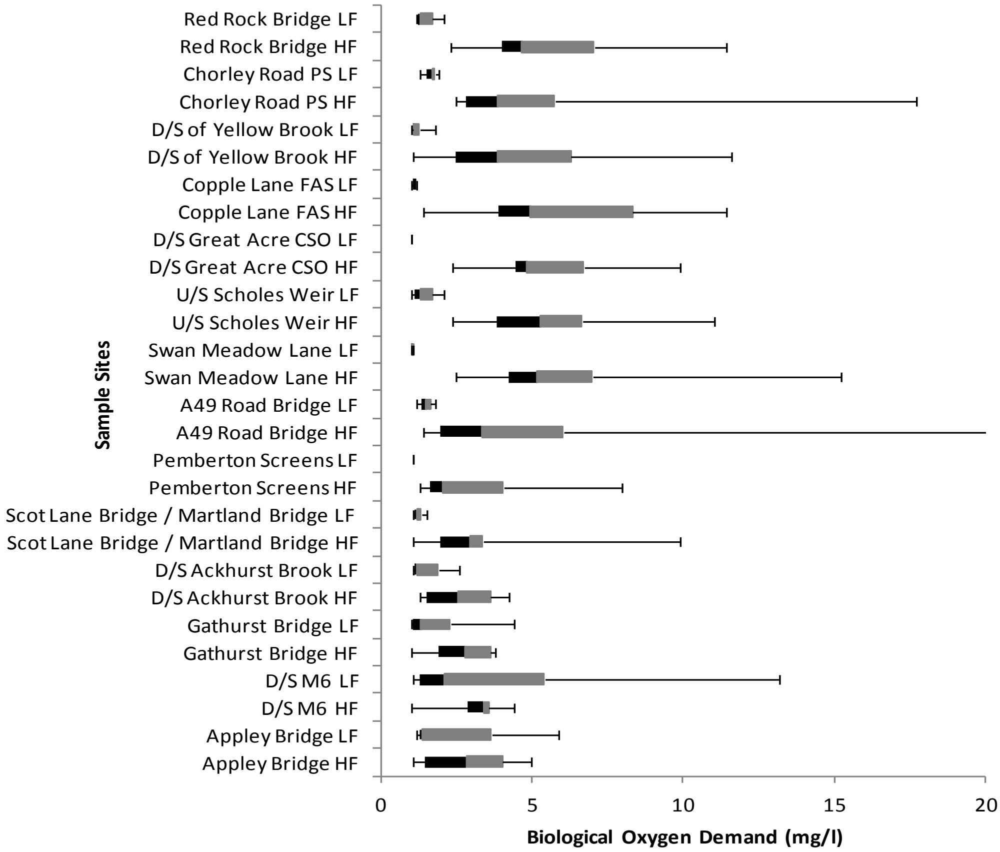


Figure 74 - Range of BOD concentrations at sample sites

All average TOC values (Figure 76) were lower in low and higher in high discharge conditions, except the site at Pemberton Screens and generally there was an obvious difference between low and high values. This is clearly evident in sites further upstream with the differences becoming less pronounced with distance downstream. The effects of increased discharge can clearly be seen on the concentrations of TOC as it measures a wide range of organic compounds which were observed in increased quantities during increased discharge.

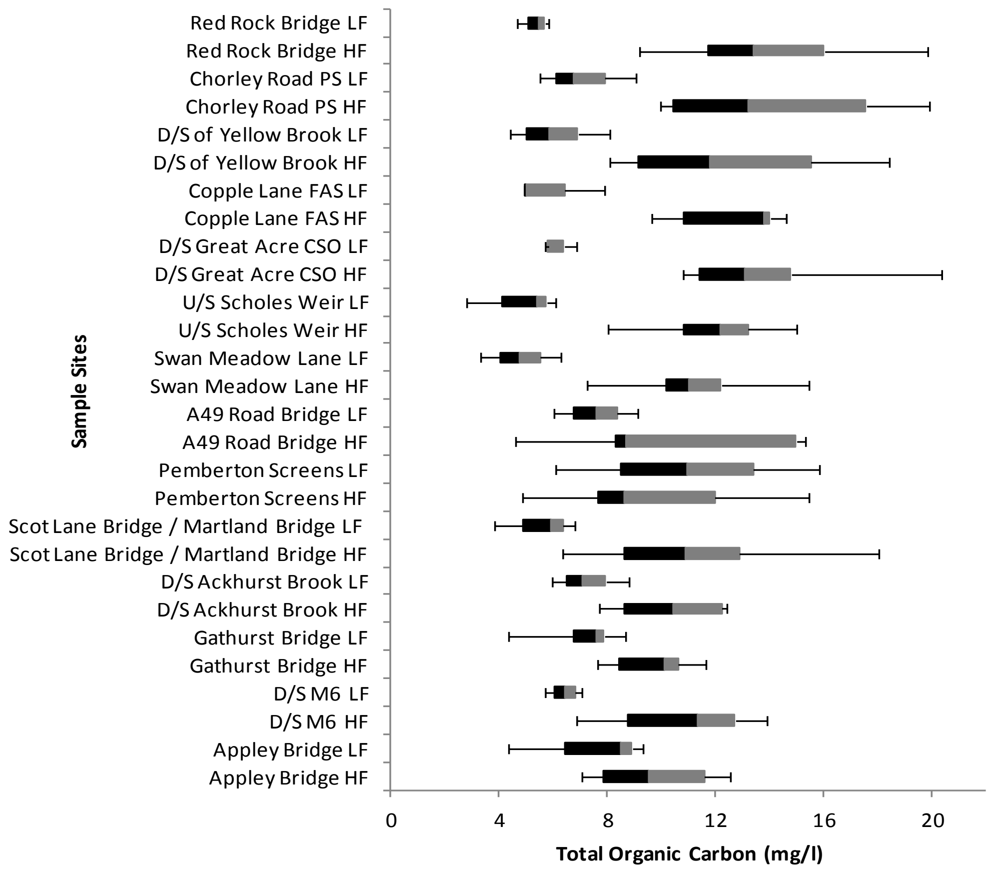


Figure 75 - Range of TOC concentrations at sample sites

4.4 SAGIS Analysis

Following the analysis of the data set collected under the project it is also important to evaluate the data collected against the EA's Source Apportionment Geographical Information System (SAGIS) modelling software in order to further explore the reliability of the results and the model. SAGIS works by collating long term monitoring data and known pollution inputs and discharges to predict concentrations of various contaminants in a river channel. It then displays them as a mixture of textual and graphical outputs.

As SAGIS is calibrated using long term data, the results from the short term intensive sample regime such as the one completed in this work, do not span a sufficiently substantial period to be directly input into the software. To allow a direct comparison, specific SAGIS outputs for the study area were obtained for WQV sampled. SAGIS predictions were then plotted, and the project results were overlaid to determine how accurately the model was predicting pollutant concentrations currently experienced in the river. Not all WQV are covered by SAGIS so a comparison of all variables was not possible. Analysis of project data against SAGIS outputs was completed for:-

- Orthophosphate,
- Nitrate,
- Zinc,
- Copper,
- Nickel.

Figure 77 shows the predictions from SAGIS for Orthophosphate plotted against the data from the project. The shaded area on the graph indicates the prediction made by the model, the upper darker shaded section indicates the upper confidence limits and the lower lighter shaded section indicates the lower confidence limits and the central line indicates the mean values. Between them they form a predicted range into which values should fall. The site ID on the x axis relates to physical locations within the catchment where discharges and pollution inputs enter the river (a full list of location and grid references is included in Appendix III). The results of the sampling conducted are displayed as high and low mean values, calculated for each sampling site. The specific location of individual sample points for the project was often different to SAGIS input nodes therefore sample results were paired with the closest available input node to display them on the graph.

As it can be seen from the graph the average values obtained from the project all fall into the range predicted by the SAGIS software. The model gives better predictions towards the higher end of the sample area whereas further downstream values are close to being outside the model's range. From the plotted project values the strong downward trend in Orthophosphate concentration from the top to the bottom of the study area is also observed in the model prediction, the point at which Orthophosphate is entering the river is clearly evident around site ID 15. Considering this site ID is just downstream of the Horwich WWTWs discharge, it seems a reasonable conclusion to attribute this rise in concentrations to the WWTWs. This supports the observations made in section 4.4.12, based on a comparison of the average concentrations of Phosphorous and Ammonia in the WWTWs discharge to average concentrations seen in samples collected at the top of the study area. It can be seen that the increase for Phosphorous is much more significant than for other variables, indicating the increased importance of the WWTWs discharge with respect to Phosphorous pollution to the river.

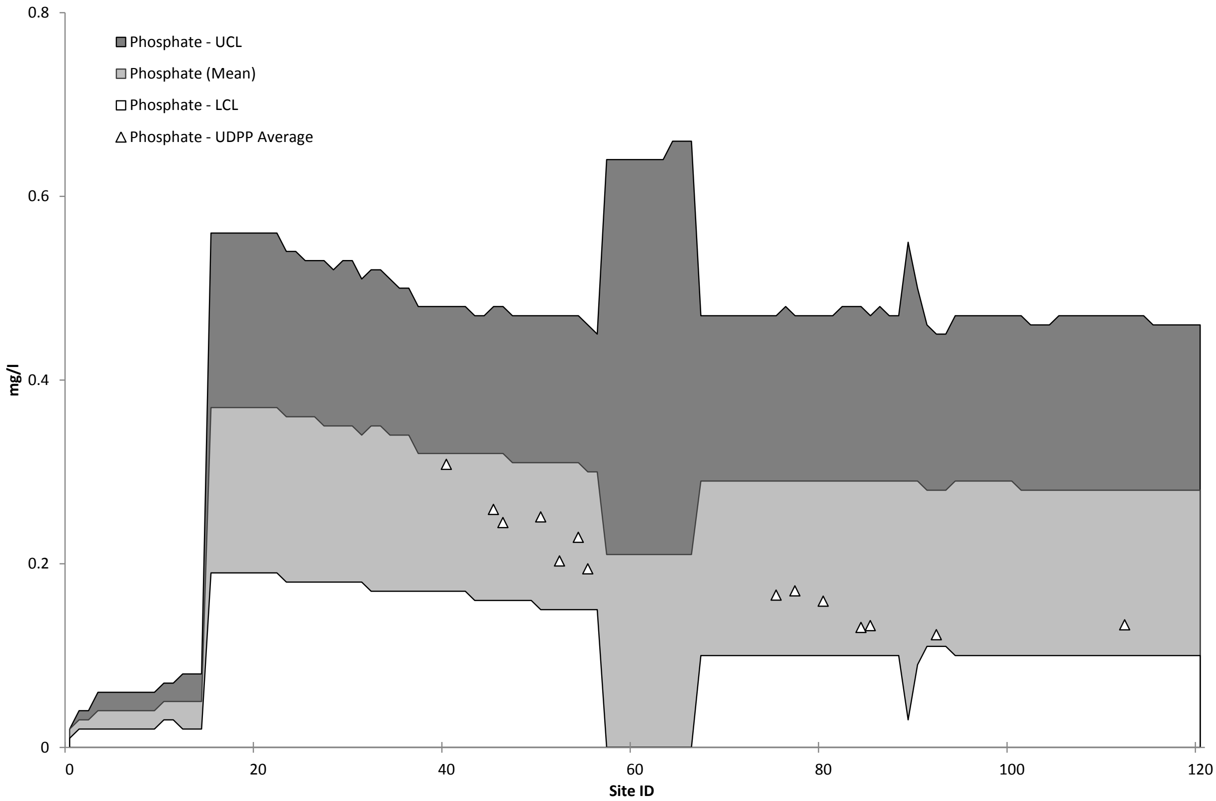


Figure 76 - Comparison of SAGIS predictions against Wigan values - Orthophosphate

Overall the model performs well and all results from the project fall within the predicted ranges made by the model. However the collected values show a more pronounced fall than the model predicts and further downstream its predictions become less accurate. This could be partly due to the seasonal effect of increased water volume in the river having a diluting effect on Orthophosphate concentrations. The EQS for 'good' quality under the WFD for Orthophosphate is 0.120mg/l so both the predicted values and actual recorded values are above this standard for much of the study area and it is only due to the absorption of Phosphate by the river system and the diluting effect of the river that the standard is achieved further downstream in the study area.

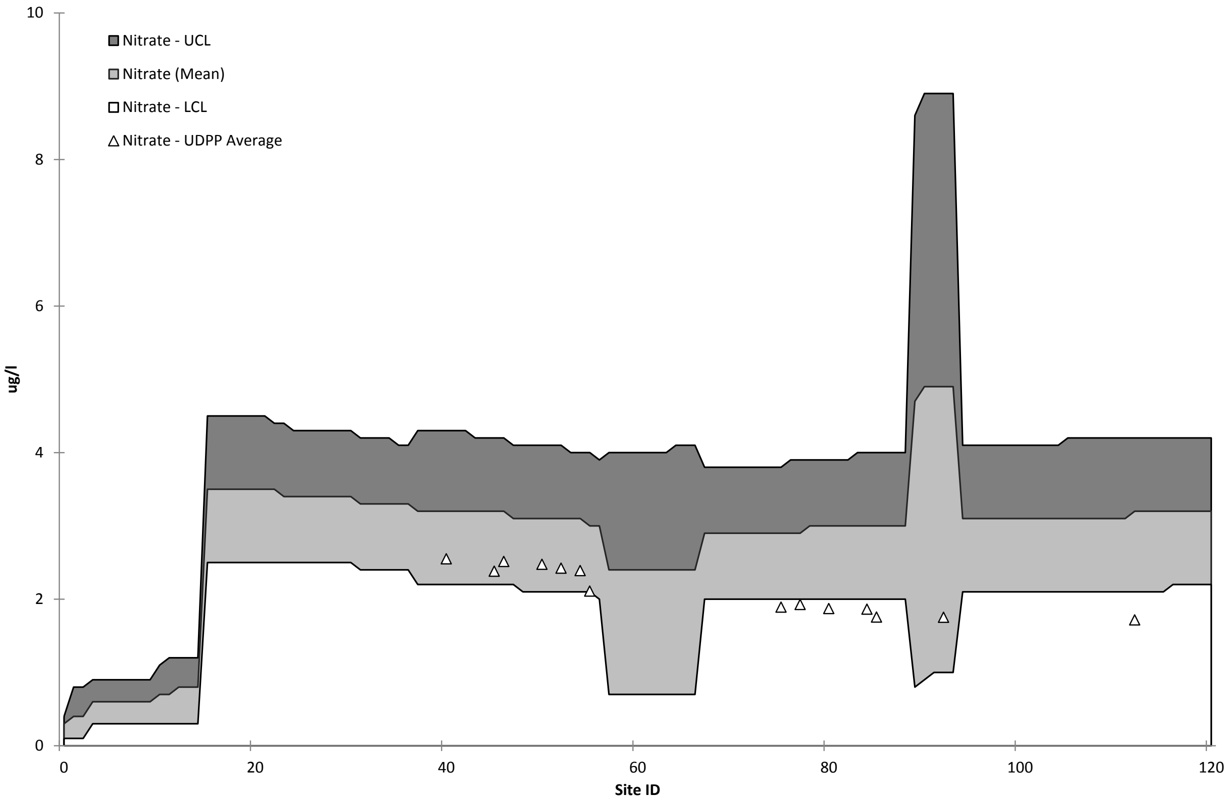


Figure 77 - Comparison of SAGIS predictions against Wigan values - Nitrate

Figure 78 shows the same data for the SAGIS Nitrate predictions. As with the Orthophosphate results there is a downward trend in concentrations from the top to the bottom of the study area. It can be seen that in the upper section of the study area values from sampling mostly fall within the SAGIS predicted range but further downstream experienced values lower than the model prediction. Again the seasonality of the data set will likely cause the values experienced to be lower due to the diluting effects of increase water volume. The EQS for Nitrate is 50mg/l which is a significantly higher concentration than both the predicted values and those identified in sampling. While the model performs well in the upper section of the study area in the lower section it over predicts the levels of Nitrate compared to the concentrations which were found in the samples. It is likely that with an increased number of inputs the variability of pollutants will increase, making accurate predictions more difficult. Again as with Orthophosphate, the contribution of discharge from above the study is evident from the model predictions.

Figure 79 shows results for Zinc. The model over predicts the levels of Zinc in the river, with actual values experienced being much lower and from the graph it can be seen that the vast majority of values recorded fall below the model prediction. With Nitrate and Phosphate being sampled on a routine basis the data set used to support the model will be much more extensive than that available for Zinc. Zinc samples are collected less frequently and this could indicate why the model predictions are not as accurate due to a reduced data set. Both predicted and actual values generally fall well below the 'good' EQS for Zinc. However as identified in section 4.4.2 the EQS is a total value whereas the collected data is dissolved figures. When considering the brief comparison between dissolved and total zinc values in section 4.4.3.1, it seems unlikely the average dissolved values observed, while their total equivalent values would be higher, are consistently greater than the 75µg/l 'good' status. As with other variables there are several sharp spikes in the model predictions along the length of the study area, these are likely to be due to the collection of samples taken during storm events which lead to over prediction by the model at these locations. Alternatively these spikes could be due to polluting discharges. However samples collected for this study did not identify the same increases in average concentrations.

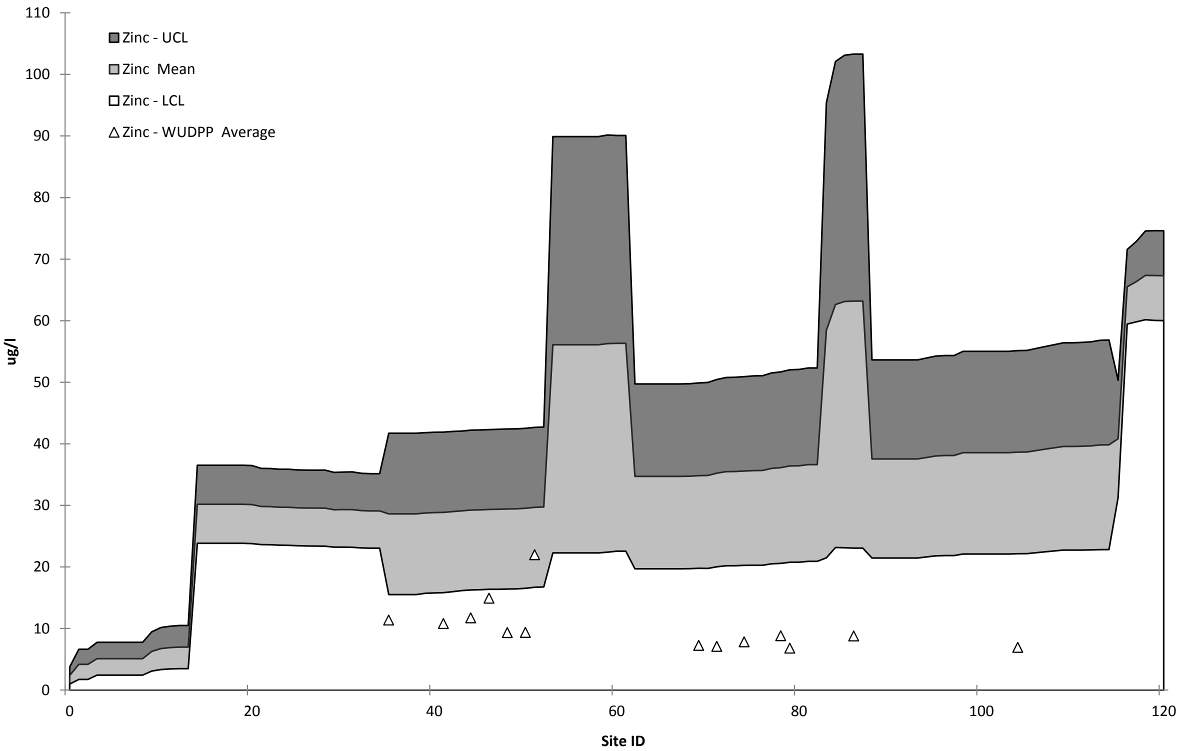


Figure 78 - Comparison of SAGIS predictions against Wigan values – Zinc

Figure 80 shows results for Copper, with which a greater portion of observed values fall into the predicted range than was experienced with Zinc. In the upper section of the study area the model provides good prediction however mean values in the lower part of the study area fall outside of the predicted range completely. Zinc, Copper and Nickel values come from common water samples and so metal concentrations often appear in similar proportions (i.e. if one metal is elevated the others will also be). So the problem of over prediction and high samples resulting in spikes in the predicted range will occur uniformly across the predicted range of all three metals.

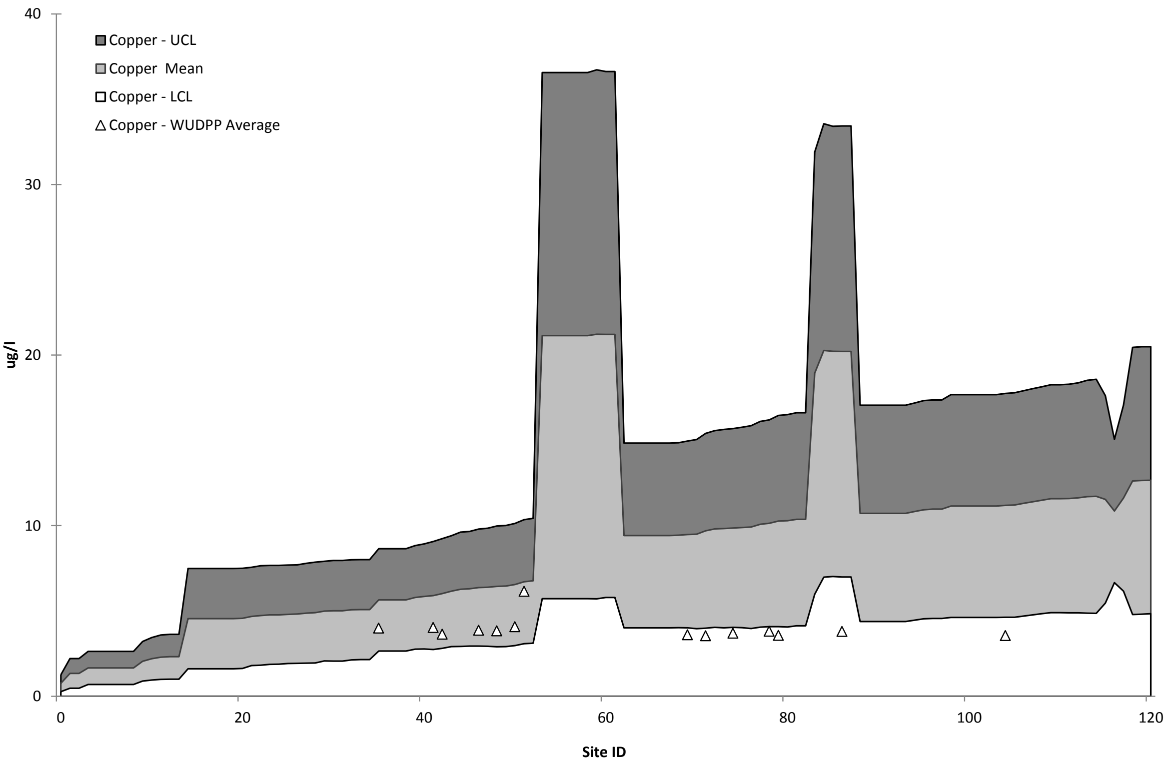


Figure 79 - Comparison of SAGIS predictions against Wigan values – Copper

Figure 81 shows results for Nickel. As with Copper, values fall well within the predicted range in the upper section of the study area but with distance downstream, observed values in samples begin to fall outside the predicted range. The EQS for Nickel is 20µg/l, which is considerably higher than either experienced or predicted values.

Overall when considering this SAGIS cross analysis, it is important to remember that the predictions made by the model are generated using much longer term data sets for average concentrations of the chemicals over a 5 year period. Also the EQS values used are designed to be used to compare data as an annual average figure and the data set collected is over a much shorter term, i.e. over only 16 weeks. One point which reoccurs through this section is that the model predictions are less accurate further downstream and here it generally over predicts values. The strong downward trend in variables that were observed in collected samples is not reflected in the prediction made by the model. This can be partially explained by, firstly, the fact that much of the sampling conducted for the project was undertaken when the river's elevated base flow resulted in comparably more dilution of the pollutants under consideration and, secondly, the absorbance of pollutants by the river system (i.e. by plants and deposition into river sediment) is not considered by the model.

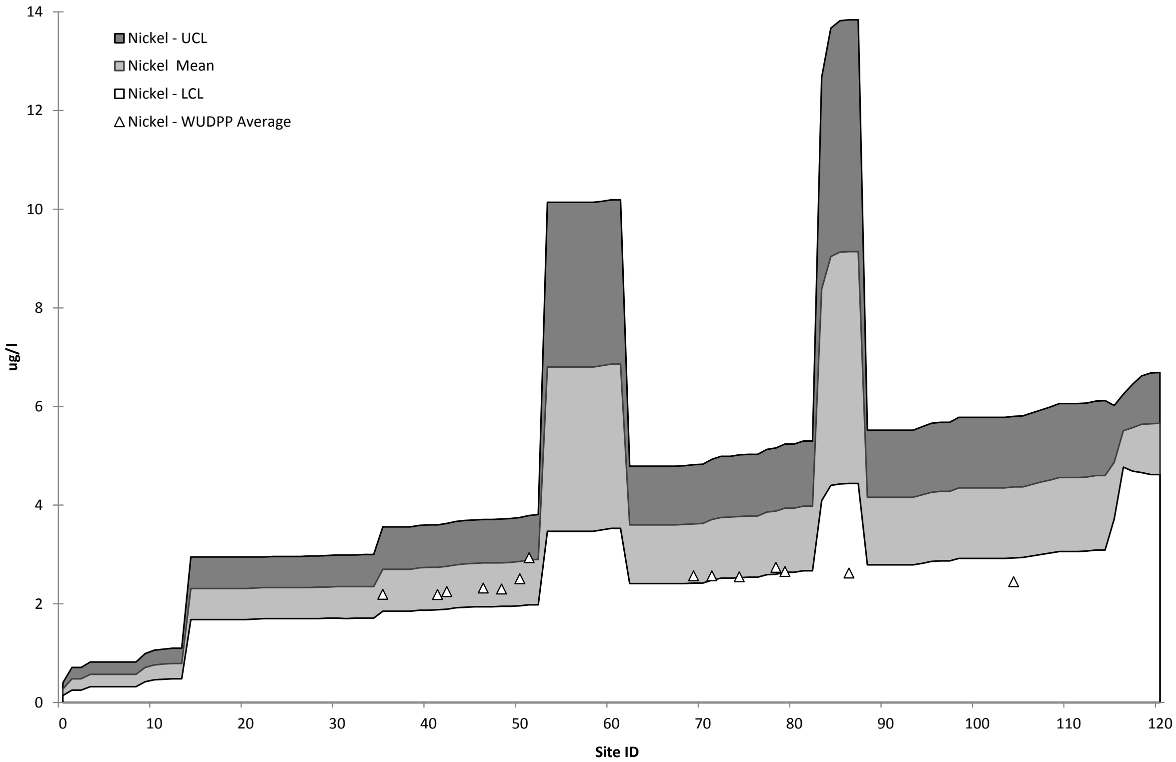


Figure 80 - Comparison of SAGIS predictions against Wigan values - Nickel

4.5 Summary of the River Sampling Results and Discussion

This Chapter has shown that interpreting the extensive results emerging from this study is difficult. The significant number of variables effecting chemical concentrations are problematic, if not impossible, to monitor accurately. To fully understand the response of chemical indicators under certain conditions even for a small section of river is very complex. The impact of the variability of discharge and rainfall is well documented. However, other important factors such as the uptake of chemicals from the water column into flora and fauna that form part of the rivers ecosystem are less well understood. The diluting effects of the watercourse are not well documented; in particular how the effect of increased water volume mitigates increases in surface pollutant wash off, and demonstrating if this results in greater or reduced chemical concentrations in the water column.

The analysis completed in this section, is based on a significant data set of over 200 manually collected water samples. Considering the relatively small dimensions of the study area and the number of samples collected, this represents a considerably dense data set, both in terms of the geographic spacing of sample locations and the frequency at which samples were collected. This density makes the results presented highly reliable and significant. Using this data, a statistical comparison between the samples collected in low and high discharge was undertaken. To do this eleven of the WQV observed in significant concentrations were subject to a T-Test and ten of these variables displayed a statistically significant difference between the (high and low discharge) groups. This is complimented by a series of graphical analyses which display this and also show the fluctuation of the WQV over the duration of typical storm events.

The key general observations made in this Chapter confirm that the fluctuation of WQV is very much dependant on rainfall and discharge, a factor frequently observed in previous studies (Neal *et al.*, 2000; Rothwell *et al.*, 2010b; Stapleton *et al.*, 2008). Further observations also identify that the location of sampling points relative to pollutant input is also very important. In several cases, as a result of the collection of a small number of exceptional samples outside the main range, the overall averages of variables at sites were not representative of typically observed concentrations. In these cases, once the effect of these extreme samples was removed, average concentrations of WQV analysed in detail reduced with distance travelled downstream. This ‘leeching’ effect on variables was more pronounced for some WQV than others, with nutrients showing the strongest change. The cause of this will differ between variables, but can be partially attributed to two factors; dilution due to increased water volumes, and absorption into the river biota. The limited geographic area covered in this study may mean this is not representative of the downstream trend in WQV along the whole of the river's course.

In addition to the general observations discussed above, a series of more specific observations can be made in relation to the water pollution observed in the River Douglas. Values for nutrients were consistently elevated at upstream sites, falling consistently across the study area, with the lowest downstream site having average concentration just 50% of the highest upstream one. This trend was also observed in FIO concentrations, although the effect was less pronounced. Collectively this strongly indicates a consistent polluting discharge above the study area, which is resulting in significant nutrient and FIO pollution. Other trends observed in the data show inputs of Ammonia and metals from several of the Douglas tributaries, principally Clarington Brook, but also Ackhurst, Calico and Barley Brooks. There is also evidence of intermittent pollution at both the Swan Lane and A40 Road Bridge sample sites. Samples with high concentrations for all determinants were collected at these sites.

In the literature review (section 2.3.4.2), the potential for bioaccumulation of metals and uptake of nutrients into the environment was discussed. The impacts can be cumulative, and by the time affected areas are identified, damage to ecosystems may be considerable (J. B. Ellis *et al.* (2002). There is no direct evidence of this from the date available in this study. However the marked downstream reduction that was observed in several WQV could be explained by the uptake of pollutants by flora and fauna. With respect to FIOs the mortality rates of bacteria is a more plausible cause for reductions in FIO sample concentrations, highlighting the need to take samples at frequent intervals (i.e. at less than 4 km spacing) to accurately identify FIO contamination.

The literature review (sections 2.1.1/2.3.7) also identified that the UK environmental quality standards (EQS's) have traditionally been utilised to control water pollution. Under this system, river catchments are divided into 'waterbodies' which often cover a significant geographic area. Typically a single sample point is considered to be representative of the whole waterbody, and a series of samples taken from these locations are used to generate annual average values, which are compared against standards set by the WFD. The sampling in this project has been undertaken using a spread of samples collected at much more frequent geographic and temporal intervals, allowing a more representative insight into the water quality of the River Douglas study area. From the data collected it can be observed that there is a statistically significant difference in the river water quality during periods of elevated discharge. It is also evident, that even across a short section of a river's course there can be significant fluctuation of pollutant concentrations.

Overall this indicates the EQS method of assessing water bodies has distinct disadvantages when observing the contribution of diffuse pollution from surface water because, as this study has shown, a single sampling location may not be representative of a larger area. Considering that EQS sampling may well fail to observe potential problems and hazards caused by sources of an intermittent but significant nature (such as discharges of urban surface water), their contribution may be masked from remote sample locations, by the effects of dilution and downstream leeching observed in this project.

This section has presented the results of sampling to observe the effects of diffuse pollution in the study area of the River Douglas. With this completed, the next section explores the results from sampling undertaken of the PTS installed in the river.

Phase 2 - Proprietary Treatment System Sampling Results

This section begins by reiterating the approach to sampling used in monitoring the selected propriety treatment systems in this study in the context of monitoring practice in other treatment system studies before presenting the results in detail. As identified in the literature review (section 2.5) a range of different approaches have been used when undertaking similar work in previous studies, including:

- Monitoring of scaled down models,
- Monitoring of full scale products in laboratory conditions,
- Field testing products under controlled conditions,
- Full field testing under natural conditions.

The actual monitoring process again differs between these options, but typically is conducted using one of the following methods:-

- Modelling of systems based on known variables and published data,
- A physical assessment of the volumes of pollutant removed over a fixed time,
- Event based influent and effluent sampling.

Each of these approaches has benefits and draw backs for monitoring and this study sought to learn from these experiences (section 2.5). As described in Chapter 3 practical field tests were seen as the most desirable approach and to use a method of monitoring which could be replicated as far as possible to facilitate a comparison between the field tests of the different products. Considering these requirements the use of automated sampling equipment was selected as the most appropriate method of monitoring, similar to that conducted by Langeveld *et al.* (2012).

This method has provided a significant and novel contribution to the knowledge around storm water monitoring and PTS, and the data presented in this section provides valuable insights, in a UK context, into the improvement of water quality through the use of such systems. Monitoring results are displayed for each of the products in turn and, two main sections of analysis are completed on the data from each system:-

- An example of a recorded storm event showing effect of PTS on the downstream concentration of TSS and heavy metals,

- A summary of all other storm events, including the quantity of various recorded pollutants removed.

Following this analysis of product performance, data from all sites is used to characterise the WQV from storm water samples and examine their response in relation to discharge, as well as observing the correlation of different metals to TSS. Using this data an estimate is made in relation to the discharge of pollutants monitored from the study area for the whole of the monitoring period. Finally a summary of the main observations from each site is given.

The method for quantifying performance of each system varied (section 3.10), due to the differences in the functionality of different PTS, and the limitations around each site. This made it unnecessary to complete all sections of analysis on each product. The WQV (such as TSS or Heavy Metals) for which collected water samples have been tested, was informed by the capabilities of each product or system. For example the DD is primarily designed to remove TSS and its associated pollutants such as heavy metals, and it has no effect on the concentrations of pollutants such as PAHs or nutrients. Therefore there would be little value in testing collected water samples for variables which would be unaffected by it. Therefore the most appropriate variables were collected for each PTS installed, with in some cases, additional supplementary analysis completed where necessary to complement the two main sections of work listed above. The duration of monitoring completed at each site is summarised in Table 17, which also shows the type of monitoring used and the total number of water samples collected at each site.

Table 16 - Summary of Monitoring Sites

Site	**Cherry Gardens**	**Coppull Lane**	**Little Wigan Theatre**	**Scott Lane**
Product	DD	X4 Heavy Traffic	Smart Sponge – Passive Skimmers	Smart Sponge – Passive Skimmers
No Water Samples	288	94	0	6
Duration of Monitoring	5 Months	3 Months	12 Months	12 Months
No of Events Captured	10	3	N/A	N/A (1)

4.6 Downstream Defender

The monitoring of the DD at Cherry Gardens was undertaken from the end of October 2013, to the end of March 2014. During this period, 10 separate storm events were captured by sampling. Each of these events was sampled using the equipment and method described in section 3.10.1, which allowed storm events to be characterised in terms of the volume of discharge passing through the system. Using this in conjunction with the concentrations of samples collected during the storm, the corresponding volumes of pollutants both upstream and downstream could be calculated. A subsequent comparison of these values, determined the efficiency with which the DD removed pollutants.

4.6.1 Example Event

The actual process of calculating the removal efficiency of the DD system was repetitive, as the same process was applied to each captured event. To demonstrate the effect of the system on the discharge at Cherry Gardens, a single event has been taken as an example and displayed graphically. Figures 82 and 83 show results for the TSS and Aluminium concentration of samples captured during a storm event on the 7/3/2014. As with the river samples, the metal concentrations in DD samples in the vast majority of cases correlated well with each other, so when one metal was elevated all the others were as well. So again to avoid repetition, Aluminium was selected for display in results as in all cases it was the metal detected most abundantly in sampling.

Metal concentrations observed in DD samples displayed similar ratios to those observed in river samples (section 4.4.3), indicating that the discharge contributed to contaminants observed in the river. There were some exceptions; Lead, Chromium and Cadmium were more commonly observed, whereas these metals were detected less frequently in the river. In order of abundance metals were, in almost all cases, seen in following order of concentration: Aluminium, Zinc, Copper, Lead, Chromium, Nickel and Cadmium. The differences between the ratios and concentrations in comparison with those from the river sampling can be attributed to the fact that DD samples were tested for total metals rather than dissolved concentrations. Secondly the concentrations in the DD samples were generally much higher as there is reduced dilution within a drainage system in comparison with the main river channel. The much high concentrations in ‘total’ samples highlight the importance of TSS as a substrate for metals uptake.

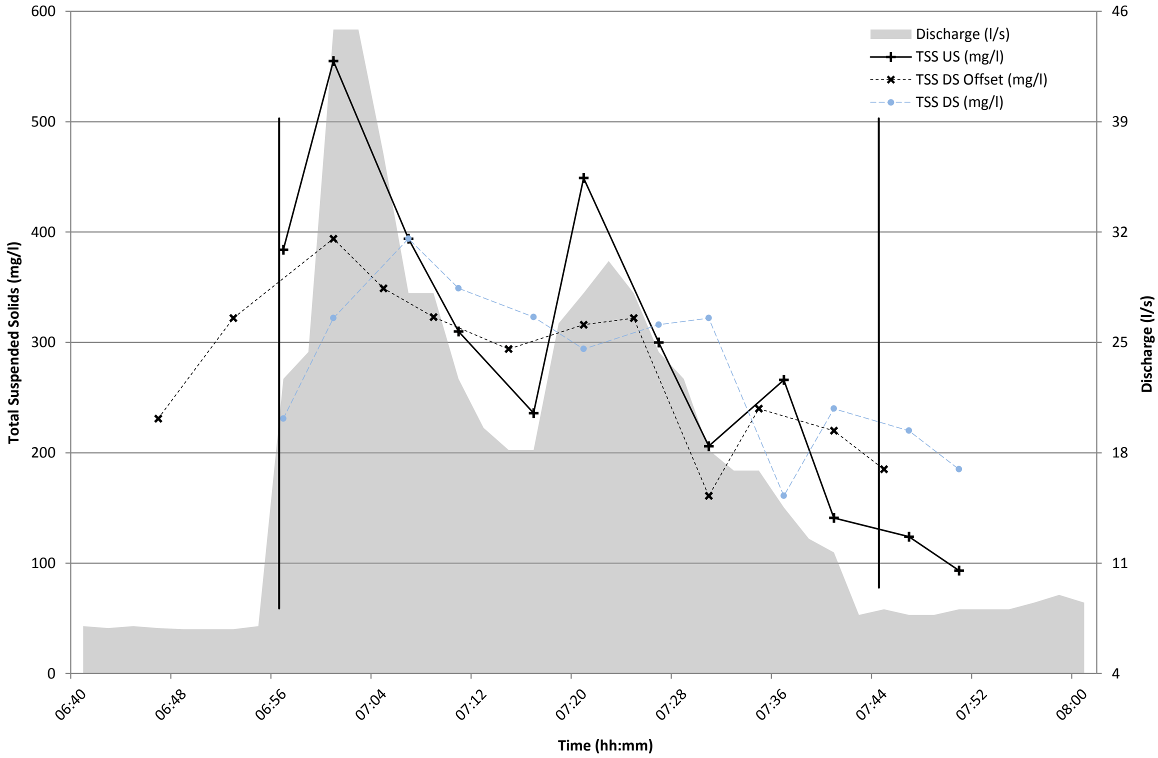


Figure 81 - TSS plotted against discharge from storm event 9 (7/3/14)

Event 9 as shown in (Figures 82 and 83) was significant, commencing at 6:55 and lasting for around 50 minutes with a total discharge of 69m^3. Samples were collected throughout the event, but were only comparable for a 40 minute period once data was offset to account for the 'lag' effect described in section 3.10.3. However, this period still accounted for 61.8m^3 of the total discharge, so the majority of the event in terms of discharge was accounted for. The event consisted of two discharge peaks, the first occurring after approximately 7 minutes (7:02) where discharge reached a maximum rate of just under 45l/s, the second was 21 minutes later (7:23) with a peak just above 30l/s. These peaks produced maximum TSS samples upstream of 555mg/l and 449mg/l respectively, and Aluminium values of 7930µg/l and 6460µg/l respectively. In comparison, the equivalent peak values downstream were 394mg/l and 322mg/l for TSS and 5930µg/l and 4600µg/l for Aluminium respectively.

The event mean concentration (EMC) upstream was 354.3mg/l for TSS and 5462.2µg/l for Aluminium and downstream was 292.6mg/l for TSS and 4722.3µg/l respectively. The total pollutant load for the storm was 23.7 kg of TSS and 358.4mg of Aluminium, of this 5.1 kg of TSS and 56.5mg of Aluminium was removed, equating to 22% of TSS and 16% of Aluminium being retained within the DD chamber.

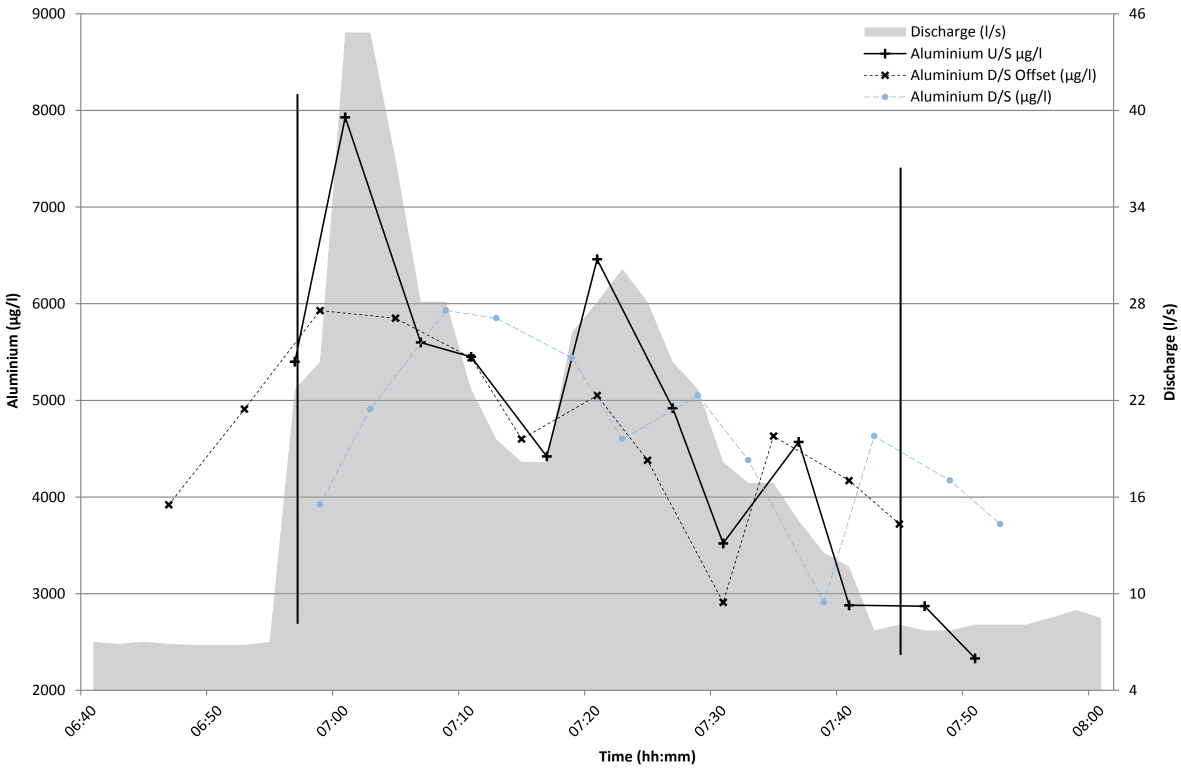


Figure 82 - Aluminium plotted against discharge from storm event 9 (7/3/14)

Figures 82 and 83 each show the upstream concentration, the original downstream concentration and the offset downstream concentration. From these figures it can be seen that there is good correlation between TSS and Aluminium, indicating the importance of TSS as a medium for the transport of heavy metals. There is also very good correlation between TSS and discharge increase, showing how surface water runoff is an important conduit for the transport of TSS and its associated pollutants into rivers. The importance of applying the offset, to account for the lag effect can be clearly seen, the vertical black lines indicate the period of the event which is comparable.

Generally it can be observed from these recorded events, that the upstream sample concentration strongly correlates with changes in discharge, rising or falling in close correlation. In contrast the downstream sample concentration fluctuates much more gradually, so for example in Figure 82 (event 9) it can be seen that following the first peak, the upstream sample concentration falls rapidly between the two peaks, whereas the downstream concentration reduces more slowly before increasing in response to the second discharge peak. To avoid repetition in reporting all the events, only this single one, no.9, has been described in detail. The following section (4.7.2) summarises the key results of other events.

4.6.2 Summary of All Events

To quantify as accurately as possible the actual volume of pollutant discharged through and subsequently removed by the DD, each event has been split into 5 minute sections for its whole duration. The volume of discharge was calculated for each section using the log of level and velocity provided by the flow sensor. Samples were typically collected at 5 minute intervals, so samples were considered to represent the mean concentration of pollutants in discharge for that 5 minute period. The mean pollutant value for each 5 minute interval multiplied by the total discharge is used to calculate the total volume of pollutants in each period.

This has been undertaken for the whole of each event for both TSS and Aluminium, to enable calculation of the total pollutant volumes discharged. The values calculated upstream and downstream can then be compared in terms of the volume of pollutant removed by the system for the duration of an event. These values for each of the storm events monitored are summarised in Tables 18 and 19. The efficiency of the DD chamber in removing pollutants has been found to fluctuate fairly significantly between different events. In all events it was found that the pollutant load downstream was lower than that upstream, showing that the DD chamber has provided a benefit in terms of TSS and heavy metal removal in all events. However the effectiveness of the system fluctuated, on average the DD system removed approximately 21% of the volume of TSS observed , but this ranged from almost 45% at best to just 9% at worst.

In the different events successfully sampled, it has been observed that those where the DD showed the greatest removal efficiency, were those where the largest proportion of the event was captured, in terms of the comparable results. In several events the efficiency of the unit in terms of the volume of pollutants it removed has been shown to be low, and in these cases the beginning of the event was not captured by sampling (i.e. events 1, 3 and 5). Capturing this section of the storm when the highest concentrations of pollutants was typically observed is important, because coupled with high discharge this means that a significant volume of pollutant was not identified in terms of the upstream load. Secondly the downstream load at this point would also be relatively low, as the incoming flux of discharge would have had time to pass through the DD system raising the pollutant concentration there.

Table 17 - Summary of Monitored DD Events (TSS)

Event	Total Event Duration (m)	Total Event Discharge (l)	Average Discharge Per Second	Duration of Comparable Results (m)	Volume of Compariable Discharge (l)	TSS EMC U/S (mg/l)	TSS EMC D/S (mg/l)	No of Samples	U/S Pollutant Load (kg)	D/S Pollutant Load (kg)	Monitored Removal Volume (kg)	% Removal of TSS
1	100	111807.2	18.6	60	98491.7	417.6	407.4	48	51.3	46.7	4.6	9.1
2	45	72613.1	26.9	45	72613.1	194.1	126.9	24	17.4	9.8	7.6	43.8
3	35	68769.3	32.7	25	53747.3	300.0	273.0	12	16.3	15.1	1.3	7.9
4	65	34406.2	8.8	45	24722.7	128.6	109.6	48	3.0	2.6	0.4	14.5
5	220	68737.3	5.2	60	34077.7	105.9	94.9	36	7.2	6.5	0.7	9.8
6	45	22447.4	8.3	40	20238.1	224.6	188.3	24	5.3	4.1	1.2	21.9
7	40	26136.4	10.9	40	26136.4	318.4	232.8	24	10.3	6.7	3.6	35.3
8	40	53300.1	22.2	40	53300.1	186.5	145.2	24	12.5	9.5	3.0	23.8
9	50	68912.1	23.0	40	61838.0	354.3	292.6	24	23.7	18.6	5.1	21.4
10	55	57643.4	17.5	45	50007.7	362.7	308.8	24	23.5	17.9	5.5	23.6
Average	**69.5**	**58477.3**	**17.4**	**44**	**49517.3**	**259.3**	**217.9**	**28.8**	**17.1**	**13.7**	**3.3**	**21.1**

Table 18 - Summary of Monitored DD Events (Aluminium)

Event	Total Event Duration (m)	Total Event Discharge (l)	Average Discharge Per Second	Duration of Comparable Results (m)	Volume of Compariable Discharge (l)	Al EMC U/S (µg/l)	Al EMC D/S (µg/l)	No of Samples	U/S Pollutant Load (mg)	D/S Pollutant Load (mg)	Monitored Removal Volume (mg)	% Removal of Al
1	100	111807.2	18.6	60	98491.7	6527.7	5104.1	48	727.8	559.1	168.7	23.2
7	40	26136.4	10.9	40	26136.4	6538.8	5497.5	12	201.7	160.6	41.1	20.4
8	40	53300.1	22.2	40	53300.1	5675.0	2827.5	12	386.5	194.1	192.4	49.8
9	55	71248.8	21.6	40	61838.0	5462.5	4722.3	24	358.4	301.9	56.5	15.8
10	55	57643.4	17.5	40	47742.0	6755	5751.5	12	387.4	298.4	89.0	23.0
Average	**58**	**64027.2**	**18.2**	**44**	**57501.7**	**6191.8**	**4780.6**	**21.6**	**412.4**	**302.8**	**109.5**	**26.4**

Five of the ten events were also monitored for heavy metal concentrations, and as has been previously discussed, Aluminium has been used as the indicator metal to report. Aluminium pollution loads have been calculated using the same method as TSS, and this was undertaken for each of the events; results are displayed in Table 18. Again the effectiveness of the DD at capturing Aluminium fluctuated between events, with on average just over 25% of Aluminium being removed, ranging from almost 50% to just below 16%. Generally for the removal of Aluminium, the DD was found to be more effective than for TSS. This can be partially explained by the fact, that four of the five events monitored were storms that were well captured by monitoring and TSS performance was also good. It also needs to be recognised that assessment of Aluminium removal in three out five events was made with samples collected at a lower frequency than that for TSS, so results will be of lower significance.

As can be seen from the Figures 82 and 83 and Tables 18 and 19, the surface water drain (SWD) at Cherry Gardens is discharging significant amounts of TSS and heavy metal pollutants on a single event basis. This discharge is fairly typical, in that it does not drain a significant area (i.e. approximately 2 km^2) and the pipe size (i.e. 0.225mm Diameter) in terms of a SWD is fairly modest. Despite this significant volumes of pollutants were observed passing through the system. In event 1 for example, over 50kg of TSS was discharged in a one hour period. It should also be noted that the proportion of the pollutants in the discharge would be decreased as a consequence of the base flow passing through it, as it would dilute the samples taken. This also means that the whole pollutant load observed in each event is due to the surface wash off, as it is unlikely that much material would settle within pipes, because of the constant discharge washing them through.

4.6.2.1 Reasons for performance fluctuation

There are a number of things to consider when explaining the reasons why the efficiency of the DD fluctuated. One of the most important reasons was the unavoidable fact that different events had different intensities, in terms of the average discharge, this varied between 5-33l/s and there was not, as would be expected, a correlation between this and removal efficiency. It was also anticipated that smaller storms with lower discharges would experience greater efficiency as lower flow rates would allow improved settlement, however this was not the case in the events recorded.

Secondly, it is important to consider the monitoring process itself, which was a complex and labour intensive process. The chief drawback was the lack of a second flow meter to allow monitoring of the discharge in the downstream sample chamber, necessitating the application of the offset equation to downstream results. Another factor was the fluctuating base flow, which made it very difficult to reliably trigger the sampling equipment and capture rainfall events in their entirety, especially the crucial first period. The level rather than the velocity output from the flow sensor, was found to be the most reliable way of triggering the sampling equipment, as the velocity readings at base flow were often intermittent due to the turbulence of the discharge interfering with the Doppler signal used by the sensor to determine velocity. Often as base flow levels fluctuated, and the threshold for the level trigger had to be reset to avoid being erroneously triggered by these fluctuations.

At least as many events that were captured successfully were missed due to the triggering of the sampling equipment through fluctuations of the base flow, which also precluded the possibility of capturing smaller events, where the DD performance may ultimately have been shown to be improved.

As well as causing difficulties with the monitoring process, the base flow through the system was detrimental to its efficiency. The presence of the continued discharge through the system following events, would cause the washing through of finer TSS material still suspended in the main DD chamber, that had not yet had time to settle, and become stored in the DD's sump section. Certainly in all monitored events, the TSS concentrations in downstream samples remained persistently high, even in the final stages of events, so it is likely that the base flow in the system contributed to this. In a system with no base flow, following the end of storm discharge, even the fine material left in suspension in the DD chamber has time to settle out, whereas the base flow in the monitored system would certainly wash a proportion of this material out.

In the literature review (2.3.4.3) the importance of particle size was emphasised, in respect of which fraction of sediments has the greatest affiliation with pollutants. There is contention over this, with some studies suggesting that over 65% of particulate matter in SWD effluents was smaller than 75 µm (Ingvertsen *et al.*, 2011; J.-Y. Kim and Sansalone, 2008). Hydro International state (Hydro International, 2014) that their DD is effective in removing TSS particle sizes of 106 µm and above. To give some insight into the particle size distribution in samples collected from the DD system, a small number of additional samples were collected and analysed for particle size (using Horiba particle size analyser la-950).

The aim of this was to observe if there was any clear difference in the particle sizes in samples from upstream and downstream. In the limited number of samples tested no significant difference was observed, with mean PSD of samples from upstream and downstream being very similar, both displaying average mean size ranging between 15-16 µmm. While the Horiba instrument could not measure the proportion of the mass each fraction of particles contributed, it identified that the proportion of particles in terms of abundance was largely below the 100 µm range of the DD, providing further reasons for the lower than expected efficiency. Figure 84 shows an example of the output of the particle size analysis and, as can be seen, the majority of particles fall under the 100 µm mark. Only 15 of the separate samples were analysed for PSD, however from each 1L sample collected three sub samples were taken to demonstrate repeatability. Figure 84 shows two results from the upstream and corresponding downstream sample from a consistent run of samples. From the graph it can be seen that no particles over approximately 250µm were detected whereas in the downstream sample a further fraction of particles between 250µm and 1000µm was observed.

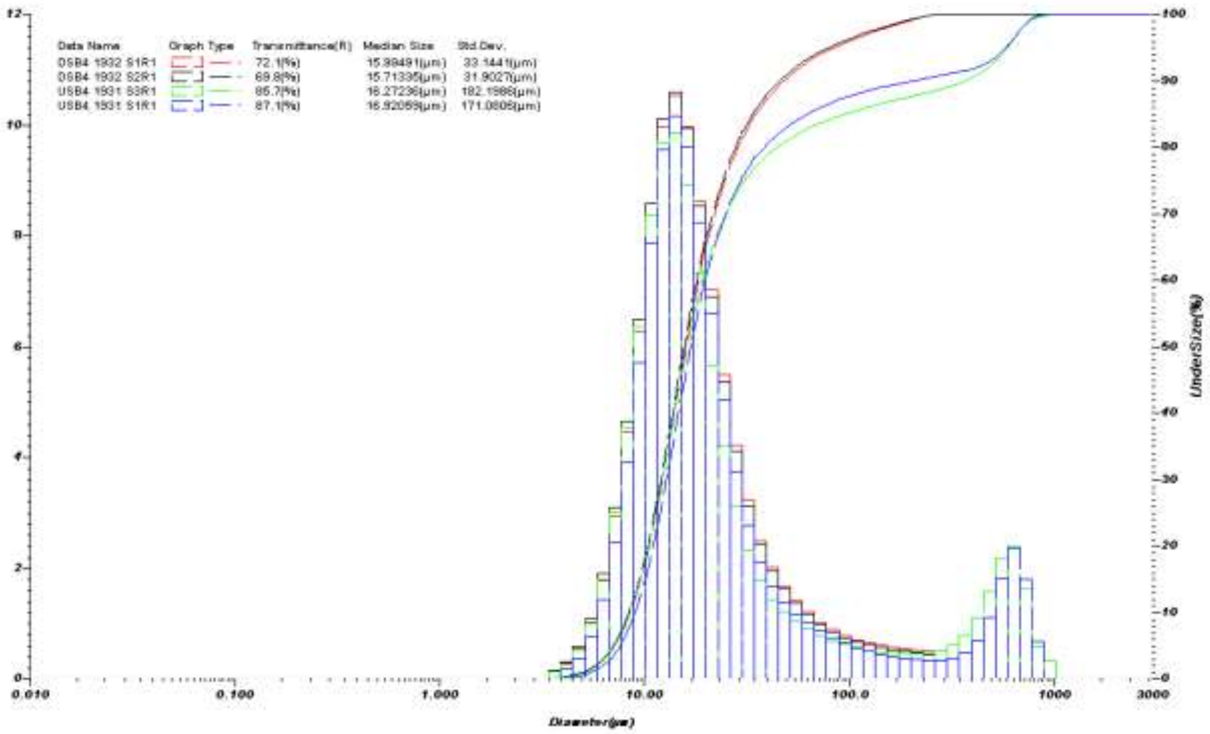


Figure 83 - PSD Graph (output from Horiba la-950)

The mean PS of the samples was not significantly different, but the standard deviation of the upstream samples was considerably higher given the presence of an extra portion of PS not observed in the downstream samples. While it is limited, this data indicates that the DD was removing the majority of the larger particles observed in samples, although further data and observations are needed to provide more significant results. This concludes the presentation of the DD results. The next section covers the results from the X4.

4.7 Storm X4 Heavy Traffic

The monitoring of the X4 at Coppull Lane was undertaken from the beginning of April 2014 until the end of June 2014. The method used to monitor the X4 was very similar to that of the DD due to the similar nature of the products in terms of their function and how the components were arranged, i.e., the way that diversion chambers and treatment chambers were arranged. Since the X4 data was collected and analysed in a similar manner to that of the DD, many of the problems and difficulties around the practicalities of sampling had been encountered before. However, there were other difficulties with the X4 sampling which meant that ultimately only 3 events were fully captured.

The first difficulty, as previously explained, was the requirement to locate the flow sensor in the downstream sample chamber. This was necessary after it was found that, due to the very low fall across the site, the inlet to the upstream manhole remained full of water at all times. This was not appreciated from the outset of sampling, and meant that using the level trigger was not possible. Secondly after a series of events were missed, it was found that the communication cable between the sample units, which allowed the downstream sampler attached to the flow sensor to trigger the upstream programme was faulty. Finally and most significantly, it was found the upstream manhole of the system surcharged, which disturbed the sample unit located there, and resulted in the loss of all upstream samples on three separate occasions. Collectively taken together, these problems meant that a disproportionate period of the time available for sample collection was wasted, and the target of collecting a minimum of five events was not achieved.

In other respects, the sampling of the X4 system was more straightforward than that for the DD because, as is more typical of SWD, there was no permanent base flow. This made capturing events much easier, as it meant equipment was not triggered accidentally as a result of base flow fluctuation. This also meant that the trigger level could be set lower, so events could be sampled more completely. The volume of the X4 system was considerably less (approximately $3m^3$) than that of the DD, and therefore there was no appreciable lag visible in the results. This removed the need to make offsets to calculate the removal efficiency of the system. These key differences between the two sites, led to some unavoidable variation in the monitoring and analysis process. However in all other respects it was completed in as similar a manner as possible, i.e., sample frequencies, collection, transport and analysis, etc.

4.7.1 Example Event

The process of calculating the removal efficiency of the X4 system is identical to that used for the DD, and again to avoid repetition an example event is explored graphically, followed by a summary of all other captured events. Figures 85 and 86 show the results of an event captured on the 2/6/2014, and as with the DD, results for TSS and Aluminium (Aluminium being used as a surrogate for other metals tested) are presented. As it is advertised as having the capability to remove dissolved metals, samples collected at the X4 site were also tested for them, so an additional graph (Figure 87) showing Aluminium Dissolved (dissolved) is also included. Metals were observed in the same ratios as in river and DD samples, with Aluminium being the most abundant and other metals occurring in lower concentrations but, in the vast majority of cases, at the same ratios.

Event 2 (Figures 85-87) commenced at 15:13 and lasted around 40 minutes, with a total discharge of 2287.1 litres. Samples were collected throughout the event, but account for a 35 minute period. This however, included 2178.7 litres of the total discharge, so the majority of the event in terms of discharge was accounted for. The event consisted of a single peak which occurred very early, just 3 to 4 minutes after the start of the event (15:17), when discharge reached a maximum rate of just 6 l/s. The storm peak produced an upstream maximum TSS sample of 613mg/l and Aluminium sample of 8020µg/l. This is in comparison to the equivalent peak value downstream, of 314mg/l for TSS and 4690µg/l for Aluminium.

The event mean concentration (EMC) upstream was 327.6mg/l for TSS and 2250µg/l for Aluminium, in comparison with downstream values of 121.5mg/l for TSS and 962µg/l. The total pollutant load for the storm was 2.1kg of TSS and 16.9mg of Aluminium, of this 0.6kg of TSS and 7.7mg of Aluminium was removed, equating to 55.6% of TSS and 45.8% of Aluminium being retained within the X4 chamber. Dissolved Aluminium (Figure 87) showed unusual behaviour, in that the downstream concentration starts fairly high but then falls gradually during the event, whereas upstream concentrations increased throughout the event before reducing in the final sample collected after the event had finished. One explanation of this anomaly could be that any standing water in the base of the pipe downstream could have already been fairly abundant in dissolved Aluminium resulting in an initially high downstream concentration.

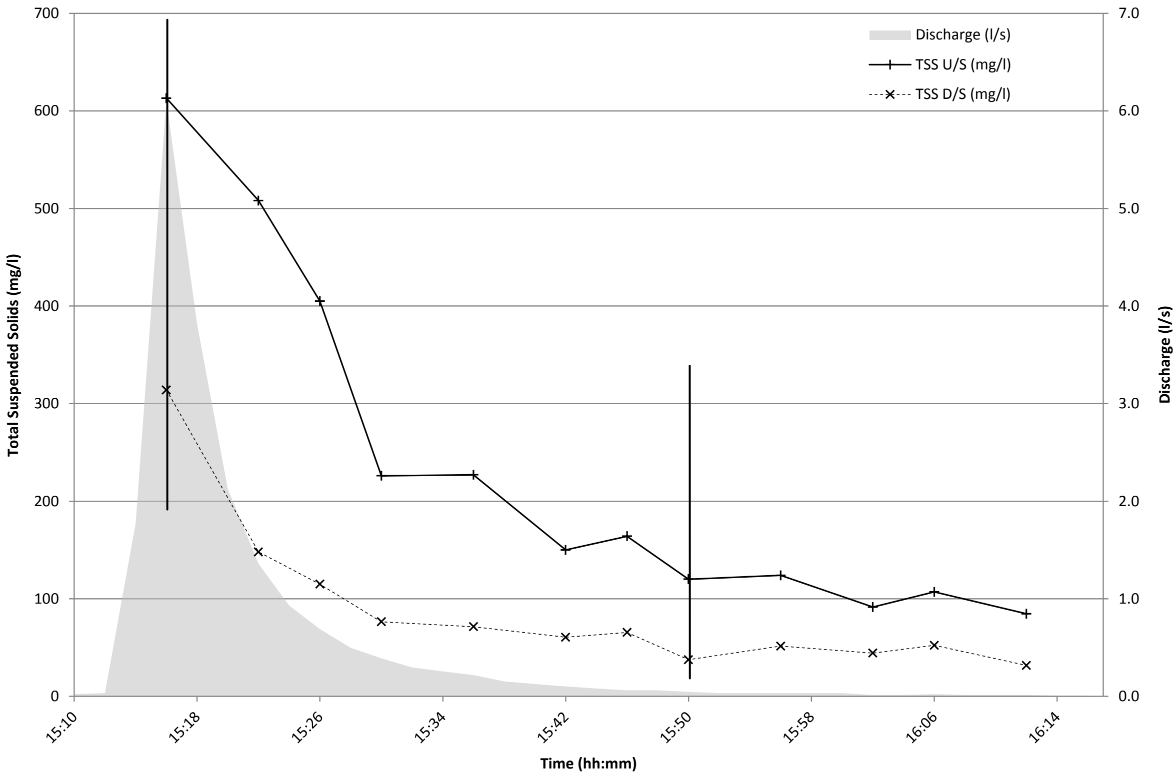


Figure 84 - TSS plotted against discharge from storm event 2 (2/6/14)

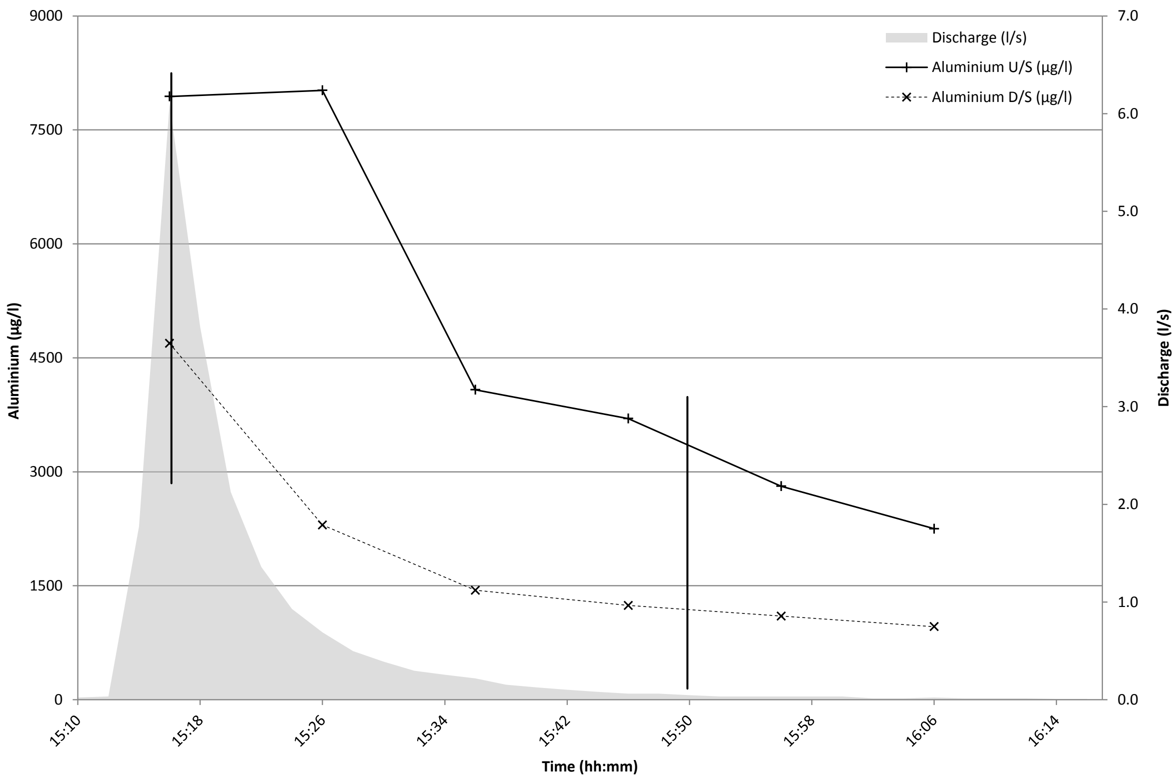


Figure 85 - Aluminium plotted against discharge from storm event 2 (2/6/14)

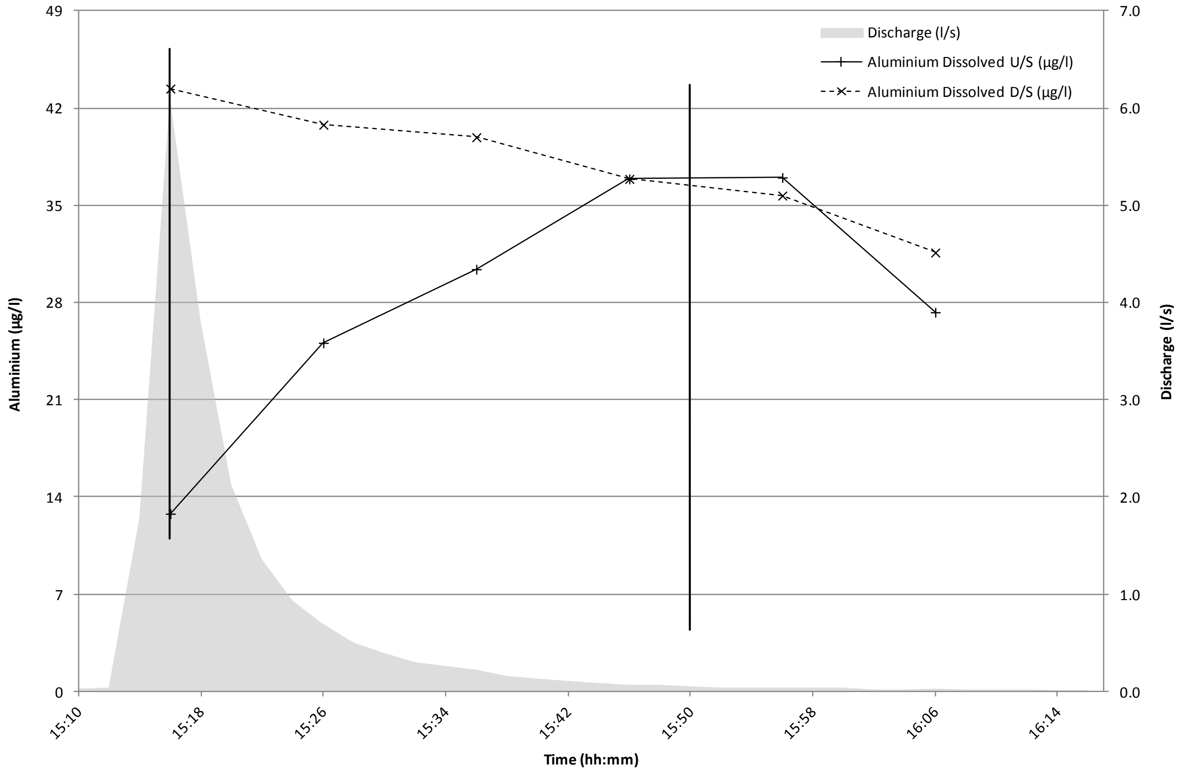


Figure 86 - Dissolved Aluminium plotted against discharge from storm event 2 (2/6/14)

As with the results for the DD from Figures 85-87 it can be seen that there is good correlation between TSS and Aluminium, indicating the importance of TSS as a substrate for the transport of heavy metals. There is also very good correlation between TSS and increased discharge, further reinforcing that surface water runoff is a conduit for the transport of TSS and its associated pollutants into rivers. Here there is less correlation between dissolved Aluminium and discharge, however other dissolved metals from the event showed more normal behaviour (increase during discharge) and this is evident when observing Figure 88 which shows dissolved Zinc concentrations during the event. Here it can be seen that both up and downstream concentrations increase in unison, remaining at similar levels throughout the event. Whilst it was not necessary to apply any data offset with respect to the up and downstream results, invariably results will not capture events perfectly. In event 2 however there was good coverage.

Again, as with the DD data, it was generally observed in events that the upstream sample concentrations strongly correlate with changes in discharge, increasing or decreasing accordingly. However, conversely, while DD downstream sample concentrations fluctuate much more gradually than the upstream, for the X4, downstream sample concentration (for

TSS and Aluminium) generally fluctuates in line with the upstream value, but in lower concentrations. This is likely to be due to the difference in design. The DD functions only as a vortex separation unit, whereas the X4 works as a vortex separator and has additional filtration capabilities. Whilst this reduces the flow rate it is able to achieve, it should result in a higher level of treatment.

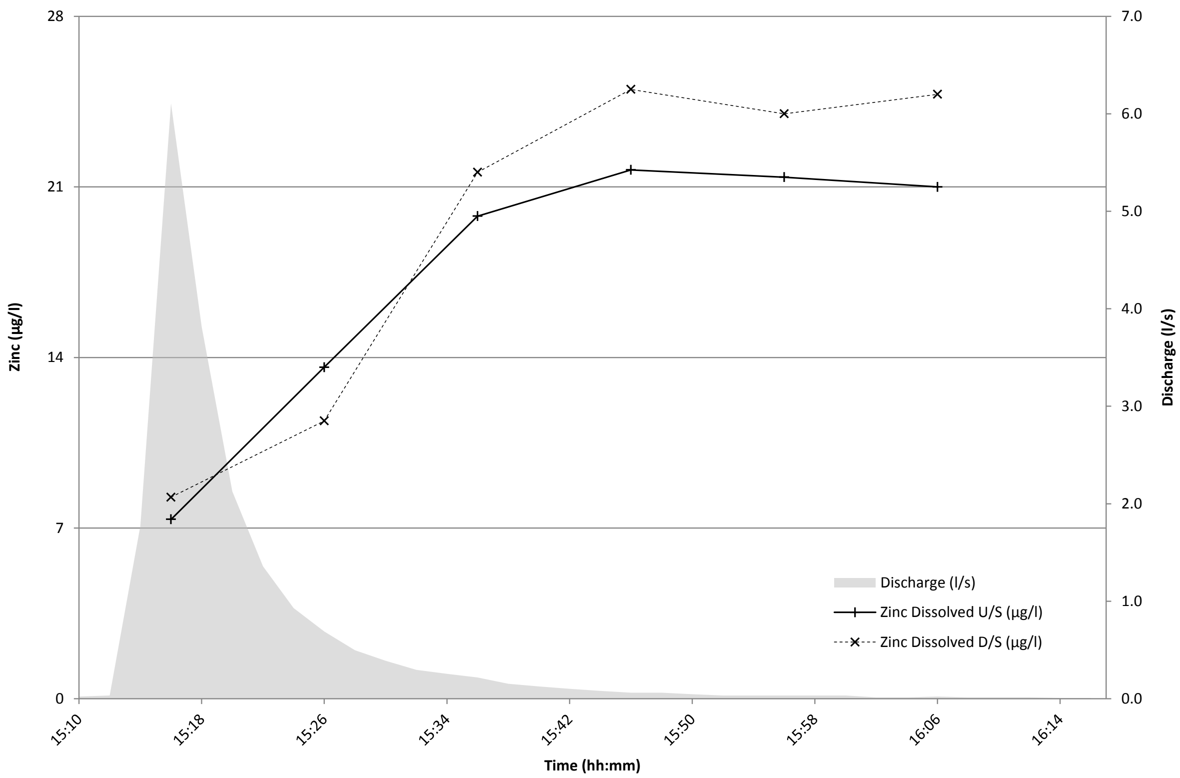


Figure 87 - Dissolved Zinc plotted against discharge from storm event 2 (2/6/2014)

4.7.2 Summary of All Events

To quantify as accurately as possible the actual volume of pollutant discharged through and subsequently removed by the X4, the same process that was undertaken for the DD was completed. Events were divided into 5 minute intervals, with pollutant loads multiplied by the discharge during each section. This was completed for the whole of each event for TSS, Aluminium and Dissolved Aluminium. The results are summarised in Tables 20-22.

The efficiency of the X4 system in removing pollutants has also been found to fluctuate between events, and in all, pollutant load downstream was lower than that upstream, demonstrating the system provides a benefit in terms of TSS and total Aluminium removal. However the effectiveness of the system fluctuates, on average the X4 system removed approximately 39% of the volume of TSS, ranging from almost 55% at best to just 21% at worst. However, it needs to be considered, that this analysis is based on a much smaller data set than that collected during the DD monitoring phase. Also important, is that two of the events monitored were much smaller in terms of average discharge, than those monitored at the DD Cherry Gardens site, so the expectation is that removal efficiency should be improved. This tendency is confirmed by the final event monitored, where average discharge was much higher and conversely efficiency was reduced to just over 21%.

Several factors suggest reasons for the increase in efficiency observed. The lack of base flow meant that events were captured more holistically. Secondly, the fact that the X4 should be capable of high removal rates, on account of its additional filtration capabilities, which is balanced against its reduced ability to treat higher discharges. For approximately six minutes of event three the flow rate was elevated above 28l/s (i.e. the X4 maximum treatment capacity) and during this period, the up and downstream TSS concentration were very similar, indicating that the high discharge resulted in water bypassing the filtration packs and passing through the internal overflow. As the X4 system had additional filtration capabilities it was able to remove TSS material with finer particle sizes, and therefore this was less important to its efficiency. So it was unnecessary to undertake additional analysis with respect to particle size distribution.

Table 19 - Summary of Monitored X4 Events (TSS)

Event	Total Event Duration (m)	Total Event Discharge (l)	Average Discharge Per Second	Duration of Comparable Results (m)	Volume of Compariable Discharge (l)	TSS EMC U/S (mg/l)	TSS EMC D/S (mg/l)	No of Samples	U/S Pollutant Load (kg)	D/S Pollutant Load (kg)	Monitored Removal Volume (kg)	% Removal of TSS
1	90	21357.1	4.0	80	18717.6	66.1	44.3	32	2.1	1.2	0.9	41.6
2	40	2287.1	1.0	35	2178.7	327.6	121.5	24	1.2	0.5	0.6	55.6
3	65	46663.5	12.0	55	45429.3	272.9	211.5	38	20.0	15.6	4.4	21.8
Average	**65**	**23435.9**	**5.6**	**56.7**	**22108.6**	**222.2**	**125.8**	**31.3**	**7.7**	**5.8**	**2.0**	**39.7**

Table 20 - Summary of Monitored X4 Events (Aluminium)

Event	Total Event Duration (m)	Total Event Discharge (l)	Average Discharge Per Second	Duration of Comparable Results (m)	Volume of Compariable Discharge (l)	Al EMC U/S (μg/l)	Al EMC D/S (μg/l)	No of Samples	U/S Pollutant Load (mg)	D/S Pollutant Load (mg)	Monitored Removal Volume (mg)	% Removal of Al (Total)
1	90	21357.1	4.0	80	18717.6	740.1	586.8	16	19.4	15.2	4.3	21.9
2	40	2287.1	1.0	35	2178.7	2250.0	962.0	12	16.9	9.1	7.7	45.8
3	65	46663.5	12.0	45	44635.5	3694.0	3107.8	30	226.5	177.9	48.7	21.5
Average	**65.0**	**23435.9**	**5.6**	**53.3**	**21844.0**	**2228.0**	**1552.2**	**19.3**	**87.6**	**67.4**	**20.2**	**29.8**

Table 21 - Summary of Monitored X4 Events (Aluminium Dissolved)

Event	Total Event Duration (m)	Total Event Discharge (l)	Average Discharge Per Second	Duration of Comparable Results (m)	Volume of Compariable Discharge (l)	Al EMC U/S (μg/l)	Al EMC D/S (μg/l)	No of Samples	U/S Pollutant Load (mg)	D/S Pollutant Load (mg)	Monitored Removal Volume (mg)	% Removal of Al (Dissolved)
1	90	21357.1	4.0	80	18717.6	27.7	27.3	16	0.6	0.5	0.1	8.8
2	40	2287.1	1.0	35	2178.7	24.8	40.7	12	0.0	0.1	-0.1	-177.6
3	65	46663.5	12.0	45	44635.5	38.3	37.1	30	1.7	1.8	0.0	-0.9
Average	**65.0**	**23435.9**	**5.6**	**53.3**	**21844.0**	**30.2**	**35.0**	**19.3**	**0.8**	**0.8**	**0.0**	**-56.6**

In all of the 3 events monitored, samples were collected for heavy metal analysis, the effectiveness of the X4 at capturing Aluminium fluctuated between events, with on average just under 30% of Aluminium being removed, ranging between a maximum of almost 46% to a minimum of just over 21%. Generally the removal of Aluminium was found to be lower than that of TSS, with the system showing better performance than that for dissolved Aluminium. Results for dissolved Aluminium (Table 22) show that the X4 did not perform well in respect to removal, with only the first event displaying a small reduction. The second displayed a very small increase but as the quantity is low, this equates to a large percentage increase. The final event again showed a small increase. Generally it was found that at best dissolved metal levels were very slightly reduced, or completely unaffected by the system.

As with the SWD at Cherry Gardens the discharge at Coppull Lane was not significant in relation to the overall river flow. Since a 0.225mm diameter pipe which drains a similar area (of approximately 2 km^2). However substantial volumes of pollutants are again evident over short storm periods with 20kg of TSS being observed in the last event monitored. Overall in terms of the monitored events, the X4 performed well with the sampling clearly showing beneficial reductions in pollutants. However due to a much smaller number of events captured, results may be of lower significance.

4.8 Principle Component Analysis of Water Quality Data

A reoccurring theme identified in the literature review is that there is a well-established causal link between increased rainfall, discharge and elevated pollution concentrations. However there is a less established relationship between different WQV. In order to examine this relationship statistically and identify the principle quality variables a Principle Component Analysis (PCA) analysis was completed.

Before conducting PCA the data was mean centred and normalised, to remove bias and allow data which maybe numerically different to be compared. Initially the whole group of 39 variables was included in the analysis. However this group was reduced to 14, as variables that did not contribute significantly to the variance of the data set were removed. PCA should be conducted with a minimum sample of 50 cases, ideally over 100 and the ratio of cases to variables should be at least 5 to 1. Also PCA requires that the Kaiser-Meyer-Olkin (KMO) measure of sampling adequacy be greater than 0.50 for each individual variable as well as the set of variables. The KMO values for all of the individual selected 14 variables included in the analysis was greater than 0.5, supporting their retention in the analysis. In addition, the overall KMO measure for the set of variables included in the analysis was 0.807, which exceeds the minimum requirement of 0.50 (Table 5). Similarly, Bartlett's test is less than 0.0001 indicating the suitability of using PCA to analyse this data.

Table 22 - KMO and Bartlett's Test form PCA

KMO and Bartlett's Test		
Kaiser-Meyer-Olkin Measure of Sampling Adequacy		0.807
Bartlett's Test of Sphericity	Approx. Chi-Square Df Sig.	3204.785 91 0

PCA results show that from the original fourteen, three of the components together account for approximately 71% of the total variance. The first component accounted for 42% the second for 15.6% and the third for 13.8%. It can be seen in Figure 52 that the analysis grouped the variables into one distinct and two less distinct groups. Closer clustering of variables within the plot indicates a stronger relationship. It can be seen from the graph that *E.coli*, IE and Biochemical Oxygen Demand (BOD) form a clear clustering group, showing that all three variables were present in samples in similar proportions. A second distinct group is formed by the 4 heavy metals (namely, Copper, Nickel, Zinc and Aluminium) that are included in the analysis along with TSS. A third less distinct group is formed by Phosphorus, Carbon, Ammonia, Alkalinity and conductivity.

Examination of the clustering identified by the PCA analysis confirms some of the observations from the literature review. For example several sources identified that there is an affiliation between heavy metals and TSS as metals bind to solid substrates (see section 2.3.4.3). Similarly the clustering of FIOs and BOD indicates that the faecal bacteria are contributing to the BOD readings in the samples. This will have connotations when considering which remediation measures would provide the greatest benefit, a system for example that removed TSS, through settlement or filtration would have an associated benefit as metal pollution with TSS would also be removed by the system.

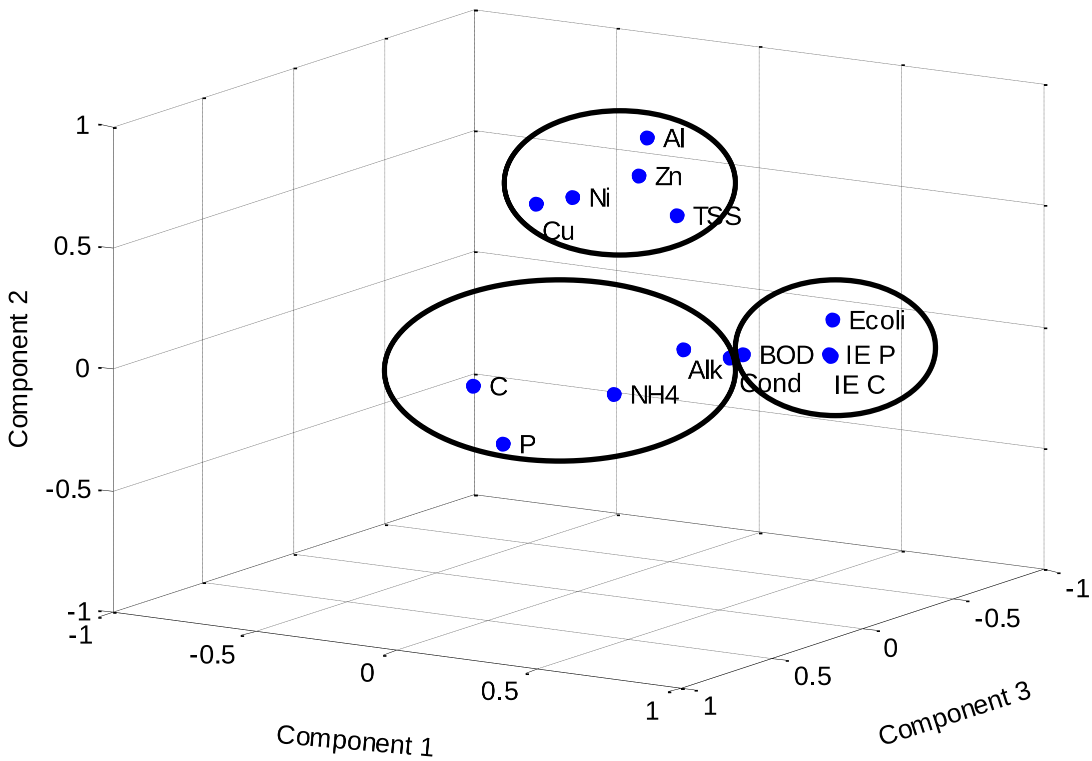


Figure 88 - PCA Component Plot and Clusters Formed

4.9 Smart Sponge

The primary function of the Smart Sponge is to remove hydrocarbons and Free Phase Oils from the water it comes into contact with. This process is described in section 3.10.4. It also intercepts sediment, which is captured by both the sponge and the bag that encloses it. Unlike the two HDVS covered in the previous sections, the Smart Sponge is a passive product, and this means observing and quantifying its effect on an event basis is not appropriate. The preferred method of quantification has been to leave the Smart Sponge containers, known as skimmers, insitu for a period of time, and comparing their before and after weights determine the quantities of hydrocarbons they have absorbed.

The heavy siltation experienced with the skimmers already described meant that subsequent analysis would clearly encapsulate the weight, not only of the hydrocarbons absorbed into the sponge structure, but also captured TSS material. Therefore, to determine the quantity of each fraction captured, the skimmers were dried, weighed, cleaned then re dried and re weighed. This process (explained in detail in section 3.10.5) was replicated for all skimmers recovered from each of the two Smart Sponge sites.

4.9.1 Scott Lane

The design, construction and monitoring of the Smart Sponge cage at Scott Lane is described in detail in section 3.10.4. Following the installation, the Smart Sponge was left in situ for just over one year. After this period, the cage (which contained 40 individual passive skimmers) was removed and Smart Sponge skimmers were cleaned, dried and weighed as described in section 3.10.5. This allowed separate quantification of the sediments intercepted by the skimmers and the volume of hydrocarbons the sponge had captured within its structure. The results of this are displayed in Figure 89, which shows the average sediment load of the skimmers to be 282.4g, with a range between 126 to 692g, and the average weight of absorbed hydrocarbons as 15.6g, with a range from 0 to 77g. Out of 40 bags, 9 recorded a reduction in weight, i.e., collectively the 12 sponge pieces within each skimmer were a lower weight than the average weight of 12 pieces of clean Smart Sponge.

The reasons for this reduction in weight are unclear; however there are a number of potential explanations. One of the first to consider is that significant volumes of hydrocarbons or other free phase Light Non-Aqueous Phase Liquid were just not present within the discharge. Water samples to identify this were not collected for either Little Wigan Theatre or Scott

Lane. Another explanation is that significant turbulence in the effluent, and very minimal contact time with the sponges, in conjunction with high blinding of the skimmers caused by build-up of sediment, has limited their ability to absorb any hydrocarbons that were present.

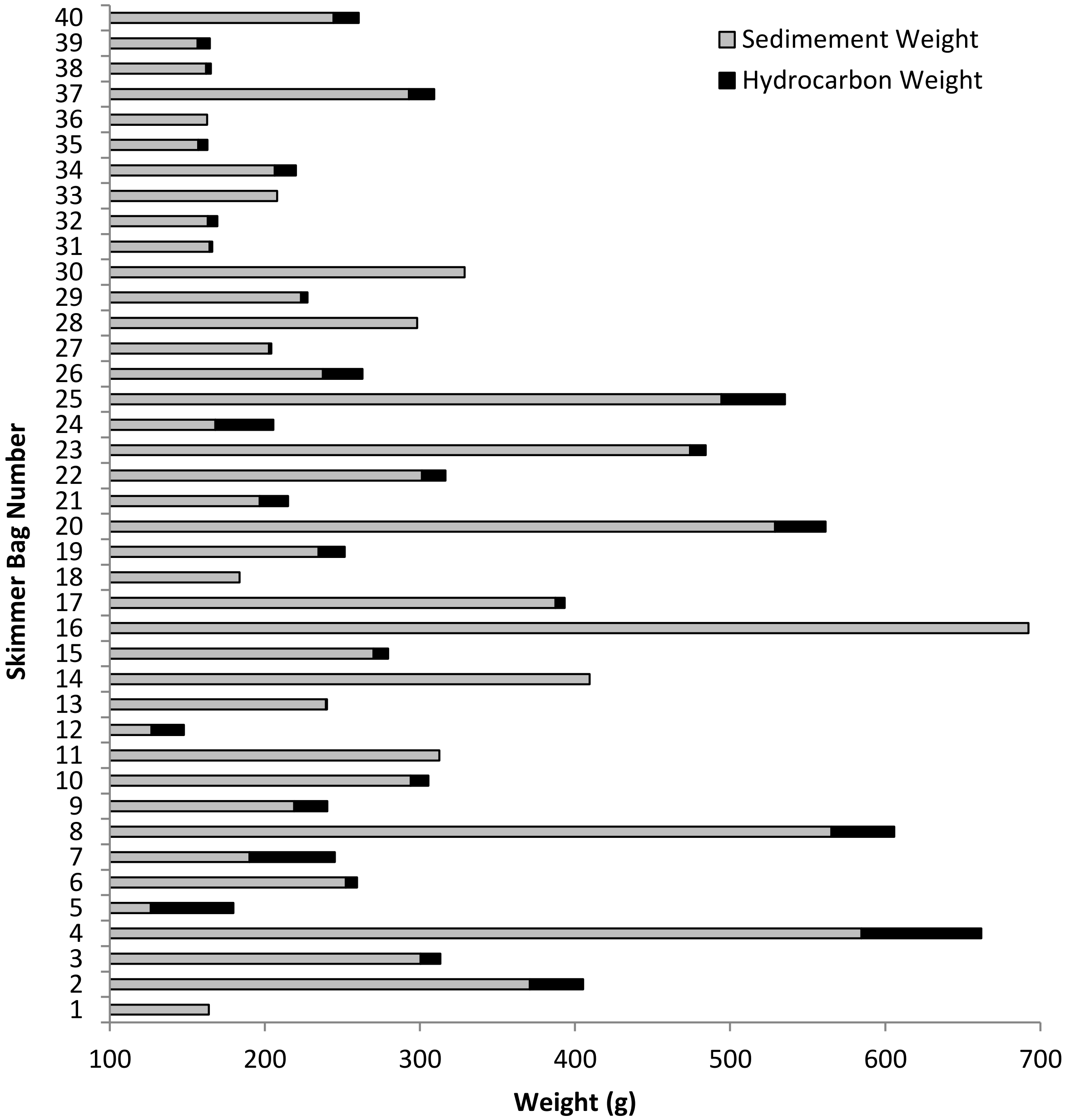


Figure 89 - Pollutant weight of skimmers removed from Scott Lane Site

While the cleaning process could have resulted some of the weight loss of sponges there was visible degradation of the sponge prior to the cleaning process, also the manufacturer confirmed that this process should not result in oil losses from the sponges. Further explanation for the loss of weight of sponges could be as the result of volatilisation, this process is particularly important when considering lighter hydrocarbons such as petroleum products, naphthalene for example has a volatilisation half-life of just 0.5-3.2 hours. Generally evaporation is the primary process behind the loss of volatile and semi volatile components of spilled oils (Lollar, 2005). Without further more rigorous testing of sponge materials following removal from sites it is not possible to determine the exact cause of perceived weight loss, however it is considered to be as a result of a combination of the process described above.

From visual inspection of the Smart Sponge, it was evident that significant degradation to the structure of some sponge pieces occurred (show in Figures 90 and 91). It is probable, that other chemicals present in the discharge, such as solvents, contributed to the degradation of some sponge pieces, and although they were protected by the netting, the considerable physical action of the discharge could also contribute to their breakdown. In such cases, it is likely that hydrocarbons were absorbed by the sponge pieces, but due to degradation their weight has still decreased. Also, due to its vigorous nature, some parts of the sponge structure could have been lost in cleaning.

Figure 90 - (Left) Disintegrated piece of Smart Sponge

Figure 91 - (Above) Some sections were significantly degraded

The fact that hydrocarbons, as a general group, will be present in the discharge in both solid particulate and liquid form also needs to be considered. Any hydrocarbon present in solid form (such as PAH particulates) would not be absorbed by the Smart Sponge. Without chemical testing of the TSS material removed from the sponge pieces during washing it is not possible to ascertain what it composed of.

4.9.2 Little Wigan Theatre

At Little Wigan Theatre 40 passive skimmers were installed, dispersed in 40 separate road gullies, as opposed to being sited within a single effluent discharge point. This is summarised in Chapter 3 (section 3.10.5) where the method of assessing and quantifying the weights of sediment and hydrocarbons captured by the skimmers is also explained. The skimmers at Little Wigan Theatre were left insitu for the same time duration as those at Scott Lane; however at the end of the test it was found that a large proportion of the skimmers had been removed and only 15 were recovered. The same process of cleaning and weighing was used on the skimmers removed from the road gullies around Little Wigan Theatre.

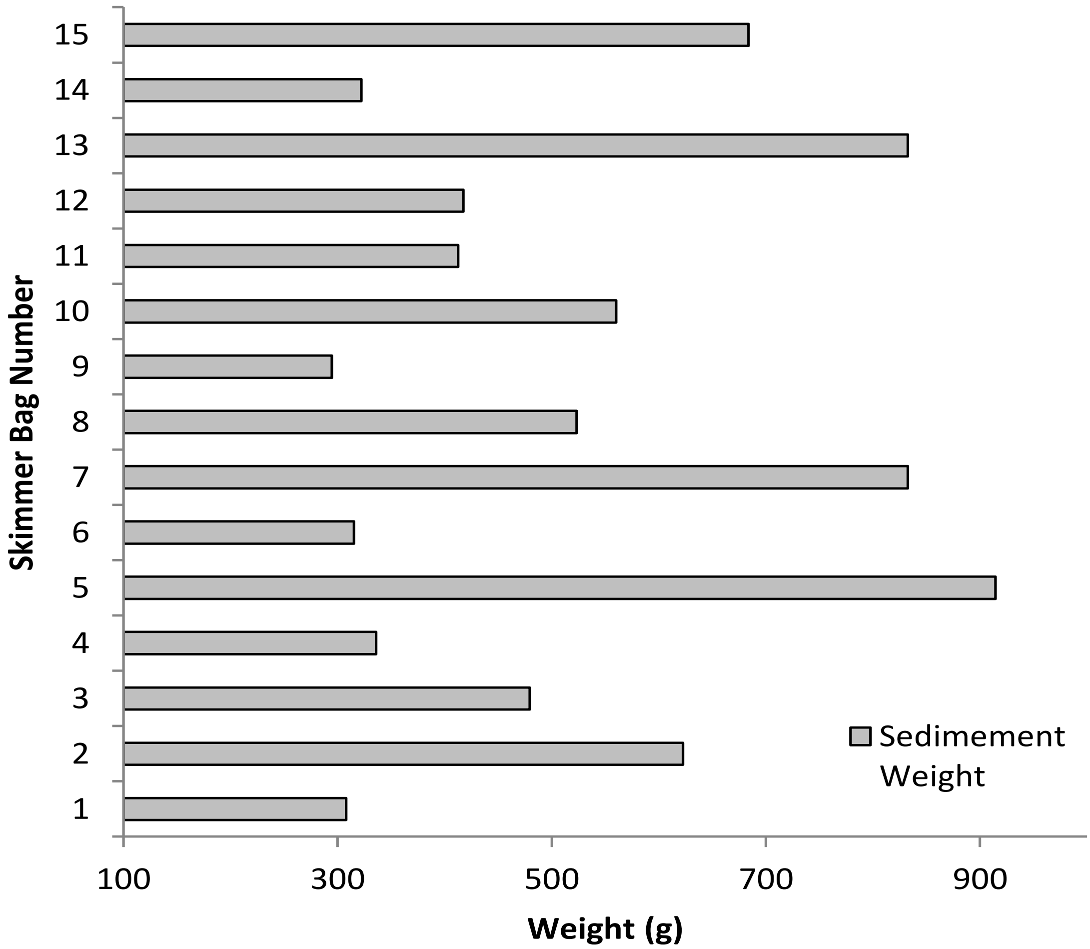


Figure 92 - Pollutant weight of skimmers removed from Little Wigan Theatre Site

The results of weighing are displayed in Figure 92. The average sediment load for the skimmers was greater than at Scott Lane at 523.6g, with a range between 294.5g and 914.4g. In terms of hydrocarbons however, the skimmers at Little Wigan Theatre did not perform as well as those at Scott Lane, with all of the bags recovered experiencing weight reduction below the average clean weight of 12 Smart Sponge pieces that were in all skimmer bags. Degradation of the Smart Sponge and other reasons already explored in the previous section are considered to be the primary reasons for the weight loss of skimmers at Little Wigan Theatre.

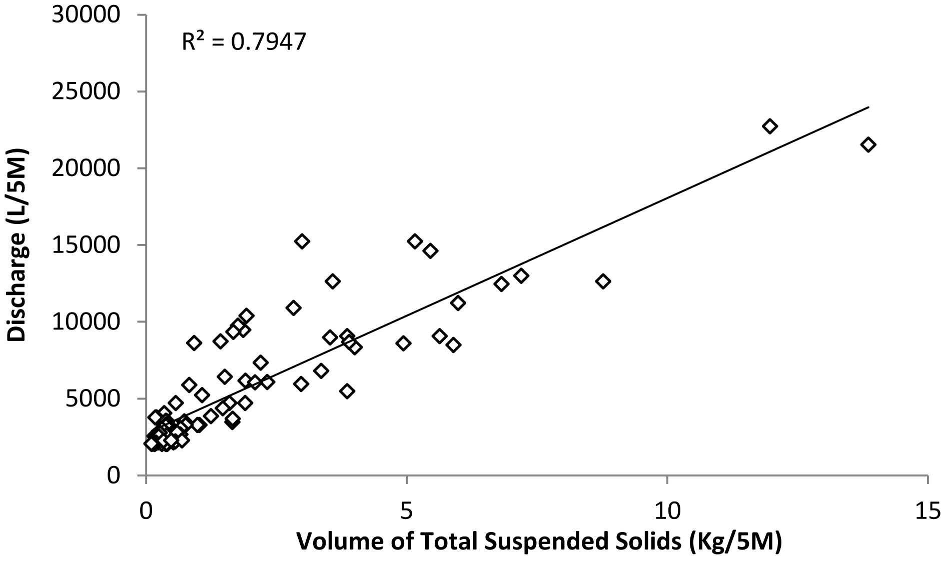
4.10 Characterisation of Stormwater Pollutants

In the river water sampling conducted in the earlier part of this Chapter, general trends between different WQV were summarised. The inaccuracies in this assessment due to discharge data only being available from the single EA gauging station in central Wigan were highlighted. In Part 2, unlike the river sampling, accurate discharge data is available for the two sites where product monitoring was completed. This enables a more substantial comparison to be made between the fluctuations in the highlighted WQV and discharge, as well as observations of the relationship between different variables. For a selection of the WQV captured by testing, this section observes relationships between them. It is divided into two sub-sections, one covering observations at the DD site and the other the X4.

4.10.1 Cherry Gardens Sample Site

Following the analysis of the DD in section 4.7, the analysis here also concentrates primarily on TSS and Aluminium, but in addition looks at the relationship between different metals and TSS to investigate if some correlate better than others. All monitored events are divided into separate 5 minute intervals and for each of these intervals the volume of discharge and corresponding volume of TSS and Aluminium summated. Figure 93 displays these results for TSS and Figure 94 for Aluminium. From both of these it is apparent there is a very good relationship between increased discharge and the increase in volume of both TSS and Aluminium, which confirms a strong correlation between washing of urban surfaces and significant TSS and heavy metal pollution

Figures 95 and 96 have been included to observe the correlation between different metals and TSS. They show Aluminium and Cadmium sample results plotted against the corresponding TSS value. As indicated in the literature review (section 2.3.4.2) and observed at multiple points in the river water sampling results presented earlier in this Chapter, TSS correlates well with heavy metals. Figure 95 provides clear evidence of this with Aluminium values plotted against TSS showing a strong, almost linear relationship. Figure 96 displays the result of plotting Cadmium against TSS which shows a weaker correlation than that with Aluminium indicating that not all metals are as strongly correlated with TSS. A potential explanation for the stronger correlation specifically between Aluminium and TSS could be that Aluminium has a larger number of free electrons in its outer shell giving it a greater electron affinity. This in turn allows it to ionise more readily therefore giving it a greater chance of forming weak electrostatic bonds with the molecules within suspended sediment.



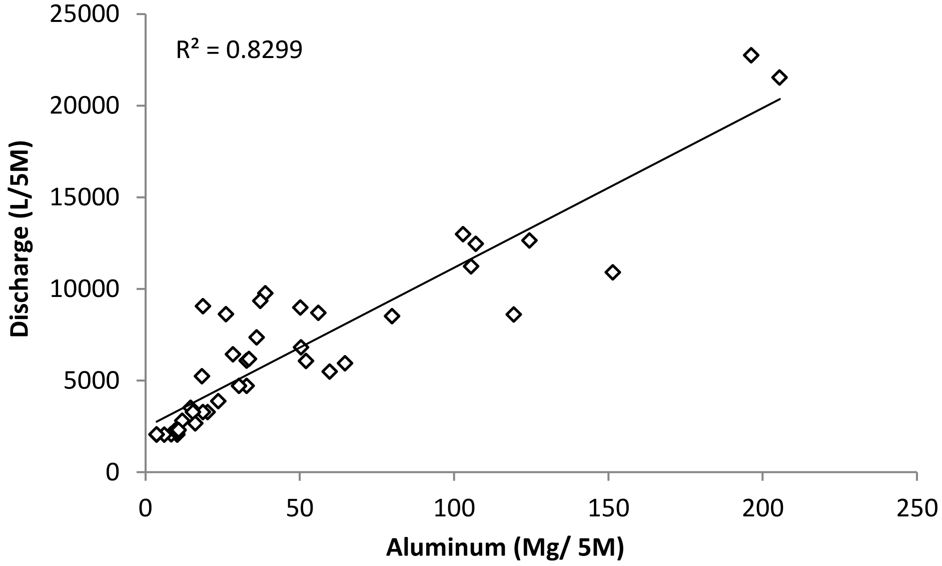
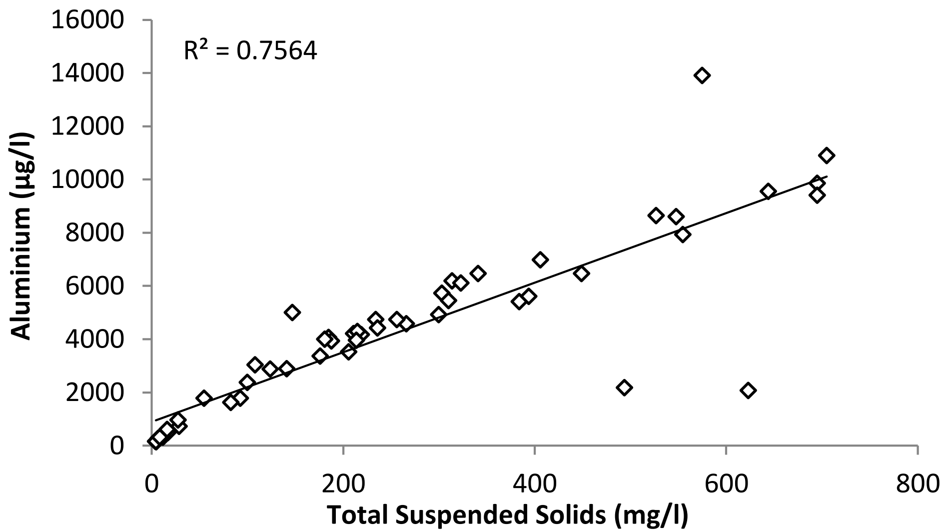


Figure 93 - 5 minute totals of discharge against TSS (DD Site)

Figure 94 - 5 minute totals of discharge against Aluminium (DD Site)



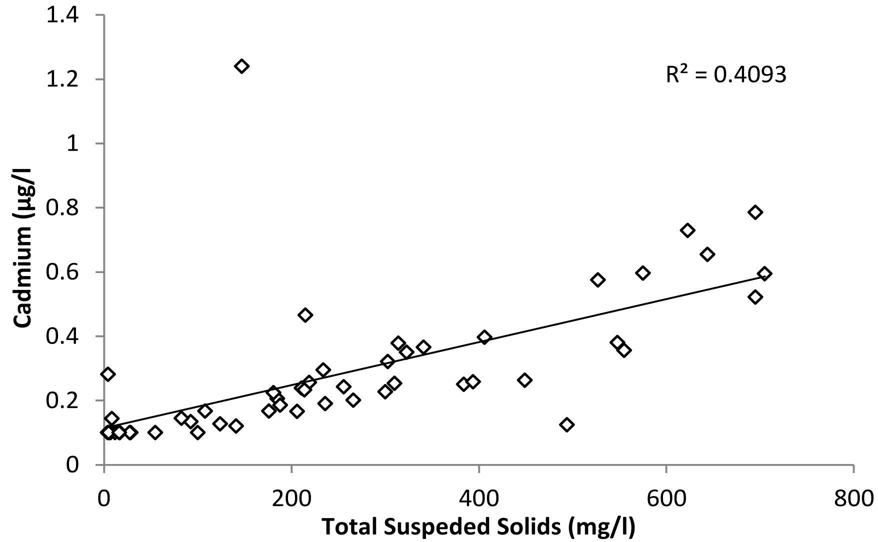


Figure 95 - Aluminium against TSS (DD Site)

Figure 96 - Cadmium against TSS (DD Site)

4.10.2 Coppull Lane Sampling Site

Using the same approach for the figures presented for the DD monitoring at Cherry Gardens, from analysis of data from Coppull Lane a series of observations about the relationships between the main WQV can be made. Figures 97, 98 and 99 show the individual 5 minute intervals into which all monitored events have been divided with the total discharge for each interval plotted against the total volume of pollutants discharged.

Figure 97 displays the results for TSS, Figure 98 for Aluminium and Figure 99 for Aluminium Dissolved. Due to the smaller number of samples collected at the X4 site the distribution of values presented is poorer than at the DD site, with a large proportion of the values for TSS and Aluminium clustered in the lower portion of the graph. However, from each of these figures it is still apparent there is a good relationship between increased discharge and the increase in volume of both TSS and Aluminium, with Aluminium Dissolved showing a better distribution and a stronger correlation. This further indicates the significance of the contribution of surface water wash off from urban areas, such as highway and residential uses. Aluminium is also evident indicating the importance of surface water discharges in the conveyance of such pollution into water courses.

Figure 100 shows Aluminium plotted against TSS, and this further corroborates the observations made in Figure 95 for Cherry Gardens, i.e., that metals show a strong correlation with TSS, and again display a strong correlation between the variables. However, conversely, in Figure 101 with Cadmium plotted against TSS, a much stronger correlation is evident (similar to all other metals) between these two WQV than seen in samples taken at Cherry Gardens. Figure 102 plots dissolved Aluminium concentrations against TSS, and from this graph it can be seen there is little correlation between these two WQV. Figure 103 shows Aluminium Dissolved plotted against Total Aluminium and again it can be seen there is no clear correlation.

These observations clearly demonstrate the link between TSS and total heavy metal concentrations highlighting the importance of removal of TSS from storm water in order to protect water courses from the effects of metal pollution. This strong relationship between TSS and metals further reinforces the relationship observed in the PCA analysis conducted on the river sampling results, indicated by the close clustering of these WQV on the component plot (Figure 52, section 4.3).

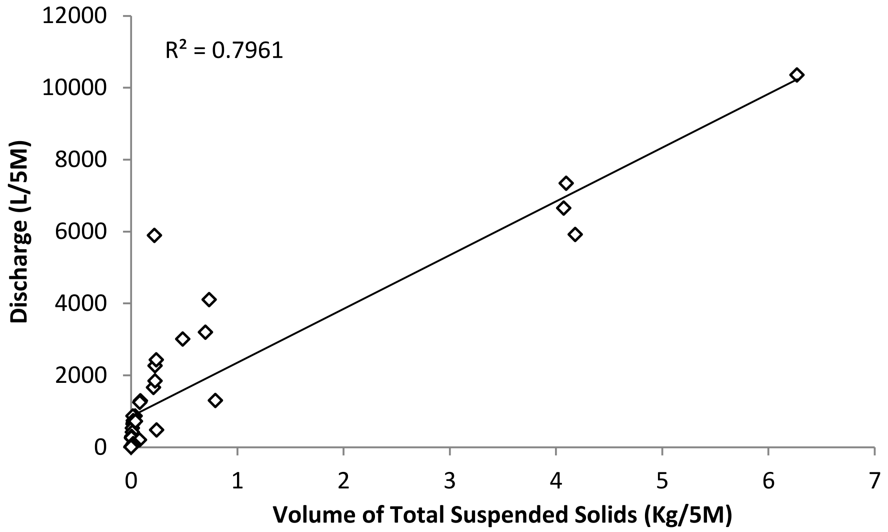


Figure 97 - 5 minute totals of discharge against TSS (X4 Site)

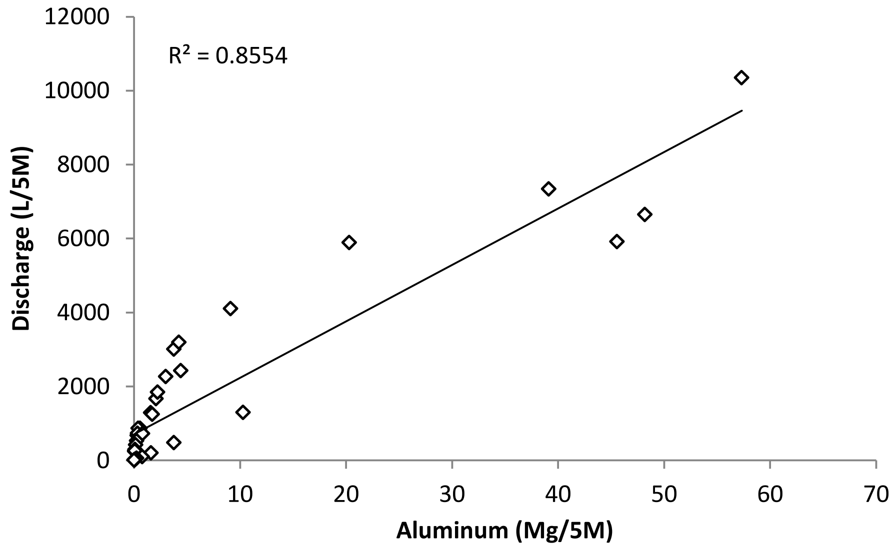


Figure 98 - 5 minute totals of discharge against Aluminium (X4 Site)

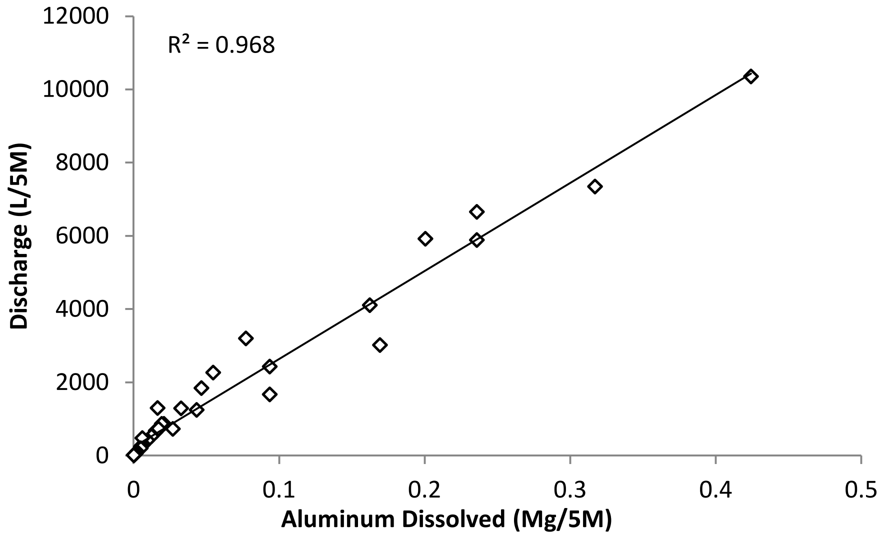


Figure 99 - 5 minuet totals of discharge against Aluminium Dissolved (X4 Site)

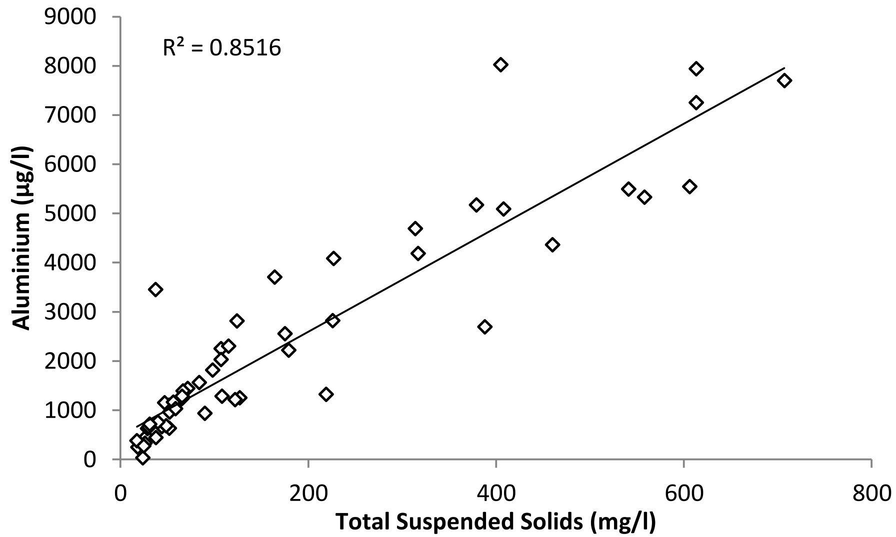


Figure 100 - Aluminium against TSS (X4 Site)

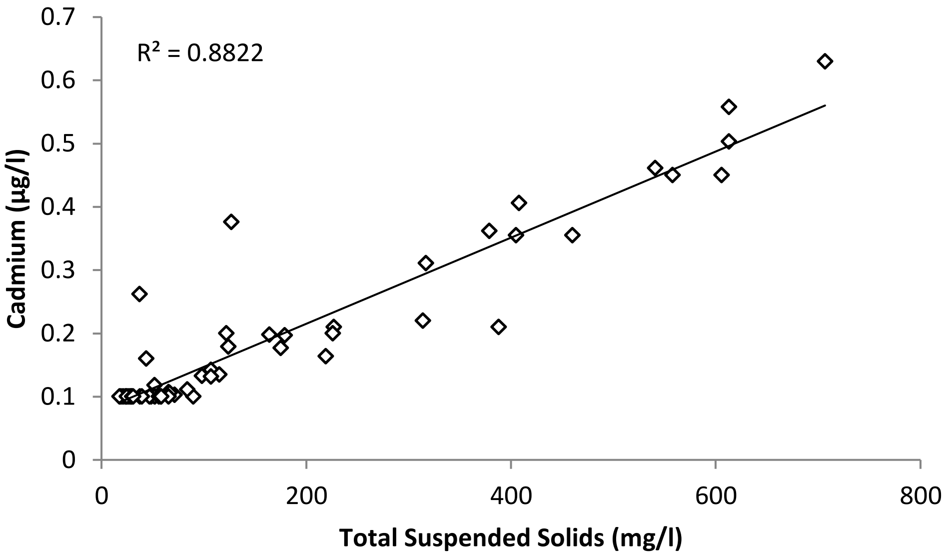


Figure 101 - Cadmium against TSS (X4 Site)

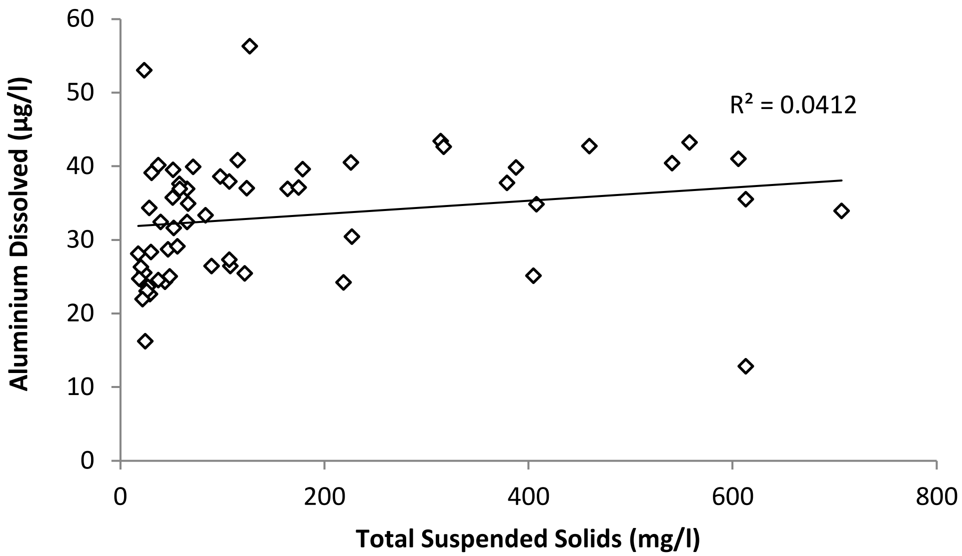


Figure 102 - Aluminium Dissolved against TSS (X4 Site)

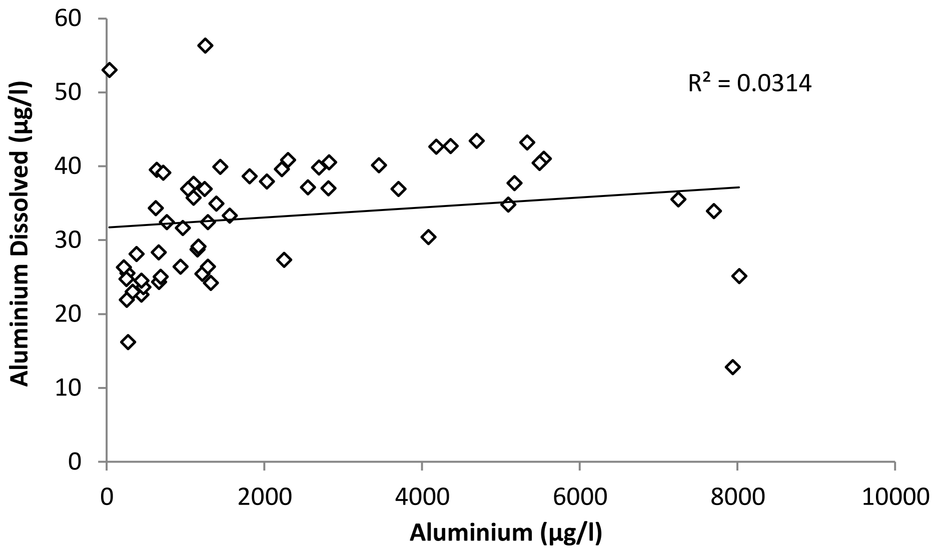


Figure 103 - Aluminium Dissolved against Aluminium (X4 Site)

4.10.3 Comparison with other Data

In order to provide some context to the concentrations seen in this study, Table 23 has been compiled using the range from several WQV reported in the literature covering stormwater monitoring from other European countries. It can be seen from Table 23 that TSS and heavy metal values fall within the range seen in the data presented from studies undertaken elsewhere. From these values it can be seen that there is significant fluctuation, not only between different countries but also at individual sites. This indicates that concentrations are very much site specific and cannot be typified to countries or regions.

Table 23 - Concentrations of WQV, and comparison with selected other studies

Study	**This Project (Wigan Study)**	**Boogaard and Lemmen (2007)**	**Fletcher and Deletic (2008) (Luxembourg Studies)**		**Daligault *et al.* (1999) (French Studies)**	
Location Figures	UK Data Mean (Range)	Dutch Data Mean (median 90 percentile)	St. Quirin Mean EMC (range)	Rte d'Esch Mean EMC (range)	Brunoy Mean (Range)	Vigneux Mean (Range)
TSS (mg/l)	188 (4-705)	49 (20-150)	592 (30-2500)	131 (30-300)	158 (11-458)	199 (25-964)
Al (µg/l)	4169 (145-13900)	-	-	-	3249 (65-10230)	3987 (89-10943)
Pb (µg/l)	51 (2-158)	33 (12-75)	80 (20-130)	50 (20-90)	52(2-210)	69 (4-404)
Zn (µg/l)	310 (14-920)	194 (95-450)	3330 (80-11700)	1170 (500-4100)	607 (210-2900)	146 (30-640)
Cu (µg/l)	72 (2-210)	26 (10-47)	170 (40-500)	70 (30-200)	23 (7-59)	24 (6-52)

Figure 104 displays the average metal concentrations from the two product sampling sites in this project, shown in proportion to each other. Also presented is the same data from the two river sample sites closest to the SWD, i.e., from Cherry Gardens and Coppull Lane discharges. The Figure has been adjusted to start at 40% to facilitate observation of metals observed at lower concentrations.

From this graph two main observations which can be made. Firstly, there is a clear difference between the average proportions seen between total and dissolved metals. All metals were higher in total as opposed to dissolved concentrations but Aluminium is the most significantly different being found in much higher concentrations proportionally than was seen in the river. Secondly, at all the sites the metals are present in almost the same proportions to each other; Aluminium is the most abundant, then Zinc then Copper and so on. The key difference to this is that Nickel was seen in higher average concentrations than Lead in the river samples taken from Coppull Lane and Great Acre but in the monitoring at Cherry Gardens and Coppull Lane Lead, was found to be in higher concentrations in its total form. The correlation between the metal ratios at each of the product sampling sites indicates that these results are relevant to other discharges to which average values are extended.

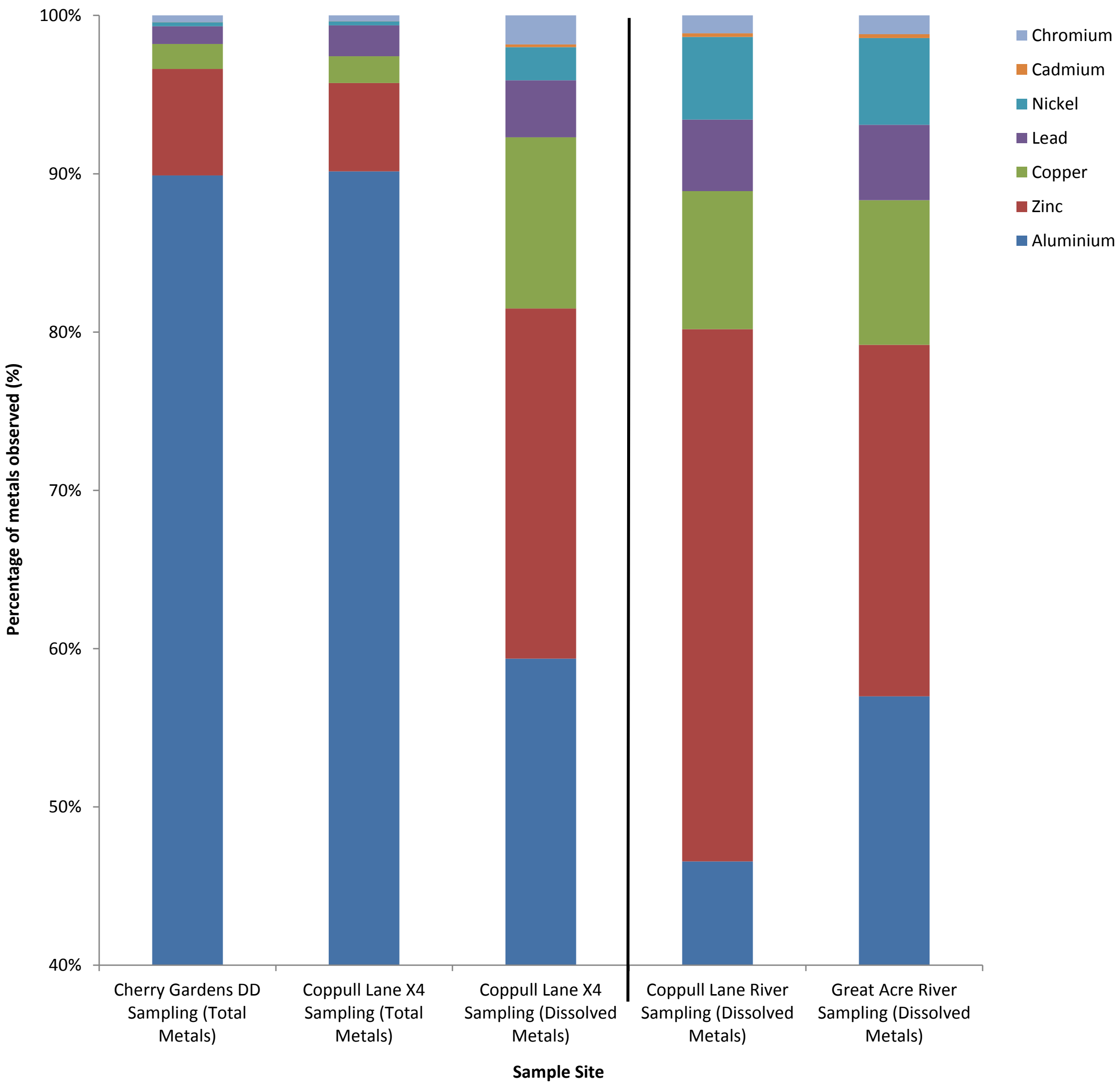


Figure 104 - Proportional Comparison of Metal Concentrations

4.11 Summary of Findings from Monitoring of the PTS

This concludes the presentation of collated data from the monitoring of the storm water treatment products installed to the Wigan Study area. From the analysis of data collected in this study, the volumes of both TSS and Aluminium with respect the DD and X4 sites, and TSS and Hydrocarbons with respect to the Smart Sponge sites, removed by the installed products were determined. Overall it was found that each product had a tangible positive impact. However this varied considerably subject to the monitoring process as well as the conditions and limitations at different sites.

4.11.1 Downstream Defender

The monitoring undertaken at the Cherry Gardens site was the most comprehensive of the three sites. It was conducted using a robust and uniform method and the use of automated sampling has reduced the potential for bias and variation present in manual sampling. Standard EA guidelines for the transport and decantation of samples were used, further reducing the potential for these elements to have any prejudicial effects on the final results.

Monitoring of 10 separate storm events was completed, each of which is covered by an average of 29 individual water samples, with a total of 288 samples collected. As with river sampling, this produced a significant data set giving results a high level of reliability. Monitoring accounted for 7.3 hours of comparable storm events, which collectively discharged 170kg of TSS and 2061 mg of Aluminium, of this 33 kg of TSS and 550 mg of Aluminium was removed from effluent. The collated results for the DD indicate that removal efficiency by the unit is not as high as might have been predicted, and that the efficiency is very much dependant on the volume of discharge, and the TSS particle sizes of samples. There are also a number of other factors which may explain why monitoring did not observe the DD functioning more efficiently. These include the large variation in the intensity of storms, difficulties associated with the monitoring, and the presence of a constant base flow.

4.11.2 Storm X4 Heavy Traffic

The same method of automated sample collection, sample transport and decantation was duplicated at the X4 site. Although the number of events recorded during the X4 monitoring was far fewer than was observed at the DD, on average the number of samples per event was actually higher (at 31), with a total of 94. While the overall significance of these results is lower than at the DD site, the samples per event and overall number of samples is still large, indicating the high reliability of the events recorded.

The three events that were monitored accounted for 2.8 hours of discharge, during this time 23.2kg of TSS and 262.8 mg of Aluminium were discharged. Of this 6 kg of TSS and 60.6 mg of Aluminium was removed from the effluent. As the number of results obtained from the site was considerably less than that obtained from the DD site, the resultant weights are reduced. Ideally more events would have been captured (as was originally stated in the methodology section 3.7), at least the same number as monitored at the DD site. As can be observed from the Figures 85-87 and Tables 20-22, the SWD at Coppull Lane is discharging significant amounts of TSS and heavy metal pollutants on a single event basis. While the average efficiency of removal was higher than the DD, the two events recorded were considerably less intense in terms of discharge. The final event monitored was the most significant with 20kg of TSS discharged over a 65 minute storm, during this event removal efficiency was lower at just over 21%.

4.11.3 Smart Sponge

A direct comparison of the Smart Sponge with the previous two sites is not possible because it was not targeted at the same pollutants as the DD and X4 and the method of monitoring was different. After being left in situ for just over a 1 year period, the 55 passive skimmer bags recovered (each containing 12 Smart Sponge pieces), collectively weighed 54.7 kg. Following cleaning and further weighing the collective weight of sediment captured by the bags was 19.2 kg, with a further 624 grams of liquid hydrocarbon being captured by the 40 skimmers removed from Scott Lane.

It is also difficult to put observed results into perspective as there are no known published studies in the UK where the effect of Smart Sponge has been quantified. A study from the US, where a similar product tested by the manufacturer to treat runoff from an airport is available, found considerably larger volumes of hydrocarbon absorbed by a different smart sponge product, the Ultra Urban Filter (unpublished grey promotional literature). These are much larger products, weighing approximately 10kg each. The US study claims to show that 16 filters captured 387.6 kg of sediment and 238.7 kg oil over a two year period. Clearly this is a considerable difference to the retention weights seen in sponges recovered by this study.

There has been a benefit to the water quality of the Little Wigan Theatre and Scott Lane sites, where the Smart Sponge was installed, however quantifying this benefit accurately has been difficult. A combination of the degradation of Smart Sponge within skimmers, blinding of skimmers with sediment, high turbulence in the effluent at Scott Lane and the likely low volume of free phase hydrocarbons occurring at sites, are the primary reasons for the lack of greater weight increases with respect to hydrocarbon in the Smart Sponge skimmers.

4.11.4 Estimation of Pollutant Load from Catchment Area

As identified in sections 4.7 and 4.8 there were a large number of events which occurred during monitoring which were not recorded. To firstly estimate the contribution of these events in terms of pollutant discharge from each site, and secondly to make an overall estimation of the volume of TSS and Aluminium discharged from the whole study catchment area, a series of calculations has been made. During the whole monitoring period at Cherry Gardens (October 2013-March 2014), the log of discharge, indicates a further 33 events could have been observed. Using average pollutant loads and removal figures, it is estimated these events would have resulted in a further discharge of 562.8 kg of TSS and 13608 mg of Aluminium and of this approximately 118.7 kg of TSS and 3592.5 mg of Aluminium was potentially removed by the system. Similarly at Coppull Lane, the monitoring period (April - June 2014), saw a further 29 events. Again using the average pollutant loads and removal figures for the site, it is estimated that these events would have resulted in a further discharge of 224.5 kg of TSS, 2541.1 mg of Aluminium, and 22.9 dissolved Aluminium; of this approximately 89.1 kg of TSS and 757.2mg of Aluminium would have been removed if the system continued to operate with the same level of efficacy throughout.

The Flood Estimation Handbook is a piece of software produced by Wallingford Hydro Solutions Ltd (1999) that has been used to give an accurate figure of the size of the catchment covered by the sampling conducted in both parts of this study. This software allows the calculation of the drained area of any individual location within all UK river catchments. This allowed the immediate area around Wigan, draining into the study area, to be determined by subtracting the catchment area given by the FEH for the furthest upstream sample point, from the lowest one. This indicates the study area catchment is 140.13 km^2; the UU asset register indicates that a further 17 SWD (including the 3 monitored) are located within this area.

These assets drain surface water for the vast majority of the study catchment area, and are considered to serve the bulk of Wigan's surface water needs (discounting combined sewers).

Considering that the land use of the two sites monitored in detail (Cherry Gardens and Coppull Lane) include a mixture of urban uses: residential, highway, light industrial and retail, etc., representative of the Wigan urban area, it can be assumed that the type and concentration of pollutants seen in other identified discharges should not be significantly different. While these surface water assets identified vary in size (meaning discharge from some would be greater and less from others) it is considered that average values from the 13 monitored events are representative of volumes of both TSS and Aluminium discharged from the 15 unmonitored SWD. Using this, an approximate figure can be calculated for the volume of these pollutants discharged from the study catchment area into the River Douglas produced by the 62 events of discharge observed over the duration of monitoring.

Using this information it is calculated that a the total of 13724 kg of TSS and 268.4 grams of Aluminium was discharged into the River Douglas by these surface water assets in Wigan during the 8 month monitoring period. Whilst simplistic, this approach allows an estimation to be made of the total volume of pollutants emitted in the study catchment area and is similar to the method employed by Wither *et al.* (2005).Overall, considering the average removal figures for both the DD and X4, wide scale retrofit of these PTS would not completely mitigate the pollution impact of storm flows on the river environment into which they discharge. While they do provide benefit to sites were SuDS retrofit is not feasible, to increase the level of treatment offered by these products they could be used in sequence or in train (as recommended by SuDS guidance section 2.4.1), or in conjunction with storage tanks to regulate discharge through systems which would increase their effectiveness.

This section of this study has provided a comprehensive analysis of the monitoring and associated sampling used to quantify the performance and effectiveness of a series of PTS in removal of TSS, Aluminium (the concentrations of which correlated well with other heavy metals) and hydrocarbons. It also provides a large amount of useful information into the practice and obstacles around stormwater monitoring. The final Chapter is the overall conclusion which summarises the main findings of this study, and makes a series of recommendations for further work which could provide greater insights and understanding in this field.

Chapter 5 – Conclusions

5 Chapter 5 – Conclusion

This final Chapter is framed by the key questions that have been consistently investigated through the previous chapters, with the primary aim of gaining a greater understanding of the contribution of diffuse pollution to rivers in urban areas, and how it can be mitigated. It begins by revisiting the problem of urban diffuse pollution (covered in detail in the literature review) before summarising the key findings which have been identified through two complimentary investigations; the first addressing the contribution of diffuse pollution to the water quality of the River Douglas; the second exploring the effectiveness of various means to mitigate diffuse pollution in the same waterway. This also includes a review of the methodology used in each case, before finally a series of recommendations are made on which further research and work needs to be focused.

5.1 What is the Urban Diffuse Pollution Problem?

In relation to environmental improvement significant progress has been made through the introduction of legislation covering the discharge of polluted and wastewater to rivers and other receiving waters. The principle of sustainable catchment management is now well established; however its complete implementation is still failing to be achieved across much of the UK. Integrated catchment management needs to link land management, and any changes to how the land is treated, to the quality of the associated water bodies. Increasing pressure on land use (from the rising population) mitigates against the maintenance of current standards, and makes further improvements to surface water quality even more difficult (McGonigle *et al.*, 2012).

There is still a lack of integration between the different groups responsible for implementing the mix of legislation affecting the protection of water quality of rivers (Macleod *et al.*, 2007). Many different groups and stakeholders involved in delivering improvements still fail to appreciate, that it is only through integrated management and better cooperation that real improvement can be made against multiple objectives, such as water quality improvement, flood protection, water demand and its control. Currently considerably more emphasis and funding is allocated for flood risk management, with comparatively little attention or priority given to the water quality risks of impermeable surface runoff (J. B. Ellis and Revitt, 2010).

The traditional method of using environmental quality standards for pollution control is not appropriate for addressing urban diffuse sources given:

- their highly variable and unpredictable nature,
- that sources are not easily traced,
- toxic components are not defined, and
- there are no EQS values for the build-up of contaminants within sediments.

Existing legislation still fails to account for the episodic nature of the problems surrounding urban diffuse pollution sources, and the fact that current large scale sampling regimes will often fail to identify the variable nature of the degradation in water quality from these sources (J. B. Ellis *et al.*, 2002),. This was also observed in Chapter 4 (section 4.6).

In the context of whole catchment management UDP contributes significant amounts of phosphates through faecal contamination of wastewater infrastructure (Crowther *et al.*, 2002; Kay *et al.*, 2008; Rothwell *et al.*, 2010a; Zhang *et al.*, 2014) and heavy metals from urban wash-off (Goonetilleke *et al.*, 2005; Miguntanna *et al.*, 2013). Clearly, tackling pollution from urban areas is critical to conform with legislation such as the WFD, RBWD, UWWTD, and there is significant evidence to suggest that the targets and requirements set in such legislation will not be achievable without the removal of the contribution made by urban water pollution (Donohue *et al.*, 2006; Haycock and Muscutt, 1995).

However, tackling this problem needs the application of a diverse and wide range of measures. Based on the results presented in Chapter 4 (sections 4.7, 4.8 and 4.9) it is apparent that currently available systems sold as end of pipe solutions cannot alone deliver satisfactory pollutant removal. These systems need to be delivered as part of a catchment wide holistic response, where many different activities are co-ordinated to deliver a wide range of benefits, as outlined in section 2.4.4/2.5. There are a number of key steps that need to be taken in conjunction with each other to deliver this change. Firstly, guidance and support (that has seen effective change in the agricultural and forestry sectors), needs to be adopted in urban areas, with similar statutory and/or financial mechanisms put in place to encourage best management practice (B. D'Arcy and Frost, 2001).

Secondly much better source control needs to be implemented across catchment areas. In the UK this is starting to happen with introduction of mandatory SuDS provision through the planning system (DCLG, 2014), but many developers still take a limited and piecemeal approach. Also this doesn't consider the primary cause of urban runoff, which is that generated by existing development. There is currently no regulation or incentive mechanism to compel/encourage property owners to manage their surface water more sustainably, there is also still an automatic right of connection for new properties with respect to discharging surface water to existing sewers. Considerably greater effort to improve source control at a local level needs to be made, for example through water retention by green roofs, rainwater harvesting systems and increased infiltration.

This needs to be paired with much better site control of pollutants, especially in urban areas, so for example premises with large car parks should regularly sweep or vacuum to remove pollutants before they can be washed into drainage systems and water courses. All of this activity removes pollutants and volume before they have chance to be transported and conglomerated into a problem. This would allow increased benefit from existing retention features and allow future ones to be smaller as well improving overall flood risk. Lower volumes and flow rates would allow PTS and SuDS to deliver much better separation and removal of pollutants

Removal of large volumes of surface water runoff would improve existing sewer capacity reducing the risk of sewer flooding and damaging discharges to waterbodies, existing practices of sewer expansion which are extremely costly and disruptive would be unnecessary. This also offers potential for reduced energy consumption as volumes to WWTWs would reduce and the need for pumping of CSO storm tanks could be removed. All of these measures would be a very large step towards urban districts and cities becoming more sustainable and environmental friendly. There is a very large body of literature showing the potential to change existing urban areas for the better, and while these changes would require a significant financial input, they would in turn deliver an extensive range of benefits.

5.2 What are the Key Findings of This Work?

In the light of the ideas identified in the literature review, the project has investigated urban diffuse pollution in rivers and subsequently what actions can be undertaken to address the problem and reduce the contaminating effect of urban runoff on river systems. A summary highlighting the key findings made in the river water sampling (section 4.6) and in the sampling of the PTS systems (section 4.11), and to avoid repetition the conclusions presented here focus on the achievement of the original objectives, and identifies where possible improvements to the methodology could be made in future studies.

5.2.1 Quantifying the Problem of Urban Diffuse Pollution

One of the original objectives of the project (section 1.2), sought to fill gaps in knowledge identified in the literature review (section 2.5), which led onto the completion of a micro level sampling programme involving the collection of 134 samples at 25 separate locations along a 16km section of the River Douglas. The program identified the WQV that were most significantly increased during storm events (section 4.4), as well as highlighting the inadequacy of the EQS method to identify diffuse pollutants (section 4.6):-

- Ammonia (On Average 163% elevated in increased discharge)
- FIOs (On average 63% elevated in increased discharge)
- TSS (On Average 111% elevated in increased discharge)
- Aluminium (On Average 228% elevated in increased discharge)
- Zinc (On Average 56% elevated in increased discharge)

It must be emphasised, that the reasons for these observations is due to pollutants either being washed from impermeable surfaces, or as a result of flushing/inundation of both existing surface water and foul drainage before being discharged into rivers. So the concentrations of WQV seen are directly connected to these 'sources' becoming 'activated' by rainfall. Another important observation from the river sampling, was the downward trend in some WQV, most significantly Orthophosphate, with average concentrations falling by 50% across the study area (section 4.4.1).

These observations demonstrate that both the location (in terms of its proximity to existing drainage infrastructure discharges), and the preceding rainfall conditions, are the two most important factors affecting the quality of a water sample. Given that there seems to be no formal structure behind the selection of locations for WFD sampling points, with most based on historical sample locations, it presents the question of how representative WFD water quality classifications are of the 'actual' water quality of their corresponding 'waterbodies'?

Considering both the wide spacing geographically, and the irregular intervals of sample collection, it seems evident that much of the event based pollution observed (such as that in this micro level sample regime), may well not be accounted for in WFD classifications. In this study this can be clearly demonstrated when considering the significant variation in average concentrations seen between sample sites. In these cases, a WFD classification based on one site would yield a different result to one just several kilometres further up or down stream.

It was intended to use river sampling to identify pollutant 'sources', however, while the river sampling is indicative of poor quality within a certain area, it has not been possible to accurately link collected sample data to specific 'source' locations, i.e. SWD, highway runoff, etc. This is primarily due to the highly intermittent nature of different pollutant sources, but also the inaccessibility of much of the river bank, as this made locating associated polluting discharges difficult. Although sources are diffuse in nature, existing drainage infrastructure is one of the most important conduits for the transport of surface wash off pollution into watercourses (strongly demonstrated through the monitoring of the PTS in the latter part of the study), and other surface waterbodies. As the cumulative effect of different land uses within a drained area is represented in the mixed effluent entering rivers, distinguishing which individual parcels of land contribute different pollutants and their volume is very difficult without monitoring and testing of the quality of runoff at an impractical multitude of locations across urban areas.

5.2.2 Evaluation of Methodology

While these are important observations to make from the collected data there are a series of improvements that could have be made to the sampling programme, which could have allowed the generation of more representative and indicative data. One of the key issues with the delivery of the sample regime was the potential for variation in conditions while multiple samples were collected over a12 to 24 hour duration. This increased the complexity of relating samples from the various locations, collected at different times and conditions, to each other.

For future studies it is recommended that samples should be collected at slightly less frequent intervals, but crucially, at the same time, either using automated sample collection or a larger number of researchers to take samples manually. This would prevent the element of time from adding to variation, allowing the change in water conditions at different sites to be observed more accurately. Being limited by the number of samples a single researcher could collect throughout a day, it is likely that increases in rainfall/discharge could easily alter water conditions between sites while the researcher travelled between them.

Furthermore, this network of samples would form part of a continuous monitoring system with each sampler location also recording specific river discharge and rainfall. As there was only a single point with respect to discharge and three with respect to rainfall within the study area where data was collected, the further a site was from this location, the less indicative this data would be of true conditions. This sampler network would need to be linked with telemetry equipment so that sample collection by separate sites was recorded. This would allow future studies to more accurately observe conditions at different sites at the time samples were taken. However such recommended improvements would require considerable resources with respect to funding and researcher time.

These recommendations would help to remove potential elements of variation from the analysis, as it is better to have data specific to a site rather than relating samples to data from a gauging station several kilometres away. In summary it is recommended, that for future studies a larger quantity of data should be collected at a smaller number of sites on an event basis, ideally the greater the number of sites where data is collected the better. However this needs to be balanced against the resources available.

Overall the objectives set for the river water sampling , i.e., to identify and quantify the most significant pollutants has been achieved, with the exception of using the data to identify specific pollution sources, where other means, such as physical inspection and scrutiny of local drainage asset registers are considered to be more appropriate.

5.2.3 Mitigating the Problem of Urban Diffuse Pollution

In the second phase of this project the focus has been on providing remediation and subsequent monitoring to demonstrate the effectiveness of different measures. Again the objectives were largely achieved, with a series of suitable locations being identified and a series of PTS being installed and monitored. As covered in Chapter 3 (section 3.6), the river monitoring data was of limited use in relation to identifying pollutant sources. Ultimately the use of the United Utilities asset register, locations where installation was feasible, and other practical considerations were more influential in site selection. In most cases these criteria precluded the possibility of locating treatment where some of the highest sample concentrations were observed in river monitoring, as gaining permission to complete works was not possible due to the potential for them to be disruptive and prohibitively expensive. So, although a series of PTS were ultimately retrofitted, the locations were not where the potential benefits would have been greatest.

It should be observed that the storm monitoring process was extremely time consuming and labour intensive, and the success of monitoring varied between sites. Certainly the most extensive results were collected from the DD at Cherry Gardens. However a greater portion of time was allocated to this site, which included fine tuning the performance of sampling equipment. Monitoring recorded 10 storm events, and the following analysis demonstrated removal of TSS (21% average) and its associated sediment bound heavy metals (26% average) in all events. Being a similar product to the DD, the monitoring of the X4 was identical, with the exception of the differences covered in section 3.10.2. As with the DD, the X4 was successfully monitored, with analysis again demonstrating the successful removal of both TSS (39% average) and associated heavy metals (29% average), although the number of events recorded, at three, was considerably fewer than those captured at Cherry Gardens

It was identified that the significance of the first flush effect is contested (section 2.3.4), as stated by Goonetilleke *et al.* (2005) an increase in pollutant concentrations in the early stages of a storm event alone cannot be considered to be adequate evidence of this phenomenon. The observations from this study (at Cherry Gardens, section 3.10.1 and Coppull Lane, section 3.10.2) were that a much larger volume of pollutants were observed to be discharged during the comparatively short period of time before and including peak discharge, indicating that not only concentrations but also the volume of discharge was higher in the earlier periods of events. It is considered then that there is a basis to say that the 'first flush' was observed for this study. However based on the definitions used in some studies this would not be definitively concluded.

The process of monitoring for the Smart Sponge was different to that for the DD and X4, with Smart Sponge skimmers left insitu for approximately a 1 year period. Following this they were removed, cleaned and weighed to quantify the TSS and free phase hydrocarbons they had intercepted. Again it was possible to demonstrate some benefit in the removal of both pollutants, however this was limited, and the volumes removed were lower than anticipated.

Another issue highlighted by the project is that of adoption and future maintenance of PTS, however this is not a problem unique to such systems. Currently local authorities are requiring developers to provide evidence of a management company who will take over the long term management of new SuDS systems. Alternatively the adoption of existing retrofitted systems can be completed using existing legislation such as Section 106 of the Town and Country Planning Act 1990 or Section 38 of the Highways Act 1980 (CIRIA. *et al.*, 2007). However with respect of retrofit schemes such as the one demonstrated within this thesis the primary barrier for their uptake is that there is no incentive or regulation to force retrofit of PTS or SuDS to existing development in the first place, let alone make provision for the on-going maintenance of such measures.

Another key problem is that enforcement with respect to diffuse pollution to water courses is not practical. As has been identified in this work the water quality monitoring in the UK does not accurately quantify many of negative impacts that urban diffuse pollution as well as other intermittent discharges make to watercourses. Secondly the effects of urban diffuse pollution may also be long term so by the time damage is evident it is too late to mitigate polluting discharges. It is therefore almost impossible to demonstrate the pollution caused by a specific outfall or discharge without further intensive monitoring and sampling which on an individual case basis is not feasible.

To provide real incentives to prevent such pollution, it is recommended that the cost of sewerage of surface water should be increased to give encouragement for action to be taken on an individual property basis with respect to source control, which would also encourage greater take up of SuDS and give owners a financial incentive to undertake the required maintenance (in that they would want to avoid discharge to sewerage). This would need to be linked to stronger powers for the EA to enforce on discharge from surface water outfalls and CSOs incentivising water companies to prevent such occurrences. This would make installation and on-going maintenance of SuDS and PTS more feasible.

The summary provided in Chapter 4 (section 4.11) provides greater detail of the success of the monitoring completed for each PTS. The objective of monitoring the efficacy of retrofitted PTS under local conditions was achieved, albeit with greater success at some sites than others, but overall the results obtained from each of the three sites demonstrate benefit in terms of water quality improvement.

5.2.4 Evaluation of Methodology

In relation to the DD, one of the most important recommendations would be the use of second flow meter to improve the reliability of results. This would have allowed a more accurate comparison of the pollutant load up and downstream. Secondly this monitoring was difficult for a single handed researcher to conduct. It is therefore recommended that future studies should employ multiple researchers working in collaboration to complete monitoring which should improve the quality of results collected. Whilst it was unavoidable, the base flow through the system at the DD site greatly complicated the analysis.

The primary improvement that could have been made to the X4 monitoring would have been to capture a greater number of events, the failure of the communication cable between samplers was difficult to diagnose and caused several events to be missed, and the surcharging of the upstream manhole greatly complicated and inhibited sampling. Ensuring greater time for monitoring would be strong recommendation for future similar studies.

Although the guidance of the manufacturers was followed to allow quantification of the SS, a series of draw backs were found with the method to determine the volume of free phase hydrocarbons absorbed by the passive skimmers installed to both Little Wigan Theatre and Scott Lane. This was mainly associated with the difficulty of observing the increase in weight for hydrocarbons only. The method used, as described in Chapter 3 (section 3.10.4/5), was subject to considerable variability. The American manufacturer, ‘Abtec’ has suggested that, rather than weighing, the Smart Sponge should be dissolved in a solvent and the volume increase observed, to more accurately quantify the hydrocarbon absorbed. This however would not remove the need for cleaning of TSS material from sponges.

It was observed that published studies often report results in different ways, in terms of benefits achieved from PTS and SuDS installations. In this study, although there were some unavoidable differences between installation sites, a partial comparison was successfully made between two HDVS and results were presented in a comparable way. It is suggested that future studies conduct monitoring in a similar fashion, to allow cross comparison to be more easily made. Complete standardisation of the monitoring process would not be possible, primarily due to the huge variation of conditions between sites, storm events and the differing function of different PTS and SuDS.

Overall, the approach to monitoring used for this part of the project has provided a significant and novel contribution to the knowledge around stormwater monitoring, certainly in a UK context. Through overcoming the difficulties associated with the retrofit application of PTS to provide water quality benefits which is not common in the UK, where there is more emphasis on SuDS. Although SuDS may offer increased levels of treatment and associated benefits, their primary limitation in urban areas is the difficulties of retrofitting and need for space. The study has demonstrated the effectiveness of PTS to provide some mitigation of pollution of untreated SWD in an existing urban area and shows that PTS are a viable option. However care should be taken in the selection of PTS which depends on the event pollution loads observed in discharges, as well as the practical limitations of sites (as observed in Chapter 4)

5.3 Recommendations for Further Work

This project has contributed to knowledge by providing new insight into a series of important areas, including

- River Sampling Methodology,
- Retrofit of storm water treatment in an urban setting,
- Storm Water Monitoring,
- The water quality performance of PTS

Specifically, the main contributions can be summarised as:

1. Quantification of the contribution of diffuse pollutants to the pollution load in the River Douglas in Wigan;
2. New insights into, and refinement around, sampling techniques and methods that have wider application in river water quality assessment;
3. Identification of a range of practical problems in the installation, monitoring and comparative evaluation of the effectiveness of PTS.

This project has highlighted a series of important areas, in relation to these fields of research, where further work is needed:

- Conduct sampling programmes in other urban areas to confirm the results observed in this study, for example metals were generally observed in proportionate concentrations. Is this the case in other areas?
- Research into how different WQV diminish with distance from their point of entry into a waterbody. For example can a reliable relationship be defined for percentage reduction in concentration over a fixed distance?
- The need for more data on the importance of particle size in relation to both the actual mass that different fractions of particle sizes contribute as well as the percentage of the volume of sediment bound pollutants that are attached to each different particle size fraction.
- Undertake further field trials of both PTS and SuDS to further demonstrate the multiple benefits they can provide. The results of such trials could then support greater dissemination of knowledge about these benefits.
- As was concluded in Chapter 4 (section 4.10) it is considered that the 'first flush' phenomenon was observed in results. There is significant further work needed to give a more robust definition of the 'first flush' effect. This needs to pay particular attention to the importance of particle size, for example does particle size fluctuate across storm events? What it the contribution of different fractions of particle size to overall pollutant load?
- The capital expenditure required for the purchase and retrofit of the HDVS in this work was considerable, there is also an on-going annual operational expenditure to consider when utilising such systems. No analysis has been completed of the cost/benefit ratio of installed PTS in this study. An interesting piece of further work could undertake a detailed comparison of the various capital and operational costs of PTS systems against the on-going benefits they provide. Such work could also include a comparative analysis of the benefits against SuDS; however such work would need to develop a method of quantifying the specific benefits offered by both systems, such as the amenity value of SuDS and the space saving potential for PTS.

- It was proposed to remove and analyse the material captured in the sump of the DD and the X4 units however this activity was not completed in time for result to be included in this work. No method to monitor the material in the sump sections of HDVS was utilised meaning that maintenance may well be conducted when it is not required. Future work could investigate if there is a method to monitor sump contents as both a tool to assess performance and to inform when cleaning is required. Potentially monitoring with a remote camera or assessing the volume of water displace from the chamber by settled sediment could be used.

In this way policy needs to be strengthened to tackle both water quality and flood risk concerns posed in existing urban areas, and to encourage proper catchment management. There is a strong need to increase the viability of retrofitting of both PTS and SuDS to urban areas. It is recommended that policy studies focus on how the current system of design and implementation works and to suggest a series of changes that should be made to existing processes to dismantle barriers and facilitate greater uptake of such systems. This needs to duplicate the policies from the agricultural and forestry sectors adapted to address diffuse pollution in an urban context.

Overall this study has added considerably to the understanding of river water pollution dynamics and the potential of PTS to reduce diffuse pollution. This knowledge is important to environmental officers, engineers and all other professions responsible for the application of the river water directives, maintenance of sewage storm water systems and the application and wider uptake of catchment management.

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Appendix Contents

Appendix I – Review of Agriculture and Forestry

1 Agriculture

In much of the UK, agriculture contributes considerably to river nutrient pollution, primarily though runoff of nutrients from fertiliser and livestock (Heathwaite *et al.*, 1996; Parker *et al.*, 1991; P. J. A. Withers and Lord, 2002). This can be attributed to a number of factors; such as the rapid intensification of agriculture over the last 50 years and because agriculture is by far the largest land use in the UK. Sources differ on the exact percentage of land devoted to agriculture, but most agree it lies between 70-77 % (Angus *et al.*, 2009; Barker, 1995; DEFRA, 2006; Khan *et al.*, 2012). This means that over the last 70 years much, of the UK landscape has been shaped by agricultural management and policy.

Nationally the land area utilised for agriculture is approximately 17.5 million ha, of this 28% is used for crops and 67% is used for grassland. Grassland areas include 4.4 million ha of sole ownership rough grazing and 1.1 million ha of common land mainly located in the upland 'disadvantaged areas', which are primarily used for grazing for sheep and cattle and the type of agriculture particularly prevalent in northern and western parts of the UK (Angus *et al.*, 2009). DEFRA (2006) identified in its annual review of the 'Agricultural Change and Environment Observatory programme', that agriculture is responsible for about 26% of phosphates and 60% of nitrates as well as up to 75% of soil sediments entering UK Rivers. Clearly, any coherent catchment management plan needs to address water pollution inputs from agricultural sources. It is also evident that any future changes or growth in agricultural land use will have a subsequent effect on water quality of surface waters.

It is important to understand that not all farming activities cause equal levels of pollution. Intensive agriculture is known to produce significant pollution outputs, particularly nutrients, FIO's and sediments (Monaghan *et al.*, 2007; Scholefield *et al.*, 1993). Water pollution from agriculture is primarily due to nutrients and sediments being transported into rivers, typically by excess surface runoff as it passes over and/or under farmland. In upland and undulating areas water runoff causes erosion by detaching soil particles and this presents an important pathway for the conveyance of pollutants to water courses. In flatter, lower lying areas this process more commonly occurs through leaching and washing of pollutants into artificial drainage channels (Chapman *et al.*, 2005; Nelson *et al.*, 2005; Schoumans *et al.*, 2014)

Concentrations of pollutants in runoff is highly dependent on a range of factors including the composition of soils, the gradient of land, the intensity of precipitation, the antecedent conditions prior to precipitation and the quantity and the timing and nature of the application of fertilisers and other agri-chemicals (Fielding and Smith, 2010; Schoumans *et al.*, 2014). High nutrient concentrations have been observed in runoff following manure and fertiliser applications (D. R. Smith *et al.*, 2007). Therefore it is important for good management that the BMP's are used to reduce pollutant runoff to the lowest possible levels.

Table 1 - Summary of Cost Action 869 Recommended Measures

1.	**Nutrient application management**	
1.1	General nutrient application management	A range of fact sheets covering details of good fertiliser management, such as, careful application rates ensuring economically optimal additions, avoiding addition of fertilisers and manures at high risk times and in high risk areas. Also the a system of recommending the most appropriate fertilisers based on soil testing for example avoidance of applying phosphate fertilisers to high phosphate index soils.
1.2	Inorganic fertilizer application management	A fact sheet advising how to reduce the phosphate content of common NPK fertilizers.
1.3	Manure production and application management	A range of fact sheets detailing good production and management techniques, including minimising the volumes of dirty water produced, increasing slurry storage capacities and move to a preference for solid manure rather than slurry systems.
1.4	Manure surplus management	A fact sheet recommending the incineration of poultry manure.
2.	**Crop management**	
2.1	Catch crops and cover crops	A fact sheet recommending the use of catch crops and cover crops to protect soils in between the growth of the main two annual crop growths.
3.	**Livestock management + production of minerals in manure**	
3.1	Overall production	Fact sheet covering optimisation / reduction overall stocking rates on livestock farms
3.2	Feeding	Fact sheets covering the reduction of phosphate and nitrogen inputs in animal nutrition and resulting reduction in concentrations in excreta.
3.3	Grazing management	Fact sheets covering the cutting of grasslands for hay and silage rather than cattle grazing, also the special management of livestock in riparian areas.
3.4	Point sources at farm scale	Fact sheets recommending the location of manure heaps away from drainage channels or impermeable areas.
4	**Soil management**	
4.1	Ploughing / tillage	A range of fact sheets detailing good ploughing and tilling practices, including using direct drilling not tillage in clay and silty soils, using practice which reduces nutrient losses, optimal cultivation times for different soils and techniques to reduce soil compaction and stratification.
4.2	Measures for preventing erosion / increase infiltration	Fact sheets with recommendations on how to reduce erosion and improve infiltration, including, cultivation to avoid uneven soils and tramlines, establishment of buffer strips, mulching and enhancement of soil organic matter content.
4.3	Miscellaneous	Sheet recommending phosphorus immobilizing amendments to soil.
5.	**Agricultural water management**	
5.1	Drainage - Reduce loss by overland flow to surface water	Facts sheets covering measures to reduce overland flow loss to surface water, including constructing pond and other retention systems, grassed waterways, sedimentation boxes and increasing soil levels in ditches.
5.2	Drainage - Reduce loss by subsurface flow to surface water	Sheets covering improvement to sub-surface drainage systems whiles ensuring reduction in losses from flows, removal of trenches and ditches and the effects of allowing field drainage systems to deteriorate.
5.3	Drainage - Reduce loss by artificial drainage to surface water	Fact sheets covering the losses via artificial drainage to subsurface water, covering methods to reduce water volumes leaving fields, such as interruption of artificial drainage and letting drainage water irrigate meadows.
5.4	Irrigation	Sheets covering mitigating nutrient loss by surface irrigation and recovery of tailwater from fields for water and nutrient cycling.
6.	**Land use change**	
6.1	Land use re-location and extensification	Fact sheet covering the beneficial effects of re-location of different farm land uses and spreading of activities.
7.	**Land infrastructure**	
7.1	Manage relationships between farm and rivers or streams	Sheets covering the management of the relationship between farms and rivers or streams, including minimisation of dirty water volumes.
7.2	Livestock and Stream management	Fact sheet covering the interactions of livestock and rivers or streams
7.3	Manage ditch, stream or river boundaries	Fact sheets covering the management of riparian zones, including information on management of field boundary vegetated buffers, maintenance and management of riparian wetlands, also the delineated function of the hydrographic systems.
7.4	Create, maintain or manage field boundaries	Sheets specifically covering the creation and management of field boundaries, detailing the buffering effects of field boundaries, hedges and their planting also the movement of field gate ways from high risk areas.
8.	**Measures in surface water**	
8.1	Channel management (ditches, streams)	Sheets covering the management of channels, including details of maintenance of vegetation and methods to protect banks and shorelines.
8.2	Restoration of surface waters	Fact sheets covering the restoration of surface waters, giving details of restoration of riparian wetlands, water course restoration, re-establishing inundated wetlands and lakes.
8.3	Technological instruments	Fact sheet covering the use of technical improvement measures such as constructed wetlands.
8.4	Options for abatement of eutrophication	Sheet covering the abatement of eutrophication, detailing the methods to inactivate excessive nutrients in lakes and ponds

Significant levels of guidance are available to farmers to help them manage the pollutant runoff from their activities. Cost Action 869 was an EU initiative running from 2006-11 in which 30 countries participated, with the aim of improving water quality through an assessment and scientific evaluation of suitability and cost effectiveness of various options to limit nutrient loss to surface waters (Chardon *et al.*, 2011).The project developed a catalogue of measures which would tackle the various causes and mitigate the effects of primarily nutrient pollution, although other pollutants were also considered. 83 measures were identified and grouped into 8 main categories (summarised in table 1). All the measures are described in detail at http://www.cost869.alterra.nl/Fs/List_of_options.htm (Schoumans *et al.*, 2014).

In England the Catchment Sensitive Farming (CSF) Scheme aims to deliver practical solutions through targeted support to enable farmers and land managers to take voluntary action to address diffuse agricultural water pollution (DWPA) in rivers, groundwater and other water sources. The scheme is run by the EA in coordination with Natural England, with funding being provided by DEFRA. Work is being carried out in 50 priority catchments as well as in 10 catchment partnerships. (CSF Evidence Team, 2011).

Natural England is also responsible for the implementation of the Environmental Stewardship schemes in the UK, these are agri-environmental schemes which provide financial incentives to farmers to encourage them to undertake environmental improvements such as reducing chemical applications (fertilisers and pesticides), lowering stocking levels per hectare, creating natural buffer strips around fields, creating ponds and tree or hedge planting. A review by Natural England (2009) identified that over 6 million hectares, over 65% of agricultural land was covered by the stewardship schemes, and approximately £400 million annually was paid to farmers for the delivery of environmental improvements. There are 4 different levels of stewardship with each level of additional improvements attracting higher levels of funding. These are, Entry Level Stewardship (ELS), Higher Level Stewardship (HLS), Organic Entry Level Stewardship (OELS) and Organic Higher Level Stewardship (OHLS). Schemes such as CSF and ES then provide a significant incentive to farmers to reduce their impact and enhance the environmental benefit of their activities.

There is a large regional variation in agricultural practice. Figures collected by DEFRA in 2010 break down agricultural land-use and show that in the North West a total of 896,000 ha of land is utilised for farming which is largely made up of 90,000 ha of Arable (10%), 659,000 ha of Grassland and Rough Grazing (74%), 17,000 ha of Forestry (2%) and 6,000 ha of horticulture (1%) (DEFRA, 2013; Ford *et al.*, 2008).

With such a large proportion of grazing and grassland used for livestock in the NW, the management of these areas is critical to the management of catchments in the region, where a large proportion of uplands and improved grasslands are heather moorland and they make a significant contribution to catchment areas. Indeed the UK as a whole contains 70% of the world's heather moorland, known as a plagioclimax community that if not managed through grazing and burning would ultimately return to woodland which is the climax of natural succession (Barker, 1995; Prosser and Wallace, 2000).

During the 1960s and 1970s much of the UK's upland areas were extensively managed to improve drainage in a process known as gripping (Holden *et al.*, 2004). Improved drainage lowered the water tables and lead to increased exposure of underlying peat layers and in turn this increased bacterial activity and oxidation leading to peat degradation (Mitchell and McDonald, 1992). Dissolved organic carbon (DOC) reduces in stream light penetration and mobilises other water pollutants such as metals, this requires additional treatment of water to be undertaken at potable water treatment works (Wallage *et al.*, 2006). Wallage *et al.* (2006) also observed that intact peats which had not been drained artificially retained significantly more DOC and associated pollutants. P. J. White and Hammond (2009) and Rothwell *et al.* (2010) indicated that nutrient concentrations in upland areas were low with effluent inputs from rural point sources such as package sewerage treatment works being the primary source of nutrient pollution to receiving waters.

The quality of water runoff from upland and moorland areas depends very much on this specific management activity. Marrs *et al.* (2000) indicated the intensity of burning as a management tool was increasing with the instances of continuous areas of moorland free from burning difficult to find, this was largely attributed to red grouse management and improvement of grazing. Burning and overgrazing can lead to significant degradation of runoff quality (McHugh, 2007), rapid erosion occurs across burned peat areas particularly around drains, here entrainment of exposed particles and material is easily washed from surfaces polluting runoff. Increases in drainage often results on increases in grazing however moorland is an inadequate to support high stocking densities (Holden *et al.*, 2004).

Haigh and Krecek (2006) indicate that although traditionally peat lands were considered to intercept and retain rainwater; unspoiled peat lands are in reality poor aquifers and generally facilitate surface flow and do not store significant volumes of rainwater. This can be exacerbated through the management practices discussed above resulting in accelerated runoff and erosion. Other hill farm practices such as the use of sheep dips and application of fertilisers and pesticides, such as asulam, used to control bracken (Scholefield *et al.*, 1993), can lead to further degradation of runoff. Pollution extent from uplands will be very dependent on the intensity of this style of management by farmers and land owners.

A range of practical measures and revised management practices are documented to mitigate the negative impacts discussed above. Some techniques take the form of direct restorative management including peat re-profiling or stabilisation, reseeding and replanting, as well as grip and gully blocking. Other techniques focus on changing management practice to adopt preventative methods such as more sustainable stocking levels, footpath building, selective vegetation removal and promotion of more sustainable use of agri-chemicals. It is also important to educate farmers, land owners and those using uplands for recreation to raise awareness of practices and activities leading to erosion and degradation and to promote sustainable upland and moorland use.(Critchley *et al.*, 2013; Milligan *et al.*, 2004; Walker *et al.*, 2008; Wallage *et al.*, 2006). The suitability of different techniques will vary depending on the original condition of the area or site in terms of the scale and severity of the restoration required as well the on-going management pressures on the site.

2 Afforestation and Forestry

It is clear that there is significant potential to improve both runoff quality and quantity from degraded uplands and that poorly managed land with exposed soils leads to greater erosion and transportation of pollutants and sediments into water courses. As well as changes in water yield, well forested and vegetated catchments retain sediment and pollutants more effectively and this can be attributed to the protection vegetation offers soils, in terms of intercepting rainfall and binding surface soils together firmly with dense root systems preventing soil and nutrient leeching (Brown *et al.*, 2005).

The UK Forestry Commission (2013) defines woodland as “land under stands of trees with a canopy cover of at least 20% (25% in Northern Ireland) or having the potential to achieve this”. This definition relates to land use rather than cover so areas of felled woodland awaiting restocking are considered woodland. It will also discount trees and woodland within areas included in a different land use classification. The latest available figures on woodland and forest (March, 2013) give the land area of the UK classified as wooded as 3.1 million hectares. This comprises of 1.4 million hectares (45%) in Scotland, 1.3 million hectares (42%) in England, 0.3 million hectares (10%) in Wales and 0.1 million hectares (4%) is in Northern Ireland. Overall approximately half of this (52%) is coniferous however this varies between regions with 26% in England and 76% in Scotland (Forestry Commission, 2013).

There is a well-established relationship between increases in runoff and erosion rates in response to changes in the amount and type of vegetation which covers an area and differences (even between tree species) of vegetation within a catchment area are important criteria which influence surface hydrological conditions (W. Zhang *et al.*, 2011). Interception and infiltrations rates vary depending on the abundance and nature of canopy and ground foliage, which provide protection and cover to soils and reducing surface water runoff (Ahtiainen and Huttunen, 1999).

It has been long proven that changes in vegetation cover can be manipulated to alter the volume of runoff yielded by an area of land. This is the effect of multiple processes associated with deforestation and reforestation interacting together. A review by (Bosch and Hewlett, 1982) into paired catchments (Paired catchment studies involve the use of two catchments with similar characteristics) reviewed 94 experimental catchments and concluded that generally increased cover reduces water yields and reduced cover increases yields. More specifically they found that:-

- Adding coniferous and eucalypt cover types caused a 40 mm change in annual water yield per 10% change in forest cover;
- Adding deciduous hardwoods are associated with a 25 mm change in annual water yield per 10% change in cover;
- Adding brush and grasslands are associated with a 10 mm change in annual water yield per 10% change in cover;

There are several key studies in paired catchments (Bosch and Hewlett, 1982; Hibbert, 1969; Hornbeck *et al.*, 1993) where increases in base flow are almost uniformly observed under catchment deforestation, although there is disagreement of how suitable forest is to facilitate infiltration of water (Wahl *et al.*, 2005). A study by Hibbert (1969) investigated a 22 acre catchment in the southern Appalachians where a hard wood forest was cleared to be replaced with a fescue grass. Levels of evapotranspiration from the grass cover correlated strongly with the amount of grass produced. Water yields from the catchment were similar or reduced to that expected from the original forest when grass production was high. With a decline in grass production water yields gradually increased until it exceeded the yield predicted from forest cover by over 120 mm annually.

Studies (Reinhardt *et al.*, 2011; Wheater and Evans, 2009) have proposed greater levels of afforestation within catchments as a low cost method to help to mitigate flood risk to the lower parts of the catchment. A paper by Wilkinson *et al.* (2014) identified that a typical situation for many small villages in the UK which are at risk of flooding but fail to meet the criteria for grant aid funding for construction of traditional flood defences, due to the high cost verses the small number of properties protected. In this situations the application of an alternative catchment based approach would offer a much cheaper option, and offer a range of additional benefits including water quality improvement (Gallart and Llorens, 2003; Reinhardt *et al.*, 2011). This is reinforced by Coulthard *et al.* (2005) who demonstrated how simulated deforestation of a catchment resulted in increased river discharge and sedimentation of runoff for a given storm event.

While re-vegetation offers water quality improvement, commercial forestry, like intensive farming, can result in a series of negative water quality issues similar to those associated with intensive arable farming. These include increased turbidity and sedimentations as a result of cultivation and soil disturbance, drainage, road constructions, cutting operations, leaching of nutrients from application of fertilisers (often applied aerially due to the size of plantations) and the increase of acid deposition in canopies resulting in acidification of runoff (Drinan *et al.*, 2013). Studies have shown that in the short term artificial drainage required in many upland commercial forestry plantations result in significant increases in river base flow as well was increasing storm peaks (Robinson, 1986).

Table 2 - Summary of Forests and Water Legal Requirements

Area	Summary of Requirements
Water Framework Directive	The water regulatory authority must give consent for any activates in or adjacent to watercourses that affects river hydromorphology, including water abstraction, impoundments, constructing culverts and extracting river gravel. If sites are subject to further special protection such as SSSI or SPA then additional authorisation from conservation agencies may be required.
Pollution control	Unless authorised by the water regulatory authority any entry of polluting materials to the water environment (with special attention given to waters containing fish) must not be caused or knowingly permitted. Scottish sites must conform to relevant general binding rules with any departures licensed or authorised by SEPA.
Control of pesticides	When the aerial application of pesticides is undertaken near or in water and designated sites, priority habitats or species may be affected, relevant regulators and conservation agencies must be consulted and if necessary authorisation obtained. Also all those employed in pesticide application must be fully trained to required standards or supervised by certified persons; all applications must comply fully with product labels.
Groundwater regulations and NVZ's	Regulatory authorities must be consulted regarding the disposal of harmful and polluting substances to land in the interests of protection of groundwaters.
Oil and fuel storage	Oil and fuel must be stored in a way that minimises the risks of leakage and pollution.
Water supplies	Forestry operations must not lead to harmful or polluting substances contaminating public or private water supplies.
Flood risk management	Appropriate regulators must be consulted for new woods next to flood defences, and the necessary consents obtained.
Waste management	Any application of wastes such as sewage sludge, waste soil or compost, waste wood, bark or other 'listed substances' to forest soils, must conform to conditions within required permissions or licences.
Aquatic habitats and species	Sites, habitats and species that are subject to legal provisions of EU directives and UK legislation must be subject to appropriate protection and conservation. Relevant authorities can offer advice to minimise impacts of management activities.

To address this the UK forestry commission developed a comprehensive set of best management practice and guidelines 'The UK Forestry Standard the Governments' approach to sustainable forest management'. This publication has been revised several times since its original introduction in 1988, the most recent update being in 2011, and this ensures it continues to embody the most up to date research and experience. Specifically within the forests and water section of the document there are two levels of compliance: Legal and Good forestry practice (UK Forestry Commission, 2011). Table 2 summarises the legal requirement practice for water protection.

The document compliments these legal requirements with 12 pieces of good practice around the areas of acidification, water quantity, water quality and buffer areas. This is further supported by 83 guidelines that inter relate to other important areas of forest management (Biodiversity, Climate Change, The Historic Environment, Landscape, People and Soils).

Compliance with best practice is a condition to obtain grant aid for forestry activities from the forestry commission (B. D'Arcy and Frost, 2001). Reviews by Carling *et al.* (2001) and Nisbet (2001) into water pollution from forestry, indicated that forest and water guidelines within UK Forestry Standard document were suitable to tackle the main issues around pollution of receiving waters but stressed the importance that guidelines were rigorously followed to ensure that negative impacts were avoided.

3 Catchment Management/ Restoration Project Case Examples

The different method and approaches to catchment restoration outlined above incorporate numerous techniques, which if applied individually may not result in significant impact and therefore need to be used in combination. The following three separate examples demonstrate differing approaches to improving water body chemical concentration through a range of different management activities. Two are based in the UK and one is based in New Zealand.

3.1 Sustainable Catchment Management Programme (SCAMP)

Organised by the North West based water company United Utilities (UU) and run in partnership with the RSPB, SCaMP was initiated in 2005. The project is funded as part of UU's AMP4 investment programme which UU have been permitted to make by OFWAT. The project aims to implement an integrated catchment management approach in two areas owed by UU, i.e., Bowland and the Peak District. These catchment areas feed a number of reservoirs that help supply over 7 million people across the NW, as well as being home to a large number of species of plants and animals. (UU, 2011)

Work is being undertaken in co-operation with farms, land managers, LA's, Government and other conservation organisations to influence how water catchment areas are managed and funded. Work undertaken is a combination of restorative measures and alterations in existing management and will include:-

- restoring blanket bogs by blocking drainage ditches
- restoring areas of eroded and exposed peat
- restoring hay meadows

- establishing clough woodland
- restoring heather moorland
- providing new farm buildings for indoor wintering of livestock and for lambing
- providing new waste management facilities to reduce run-off pollution of water courses
- fencing to keep livestock away from areas such as rivers and streams and from special habitats

The SCaMP project is set to run to 2015 but has already improved many upland areas with several reports showing improvement in the colour of and the amount of suspended solids in samples of runoff and river discharge (UU, 2010).

3.2 Lake District, Windermere Catchment Restoration Programme (WCRP)

The programme was established as a whole catchment approach to address a range of environmental issues affecting the Windermere catchment comprising of 7 major lakes and tarns. Founded in 2009 the WCRP is a partnership of 8 member organisations with responsibilities to protect and return lakes to their former high water quality. Incorporated in to the WCRP is a 5 year plan which outlines the long term vision for the catchment, it outlines the how the partnership operates and the responsibilities of different groups. Within this plan are 3 key strategic objectives which are

- Ecology – Improve and protect the water quality and natural ecology of the catchment
- Awareness – Educate residents and visitors about the environmental pressures affecting the lakes.
- Economy – Ensure that improvements to catchments support the local economy.

The specific management activities to achieve these goals differ between lakes and the 3 key objectives expand into 7 key technical objectives:-

- Controlling phosphorous concentrations within lakes
- Manage sedimentation
- Improving biodiversity and controlling invasive species
- Restore natural fish communities
- Ensure recreational access is managed sustainably
- Provide water quality suitable to meet public needs
- Locally mitigate the effects of climate change where possible

The plan identifies the links between work of the WCRP and the corporate strategies of partners. Each year a 1 year plan is produced to detail specific and measurable actions for that year.

3.3 Lake Okaro, New Zealand

This example represents a more intensive approach addressing not only the external contribution to nutrient pollution of a water body but treatment of that water body to address the existing pollution issues. It provides a good demonstration of how a combination of different techniques can be used to drastically reduce nutrient levels. Lake Okaro is a warm monomictic lake, meaning that for much of the year it is thermally stratified and the warmer top layer doesn't mix with the colder lower layer. During winter when the upper layers cool they mix thoroughly with the lower cold water. In the 1960's Okaro altered from an oligotrophic to eutrophic condition and during the period 2003-08 multiple restoration efforts were made to reverse the trend in the phosphorus concentrations in the lake. These efforts included the construction of a 2.3 ha purpose made wetland and protection of riparian margins reducing external loading. This was in conjunction with application of Alum (December 03) and modified zeolite (September 08) chemicals used to purify the water, to reduce internal loading. As a result of these measures total phosphorus concentration (TP) in the lake decreased by 41% between 2004 and 2008. (Özkundakci *et al.*, 2010)

The relatively rapid response of total phosphorous concentrations following the reduction of internal loading using modified zeolite suggests that this technique can be used to effect rapid decrease in TP concentrations in lakes. The trophic state of Lake Okaro show high resilience to the reduced Phosphous (P) loading. The paper concluded that the combination of approaches used in this case worked effectively together with treatment of water to reduce internal P concentrations, which could then stay lower after external measures reduced the rate at which P was entering the lake system (Özkundakci *et al.*, 2010).

These three different examples highlight the importance of not only ensuring that all affected stakeholders are engaged but that a divers mixture of different management activities, including both broader change to land practices and structural measures are required for goals to be achieved. This section has covered the two land uses in the UK which cover the greatest area of land, how this use can negatively affect water quality of surface and ground waters and the regulation in place which seeks to reduce and mitigate these effects. It demonstrates need for coordinated action between all affected stake holders across the catchment and that there is much common ground in recommended good practice from both the agriculture and forestry sector.

Appendix II – Sample Site Details

Sampling Locations for Wigan UDP Project

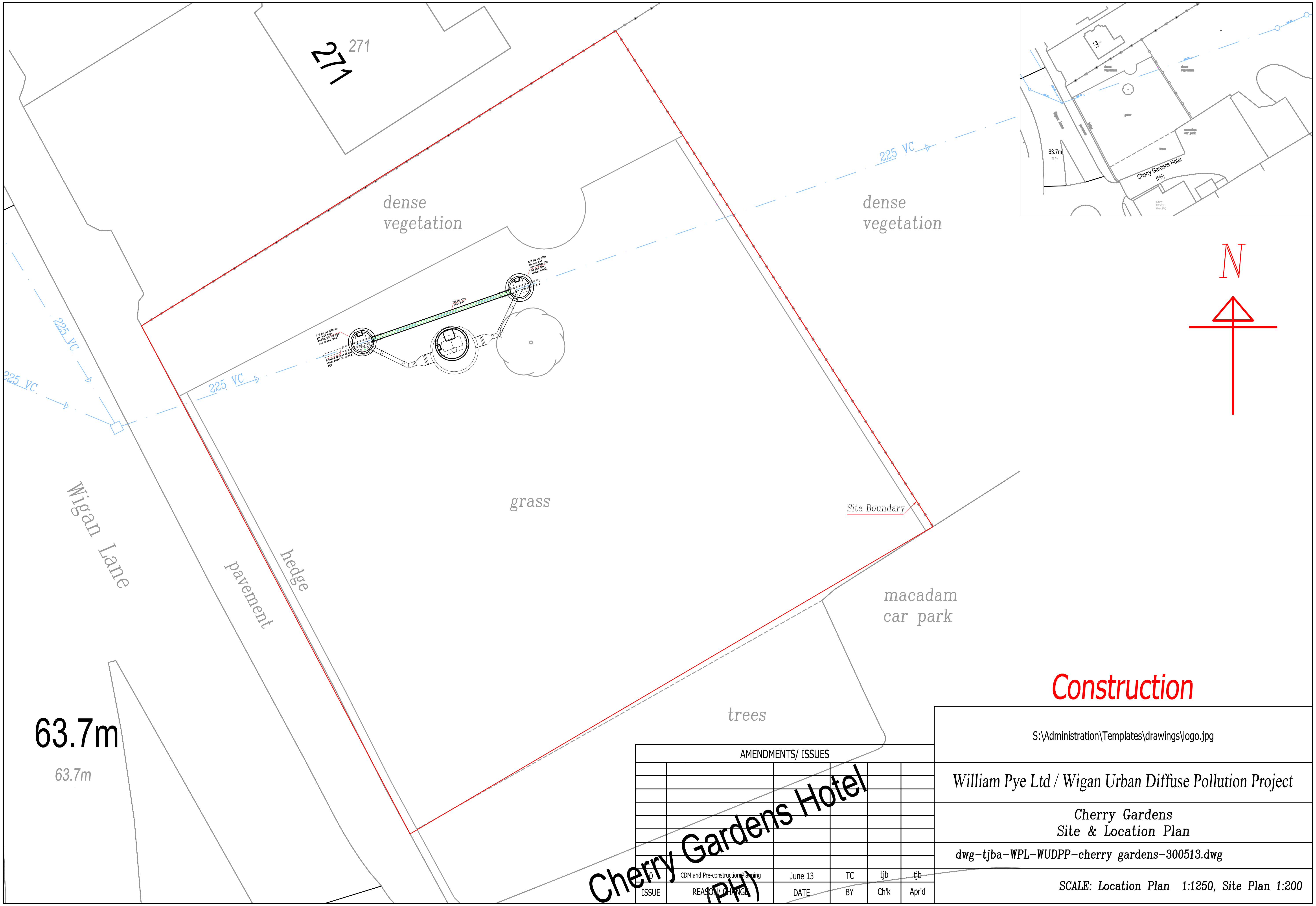
No	Site Name	Grid Reference	Latitude	Longitude	Geographic Order	Point Classification	EA Point Reference
	Key Points						
1	Douglas at Red Rock Bridge	SD57913 09867	53°35'1.31"N	2°38'13.90"W	2	Main Channel	88003093
2	Douglas @ Copple Lane Flood alleviatio	SD58779 06697	53°33'17.72"N	2°37'25.29"W	6	Main Channel	88023088
3	Douglas above Scholes Weir	SD58607 05404	53°32'42.29"N	2°37'36.38"W	8	Main Channel	88003102
4	Douglas @ Scot Lane Bridge	SD56177 06858	53°33'23.44"N	2°39'46.75"W	14	Main Channel	88003131
5	R Douglas @ Gathurst Bridge	SD54031 07353	53°33'41.38"N	2°41'43.74"W	19	Main Channel	88023090
6	R Douglas @ Appley Bridge	SD52309 09200	53°34'39.60"N	2°43'17.79"W	26	Main Channel	88023091
	Secondary Points						
7	R Douglas d/s Chorley Rd PS	SD58055 09386	53°34'45.77"N	2°38'5.90"W	3	Main Channel	88023092
8	R Douglas above Yellow Brook	SD58256 08162	53°34'7.13"N	2°37'58.44"W	4	Main Channel	88003096
9	R Douglas, Great Acre CSW, nr Tesco	SD58725 06174	53°33'2.05"N	2°37'27.94"W	7	Main Channel	88023093
10	R Douglas at Swan Meadow Lane	SD57798 04829	53°32'18.21"N	2°38'17.34"W	10	Main Channel	88003105
11	R Douglas at A49 Road Bridge	SD57113 05168	53°32'33.51"N	2°38'56.89"W	13	Main Channel	88003127
12	R Douglas at Pemberton Screens	SD56896 05898	53°32'54.39"N	2°39'11.22"W	17	Main Channel	88003128
13	R Douglas below Ackhurst Brook	SD54757 07451	53°33'42.19"N	2°41'4.22"W	21	Main Channel	88023094
14	R Douglas below Dean Brook	SD53378 07585	53°33'46.08"N	2°42'19.25"W	24	Main Channel	88023095
	Tributary Points						
15	Bradley Brook PTC R Douglas	SD57891 10165	53°35'12.01"N	2°38'16.97"W	1	Tributary	88003092
16	Yellow Brook PTC R Douglas	SD58765 06986	53°33'21.36"N	2°37'24.90"W	5	Tributary	88003108
17	Clarrington Brook	SD59489 05749	53°32'48.51"N	2°36'46.24"W	9	Tributary	88023110
18	Poolstock Brook	SD57728 04273	53°32'0.24"N	2°38'21.18"W	11	Tributary	88003126
19	Smithy Brook @ Lady Lane	SD56489 03608	53°31'38.17"N	2°39'32.26"W	12	Tributary	88003110
20	Sled Brook	SD57139 04958	53°32'22.26"N	2°38'53.51"W	13	Tributary	88023111
21	Barley Brook PTC Douglas	SD56981 05875	53°32'51.77"N	2°39'4.40"W	16	Tributary	88023097
22	Close Brook @ Stadium Way	SD56621 06019	53°32'56.43"N	2°39'22.21"W	18	Tributary	88023096
23	Ackhurst Brook PTC Douglas	SD54864 07308	53°33'37.61"N	2°40'57.64"W	20	Tributary	88003134
24	Dean Brook PTC Douglas	SD53394 07470	53°33'41.88"N	2°42'16.51"W	23	Tributary	88003135
25	Calico Brook PTC Road	SD52557 09179	53°34'36.91"N	2°43'5.01"W	25	Tributary	88003136

Appendix III – SAGIS Sample Point ID No's

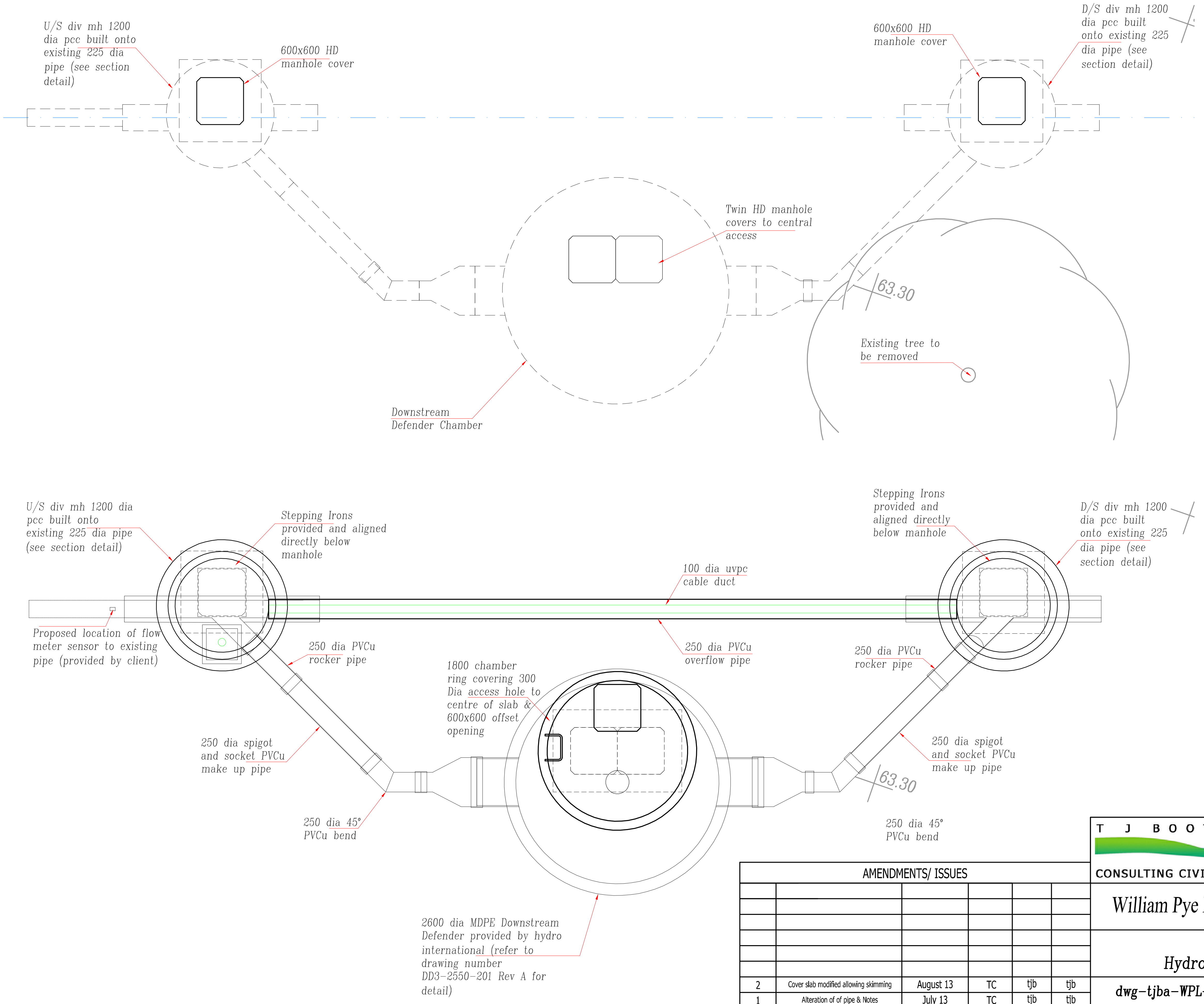
Default Values Taken from modle database			
Reach	**Feature**	**Site ID**	**Relevant Grid References**
River Douglas	WQ 88009654	0	
River Douglas	WQ 88009654	1	
River Douglas	End of reach River Douglas	2	
River Douglas	Start of reach River Douglas	3	
River Douglas	OSWwTW Input to Reach	4	
River Douglas	Urban Input to Reach	5	
River Douglas	Background Input to Reach	6	
River Douglas	Atmospheric Input to Reach	7	
River Douglas	Highway Input to Reach	8	
River Douglas	Extra Plot Point - Reach 221 No 1	9	
River Douglas	WQ 88003080	10	
River Douglas	CSO 574	11	
River Douglas	CSO 574	12	
River Douglas	End of reach River Douglas	13	
River Douglas	Start of reach River Douglas	14	
River Douglas	OSWwTW Input to Reach	15	
River Douglas	Urban Input to Reach	16	
River Douglas	Background Input to Reach	17	
River Douglas	Atmospheric Input to Reach	18	
River Douglas	Highway Input to Reach	19	
River Douglas	UN-NAMED WATERCOURSE FEEDING CONNECTE	20	
River Douglas	WQ 88003085	21	
River Douglas	CSO 571	22	
River Douglas	RESERVOIR AT ADLINGTON	23	
River Douglas	RESERVOIR AT ADLINGTON	24	
River Douglas	RESERVOIR AT ADLINGTON	25	
River Douglas	RESERVOIR AT ADLINGTON	26	
River Douglas	CSO 570	27	
River Douglas	CSO 568	28	
River Douglas	The Holmes	29	
River Douglas	CSO 569	30	
River Douglas	The Holmes Storm Tank	31	
River Douglas	Extra Plot Point - Reach 223 No 1	32	
River Douglas	Extra Plot Point - Reach 223 No 1	33	
River Douglas	End of reach River Douglas	34	
River Douglas	OSWwTW Input to Reach - Red Rock	35	SD5803610355
River Douglas	Urban Input to Reach	36	SD5803610355
River Douglas	Background Input to Reach	37	SD5803610355
River Douglas	Atmospheric Input to Reach	38	
River Douglas	Extra Plot Point - Reach 230 No 1	39	SD5805909475
River Douglas	CSO 699	40	SD5799509268
River Douglas	CSO 700 - CR Pump Station	41	SD5805009186
River Douglas	CSO 706 - Above Yellow Brook	42	SD5813508417
River Douglas	LEYLAND MILLS	43	
River Douglas	CSO 704	44	
River Douglas	WQ 88003108 D/S YB	45	SD5873206960
River Douglas	CSO 701 - Coppull Lane	46	SD5875206530
River Douglas	CSO 703	47	SD5875206530
River Douglas	CSO 702 - Tescos	48	SD5875306329
River Douglas	FS Douglas Central P	49	
River Douglas	WQ 88003102 - Scoles Weir	50	SD5860405398
River Douglas	GB112070064780 Boundary - Swan Meadow Lane	51	SD5755904897
River Douglas	End of reach River Douglas	52	
DOUGLAS	Start of reach DOUGLAS	53	
DOUGLAS	OSWwTW Input to Reach	54	
DOUGLAS	Urban Input to Reach	55	
DOUGLAS	Background Input to Reach	56	

DOUGLAS	Atmospheric Input to Reach	57	
DOUGLAS	Highway Input to Reach	58	
DOUGLAS	CSO 853	59	SD5748504861
DOUGLAS	CSO 853	60	SD5737005003
DOUGLAS	End of reach DOUGLAS	61	
River Douglas	Start of reach River Douglas	62	
River Douglas	OSWwTW Input to Reach	63	
River Douglas	Urban Input to Reach	64	
River Douglas	Background Input to Reach	65	
River Douglas	Atmospheric Input to Reach	66	
River Douglas	Highway Input to Reach	67	
River Douglas	WQ 88003126	68	SD5736705005
River Douglas	CSO 851 - A49 Road Bridge	69	SD5710305181
River Douglas	CSO 852	70	SD5710305181
River Douglas	CSO 849 - Pemberton Screens	71	SD5658506130
River Douglas	CSO 847	72	SD5624606682
River Douglas	CSO 850	73	SD5624606682
River Douglas	WQ 88003131 - MM Bridge	74	SD5606206949
River Douglas	CSO 846	75	SD5601007012
River Douglas	CSO 848	76	SD5601007012
River Douglas	CSO 844	77	SD5504807245
River Douglas	CSO 838 - Canal Overflow	78	SD5479207447
River Douglas	CSO 842 - Gathhurst Bridge	79	SD5399807368
River Douglas	CSO 843	80	SD5341907473
River Douglas	CSO 843	81	
River Douglas	End of reach River Douglas	82	
DOUGLAS	CSO 840	83	SD5254805536
DOUGLAS	CSO 841	84	SD5316106636
DOUGLAS	WQ 88003135	85	SD5339807459
DOUGLAS	WQ 88003135 - Dean Brook Join	86	SD5341907473
DOUGLAS	End of reach DOUGLAS	87	
River Douglas	Start of reach River Douglas	88	
River Douglas	OSWwTW Input to Reach	89	
River Douglas	Urban Input to Reach	90	
River Douglas	Background Input to Reach	91	
River Douglas	Atmospheric Input to Reach	92	
River Douglas	Highway Input to Reach	93	
River Douglas	Extra Plot Point - Reach 236 No 1	94	SD5292308155
River Douglas	Extra Plot Point - Reach 236 No 2	95	SD5265609015
River Douglas	Extra Plot Point - Reach 236 No 2	96	SD5247709054
River Douglas	End of reach River Douglas	97	
River Douglas	Start of reach River Douglas	98	
River Douglas	OSWwTW Input to Reach	99	
River Douglas	Urban Input to Reach	100	
River Douglas	Background Input to Reach	101	
River Douglas	Atmospheric Input to Reach	102	
River Douglas	Highway Input to Reach	103	
River Douglas	CSO 836 - Appley Bridge	104	SD5225909204
River Douglas	CSO 839	105	SD5225909204
River Douglas	Extra Plot Point - Reach 238 No 1	106	
River Douglas	Extra Plot Point - Reach 238 No 2	107	SD5135409486
River Douglas	Extra Plot Point - Reach 238 No 3	108	SD5066109968
River Douglas	Extra Plot Point - Reach 238 No 4	109	SD4973110086
River Douglas	WQ 88003137	110	
River Douglas	CSO 835	111	
River Douglas	CSO 833	112	
River Douglas	CSO 834	113	
River Douglas	CSO 832	114	
River Douglas	SKELMERSDALE STW	115	
River Douglas	WIGAN WWTW	116	
River Douglas	Skelmersdale Storm Tank	117	
River Douglas	Wigan Storm Tank	118	
River Douglas	WQ 88003146	119	
River Douglas	WQ 88003146	120	

Appendix IV – Downstream Defender Plans and Technical Drawings



271
271
dense vegetation
dense vegetation
225 VC
225 VC
225 VC
225 VC
grass
Site Boundary
hedge
pavement
Wigan Lane
macadam car park
trees
63.7m
63.7m
Cherry Gardens Hotel (PH)
N
Construction
S:\Administration\Templates\drawings\logo.jpg
William Pye Ltd / Wigan Urban Diffuse Pollution Project
Cherry Gardens
Site & Location Plan
dwg-tjba-WPL-WUDPP-cherry gardens-300513.dwg
SCALE: Location Plan 1:1250, Site Plan 1:200
AMENDMENTS/ ISSUES
0
CDM and Pre-construction Planning
June 13
TC
tjb
tjb
ISSUE
REASON/ CHANGE
DATE
BY
Ch'k
Apr'd



Construction

T J BOOTH ASSOCIATES

CONSULTING CIVIL & GEOTECHNICAL ENGINEERS

20A Eastgate Street
Rochdale
Lancashire
OL16 1DH
Tel:01706 868288

William Pye Ltd / Wigan Urban Diffuse Pollution Project

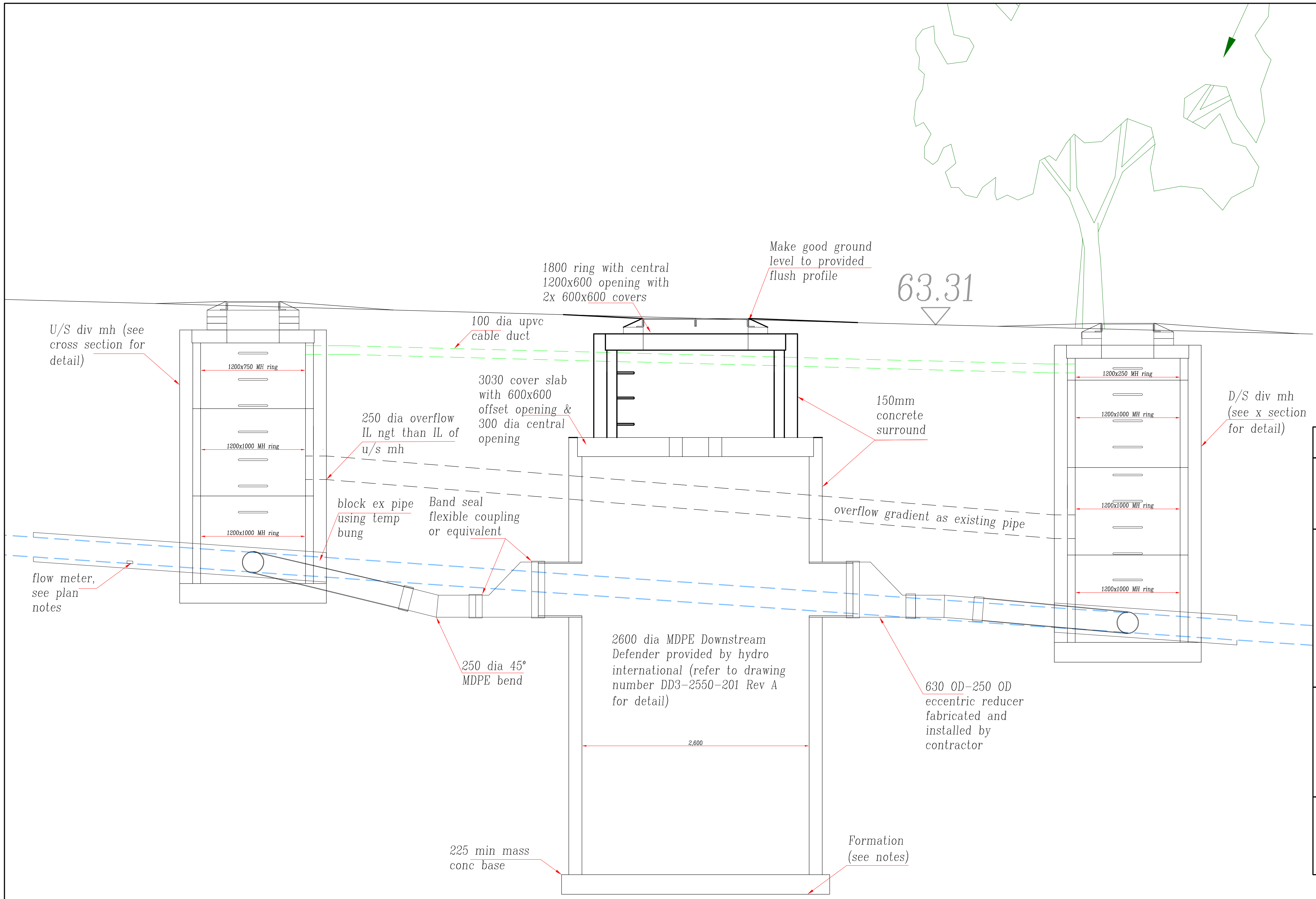
Cherry Gardens
Hydro Intn'l Downstream Defender Plan

dwg-tjba-WPL-WUDPP-cherry gardens-300513.dwg

SCALE: 1 : 50 @ A3

AMENDMENTS/ ISSUES

ISSUE	REASON/ CHANGE	DATE	BY	Ch'k	Apr'd
2	Cover slab modified allowing skimming	August 13	TC	tjb	tjb
1	Alteration of of pipe & Notes	July 13	TC	tjb	tjb
0	CDM and Pre-construction Planning	June 13	TC	tjb	tjb



Safety, health and environmental information

In addition to the hazards/risks normally associated with the types of work detailed on this drawing, note the following significant residual risks

Construction
- Concreting operations
- Heavy lifting operations
- Deep excavations
- Working from height
- Buried services
- Unforeseen ground conditions
- Sewer flooding

Maintenance/cleaning/operation
- Manhole access and inspection
- Placing & checking sample line & flow meter
- lifting sampler from manhole
- Desludging of defender

Decommissioning/demolition
- Closure of sewer diversion
- Reinstatement of main sewer flow

Construction

AMENDMENTS/ ISSUES

ISSUE	REASON/ CHANGE	DATE	BY	Ch'k	Apr'd
3	Cover slab modified allowing skimming	August 13	TC	tjb	tjb
2	Alteration of of pipe & Notes	July 13	TC	tjb	tjb
1	SHE Box addition	July 13	TC	tjb	tjb
0	CDM and Pre-construction Planning	June 13	TC	tjb	tjb

T J BOOTH ASSOCIATES
CONSULTING CIVIL & GEOTECHNICAL ENGINEERS

20A Eastgate Street
Rochdale
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Tel:01706 868288

William Pye Ltd / Wigan Urban Diffuse Pollution Project

Cherry Gardens
Hydro Intn'l Downstream Defender Long Section

dwg-tjba-WPL-WUDPP-cherry gardens-300513.dwg

SCALE: 1 : 50 @ A3

DO NOT SCALE

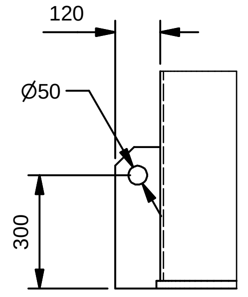
IF IN DOUBT ASK

PROJECTION

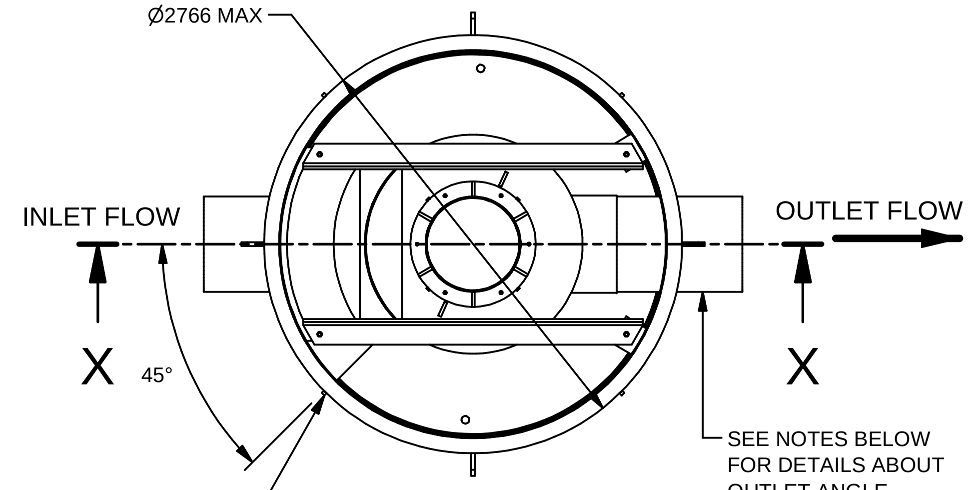
Notes
1. THIS DRAWING IS AN A3 SIZE ORIGINAL
2. ALL DIMENSIONS IN MILLIMETRES UNLESS OTHERWISE STATED.
3. THIS DRAWING SHALL BE READ IN CONJUNCTION WITH ALL RELEVANT GENERAL ARRANGEMENT & DETAIL DRAWINGS.

Parts List

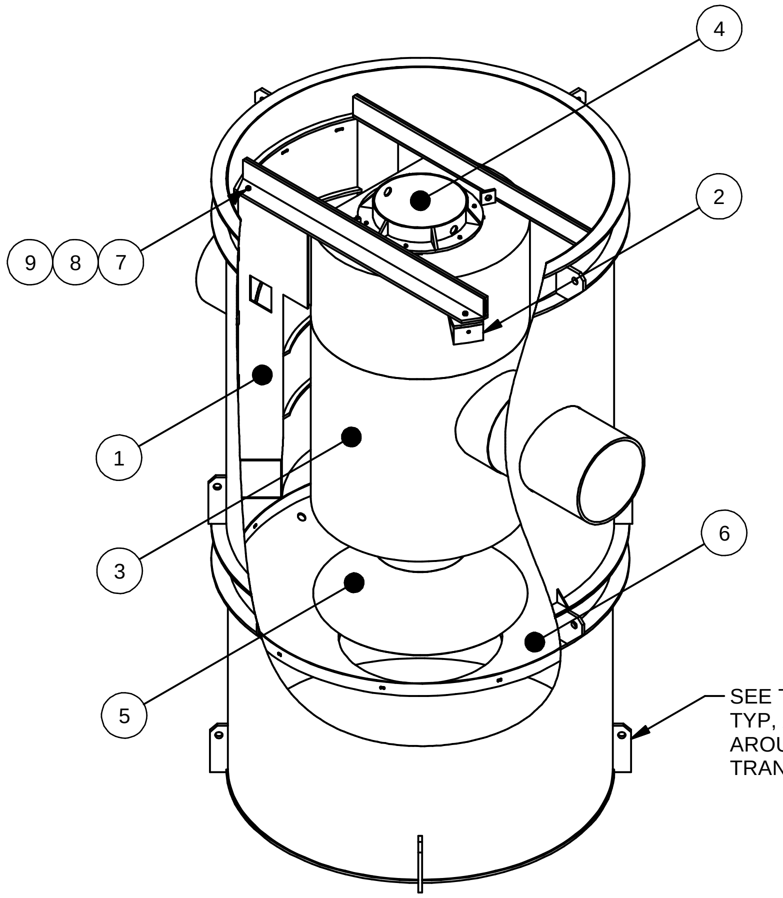
ITEM	QTY	COMPONENT DESCRIPTION	DRG NO	MATERIAL
1	1	Tangential Chute	DD3-2550-10	HDPE Plastic
2	2	Ledger Angle	DD3-2550-18	HDPE Plastic
3	1	Dip Plate Sub Assembly	DD3-2550-11	HDPE Plastic
4	1	Centre Shaft	DD3-2550-12	HDPE Plastic
5	1	Centre Shaft Cone	DD3-2550-13	HDPE Plastic
6	1	Benching Skirt	DD3-2550-20	HDPE Plastic
7	4	M16x60 Bolt		SS 304
8	8	M16 Washer		SS 304
9	4	M16 Nut		SS 304

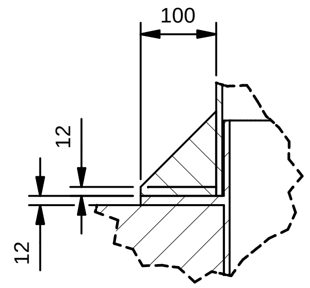


TRANSPORT LUG DETAIL

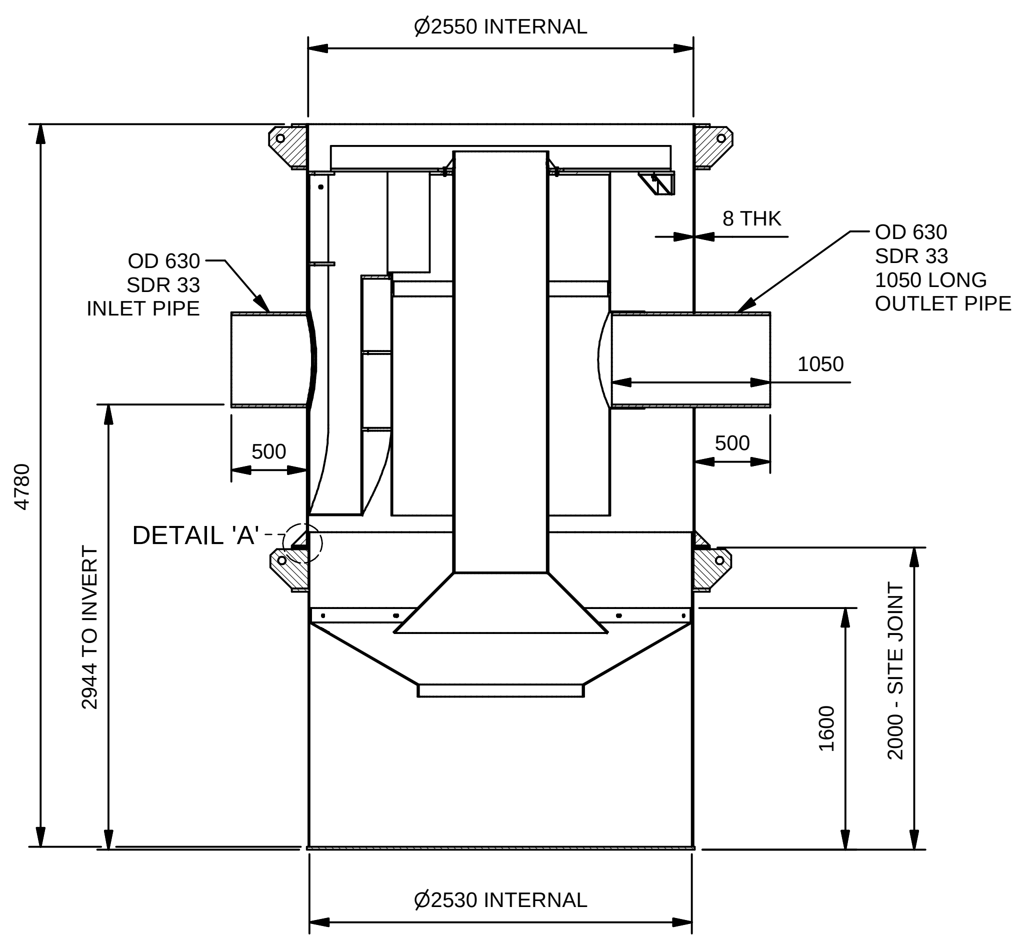


PLAN VIEW





DETAIL 'A'
(1:10)



SECTION X-X

ADDITIONAL NOTES:

1) CHECK INDIVIDUAL PART DRAWINGS FOR NOTES ABOUT DIRECT ATTACHMENT TO THE PLASTIC CHAMBER.

2) THE OUTLET PIPE CAN BE POSITIONED AT ANY ANGLE ACCORDING TO CUSTOMERS SITE REQUIREMENTS.

OUTLET HOLE POSITION SHOULD BE MEASURED CLOCKWISE FROM INLET AS DEFINED BY CUSTOMER OR AT 180° TO INLET (STRAIGHT) IF NO ANGLE IS PROVIDED. DRAWING SHOWS 180° STRAIGHT POSITION AS DEFAULT.

TOTAL WEIGHT = 870 Kg.

Rev	By	Date	Description
A	APN	25.02.09	First Issue Previously DD3-2550-PGA-E

Date: 25.02.09
Scale: 1:50
Drawn By: APN
Checked By: KGH
Approved By:

Title
Downstream Defender® III
2.55M HDPE Chamber
HDPE Internals

Manufacturers
General Arrangement

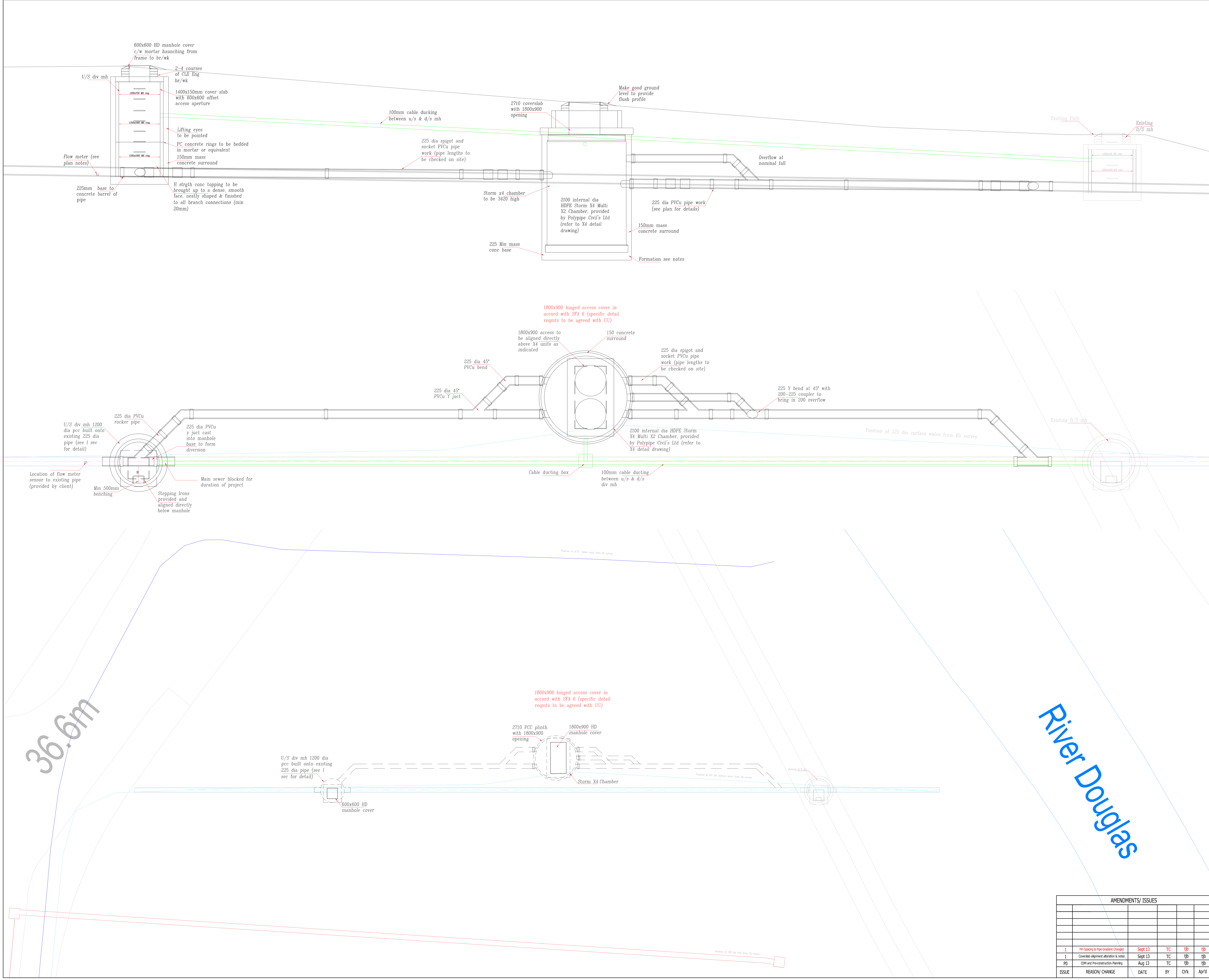
Hydro International

Shearwater House,
Clevedon Hall Estate,
Victoria Road, Clevedon,
North Somerset. BS21 7RD
Tel: 01275 878371
Fax: 01275 874979
email:
enquiries@hydro-international.co.uk

Ref. No.

Drawing No. DD3-2550-201 Rev A

Appendix V – Storm X4 Plans and Technical Drawings



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0.1 THIS DRAWING IS TO BE READ IN CONJUNCTION WITH ALL RELEVANT CONTRACTUAL DOCUMENTS.
0.2 The drawings will take precedence over the Sewers for Adoption (SFA) & Civil Engineering specification for the Water Industry (CESWI), if in doubt seek clarification.
0.3 All dims in mm unless o/w stated.
0.4 All levels in m aOD unless o/w stated.
0.5 Datum from surface water outfall IL from EA survey drawing number 52957
0.6 Contractor to confirm ALL SOP's & datum prior to ordering material/ construction.

1.0 WORKMANSHIP
1.1 All workmanship and materials to comply with Building Regulations British Standards and Codes of Practice. All materials to be stored, mixed and fixed in accordance with manufacturer's instructions and recommendations.
1.2 All provisions for temporary means of support & regulations for Health & Safety at Work to be complied with.
1.3 All works to be carried out to the complete satisfaction of the Environment Agency
1.4 All works to be carried out in strict accordance with HSE guidance and CDM regulations.
1.5 Do not scale, the contractor must verify all dims prior to commencing shop drawings or work on site.
1.6 All services above and below ground to be located, identified, de-commissioned, diverted or protected as necessary prior to and during the course of the work and reinstated on completion, in accordance with relevant codes of practice and statutory undertakers requirements.

2.0 MASONRY
2.1 Engineering Brick: Class 2, second quality, used in areas noted.
2.2 Mortar mix: 1:1:6 mix cement: lime: sand, proportions by volume, designation (iii) to BS 5628.

3.0 EXTERNAL WORKS & FINISHES
3.0 Ground levels to brought flush to all manhole surfaces

4.0 EARTHWORKS to comply with BS 6031.
4.1 Topsoil remove existing top soil and set a side for reuse in a bund rolled/ sealed faces
4.2 Formations: remove soft spots and any deleterious materials, replace with clean compacted granular fill or weak conc (as noted).
4.3 Back filling: clean surplus excavated material carefully compacted in layers.
4.4 Weak Concrete provide C10 concrete, nlt 100kg OPC per cum, w/c ratio 0.65. Temporary propping required until concrete cured.
4.5 Landscaped Areas; 100 re use top soil with grass seeding broad cast in accordance with BS 4428:1989.
4.6 Pipe bedding and surrounds back fill to trenches refer to EA Drawing number TD21A - Pipe Bedding Detail

5.0 CONCRETE.
5.1 Reinforced concrete slabs & walls to BS 8110 & BS 8500-2
5.2 Mass Concrete. Comp strength class C20, max w/c ratio 0.65, min cement combination content 250kg/cu m, consistency S3.

6.0 MECHANICAL & ELECTRICAL
6.1 Trenches: provide min 450 cover to cable ducts.
6.2 Ducts: 50-150ø green cable duct with "Cable" embossed in black to outer walls. Provide draw rope in each. sand backfill to trench around ducts. Marker tape bearing the legend "Caution Cable Below" provided above.

7.0 EXCAVATIONS & TEMPORARY WORKS
7.1 Excavations thro' gravel: adequately support at all times the sides of pits and trenches.
7.2 Contractor shall arrange for the rapid dispersal of water shed onto the earthworks or completed formation or into excavations during construction.

8.0 PIPE WORK
8.1 All 225 & 200 dia pipe work shall be Wavins Osma (PVCu) pipe, or equivalent

Referenced Drawings

1 Existing Plan, Section & Location Plan (rev1)
2 Proposed Plan, Section and External Plan (rev2)
3 Pipe Bed Detail
4 Storm X4 Detail (revA)

Site investigation Notes
1 11/03/2013 Trial hole by William Pye Ltd
2 16/09/2013 U/S & D/S Trial holes by William Pye to establish pipe gradient and ground conditions

Safety, health and environmental information

In addition to the hazards/risks normally associated with the types of work detailed on this drawing, note the following significant residual risks

Construction
- Concreting operations
- Heavy lifting operations
- Deep excavations
- Working from height
- Buried services
- Unforeseen ground conditions
- Sewer flooding

Maintenance/cleaning/operation
- Manhole access and inspection
- Placing & checking sample line & flow meter
- lifting sampler from manhole
- Desludging of defender

Decommissioning/demolition
- Closure of sewer diversion
- Reinstatement of main sewer flow

AMENDMENTS/ ISSUES

ISSUE	REASON/ CHANGE	DATE	BY	Ch'k	Apr'd
1	MH Spacing & Pipe Gradient Changed	Sept 13	TC	tjb	tjb
1	Coverslab alignment alteration & notes	Sept 13	TC	tjb	tjb
P0	CDM and Pre-construction Planning	Aug 13	TC	tjb	tjb

Construction

T J BOOTH ASSOCIATES
CONSULTING CIVIL & GEOTECHNICAL ENGINEERS
20A Eastgate Street
Rochdale
Lancashire
OL16 1DH
Tel:01706 868288

William Pye Ltd / Wigan Urban Diffuse Pollution Project

Coppull Lane
Proposed Plan & Sections

dwg-tjba-WPL-WUDPP-coppul lane-290713

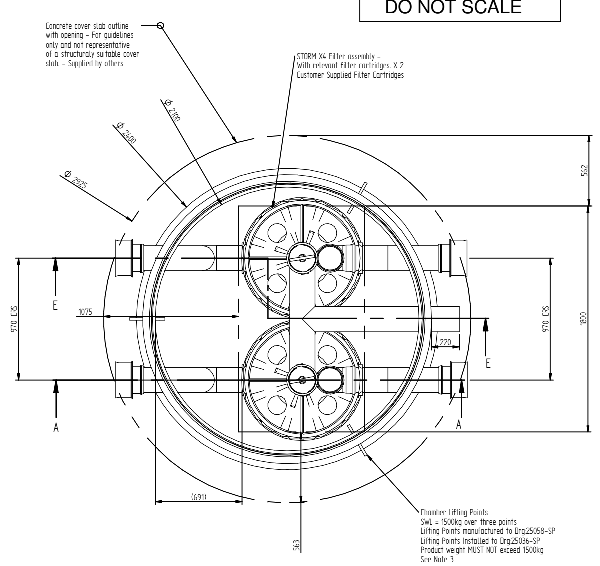
SCALE: 1 : 50 @ A1

DO NOT SCALE

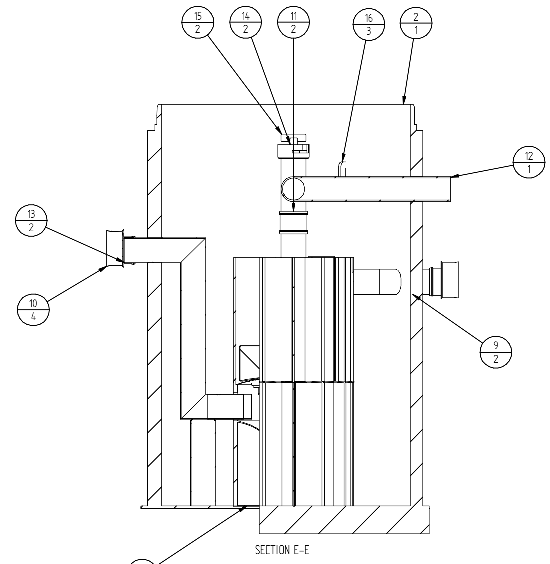
IF IN DOUBT ASK

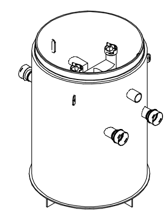
DRG N°

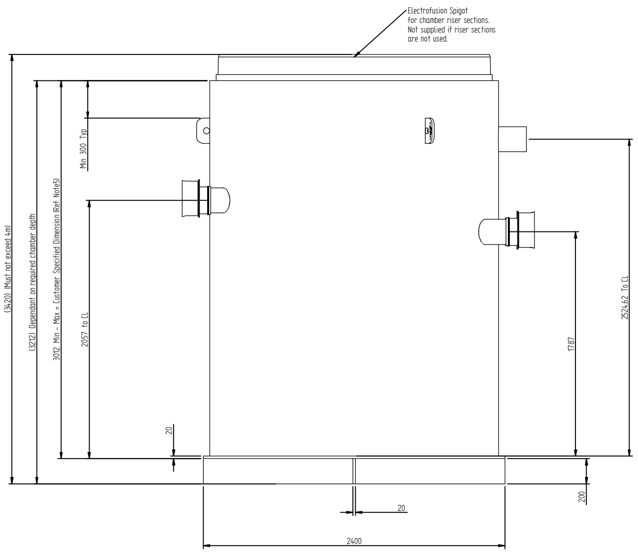
ISSUE	CHANGE	DATE
1	AS DRAWN	07/11/2008
2	Seperation of inlet Pipes from Manafold system	27/02/09
3	Pipe work change from RD to SDR41	16/04/13
4	SX4 units moved to aid manufacturing	22/04/13
5	Addition of Cover slab guidance detail	05/05/13

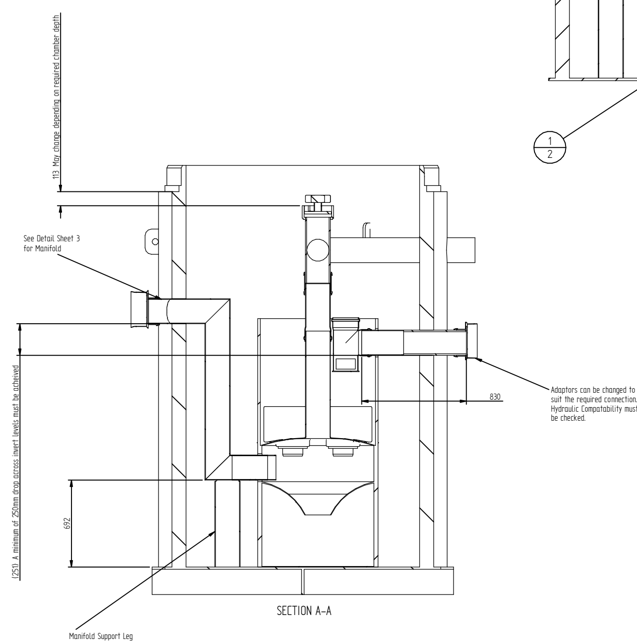


NOTE 1: Where the chamber assembly is to be transported as a single unit, it will have to be transported in the vertical plane. Thus special transport will be required due to the height of the chamber.
NOTE 2: Where chambers higher then the standard chamber are required, chamber riser sections should be electrofusion welded on to the chamber.
NOTE 3: Lifting Points must be installed and proof tested in accordance with the correct Polypipe procedure. Official Polypipe lifting guidance must be supplied to the customer and more specifically the competent person responsible for on site lifting.
The Polypipe lifting guidance should be stuck to the chamber in a sealed plastic bag.
NOTE 4: Lifting points available from April 30th 2013. Chambers before this date will not be fitted with them.
NOTE 5: If the chamber exceeds 1500kg it can not be supplied with fabricated lifting points and an alternative lifting method will have to be used.









Item Number	Document Number	Title	Material	Quantity
1	21032	Storm X4 Fabricated Chamber		2
2	20225	Storm X4 Multiple Chamber	Polyethylene ,	1
9	N/A	Φ200mm-SDR41-Pressure Pipe	Polyethylene ,	2
10	21033	20137-225mmΦ to SDR41 Adaptor	Polypropylene, general purpose	4
11	23077	200mmΦ Pressure Pipe Coupler - Pipelife	Polypropylene, general purpose	2
12	25031	StormX4 - Overflow Manifold - SDR41	Polyethylene , low density	1
13	25064-P	SX4 Multi - Backdrop Pipe work	Polyethylene ,	2
14	24138P	200mm-BayonetEndCap-Fitting	Polyethylene ,	2
15	24139P	200mm-BayonetEndCap-HatchPlate	Polyethylene , low density	2
16	25058-SP	Manhole Lifting Lug	Polyethylene ,	3

ITEM	DESCRIPTION	MATERIAL	QTY

Polypipe Civils Ltd.
Civils & Agricultural Products
UNION WORKS, BISHOP MEADOW ROAD, LOUGHBOROUGH, LEICESTERSHIRE, LE11 5RE.

TOOL N°	MATERIAL	EST. WEIGHT	GENERAL WALL THICKNESS
N/A	HDPE	1456kg	N/A

ALL DIMENSIONS IN MILLIMETRES UNLESS OTHERWISE STATED

ALL IMPRESSIONS TO BE CLEARLY MARKED

GENERAL MACHINE 3.2
FINE MACHINE 1.6
GRINDING 0.8
THIRD ANGLE

TOLERANCES (UNLESS OTHERWISE STATED)
OPEN DIM ± 0.40 — ANGLES
.0 ± 0.25 — ± 0° 30'
.00 ± 0.10

SCALE 1:20

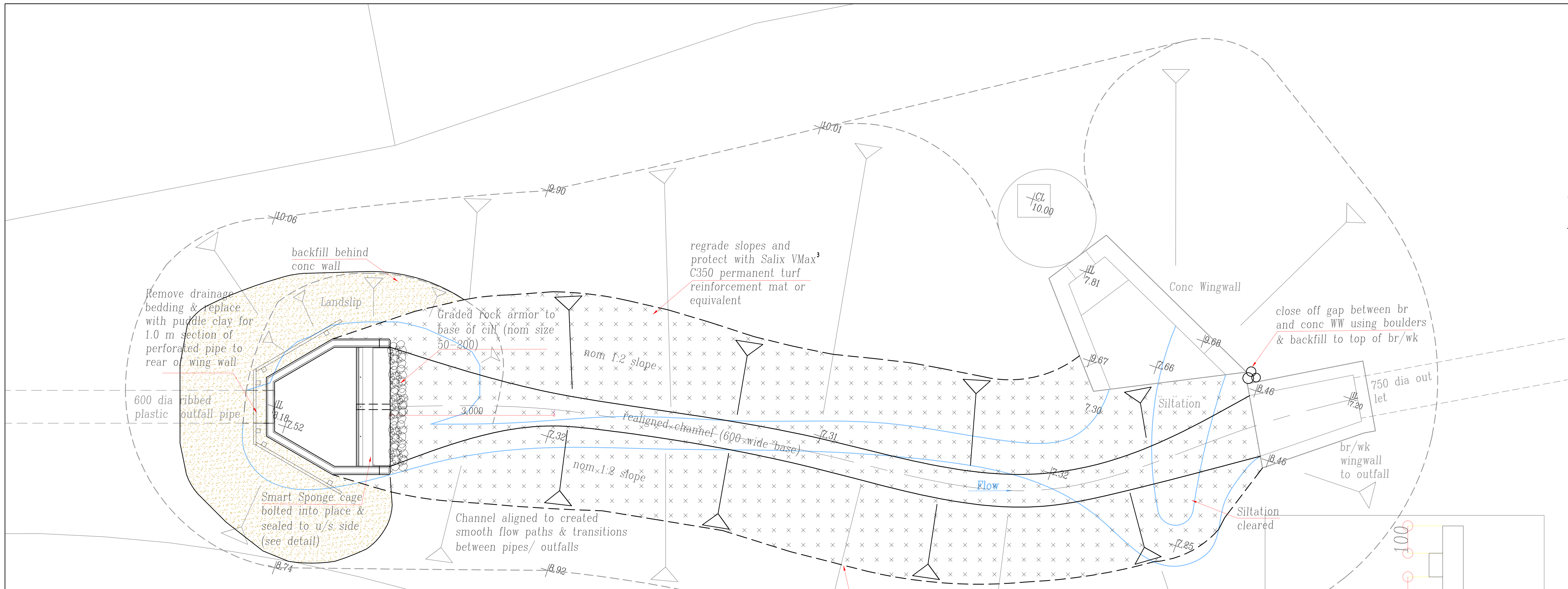
DRAWN BY stuart.ramella DATE 07/11/2008
CHK'D C Ness DATE 19/04/13
APPROVED S Ramella DATE 19/04/13

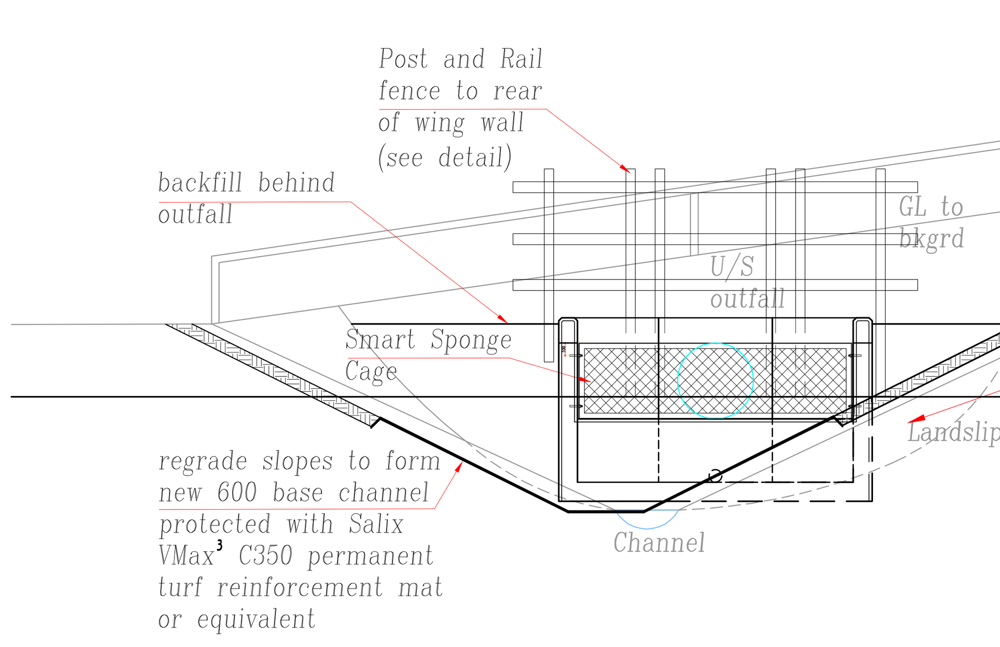
TITLE
Storm X4 Multi X2 Chamber (2100mmΦ)

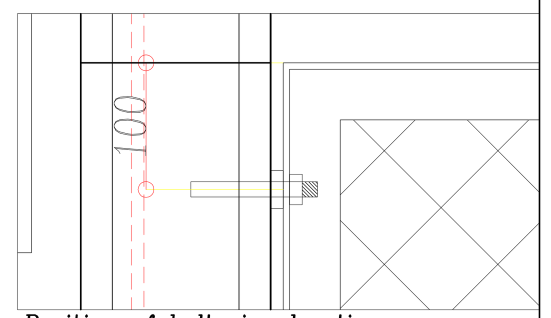
DRAWING No.

SHEET 2 OF 5

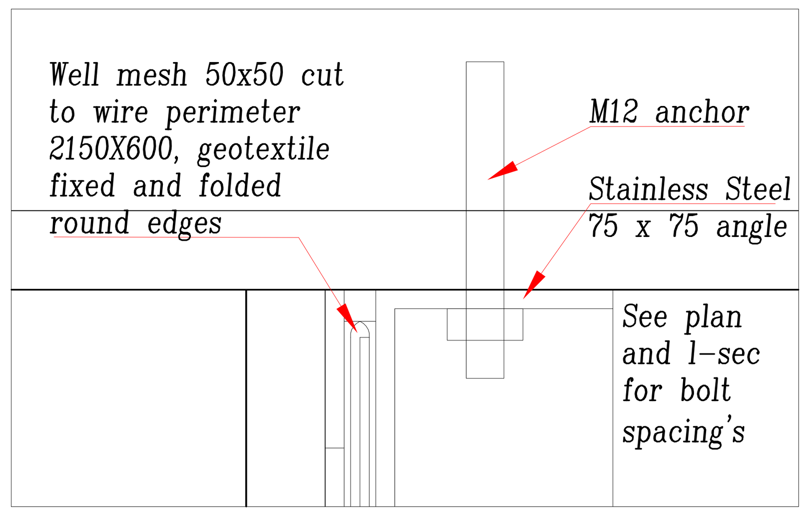
Appendix VI – Smart Sponge Scot Lane Drawings



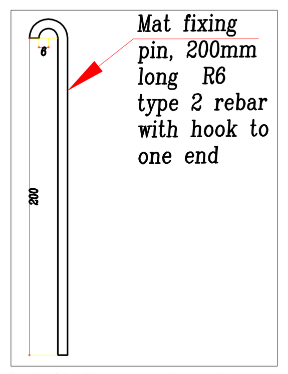




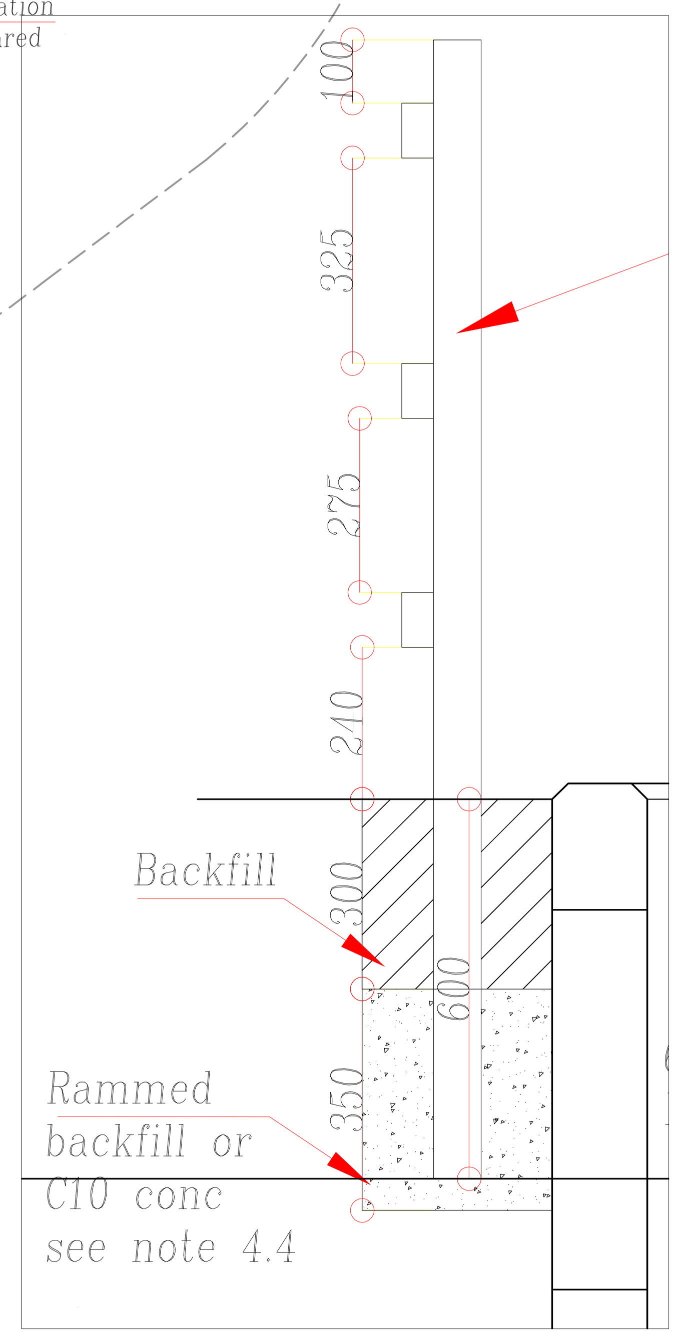
Position of bolts in elevation



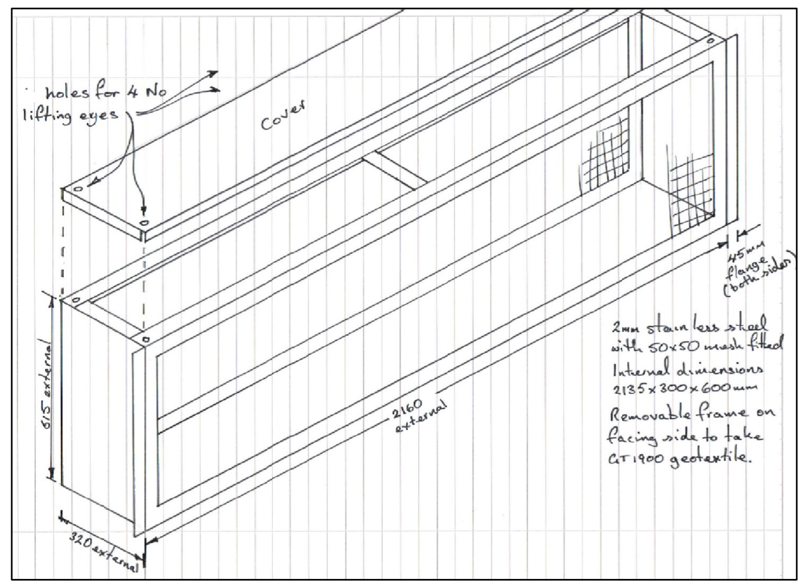
Silt Screen Fixing Detail



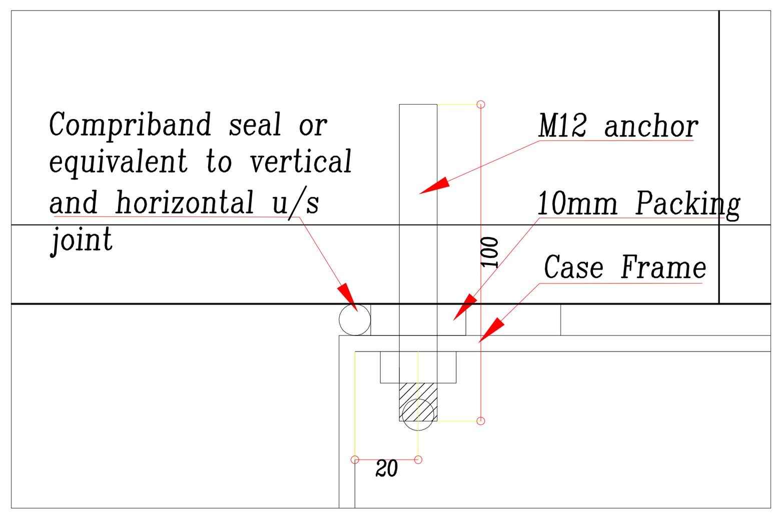
VMAX Fixing Pin Detail



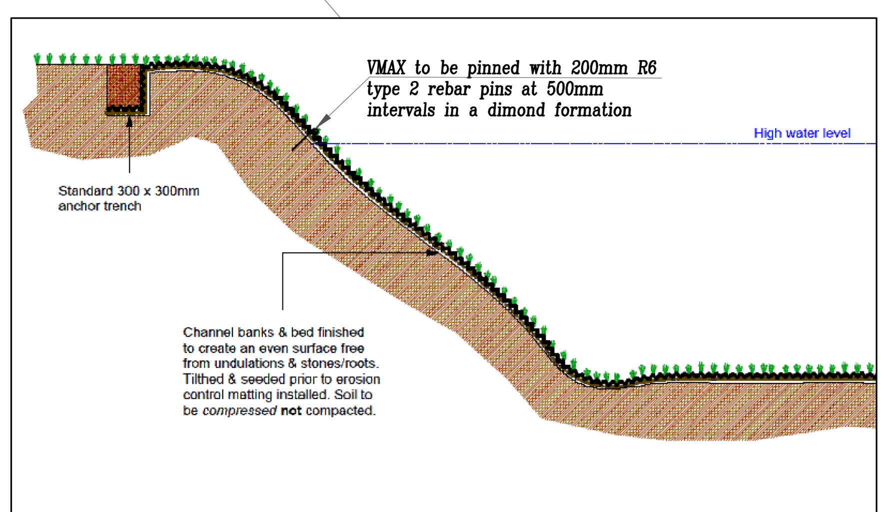
Post Casting Detail



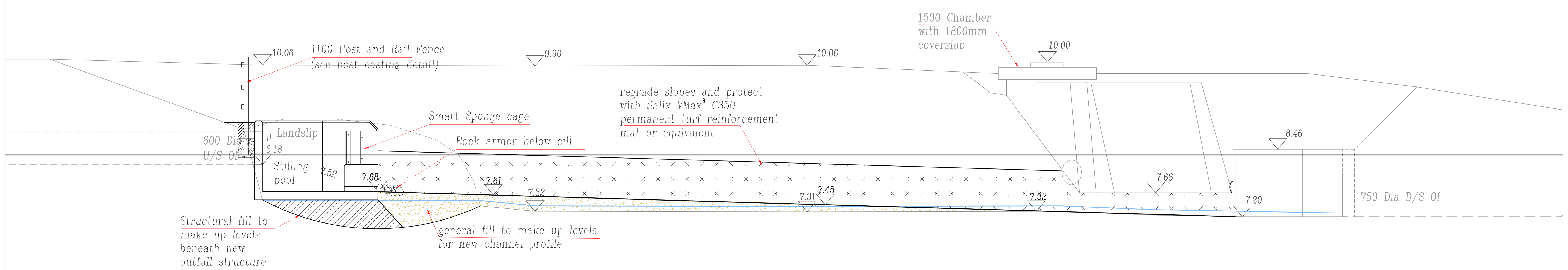
Smart Sponge Frame Detail



Frame Fixing Detail



Matting installation typical cross section



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0.1 THIS DRAWING IS TO BE READ IN CONJUNCTION WITH ALL RELEVANT CONTRACTUAL DOCUMENTS.

0.2 The work should be completed in accordance with Civil Engineering Specification for the Water Industry (CESWI)

0.3 The drawings will take precedence over the Civil Engineering Specification for the Water Industry (CESWI), if in doubt seek clarification.

0.4 All dims in mm unless o/w stated.

0.5 All levels in m aOD unless o/w stated.

0.6 Datum existing from manhole ID

0.7 Contractor to confirm ALL SOP's & datum prior to ordering material/ construction.

1.0 WORKMANSHIP

1.1 All workmanship and materials to comply with Building Regulations British Standards and Codes of Practice. All materials to be stored, mixed and fixed in accordance with manufacturer's instructions and recommendations.

1.2 All provisions for temporary means of support & regulations for Health & Safety at Work to be complied with.

1.3 All works to be carried out to the complete satisfaction of the Environment Agency

1.4 All works to be carried out in strict accordance with HSE guidance and CDM regulations.

1.5 Do not scale, the contractor must verify all dims prior to commencing shop drawings or work on site.

1.6 All services above and below ground to be located, identified, de-commissioned, diverted or protected as necessary prior to and during the construction of the work and reinstated on completion, in accordance with relevant codes of practice and statutory undertakers requirements.

2.0 EXTERNAL WORKS & FINISHES

2.1 Finishes to brought flush to existing ground levels

2.2 Salix VMax3 C350 or equivalent to be used to line reformed channel, to be roled at right angles to the direction of flow and overlapped on the d/s site by 100mm.

3.0 EARTHWORKS to comply with BS 6031.

3.1 Topsoil, remove existing top soil and set aside for reuse in a bund with rolled/ sealed faces

3.2 Formations: remove soft spots and any deleterious materials, replace with clean compacted granular fill or compacted 6F1 beneath structures.

3.3 General filling: clean surplus excavated material set aside for reuse in carefully compacted layers.

3.4 Back filling: clean surplus excavated material carefully compacted in layers.

3.5 Filling beneath structures: well compacted granular material in carefully compacted layer (6F1)

3.6 Landscaped Areas; 100 re use top soil with grass seeding broad cast in accordance with BS 4428:1989.

4.0 CONCRETE.

4.1 Reinforced concrete slabs & walls to BS 8110 & BS 8500-2

4.2 Mass Concrete. Strength class C20, max w/c ratio 0.65, min cement combination content 250kg/cu m, consistency S3.

4.3 Reinforced concrete strength class C35, min cement 300kg/cu m max w/c ratio 0.55

4.4 Weak Concrete grade C10, nlt 100kg OPC per cum, w/c ratio 0.65. Temporary propping required until cured.

4.5 Reinforcement: grade 500, type 2 reinforced bars to BS 4449, Welded fabric to BS 4449, Cover 40

4.6 Rebar, min lap 400, wall kicker nlt 100,

4.7 Starter bars, bent fabric OR H10 @ 200 base to wall.

5.0 EXCAVATIONS & TEMPORARY WORKS

5.1 Excavations thro' silts, sands and gravel: adequately support at all times the sides of pits and trenches.

5.2 Contractor shall arrange for the rapid dispersal of water shed onto the earthworks or completed formation or into excavations during construction.

Referenced Drawings

1 Existing Plan, Sections & Location Plan (revP0)

2 Proposed Plan & Sections (revP0)

3 Proposed Outfall Detail (revP0)

4. Highways Agency - Post and Rail Fence detail (H15)

Safety, health and environmental information

In addition to the hazards/risks normally associated with the types of work detailed on this drawing, note the following significant residual risks

Construction
- Buried services
- Unforeseen ground conditions
- Sewer flooding

Maintenance/cleaning/operation
- cleaning trash screen
- changing sponge
- manual handling

Decommissioning/demolition
- removal of sponge cage

Construction

AMENDMENTS/ ISSUES

ISSUE	REASON/ CHANGE	DATE	BY	Ch'k	Apr'd
4	As built changes & silt screen details added	Sept 13	TC		
3	Fence & Erosion Control detail added	Sept 13	TC	tjb	tjb
2	Clay backfill note added	Sept 13	TC	tjb	tjb
1	rc detail added	Aug 13	TC	tjb	tjb
P0	CDM and Pre-construction Planning	June 13	TC	tjb	tjb

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Scot Lane
Smart Sponge, Proposed Plan & Sections

FILE NAME; dwg-tjba-WPL-WUDPP-scot lane-240613

SCALE: 1 : 50 @ A1

Appendix VII – Wigan Fact Sheet

Wigan Fact Sheet	
Town Population (2011 Census)	97000
Growth Since 1900	37000
Principle River	The Douglas
Catchment Area	$220km^2$
Principle Industries (Historically)	Coal Mining
	Iron and Steel Works
	Textiles
Expansion	25,000 people in 1800 to over 100,000 people in 1870
Decline	1930's onwards
Historical Water Abstraction	Known as a 'Spa' town in the 1700's significant pollution from growning industry prevented this. Anecdotal evidence indicates significant abstraction was used in the Textile industry, particularly in dying and finishing.
Reference	Fletcher, Mike, The Making of Wigan (Wharncliffe Books, 2005)

Appendix VIII – Horwich WWTW Details

Horwich WWTW Fact Sheet	
Location	SD 6221 1099
Treatment Stages Utilised	Screens, Detritor, Primary Settlement Tank, Ferric dosing plant, Activated Sludge Process Plant, Final Settlement Tank, Percolating filters.
Average Daily Treatment Volume (ML/d)	14
Maximum Flow to Works (ML/D)	44.5
Daily Flow to Full Treatment Capacity(ML/D)	33
Storm/ Detention Tank Volume (m^3)	422
Population Served	31,000