HYDROLOGICAL AND SEDIMENTOLOGICAL (DIS)CONNECTIVITY IN AN URBAN SUDS SYSTEM BASED ON HIGH RESOLUTION, MULTI-EVENT MONITORING.

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Abstract

Sustainable Urban Drainage Systems (SUDS) are an integral part of the design in new housing developments. They are designed to manage surface water flooding and provide temporary storage for urban water runoff as well as allowing for the improvement of water quality through the natural operation of physical, chemical, and biological processes and through the trapping of sediments that often carry undesirable contaminants at above background concentrations.

The study site was based in the Upton housing development, on the western side of Northampton, UK, which was designed in the 1990s based on key principles to promote sustainable urbanism. The SUDS formed an integral part of the development and comprises of a number of swales that discharge into a series of ponds linked together with overspill-weirs. The ponds also receive inputs from a car park, local roads and a major road on the northern edge of the new housing development. It was one of the first developments in the UK to provide a "roof to river" surface water management strategy, with the SUDS designed to provide a "treatment train" before surface run off eventually discharges into the River Nene.

The efficiency of SUDS has been reported in the literature in terms of sediment and contaminant retention, but such studies do not always address the long-term source/ pathway/ sink receptor relationships. Little is known about the effects of how rainfall driven transient events (in terms of water levels) affect the performance of such systems. The aim of this research was to assess the hydrological and sedimentological (dis)connectivity within the system based on high resolution, multi-event monitoring. Rainfall was measured at 5-minute intervals local to the SUDS and pressure transducers were also installed in order to capture 5-minute water level data in up to 3 locations simultaneously. Data were collected over a four-year period from 2014 to 2018. The high-resolution water level data were used to produce hydrographs and the water level rise and fall dynamics were analysed for a series of individual events to illustrate performance over a range of rainfall intensity / durations. The impact of rainfall on water levels within the system varied over the study period and

highlighted the complex response of SUDS to rainfall input. It also highlighted the lack of connectivity within the system unless significant rainfall (storm) events occurred.

Time-integrated sediment samplers allowed the collection of sediment throughout the study period at various points in the SUDS. The sediments were analysed for heavy metals (Cd, Cr, Cu, Ni, Pb and Zn), gamma emitting radionuclides and mineral magnetic parameters (to establish if these could be used as a rapid inexpensive alternative to heavy metal analysis). A variety of statistical methods were used to establish the similarity between the sediments trapped in the samplers in terms of their contaminant concentration and radionuclide activities as well as aiding the identification of potential sediment sources. There was little evidence to suggest that environmental magnetism could be used as a surrogate for heavy metal concentrations in this particular SUDS.

The results from both the hydrological and sedimentological data suggested that little connectivity exists between all of the SUDS components and instead of acting as a "treatment train" most of the sediment is deposited in only one of the ponds, potentially creating a "sink" for pollutants. Sedimentation within this pond was also complex with high rates of accumulation at points close to drain inflows suggesting that there was insufficient energy most of the time to suspend and redistribute the sediment across the bed of the pond. A series of summary models were created based on 3 different rainfall scenarios and demonstrated that equal consideration should be given to both the hydrological and sedimentological response for such a system at the initial design stages. In addition, the siting of SUDS should consider the nature of the surrounding landscape and land use and ascertain whether potential local inputs could further contribute to the sediment and pollutant loads at different points in the system.

The metal concentrations of the sediment within this system were found to significantly exceed background levels. Rates of accumulation and the potential for exceedance of sediment quality guideline levels for soils need to be further investigated as well as the potential for pollutant linkages.

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For my Dad.

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Chapter 1: Introduction

The global population is expected to grow by around 2.9 billion in the next 30 or so years and it is anticipated that, due to issues such as economy and climate change, 68% of people will be living in cities and large towns by 2050 (UN, 2018). It is estimated that more than half the global population live in urban areas and that in the region of 1.5 million people are added to the global urban population every week (UN, 2014). The increase in population and the need for housing on a global scale has led to the expansion of towns and cities.

Development of urban housing is required to accommodate this increase in population and consequently will have an impact on the structure and the function of natural systems, in particular the hydrological cycle (Hall, 1984; Charlesworth *et al.,* 2003a). Increased peak flows, absence of permeable surfaces, the presence of straight channels (pipes, culverts) diverting excess water out of the area and the consequential decrease in lag time of the water reaching surface water courses, all account for a decrease in water quality and associated contaminant fluxes (Charlesworth *et al.,* 2003a). Such impacts are expected to lead to an overall reduction in hydrologic amenity value and an increase in flood risk. The quality of both the surface water and seepage water in urban areas are influenced by the pollutants that collect on impervious surfaces and are then transported by urban storm water run-off. Heavy metals (e.g., Pb, Zn, Cu, Cd) along with hydrocarbons are just some of the urban pollutants which are transported through the urban stormwater networks, both as dissolved phase and particle associated pollutants (Horowitz, 1991; LeFevre *et al.,* 2015). The behaviour of washed off metals from surfaces, such as concrete and tarmac roads, is a complex process which is influenced by the nature and volume of local road deposited sediments (RDS), storm water intensities, duration of dry weather periods, duration of storm events and the physical form of the draining surfaces (Xanthopoulos and Hahn, 1990). The concentration of the dissolved phase of heavy metals is generally low (although dependent on water chemistry) compared to the particulate fraction (Horowitz, 1991). Heavy metals coming directly from traffic activities are mainly found as particulate matter in urban runoff (Sansalone and Buchberger, 1997; Göbel *et al.*, 2007).

The need for management of urban surface water flooding was highlighted as a priority after the 2007 UK summer floods. It was estimated that at least 40% of urban flooding in the UK was a result of the failure of current urban drainage systems (Ellis and Viavattene, 2013) and the management and drainage of storm water continues to present a serious challenge (Davis and Naumann, 2017). Recommendations highlighted in the Pitt review (2008) along with the Flood Risk Regulations (2009) (transposed from the European Flood Directive 2007/60/EC) and Flood and Water Management Act 2010 emphasised the need for Stormwater Management Plans (SWMPs), for flood control, and the need for an integrated approach to urban surface water flood management (Ellis and Viavattene, 2013). The Governments planning and development processes incorporated Sustainable Urban Drainage systems (SUDS) at a strategic level within the National Planning Policy Framework (NPPF) (Gov.uk, 2012) (which replaced Planning Policy Statement 25 (Gov.uk, 2009) in 2012). The NPPF states that priority should be given to sustainable drainage and managing flood risk and that local planning authorities need to address such issues in Local Plans which should be supported by Strategic Flood Risk Assessments (Susdrain, n.d.).

SUDS are designed to mimic the behaviour of natural drainage and alleviate the problems highlighted by decreasing flow rates to watercourses and improving water quality (Charlesworth *et al.,* 2003a). SUDS use a range of techniques which allow the control of velocity and the removal of pollutants. Such techniques include swales, retention ponds and wetlands providing storage, both temporary and permanent, to storm runoff and allow for the improvement of water quality through the operation of physical, chemical, and biological processes. As a result, SUDS inevitably accumulate sediment and associated contaminants over a period of time (Heal and Drain, 2003).

Sediment accumulation within such systems is a concern, not only due to the reduction in storage volume over time (eventually decreasing performance with respect to reduced residence time and decreasing trap efficiency) but also with regard to the accumulation of potential contaminants within the systems. It is likely that future management of SUDS will require sediment removal in order to ensure maximum performance and protection of water quality within and downstream of the system (Heal and Drain, 2003).

Research has highlighted the retention of total suspended solids in various single runoff events in different vegetated SUDS as well as looking at retention within ponds and wetlands (Heal, 2000; Backström, 2002; Birch *et al*., 2004; Hossain *et al*., 2005; Deletic and Fletcher, 2006; Woods-Ballard *et al*., 2007; Allen *et al*., 2017a; Allen *et al*., 2018). Most research is based on single simulated or observed events and multiple, long term event analysis has not been undertaken in any detail. Allen *et al.* (2017a) assessed treatment efficiency by monitoring fine sediment transport (<2 mm) over a 12-month period using a novel rare earth oxide tracing methodology. It was noted that, as suspected, there is resuspension and redeposition which can continue over an extended period (in this case 52 weeks) and that a linear wetland has the capacity to perform much better than the swales with regard to fine sediment detention. Further modelling within the field is required to be able to ascertain the complex fine sediment transport processes but initial data has indicated that particle size of deposited sediment decreases through the network (Allen *et al.,* 2017b). The UK current design guidance for SUDS (Woods-Ballard *et al*., 2015) is based on information which is generally complied from single event monitoring and analysis which is assumed to represent long term removal efficiencies (Allen *et al.,* 2018).

1.1 Research Aim and Objectives

Aim:

To investigate the potential sources, transport, and deposition of sediment within a SUDS wetland system and identify connectivity within and between wetlands based on multiple event analysis and long-term monitoring.

Objectives:

1. To review long term data with regard to water levels within the SUDS and assess the impact of rainfall events on the hydrology and connectivity of the system.
2. To investigate the potential sources, and connectivity of sediment and sediment-associated contaminants between sampling sites and establish whether environmental magnetic measurements could be used as a cheaper alternative to heavy metal analysis for contamination assessment.

1.2 Format of the Thesis

Chapter 1

Introduction to the research, aims and objectives, format of the project.

Chapter 2

Literature review: providing context with regard to previous research in order to draw comparisons for discussion and critical review.

Chapter 3

Study Site: An overview and background of the SUDS at Upton and justification for site selection and for location of instrumentation and sampling strategy. Methods: Field, laboratory, and statistical methods statistical methods.

Chapter 4

Results and Discussion (**Objective 1**)- an overview of the hydrology of the system will be presented focusing on identified rainfall events. Reactions within the system are identified and temporal variations discussed.

Chapter 5

Results and Discussion (**Objective 2**)- analysis of the physio-chemical data obtained from the tube samplers within the SUDS. Similarities between sites will be explored as well as identifying influencing factors within the samples which could be used to indicate potential sources of sediment.

Chapter 6

Conclusion, summary, and future research.

Chapter 2: Literature Review

The following chapter comprises a review of the literature with regard to sediment transport and deposition within the urban environment, sustainable urban drainage systems (SUDS), and the geochemistry of such sediments in the context of this research, outlined in the first aim (see section 1.1). After introducing some of the key aspects of the nature and origins of sediments in the urban environment, this review will look at the input and behaviour of anthropogenic pollutants, in particular heavy metals, and the influence of physio-chemical properties, e.g., particle size. The role of SUDS and existing literature and research on their effectiveness and current knowledge will also be presented. Finally, the potential use of magnetic signatures as a surrogate for heavy metal analysis and the additional use of gamma emitting radionuclides to help determine sediment sources is reviewed and discussed.

2.1 Introduction

Sediments that accumulate within any environment are primarily a reflection of the sediment sources and processes of detachment, transport, and deposition (connectivity), and chemical processes occurring within both the water column and the deposited sediment (Perry and Taylor, 2007). Increasingly important components of these sediments are inputs of anthropogenic material and notable contaminants associated with human impacts including metals, inorganic elements, nutrients, organic compounds, and radionuclides (Perry and Taylor, 2007). These contaminants within the sediment are predominantly in the form of particulates but can also exist in dissolved or in gaseous forms and sources can be both point and diffuse. As sediment is vital to contaminant transportation and accumulation, the sources and delivery mechanisms of contaminants should be considered in an urban environment due to a high level of connectivity within fluvial systems (Horowitz, 1991).

2.2 The Sediment Cascade Approach and Urban Sediments

The categorisation of sediments within urban river basins into two main types was detailed by Taylor (2007) as those primarily acted upon by sub-aerial processes (road deposited sediments) and those deposited within, and transported through, aquatic systems. As can be seen in Figure 2.1 these two primary sources can be linked through the development of an urban sediment cascade (Taylor 2007).

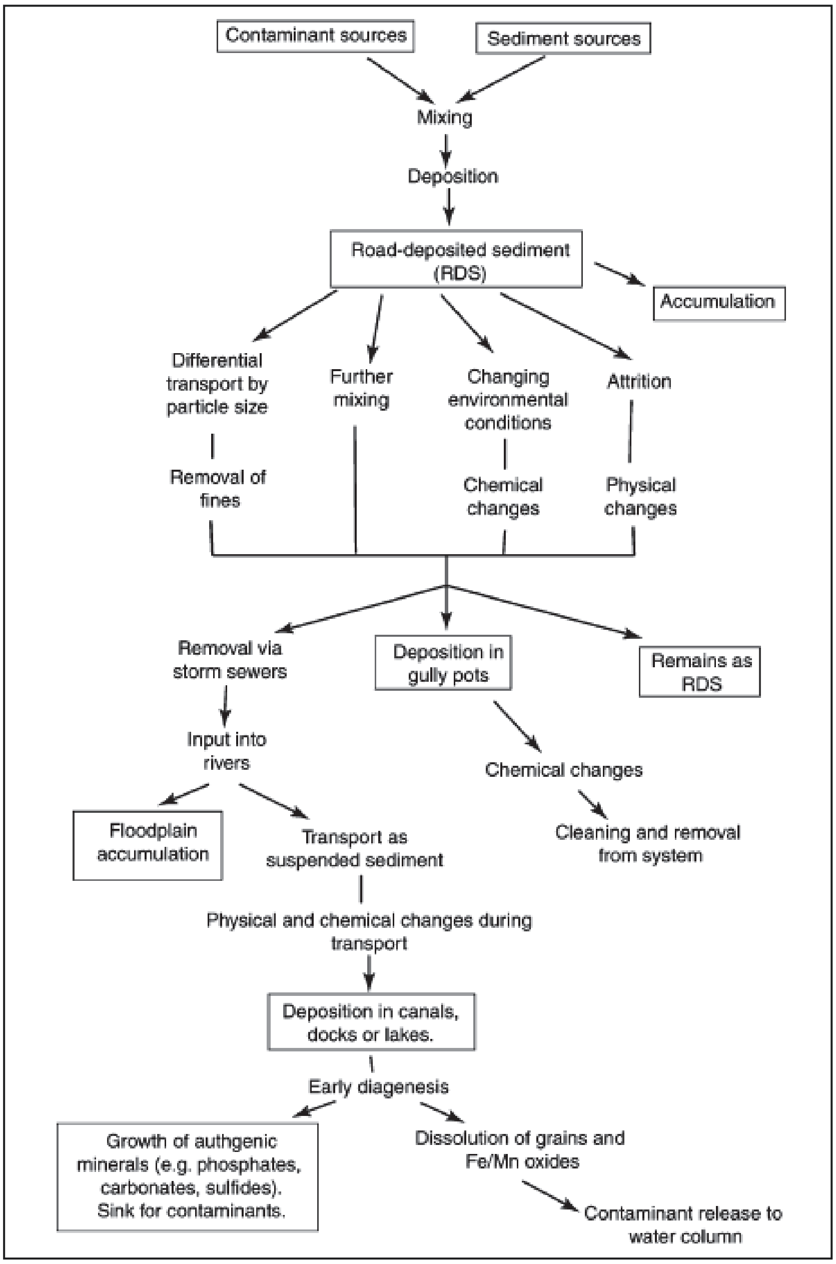

Figure 2.1: The Sediment Cascade (Taylor 2007).

The "sediment cascade" conceptual model approach also assesses the effectiveness and nature of sediment stores and, in turn, reflects the routes and distances of sediment transport within the landscape of the catchment (Fryirs *et al.,* 2007). If the landscape is considered compartmentalised, with all compartments contributing to the sediment cascade, then connectivity between these compartments could be assumed. Connectivity in this instance can describe (rather than define) the transfer of energy or matter between the system as a whole or between just two of the fluvial compartments such as the riverbed and water column (Chorley and Kennedy, 1971). Often there is discontinuity within this transfer process with buffers (preventing sediment from entering the channel network), barriers (disrupting sediment moving along the channel) and blankets (features which smother other landforms) leading to dis-connectivity between linkages within the catchment (Fryirs *et al*., 2007). The presence of buffers, barriers, and blankets, should be considered significant when validating an urban sediment cascade (Fryirs *et al.,* 2007). While the sediment cascade model attempts to demonstrate the potential buffers, barriers, and boosters to sediment flux, this is the first known study of urban fluvial geomorphology to explicitly explore the value of the connectivity concept in an urban context.

A recognition that sediment is a problem in urban environments, having direct and indirect effects on urban streams and their ecology, has been a much-debated issue within the framework of urban development between engineering and socio-economic growth and the acknowledgement that such sediments have a lasting impact in an environmental context (Table 2.1).

Table 2.1: Some of the most notable issues with regard to sediment in urban environments (adapted from Guy, 1970).

Issues	**Potential consequences**
Public health	the sorption of bacteria, chemicals radionuclides and the consequential fate of the sediments during their transportation either to new locations or public water supply
Gully erosion and associated deposition	the increase in sediment yield caused by construction practices
Reduction in infiltration	caused by compaction and changes in land surface thereby increasing runoff and sediment flow
Increase in the deposition of coarse sediments	potential increase from construction materials causing reduction in flow, impediment of channels
Risk of flood and an increase in floodwater damage	the transportation and disturbance of excess sediments
Aesthetic damage to varying water bodies	fine sediments in suspension as well reduction in amenity/ recreation value
Increase in costs for wastewater treatment	whether removal of excess sediment or removal of excess nutrients
Ecological damage	change in species composition and reduction in diversity
Maintenance costs for publicly used area	the increase in charges to the householder in order to maintain non adopted streets

The overall impact of sediment within the urban environment may not always be obvious but it is evident that the issues and problems it causes, visual or not, will likely cause long term ecological damage (Guy, 1970)

The term “urban sediments” has been used within the literature to reflect different concepts and, most notably, road deposited sediments (RDS) (Taylor and Owens, 2009). RDS is used to refer to the accumulation of all particulate matter on street surfaces (Sutherland 2003; Taylor 2007; Taylor and Owens, 2009) and not just road dust which implies a composition of very fine particles (<10 µm) although the terms have been used interchangeably in some of the published literature. The term RDS will be adopted within this chapter to reflect any sediments within the urban environment, incorporating urban soils, which are an important source of both airborne particulates and sediments (Taylor and Owens, 2009). Anthropogenic (e.g., vehicle exhaust emissions, vehicle tyre, body wear, brake material, building and construction material, road salt, road paint), pedestrian debris (Taylor and Owens, 2009; Sutherland and Tolosa, 2000) and natural sources (e.g., soil material, plant and leaf litter, atmospheric deposition of particles, animal material) will all be considered to be part of RDS and have the potential to reach urban fluvial systems.

2.3 Heavy Metals in the Environment

Heavy metals are widely found within the urban environment and tend to arise from many anthropogenic activities entering the environment via both point and diffuse sources (Sutherland *et al.*, 2004). Emissions of heavy metals to the environment occurs from a wide range of processes. Pathways include to the air (during combustion and processing), to surface waters (through runoff and releases from storage and transport) and to the soil. Atmospheric emissions have been the greatest concern, in terms of human health (accounting for about 7 million deaths per year (WHO,2018)), because of the quantities involved and the widespread dispersion (transboundary pollution). In addition to industrial and vehicular emissions, inappropriate waste treatment/ management techniques, with regard to municipal and industrial wastes, have been significant contributors to air, soil and water pollution since the middle of the 20th Century.

The variation in the concentration of metals being transported in the natural environment, is mainly dependent on geographical location and the proximity to industry and / or major roads. Due to their conservative nature these metals accumulate in the surface environment, subsequently contributing to air pollution through resuspension and / or eventually entering drainage systems (Duzgoren-Aydin *et al.*, 2006) causing the enrichment of heavy metals in receiving freshwater systems (Wilber and Hunter, 1979).

At the end of the 20th Century the emission of heavy metals started to decrease in developed countries and falling by over 50% between 1999 and 2000 in the UK (Järup, 2003) and by 2017, emissions for Cadmium (Cd), Mercury (Hg) and Lead (Pb) in EEA member countries, had declined by approximately 35%, 30% and 90% respectively, since levels observed in 1990 (EEA, 2019). Sector contribution of emission of Cd, Hg and Pb is summarised in Table 2.2.

Table 2.2: Sector split (%) of emissions of selected heavy metals (EEA, 2019).

Sector	Cd	Hg	Pb
Non road transport	0.3	0.4	1
Other	0.2	0.1	0.4
Industrial processes and product use	33.2	24.9	38.2
Waste	2.4	6	1
Road Transport	3.4	2.9	19.6
Energy Use in Industry	24.2	16	18.9
Commercial, institutional and households	21.3	12	13.1
Energy production and distribution	13.3	37.4	7.8
Agriculture	1.7	0.3	0
Total	100	100	100

However, since the mid 2000's severe heavy metal pollution has arisen due to the lack of facilities and treatment for e-waste, particularly, though not exclusively, in developing countries (Ozaki *et al.*, 2019). Waste Electronic and Electrical Equipment (WEEE) contains many rare elements of the periodic table, of which, the environmental behaviour and impact on human health is largely unknown. Increased environmental concentrations of elements such as Indium (In) and Gallium (Ga) are enhancing the risk of environmental exposure in the urban environment. Mounting evidence is indicating that these elements can present substantial toxicity and In, for example, has been linked to lung disorders and it is recognised as potentially fatal if In dust is inhaled (White and Shine, 2016). Much is unknown about the natural and anthropogenic cycling with regard to In and Ga. However, there is evidence that environmental concentration of In is changing as a result of anthropogenic activity and anthropogenic fluxes already appear to be exceeding natural fluxes (White and Hemond, 2012).

One of the growing concerns is the potential adverse health effects of exposure to urban pollutants (Charlesworth *et al.*, 2010; Duzgoren-Aydin *et al.*, 2006) as well as the underlying environmental effects of increasing concentrations of metals accumulating within sediments (Heal, 1999) with the most commonly reported urban pollutants within the literature being Cd, Cr, Cu, Ni, Pb and Zn (e.g. Harrison *et al.*, 1981; Cullbard *et al.*, 1988; Kim *et al.*, 1998; Heal, 1999; Wei and Yang, 2009; Sutherland *et al*., 2012; Zhao and Li, 2013; Tedoldi *et al.*, 2016; Venvik and Boogaard, 2020). Indeed, the accumulation of heavy metals within urban drainage structures can have important implications for the management of such drainage systems as they in turn affect storm water quality (Yuan *et al.*, 2001).

Reducing heavy metal inputs to soil is one of the strategic aims of soil protection policies within the EU (EC, 2002) and the UK (Defra, 2009). Baseline pollutant levels for soil and herbage in the UK were developed in 2007 and values for Cd, Cr, Cu, Ni, Pb and Zn are reported in Table 2.3.

Table 2.3: Range of Heavy Metals in Soils reported in UK Soil and Herbage Pollutant Survey mg kg^{-1} (adapted from Environment Agency., 2007).

	Rural	**Urban**	**Industrial**
Cadmium	0.39	11	18.1
Chromium	34.4	34.3	41.1
Copper	20.6	42.5	59.9
Lead	52.5	110	145
Nickel	21.1	28.5	37.1
Zinc	82.1	121	211

UK Soil Guideline Values (SGVs) and associated framework documents can provide evidence-based assessment of risks to human health from heavy metals. They were developed as a non-statutory technical guidance in support of the statutory regimes which address contaminated land, Part 2A of the EPA 1990 (Cole and Jeffries, 2009). While SGVs do not take into account other non soil-based sources of contamination, such as contamination in groundwater, surface water and drinking waters, they are guidelines and represent trigger values which can act as indicators that soil concentrations above a certain level may pose a “possibility of significant harm” to human health (Defra 2009). The

UK SGVs for Cd, Cr, Cu, Pb, Ni, and Zn are reported in Table 2.4 along with the Target and Intervention values used in the Netherlands. These values along with those presented in Table 2.3 can provide an indicator of relative heavy metal contamination in soil.

Table 2.4: Heavy Metal Guidelines in Soil- UK and Netherlands mg kg^{-1} (Adapted from Esdat, 2000; CL:AIRE, n.d.).

Metal	**UK Soil Guideline Values**				**Netherlands Target and Intervention Values**		
	Residential with plant uptake/ home grown produce	Residential without home grown produce	Allotment	Commercial	National Background Concentration	Target Value	Intervention Value
Cadmium	22	150	3.9	410	0.8	0.8	12
Chromium	130	200	-	5000	100	100	380
Copper	-	-	-	-	36	36	190
Lead	200	310	80	2300	85	85	530
Nickel	130	230	-	1300	35	35	210
Zinc	-	-	-		140	140	720

2.3.1 Road Deposited Sediment

Sediments on road surfaces and associated areas constitute a major contribution to urban and suburban drainage areas (Sutherland and Tolosa, 2000). The metal characteristics, and their availability in both the urban environment and these drainages systems, could constitute a major controlling factor in their transport downstream and play a critical role in the contamination and degradation of receiving waters. Roadside sediments and their associated contaminants are generally available for mobilisation and transportation to subsurface drainage systems by rainwater (Sutherland and Tolosa, 2000). The link between road run off and the effect of inorganic metal pollution on the benthic community in receiving water bodies has been well documented (Maltby *et al.*, 1995; Pitt *et al.*, 1995). Studies have shown declining macroinvertebrate assemblages down stream of urban runoff discharge points with copper, zinc and nickel influencing the diversity of the populations and demonstrating sublethal effects such as deleterious changes in reproductive rates, growth rates and enzyme activity (Maltby *et al.*, 1995; Beasley and Kneale, 2002).

The elevated concentration of metals in RDS tends to be reflective of vehicle emission, disintegration of vehicle brakes and tyres as well as atmospheric deposition, road surface wear and residential solid fuel heating, in certain countries (Zhao and Li, 2013). Some of the sources of heavy metals within RDS are noted in Table 2.5.

Table 2.5: Sources of Heavy Metals within RDS.

Sources	**Elements present**
Exhaust Emission	Ni, Cu, Zn, Pb (Castanheiro *et al.,* 2016)
Abrasion of Tyres	Cu, Zn, Cd (Castanheiro *et al.,* 2016)
Brake Pads	Sb, Cu (Castanheiro *et al.,* 2016)
Corrosion	V, Fe, Ni, Cu, Zn, Cd, Ce (Castanheiro *et al.,* 2016; Ward 1990)
Lubricating Oils	Cd, Cu, V, Zn, Mo (Ward 1990; Castanheiro *et al.,* 2016)
Fuel Additives	V, Zn, Cd, Pb, (Sansalone and Bucherger, 1997)

Pb is generally associated with historic uses (e.g., paint pigments, leaded fuel) although high concentrations of Pb in soils could possibly indicate remobilisation of Pb which has previously be stored in soils and transported to urban surfaces by erosion processes (Sutherland and Tolosa, 2000). Levels of Pb are still found to enrich sediments in receiving water courses, adding to the metal burdens of some streams indicating, that despite its reduction in use, it is still an active contributor to road surfaces and from surrounding soil erosion that was once stored during the leaded fuel era (Sutherland and Tolosa, 2000). Ondov *et al.* (1982) suggest that Br/Pb ratios can be used to determine “fresh” and historic particulate matter from vehicle exhaust emissions with “fresh” matter having values ranging from 0.39 to 0.47 and decreasing to 0.1 for “aged” particles, having lost Br to the gas or soluble phase over a period of time. EU Member states phased out the use of leaded petrol which was regulated by the Directive on the Quality of Petrol and Diesel Fuels (98/70/EC) and a ban on leaded fuel was implemented 1st January 2000. This said, the road transport sector, in Europe, remains an important source of Pb contributing approximately around 20% of the total Pb emissions, arising from engine lubricants and parts, tyre and brake wear (EEA, 2019).

It is now thought that RDS is a significant carrier for potentially toxic elements such as metals but the transport mechanisms via urban run-off still needs to be explored in detail (Section 2.6 examines previous studies specifically related to RDS and SUDS). Table 2.6 compares mean heavy metal concentrations in RDS in varying worldwide locations.

Table 2.6: Heavy Metal concentrations in RDS in cities worldwide (adapted from Charlesworth et al., 2003b; Li et al., 2015).

City	Cd (mg kg^{-1})	Cu (mg kg^{-1})	Pb (mg kg^{-1})	Zn (mg kg^{-1})	Reference
Newcastle upon Tyne	1.0	132	992	421	Okorie *et al.*, 2012
Birmingham, UK	1.62	466.9	48.0	534.0	Charlesworth *et al.*, 2003b
London, UK	6.5	197	3030	1174	Fergusson and Ryan, 1984
London, UK	6250	61-323	413-2241	ND	Leharne *et al.*, 1992
Barcelona	3	1332	48.0	534.0	Amato *et al.* 2011
Buenos Aires, Argentina	NA	190	208	751	Fujiwara *et al.*, 2001
Banja Luka, Bosnia Herzegovina	1.39	77.7	608	272	Škrbić *et al.*, 2012
Istanbul, Turkey	3.9	1039	222	229	Sezgin *et al.*, 2004

Early studies looking at metal accumulation in catchments tend to have focussed on specific land use areas or within large cities (Zhao and Li, 2013) as demonstrated in Table 2.6. However, there is a need to look at varying land uses coupled with population density, the location of impervious surfaces, traffic density, and sediment stores to allow for the spatial and temporal distribution of heavy metals to be quantified. This would allow potential risks to be further analysed in relation to the connectivity model described earlier as SUDS systems provide excellent examples of where connectivity in the fluvial system is being managed. RDS could be considered as a function of inputs, outputs, and storage changes along with all associated processes (Sutherland and Tolosa, 2000). The inputs themselves arise from extrinsic sources such as water transported sediments and dusts from soils, via both dry and wet deposition as well as from biological inputs from surrounding vegetation (Muschack, 1990; Sutherland and Tolosa, 2000; Herngren *et al.*, 2006). In addition, inputs would equally arise intrinsically from road surfaces (both wear on the actual surface and from vehicles and associated particulate emissions). These intrinsic and extrinsic inputs are seen as the dominant source of sediment accumulation on paved surfaces in urbanised areas. Outputs would arise from the re-suspension of RDS from high traffic speeds, from street sweeping operations and other physical movements such as aeolian processes and also from the transport of material to subsurface drainage. It is important to note that the RDS would be a direct reflection on a proportion of the inputs for a given time period. (Sutherland and Tolosa, 2000).

If we consider the input of RDS then the nature of the input is critical to our understanding of downstream pollution patterns (Sutherland and Tolosa, 2000) and would provide a level of prediction/ assessment with regard to pollution, although perhaps only when the bioavailable fraction is quantified (Sutherland and Tolosa, 2000). Therefore, understanding and measuring pollutant characteristics of impervious surfaces could be essential to estimate pollutant run off (Vaze and Chiew, 2002). It has been suggested that there are two major factors to be considered when looking at RDS; pollutant build up and pollutant wash off. The pollutant build-up is a factor of accumulation within the catchment surface during dry periods and the pollutant wash off is the removal of such RDS by precipitation and is a function of intensity and volume. While it is possible to draw the conclusion that surface pollutant load increases with the longevity of dry periods, the assumption that these pollution loads reset to zero after a rainfall event is contentious. Some studies have shown that storm events will only typically remove a small proportion of the surface pollutant load (Malmquist, 1978; Chiew *et al*., 1997). This has a significant effect when models are applied to ascertain potential pollutant loading as shown in Figure 2.2.

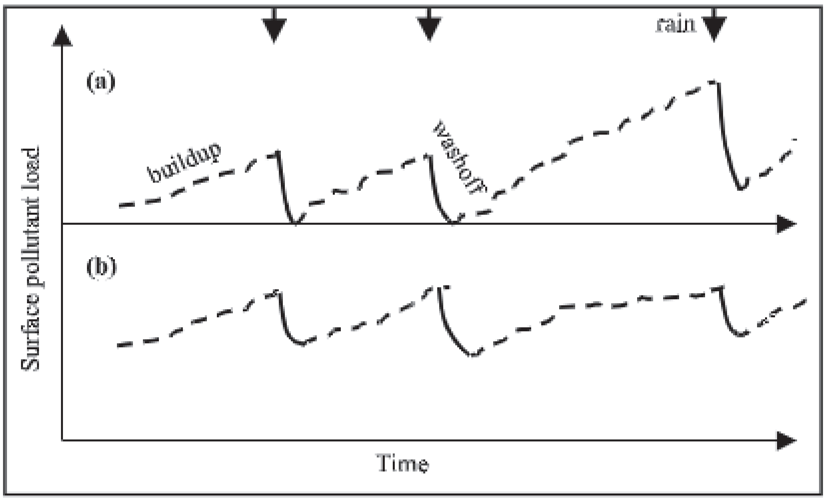

Figure 2.2: Hypothetical representations of surface pollutant load over time. (a) assuming that the pollution load resets to zero and b) an assumption that a proportion of the pollution loading remains between events. (after Vaze and Chiew., 2002).

2.3.2 Particle Size

It has been observed that metal concentrations (and loads) can increase in the sediment through an urban area mainly as a function of particle size (Horowitz, 1991). While the amounts of metals have been found to be greatest in fractions >125um, this is only due to the increased proportion of the `larger particle size in the sediment and not a function of concentration in relation to particle size (Wilber and Hunter, 1979). Although a number of factors, such as surface area and specific gravity, have substantial effects on trace element concentration, particle size is the most important (Horowitz, 1991) and a key factor in determining potential heavy metal transport with regard to RDS (Zhao and Li, 2013).

While the load of such sediments is crucial in the management of urban runoff, knowledge of the particle size distribution of the sediment is essential in order to assess the transport dynamics of such pollutants (Vermette *et al*., 1987). The input from varying different sources and consequential particle size has significant effects; smaller particles are more likely to remain in suspension and therefore be transported over greater distances (Deletic *et al*., 1997).

However, it is not only the load that needs to be considered but also the size distribution and pollutant transport dynamics including information pertaining to the magnitude of the sediment load, particle size distribution and particulate input (Herngren *et al*., 2006). The larger particles are relatively easy to remove, if that is the purpose, but when the finer particles are contributing to the stormwater runoff, they have the ability to remain in suspension for a longer period of time and are transported a greater distance. Solid particles, which are greater that 100µm in diameter, are relatively easy to separate by settling but smaller particles tend to remain in suspension in run off and need much longer settling times. Particles of <50µm have been shown to represent around 75% of the total weight of solids in urban runoff (Andral *et al*., 1999). This fraction of particles has also been observed to settle at speeds of around 2.5-3.3 m hr^{-1} compared to those >100µm which will settle at velocities of 5.7-13.1 m hr^{-1} (Andral *et al.,* 1999). In addition, due to the cohesiveness of fine sediments (<63µm) they tend to resist the process of resuspension but at higher velocities are remobilised and then tend to remain in suspension for a long period of time and travel long distances (Horowitz, 1991).

The smaller particles are of greatest concern as they have a relatively high specific surface area which facilitates adsorption of pollutants (including heavy metals), highly magnetic particles produced by high temperature combustion and atmospheric fallout radionuclides such as excess ^{210}Pb ($^{210}Pb_{un}$), ^{137}Cs and ^{7}Be. and the adverse effects of such particles is more notable where anthropogenic sources of water pollutants are greater (Herngren *et al*., 2006).

2.3.3 Organic Matter and Heavy Metals

Positive linear relationships between organic matter concentration and heavy metal concentration / load have been observed (Wilber and Hunter 1979) and this organic fraction plays a vital role in the partitioning of metals into varying particle sizes (Hamilton *et al*., 1984). Higher organic content in fine particulates, as opposed to coarser sediments, has been noted and it is therefore possible that the crucial role played by these finer particles is further enhanced by the presence of high organic carbon concentrations (Herngren *et al*., 2006).

The pathways of heavy metals through the urban environment will be considered in this thesis and needs to consider not only the transport but also the deposition and storage of sediments (both long and short term) (Charlesworth and Lees, 1999).

2.4 Radionuclides in the Environment

A significant amount of background radiation within the environment can come from natural sources including long-lived radionuclides present within natural minerals contained in river sediments. Natural radionuclides can also come from cosmogenic sources that fall out of the atmosphere and accumulate on urban surfaces (Cooper *et al*., 2003). These nuclides are often rapidly and strongly sorbed to sediment (Walling, 2004) and are transported through the fluvial system largely in particulate form. Additionally, global fallout from atomic weapons testing and nuclear accidents have elevated the natural radioactivity and added new isotopes of common and rare earth elements, in most environments. Information on the concentration and distribution of naturally and artificially produced nuclides proves useful in the monitoring of environmental contamination and in assessing potential risks to human health hazards by such radioactivity (Suresh *et al.,* 2011). A summary of the sources and uses of radionuclides commonly used in environmental monitoring is presented in Table 2.7.

Table 2.7: Radionuclides, sources and uses (adapted from Foster et al., 2007)

Isotope	Half Life	Origin	Notes
$^{210}Pb_{un}$	22.26 yr	Atmospheric fallout	Atmospheric from ^{222}Rn (Radon Gas) ^{222}Rn is formed from the ^{238}U decay series. Transport to the ground occurs in rainfall and it has been used as a tracer and to date sediment sequences
^{137}Cs	30 yr	Fission: Weapons fallout and nuclear accidents	First occurrence, 1954 with peak in 1963; Chernobyl 1986, Fukishima 2008. Commonly used as a tracer as independent of lithology and soil type (Collins and Walling, 2004).
^{234}Th	24.1 day	Natural	^{238}U decay series Activity ratio of ^{234}Th/ ^{238}U has been used in aging drinking water samples (Waples *et al.*, 2015)
^{226}Ra	1600 yr	Natural	^{238}U decay series Application as a tracer for processes occurring in estuaries and salt marshes (Szymczak, 2012), groundwater discharges (Moore, 1996)
^{235}U	7.04 x 10^8yr	Natural	^{235}U decay series
^{228}Ac	6.14 hr	Natural	^{232}Th decay series
^{212}Pb	10.6 hr	Natural	^{232}Th decay series
^{40}K	1.28 x 10^9 yr	Natural	Primordial

Radionuclides can enter the aquatic environment through point and diffuse discharges. While suspended material can be deposited as sediment and immobilised over time, this material can also be re-suspended and transported over large distances, progressively becoming more dilute and eventually reaching the oceans (UNSCEAR, 2008). Sediment can accumulate within the urban drainage system and therefore it is assumed that radionuclides which have been deposited and sorbed onto particulates will also accumulate**.**

^{210}Pb, a daughter product of ^{222}Rn, attaches to the surface of aerosols and dust particles which then reach the soil and other surfaces by both wet and dry deposition. The final activity in a sediment sample is a balance between fallout rates, wash off and radioactive decay (Table 2.8). In consequence, the surface layers of road dusts would be expected to contain higher $^{210}Pb_{un}$ activities than that which is in equilibrium with ^{226} Ra. (The ^{210}Pb which is in equilibrium is called “excess”

$^{210}Pb_{ex}$). The $^{210}Pb_{ex}$ provides the basis of tracer applications (Mabit *et al.*, 2014) as seen in Figure 2.3 and also of $^{210}Pb_{un}$ dating of lake and floodplain sediment sequences (Appleby, 2001).

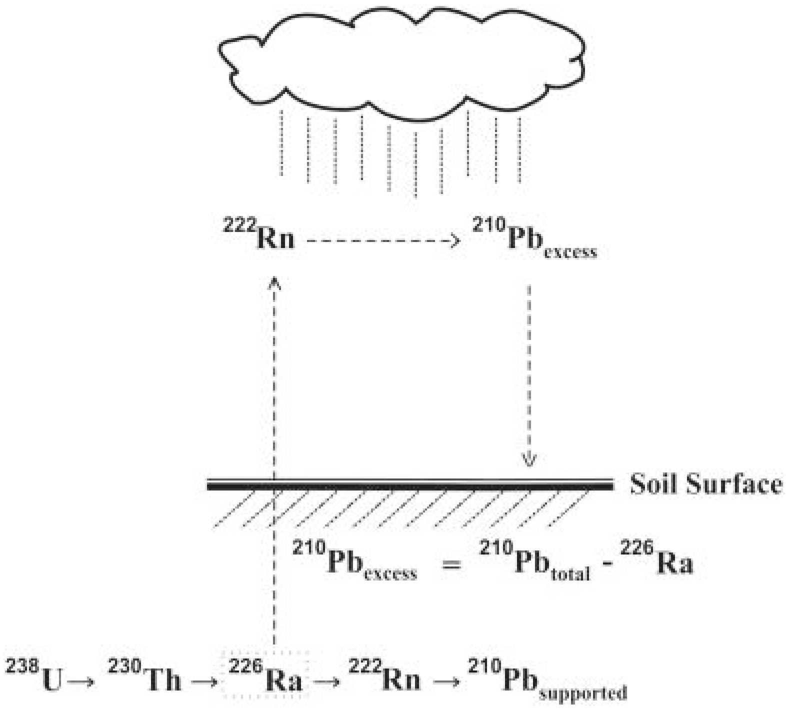

Figure 2.3: The Origin of geogenic $^{210}Pb_{un}$ and fallout $^{210}Pb_{ex}$ (Mabit et al., 2014).

Unsupported ^{210}Pb is likely to accumulate on surfaces during dry weather periods. It binds strongly with soil particles (both mineral and organic) and is chemically stable; therefore, an assumption is made that the major processes causing redistribution are mechanical processes such as wind or water (runoff) and its subsequent redistribution is controlled mainly by soil and sediment erosion (Mabit *et al.*, 2014).

2.5 Environmental Magnetism

Many studies which have looked at RDS and urban sediments have reported heavy metal concentrations as a way of indicating potential sources and sinks of pollution (e.g., Kim *et al.*, 1998; Göbel *et al.*, 2007; Wei and Yang, 2009; Charlesworth *et al.*, 2011; Allen *et al.*, 2015; LeFervre *et al.*, 2015). However, since Oldfield *et al.* (1985) demonstrated that it was possible to use simple, rapid and non-destructive magnetic measurements in the characterisation of sediments, there have been a number of studies using such measurements to identify heavy metal pollution (e.g. Thompson and Oldfield, 1986; Heller *et al*., 1998; Bityukova *et al*., 1999; Shilton *et al*., 2005; Bai, 2006; Karimi *et al*., 2011; Meena *et al*., 2011; Wang, 2013a, 2013b) Lu and. In addition, magnetic minerals have been used in sediment source fingerprinting (e.g., Walling, 2005; Foster *et al.*, 2007; Pulley, 2014, Biddulph, 2017).

Measurements such as magnetic susceptibility (χ_{LF}) have been used to map pollution (Wang, 2013a, b) and provide a cost-effective way to identify industrial and traffic related atmospheric particulate pollution (Chaparro *et al*., 2006; Gargiulo *et al*., 2016). Magnetic particles have been associated with atmospheric emissions as fossil fuels contain traces of iron which may melt to form magnetic spherules during a range of combustion processes (Hunt, 1986). Other studies have demonstrated positive correlations between anthropogenic magnetic enhancement and heavy metal pollution (Desenfant *et al*., 2004; Lu and Bai, 2006; Lu *et al.*, 2007; Zhang *et al*., 2012; Jaffar *et al*., 2017; Karimi *et al*., 2017) suggesting that magnetic particles can act as hosts (through incorporation into its crystalline structure or sorption onto the surface (Vassilev, 1992)) of heavy metals and other pollutants. Therefore, mineral magnetism is often deemed advantageous over other methods for sediment characterisation, as it involves relatively quick measurement, is non-destructive and the equipment required is inexpensive in contrast to detailed geochemical analyses which may require complex and expensive chemical analyses (Bityukova *et al*., 1999).

2.6 Sustainable Urban Drainage Systems

2.6.1 An Overview of SUDS

The term “Sustainable Urban Drainage systems” first appeared in the scientific literature (according to a systematic search in Google Scholar) in 1982 in an article entitled “River rehabilitation and management” (Braune and Hinsch, 1982) which focussed on the Braamfontein Spruit river flowing through Johannesburg and reviewed potential management systems to improve the natural habitat and surrounding infrastructure. Within the scientific literature the term sustainable urban drainage has appeared within articles 5820 times and within titles of articles 247 times with date ranges as shown in Table 2.8. There has clearly been an exponential increase in the use of the term in the decades from 1980 to 2020.

Table 2.8: The appearance of "Sustainable urban Drainage systems" in Google scholar searches.

Dates	Number of appearances within the title only	Number of appearances within the article
1980-1990	0	3
1991-2000	7	33
2001-2010	72	1090
2011-2020	150	4360

However, the concept of surface water management is not new and there is evidence that early civilisations, such as the Babylonians and Mesopotamians, had water drainage systems and similar techniques to those used in sustainable drainage systems. These were utilised as early in human history as the Early Bronze Age and engaged in such practices as rainwater harvesting, water storage in ponds and in the basements of houses and municipal buildings, and the control and conveyance of surface water (Charlesworth *et al.*, 2018).

Traditional (20th/21st century) urban drainage systems consist of a network of underground pipe systems that convey water away from built up areas. The high density of impermeable surfaces, between 60-99% (Donovan *et al.*, 1992), and existing design of drainage networks have contributed to the increase in peak flow in river systems downstream of urban areas. In addition, these areas have been demonstrated as one of the main pollutant sources of heavy metals within river systems (Brown and Peake, 2006). SUDS offer an alternative to conventional drainage systems where the singular objective is based around water quantity (flood) control. The use of SUDS embraces aspects of water management for the urban environment run off quality as well as visual amenity recreational value and ecological/ biodiversity benefits. Water quality has also become increasingly important in the design of such systems as a result of legislative and political tools such as The Water Framework Directive (2000/60/EC (2000), which seeks to prevent the deterioration in surface water quality and sets out objectives for the attainment of good ecological status for all watercourses. SUDS can be described as engineered solutions (Waite, 2010) or a range of techniques to support the management of water resources by reducing pollution loads and stemming the rapid flush of waters entering urban rivers after storm events. Increasing urbanisation, and the consequential increase in impermeable surfaces (Balmforth *et al.,* 2006) and loss of free draining land, results in large amounts of surface run-off being directed into local water courses or inadequate sewerage systems (CIRIA, 2008). In addition to the issues concerning flood risk, it is acknowledged that contaminant accumulation and abundance within urban areas can also result in high pollution loadings within aquatic systems (Wong *et al*., 2006; Duh *et al.*, 2008; Horowitz and Stephens 2008; Laidlaw and Filipeli, 2008).

2.6.2 SUDS Design

A sustainable urban drainage system is, by its nature, designed to manage and drain surface water (CIRIA, n.d.) and utilises such structures as pervious pavements, wetlands, swalesswales, and ponds (Lawrence *et al*., 1996). This use of multiple techniques and the natural catchment is often referred to as the SUDS management train (treatment train), and it can be used to change the flow and quality of the runoff in a series of stages (Figure 2.4) (Susdrain, n.d.(a))

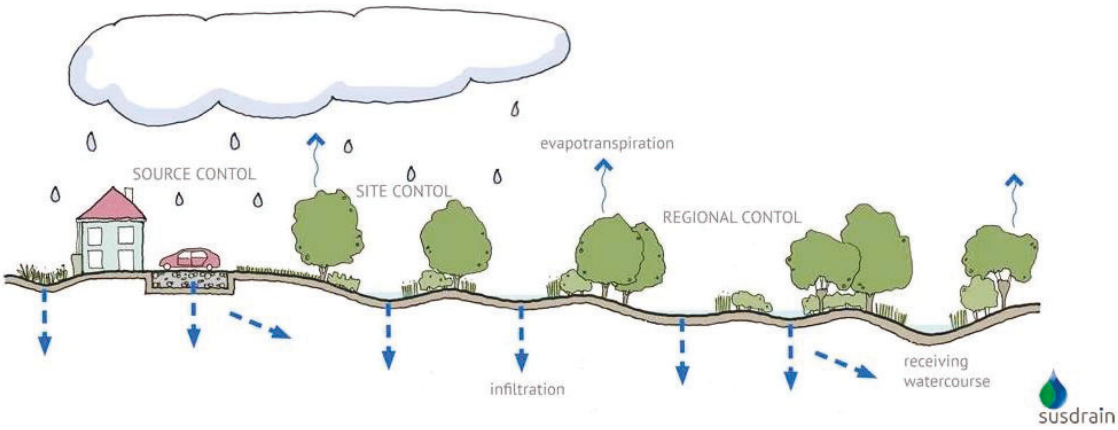

Figure 2.4: The SUDS management train (treatment train). (Susdrain, n.d. (a)).

The concept of the management (treatment) train is fundamental in the design of successful SUDS and the use of varying drainage and storage techniques is intended to reduce pollution, flow rates and volumes of surface water reaching major rivers (Woods-Ballard *et al*., 2007). Design criteria are presented within the SUDS manual (Woods-Ballard *et al*., 2007) and, while it provides a framework for the design of effective systems to protect both public health and safety, and the environment, it does state that it is impossible to design for all events and therefore there is potential for the design criteria to be exceeded in extreme floods. The design criteria focus on hydraulics, water quality amenity and ecology.

Hydraulic design methods include structures store runoff within the SUDS train which is essential in providing for the detention of flow, including the ability to attenuate flow for downstream flood protection as well as for water quality. Water storage can be an on-site system which is provided using both landscaped and structural features. The storage options can be divided into those that both attenuate and retain. Attenuation storage aims to reduce peak discharges from the site, e.g., swales, which are designed to drain at a rate which can be dictated by the outlet pipe but will not necessarily improve water quality leaving the site. Swales are shallow broad vegetated channels

(Figure 2.5) and are designed to reduce velocity and therefore allow infiltration and evaporation and consequently allow any suspended solids to be deposited (Wilson *et al.*, 2004; Kirkby. 2005).

Figure 2.5: Example of a vegetated swale, Upton, Northampton (Copeland-Phillips, 2021).

Retention storage areas are designed in order to contain a permanent pool of water (potentially in ponds and wetlands), and these are used to provide water quality treatment (Woods-Ballard, 2007). Retention ponds often comprise open areas of shallow water, accommodating rainfall and providing a temporary storage for excess water. Generally, the water levels increase during rainfall events but there is also a permanent body of water present and while similar to wetlands, they are generally designed as storage facilities (Susdrain, n.d.(b)).

Wetlands used within SUDS (Figure 2.6) are generally densely vegetated and provide opportunities for sedimentation and filtration of water runoff thus improving the water quality. Wetlands should generally be the last stage in the “treatment train” as without upstream treatment they can be subject to siltation when their function would rapidly deteriorate over time. Generally, wetlands are used to remove fine sediments and varying contaminants and nutrients, but they can also enhance biodiversity and amenity within a development (Susdrain, n.d.(c)).

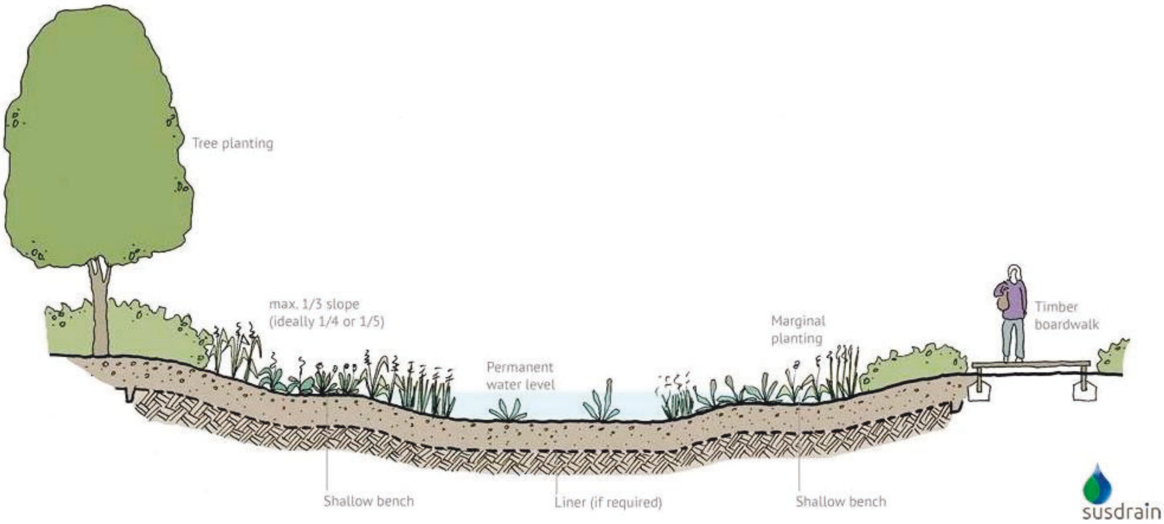

Figure 2.6: Example of a wetland within SUDS (Susdrain, n.d.(c)).

The structures within the SUDS treatment train disrupt the connectivity between source and sink in fluvial systems (Fryirs *et al*, 2007). Since most point source discharges are regulated in the UK, diffuse pollution has become a major focus and SUDS is identified as a central approach to deal with urban diffuse pollution (CIRIA, 2008). The need to enhance the capacity and flexibility of conventional sewer systems to adapt not only to climatic change but also to the increase in urbanisation means that SUDS can be promoted as a complementary or even alternative approach ensuring the long-term sustainability of such systems.

In 1997 the Environment Agency for England and Wales published a report (“Liaison with Local Planning Authorities”) (Environment Agency, 1997) which promoted the use of SUDS to tackle contaminated surface water and associated sediments. This arose from Agenda 21 (UN, 1992) which established a global plan for sustainable development. Further to this in 2008 the UK government laid out their “Future Water” strategy which encouraged the use of SUDS to mitigate the potential effects of rainwater and flooding on water courses and in 2009 the Flood Water Management Bill called for flood risk management. The resultant Flood and Water Management Act (2010) requires developers to utilise SUDS to control surface drainage and therefore transfer responsibility for flood risk from central to local governments

However, a recent review looking at the impacts of urbanisation and climate change on urban flooding and urban water quality, has identified that there is a lack of national research focussed on the impacts of climate change, flooding, and water quality in the UK (Miller and Hutchins, 2017). Figures often put forward for use in planning for flooding are often generic and that approaches for the development of Depth Duration and Frequency (DDF) curves and the estimation of flooding rarity are often static in their nature. There tends to be an assumption that rainfall and runoff models calibrated over historical periods, e.g., 20 years, remain valid for future use but it has been demonstrated that this is not the case should the mean annual rainfall be more than 15% drier or 20% wetter than the calibration period (Vaze *et al*., 2010).Also these models tend not to account for seasonal and regional variation (Prudhomme and Reynard, 2009) and current levels of adaptation are not yet sufficient to mitigate increases in flood risk with 2-4°C climate change projections. For towns like Northampton, winter rainfall events could become more frequent (reducing the present-day return period) with the biggest changes predicted between 2010-2040 (Sanderson, 2010). With uncertainty surrounding climate change and frequency of extreme rainfall events it is imperative that modelling of flood risk and the use of SUDS within urban environments is considered at a Local Authority level. Long term monitoring of SUDS would provide further insight into the performance of such systems under different rainfall regimes. The effect of data resolution can be important particularly in urban areas where there is a mix of both pervious and impervious surfaces resulting in variation in time periods with regard to run off often giving rise to a “flashy” system in terms of hydrological response. Fine temporal data has been used to capture rising limb transients in stream hydrology (Shuster *et al*., 2008) and therefore could help in the identification of connectivity and performance in SUDS.

2.6.3 SUDs and sediment management

As previously discussed, SUDs consist of multiple approaches to provide effective surface water drainage, ideally providing flood risk protection and pollution mitigation as an integrated intervention. In contrast to traditional drainage networks, they aim to store and divert surface water and decrease flow rates to watercourses as well as remove sediment. These methods are not generally used in isolation but instead are used to create a “treatment train”; a range of different stages designed to; reduce volumes of water entering local watercourses through source control (e.g. rainwater harvesting or evapotranspiration via green roofs); vegetated swales or trenches designed to remove sediment and associated contaminants; storage ponds designed as retention systems which aid in the delay of discharge (e.g. wetlands); soakaways and other infiltration systems which aim to mimic natural groundwater recharge. As such SUDs could be perceived as “jerky conveyor belts”, a term first coined by Ferguson (1981) when describing river systems in terms of sediment transport. Like a river system, SUDs have different zones where sediment can be deposited and kept in storage (within ponds, wetlands etc.) and potentially removed or eroded and further added to the “conveyor belt” in a spatial sequence of source, transfer, and accumulation. Much like a river system, the nature of this sequence is unlikely to be straightforward (Thompson *et al.*, 2016).

The likely efficiency of SUDs, regarding sediment deposition and storage, has been reported in the literature both pertaining to small scale laboratory studies and in field trials under single event conditions, often restricted to a very particular area. It has been reported that a vegetated swale, for example, can contribute up to 86% (Deletic, 2001) in suspended solids load reduction but such studies are limited and do not address the long-term source / pathway / sink relationships (Allen *et* *al*., 2015) which form this “jerky conveyor belt” and constitute an important part of both long-term performance and maintenance.

Sediments tend to accumulate within SUDs ponds and wetlands over periods of time (Heal *et al*., 2006) and it is also assumed that some of these sediments will accumulate within the swales and the gulley pots which constitute the primary drainage areas within the system. While the contaminant removal capabilities of SUDs are reported, the fate of contaminants is generally not (Heal, 1999). Table 2.9 shows a range of studies and reported metals within different compartments of SUDs.

Table 2.9: Metals reported in SUDs.

Geographical Region	Metals Reported	Compartment sampled	Reference
Scotland, UK	Cd, Cr, Cu, Ni, Pb, Zn*	Sediment	Heal (1999)
Scotland, UK	Cd, Cr, Cu, Fe, Ni, Pb, Zn	Sediment	Heal *et al*. (2006)
Netherlands	Cu, Pb, Zn***	Topsoil	Venvik and Boogaard (2020)
Paris, France	Cu Pb, Zn	Topsoils	Tedoldi *et al*. (2016)
Scotland, UK	Cd, Cu, Pb, Ni, Zn	Soil and Sediment	Napier *et al*. (2009)

**Metals reported as bioavailable, data obtained through EDTA Extraction*
***Metal concentration obtained via flame atomic absorption*
****XRF portable*

The fate and concentrations of heavy metals has been addressed in several studies which have concluded that elevated levels have been found within aquatic sediments and in infiltration systems (Lind & Karro, 1995; Mikkelsen *et al*., 1996; Heal, 1999). It is also noted that the swales, vegetated structures designed to reduce water velocity and allow infiltration, within the systems are good at the removal of total suspended solids (TSS) with efficiency of removal dependent on several factors (Wilson *et al.*, 2004). In addition, vegetated swales can be effective in the removal of heavy metals, through uptake by a variety of grass species, although this is highly dependent on the design of the vegetative system (Waite, 2010). Often highest contaminations of metals within SUDs are detected at, or near, the water arrival point (Napier *et al*., 2009; Jones and Davis, 2013;). A range of metal concentrations have been found within these studies and are reported in Table 2.10.

Table 2.10: Concentrations of Metals (mg kg^{-1}) found within SUDS from previous studies.

Reference	Cd	Cr	Cu	Fe	Ni	Pb	Zn
Rownay *et al.* (1986)	2.55 ± 1.03	-	25.7 ± 9.97	-	-	58.3 ± 32.9	114 ± 38.9
Striegl (1987)	4		250	1.94		1590	210
Mesure and Fish (1989)	4	-	17 ± 4 47 ± 16 18 ±5 17 ± 4	-	-	16	-
Yousef *et al.* (1990)	5 ± 6	19 ± 17	10 ± 14	-	10± 10	92 ± 193	37 ± 101
Färm (2002)	0.43 ± 0.37	25.7 ± 9.07	51.3 ± 24.4		38.7 ± 12.4	34 ± 9.85	189 ± 73.7
Mallin *et al.* (2002)	0.15 ± 0.15	1.72 ± 1.31	3.58 ± 6.00	0.03 ± 0.032	0.72 ± 0.76	3.35 ± 3.71	25.7 ± 45.2
Heal *et al.* (2006)	0.21 ± 0.54 0.22 ± 0.42 0.32 ± 0.39 0.39 ± 0.94	70.7 ± 65.8 78.2 ± 87.0 118 ± 110 76.7 ± 102	18.8 ±9.22 10.9 ± 15.3 16.3 ±6.42 17.4 ± 7.44	4.41 ± 1.1 4.74 ±1.68 3.87 ± 0.873 7.16 ±3.04	63.3 ± 48.4 68.4 ± 39.8 83.9 ± 61.4 63.6 ±57.5	26.3 ± 31.5 25.4 ± 19.6 18.2 ± 9.46 22.6 ±17.2	78.4 ± 72.9 110 ± 89.4 77 ± 24.8 93.1 ± 43.1
Marsalek *et al.* (2006)	1.2 ± 0.3	110 ± 25	63 ± 26	2.98 ± 0.21	32 ± 5	125 ± 50	319 ± 124
Napier *et al.* (2009)	0.15 - 0.4	-	21 - 109	-	32 - 48	25 - 60	127 - 388
Jones *et al.* (2007)	-	22.6 ± 2.6	21.9 ± 3.1	4.32 ± 1.18	22.4 ± 6.4		69.7 ± 9.5
Tedoldi *et al.* (2016)*	-	-	~50-~200	-		~20-~240	~200-~1100

**Values approximated from graphs as no values cited directly.*

The effectiveness of SUDs in the removal of Total Suspended Soilds (TSS) and contaminants is documented but in comparison with traditional urban drainage methods there are uncertainties surrounding the “sustainability” of these systems. One of these uncertainties is the long-term performance and management of SUDs with the necessary requirement for regular maintenance (Wood-Ballard *et al*., 2007). The accumulated sediments within SUDs should be removed regularly (not yet quantified although suggested as 25-30 years (Heal, 1999)) to prevent reduced performance of storage volume (lowering trap efficiency) and prior to the potential for accumulated contaminants to reach unacceptable levels (McKissock *et al.,* 2003; Heal *et al.,* 2006). Heal (1999) reported the accumulation of Cd, Cr, Cu, Ni, Pb and Zn (Table 2.12) in comparison with both Swedish EPA scores and that of the ICRCL (Inter Departmental Committee for the Redevelopment of Contaminated Land). The excavation of the sediments has the added complication of the classification of the waste material and the consequential nature and cost of disposal (Heal, 1999).

Table 2.11: Comparison of metal contamination in sediments with Swedish EPA score and ICRCL score (Source: Heal, 1999).

Site	Cd	Cr	Cu	Ni	Pb	Zn	Total
Wetland (operational 1997)							
Max concentration (mg kg^{-1})	0.085	1.6	7.05	20.0	5.8	26.6	
EPA Score*	1	1	1	2	2	1	**8**
ICRCL Score	2.8	0.27	5.42	27.9	1.16	8.85	**46.5**
Pond Industrial (operational							
1995)	0.045	1.7	13.7	24.6	13.7	7.1	
Max concentration (mg kg^{-1})	1	1	2	2	2	1	**9**
EPA Score*	1.50	0.28	10.5	35.1	2.74	2.37	**52.6**
ICRCL Score							
Pond Residential (operational							
1987)	0.155	2.9	7.55	20.8	133	33.4	
Max concentration (mg kg^{-1})	1	1	1	2	4	1	**10**
EPA Score*	5.17	0.48	5.81	29.7	26.6	11.2	**78.9**
ICRCL Score							

**The Swedish EPA classification scoring- 1 = "background", 2 = "low", 3 = "moderate-high", 4 = "high", 5 = "very high".*

Further studies (Heal *et al*., 2006) characterised sediment accumulation and quality, sediment particle size and metal concentration and provided recommendations for the design and maintenance with specific regard to ponds and sediment issues. They concluded that the removal of the sediment should be related to lack of capacity / storage within the system as opposed to contaminant accumulation. Interestingly it was also noted that while metal concentrations increased year on year, that only a few of the samples taken presented any effect level for aquatic organisms. In this particular case, the metal flux into the ponds was associated with the coarse sediment suggesting a need for the control of runoff from construction developments nearby.

The implementation of sustainable urban systems as an effective management tool to minimise impact of urban developments on receiving watercourses has always been surrounded by questions of maintenance and uncertainty over costs and effective practices (McKissock *et al*., 2003). The sediment “traps” engineered as part of the systems are designed to accumulate such sediments and thus the question of sediment removal should be one which is addressed at the outset of the planning process. In order to develop a maintenance schedule, the frequency of such sediment removal in addition to the volumes and the quality should be considered.

However, the long-term effects of multiple rainfall and runoff events through swales and the resultant long-term sediment deposition and retention is not well understood and therefore much uncertainty surrounds long term maintenance. Current practice states that a swale design life is around 25-30 years and aside from vegetation cutting and litter removal the maintenance required is unknown (Allen *et al*., 2015). With traditional drainage systems, local catchment-based approaches to predict run off require the interpretation of varied and many intercorrelated drainage basins (Wharton, 1994). The prediction of run off within SUDs requires a similar approach, albeit on a much smaller scale, but the use of different structures within the system makes prediction and retention complex.

Recent field research has been undertaken to attempt to quantify the sediment retention rates within swales from a single release event and to look specifically at the deposition, resuspension, and subsequent conveyance of such sediment. Rare earth oxides were used to provide trace signatures within the study (Allen *et al*., 2015). After initial deposition the sediment was observed to continue moving through the vegetated swales with further resuspension and transport during subsequent events. The initial sediment deposition zone, suggested as the impervious / vegetated interface (Hussien *et al*., 2007), potentially only provides a temporary sink and continued events would cause resuspension and further movement of sediment through the swale. Further research is needed to consider multiple events over extended periods of time. It can be assumed that a proportion of sediment will move through the swale at some point and therefore efficiencies of a swale cannot be predicted. Attentions should be turned, as suggested by Heal (1999), to the retention within any ponds or wetland which receive inputs from swales and other sources.

2.6.4 Ecological Considerations

SUDS can benefit many priority habitats and species contained within local Biodiversity Actions Plans (BAPs), as well delivering some of the objectives contained within each of the UK four national biodiversity strategies (RSPB, 2012). Much of the literature focusses on the planning and design stage ensuring that ecological advice and best practices are incorporated ensuring a positive outcome for biodiversity (e.g., Mak *et al*., 2017; Monberg *et al*., 2018).

It has been demonstrated that constructed wetlands can be rapidly colonised and provide ideal habitats for specialised species (Noon, 1996) such as amphibians, where the value of SUDS is noted particularly in relation to gene flow (O'Brien *et al*., 2020). While the biodiversity of SUDS can be lower than that observed in natural ponds (Krivstov *et al*., 2019), there is overwhelming evidence that they can offer conservation value providing suitable habitats for pollution tolerant invertebrate taxa (Sun *et al*., 2019). Ecological assessments pertaining to wet basins within SUDS have been documented but those related to other SUDS elements are relatively scarce (Kazemi *et al.*, 2011). SUDS systems present structural heterogeneity in habitats, often associated with high biological diversity (Rosenzweig, 1995), with differences in water quality, water dynamics, vegetation and size / volume. Larger areas such as swales, retention ponds and wetlands therefore present the opportunity to support higher biodiversity (Beninde *et al*., 2015) if correctly managed.

One of the influencing factors for invertebrate communities within these drainage systems appears to be the presence of gravel and the retention of leaf / plant litter coupled with a large number of plant taxa (Kazemi *et al*., 2009a & b). The invertebrate diversity of stormwater wet detention ponds (SWDP) has demonstrated that structures designed for treatment and flood protection purposes, similar to those observed within SUDS, become aquatic environments which play a local role for biodiversity similar to that of natural small and shallow lakes (Stephansen *et al.,* 2016). However, there is a lack of consideration of temporal variation, long-term impacts (Kazemi *et al*., 2009b) and the impact of pollution levels on the biodiversity of such systems (Sun *et al.,* 2019). As these structures are generally designed for the retention of sediments and, consequentially, their associated urban contaminants, toxicity to aquatic invertebrates and particularly to sediment-associated organisms which feed and spawn close to, or within, the bed (Garpentine *et al*., 2002, 2008) needs further investigation. Bioaccumulation of contaminants within the biota of detention ponds has been observed in both vertebrates (Campbell, 1994; Salem *et al*., 2014) and invertebrates (Karouna-Renier and Sparling, 2001; Stephansen *et al*., 2016). These long-term impacts and combined acute toxicity, have been reported in the literature with respect to urban watercourses and wetlands subjected to stormwater run-off (e.g., Fleeger *et al*., 2003; Carew *et al*., 2007; Johnson *et al*., 2011). However, clear cause and effect relationships between specific contaminants and changes to the ecosystems have not been demonstrated (Stephansen *et al*., 2016).

Regarding vegetation use within SUDS, phytoremediation is considered a key treatment process (Allen *et al*., 2017b) but the potential for SUDS to act as contaminant sinks requires further research to fully assess the impact of such contaminants on biodiversity in situ. In addition, resuspension, and movement of sediment through this "jerky conveyor belt" and connectivity between SUDS and receiving waterways, further highlights the risks that contaminated sediments could have on receiving ecosystems, with direct links between metal concentrations and microbial bacterial populations noted within the literature (Havens, 1994a & b; Forrow and Maltby, 2000)

2.7 Summary and Conclusion

This literature review has shown that there is a relatively poor understanding of the pathways that sediments take through drainage systems to receiving waters (Taylor, 2007). It demonstrates that there is a lack of detailed understanding about the transport and accumulation of RDS and associated contaminants within and through SUDS systems following multiple rainfall events. SUDS are specifically designed to reduce connectivity in urban fluvial systems to effect flood control (aim 2) and a consequence of this is the potential for sediment accumulation with the system (aim 3).

There is lack of studies, within the scientific literature, which have looked at the long-term performance of SUDS systems and current design guidance has been based on information modelled from single event monitoring. This research aims to apply a sediment cascade (connectivity) approach to the context of a SUDS system, in relation to other urban infrastructure. Attention will be given to the individual relationships between sources, transport, deposition and resultant modification of sediments and contaminants and to explore the catchment configuration and (dis)connectivity within the sediment cascade.

The next chapter outlines and justifies the choice of SUDS and the monitoring and sampling methodology employed to deliver the objectives as set out in Chapter 1.

Chapter 3 Methods

3.1 Introduction

This chapter initially justifies the choice of the Upton development SUDS in Northampton for field work and subsequently describes the field, laboratory and statistical methods used to undertake the research project.

3.2 Study Site

The study site selected for this project was based at Upton, located in the Southwest District of Northampton (Figure 3.1 and Figure 3.2).

Figure 3.1: Map (OS 1:800000) showing study site near Northampton, UK (Ordnance Survey, 2021).

Figure 3.2: Aerial Map (Scale 1:4000) of the study site within Northampton (GetMapping, 2021).

Upton was identified as a key area of strategic urban development in Northampton, and outline planning permission was granted in 1997. After the establishment of working groups and an "Enquiry by Design", a planning tool that brings together key stakeholders to collaborate on a vision for a new community (Isherwood, 2013), variations to the planning were submitted between 2001 and 2005. In 2001 an urban extension project was started with partners including Northampton Borough Council, English Partnerships (the landowner) and the Prince's Foundation, to promote best practice in sustainable urban growth (Isherwood, 2013).

One of the principal aims of Upton was to provide more and affordable housing in Northampton but, creating a neighbourhood based on sustainable urbanism, also presented a difficult challenge. At the time, 1997, housing developments were still designed around the low-density suburban model. Upton was designed to promote good design and development practices for developers and house designers (ADS, 2011) combining green building technologies and traditional architecture with the integration of more contemporary design (Farr, 2008).

Upton was to be the first coding project undertaken by English Partnerships (Momoh and Medjdoub, 2018) and an Urban Framework and Design code was developed based on key developmental principles promoting sustainable urban growth. This included permeable streets, good street views, quality public spaces, in addition to environmental sustainability, in terms of housing (BREEAM standards) and sustainable urban drainage systems (SUDS). The codes enabled the development of a design guide on the inter- relationship between infrastructure and urban developments (Momoh and Medjdoub, 2018).

The SUDS at Upton not only offered an opportunity to manage rainwater run-off but also to promote biodiversity and create an ecological network linking Upton with the Upper Nene River Valley and thus forming a central part of the unique design for sustainable urban growth, highlighted as one of the benchmarks and key sustainable indicators for the project. The Upton project was considered innovative in terms of stakeholder engagements and the Enquiry by Design, the large-scale use of design code and the implementation of sustainable urban drainage systems (ADS, 2011). The network of linked swales, having both storage and infiltration functions, were perceived to enhance open green spaces providing “green fingers” from the neighbouring country park deep into the development and public realm and thus enhancing biodiversity (Dickie *et al*., 2010).

Development C (the study site) had planning permission approved in 2005. The wetlands, shown in figure 3.3, were designed prior to the development of site C, and constructed by English Partnerships during the development of Sites A and B.

Figure 3.3: Wetlands within the study site near development Site C (Google Earth (2021a)).

While the site was not considered to be at risk of flooding from the River Nene, the development of the site needed to alleviate any flood risk to adjacent lands. The 1 in 200-year flood level for the site was predicted (based on modelling carried out by the Environment Agency) to be 62.30m AOD (Above Ordnance Datum). The level of the open space within the country park (Figure 3.4) was designed to be approx. 65.5m AOD with the bottom of the wetland ranging between 63.36m AOD and 63.8m AOD The design of the wetland was as series of interconnected ponds to form one large wetland. The range of contours in the original design for the wetland is shown in Figure 3.4 which is taken from the original Constraints Plan submitted as part of the planning application, to Northamptonshire Council, produced by Pell Frischmann (2004). Sediment depths of pond 1 were measured in 2014 as part as an undergraduate dissertation (supervised by Copeland-Phillips). Those results are presented in Figure 3.5 to provide a representation of the changing depths of the pond due to sediment accumulation.

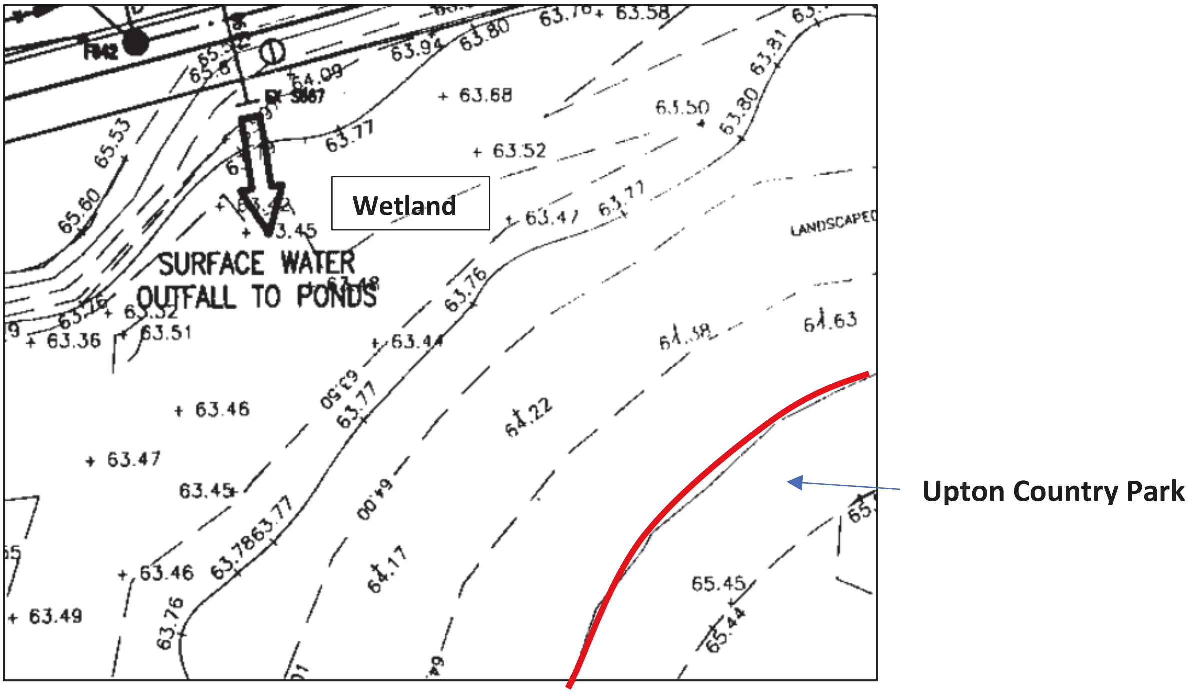

Figure 3.4: Part of the original plans for the wetlands showing different depths and contours along the wetland. Adapted from Pell Frischmann (2004).

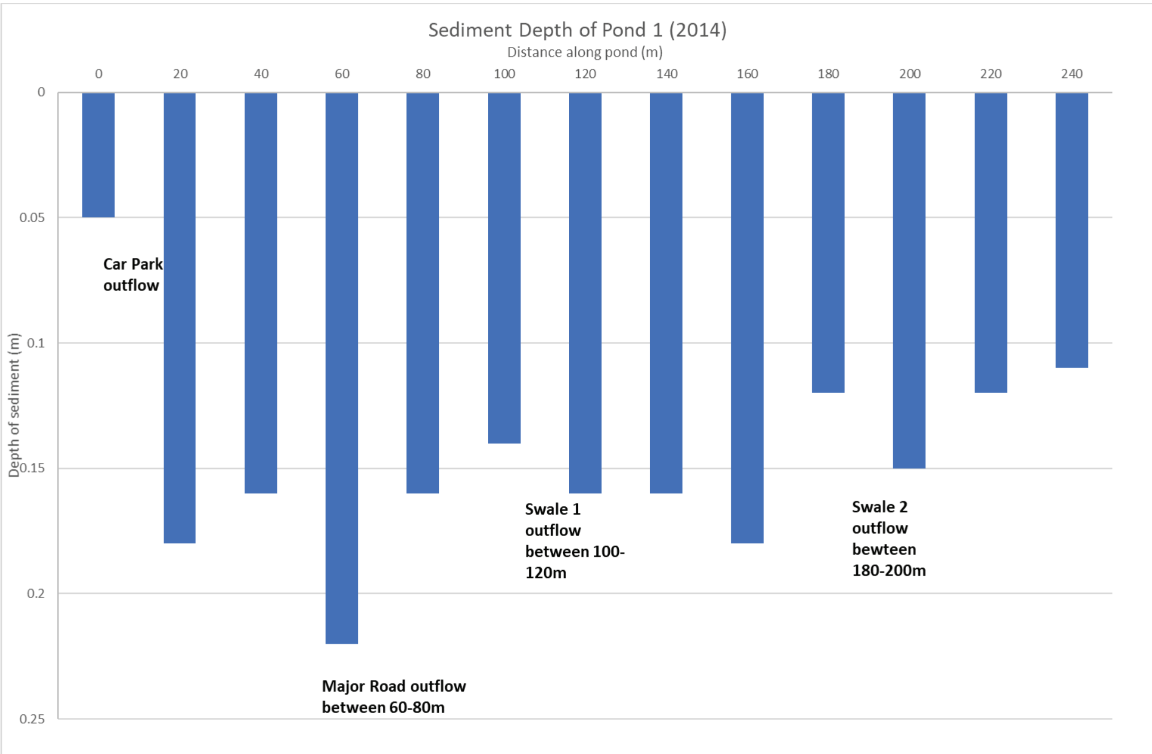

Figure 3.5: Sediment depths of Pond 1 taken in June 2014. Compared with the original surface in Figure 3.4.

The swales (see Figure 3.6) within the new development would directly discharge into the constructed wetland (pond 1), making them an attractive study site as the swales would only receive input from the surrounding impermeable pavements and associated road drains and were not connected to previous parts of the development. The wetland would also receive input from a car park (solely for the use of the Elgar Community Centre) at the eastern end of the of the wetland and a road drainage pipe (connected to the north of the development near the A4500) (Figure 3.7).

Figure 3.6: Example of the vegetated swales placed within Site C. The alignment of the swales within this section of the SUDS is north to south. The vegetated swales are positioned in the middle of the street for maximum sunlight (Google Earth, 2021b).

Figure 3.7: The inputs into the Wetland (pond 1) (Google Earth, 2021c).

The SUDS at Upton consists of swales, wetlands, ponds, and green field attenuation areas. The direction of the system from Site C (white arrows) is shown in Figure 3.8 as well as the direction of flow from Site A and B (red arrows).

Specific monitoring in this area will be further discussed in the rest of this chapter.

Figure 3.8: Representative diagram of the water movement round the site prior to discharging into the River Nene (Google Earth, 2021d).

As previously mentioned, the SUDS at Upton were designed to managed rainwater and promote connectivity between habitats, throughout the development and with the Upper Nene River Valley. The network of connected swales was designed to have both a storage and infiltration function and therefore provided an opportunity to investigate both hydrological (aim 1) and sedimentological connectivity of the system (aim 2). Pervious modelling of SUDS systems has been cited in Chapter 2, but none have addressed the varying connectivity in real time, instead often modelled on single event monitoring. The Upton site offered a unique opportunity, in terms of its novel design which aimed to provide a blueprint for the development of further sustainable communities and thereby long-term monitoring of both hydrological connectivity and sediment retention could also provide a basis for design and future modelling of SUDS performance taking into account varying different climate changes and weather unpredictability. A further reason was this site selection was based on the practicalities of research and field work. The rapid response (flashiness) of urban hydrological systems, meant that access might have been needed in less than an hour's travel time from the University of Northampton

3.3 Field Sample Collection

In order to meet the aims and objectives of this research, several different methods of field sampling and laboratory analytical techniques have been employed which will be detailed in this chapter.

There is the need to investigate the appropriate strategies which are adopted to control the flow and consequential storage of sediments within the SUDS and partly inform the measures which are taken in the maintenance of such systems. Suspended sediment is a vector for the transport of sediment-associated nutrients and contaminants (e.g., heavy metals) in fluvial systems (Horowitz, 1991; Owens *et al*., 2001). As SUDS are designed to trap a proportion of such sediment, it is important that such sediments are characterised to determine appropriate management strategies as well as the lifetime expectancy and efficiency of SUDS. The following methods have been adopted within this study to attempt to identify the types of sources involved as well as sediment transport mechanisms.

The sampling programme for this study was designed to collect representative samples of suspended sediments at the field site at Upton, Northampton. In addition, several monitoring devices were placed *in situ* to provide further parameters, which would provide complimentary data and allow for the capture of hydrological data. A table of field sampling is detailed in Table 3.1. linked to the specific objectives of the thesis.

Table 3.1: Sampling regime and associated analysis.

Nature of Sample	Method employed	Dates Collected	Analysis (inc. associated software)	Objective
Suspended sediment sampling	Time integrated sampler	February 2016 March 2016 March 2017 October 2017 February 2018 July 2018	Particle size Geochemistry Mineral magnetic parameters radionuclides	Objective 3
Background Sampling	Gouge corer	2018	Particle size Geochemistry Mineral magnetic parameters radionuclides	Objective 3
Water level	In-Situ Rugged TROLL 100®	October 2014-August 2018	Win-Situ® Baro Merge Microsoft Excel; Microsoft Office 365 Pro Plus Minitab®19	Objective 2:
Rainfall	WatchDog 1120 Data-Logging Rain Gauge	November 2014-August 2018	Microsoft Excel; Microsoft Office 365 Pro Plus Minitab®19	Objective 2:

3.4 Rainfall

Rainfall was measured within the catchment using a WatchDog 1120 Data-Logging Rain Gauge. (Figure 3.9) The gauge was installed according to the specific instructions and mounted on a rigid (25-32mm) pipe at a height of 1.25m above ground level in an open area (Spectrum Technologies, Inc, n.d.). The gauge was located approximately 170m from the Baro®TROLL on a local farm, 52°13’43’N 0°56’49’W. The field in which it was located is used only for occasional grazing, especially during the lambing season. Data were collected at 5-minute intervals in order to ensure that the data would complement the hydrology data obtained from the Rugged ®Trolls (See section 3.3). The rain gauge was calibrated twice a year when it was removed from site to ensure that the collector was clean and operating efficiently.

Figure 3.9: WatchDog 1120 Data logging Rain Gauge (Spectrum Technologies, Inc., n.d.).

3.5 Water Level

Water Level was recorded between 2014 and 2018 using an In-Situ Rugged TROLL 100® (Figure 3.10). It provided an economical way in which to measure the water level, water pressure and temperature at each of the sites. Each TROLL was set up to record data every 5 minutes which would provide high temporal resolution data which could be related to local rainfall data collected at the same timestep. "Flashiness" or storm transients, representing the relatively quick rises in water level (rising limbs) which tend to be less studied than aspects of stream flow such as magnitude, frequency and duration and was focused on in this study (Shuster *et al*., 2008). The rate of recession is also important dependent partly on the hydraulic connectivity between the landscape features of a catchment. The SUDS ponds are designed to interconnect at times of high-water level, water flow between the systems is rarely visible and difficult to record. As noted within section 2.7.3, the transport and movement of sediment through SUDS is often presented in terms of single event conditions and this study aims to address the long-term source/ pathway /sink relationships which are important in understanding performance and maintenance. Therefore, the use of data loggers presented an ideal data opportunity to measure hydrological flux within the system and coupled with sediment studies could provide some understanding about the movement of sediment within this "jerky conveyor belt".

Figure 3.10: In Situ Rugged Troll 100 and Baro Troll 100. (Source: In-Situ, 2013).

The Rugged TROLL® 100s were complemented by the installation of a Rugged Baro TROLL® which measured and logged barometric pressure and air temperature. The data obtained was corrected by compensating for barometric pressure effects during the course of a logged event using Win-Situ® Baro Merge.

The locations of the Rugged TROLL® 100s are shown in Figure 3.11. The Rugged Baro TROLL® was placed under a foot bridge between Pond 2 and Pond 3 near the Rugged TROLL® that was placed in Pond 2.

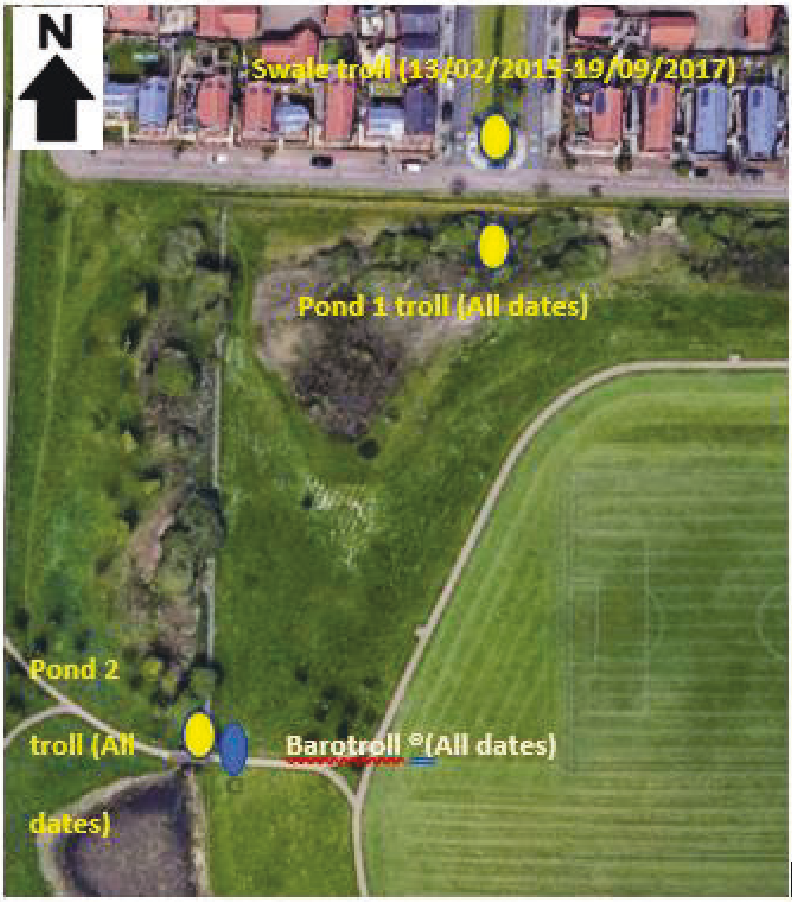

Figure 3.11: The locations of both the Rugged TROLL® 100s (shown in yellow) and the Rugged Baro TROLL® (shown in blue) (Google Earth, 2021e).

3.6 Suspended sediments within the SUDS

As this research was concerned with a small area of the SUDS which, was inherently interconnected it was pivotal to characterise the sediment which potentially moves between the varying swales and ponds. It is often assumed that the different surface systems within the SUDS are interconnected and offer the potential for a cascade of sediment traps.

It is noted that it is often the finer grained sediment (<63um) that is responsible for the sediment associated transport of anthropogenically derived substances within these types of drainage systems as well as in other fluvial systems (Meybeck 1982; Allan, 1986; Walling and Moorehead 1989).

Various sediment sample types have been used in source tracing studies. With a particular focus on contemporary timescales (as opposed to historic), investigations have used instantaneous suspended sediment samples (e.g., Peart and Walling, 1986; Walling and Woodward, 1992; Collins *et al*., 1997a; Collins *et al*., 2001; Russell *et al*., 2001; Carter *et al*., 2003) or those from on the channel bed (e.g. Collins and Walling, 2007; Collins *et al*., 2012a and b). The former was collected in the form of representative bulk water samples, during events, using submersible pumps powered by a portable generator and then recovered using continuous flow centrifugation (Collins *et al*., 1997a) or the deployment of automatic samplers (Russell *et al.*, 2000). The latter were commonly collected using a re-suspension technique (Lambert and Walling, 1988; Duerdoth *et al*., 2015).These traditional approaches have been noted as having three main problems (Russell *et al*., 2000); manual sampling is limited by manpower or inhibited by the cost of automatic equipment; the automatic samplers tend to only collect small quantities of the sample and therefore hindering potential geochemical analysis of the sediment; the samples are representative of a moment in time and not of longer periods which then does not take into account longer term flux of nutrient and contaminant concentrations (Russell *et al*., 2000). The design and principle of operation of the sediment sampler is described in Phillips *et al*. (2000) and offered a potential solution and presented an *in-situ* sampler which could continuously sample suspended sediment during the period of deployment. This simple sampler has been shown to collect a representative time-integrated sample of suspended sediments (Russell *et al*., 2000) and that the particle size characteristics of that sediment collected are statistically representative of the ambient suspended sediment (Russell *et al*, 2000).

A total of 4 sediment samplers (at any one time), as described in Phillips *et al.*, 2000 (shown in Figure 3.12) were utilised to collect samples at the inlet and outlet of swales and within each of the retention ponds at different periods of time in the overall sampling campaign. All locations over the sampling period are shown in Figure 3.13 and Table 3.2 Although the study area comprised of two swales as shown in Figure 3.7 only one swale had been observed to carry water through the system and was the only one directly sampled. Table 3.2 provides the coding for the samplers used during the study period.

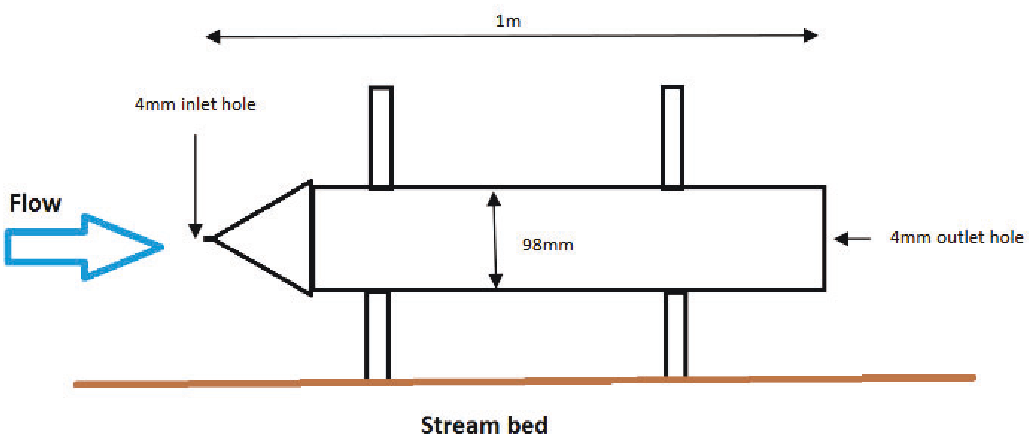

Figure 3.12: Time integrated sampler as described by Phillips et al. 2000 and reproduced in Pulley, 2014.

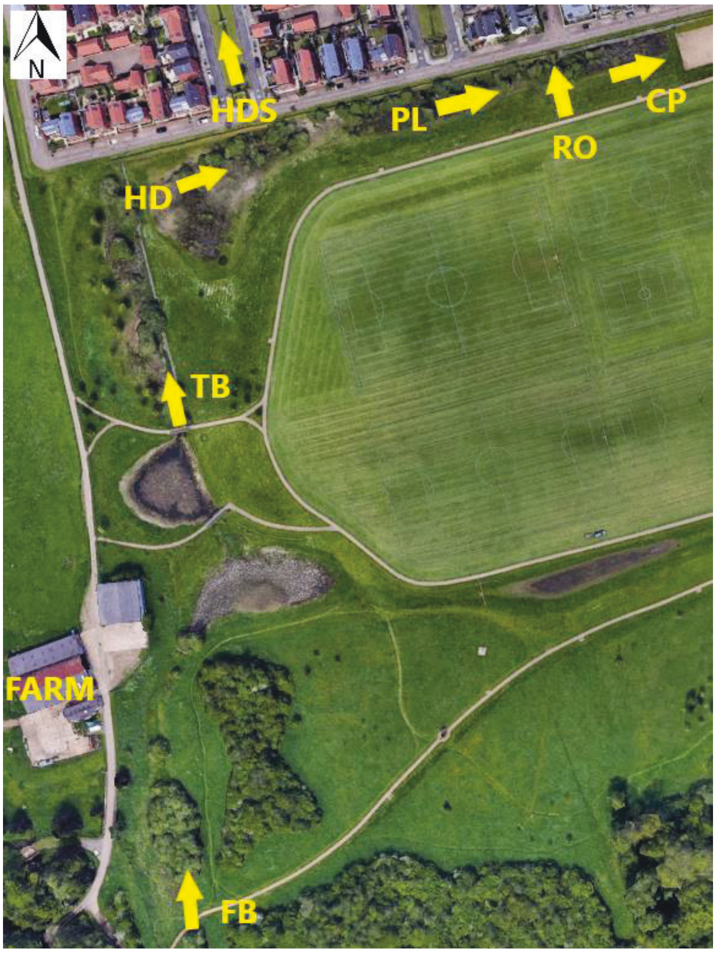

Figure 3.13: Yellow arrows indicate the location and direction of the inlets of the sediment samplers (Google Earth, 2021f).

Table 3.2: Codes used to identify sampler position within the SUDS.

Code Adopted	Position of Sampler	Dates of Collection
CP	Car Park Pond 1	Feb 16, Mar 16, Mar 17, Oct 17, Feb 18, July 18
PL	Padding Lane swale out Pond 1 (downstream of the RO sampler)	Feb 16, Mar 16, Mar 17, Oct 17, Feb 18, July 18
RO	Road Out Pond 1 (perpendicular to the pond)	Oct 17, Feb 18, July 18
HDS	Harrington Drive Swale	Feb 16, Mar 16, Mar 17(no water was observed in this swale following the last sampling date, this sampler was moved to RO).
HD	Harrington Drive swale out Pond 1	Feb 16, Mar 16, Mar 17, Oct 17, Feb 18, July 18
TB	Troll Bridge (between Pond 2 and Pond 3)	Feb 16, Mar 16, Mar 17, Oct 17, Feb 18, July 18
FB	Farm Bridge	Feb 18, July 18 (a sampler was placed in this location towards the end of the study period)

Water and sediment captured by the time-integrated samplers were removed from the field in 20 litre containers and stored in a cold room for 24hrs. This allowed settling and extraction of most of the water by siphon prior to decanting and oven drying.

3. 7 Laboratory Methods

3.7.1 Preparation of samples

All sediment samples obtained were transported to the laboratory for analysis. Suspended sediment samples from the tube samplers were allowed to settle for excess water to be siphoned off. All soil/ sediment samples collected from were oven dried at 40°C (Dearing, 1999). This prevented any thermal changes to mineral magnetic signatures occurring. After being dried the samples were then manually disaggregated using a pestle and mortar and sieved to <63µm. By sieving to <63µm it is possible to minimise the issues of differences in particle size between sources and sediments assuming that this is the fraction that contains the bulk of the suspended sediments (Phillips *et al*., 2000).

3.7.2 Organic Matter Content

There are several methods to determine OM, each method presenting advantages and disadvantages regarding convenience, accuracy, and expense (Nelson and Sommers, 1982). Methods to determine OM include a wet oxidation procedure followed by titration with ferrous ammonium sulphate or combustion, followed by the collection and measurement of evolved carbon dioxide (Schumacher, 2002) and the use of elemental carbon analysers, which are expensive to purchase and maintain (Salehi *et al*., 2011). Loss on ignition (LOI), involves the combustion of samples at high temperatures and is generally accepted as an inexpensive method for the estimation of organic matter (OM) (Salehi *et al*., 2011), avoiding the need for chromic acid and the disposal of resultant wastes (Konen *et al*., 2002). Fairly consistent results, from LOI (~2% error) can be observed providing sample mass, heating time and temperature are carefully controlled (Heiri *et al*., 2001). LOI is accepted as a reliable method (Howard and Howard, 1990; Dean, 1999; Brunetto *et al*., 2006; Abella and Zimmer, 2007; Escosteguy *et al*., 2007, Wang *et al.*, 2012) and therefore has been adopted in this study. However, the optimal temperatures involved in the heating are difficult to determine and can affect LOI results (Ben-Dor and Banin, 1989; Schulte *et al*., 1991). High temperatures have been observed to eliminate structural water from inorganic constituents and clays (Ball, 1964; Salehi *et al*., 2011) resulting in an overestimation of organic content (Pulley, 2014). Various temperatures (300, 360, 450 and 600°C) (Schulte *et al*., 1991; Konen *et al*., 2002; Brunetto *et al*., 2006; Abella and Zimmer, 2006; Escosteguy *et al*., 2007; Yerokun *et al*., 2007) have been reported within the literature with temperatures around 360 ◦C (Salehi *et al*., 2011), identified as optimum, destroying less inorganic carbon and less clay structural water loss but burning most organic carbon and using less electrical energy.

Ceramic crucibles were preheated to 105°C to remove any residual moisture. Crucible mass, to 3 decimal places (g), was recorded. Samples were weighed (2g ± 0.005) into each crucible and oven dried at 105°C for 2 hours, after which the mass of the crucible plus sample were recorded. Crucibles were transferred to a Carbolite muffle furnace at 360 °C for 4 hours. The crucibles were allowed to cool in the furnace for a short period before being placed in a desiccator and re-weighed. LOI was calculated as follows.

*Loss on Ignition (%) = ((pre-ignition weight - post–ignition weight) / pre-ignition weight)) *100*

3.7.3 Particle Size Analysis

The particle size distribution of sediments provides information which allows for assumptions regarding transport and deposition as well as entrainment within a system. More notably it can also provide clues about provenance of the sediment and has been used to characterise sediments in forensic and fingerprinting studies (Junger, 1996; Sugita and Marumo, 2001; Pye and Blott, 2004).

SUDS are designed to provide important storage zones for sediment arising from various urban sources. Many of the geochemical properties of this deposited sediment are known to be strongly related to particle size and there is a very strong positive correlation between decreasing particle size and increasing trace element concentrations (Horowitz, 1991). The movement and fate of these particles are ultimately responsible for the transfer of anthropogenic pollutants (e.g., heavy metals, radionuclides) and nutrients through any aquatic system. The concentration of these resultant pollutants within the bottom and suspended sediments is often substantially higher (up to 100, 000 times) than the dissolved levels (Horowitz, 1991). One of the most important factors in trace element-sediment interactions is the surface chemical reactions. Since fine particles present such a large surface area, they are considered the main vehicle for trace elements transport. Other factors such as surface charge and cation exchange capacity have a substantial effect on trace element concentrations but as Horowitz (1991) suggests it is somewhat difficult to differentiate between these effects. If only physical property is to be determined, then:

> "*grain size is by far the property of choice, because it seems to integrate all the others"*
>
> (Horowitz, 1991, p 16)

There are many different techniques for determining particle size including, dry and wet sieving and the use of settling columns. Laser granulometers, laser diffractometers or laser diffraction spectrophotometers measure the size of particulate materials using laser diffraction. The instruments vary in their sensitivity and in the range of particle sizes that can be measured however they offer a rapid determination of size distribution (Pye and Blott, 2004). As with any other technique, accuracy and reproducibility are key and therefore it is essential that standardised sample handling and pre-treatment should be ensured. In addition, techniques such as these assume that all particles are spherical so in the case of platy particles, such as mica, individual particles may appear larger than the actual equivalent spherical diameter and thereby provide an overestimation of such volumes (Hayton *et al.,* 2001). There is also the assumption that there is a uniform density/ composition of the material and therefore there is a tendency for over estimation of the lower density particles. Finally, it is imperative that the optical model utilised is relevant to the diffractive and refractive properties of the material which is to be analysed and the liquid in which the sediment is suspended in the instrument.

For the purpose of this analysis, a Malvern Mastersiser 2000 was employed. Unlike other instruments, which employ the Fraunhofer optical model, the Malvern uses the Mie theory. The Fraunhofer model does not describe scattering exactly and is less appropriate for small particle sizes, especially where the particle size is similar to the light wavelength (0.75µm). Therefore, it is possible that light diffraction can underestimate the volume of clay materials (Loizeau *et al*., 1994). The Mie theory was developed to predict how light is scattered by spherical particles and ascertains the way in which light is carried through or even absorbed. While this is considered a more appropriate theory it does assume some very specific information about the particle such as its refractive index and its absorption (Malvern, 2010).

As per Pulley (2014) the nature of the sediment within the study area was assumed to be mostly discrete particles, and to prevent an underestimation of the fine particle size fractions (Di Stefano *et al.*, 2010) caused by the aggregation of particles by organic matter, the organic fraction was removed using hydrogen peroxide prior to analysis. 10ml of 30% hydrogen peroxide was added to 0.1g of sediment and left for 24 hours at room temperature. Samples were heated at 70°C until bubbling had ceased and the samples were allowed to cool. 5 ml of 3% sodium hexametaphosphate solution was added to the samples which were left to stand for 5 minutes before analysis (Gray *et al.,* 2010**;** Pulley, 2014).

The samples were added to 500ml Ultrapure water in a Malvern 2000 unit. The sample was then subject to 2 minutes of ultrasonic dispersion (Blott *et al.,* 2004) prior to being measured over a 60 second timeframe at 8-12% obscuration. Malvern suggest that an optimum level of obscuration is between 3-20%. Each sample was measured once but every 100 samples were measured 3 times to ensure and confirm consistency of the results.

3.7.4 Gamma -emitting Radionuclides

Characterising fine sediments is often based on a suite of properties, which represents unique characteristics that can be used to identify sediments that are derived from a particular source, and a comparison of these properties with other deposited and transported sediments. There are a range of properties which can offer the potential for discrimination including geochemistry, mineral magnetic signatures radionuclides, particle size and geochemistry as well other properties such as colour, mineralogy and the use of microfossils, macrofossils, and other biogenic properties.

Samples were prepared as noted in 3.2. This fraction was then packed to a depth or approximately 4 cm in a PTFE pot and sealed with a turnover cap and paraffin wax. The samples were left to equilibrate for 21 days in order to allow the growth of ^{222}Ra (return of equilibrium between ^{214}Pb/ ^{214}Bi and ^{226}Ra) in accordance with USEPA method 901.1. Radionuclide activity was determined by high resolution, low-level gamma spectrometry using Ortec hyper-pure germanium (HPGe) detectors.

3.8 Geochemistry

Common utilised methods for the measurement of geochemical tracers include inductively coupled plasma (ICP) atomic absorption spectroscopy (ICP-AAS), ICP optical emission spectroscopy (ICP-OES), ICP mass spectroscopy (ICP-MS) and X-ray fluorescence (XRF). ICP-OES and ICP-AES (Atomic emission spectroscopy) both utilise a plasma for elemental analysis, and a sample solution, carried by argon gas into a torch (approximately 10 000°C) ionises the argon discharge. In both ICP-OES and ICP-AES the ionised gaseous mixture emits photons that are collected by a concave mirror or lens and the emission of the photons provides the technologies with their hyphenated sub names, OES and AES. The names represent the same method and technology and have been used interchangeably over the years. ICP-OES has been used in many studies within the literature regarding heavy metal concentrations of RDS (e.g., Lanzerstorfer, 2018; Zhao and Li, 2013) and SUDS soils, sediments, and water (e.g., Charlesworth *et al.*, 2017; Allen *et al*., 2017b; Tedoldi *et al*., 2017; Bird *et al*., 2019). ICP-OES was available at the time of study and as the literature showed presented an effective and comparable method of analysing heavy metal concentration within the samples obtained from the study site and therefore was selected as the preferred method. While it is not as sensitive at low element concentrations as ICP-MS and is often affected by spectral overlap, this was noted, and careful examination of the resultant spectra was undertaken to eliminate the problem of spectral overlap.

Samples were prepared using a microwave (The Microwave Accelerated Reaction System, Model MARS 6®). Approximately 0.8g (+- 0.005g) was weighed into tetraflouromethacrylate tubes. 10ml aqua regia (3:1 hydrochloric acid and nitric acid respectively) was used in order to extract the total metal content of each sample.

The procedure used by the CMS MARS 6® is outlined in Table 3.3.

Table 3.3: Stages and methods employed for the digestion of sediment samples in CMS MARS 6® Microwave Digestor (after Pulley, 2014).

Stage	Temperature (°C)	Power (watts)	Duration (minutes)
1	Ramp to 120	1000	8
2	Hold at 120	1000	3
3	Ramp to 170	1500	10
4	Hold at 170	1500	3
5	Ramp to 180	1500	4
6	Hold at 180	1500	20
7	Cool down	0	20

Once digested the samples were filtered through a Whatman No.1 filter paper into a 50ml volumetric flask (a fine, slow filter paper was used to remove particles which may have blocked the spectrophotometer. The sample was diluted to 50ml using Type 1 ultrapure water, a subsample of which was decanted into a 10ml polypropylene tube for analysis. All samples were analysed using a Thermo iCAP 6500 Duo View ICP-OES. A series of standards (Reagecon ICP19A10) were made to appropriate volumes and concentration with Type 1 ultrapure water. The ICP-OES was calibrated before each use and the first sample was repeated at the end of each session to check for drift. Each sample was measured three times during the analysis and an average taken. Each batch of samples (40) contained a blank which would provide a basis for estimating contamination during the digestion procedure. The contamination for each batch of samples was considered in the final calculations of total element concentration. The elements, wavelengths, calibration correlation coefficient, predicted method detection limits (MDL) and method quantification limits (MQL) are shown in Table 3.4.

Table 3.4: Elements, Wavelengths, Correlation, MDL and MQL.

Element	Wavelength	Correlation	Predicted MDL	Predicted MQL
Ag	328.068	0.99	0.0023	0.0076
Al	396.152	0.99	0.0014	0.0047
B	249.773	0.99	0.0015	0.0050
Ba	455.403	0.97	0.0001	0.0003
Be	313.042	0.99	0.0001	0.0002
Bi	223.061	0.99	0.0041	0.0139
Cd	228.802	0.99	0.0008	0.0009
Co	228.616	0.99	0.0004	0.0013
Cr	283.563	0.99	0.0008	0.0029
Cu	324.754	0.99	0.0011	0.0038
Fe	259.940	0.99	0.0010	0.0035
Ga	294.364	0.99	0.0065	0.0218
In	230.606	0.99	0.0051	0.0187
Mn	257.610	0.99	0.0001	0.0005
Ni	221.647	0.99	0.0004	0.0014
Pb	220.353	0.99	0.0019	0.0065
Sr	404.771	0.99	0.0001	0.0001
Ti	334.941	0.99	0.3002	1.0007
Zn	213.856	0.99	0.0002	0.0007

3.9 Mineral Magnetic Measurements

Magnetic measurements have increasingly been used to detect heavy metal contamination in soils and sediments (Lu and Bai, 2006) particularly those related to industrial emissions, e.g., metallurgical industries and fly ash of coal combustions (Thompson and Oldfield, 1986).

The correlation of magnetic susceptibility and heavy metal content has been noted in atmospheric dusts, more specifically relating to the correlation between SIRM and the concentration of Pb, Cu, Zn and Cd in atmospheric dusts (Hunt *et al*., 1984). In addition, magnetic susceptibility has also been used to evaluate traffic and roadside related heavy metal pollution (Hoffmann *et al*., 1999; Gautam *et al*., 2004; Goddu *et al*., 2004), atmospherically derived pollution of topsoil (Hay *et al*., 1997; Karimi *et al*., 2011) and for the discrimination between anthropogenic and lithogenic fractions in urban soils (Meena *et al*., 2011).

Magnetic measurements are dependent not only on the various types of iron bearing minerals but also by particle size (Walden, 1999). It is the formation of iron bearing minerals in different environments, and relating to the mineralogy of the underlying bedrock, which provide the basis for their potential to be used to discriminate between sediment sources.

Equipment required to make magnetic measurements are relatively inexpensive. The measurements themselves are quick and non-destructive which lends to a straightforward and widely adopted method for differentiation of sediment and dust sources. The measurements are often affected by the particle size distribution of the sample (Thompson and Morton, 1979) and benefit from organic corrections (Pulley, 2014).

A suite of mineral magnetic measurements was made on the sediment samples collected within this study. The samples were prepared by packing 10 ml sample pots with the sediments. Each pot was packed to an approximate 2cm depth (~10g) within the pots. If insufficient sample was available, the pots were packed with cotton wool and the sample contained within a central position using circles of GFA filter paper, to maintain a consistent band of sample in an appropriate geometry relative to the position of the coil in the susceptibility meter. The measurements performed and associated equipment are shown in Table 3.5. Protocols for such measurements are noted in Lees (1999) and Walling and Foster (2016).

Table 3.5: Magnetic measurements (Foster et al., 2008).

Property	Measured (M) or Derived (D)	Units	Instrument or Calculation used
Low frequency susceptibility (χ_{LF})	M	10^{-6} m^3 kg^{-1}	Barington Instruments MS2b sensor (470Hz)
High frequency susceptibility(χ_{HF})	M	10^{-6} m^3 kg^{-1}	Barington Instruments MS2b sensor (4700Hz)
Frequency dependant susceptibility (χ_{FD})	D	10^{-9} m^3 kg^{-1}	((χ_{lf} -χ_{hf})/m)*100 (m= sample mass (g))
Anhysteric Remanent Magnetism (ARM $_{(40\mu T)}$)	M	10^{-3} Am^2 kg^{-1}	Molspin® anhysteretic remanent magnetiser, Molspin® slow-speed spinner magnetometer
Saturation Isothermal Remanent Magnetisation (SIRM)	M	10^{-3} Am^2 kg^{-1}	Molspin® pulse magnetiser, Molspin® slow-speed spinner magnetometer
Soft Isothermal Remanent Magnetisation ($IRM_{(-100mT)}$)	M	10^{-3} Am^2 kg^{-1}	Molspin® slow-speed spinner magnetometer
Susceptibility of ARM (χ_{ARM})	D	10^{-6} m^3 kg^{-1}	ARM x 3.14 x 10
S Ratio	D		-1 x (IRM_{100mT} / $IRM_{0.88T}$)
Hard Isothermal Remanent Magnetisation (HIRM)	D	10^{-3} Am^2 kg^{-1}	IRM1T/(1-Sratio))/2

3.10 Conclusion

This chapter outlines the methods which have been used within this research project. Fieldwork and data collection was carried out between 2014-2018. Long term data with regard to water levels within the SUDS is discussed in the following chapter. The main objective of data collection was to assess the impact of rainfall events on the hydrology of the system. In Chapter 5 the particle size, LOI, geochemistry, magnetism and presence of radionuclides is reported to investigate the potential sources of sediment as well as the connectivity between the sites. The use of environmental magnetic measurements has been used in place of more expensive geochemical techniques in some studies (e.g., Thompson and Oldfield, 1986; Walden *et al*., 1999; Xie *et al*., 1999; Shilton *et al*., 2005) as a proxy for environmental and atmospheric anthropogenic pollutants.

Chapter 4: Hydrology of the system using real time data

4.1 Introduction

Previous research, mostly based on single (real and simulated) events, has examined the retention of sediments and associated contaminants within SUDS (Chapter 2.6). Long-term event analysis has not been undertaken previously by looking at real time data. This chapter aims to examine, in detail, the impact of rainfall and associated connectivity between SUDS (Objective 1), over 10 events during November 2015 and May 2017. The Data were obtained in 5-minute intervals and has not been smoothed to provide general trends. Data with lower temporal resolution may not be adequate to fully represent the dynamics of storm runoff rates in these rapidly responding systems (Shuster *et al*., 2008). Examination of the raw data has provided an insight into the temporal responses, in terms of water level, within each of the monitored SUDS components.

Data were collected using In Situ Rugged Troll® 100 Data Loggers non vented (absolute water level logger) and corrected using data from a Rugged Baro Troll® (figure 3.10) which measured barometric pressure to compensate for water level changes due to barometric fluctuations. This correction was performed using the In Situ Baro Merge® software which automatically post-corrected water levels. Figure 4.1 shows the pre and post correction pressure data for the swale during a 12-hour example period. Note that both y axes show a range of 0.5 psi for the data sets to enable a direct comparison.

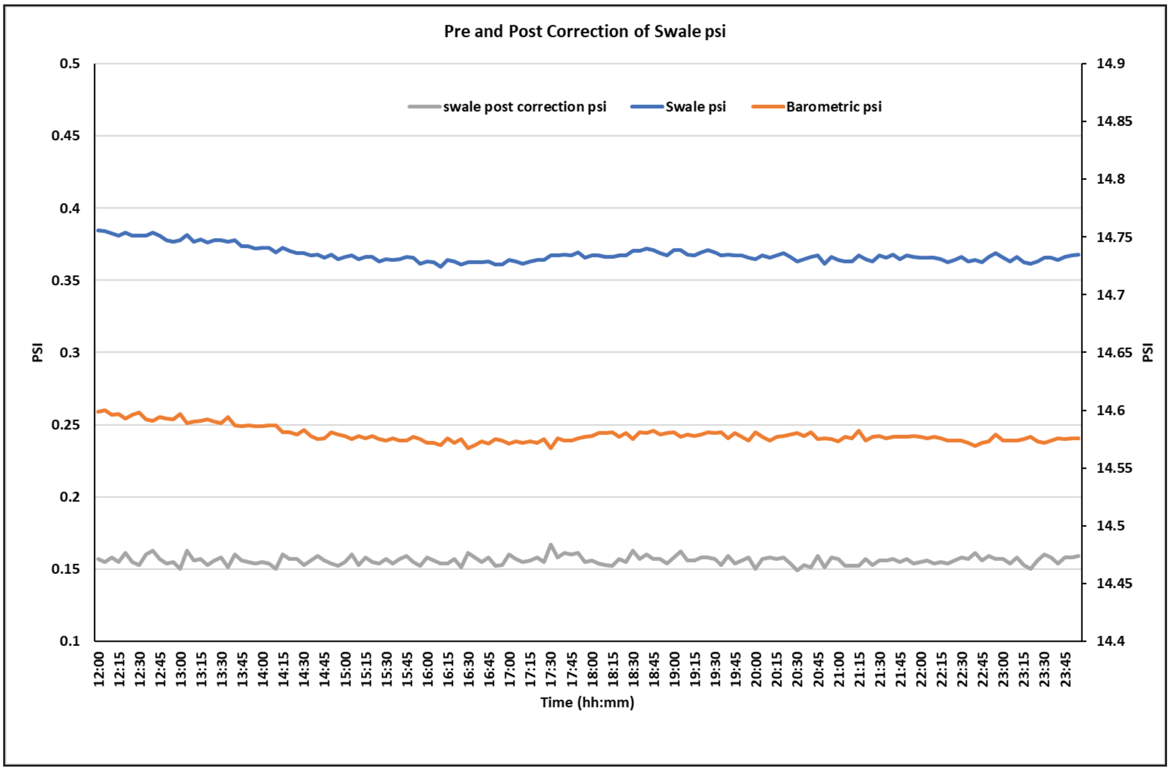

Figure 4. 1: Psi data obtained from the Troll® within the swale and from the BaroTroll® (range from 14.4-14.9 psi) and the converted data using the Baromerge® software (range from 0.1-0.5 psi).

The data from the Trolls was obtained in psi and converted to depth using a level mode non-vented equation. Depth refers to the depth of water over the pressure sensor plus the barometric pressure, but it is possible to compensate for barometric data using the Baromerge® software the equation becomes:

Equation 4.1 Depth (metres)= (0.703073) x (CrP)/ Specific gravity of water

Were CrP= corrected pressure using Baromerge® (psi)

Specific gravity of pure water at 4°C= 1 and at 20°C= 0.998

The data used in the Figure 4.1 was converted to depth using equation 4.1 and is shown in Figure 4.2.

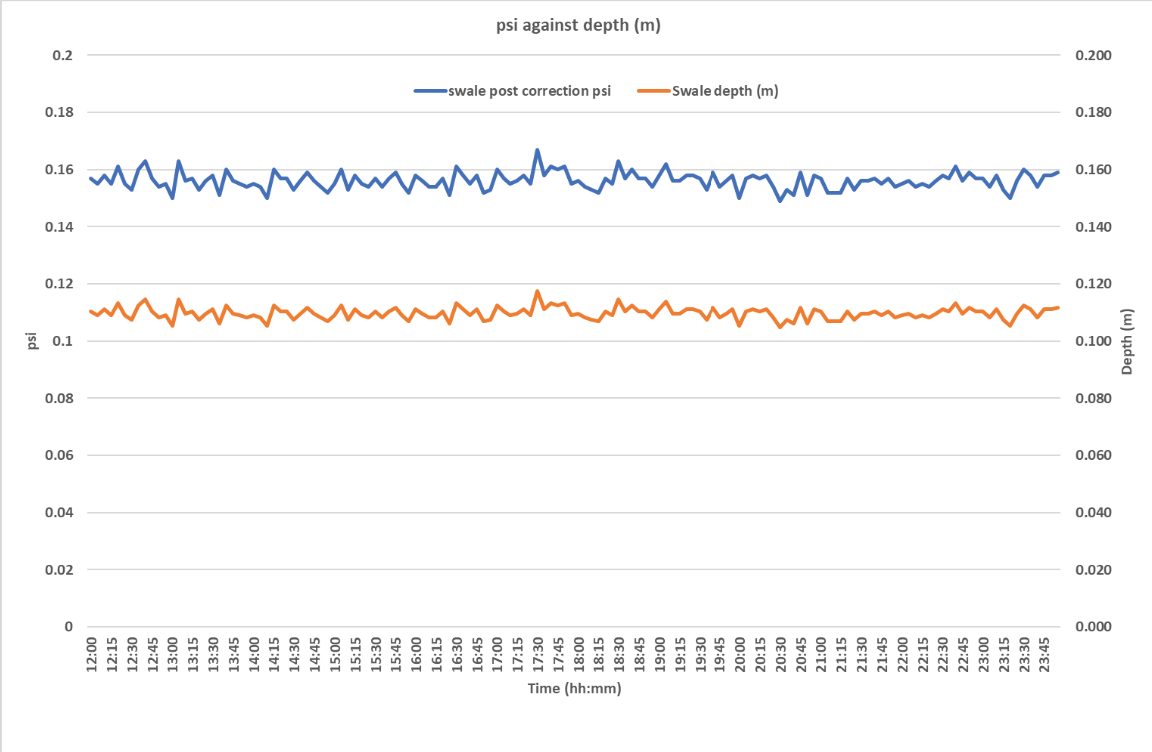

Figure 4.2: Baromerge Troll® data (psi) converted to depth (m).

Despite performing the correction there were instances (particularly in summer months) when interference from Barometric pressure appeared to affect the data as noted in Figure 4.3. In these instances, the noise in the data has been commented on.

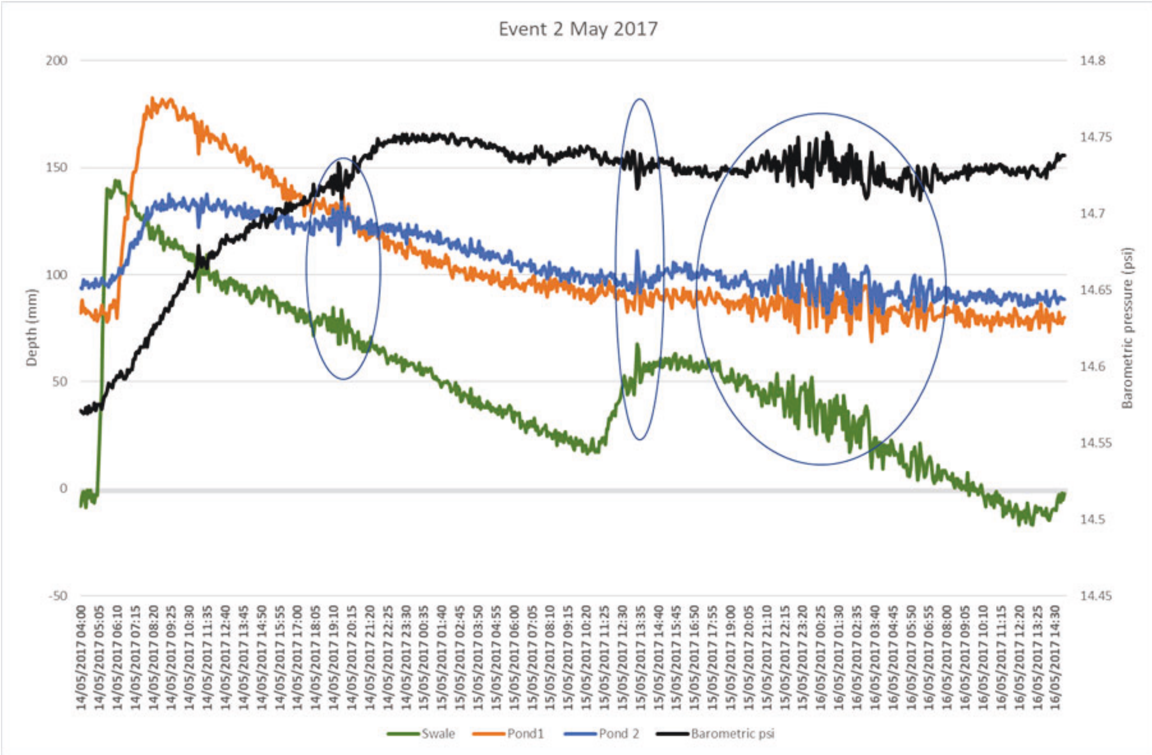

Figure 4.3: Data from May 2017, and example showing the noise in the data, water depth (mm) as a result of barometric data (psi).

Data were obtained using the In Situ Trolls® at locations and dates specified in Table 4.1. At the start of the data collection only two Trolls® were available along with the BaroTroll®. A further Troll® was purchased and placed in the swale From March 2015 but was removed from the swale in February 2018 as data recorded over the period 23/10/2018-23/01/2018 showed that there had been no water within the system over that winter period.

Table 4.1: The dates each of the loggers were in situ within the SUDS.

Dates	Swale	Pond 1	Pond 2	Baro Troll®
02/10/2014-06/11/2014		X	X	X
12/11/2014-12/02/2015		X	X	X
13/02/2015-26/03/2015		X	X	X
26/03/2015-01/05/2015	X	X	X	X
15/05/2015-15/06/2015	X	X	X	X
26/10/2015-25/01/2016	X	X	X	X
02/02/2016-02/05/2016	X	X	X	X
18/07/2016-23/09/2016	X	X	X	X
23/09/2016-24/10/2016	X	X	X	X
21/10/2016-21/02/2017	X	X	X	X
10/05/2017-10/07/2017	X	X	X	X
19/07/2017-19/09/2017	X	X	X	X
23/10/2017-23/01/2018	X	X	X	X
29/03/2018-28/05/2018		X	X	X
05/07/2018-08/10/2018		X	X	X

The location of the data loggers at the study site is shown in Figure 4.4. The BaroTroll® was placed under the footbridge at the end of Pond 2 near the datalogger.

Figure 4.4: The position of all the Trolls® (Rugged Trolls® in yellow; BaroTroll® in blue) placed in the SUDS at Upton during the study period (Google Earth, 2021e).

In addition, rainfall data were measured using a Datalogging Rain Gauge with a low maintenance tipping bucket rain collector (as explained in Chapter 3). Rainfall intensity was measured at 5-minute intervals, to complement the data obtained from the Trolls®. The rainfall gauge was not as reliable as expected and as a result the rainfall data for the entire study period was not continuous. The data were collated using *SpecWare 9 Pro* software, supplied by the rain gauge manufacturer.

Due to the large amounts of data collected over the study period (~8640 data points a month) individual months were plotted and selected results are presented in the Chapter to illustrate the hydrological responses of the system to different storm magnitudes. Events presented within this chapter (November 2015 and May 2017) represented the most interesting in terms of "flashiness" of the system, showing the stormflow transient behaviour which provided a relative measure of the magnitude of change in the water levels within the system. The rise and fall rates seen within these data represent the direct impact of rainfall on the system and allow for an insight into connectivity between different parts of the system.

4.2 November 2015

In November 2015 two rugged Trolls® (Table 4.1) were in place within the system, in both pond 1 and pond 2 as shown Figure 4.4. The data from both Trolls® and rainfall gauge are reported in figure

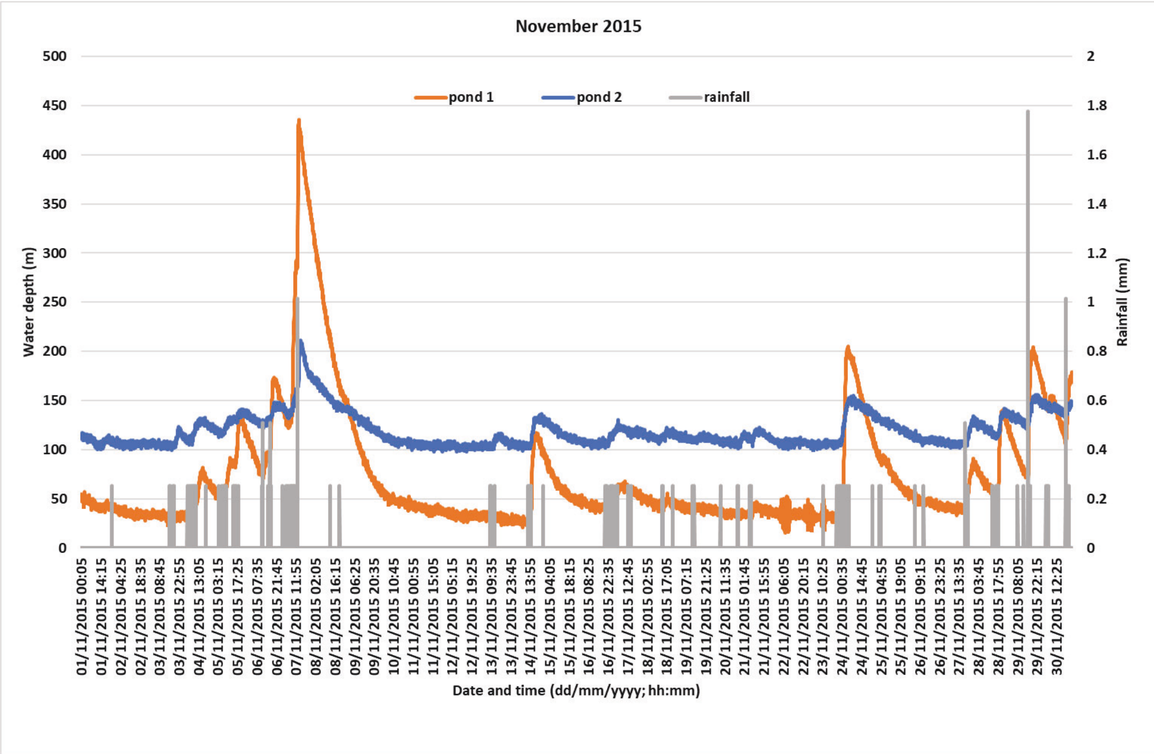

Figure 4.5: Water levels (mm) in pond 1 and pond 2, rainfall (mm) in November 2015.

Due to the large number of data points, apart from seeing trends in response to rainfall, a much higher resolution of the plot is required to demonstrate the temporal differences between the response of the swale and that of the pond with regard to the rainfall input.

The data were sub-divided into 4 sections regarding rainfall periods as shown in Figure 4.6.

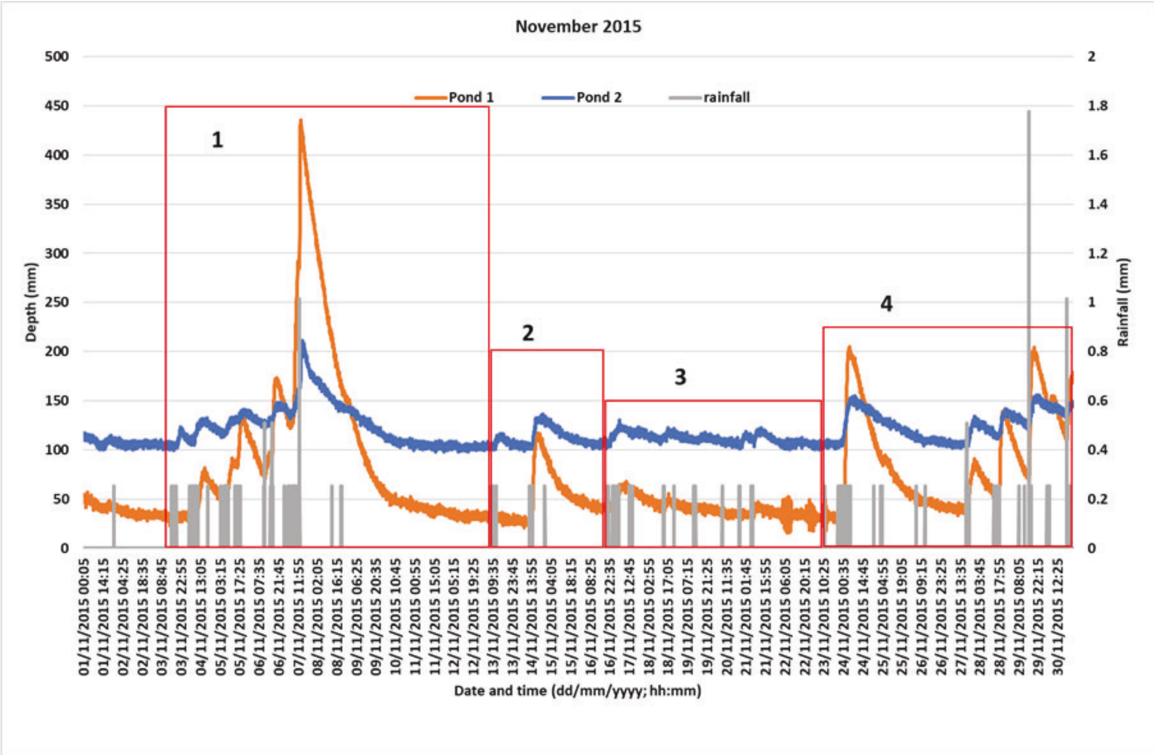

Figure 4.6: November 2015 data divided into 4 sections: 1,2,3 and 4.

Within Event 1 each rainfall period was subsequently divided into shorter time periods as shown on figure 4.7 (A, B, C, D and E). Each event starts when the rainfall was first recorded and ends before the next rainfall occurred. In this way the reaction before and after the rainfall event could be investigated in more detail (Figure 4.7).

4.2.1 Event 1

Event 1 from November 2015, between 01/11/2015 08:45 to 13/11/2017 07:30, is shown in Figure 4.7.

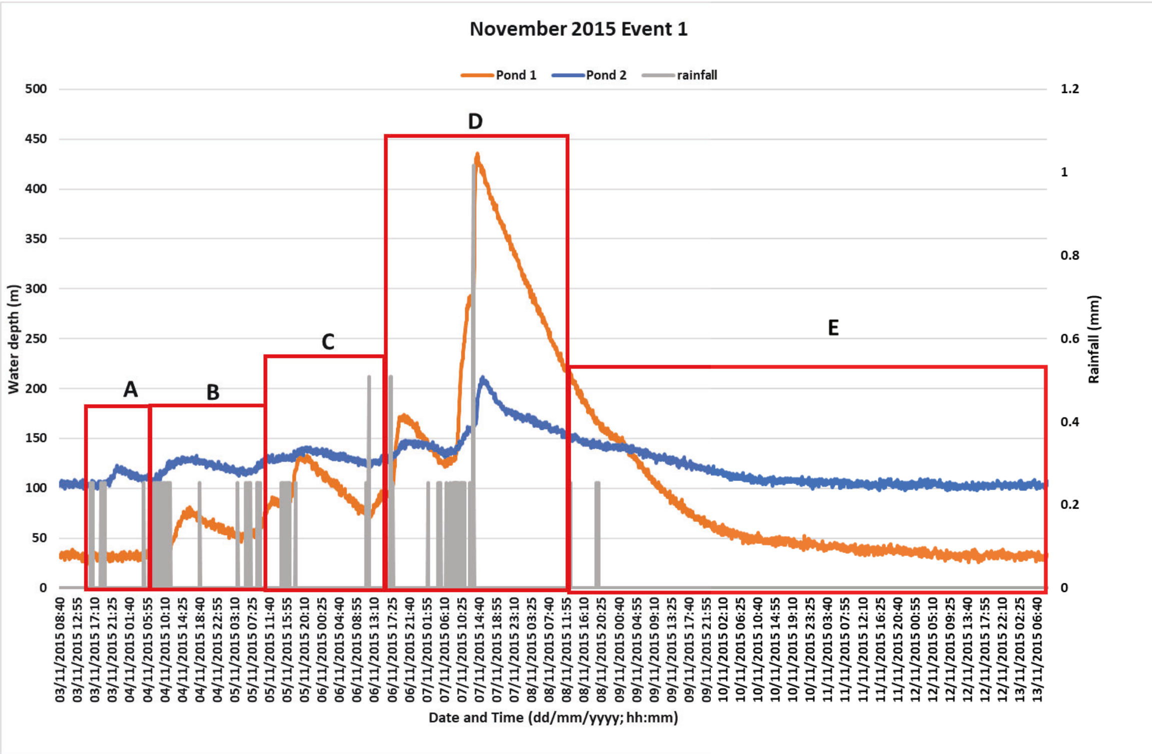

Figure 4.7: Event A (taken from Figure 4.5) 03/11/2015 08.45 until 10/11/2015 08.45.

Event A

Event A between 03/11/2015, 16:15 and 04/11/2015, 05:00, is shown in figure 4.8.

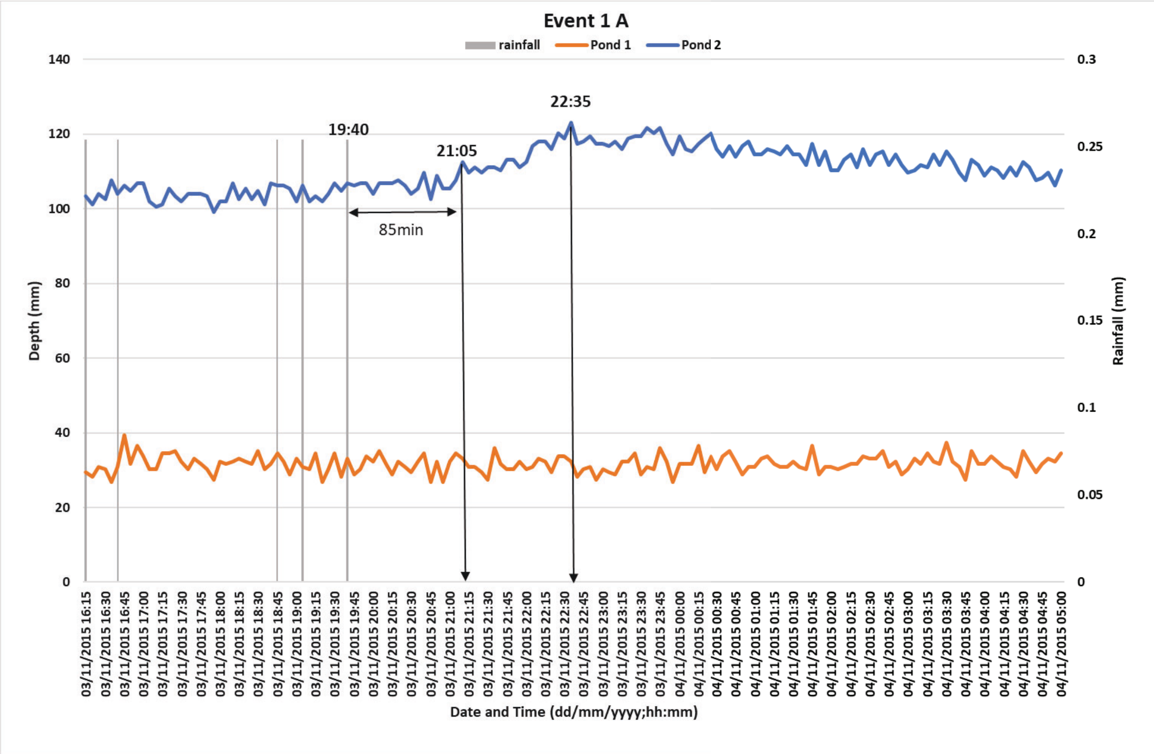

Figure 4.8: Event 1A (taken from Figure 4.6) from 03/11/2015 16.15 until 04/11/2015 05.00.

Rainfall (total of 1.27 mm) stopped at 19.40 on 03/11/2015 and pond 2 started to respond at around 21.05 until it reached a peak height (123mm) at 23.35 (a total of 150 minutes after the rain stopped). The weir at the end of Pond 2 is 120mm, in height, so it can be assumed discharge occurred into pond 3 when water levels reach this elevation (Figure 4.9). Due to the presence of a surface water pipe which provides inflow to pond 2 (Figure 3.6) and the longitudinal shape of pond 2, when water entered at the inflow to the pond it induced unidirectional flow towards the downstream weir. In contrast, pond 1 was designed for water storage therefore, the impact of water ingress affected water depth and in addition provided a large basin area for the water pool as opposed to inducing unidirectional flow. There was no indication from Figure 4.8 and the data reported here that there was an observable response in pond 1. As pond 1 has a much larger area than pond 2 it was less likely to respond to such small rainfall events, particularly if there was little or no water initially within the pond prior to the start of rainfall.

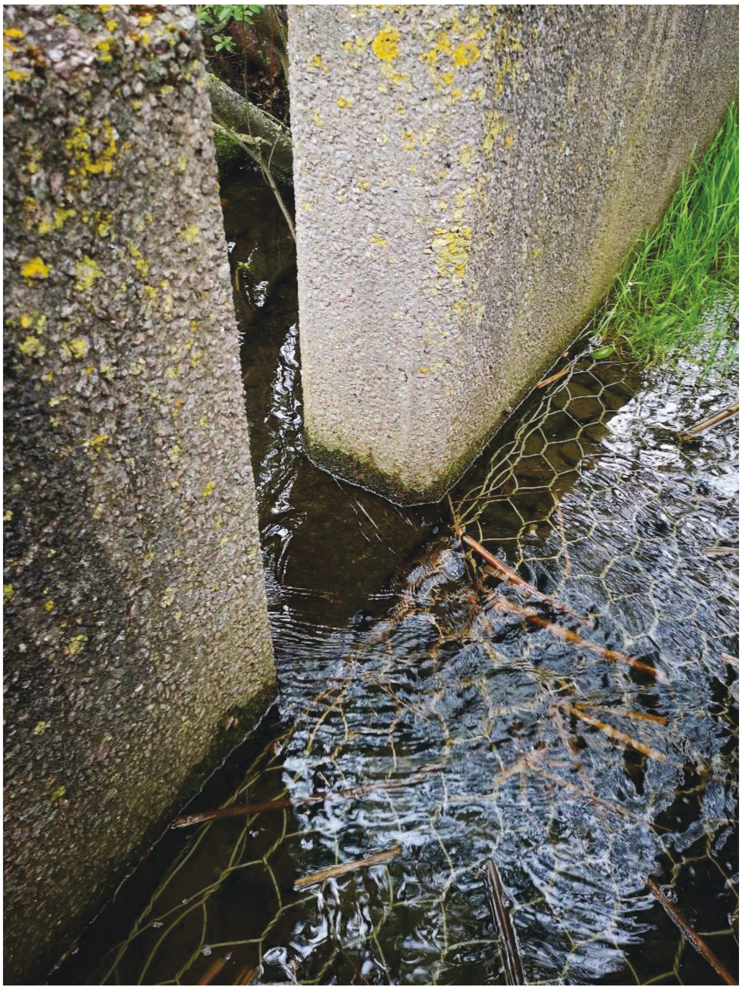

Figure 4.9: Weir at the end of Pond 2. (Copeland-Phillips, 2021)

Event 1 B

Event 1 B, between 04/11/2015, 05:55 and 05/11/2019, 03:55 is shown in Figure 4.10.

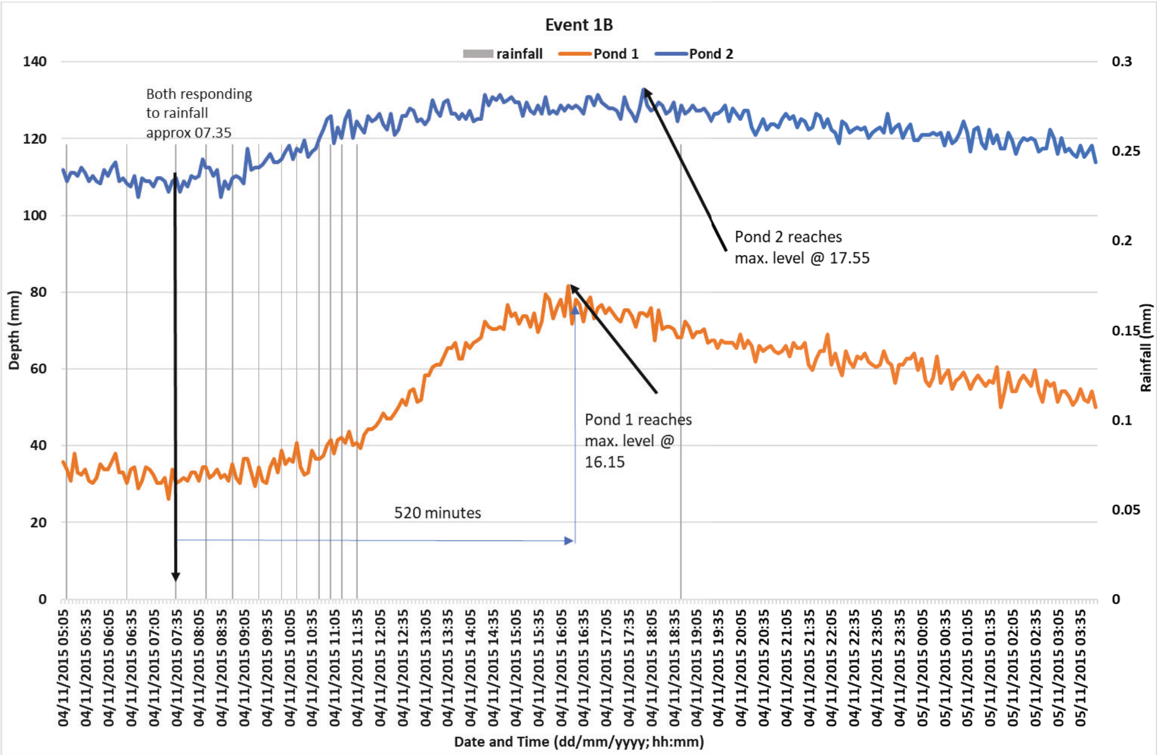

Figure 4.10: Event 1B- Water levels (mm) and rainfall data (mm) 04/11/2015 05:55-05/11/2015 03:55.

A total rainfall of 7.36mm fell between 05:10 and 11:35. Both pond 1 and pond 2 responded at the same time (07:35). Pond 1 reacted with a greater rise in water level (from ~31mmm to ~80mmm) than observed in pond 2. A positive correlation ($p<0.05$; $r=0.871$) between pond 1 and pond 2 was found regarding water levels over this time period. If there was any connectivity between the ponds a time lag would have been expected as pond 1 would have to discharge into pond 2 after reaching maximum depths during the event, therefore, creating an additional water level rise after the initial response to rainfall. The rainfall intensity at this point was not high enough to cause ac water level rise in pond 1 which would lead to overflow into pond 2.

Event 1C

Event 1C between 05/11/2015, 04:00 and 06/11/2015, 10:45 is shown in figure 4.11.

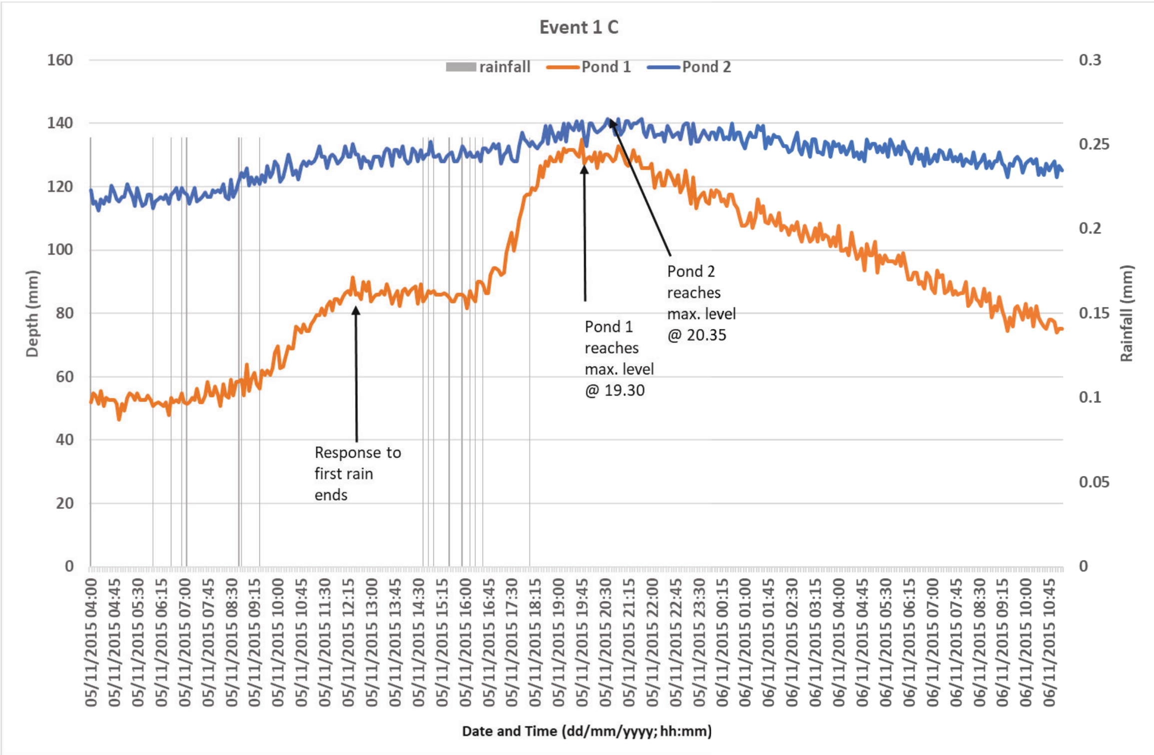

Figure 4.11: Event 1C- Water levels (mm) and rainfall data (mm) 05/11/2015 04:00 06/11/2015 10:45.

2 mm of rain fell between 04:00-09:15 and a further 2 mm fell between 14:30-18:15. Pond 1 water levels increased to a greater extent than that of pond 2 over both rainfall periods. The decline in water level was more pronounced in pond 1 after it reached a peak water level at around 19:45 05/11/2015, 100 minutes after the rain had stopped. As water levels during this event were >120mm a continuous discharge to pond 3 is assumed, considering the height of the weir (figure 4.9). This constant flow of water between these two ponds provided a marked contrast to the sharp water level increases, often called a rising limb (Konrad *et al*., 2005) observed in pond 1. The difference between the reaction in terms of water levels in the ponds from these events, provide further indication that their hydrological behaviour and roles within the SUDS were different. It is worth noting that pond 2 can maintain a more constant water level during storm runoff events, again possibly due to the unidirectional flow and the connectivity with pond 3.

Event 1D

Event 1D between 06/11/2015, 11:15 and 08/11/2015, 12:45, is shown in figure 4.12.

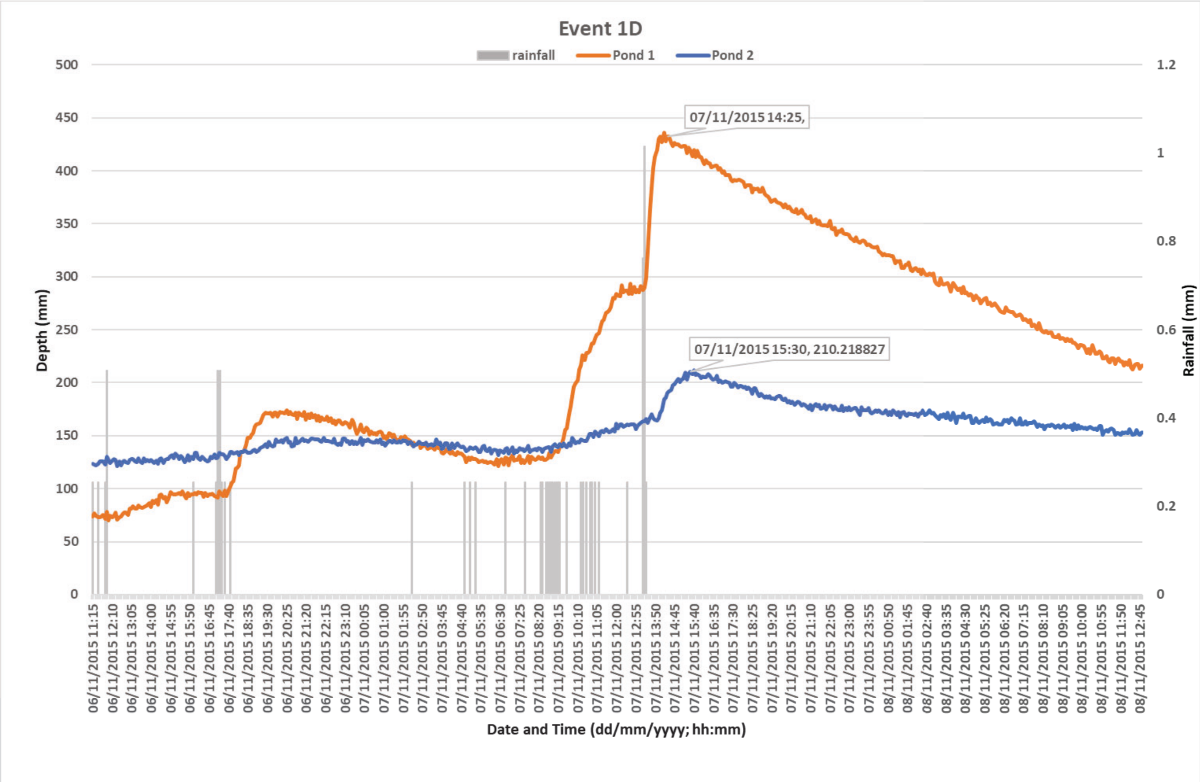

Figure 4.12: Event 1D Water levels (mm) and rainfall data (mm) 06/11/2015.

Two rainfall events are reported in Figure 4.12. On 06/11, between 11:15 and 17:40, 3.6mm of rain was recorded with a further 8.6mm recorded on 07/11/2015, from 01:55 to 13:50. Pond 1 responded quickly to rainfall with an increase in water levels in resopnse to both rainfall events. The response of the water levels in pond 1 appeared to be dependent on the intensity of rainfall, however pond 2 responded to the same rainfall with a constant rise in water level rather than a rise proportional to rainfall intensity. The position of the data logger in pond 1 was near the outflow from the vegetated swale and the observed increases in water level could have been attributed to an inflow of water from the swale (although this was not monitored at the time).

Pond 1 reached maximum water levels at 14:25, 50 minutes after the last recorded rainfall, potentially due to the inflow from the swale yet pond 2 increased in water level until 15: 30, 125 minutes post rainfall again potentially responding to water derived from the surface water inflow pipe.

Event 1E

Event 1E, between 08/11/2015, 12:55 and 13/11/2015, 07:30 is shown in figure 4.13.

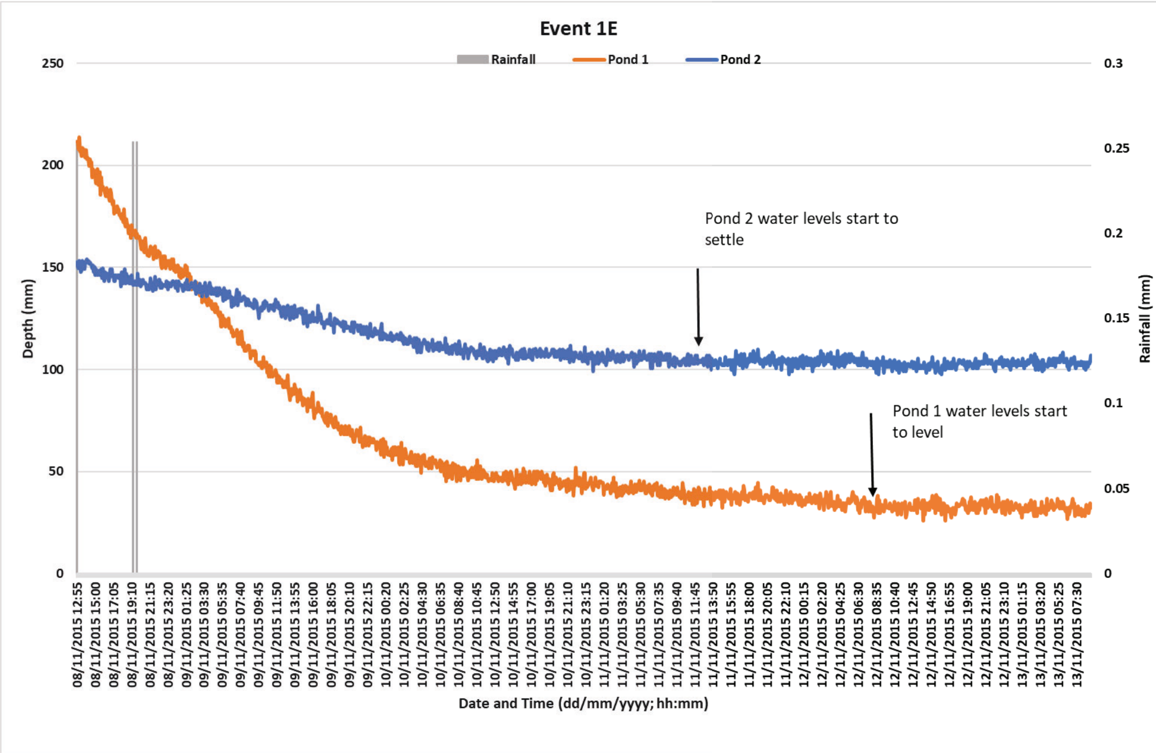

Figure 4.13: Event 1E Water levels (mm) and rainfall data (mm) 08/11/2015-13/11/2015.

0.5mm of rain was recorded between 19.20 and 19.45 but had no measurable impact, in terms of water level, on either of the ponds. Water levels declined during this period, pond 2 reached water levels <120mm, around 11/11/2015 at 11:15. After discharge to pond 3 had ceased (water levels <120mm) a more constant water level was observed. The water levels in pond 1 retruned to levels reported at the start of event 1 at 09:30, 12/11/2015 Taken in conjunction with Event 1D (Figure 4.12), the time taken to return to initialwater levels in each of the ponds appeared to take, days rather than hours to achieve. Pond 1 was observed to be more responsive in terms of the rise and fall of water levels than pond 2 at this time.

4.2.2 Event 2

Event 2, between 13/11/2015, 09:15 and 16/11/2015, 19:45 is shown in figure 4.14.

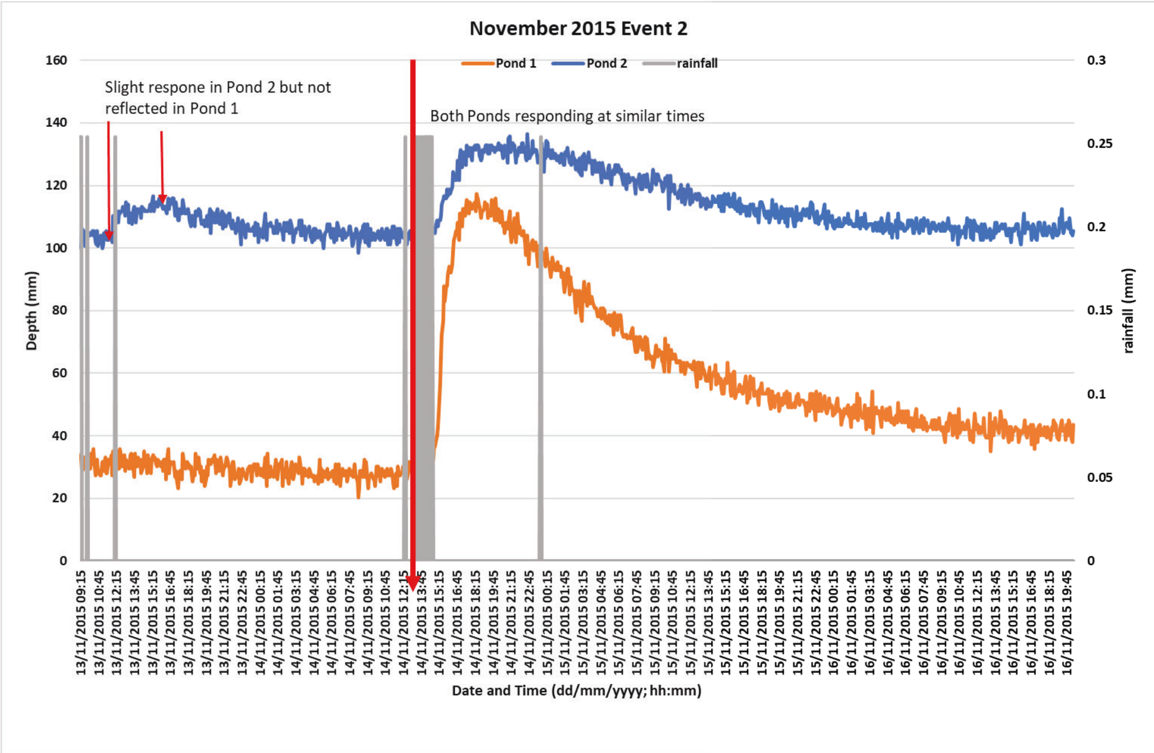

Figure 4.14: Event 2- Water levels (mm) and rainfall data (mm) 13/11/2015-16/11/2015.

1 mm of rain was recorded on the 13/11 (09.15-12.15). There was no observable impact on pond 1, however a slight rise in water levels was reported in pond 2. A further 2.7mm of rain was recorded between 13:15-14:40 on the 14/11. Both pond 1 and pond 2 water levels started to rise at a similar time of 15:00 which was in contrast to the response noted in Figure 4.12.

4.2.3 Event 3

Event 3, between 16/11/2015, 20:05 and 23/11/2015, 11:05 is shown in figure 4.15.

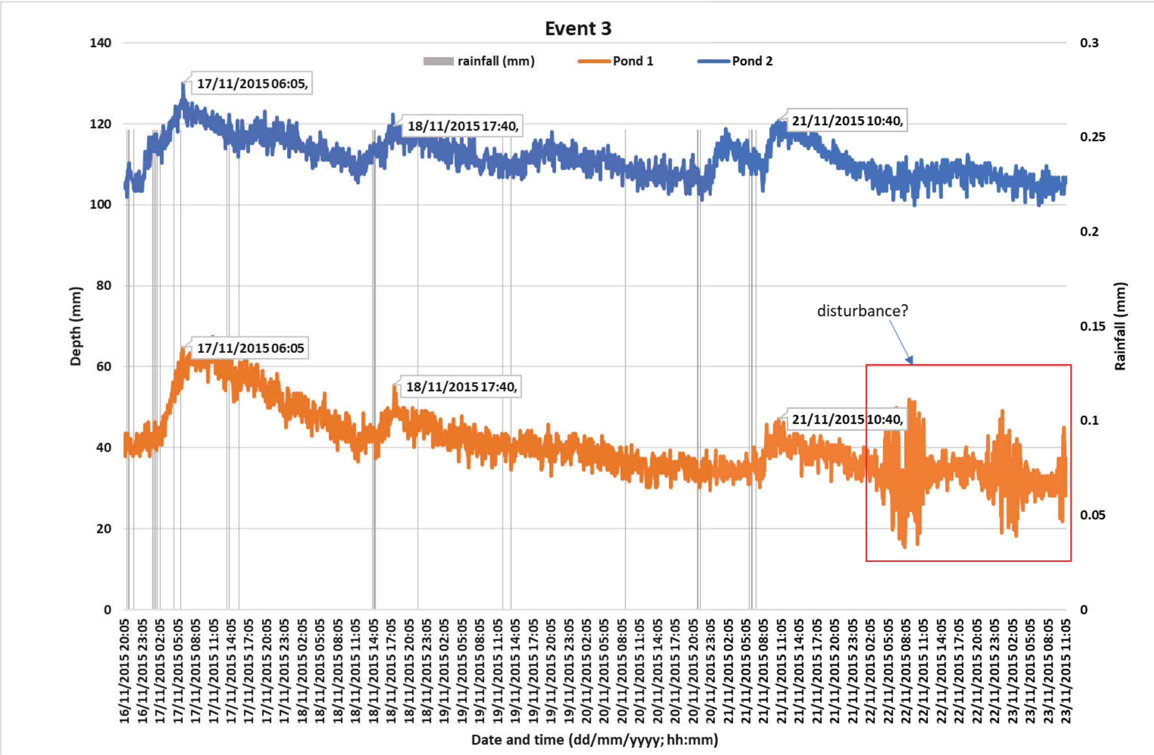

Figure 4.15: Event 3- Water levels (mm) and rainfall data (mm) 16/11/2015-23/11/2015. Interference in the data is shown in the red box.

Event 3 from 16/11/2015 until 23/11/2015 shown in Figure 4.15) illustrates the temporal synchronicity of the response of pond 1 and pond 2 to different rainfall events. A total rainfall of 7.4mm was recorded during this time with 2.3mm between 16/11/2015 20:05 and 17/11/2015 02:05 when both ponds started to respond. The water levels in both ponds were significantly correlated ($p<0.01$ $r=0.73$) again suggesting at this point that the water levels in the ponds were responding to rainfall input, and not to other external inputs (e.g., the swale). Water levels in both ponds were similar to each other. The highlighted area (shown in the red square) in pond 1 shows fluctuations in the data which cannot be explained by rainfall or slight changes in barometric pressure (this would be observed across the two data sets) but could be due to a number of factors such as disturbance by animals, especially dogs, within the pond.

4.2.4 Event 4

Event 4, between 23/11/2015 11:20 and 30/11/2015, 22:05 is shown in figure 4.16.

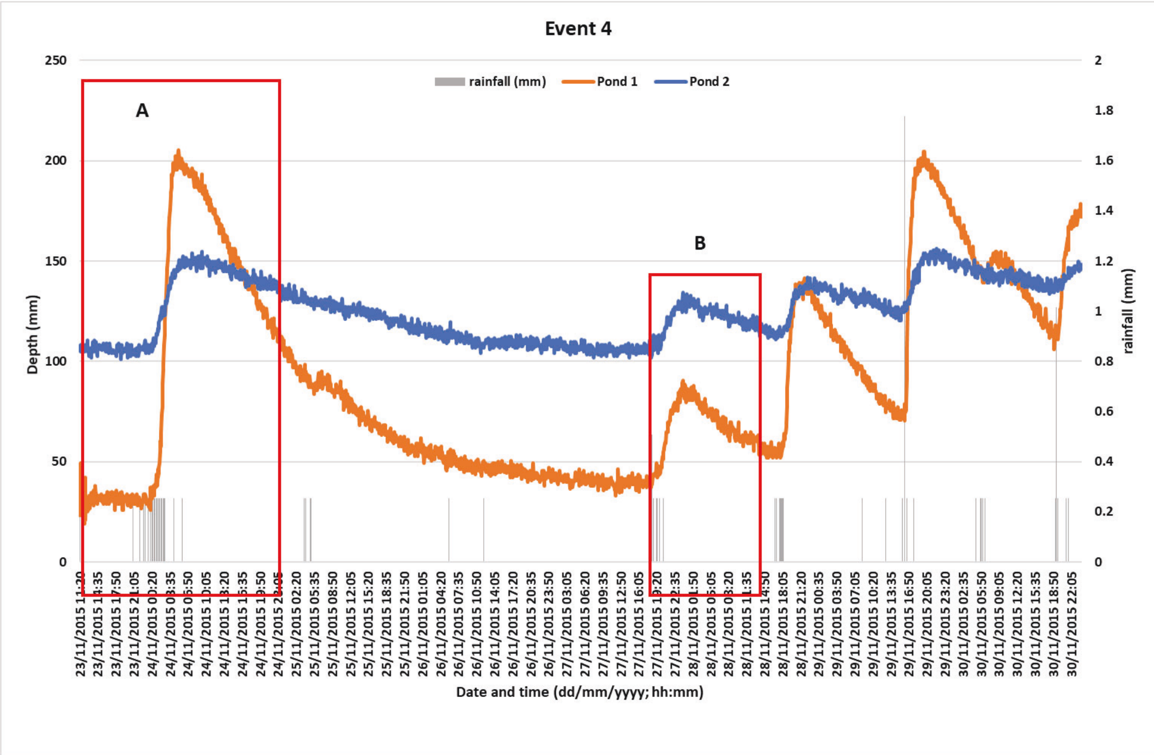

Figure 4.16: Event 4- Water levels (mm) and rainfall data (mm) 23/11/2015 11:20-30/11/2015 23:55.

Figure 4.16 again shows synchronicity between the ponds but unlike Figure 4.15 there was a difference in the timing of the peak water levels between the ponds. During this event there was also a difference in the magnitude of response in terms of water level. A total of 6.1mm of rain was recorded between 23/11/2015 11:20 24/11/2015 05:50; 1.3mm 24/11/2015 23:10 and 05:00 25/11/2015; 2.3mm 27/11/2015 18:40- 22:30; 2.5mm 30/11/2015 19:10-21:30.

Within event 4, 4A and 4B the different responses of pond 1 and pond 2 to different rainfall intensities were observed and are reported below.

Event 4A

Event 4A between 23/11/2015. 11:20 and 30/11/2015, 22:05, is shown in figure 4.17.

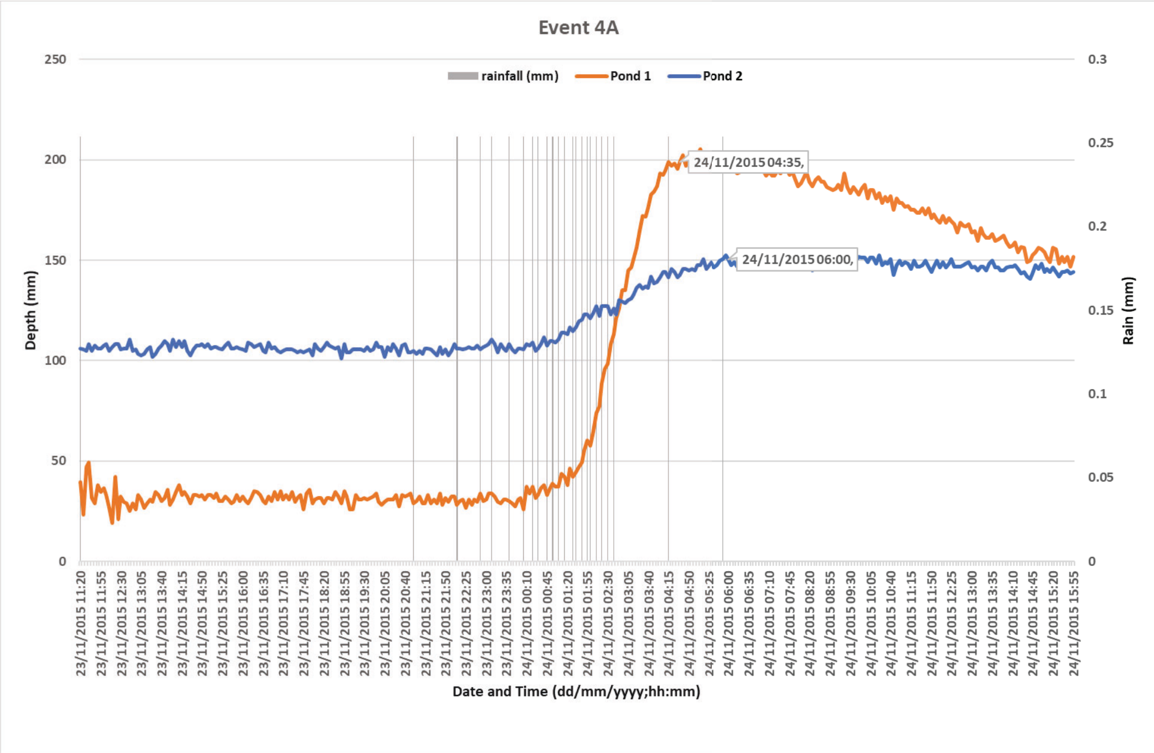

Figure 4.17: Water levels (mm) and rainfall data (mm) 23/11/2015 11:20 and 24/11/2015 15:55.

Event 4 A is expanded in Figure 4.17. The water levels in pond 1 increased to ~270mmm as opposed to only ~50mm in pond 2. This change in magnitude in response in pond 1, since event 3, could indicate that pond 1 was receiving an additional input of water from elsewhere in the SUDS (refer to Event 1D) potentially from the swale. Rainfall intensity reported in 4A (6.1mm) is less than that observed in event 1D (12.2mm). If pond 1 were receiving direct input from the swale this would also explain the shorter time taken for water level to reach a maximum in pond 1.

Event 4B

Event 4B, between 27/11/2015, 16:00 and 28/11/2015, 14:30 is shown in Figure 4.18.

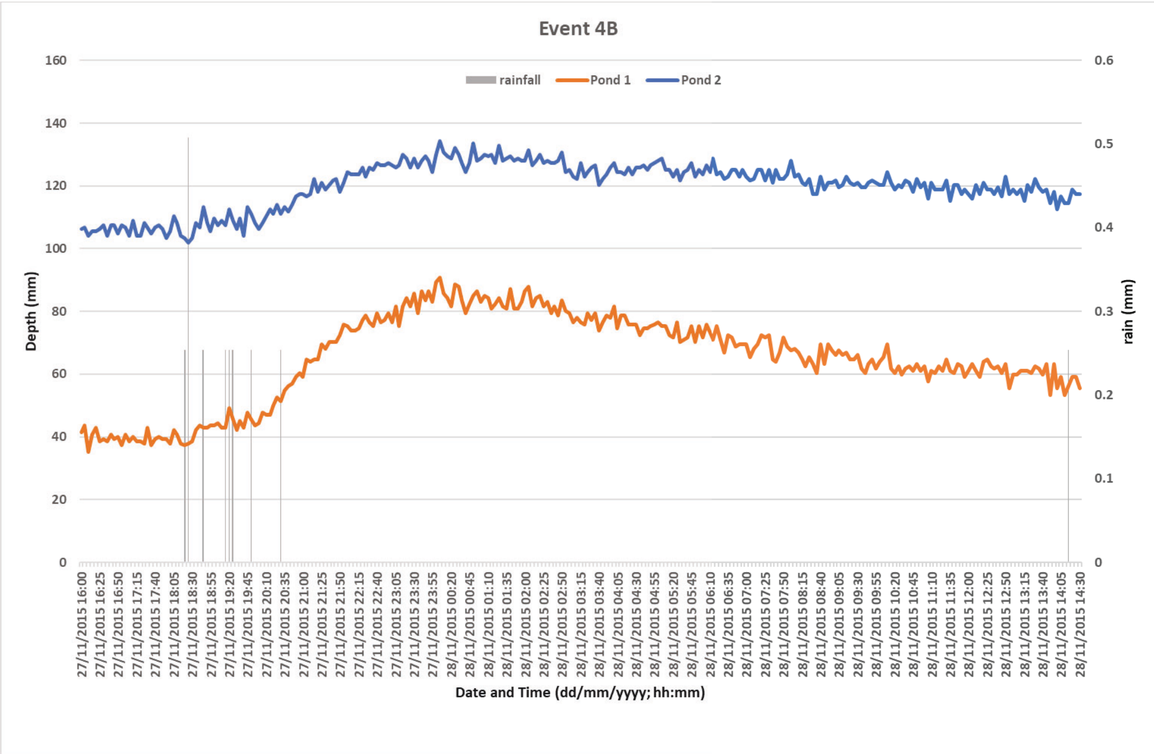

Figure 4.18: Water levels(mm) and rainfall data (mm) 27/11/2015 16:20 and 28/11/2015 14:30.

Event 4B (Figure 4.18) shows that pond 1 and pond 2 returned to time-synchronous responses to rainfall (2.3mm) and, while pond 1 reacted with to a higher water level than pond 2, rapid rising limbs (as shown in Figure 4.17) were not observed during this time.

4.2.5 November summary

From the data it is evident that pond 1 was more responsive to rainfall intensity and duration than pond 2. There is no evidence from the data that there was any connectivity between pond 1 and pond 2 during November 2015. Changes to water levels within pond 2, would have been observed on the hydrographs, that could not have been attributable to rainfall e.g., potentially delayed responses to water levels which were greater in magnitude than those that were observed during this time. However, these high-resolution data provide an insight into the rise and recession of water levels in response to rainfall events, often referred to as storm transients within the literature (Shuster *et al*., 2008).

A very rapid a rising limb can lead to increased scouring of the sediment as opposed to a more gradual rise (seen in pond 2). If the idea of storm flow transient behaviour is applied to SUDS then a direct flow through the system would be expected. However, the November data has provided an insight into the behaviour of pond 1 and the demonstration of this storm flow behaviour helped to characterise the hydrology. It also demonstrates the potential for the mobilisation of nutrients and other compounds within pond 1, which would not, due to the lack of connectivity at this point, be transferred downstream to other ponds; therefore, the concept of a "treatment train" is not applicable for the majority of the time suggesting that pond 1 was acting as a retention pond (a permanent pool which allowed deposition of sediment most of the time rather than a detention pond which is designed to help control the rate of flow) (Susdrain, n.d.).

4.3 May 2017

In terms of weather, May 2017 was the first month of 2017 with above average rainfall. April 2017 (apart from the 30th) had been an exceptionally dry month and little water was observed within the SUDS system. As the swale and pond 2 had no water the flashiness of the system was observed in direct response to rainfall. Pond 1 initially had very low levels of water. During May 2017, the response of the swale, pond 1 and pond 2 were recorded and compared (see Figure 4.19 for troll locations and Figure 4.20 for the responses).

Figure 4.19: The position of all the trolls (Rugged Trolls in yellow; BaroTroll in blue) placed in the SUDS at Upton during May 2017(Google Earth, 2021e).

The data set recorded for May 2017 was divided into five “events” as reported in Figure 4.20.

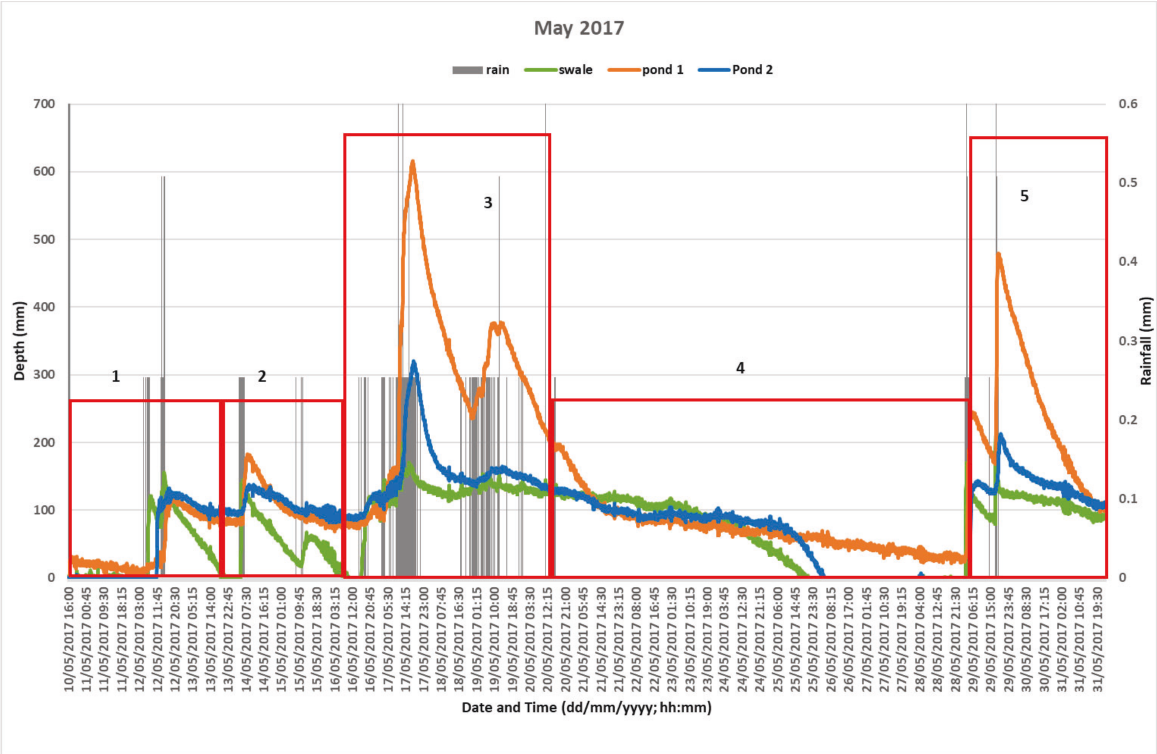

Figure 4.20: Data for May 2017 with regard to water level (mm) of the swale, pond 1 and pond 2 along with rainfall data (mm).

4.3.1 Event 1

Event 1 between 10/05/2017 16:00 and 14/05/2017 02:30 is reported in Figure 4.21.

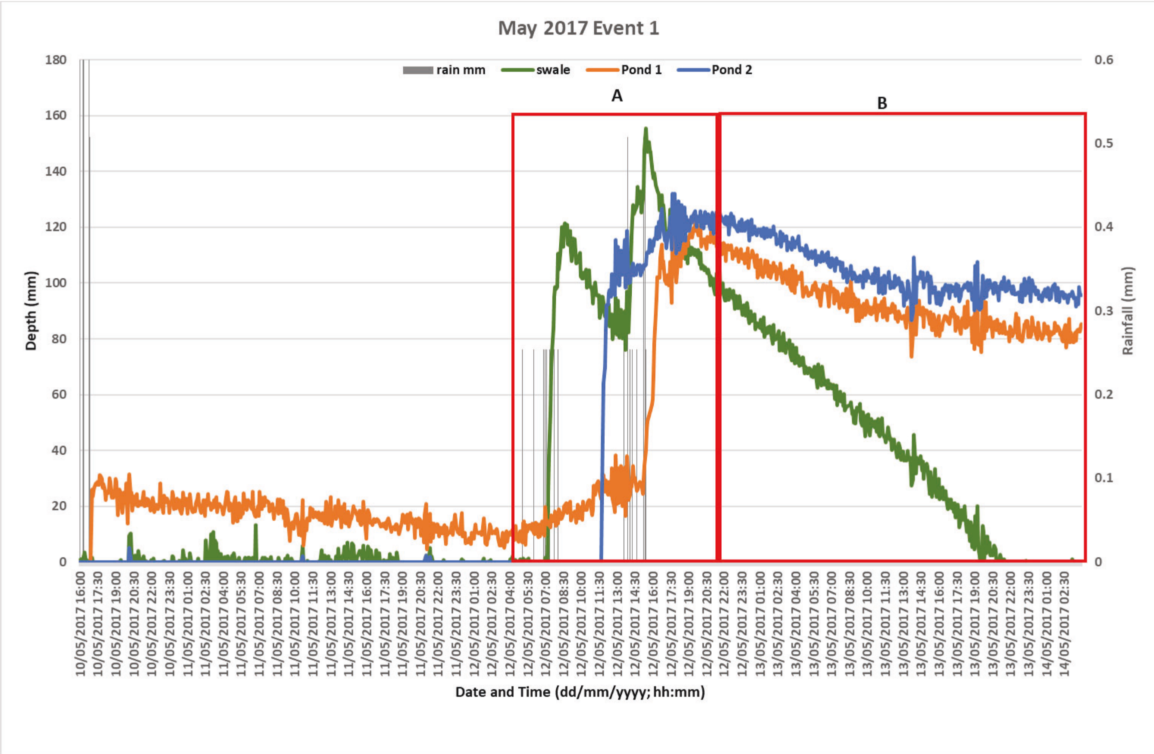

Figure 4.21: Event 1 May 2017 Water levels (mm) and rainfall data (mm).

At the start of this event the swale and pond 2 water levels were recorded at 0mm. Due to correction of the data for barometric pressure and the subsequent conversion to depth there were some negative values within the data set, but these were taken also recorded as 0mm depth values.

Event 1A

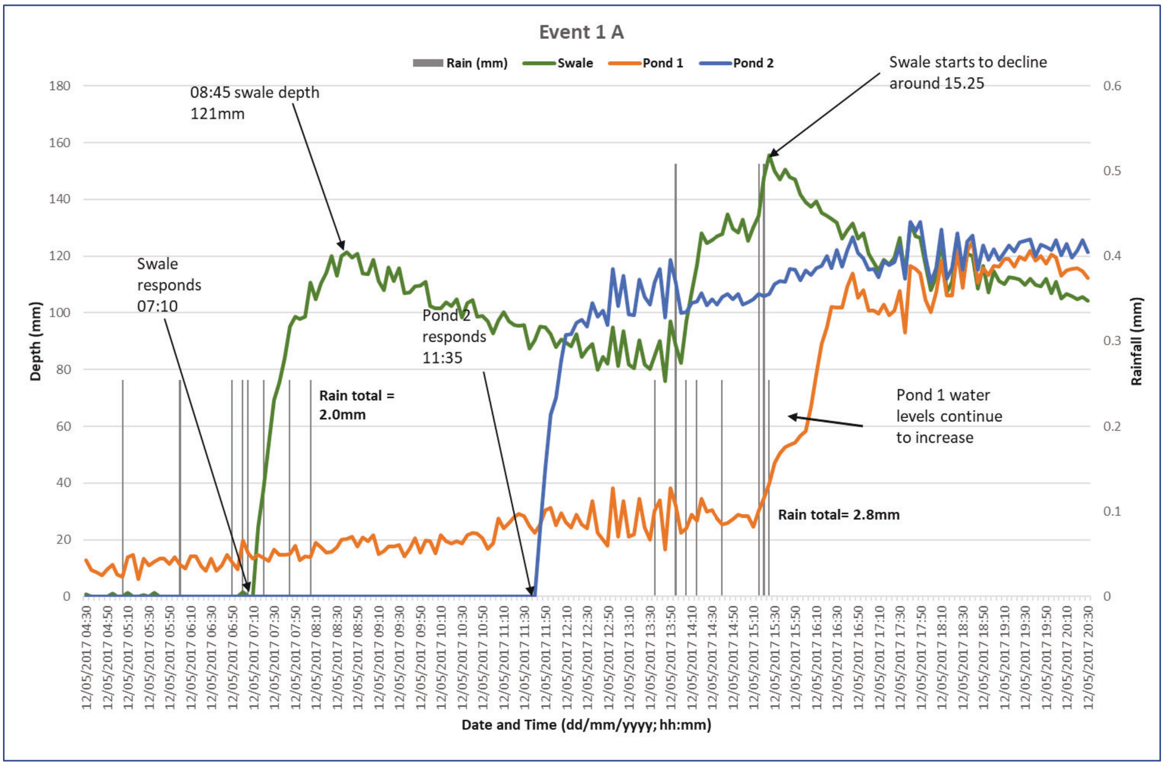

Figure 4.22: Event 1A 12th May 2017 Water levels (mm) and rainfall data (mm).

Rain fell between 05:05 and 08:05 on the 12th of May. The swale responded first at 07:10 by rising to a water level of 121mm at 08:45. From that point water levels declined to 80mm before the next rainfall at 13.30. The recession of the water in the swale (-40mm) took ~265min, which would suggest that there was no significant outflow to pond 1 at this time (given that the height of the outflow pipe within the swale is ~150mm, (Figure 4.23). Pond 1 showed a gradual increase in water levels starting at ~10mm and rising to ~20mm. Pond 2 responded at 11.35 (approx. 200 minutes after the last recorded rainfall). The water level in this part of the system is affected by the surface water drain as shown in Figure 3.6.

Figure 4.23: The swale outlet pipe at Upton. (Copeland-Phillips, 2021)

Despite the time delay between the swale and pond 2, the response showed similar trends. Both responded with a constant rate of increases in the water level over a 30-minute period (seen by the almost linear trend on the chart). These data were plotted (Figure 4.24) to show the similarities in response over a 60-minute period (swale 07:15 and 08:15; pond 2 11:45-12:45).

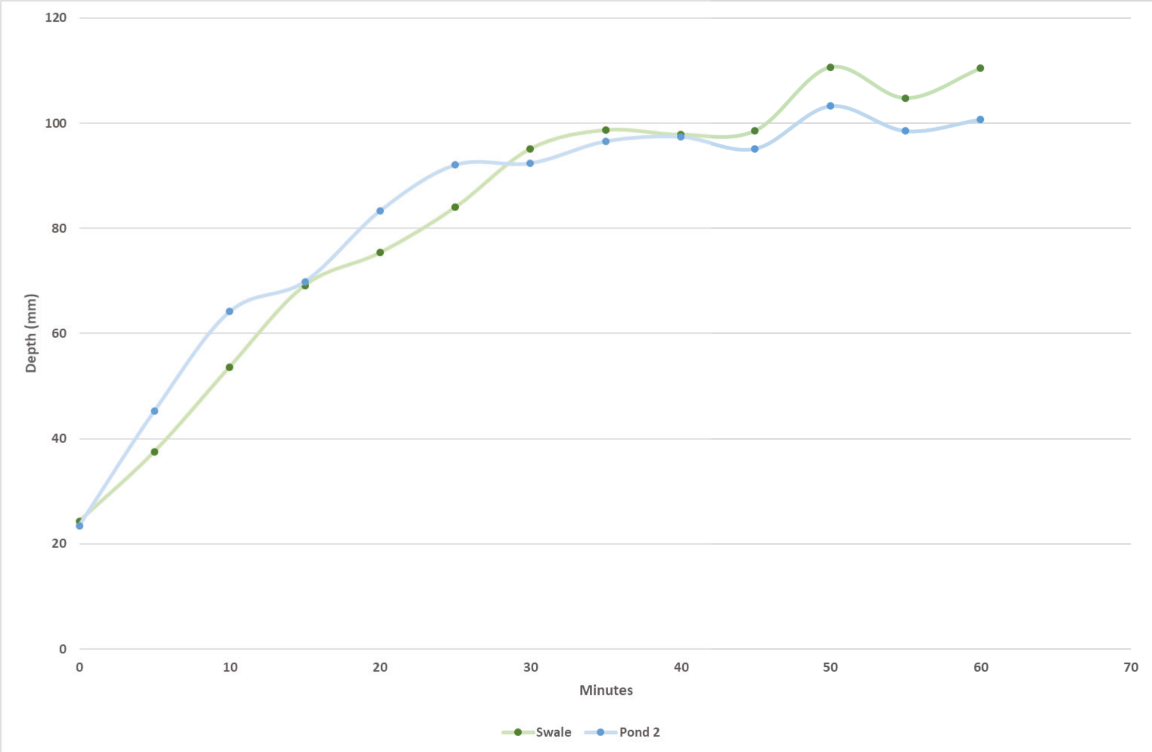

Figure 4.24: The response (depth (mm)) from the swale and pond 2 in response to the same amount of rainfall over a 60-minute time period, although data sets are 4.5 hours apart in real time.

Figure 4.24 shows that although the swale and pond 2 responded in the same way, there was a delay of nearly 4 hours and 30 minutes in the reaction time of pond 2. Both receive surface run off, but the delay was difficult to explain other than the fact that the data logger was at the outflow end of pond 2. 2mm of rain would not have been sufficient to produce fast flowing conditions in what was a dry system, so a slow rise in water level, and some infiltration, potentially accounted for this delay in pond 2. In addition, the swale was directly linked to surface road drains and therefore the input from this source was almost immediate as is reflected in the much shorter response time.

Rainfall was again recorded between 13:35 and 14:40. Barometric interference was observed in all data loggers at this time although the water levels in the swale started to rise at around 14:00. Further rainfall was recorded (2.8mm), and the swale responded with an increase in water level to 158mm. However, as the water level in the swale started to decline around 15.25 a continued increase in water level was recorded within pond 1 until 17.10. In contrast to the earlier scenario a drop of -40mm was observed in water levels from the swale, over a relatively short period of time

(~110min). The information presented here coupled with the knowledge that the discharge pipe from the swale was sited ~150mm above ground level, prompts the conclusion that unless water levels within the swale are > 150mm, water is retained within the swale. From a peak of 155mm within the swale at 15:25 a drop to 147mm (8mm decline) took only 8 minutes as opposed to 25 minutes it took for water levels to further recede a further 8mm (139mm depth @ 16:00). The nature of the data has provided an opportunity to look at short term and subtle temporal changes in the hydrology of the swale which may have been lost by smoothing the data or if data had been collected at a coarser temporal resolution. Barometric interference was recorded between 17:10 and 19:30.

Event 1B

Event 1B, between 12/05/2017, 20:30 and 14/05/2017 03:25 is shown in Figure 4.25.

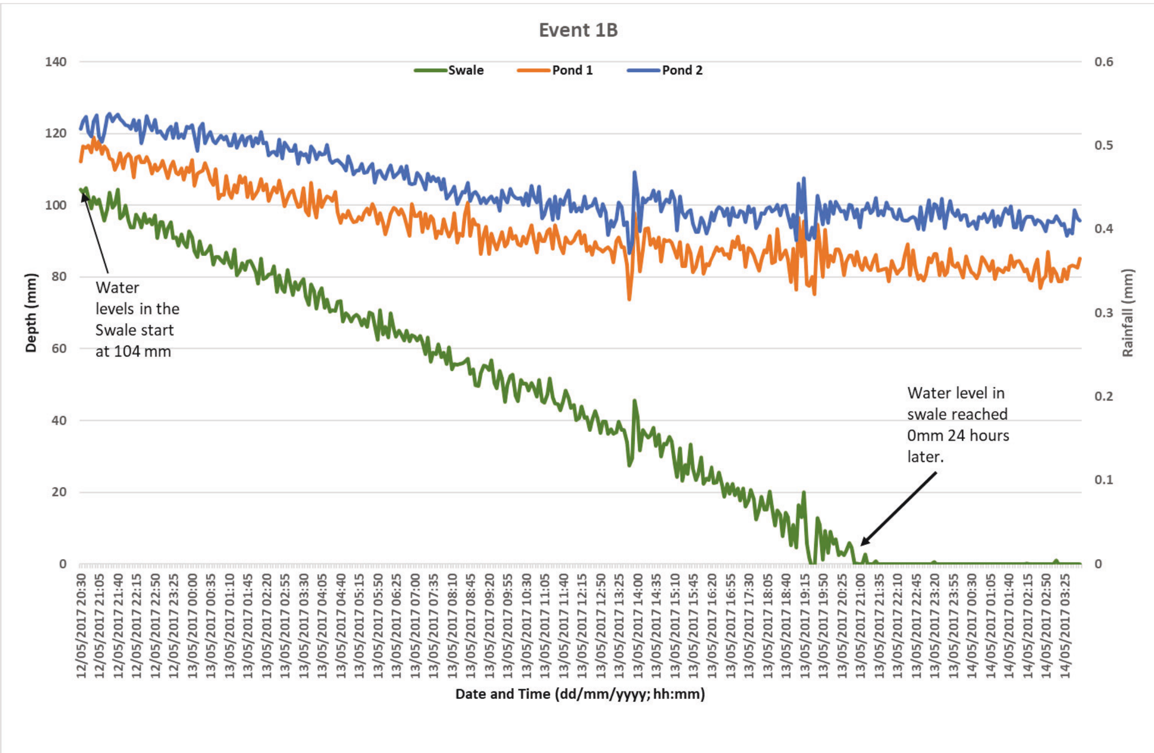

Figure 4.25: Event 1B May 12th-14th May 2017 Water levels (mm) and rainfall data (mm).

While there was no rainfall during Event 1B (Figure 4.25) the decline in water levels was recorded. In this instance pond 1 and pond 2 reacted almost synchronously in time and approximated static water levels at the same time. The swale water levels declined from 104mm to 0mm over a 24-hour period. If it is assumed that the swale did not discharge further quantities of water into pond 1, the decline in levels observed in the swale could be attributed to infiltration and/ or evapotranspiration, although the latter is less likely to be significant, as it is dependent on the nature of the vegetation.

4.3.2 Event 2

Event 2 between 14/05/2017, 04:00 and 16/05/2017, 14:30 shown on Figure 4.26.

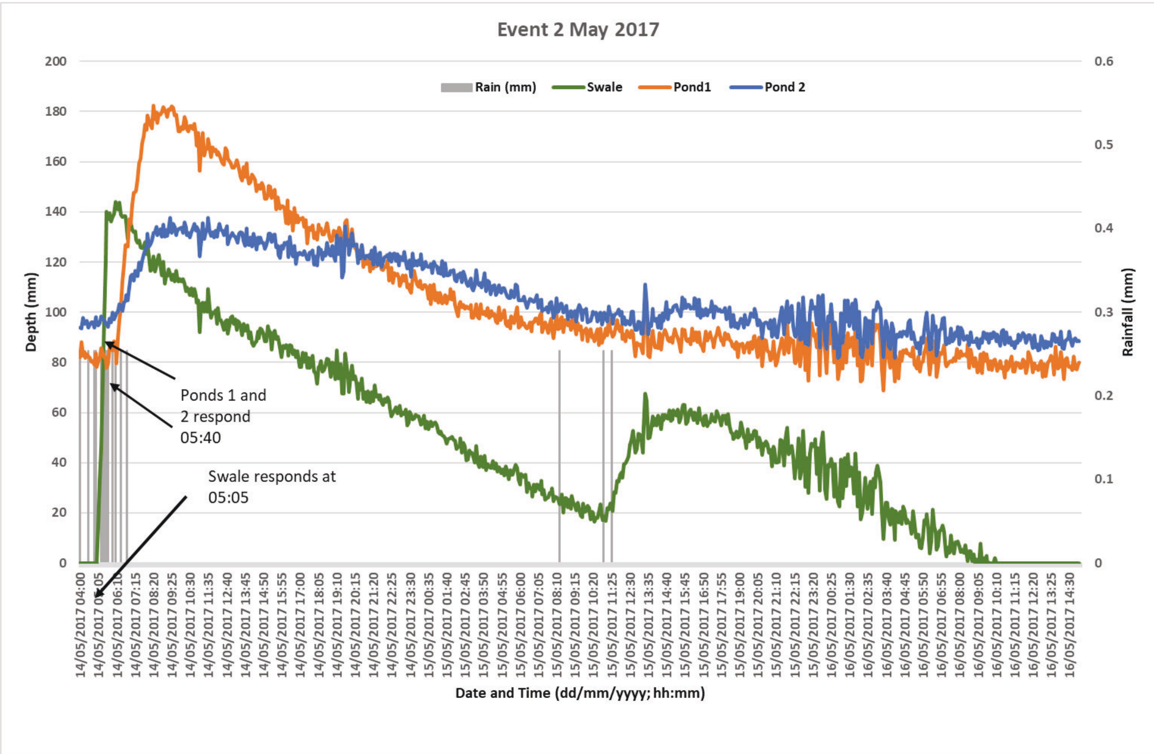

Figure 4.26: Event 2 May 2017 Water levels (mm) and rainfall data (mm).

3.56mm of rain fell between 04:00 (14/05/2017) and 06:45. The swale responded at 05:05 and increased water levels from 0mm- 143mm by 06:10. Levels declined until further rain fell at 11:25 on 15/05/2017. Pond 1 did not react until 05:40, increasing water levels from 79mm to 180mm at 09.00. The swale potentially started to discharge to pond 1 at 06:05 when the water levels started to drop as rainfall continued until 06:45 yet an observable water level decrease was recorded in the swale.

Pond 2 also responded around 05.40 and water levels rose until 09:35 although levels only increased from ~96mm to ~135mm, almost 50% less than those observed in pond 1. If the swale had discharged, due to water levels over 150mm, this again would explain the larger increase in water level recorded in pond 1.

Ponds 1 and 2 showed little response to the rain recorded on 15/05/2017 (<1mm). The rapid swale reactivity in terms of timing and water levels, even to small amounts of rainfall, is shown in Figure 4.26.

Like event 1B a decline to zero water levels within the swale no rainfall was recorded, declining from 60mm to 0mm in ~17 hours. Within event 1B a similar decline from 60mm – 0mm took around 13 hours suggesting that infiltration into the vegetated swale took a longer period of time, possibly due to previous soil wetting.

4.3.3 Event 3

Event 3 between 16/05/2017, 15:10 and 20/05/2017, 15:25 is shown in Figure 4.27.

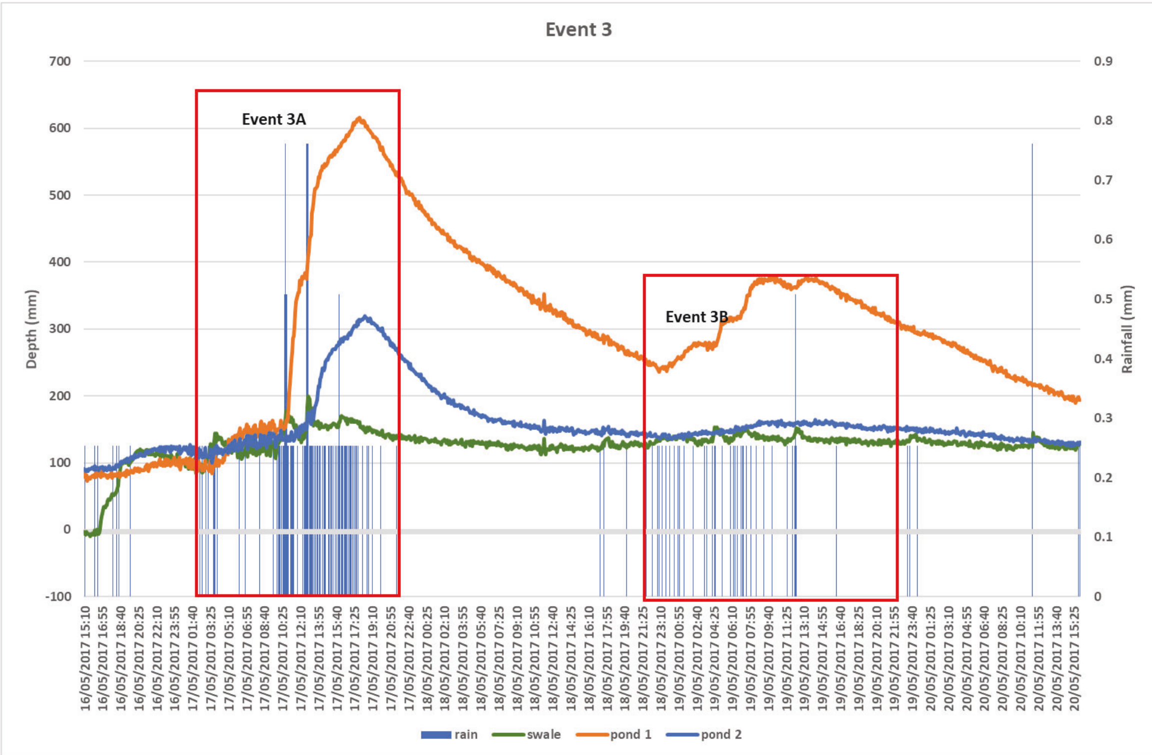

Figure 4.27: Event 3 May 2017 Water levels (mm) and rainfall (mm).

Two rainfall events were observed within Event 3. This event has therefore been sub-divided into Event 3A and 3B to enable a more detailed reporting of the responses.

Event 3A

Event 3A occurred between 17/05/2017, 02:20 and 17/05/2017, 21:30 and is shown in Figure 4.28.

During this event there was rainfall of 24mm between 02:20 and 21:30 on 17/05/2017.

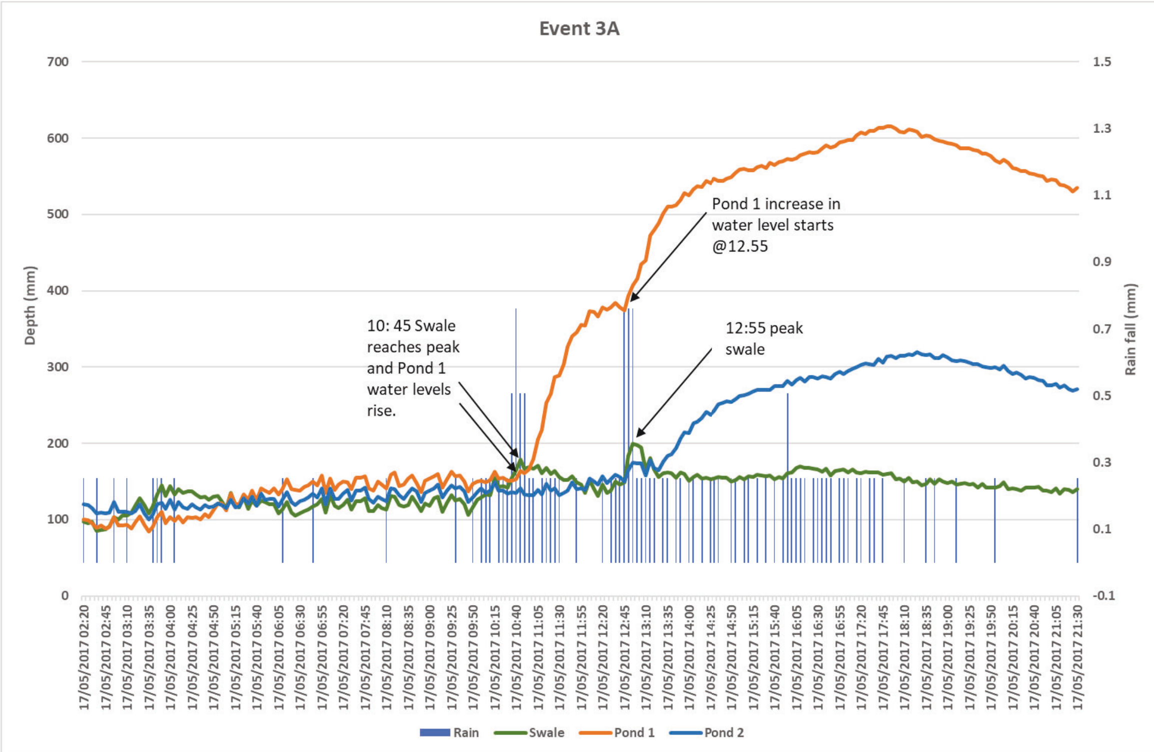

Figure 4.28: Event 3A May 2017 Water levels (mm) and rainfall (mm).

The swale, pond 1 and pond 2 had similar water levels at the start of the rainfall for a period of ~2.5 hrs. The swale started to respond at around 09:40 and reached a peak level at 10:45 of around 178mm. Assuming connectivity at this point (between the swale and pond 1 due to high water levels), the decline in levels within the swale (dropping to 145mm during reduced rainfall up to 12.15) corresponded with the increased water level in pond 1 (161mm @ 10:50 to 366mm @ 2:15). Further increases in the swale water levels were recorded when the rainfall started again at 12:20. This time water levels in the swale reached 199mm. As soon as the swale water levels started to decline a further increase in water levels in pond 1 was recorded (374mm to 525mm at 14.00). The water levels in the swale started to level out (~ 158mm) at 14:00, being maintained around this level due to the continued rainfall. The pond levels continued to rise until a level of 615mm was reached at 17:55. The rain became more sporadic at 18.20 and all levels within the ponds and the swale began to decrease.

Pond 2 appeared to respond to rainfall although there were no rapid changes in water levels unlike those observed in the swale and pond 1. The response of pond 1 and pond 2 between 14: 10 and 21:20 was time-synchronous and significantly positively correlated ($p<0.05$).

Event 3B

Event 3B between 18/05/2017, 21:25 and 19/05/2017, 21:25 is shown in Figure 4.29.

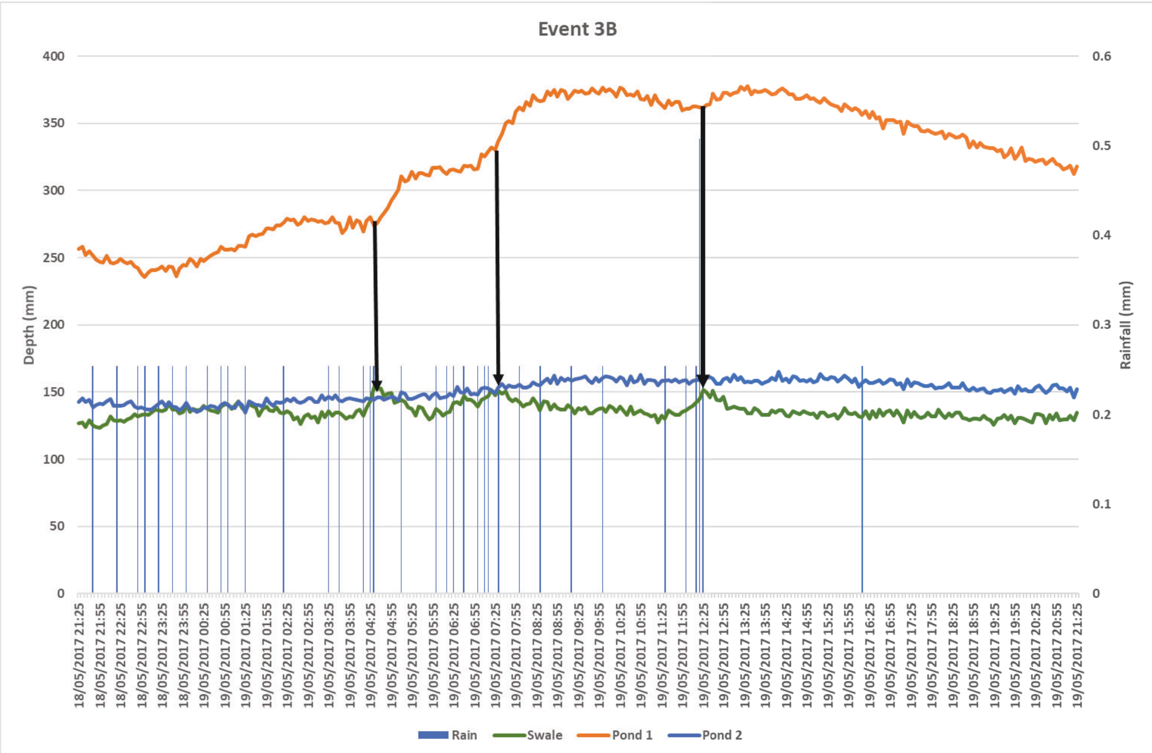

Figure 4.29: Event 3B May 2017 Water levels (mm) and rainfall (mm).

Rainfall continued during this period (18/05/2017 21:25- 19/05/2017 21:25) and consistent water levels in pond 2 were recorded. The water levels within the swale rose and fell with changing rainfall intensity steady increases were recorded in pond 1. The black arrows on Figure 4.29 highlight times when there were observed increases in the swale water levels which corresponded to a further rise in the water levels of pond 1 in a similar way to that described Event 3A, possibly indicating discharge occurred from the swale directly into pond 1.

4.3.4 Event 4.

Event 4 between 20/05/2017, 16:05 and 29/05/2017, 00:45 is shown in Figure 4.30.

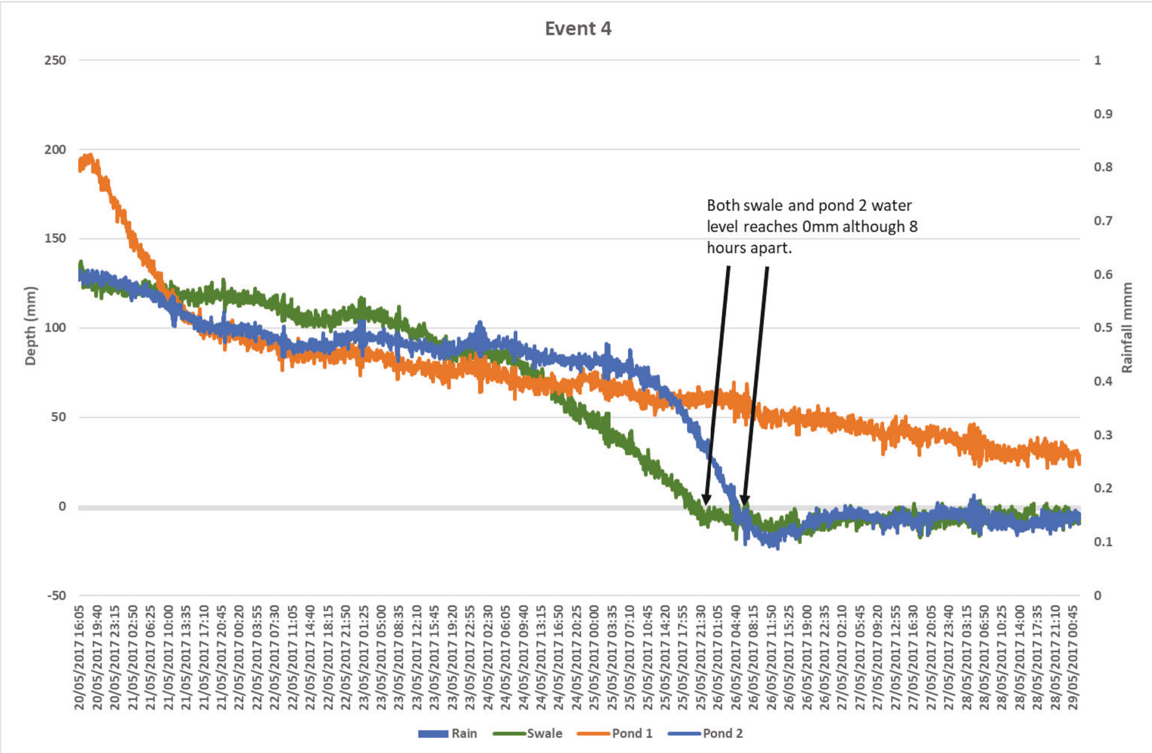

Figure 4.30: Event 4 May 2017 Water levels (mm) and rainfall (mm).

During Event 4 there was no rainfall, so the water levels in each part of the system declined. Water within the swale decreased to 0mm by 25/05/2017 at around 21.00, 6 days 4 hours and 45 minutes after any rainfall. Pond 2 water levels reached 0 mm by 26/05/2017 around 05:00 (6 days 12 hours after rainfall) and Pond 1 levels continued to decline although it retained some water (depth ~28mm).

4.3.5 Event 5

Event 5 between 29/05/2017. 02:35 and 31/05/2017, 23:10 is shown in Figure 4.31.

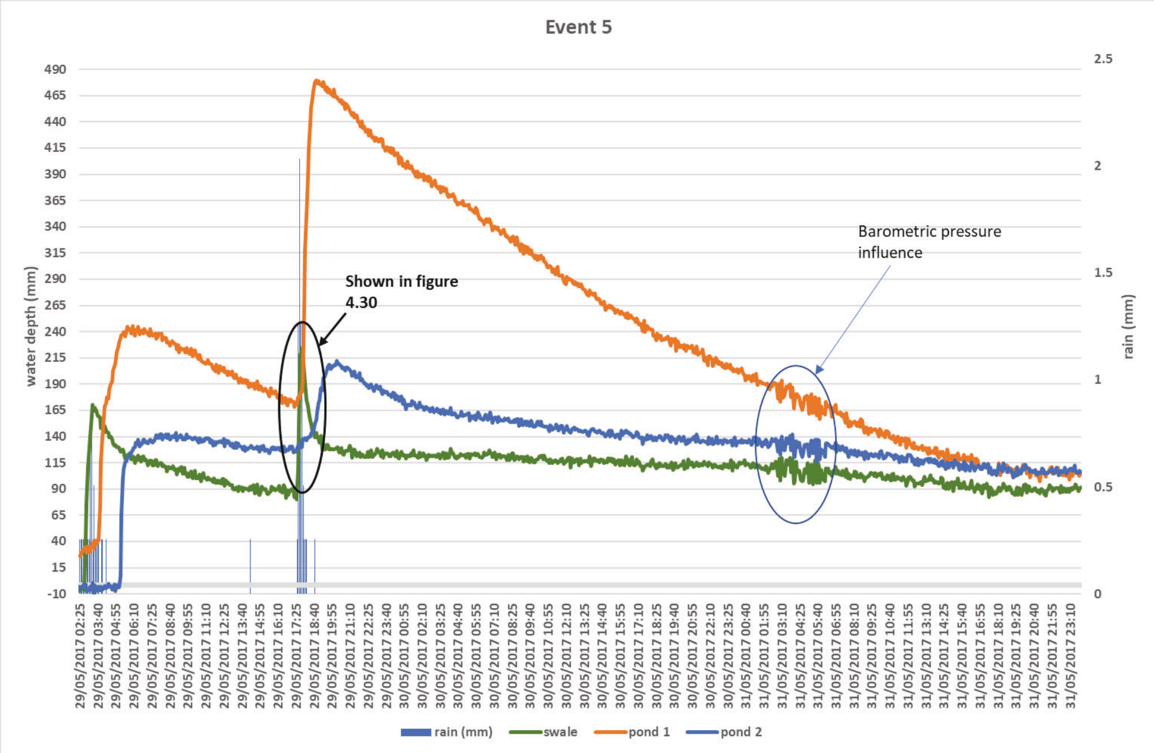

Figure 4.31: Event 5 May 2017 Water levels (mm) and rainfall (mm).

At the beginning of this event, 6.35 mm of rain was recorded between 02:25-04:15 with a further 7.62mm between 17:30-18.45. The swale reacted at around 02:50 and reached a maximum level of 167mm by 03:30. The water level declined until the second rainfall event at 17:30 and water levels reacted immediately by rising from a level of 80mm to 109mm at 17:35 and up to a maximum of 219mm at 17:55. When the rainfall intensity decreased, the swale levels declined to 140mm within 50 minutes. The water levels within the swale continued to decrease reaching 90mm by 31/05/2017 23:55, a period of 52 hours after the cessation of rainfall.

Pond 1 also reacted to rainfall although not until 03:45. Like the responses reported in Event 3 when the water levels in the swale declined, an increase in water levels was recorded in pond 1. The drop in water levels in the swale and the consequent rise is shown in more detail in Figure 4.32.

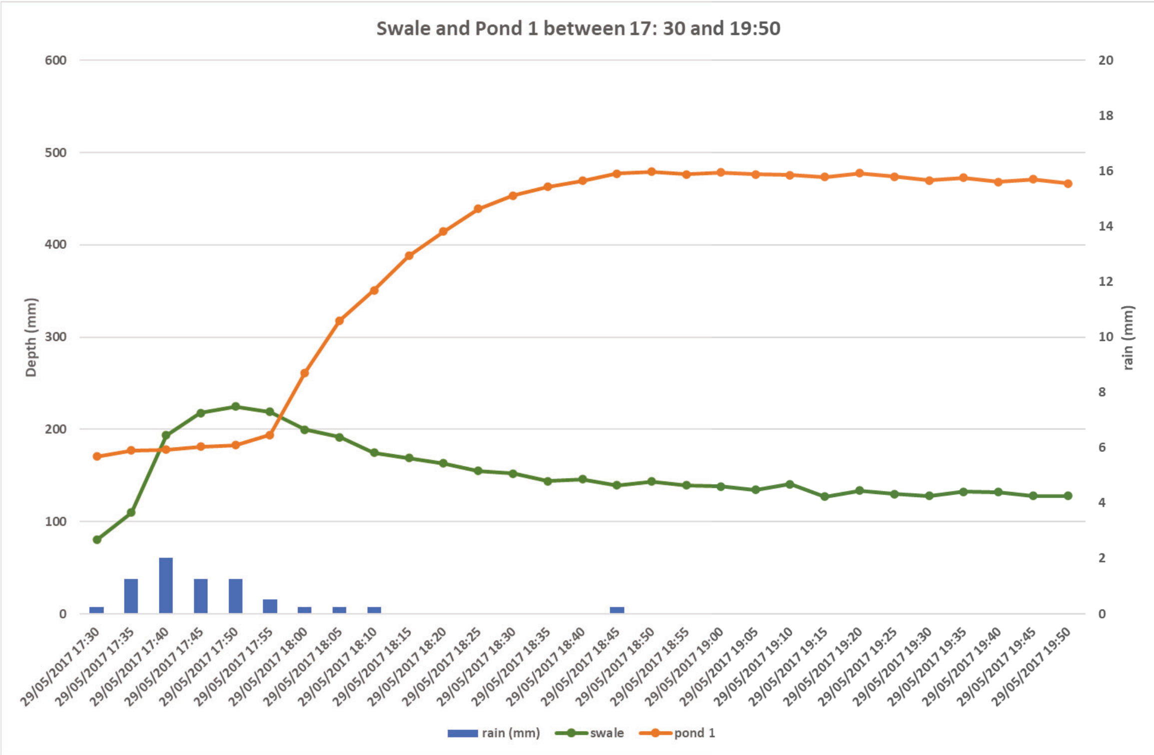

Figure 4.32: Event 5: 17:30- 19: 50, 29 May 2017 Water levels (mm) and rainfall (mm).

Between 17:35 17:40 the pond levels increased from 109mm to 177mm. At 18:00, as the levels in the swale decreased, water levels rose from 193mm to 261mm within the pond. As previously suggested, this was most likely due to discharge from the swale into pond 1.

In Figure 4.31 Pond 2 had shown a delayed reaction to the first rainfall with water levels not rising until around 04:55, nearly 2 hours after the rainfall started and 40 minutes after the rainfall stopped.

4.4 Notable Events.

Data were collected between 2015-2018 and using November 2015 and May 2017, as an indication as to the behaviour of the system, the hydrographs across all the dates were produced and examined for potential connectivity between pond 1 and pond 2. One event was identified in November 2016 which is shown in Figure 4.33.

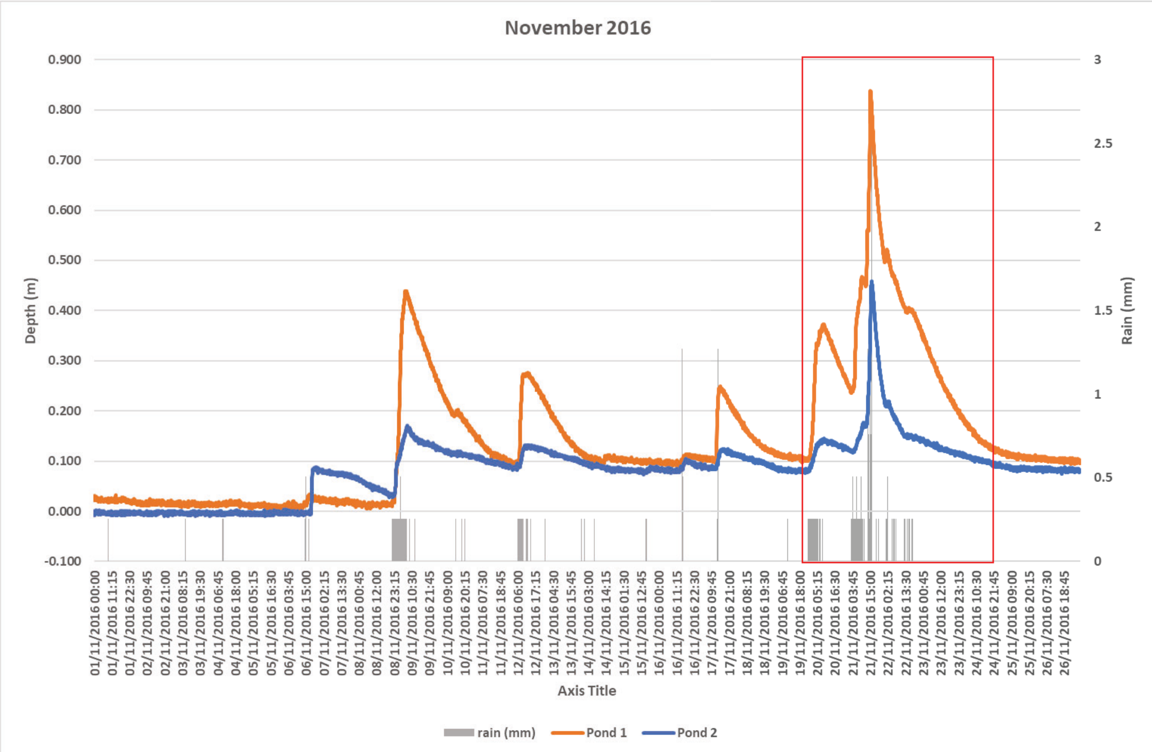

Figure 4.33: November 2016 between 01/11/2016 00:00 and 26/11/2016 18:45.

Figure 4.33 shows that water level rose in pond 1 on 21/11 of up to ~0.84m. This event is shown in more detail in Figure 4.34.

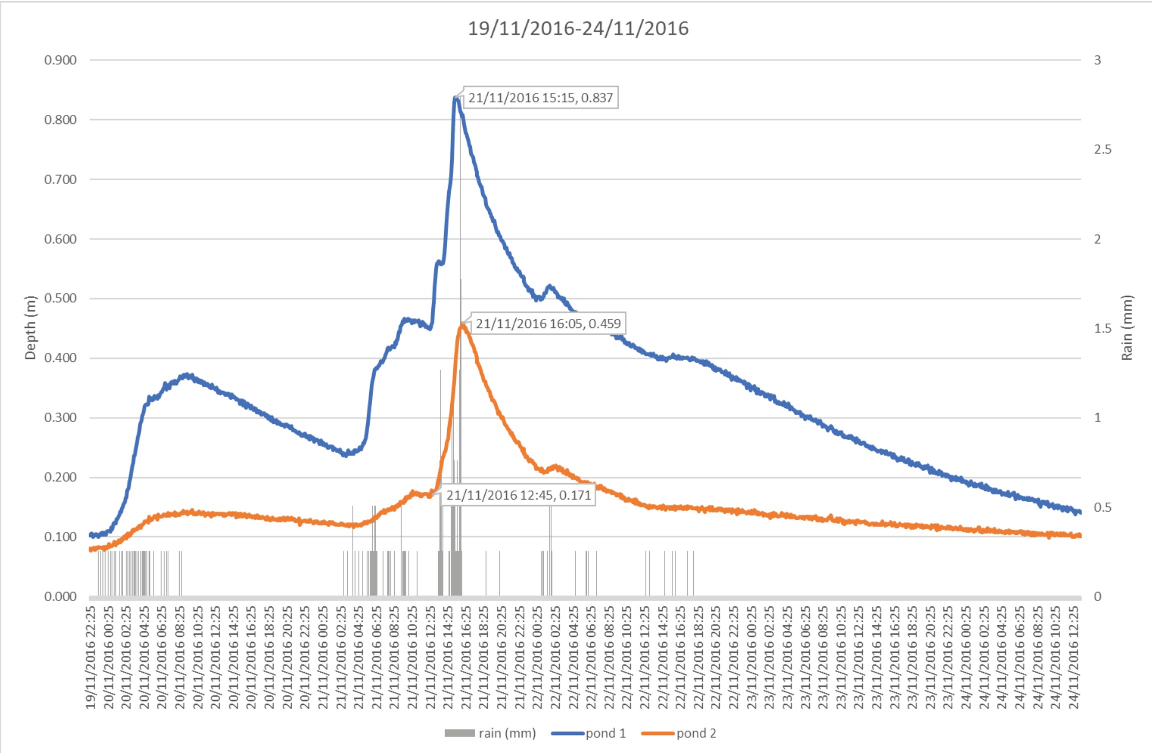

Figure 4.34: November 2016 between 19/11/2016, 22:25 and 24/11/2016 12:25.

In November 2016 there was a total rainfall of 75mm of rain with 35.5mm falling between 21/11/2016 at 02:25 and 22/11/2016 at 18:25. As shown on Figure 4.33 a rapid rise in water levels in pond 1 and pond 2 was observed ~ 13:20 (21/11) in response to the 17mm of rain that fell between 13:20 and 16:00. A rapid rise in water levels within pond 1 and pond 2 had been observed in previous data sets but, from the original designs of pond 1, it wass estimated that the levels would need to reach around ~0.8m in order connect with pond 2. In Figure 4.34 pond 1 levels reached ~0.837m,the highest levels observed during the study period. After only ~45 minutes a water level of ~0.4m were recorded in pond 2. The synchroneity in the storm transient rising limbs between the 2 ponds and the short temporal nature of these changes indicated that only after a significant amount of rainfall would the conveyance of water within the system be expected. The potential connectivity between the ponds at this time, was demonstrated by the sharp rising limb in Pond 1, only lasted for a few minutes before a rapid decline in the water was observed.

4.5 Discussion

The water levels in pond 1 were clearly influenced by inputs from the swale, by rainfall and by other local inputs. Pond 1 was originally designed as a wetland with multiple inputs as explained in Chapter 3.1. The wetland was designed to have varying water depths and now has varying depths of sediment (see Chapter 5). Pond 2 received one input and was connected, via a weir, directly to pond 3 which was similarly connected to pond 4. While referred to in this research as ponds, both Ponds 1 and 2 are essentially designed as wetlands because they comprise of marshy areas and are covered in aquatic vegetation. The purpose of such a wetland is to detain flow for a period of time and allow sediments to settle out at low water velocities (<0.1 m s^{-1}) (Woods-Ballard *et al*., 2007).

The data from November 2015 (Figure 4.5) shows that the responses in ponds 1 and 2 to rainfall are similar although the magnitude of the response (in terms of water level) differs. The greater increase in water levels in pond 1 could be a result of two factors; the slow movement of water within pond 1 itself, or the input of water from the swale if the rainfall intensity was great enough to cause discharge from the swale. The rate of rise of water levels can be affected by a number of attributes that alter the time frame in which storm runoff is delivered to the channel (Shuster *et al*., 2008). Direct evidence for discharge from the swale was reported in May 2017 when a data logger was placed in the swale and water levels were subsequently monitored (e.g., Figure 4.22). The hydrological connectivity between these two components occurred frequently during rainfall events but the connection between the two components was short lived due to rapid discharge to the swale. Importantly no connectivity was reported if the water levels, within the swale, remained below 150mm. The quick responsiveness of the swale was demonstrated by an increase in water levels from 0mm to 120mm with as little as 2mm rain over 3 hours which produced a 121mm rise in water levels in a 95-minute period. The total area of impermeable surfaces contributing to the swale is ~2900m^2 (Chapter 3.1) and it is the extent of such surfaces, directly connected to the swale, which can lead to faster conveyance of greater quantities of storm runoff (Shuster *et al*., 2008) resulting in a greater magnitude of water level rise. Rapid rising limbs, as reported in some of the hydrographs, can cause increased transportation and remobilisation of nutrients and sediment in streamflow (Royer *et al*., 2006). The longitudinal nature of pond 2, and the fact that the inputs are only at one end, could provide a greater opportunity for infiltration prior to water reaching the data logger, which could provide an explanation for the differences in magnitude (e.g., Figure 4.11, Figure 4.25).

On some occasions (e.g., Figure 4.14) the synchronicity between pond 1 and pond 2 in response to rainfall is observed. This synchronicity occurred when rainfall intensity was low, approximately <2mm occurring over a >60min period. However, this is dependent on the previous and subsequent amount of rainfall. The impact of the rainfall on water levels in pond 2 sometimes lagged behind the response reported in pond 1 (Figure 4.10) but this was not a consistent pattern (Figure 4.8). The location of the data logger in pond 2 (at the outflow end of the pond and at the furthest point from the input) and could have resulted in a delay of water movement through the pond giving rise to a lag in the recorded data.

The decline in the water levels after rainfall was also recorded (e.g., Figure 4.13) and provided an insight into the dispersal of water in each pond and the swale (Figure 4.25). The rate of recession of the water depends partly on the extent of the hydraulic connectivity between the surrounding landscape and the SUDS (Shuster *et al*., 2008). The water levels in the swale declined much more rapidly than those ponds 1 or 2 (Figure 4.25) particularly in the summer months after a period of no rainfall. Evaporation was considered to be relatively unimportant for modelling individual storm events but with regard to long-term water balance studies it could be an important contributor to water loss (Wang *et al*., 2008). In such densely vegetated systems, evapotranspiration would also be of significance. There was a greater capacity for dispersal and possible infiltration in pond 1 compared to pond 2, due to the volume and size differences and a quicker decline in water levels was also observed in pond 1 (Figure 4.13). Pond 2 is smaller than pond 1 and therefore retained less water volume even though depth was greater.

The water depth provided an indication of the potential for connectivity between the ponds. The water levels in pond 1 would need to increase to around 800mm to be directly connected (based on the original designs and heights shown in Figure 3.4). The design of pond 1 included a series of deeper areas (pools) which were interconnected but over time, and as a result of sediment accumulation, a fairly uniform bed level has developed although in some areas distinct channelled flow was observed at times of high rainfall. If areas are experiencing fast flow or inputs from lateral inflows, then some redistribution of sediment would be expected but potentially (dependent on flow) only to the immediate downstream area as point source piped inputs enter a wide and slow flowing environment. If connectivity is represented by the transfer of energy and matter between systems (Chorley and Kennedy, 1971), then the energy from the discharge will be pivotal in determining the amount and particle size distribution of sediment that could be redistributed in these ponds. The nature of discharge from the swale is shown in the data from May 2017 (Figure 4.25). The levels in pond 1 continued to rise as the swale level declines suggesting that the discharge from the swale was having an impact on the water levels of pond 1 despite the lack of rainfall at this time.

The sediment cascade approach as described in Chapter 2.2 and Figure 2.1 (Taylor 2007), assumes that there is connectivity with regard to the transfer of urban sediment. The results presented in this chapter show there was a discontinuity in flow for most of the time (the exception being under heavy rainfall conditions e.g., >8-10mm hour). Therefore, for most of the time, pond 1 acted as a depositional zone and collected and retained road runoff and inputs from the swales.

Dis-connectivity within the urban sediment cascade is exactly what a SUDS system is designed to achieve and therefore it can remove a major proportion of the sediment and sediment-associated pollutants (Woods-Ballard *et al*., 2007). As the data have shown there is little evidence for connectivity during most of the rainfall events which occurred over the study period. However, it is the small and frequently occurring storms that produced the majority of runoff from developed sites, and these affect the quality of stormwater entering the SUDS. The fine temporal resolution data has proved invaluable in the capturing details about the rising water levels and potentially further study of these transients is required in terms of localised rainfall data and climatic contributions to further inform the design of such systems.

Understanding of the temporal nature of the hydrological connectivity of the system will be used in Chapter 5 to examine connectivity in sediments and potential transport of contaminants through the system.

4.6 Summary

In order to summarise the hydrological connectivity with the SUDS at Upton a simple conceptual diagram of the water flow between the components of the SUDS systems has been developed and is given in Figure 4.35. It shows the water flow based on 3 scenarios depending on rainfall intensity.

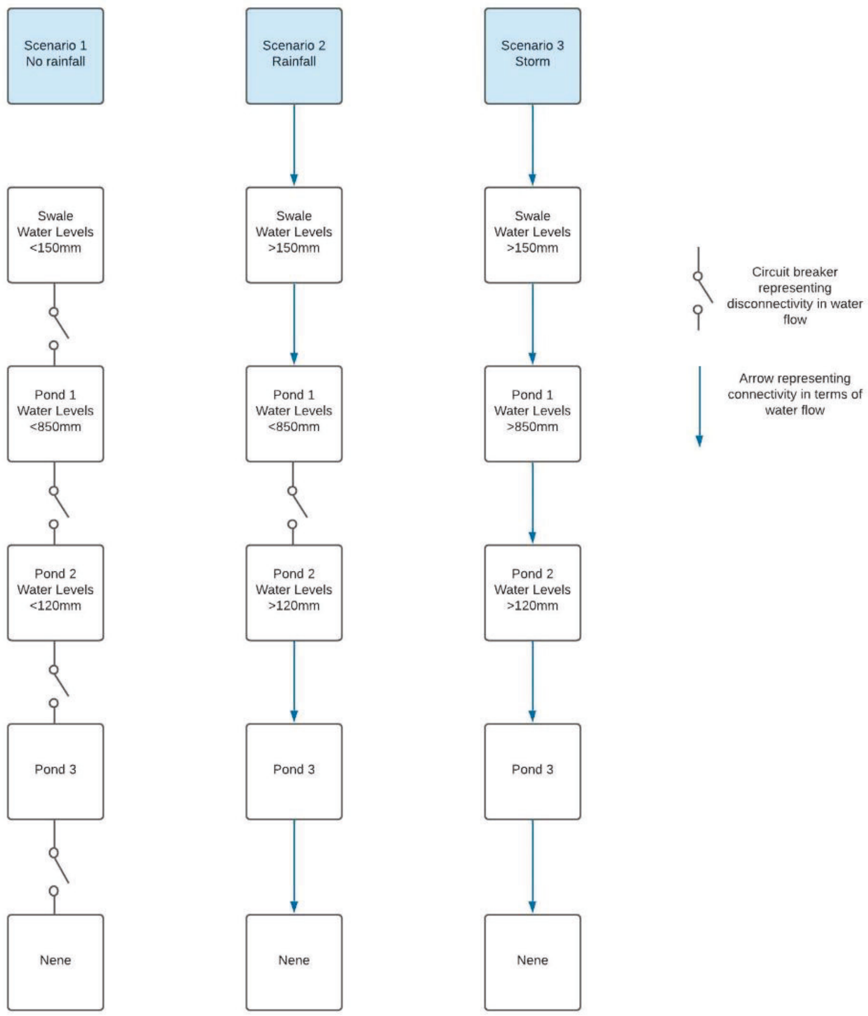

Figure 4.35: A conceptual diagram based on water flow and connectivity between the swale, pond 1 and pond 2.

Scenario 1:

No/low rainfall: Events as reported in Figure 4.22, demonstrated that even with small amounts of rainfall (2mm over 180min), not all conditions induced connectivity. For water to flow between the swale and pond 1, water levels need to exceed 150mm.

Scenario 2:

Rainfall similar to events in May 2017 caused water levels within the swale to reach >150mm and leads to outflow of water into pond 1. In addition, this magnitude of rainfall causes rising water levels, >120mm, to be observed in pond 2 which led to a connectivity with pond 3. (N.B. there were no data loggers placed in pond 3 which led to conjecture about the connectivity with the rest of the system). However, unlike pond 1 and pond 2, pond 3 and downstream (a series of vegetated channels) present a gravity fed "train" of ponds and channels that eventually connect to the Nene.

Scenario 3

All ponds were connected during events of the type depicted in November 2016. Given the long-term monitoring of the system it was anticipated that this connectivity would have been demonstrated on several occasions however, this was not observed within the monitored data. .

SUDS are designed to manage surface water and uses various component to reduce velocity and allow infiltration and evaporation. They are effectively providing a "jerky conveyor belt" for surface water and disrupting the flow from urban areas to rivers. The SUDS are also designed to provide a break within the sediment cascade as an effective way of managing sediments associated with surface water drainage. However, as shown in Figure 4.33, sediment accumulation is likely within pond 1, due to the disconnectivity with pond 2 and may require future management to remove contaminated sediment and maintain a high trap efficiency.

The dynamic response of the system is dependent on the intensity of rainfall (both volume and duration) and the accurate modelling of such a system would be difficult and might only be achieved through long-term high-resolution monitoring studies such as those reported here. Factors such as vegetation, and associated evapotranspiration, conditions (connectivity and effectiveness of water delivery systems) pertaining to inflow connections (such as road drainage) and changes in sediment accumulation and storage capacity with the SUDS themselves are all important considerations which can influence the movement and connectivity of the system. Design of SUDS needs to be based on multiple events rather than single event monitoring as it is unlikely that such an approach would represent long term behaviour of SUDS and while the hydrology is important, multiple factors related to sediment and management of vegetation as well as and overall assessment of connectivity of the integral system is required.

In this chapter the variability in hydrological responses of a SUDS system have been reported. It is evident from this long-term monitoring study that design and modelling of such a system would need to be based on varying scenarios of rainfall duration and intensity. With climate change and the reported high spatial and seasonal variability between catchments with regard to precipitation it is important to consider localised modelling with regard to the mitigation of flood risk as well as considering urbanisation. The author has noted, since completing the field research that further housing development has been approved on the land adjacent to the site studied. This rapid development of housing and a continued need for the management of surface run off remains a vital consideration within new developments at the Local Authority level. This chapter has identified the complexity of the connectivity of such systems and has highlighted that SUDS should be designed for specific catchments and therefore it is suggested that long-term monitoring should be used to provide a more organic approach for estimating urban drainage and flood mitigation within the UK. . The design of future systems should take into account current research which implies that urban drainage systems are more likely to be frequently exceeded and that storms with higher return periods should be used to account for climate change at a localised level.

Chapter 5 Sediment Analysis

5.1 Introduction

Sediments tend to accumulate within SUDS (swales, ponds, and wetlands) over periods of time (Heal *et al*., 2006) and while the contaminant removal capabilities are often reported in the literature the fate of the contaminants are not. Elevated levels of contaminants within SuDs have been reported (Table 2.10) and this has implications for long term management of such systems and the potential for SUDS to accumulate contaminants to above critical threshold- levels (McKissock *et al*., 2003; Heal *et al*., 2006). The purpose of this chapter is to present data relating to contaminant levels but also to look at the sources and resultant connectivity of the sediment and associated contaminants within the SUDS. In addition, the potential relationships between environmental magnetic measurements and contaminants will be explored which would allow a more cost-effective tool for identifying potential sources and sinks of pollution (chapter 2.5) (Objective 2).

5.2 Sampling strategy

Chapter 3 (Figure 3.13 and Table 3.2) show the positions, dates and coding of the samplers and is replicated here for clarity shown in (Figure 5.1 and Table 5.1).

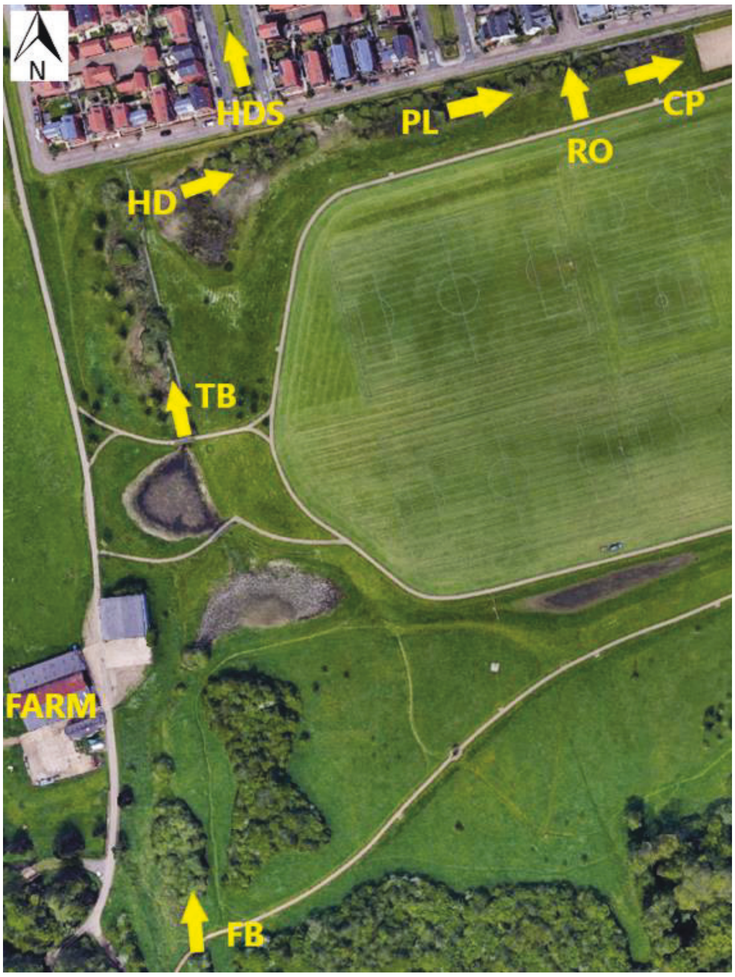

Figure 5.1: Copy of Figure 3.13: Yellow arrows indicate the location and direction of the inlets of the sediment samplers (Google Earth, 2021f).

Table 5.1: Copy of Table 3.2 Codes, positions, and dates of samples.

Code Adopted	Position of Sampler	Dates of Collection
CP	Car Park pond 1	Feb 16, Mar 16, Mar 17, Oct 17, Feb 18, July 18
PL	Padding Lane swale out pond 1 (downstream of the RO sampler)	Feb 16, Mar 16, Mar 17, Oct 17, Feb 18, July 18
RO	Road Out pond 1 (perpendicular to the pond)	Oct 17, Feb 18, July 18
HDS	Harrington Drive Swale	Feb 16, Mar 16, Mar 17 (no water was observed in this swale following the last sampling date, this sampler was moved to RO).
HD	Harrington Drive swale out pond 1	Feb 16, Mar 16, Mar 17, Oct 17, Feb 18, July 18
TB	Troll Bridge (between pond 2 and pond 3)	Feb 16, Mar 16, Mar 17, Oct 17, Feb 18, July 18
FB	Farm Bridge	Feb 18, July 18 (a sampler was placed in this location towards the end of the study period)

5.3 Organic Matter

% Organic matter (OM) was derived using LOI and is shown in Figure 5.2. Note that the sampler HDS was removed in October 2017 as the swale had not received any water (Table 5.1). The samplers RO and FB were placed in their locations in November 2017.

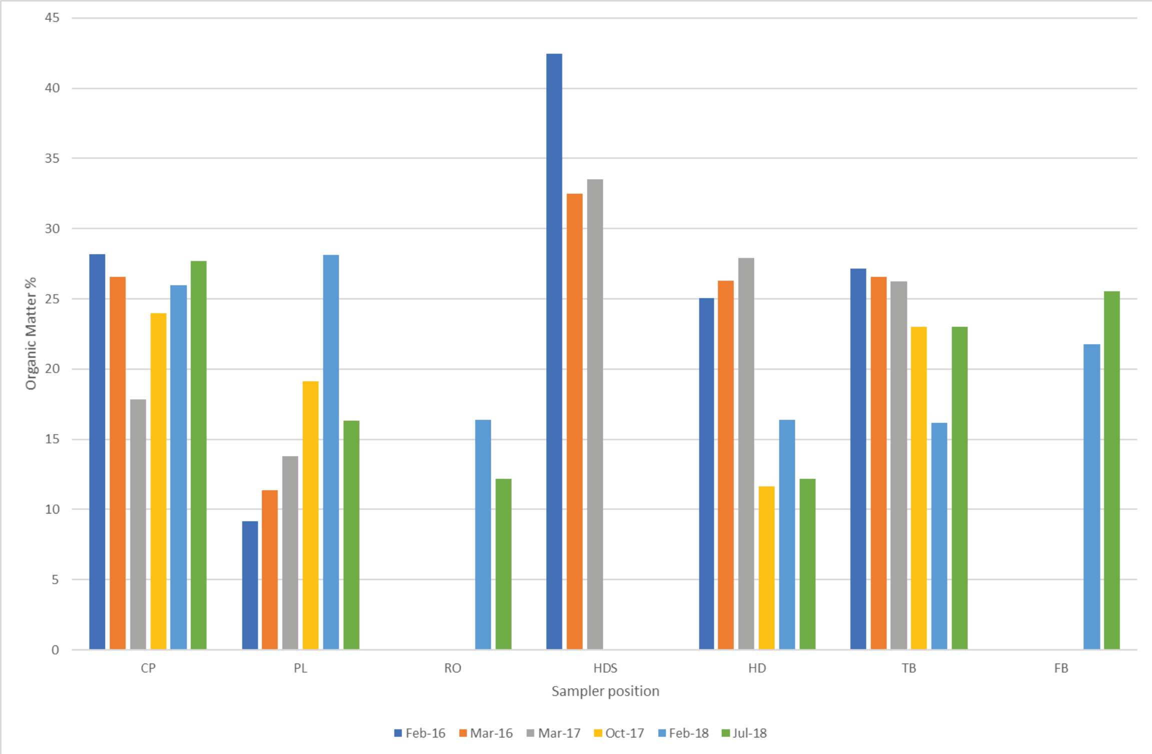

Figure 5.2: 5.1 % Organic matter in each sampler at each sampling date.

Organic matter varied from less than 15% to over 40% across the sites over the sampling period. HDS had a higher % of OM during the sampling period than the rest of the samplers. Over the summer of 2017 no water was observed in the swale and the sampler (HDS) was removed. As shown in Figure 5.2 the %OM in HD sampler declined when the swale (HDS) sampler was removed.

5.4 Particle Size

5.4.1 Definitions

Specific Surface Area (SSA).

A fundamental material property of solids is often expressed as total surface area per unit mass in both soil and sediments. SSA is a dominant factor controlling particle surface reactions and is closely related to a range of chemical properties which also influence biological processes e.g., SSA is strongly correlated with cation exchange capacity (Peterson *et al*., 1996) and larger quantities of organic matter tend to occur in sediments with greater SSAs (Mayer, 1994).

Percentiles

For volume weighted particle size distributions, such as those measured by laser diffraction, it can also be useful to report the parameters based on the maximum particle size for a given percentage of the volume of the sample (Malvern, 2010).

- D(0.1)- the maximum particle size diameter below which 10% of the sample volume exists
- D(0.5) the maximum particle size diameter below which 50% of the sample volume exists (also known as the median particle size by volume).
- D(0.9) the maximum particle size below which 90% of the sample volume exists.

5.4.2 Particle Size Results

Percentiles were plotted with SSA for each of the sites during the sampling periods and is shown in Figures 5.3-5.8.

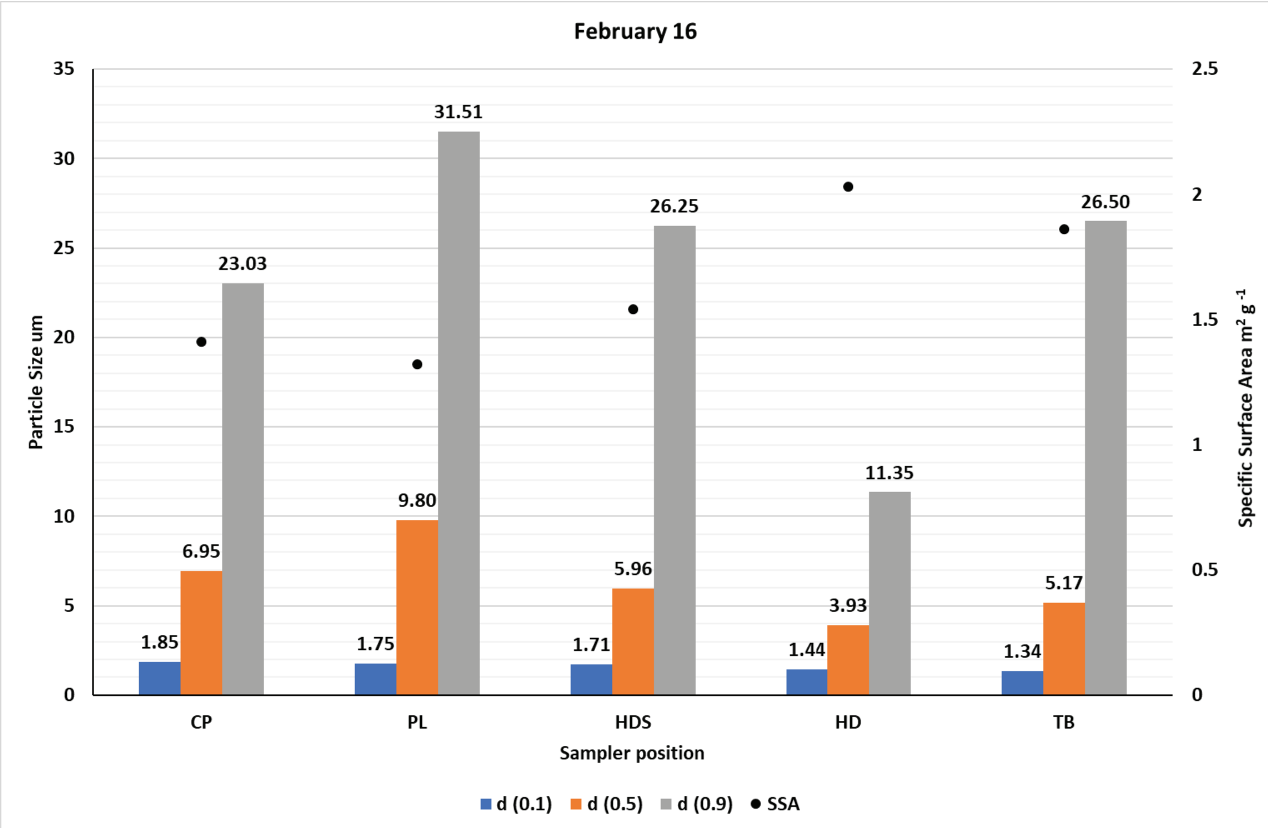

Figure 5.3: February 2016: Particle size percentiles (µm) d(0.1) d(0.5) d(0.9) and SSA m^2g^{-1}.

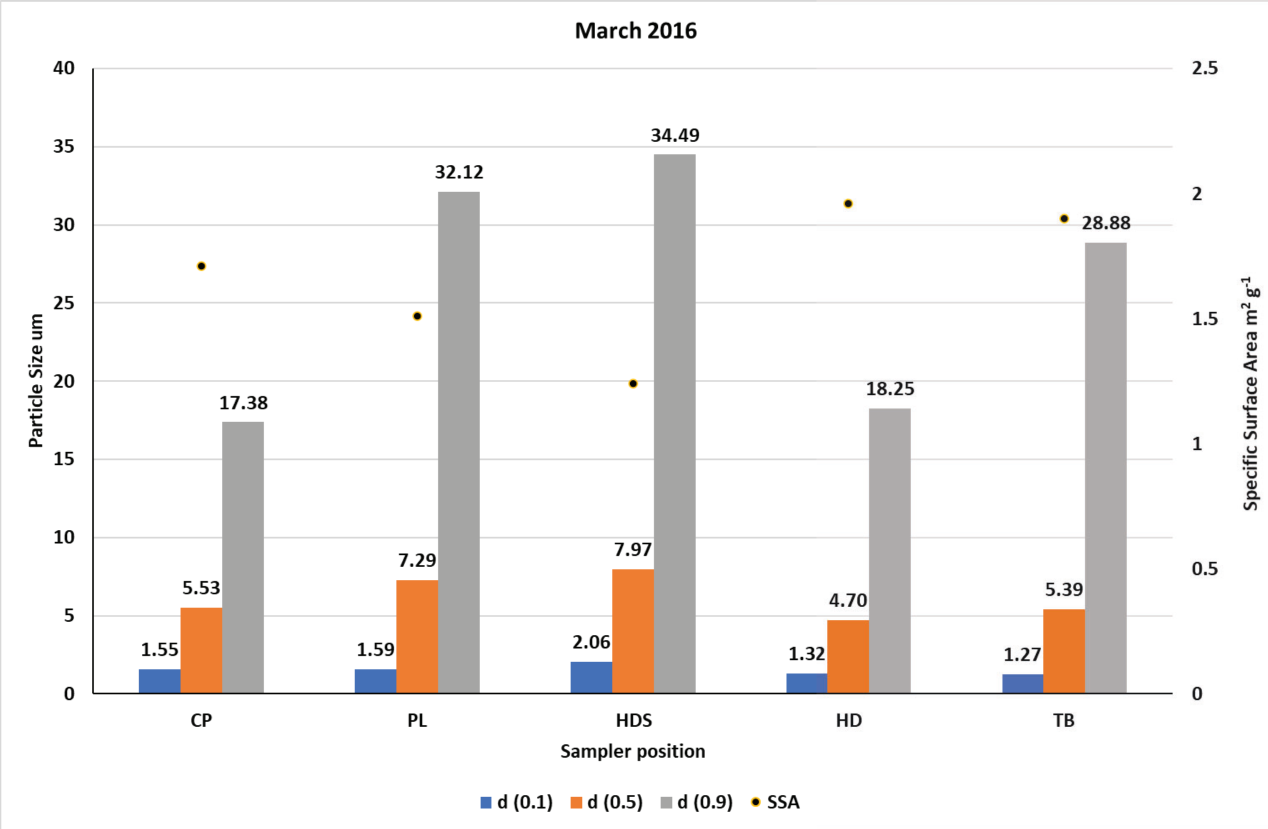

Figure 5.4: March 2016: Particle size percentiles (μm) d(0.1) d(0.5) d(0.9) and SSA m^2g^{-1}.

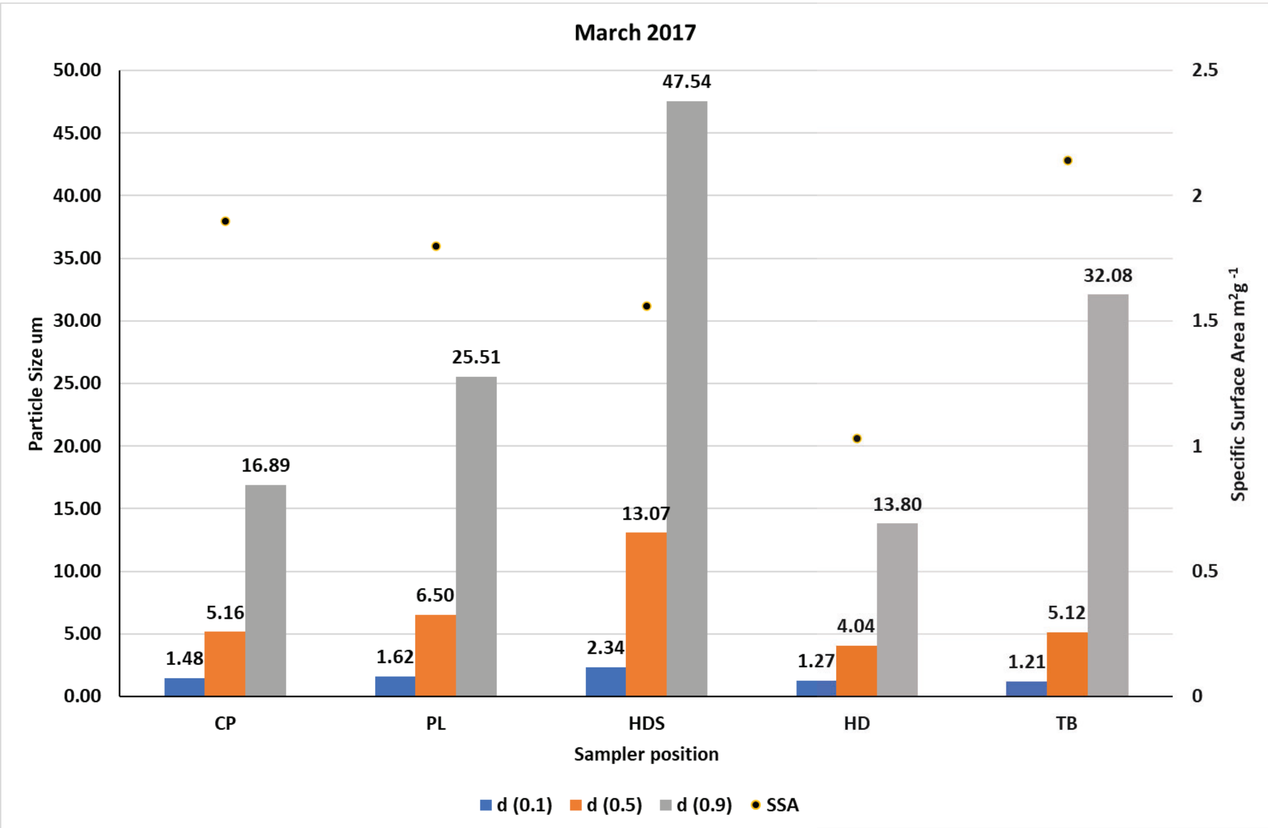

Figure 5.5: March 2017: Particle size percentiles (μm) d(0.1) d(0.5) d(0.9) and SSA m^2g^{-1}.

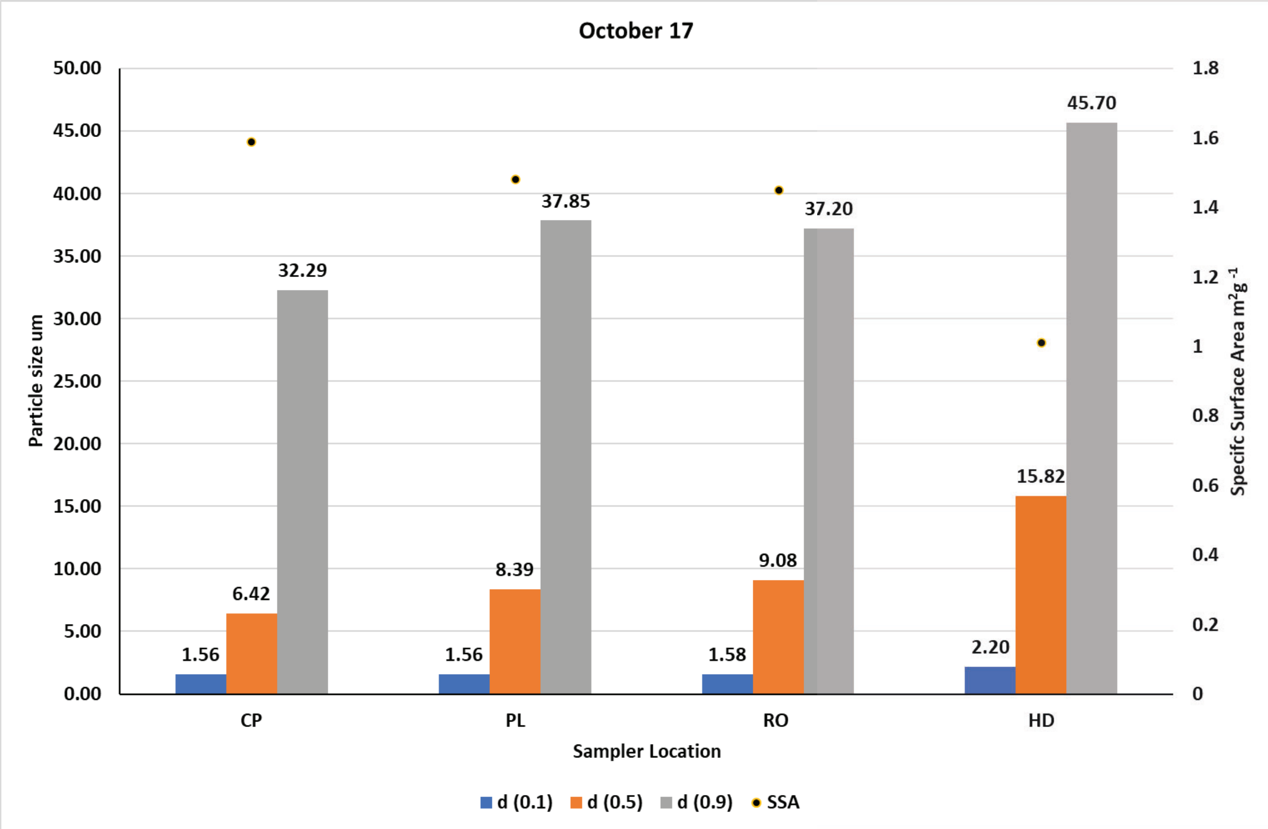

Figure 5.6: October 2017: Particle size percentiles (µm) d(0.1) d(0.5) d(0.9) and SSA m^2g^{-1}.

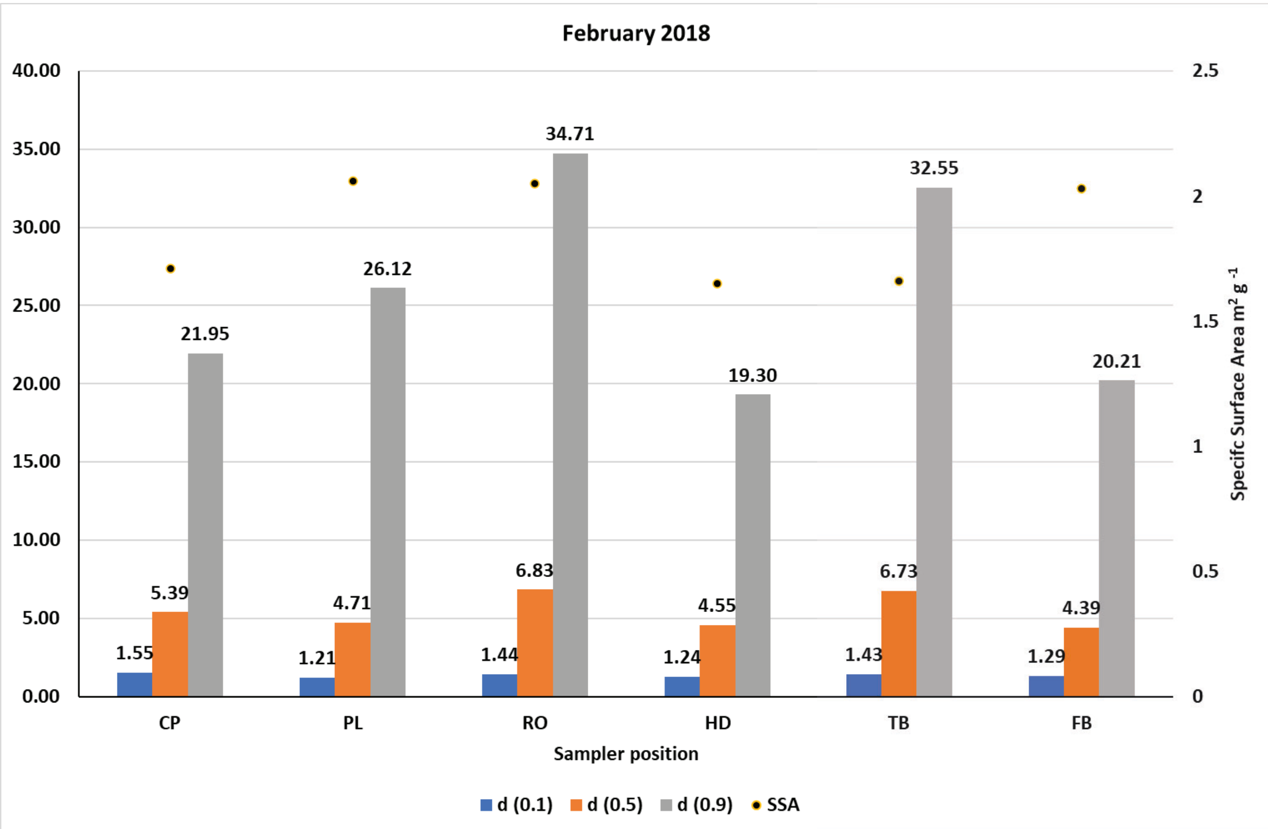

Figure 5.7: February 2018: Particle size percentiles (µm) d(0.1) d(0.5) d(0.9) and SSA m^2g^{-1}.

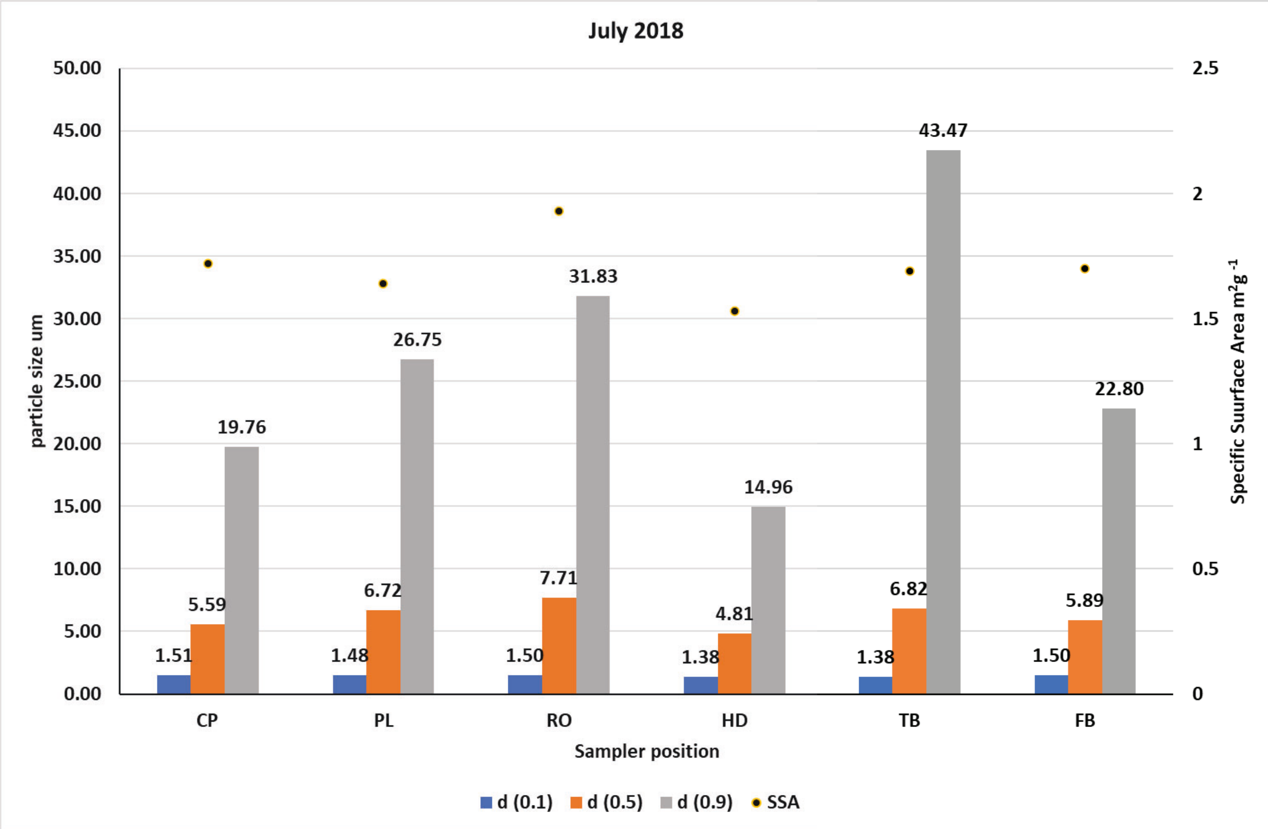

Figure 5.8: July 2018: Particle size percentiles (µm) d(0.1) d(0.5) d(0.9) and SSA m^2g^{-1}.

SSA is generally used in fingerprinting but the purpose here was to look at the physical nature of the sediment and therefore it was important to examine the particle size distribution. SSA has been used as a proxy for particle size and when the surface area increases in samples, trace element concentrations also increase (Horowitz and Elrick, 1987). Within the samples SSA was larger when the particle sizes were smaller with the largest SSA associated with sediment at HD. The relationship between SSA and d(0.1), d(0.5) and d(0.9) was examined. Due to the non-parametric nature of the data relating to d(0.1), d(0.5), d(0.9) ($p<0.05$), a Spearman's Rank correlation was performed. Negative correlations ($P<0.01$) between SSA and d(0.9) d(0.5) and d(0.1) are reported in Table 5.2. The strongest correlations are between d (0.1) and d(0.5) and SSA although all relationships are statistically significant.

Table 5.2: Correlation and p values between SSA and d(0.9) d (0.5) and d(0.1).

	SSA	
	Correlation	p-Value
D(0.9)	-0.699	0.00
D(0.5)	-0.952	0.00
D(0.1)	-0.913	0.00

Samples obtained from the swale (HDS) had a lower specific surface area and a larger d(0.9) than the rest of the samples. When the sampler was removed from the swale (Summer 2017-due to the lack of water), the particle size range found within the HD sampler changed with 90 % of the sample <37 µm as opposed to <18 µm previously measured. A return to a smaller range of particle sizes was observed after this date in February and July 2018.

CP only received local input and run off from a small carpark. Over the sampling period, the samples had smaller particle sizes in comparison to PL which was situated near to the road outlet. The RO sampler (Feb and July 2018) was placed immediately in front of the discharge pipe and larger particle sizes than the other samplers located within pond 1 were observed at this location.

TB was situated at the end of pond 2 ~110m from the inflow (surface water drain). As discussed in Chapter 4, pond 2 was potentially unconnected to pond 1 for most of the time. The particle sizes within the sampler ranged from 45.7-28.9 µm d(0.9); 4.4-15.8 µm d(0.5); 1.2-2.2 µm d(0.1). The d(0.9) of the sample was larger than the samples from pond 1 and was comparable to the particle sizes found within the swale (HDS), potentially indicating similar sources or flux of sediment being received in both the swale and the surface water drain. The d(0.5) and d(0.1) were comparable with the other samples. Transport of sediment through the pond and/or local deposition of particulate matter could account for this slightly larger range of particle sizes. FB (Feb and Jul 2018) was downstream from pond 2 and had no direct input of sediment other than local deposition/ erosion and potentially from further upstream in the system. A decrease in both d(0.9) and d(0.5) was observed from those samples obtained from TB.

Allen *et al.,* (2017a) demonstrated that the particle size distribution of bed sediments showed a consistent decrease in the size of the material with increasing distance downstream through a SUDS network. Larger sediment was observed at the upstream end of the SUDS network and the use of the SUDS as a "treatment train" was reported, as opposed to the system being a series of essentially disconnected 'independent' assets. However, within the SUDS at Upton there is potential for dis-connectivity within the system for most of the time as flow is not continuous and there is potential, except for extreme storm conditions (Chapter 4), for the ponds to act as "disconnected assets". In addition, the nature of pond 1, in terms of size, bed profile and depth (Chapter 3) would appear to provide an opportunity for the creation of deposition zones as well as vegetation trapping. Pond 1 and pond 2 were densely vegetated and the trapping efficiency of vegetation normally applies to the particle size fraction >53µm (Deletic, 2005). 90% of all samples within this study period, however, were <45.7µm in diameter. The trapping efficiency of sediment within pond 1 was demonstrated by a parallel study that was carried out in the summer 2018 (for an undergraduate project, supervised by Copeland-Phillips, 2019). Samples were taken from the upstream end of pond 1 (near the car park-CP) in the middle of the pond 1 (between PL and HD) and the downstream end of pond 1 (downstream of HD). The particle size distribution within the sediments and the rhizosphere (of the sampled vegetation) was measured by laser granulometry and is shown in Figures 5.9-5.11. At the upstream end of the pond (A), which was less well vegetated (predominately soft rush e.g. *Juncus spp*.), the majority of the particles d(0.9)) within the surrounding sediment were <322 .25 µm diameter whereas within the rhizosphere they were <51.6µm in diameter. In the middle of the pond (B), representative of the section upstream of HD, the particle size d(90)) of the surrounding sediment and the rhizosphere was similar <23.9µm diameter. This section of the pond was densely vegetated by willow (*Salix spp.)*, as well as bulrush (*Typha latifolia)* and reed sweet grass (*Glyceria* *maxima)*. In section C, downstream of HD at the end of the pond adjacent to the outflow, a more sparsely vegetated area dominated again by *Juncus sp.*, particle size of the sediment was <53 µm as compared with <41µm within the rhizosphere.

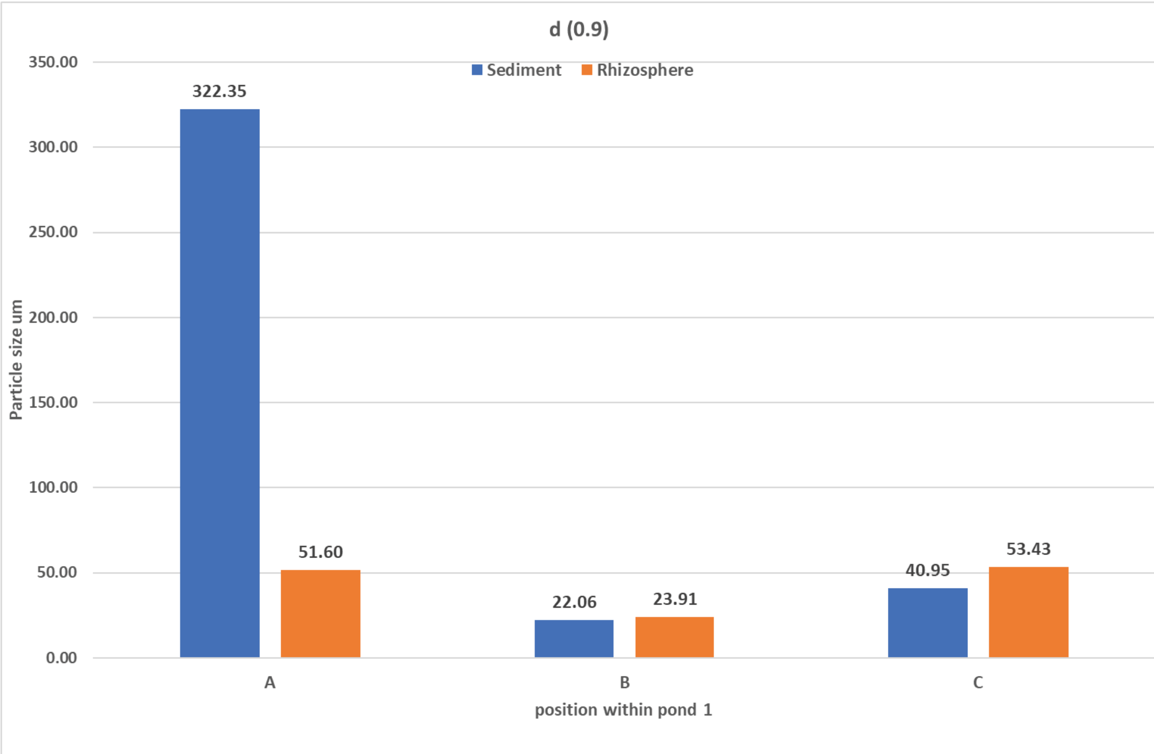

Figure 5.9: d(0.9) µm for sample points A , B and C for both the surrounding sediments and rhizosphere.

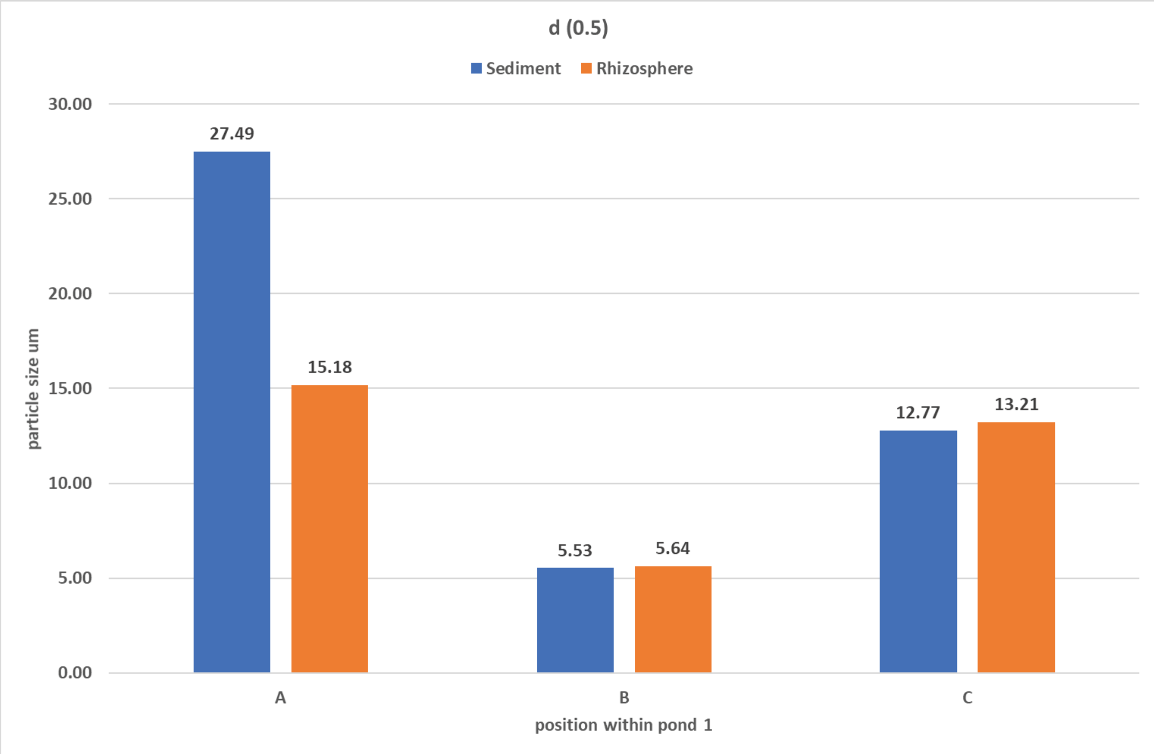

Figure 5.10: d(0.5) µm for sample points A , B and C for both the surrounding sediments and *rhizosphere.*

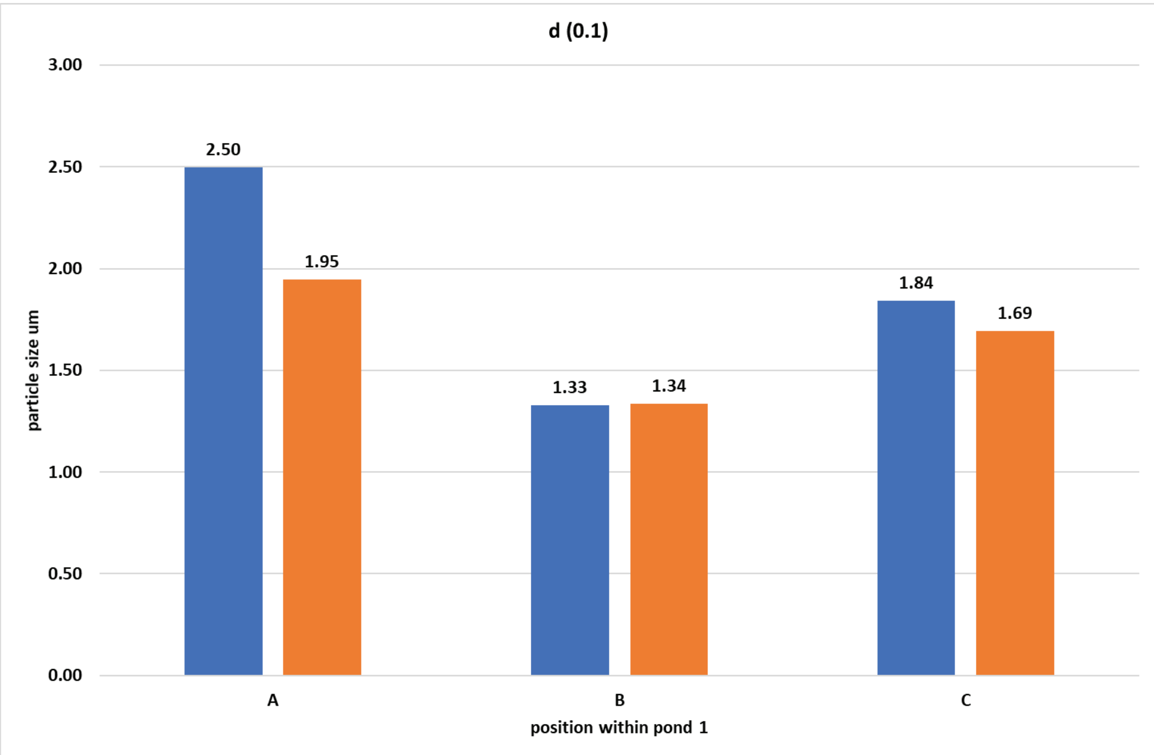

Figure 5.11: d(0.1) µm for sample points A , B and C for both the surrounding sediments and rhizosphere.

5.5 Heavy Metals

While a range of metals was tested using the methods presented in Chapter 3, for the purpose of comparison (in line with previous literature) Cd, Cu, Cr, Pb, Ni, Zn, as previously noted in chapter 2), will be reported in terms of the metal contamination in the SUDS.

All tube samplers were collected after events as reported in Table 3.3. Sampling was undertaken to obtain a temporal record of potential sediment transfer to and within the SUDS retention pond system.

5.5.1 Heavy metal concentrations across the sampling period.

A range of concentrations of heavy metals (Table 5.3) was found at the site during the study period (2015-2018). Sediment cores from floodplains in both Upton and the nearby Kingsthorpe, as well as a core from Sywell Reservoir, were used to provide background estimates of metal concentrations (N=58).

Table 5.3: Metals found within whole study site (mg kg $^{-1}$) in comparison with background levels in Northampton (where levels in the SUDS were higher than the background sample they are shown in bold).

Metal	Mean		Range		SE Mean		Median	
	SUDS N=34	**Background N=58**	**SUDS**	**Background**	**SUDS**	**Background**	**SUDS**	**Background**
Cd	**0.67**	0.45	0.52-0.03	0.39-1.50	0.12	0.20	0.50	0.60
Cr	**51.78**	45.25	0.00-108.63	30.18-61.56	5.23	1.00	61.00	61.56
Cu	**76.9**	18.25	23.56-169.72	8.05-27.09	7.09	0.71	70.1	27.09
Ni	9.01	31.03	1.47-28.17	22.17-39.84	1.87	.44	5.95	39.84
Pb	**56.97**	42.29	1.08-143.81	22.02-66.37	6.02	1.92	52.6	66.37
Zn	**330.5**	70.37	31.95-746.56	6.34-170.07	32.4	8.63	326.9	170.07

Within the samples the mean concentration of Cd, Cr, Cu, Pb, Zn where higher than the background. The median concentration (Cd, Cr, Cu, Ni, Pb and Zn) of metals at each of the sampling points was calculated and shown in Table 5.4.

Table 5.4: Median concentrations (mg kg^{-1} of metals in each of the tube samplers over the study period).

	Median values (mg kg^{-1}) (N=34)					
Sampler	Cd	Cr	Cu	Ni	Pb	Zn
CP	0.383	12.56	81.5	1.7	53.9	312
PL	0.67	16.69	76.5	10.8	88.8	498.7
RO	0.924	13.245	124.9	6.17	73.29	351.4
HDS	0.373	2.34	40.7	0	26.84	367
HD	0.558	16.13	64.9	7.28	71.96	314.7
TB	0.7	12.57	60.9	19.22	41.28	322.4
FB	0.53	24.28	128.8	5.4	96.7	720

None of the metals exceeded current UK SGVs (Table 2.4) (no SGVs are reported for Cu and Zn) and while Zn exceeded the Dutch target values, only one site (FB) equalled the Dutch intervention levels. In terms of the reported heavy metals across the UK (Table 2.3) Cu and Zn exceeded the range of values cited for rural, urban, and industrial soils (Environment Agency, 2007).

Lower concentrations of metals, with the exception of Zn, were reported from the HDS sampler (swale). The swale is not connected to any of the wider SUDS and primarily received input from precipitation and RDS from the adjacent roads. Ponds 1 and 2 received inputs from various sources (swales, surface runoff, road runoff and a car park) shown on Figure 3.6.

Ni was below the limits of detection (LOD) in the swale over the study period and significantly lower than background levels in the rest of the samplers. Ni in soils arises from both natural and anthropogenic sources including combustion of hydrocarbons in particular diesel and petrol (Cempel and Nikel, 2006; Defra, 2019). Air quality background concentrations of Ni are generally less than 2 ng m^{-3} but Ni has been found in RDS particularly on major road routes (Defra, 2019).

5.5.2 Heavy metals at each of the sites

The metals of concern Cd, Cr, Cu, Ni, Pb and Zn for each site are reported in terms of a cluster analysis and PCA. Cluster analysis was performed using Minitab (v.19, 2019) to group the data for each sampling site and to look for any similarities between sites, therefore ascertaining whether there was any potential connectivity between the sites. Cluster analysis is a technique which groups similar observations into a number of clusters based on observed values of several variables for each individual site. All cluster analyses were performed as Complete linkage, Euclidean distance measures of similarity (Minitab, 2020). Distance within cluster observations refers to the distance between the observations, using complete linkage (furthest neighbour). The distances between two clusters is the maximum distance between an observation in one cluster and an observation in another cluster. Euclidean is one of the most common distance measures which calculates the square root of the sum of squared differences between the observations (Minitab, 2020).

February 2016

The raw data for the samplers emptied in February 2016 is shown in Table 5.5, the cluster dendrogram in Figure 5.12 for and the PCA in Figure 5.13.

Table 5.5: Metal concentrations found at each of the sites in February 2016 (units are mg kg^{1}).

	Concentration mg kg^{-1}					
	Cd	**Cr**	**Cu**	**Ni**	**Pb**	**Zn**
CP	0.81	11.50	76.98	5.76	45.18	244.01
PL	0.68	14.05	59.53	27.65	51.28	251.35
HDS	0.79	2.34	40.74	0.00*	31.82	366.85
HD	0.50	9.90	39.70	25.53	42.71	209.72
TB	0.78	13.80	57.99	28.17	42.69	335.53

**below LOD*

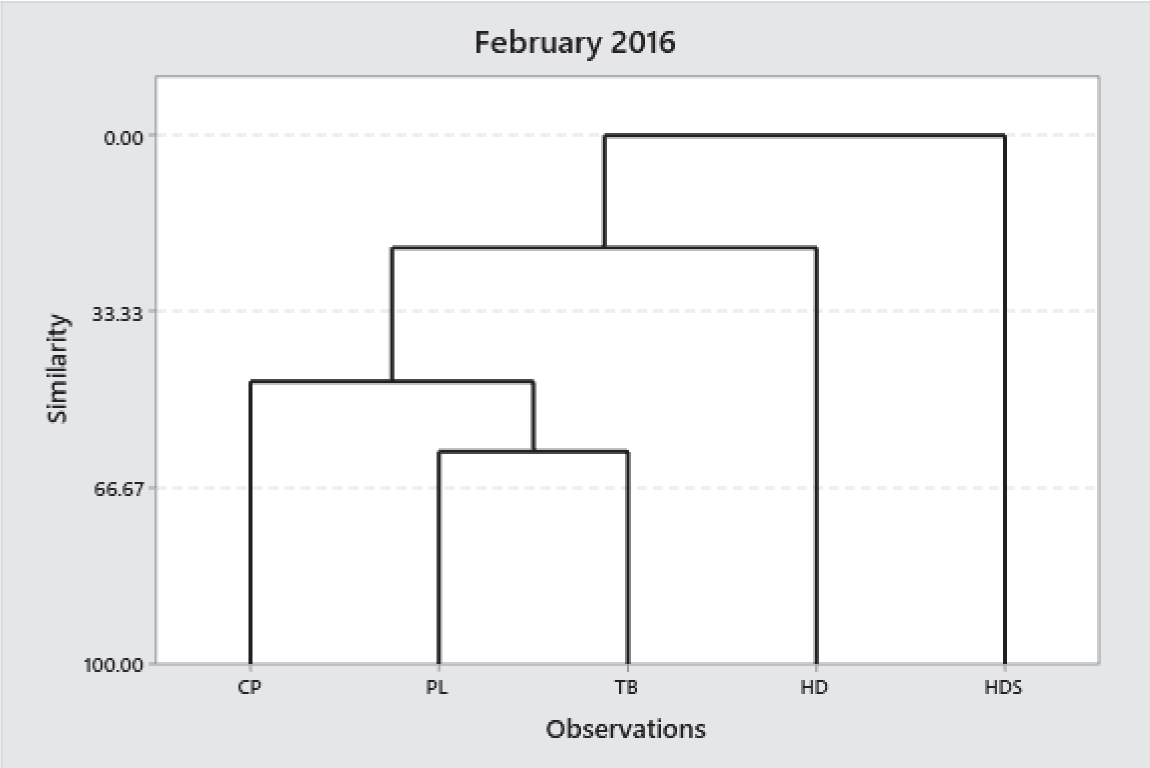

Figure 5.12: Cluster Dendrogram February 2016 between CP, PL, TB, HD and HDS for metals Cd, Cr, Cu, Ni, Pb and Zn.

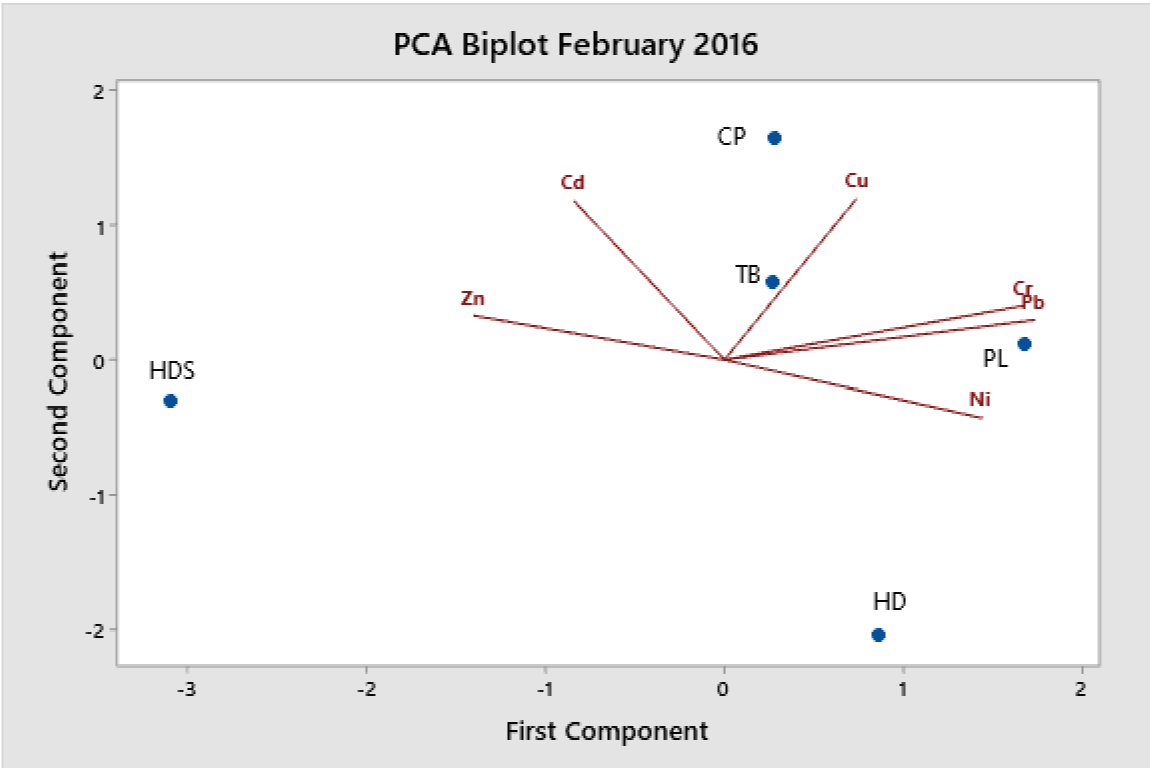

Figure 5.13: PCA plots of metals and sites February 2016.

Figure 5.12 shows that the sample collected from the swale (HDS) is almost totally independent of those samples collected from the retention pond system and shows no similarity to the other sites. The swale system is designed to catch runoff from the surrounding road/ pavement drainage and is not designed to be in receipt of major inflows from a wide area. Therefore, it might be expected that the nature of the sediment would be different from other parts of the system. Table 5.5 shows samples to have relatively low concentrations of Cr, Pb and Ni in comparison with the other sites although higher levels of Zn. CP, HD and PL were clustered together (21.3% similarity) but as they are all part of the same system this is to be expected.

Figure 5.13 shows that CP, PL, TB and HD are influenced by the first component. Lower levels of Zn were seen in HD. Similar concentrations of Cr and Cu were observed in PL and TB which probably accounts for their similarity (60%).

March 2016

The metal data for the samples collected in March 2016 is shown in Table 5.6, the cluster dendrogram in Figure 5.14 and the PCA in Figure 5.16.

Table 5.6: Metal Concentrations found at each of the sites in March 2016.

	Concentration mg kg^{-1}					
	Cd	**Cr**	**Cu**	**Ni**	**Pb**	**Zn**
CP	0.13	1.8	49.36	0.00*	1.08	31.95
HDS	0.03	0.05	23.56	0.00*	0.00*	0.00*
HD	0.98	16.03	63.45	14.50	68.55	311.10
TB	0.20	3.29	19.50	1.46	6.8	88.77

**below LOD*

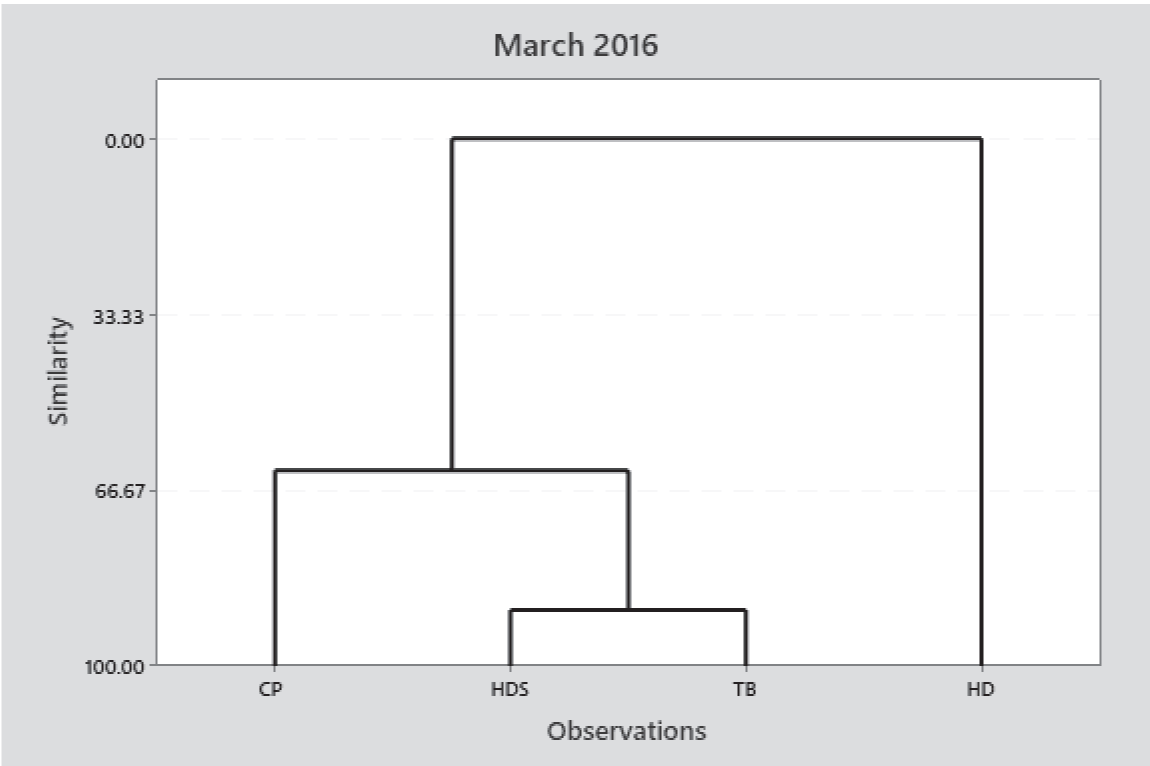

Figure 5.14: Cluster Dendrogram March 2016 between CP, PL, TB, HD and HDS for metals Cd, Cr, Cu, Ni, Pb and Zn.

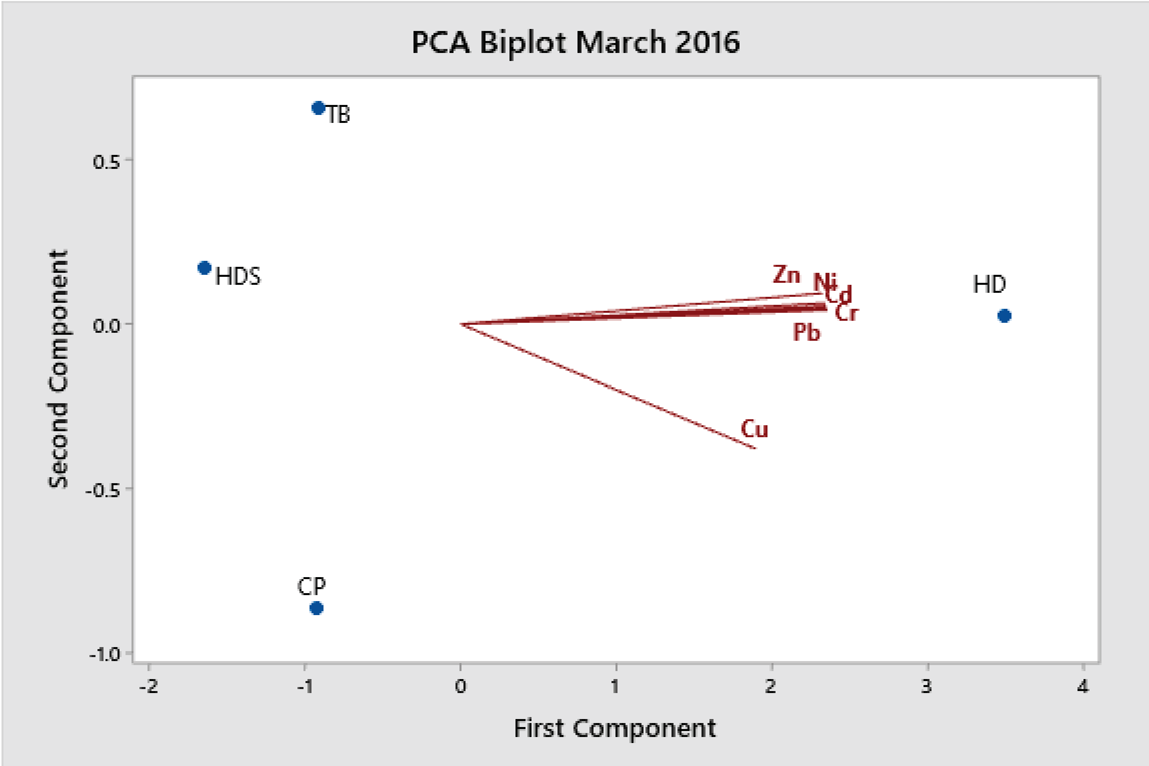

Figure 5.15: PCA plots of metals and sites March 2016.

HDS and TB are shown to have a high similarity (82%). Both samples contained lower levels of Cu than the other sites. HDS and TB were also grouped with CP (70%) as all three of these sites had lower concentrations of all the metals than in HD. Two distinct grouping of metals are seen in figure 5.15. It is unlikely that the HDS (swale) and TB (pond 2) are connected although similarity, regarding metal concentration was seen but, this may be more related to the absolute concentrations as opposed to showing similarities due to connections within the SUDS.

March 2017

The metal data for the samples collected in March 2017 are shown in Table 5.7, the cluster dendrogram in Figure 5.16 and the PCA in Figure 5.17.

Table 5.7: Metal Concentrations found at each of the sites in March 2017.

	Concentration mg kg^{-1}					
	Cd	**Cr**	**Cu**	**Ni**	**Pb**	**Zn**
CP	0.96	16.72	86.03	19.02	67.45	328.94
PL	0.52	16.69	76.49	11.14	88.78	404.08
HDS	0.37	3.48	31.04	0.00*	26.84	403.08
HD	0.52	16.44	66.29	23.08	76.44	318.26
TB	0.88	10.75	63.8	22.7	39.88	309.35

**below LOD*

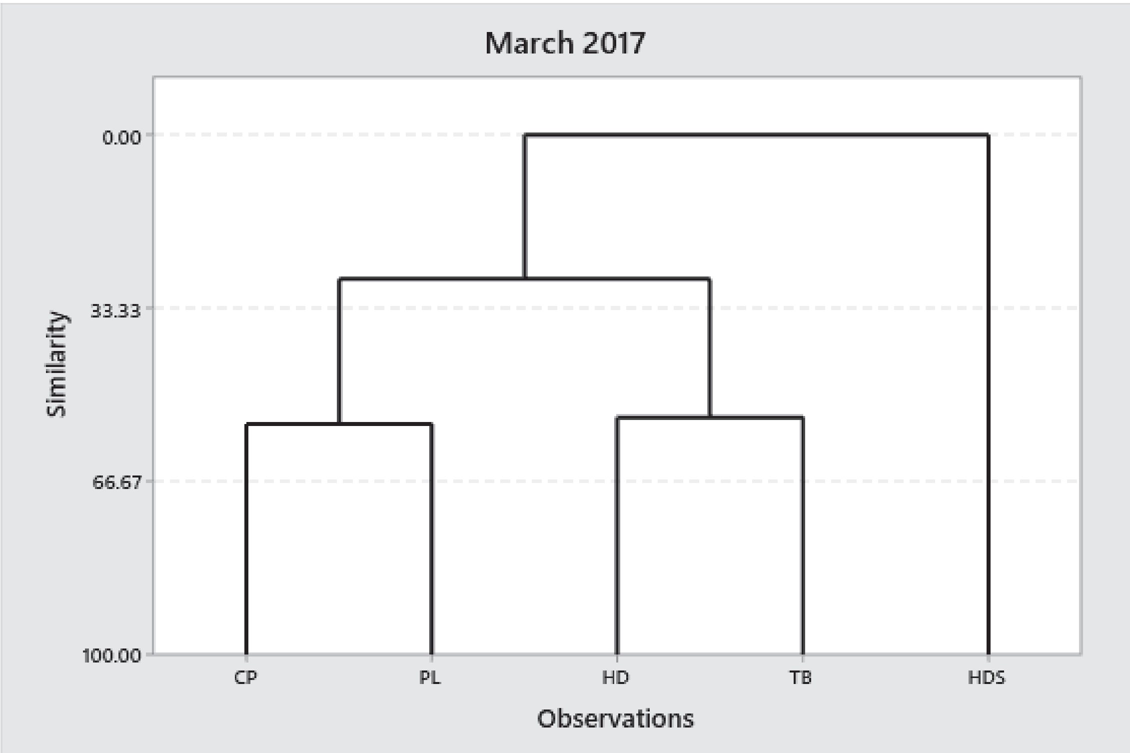

Figure 5.16: Cluster Dendrogram March 2017 between CP, PL, HD, TB and HDS for metals Cd, Cr, Cu, Ni, Pb and Zn.

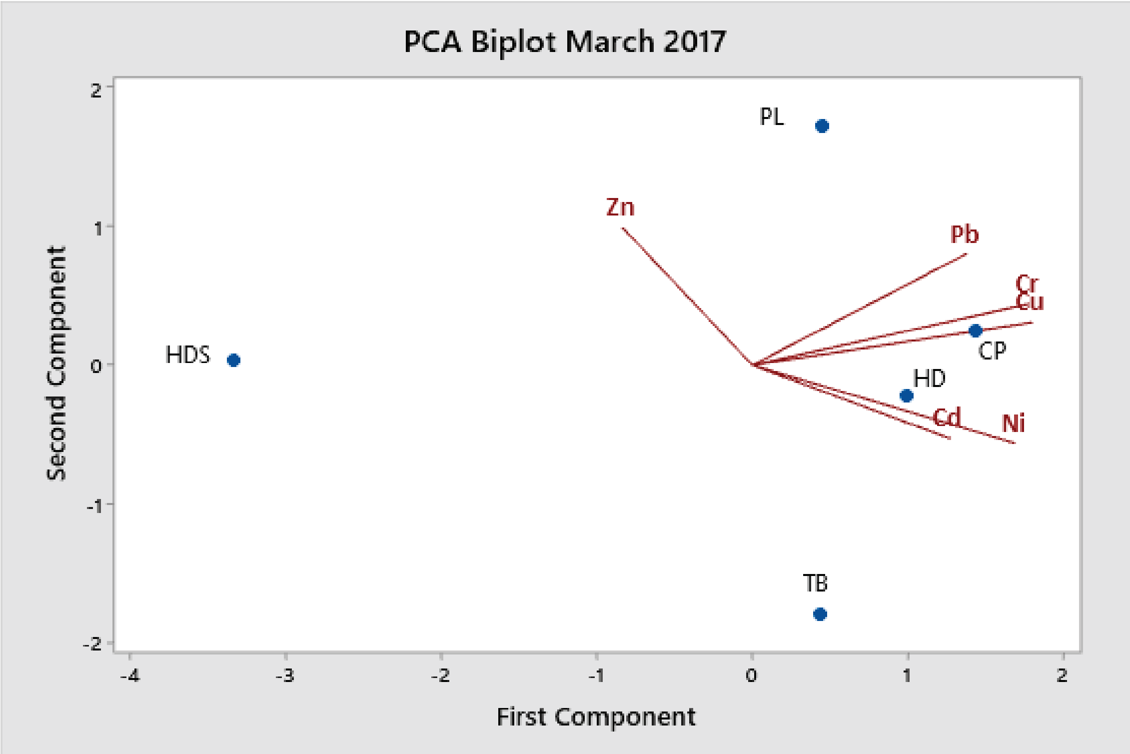

Figure 5.17: PCA plots of metals and sites March 2017.

Three groupings are shown in Figure 5.16 and Figure 5.17. HDS is not significantly related to any of the other sites and while it had lower values of Cd, Cr, Cu, Ni and Pb, a concentration of 403.08 mg kg^{-1} of Zn was recorded, which is similar to that of PL (404.08 mg kg^{-1}). 54% similarity was observed between CP and PL and between HD and TB with all four sites showing a 27% similarity. CP and PL had similar levels of Cr and Cu (Table 5.7) and, due to the close proximity of these sampling sites within pond 1, there is clear potential for connectivity between these sites. HD and TB had a similar level of similarity related to Zn and Ni concentrations. However, as both sites received direct surface run off (HD from the swale; TB from the inflow pipe) the similarities observed could be related to the nature of the sediment input as opposed to connectivity between these sites.

October 2017

The metal data for the samples collected in October 2017 are shown in Table 5.8, the cluster dendrogram in Figure 5.18 and the PCA in Figure 5.19.

Table 5.8: Metal Concentrations found at each of the sites in October 2017.

	Concentration mg kg^{-1}					
	Cd	Cr	Cu	Ni	Pb	Zn
CP	0.32	16.42	92.26	0.00*	106.09	738.41
PL	0.74	20.25	89.9	8.86	143.81	557.28
RO	3.96	16.24	58.59	8.44	75.37	344.65
HD	0.8	21.52	100.95	15.75	79.73	376.2

** below LOD*

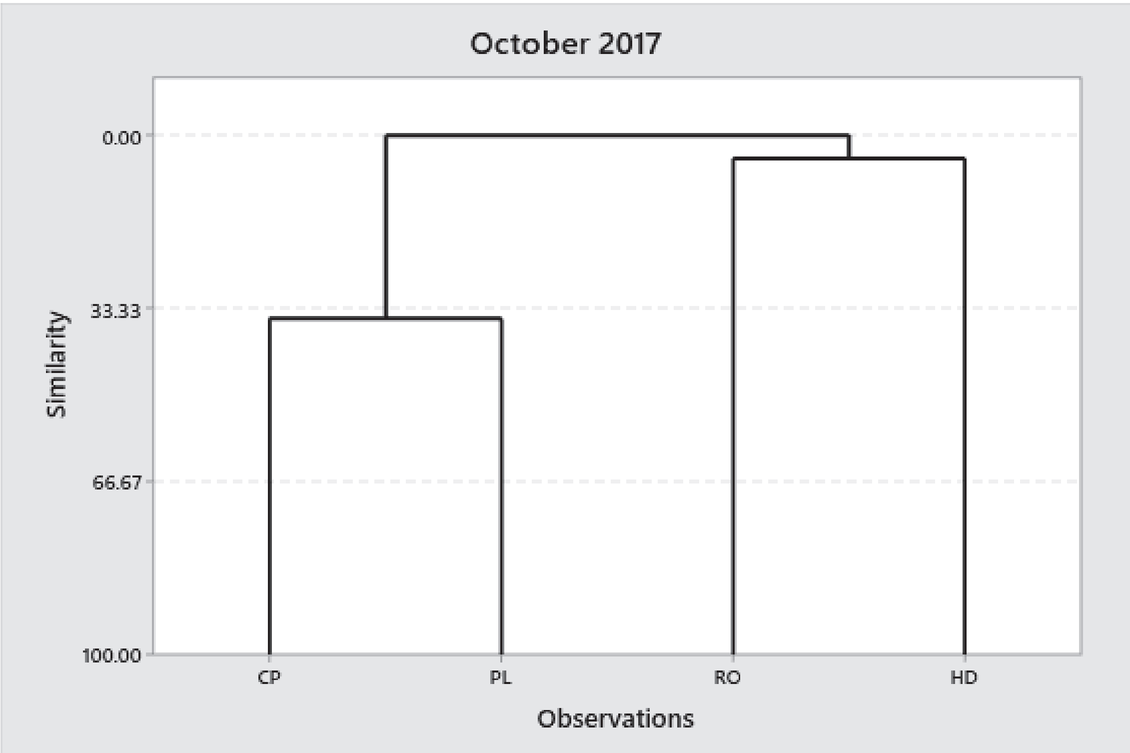

Figure 5.18: Cluster Dendrogram October 2017 between CP, PL, RO and HD and for metals Cd, Cr, Cu, Ni, Pb and Zn.

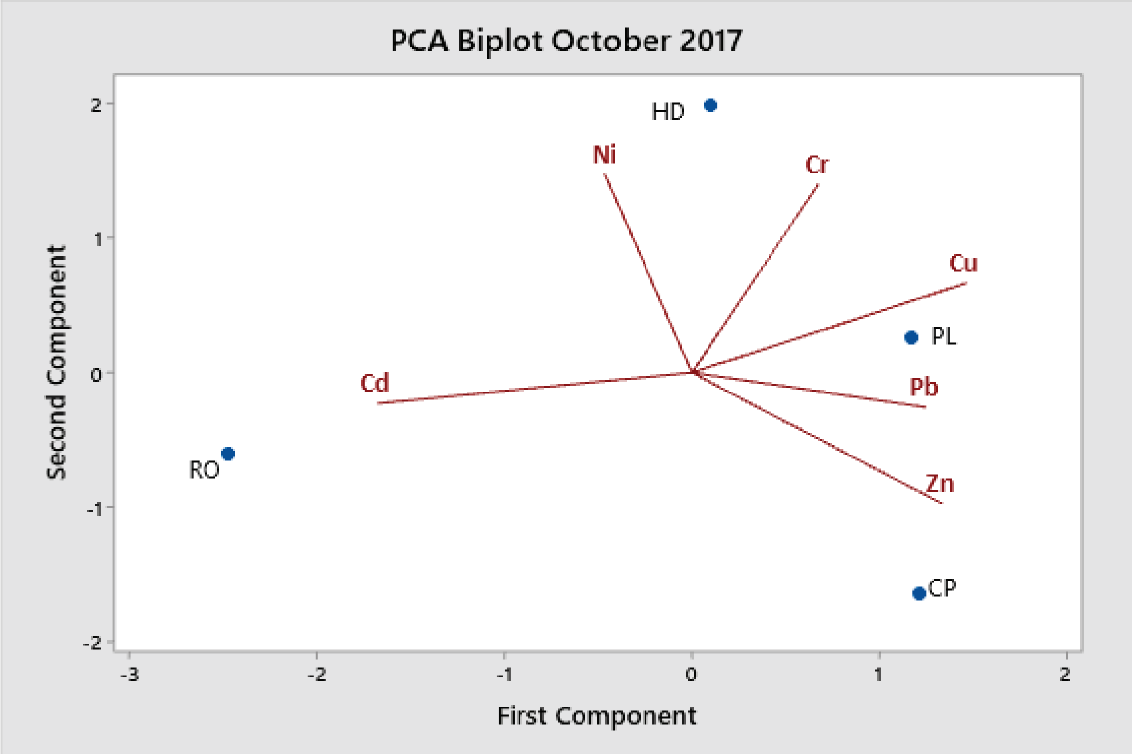

Figure 5.19: PCA plots of metals and sites October 2017.

Weak similarities are shown between sites (Figure 5.18) and there are no distinct groupings in the metals in the PCA plot (Figure 5.19). This lack of similarity between any of the sites could be due to the lack of water in the system over the summer of 2017. No sediment was collected from TB during this sampling period. Both CP and PL did have high levels of Cu (92.26 mg kg^{-1} and 89.9 mg kg^{-1} respectively) as did HD (100.95 mg kg^{-1}). High levels of Zn (738.4 mg kg^{-1}) were recorded at CP and the sediment from the sampler at RO had the highest concentration of Cd that was recorded over the sampling period (3.96 mg kg^{-1}) at any site.

February 2018

The metal data for the samples collected in February 2018 are shown in Table 5.9, the cluster dendrogram in Figure 5.20 and the PCA in Figure 5.21.

Table 5.9: Metal Concentrations found at each of the sites in February 2018.

	Concentration mg kg^{-1}					
	Cd	**Cr**	**Cu**	**Ni**	**Pb**	**Zn**
CP	0.00*	1.38	27.5	0.00*	3.35	39.83
PL	1.09	12.86	104.77	12.35	66.82	296.57
RO	0.59	16.54	131.73	20.45	86.78	434.75
HD	0.43	12.59	73.89	0.00*	49.50	247.59
TB	0.55	11.34	47.06	3.93	35.22	253.58
FB	0.53	21.7	87.9	6.14	121.49	746.57

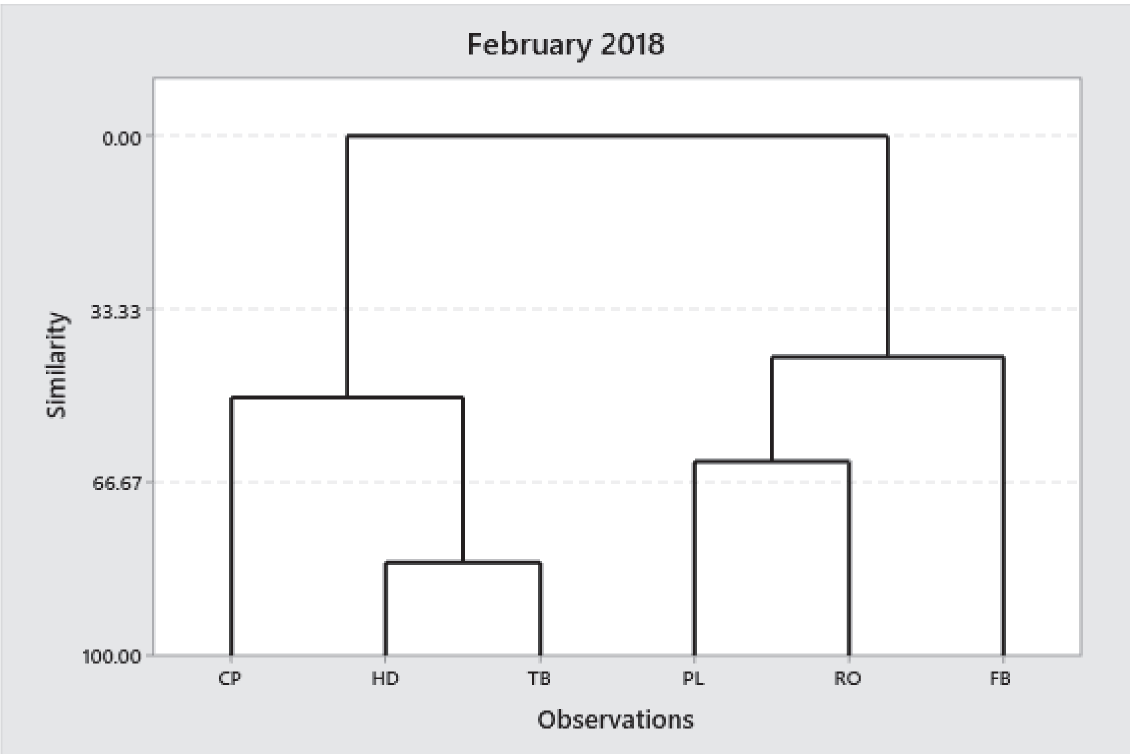

Figure 5.20: Cluster Dendrogram February 2018 between CP, PL, TB, HD, Tb and FB for metals Cd, Cr, Cu, Ni, Pb and Zn.

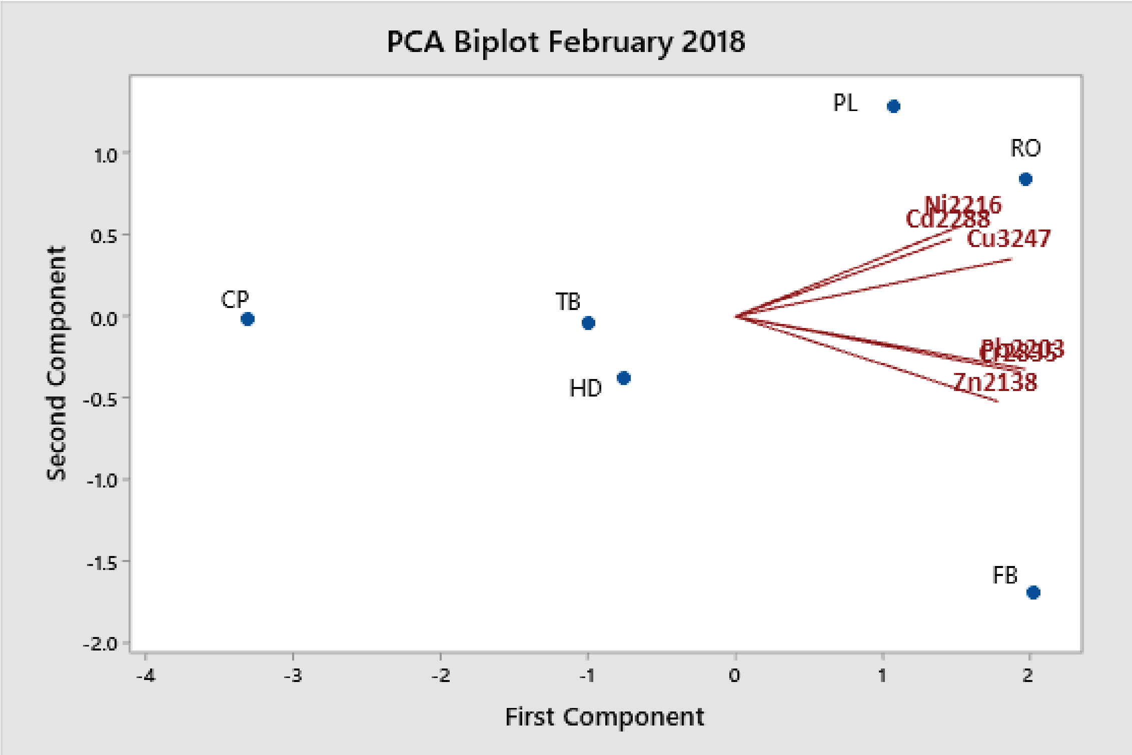

Figure 5.21: PCA plots of metals and sites February 2018.

There are two distinct and unrelated clusters shown in Figure 5.20. HD and TB showed 82% similarity and had similar concentrations of Zn (247.6 and 253.6 mg kg^{-1} respectively) and similar levels of both Cd and Cr. Both sites were 50% similar to CP, although CP had lower concentrations of metals. The second grouping showed 63% similarity between PL and RO, probably due to the similarly high Cu concentrations (104.77 and 131.73 mg kg^{-1} respectively), and 43% similarity with FB. PL, RO and FB due to high levels of Pb, and Zn. The grouping of metals, Figure 5.21 shows two distinct groupings of Ni, Cd, Cu and Zn, Pb and Cr. It is unlikely that there was any connectivity between the three samplers PL, RO and FB, although potential connectivity between PL and RO was likely due to the close proximity to each other and to the road outlet into pond 1. The high values seen at FB could indicate that there is a source of heavy metals close to this site, yet independent of the SUDS. FB is located close to the farm at Upton and within close proximity (<200m) of storage barns and a concreted yard which could potentially be a source of sediments/contaminants to this area of the SUDS. The highest level of Zn over the sampling period is recorded within the FB sampler at 746.57 mg kg^{-1}.

July 2018

The metal data for the samples collected in July 2018 are shown in Table 5.10, the cluster dendrogram in Figure 5.22 and the PCA, in Figure 5.23.

Table 5.10: Metal Concentrations found at each of the sites in July 2018.

	Concentration mg kg^{-1}					
	Cd	**Cr**	**Cu**	**Ni**	**Pb**	**Zn**
CP	0.44	13.62	110.62	3.4	62.54	379.9
PL	0.82	16.72	147.59	10.8	91.29	440.15
RO	0.76	13.63	144.95	0.00*	79.77	406.19
HD	0.38	10.88	90.1	3.79	40.2	212.96
TB	0.62	16.49	152.71	23.2	53.91	433.93
FB	0.52	26.86	169.73	4.65	71.91	693.66

**below LOD*

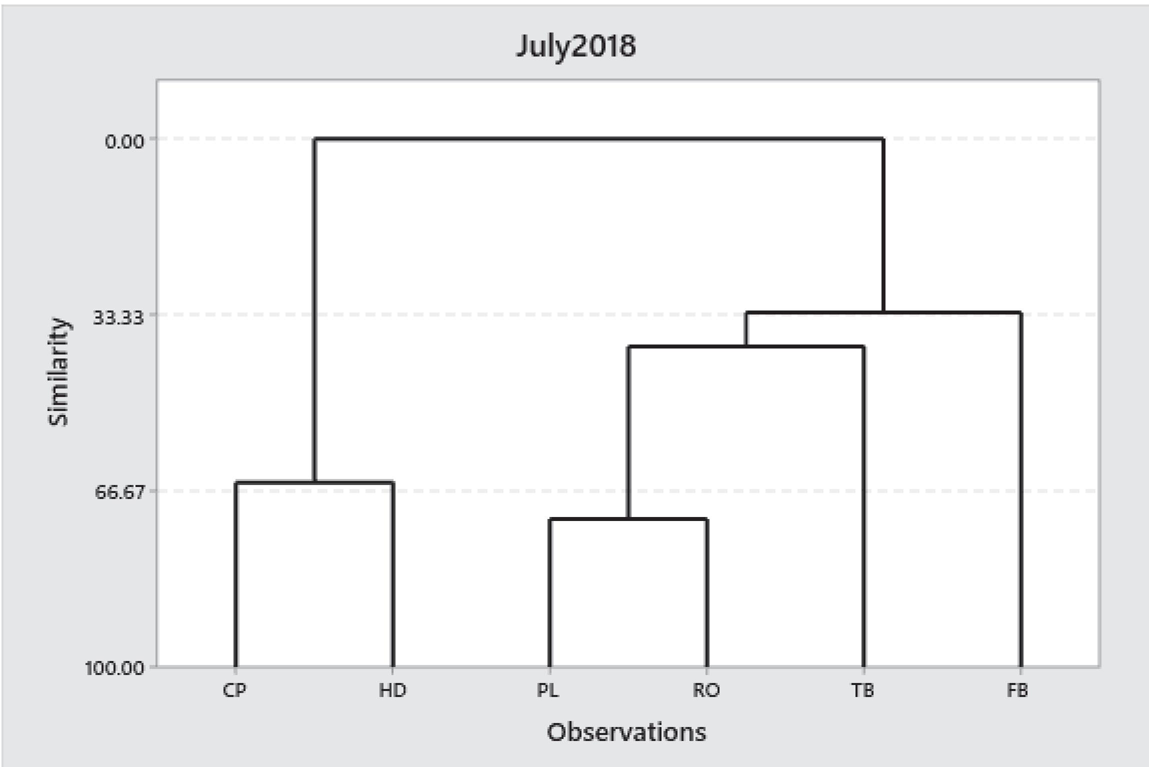

Figure 5.22: Cluster Dendrogram July 2018 between CP, PL, TB, HD and HDS for metals Cd, Cr, Cu, Ni, Pb and Zn.

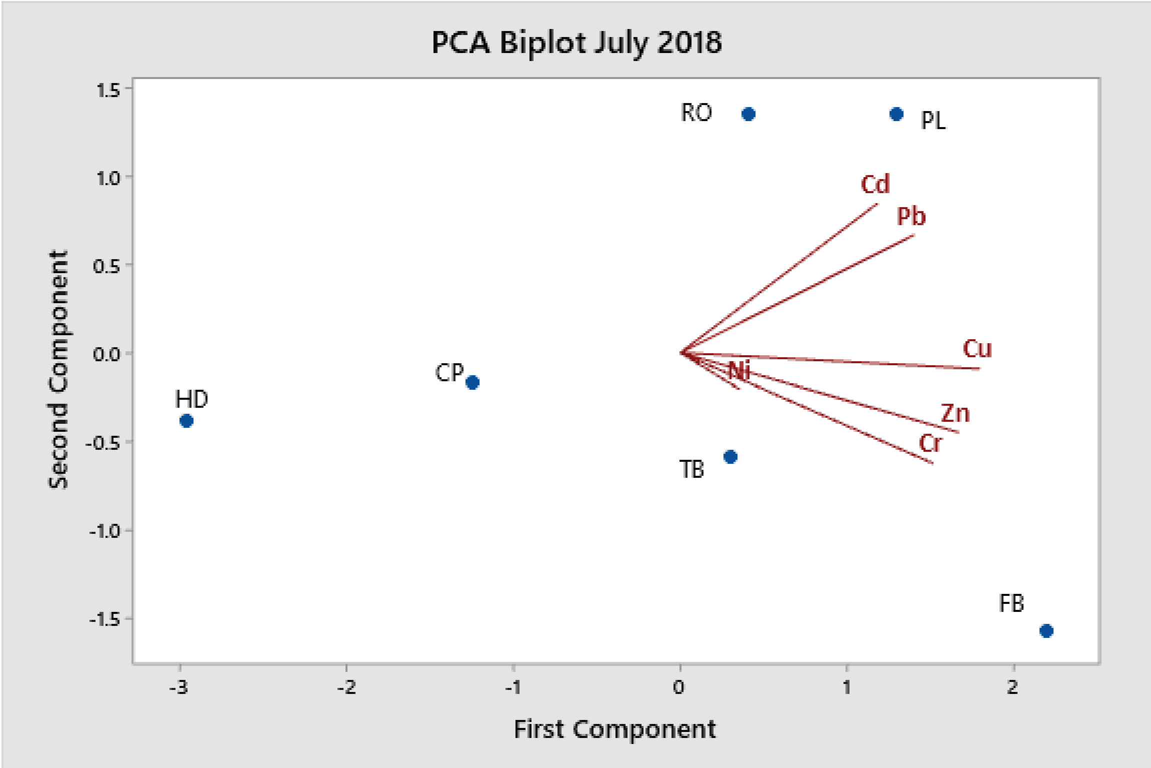

Figure 5.23: PCA plots of metals and sites July 2018

Figure 5.23 shows three groupings of metals Cd and Pb; Cu and Zn; Cr and Ni. Again, the sites were divided into two groups (Figure 5.22). CP and HD had a 65% similarity due to similar levels of Cd, Cu and Ni (Table 5.10). In the second cluster group (PL, RO, TB and FB), 72% similarity is shown between PL and RO (due to similarities in metal concentration), as was also reported for February 2018 (Figure 5.20). The 40% similarity (between PL, RO and TB) and the 33% similarity (between PL, RO, TB and FB) was more than likely related to the high levels of Zn observed. A connection between PL and RO had previously been suggested due to their close proximity to the road outlet. However, it is unlikely that there is a direct connection between these two sites and TB and FB. It is possible that TB and FB were receiving other inputs that were similar to those in pond 1 (from the farm) but not as part of the SUDS.

5.5.2 Correlations between Metals

All metals analysed (Cd, Cr, Cu, Ni, Pb and Zn) were tested for normality using the Anderson Darling test (Minitab, 2020). (Cd p<0.05; Cr p>0.05; Cu p>0.05; Ni p<0.05; Pb p>0.05; Zn p>0.05). Due to the non-parametric nature of the data, a Spearman's Rank correlation was conducted to examine relationships between the metals. The relationships are plotted in Figure 5.24.

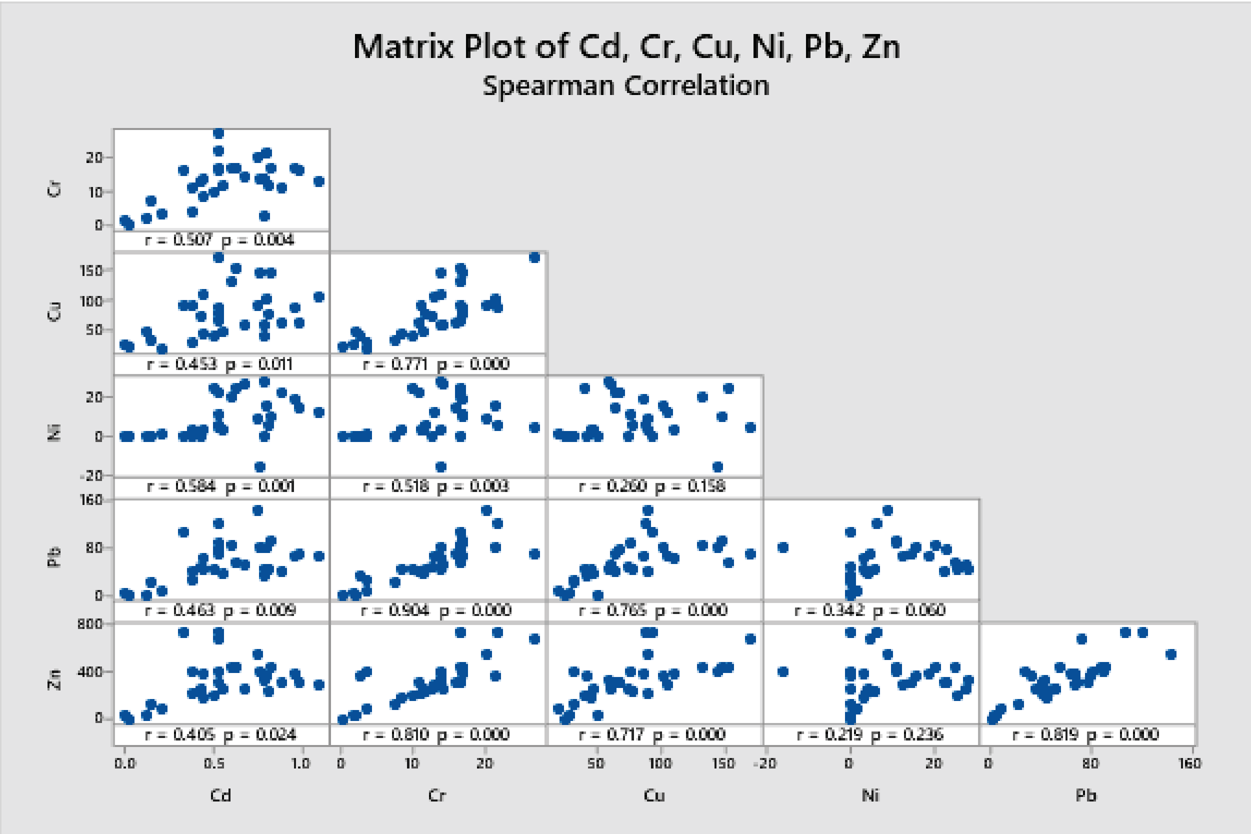

Figure 5.24: Matrix plots for Pairwise Spearmans Correlations between Cd, Cr, Cu, Ni, Pb, Zn.

Significant correlations (p<0.01) in order of correlation coefficients are noted (Figure 5.24) as Pb/Cr (r= 0.904 p<0.01); Zn/Pb (r=0.82, p<0.01); Zn/Cr (r=0.81 p<0.01) Cu/ Cr (r= 0.77 p<0.01);Pb/Cu (r=0.76 p<0.01); Zn/Cu (r=0.71 p<0.01); Ni/ Cd(r=0.58 P<0.01); Ni/ Cr (r=0.52 p<0.01), Cd/Cr (r=0.51 p<0.01); Pb/Cd (r=0.46 p<0.01). Significant correlations (p<0.05) were seen for Cd/ Cu (r=0.45 p<0.05) and Zn/Cd (r=0.41 p<0.05). There was no significant relationship observed between Ni/ Pb (p>0.05); Ni/ Zn (p>0.05) Ni/Cu (p>0.05)

5.5.2 Principal Component Analysis

A principal component analysis (PCA) was conducted on the metal data to determine groupings of metals within the system. Such multivariate techniques (including PCA) have been widely used within geochemical applications to identify not only pollution sources but also distinguish between anthropogenic and natural sources (Dragović, and Mihailović, 2009; Yang *et al*., 2014). PCA has been shown to reduce the dimensionality of the data and extract a small number of Principal Components (independent factors) which aid the determination of relationships between variables (Ruiz *et al.,* 1998; Yang *et al.,* 2009; Hu *et al*., 2011). The PCA for the heavy metals is shown in Table 5.11.

Table 5.11: Eigenvalues for the first 6 Components. (Note only 2 Principal Components > 1).

Eigenvalue	**3.3808**	**1.0924**	0.8343	0.4232	0.1668	0.1025
Proportion	**0.563**	**0.182**	0.139	0.071	0.028	0.017
Cumulative	**0.563**	**0.746**	0.885	0.955	0.983	1.000

The first two eigen values accounted for ~75% of the total variation and on further observation of the eigen vectors (Table 5.11) and the loading plot (Figure 5.25) it can be demonstrated that Cr, Cu, Pb and Zn have patterns of concentration closely associated with each other and the first Component, accounting for 56% of the total variance. The second Principal Component includes Cd and Ni and therefore suggests there were potentially two discrete sources of these two groups of metals represented by the first two Principal Components.

Table 5.12: Component Loadings (correlation between each variable and each component). Significant p<0.05 correlations are in bold).

Variable	PC1	PC2
Cd	0.198	**0.584**
Cr	**0.518**	0.059
Cu	**0.432**	-0.266
Ni	0.164	**0.724**
Pb	**0.498**	-0.076
Zn	**0.481**	-0.235

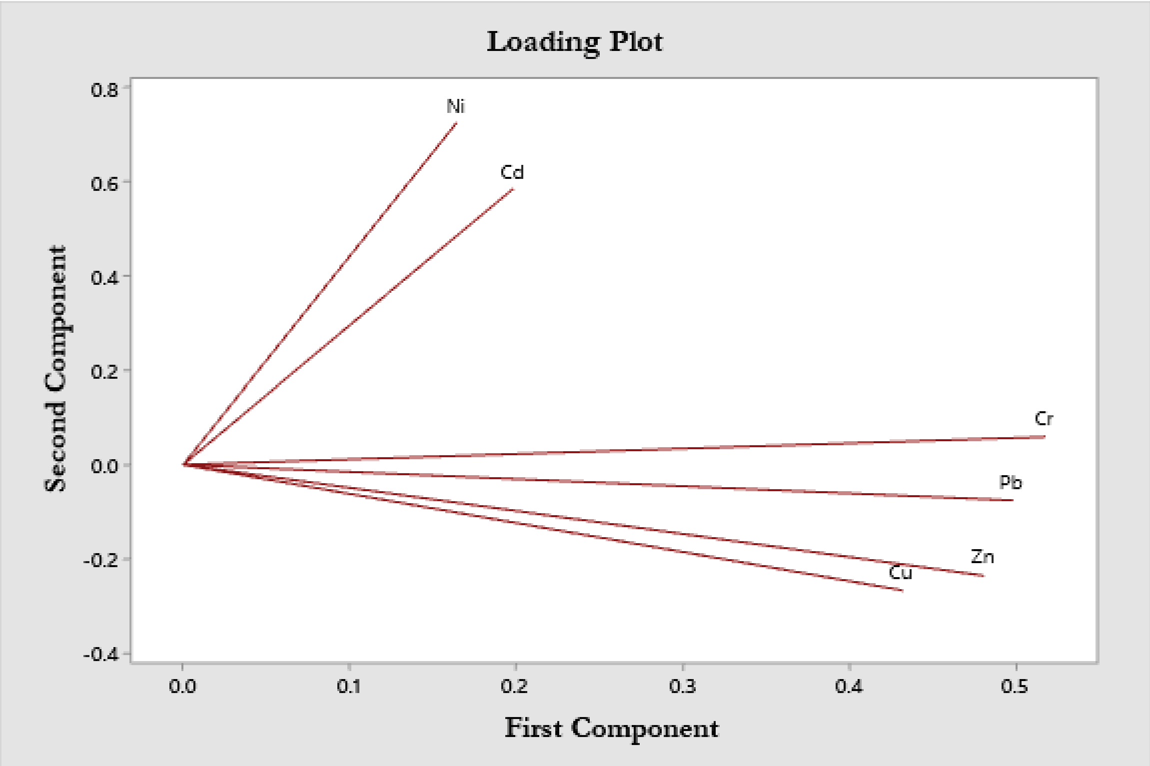

Figure 5.25: PC loading plot showing separation of 2 metal groups Group 1 (Cr, Cu, Pb, Zn).

5.5.3 Emerging contaminants

Emerging contaminants or "contaminants of emerging concern" is a term used to describe pollutants that have been detected in water bodies but, which are not yet regulated under current environmental laws. In addition, they may cause human health or ecological impacts. The concentrations of the two emerging contaminants (Ga and In) are reported in Table 5.13.

Table 5.13: Mean and median concentrations (mg kg^{-1}) of Gallium (Ga) and Indium (In).

Metal	Mean		Range		SE Mean		Median	
	SUDS	Background	SUDS	Background	SUDS	Background	SUDS	Background
Ga	4.1	4.9	0.1-8.3	3.9-6.3	0.4	0.4	4.5	4.8
In	5.3	3.8	1.73-12.28	1.8-4.9	0.6	0.5	5.6	4.1

Both Gallium and Indium were observed in SUDS and background samples. Ga and In are recognised as emerging contaminants which are likely to accumulate in soils due their low solubility in comparison to other trace elements such as Cd, Cu and Zn (Jensen *et al*., 2018). Their presence within the SUDS and surrounding soils and sediments is of interest and concern, and potentially requires further investigation but that is beyond the scope of this research. Gallium is naturally present at 3-70 mg kg^{-1} across a range of soil types, and Indium at 0.01-0.5 mg kg^{-1} (Kabata-Pendias & Mukherjee, 2007).

5.5.4 Heavy Metals and Particle SSA relationships

A Spearman's rank correlation was conducted for Cd, Cr, Cu, Ni, Pb and Zn with SSA. No significant correlation ($P<0.05$) was found. Surface area is important in controlling sediment trace element concentrations as they have a good capacity however, the relationship between SSA and trace element concentration is not often linear (Horowitz, 1991).

5.6 Radionuclides

5.6.1 Introduction

Both fallout and lithogenic radionuclides have been used as tracers in numerous soil erosion studies (Chapter 2.4). Within the context of this research, radionuclides were measured and used to look at potential sources of the sediment entering the ponds.

The gamma-emitting radionuclides which were detected, and their origins are shown in Table 5.14.

Table 5.14: Radionuclides used within this study (Foster et al., 2007).

Isotope	Half Life	Origin	Measured Decay Energy (Kev)	Notes
$^{210}Pb_{tot}$*	22.26 yr	Atmospheric fallout	46.52	Atmospheric from 222 Rn (Radon Gas) ^{226}Ra is formed from the ^{238}U decay series.
^{137}Cs	30 yr	Fission: Weapons fallout and nuclear accidents	661.62	First occurrence, 1954 with peak at 1963; Chernobyl 1986, Fukushima, 2008
^{234}Th	24.1 day	Natural	63.29	^{238}U decay series
^{226}Ra**	1600 yr	Natural	295 & 351	^{238}U decay series
^{235}U	7.04×10^{8}yr	Natural	185.72	^{235}U decay series
^{228}Ac	6.14 hr	Natural	338.40, 911.07	^{232}Th decay series
^{212}Pb	10.6 hr	Natural	238.63	^{232}Th decay series
^{40}K	1.28×10^{9} yr	Natural	1460.75	Primordial

* Unsupported Pb-210 ($^{210}Pb_{un}$) is derived from $^{210}Pb_{tot}$ – ^{226}Ra

** Ra-226 does not emit a gamma ray and is measured indirectly from its daughter Pb-214 at 295 and 351 keV

The range of radionuclides measured at the sampling sites over the sampling period is displayed in Table 5.15. While there is not a definable LOD for radionuclides, as the uncertainty in the measurements is a function of activity and count time, nuclide activities have been reported when the count is at least twice the counting error. ^{137}Cs was only detectable in 15 samples.

Table 5.15: Range of radionuclide activities reported in tube samplers at Upton. All values in mBq g^{-1}.

	^{210}Pb$_{un}$	^{226}Ra	^{137}Cs	^{228}Ac	^{40}K	^{234}Th	^{235}U	^{212}Pb
Min	33.74 ± 2.60	5.80 ± 0.71	0.57 ± 0.20	6.30± 0.86	83.50± 3.70	3.84± 1.10	1.11± 0.60	16.23± 0.70
Max	620.93 ± 0.36	66.80± 4.02	2.48 ± 0.70	73.70 ± 5.60	734.10 ± 63.40	35.80 ± 2.90	6.13 ± 0.40	66.10 ± 2.40

As there were large variations in the measured activities between sites and between sampling periods a Principal Component Analysis (PCA) was used for each of the sampling periods in order to assess whether there were any relationships between the sites with regards to the activities of the radionuclides.

A PCA biplot is shown for each of the sampling times in Figures 5.26-5.31 below. These plots can be used to assess the data structure and the loadings on the first two components, the second principal components score versus the first principal component as well as loadings for both components.

February 2016

The data from the samplers in February 2016 are presented in Table 5.16 and the corresponding PCA plot is shown in Figure 5.26.

Table 5.16: February 2016 radionuclide activities in each of the samplers.

	Unsupp Pb-210 mBq g^{-1}	Ra-226 mBq g^{-1}	Cs-137 mBq g^{-1}	Ac-228 mBq g^{-1}	K-40 mBq g^{-1}	Th-234 mBq g^{-1}	U-235 mBq g^{-1}	Pb-212 mBq g^{-1}
CP	178.15	14.176	0.00	35.915	456.66	35.81	6.131	35.62
PL	33.74	37.344	0.00	27.087	448.15	14.48	2.564	30.64
HDS	455.03	43.944	0.00	7.578	280.55	9.81	0.000	16.53
HD	85.80	54.159	1.92	36.558	548.05	25.02	4.005	46.18
TB	59.22	58.339	0.00	36.624	373.77	10.87	0.986	43.51

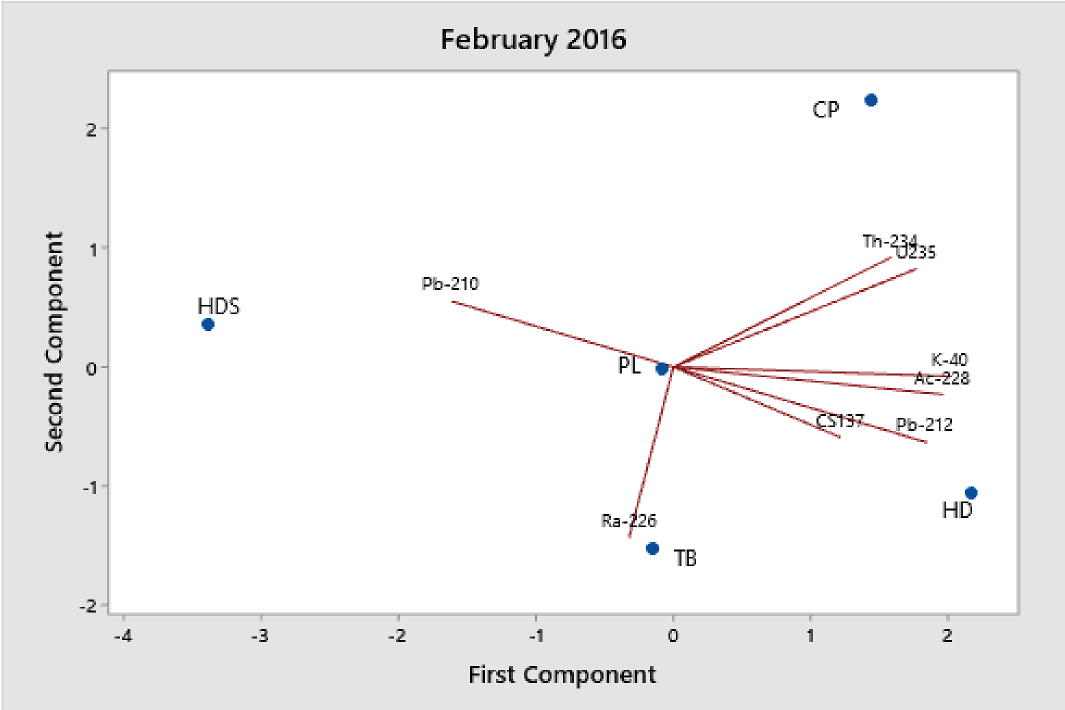

Figure 5.26: PCA biplot of radionuclides and sampler sites February 2016.

The position of the samplers on the plot indicates their grouping in terms of the radionuclides. $^{210}Pb_{un}$ is a strong influence on the second component and was grouped near HDS (the swale) which had high activities of $^{210}Pb_{un}$. ^{137}Cs was only found in one sampler (HD) on this date.

March 2016

The data from the samplers in March 2016 are presented in Table 5.17 and the corresponding PCA plot is shown in Figure 5.27.

Table 5.17: March 2016 radionuclide activities in each of the samplers.

	Unsupp Pb-210 mBq g^{-1}	Ra-226 mBq g^{-1}	Cs-137 mBq g^{-1}	Ac-228 mBq g^{-1}	K-40 mBq g^{-1}	Th-234 mBq g^{-1}	U-235 mBq g^{-1}	Pb-212 mBq g^{-1}
CP	220.73	6.895	0.00	45.22	612.20	8.52	2.94	43.07
PL	85.50	28.70	0.00	33.39	481.97	16.61	4.08	43.94
HDS	258.09	14.14	0.85	11.91	261.10	32.27	3.45	20.49
HD	126.32	23.71	1.05	34.40	483.02	18.85	2.03	46.69
TB	75.95	23.95	0.00	27.59	482.52	17.51	2.97	37.04

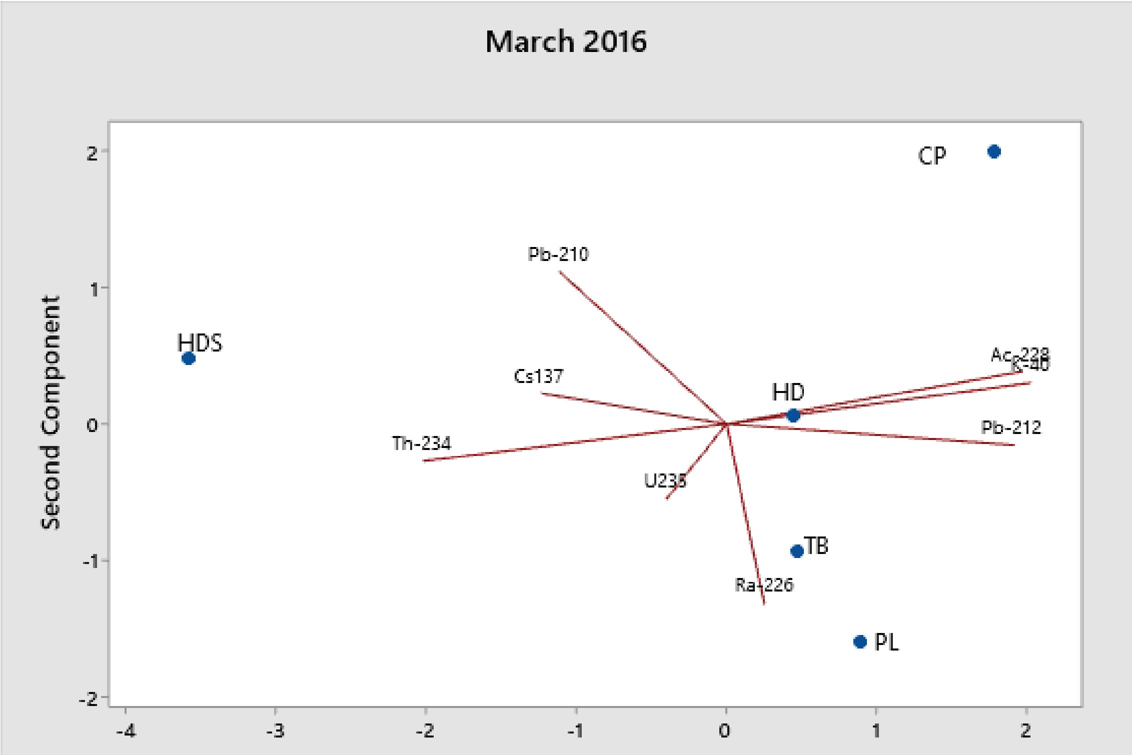

Figure 5.27: PCA biplot of radionuclides and sampler sites March 2016.

^{137}Cs was only measured in both HDS and HD samplers. ^{137}Cs is reported in measurable activities in many studies of UK soils (Walling and Foster, 2016) and due to its relative immobility has been used in soil erosion tracer studies. It is present in rural soils with higher activities near the soil surface in grassland soils but in arable soils ^{137}Cs is mixed to the depth of ploughing. The presence of ^{137}Cs in subsoils and RDS is rare unless it has been transported there by e.g., trafficking agricultural vehicles. It's presence in both HDS and HD not only implies connectivity between the swale and that area of pond 1, but also indicates that local sediment, potentially from agricultural activities, is being deposited in this area of the SUDS. $^{210}Pb_{un}$ was measured at all sites although the highest activities were seen at HDS and the CP.

March 2017

The data from the samplers in March 2017 is presented in Table 5.18 and the corresponding PCA plot is shown in Figure 5.28.

Table 5.18: March 2017 radionuclide activities in each of the samplers.

	Unsupp Pb-210 mBq g^{-1}	Ra-226 mBq g^{-1}	Cs-137 mBq g^{-1}	Ac-228 mBq g^{-1}	K-40 mBq g^{-1}	Th-234 mBq g-1	U-235 mBq g^{-1}	Pb-212 mBq g^{-1}
CP	174.55	9.78	1.01	10.611	526.97	21.99	5.473	50.44
PL	164.38	38.88	0.00	43.206	472.49	15.10	4.401	44.35
HDS	133.91	20.04	0.68	8.535	165.45	3.84	1.452	16.23
HD	159.89	30.8	1.35	37.049	445.51	25.03	2.318	43.96
TB	76.25	40.5	0.00	36.910	403.72	23.48	1.387	40.41

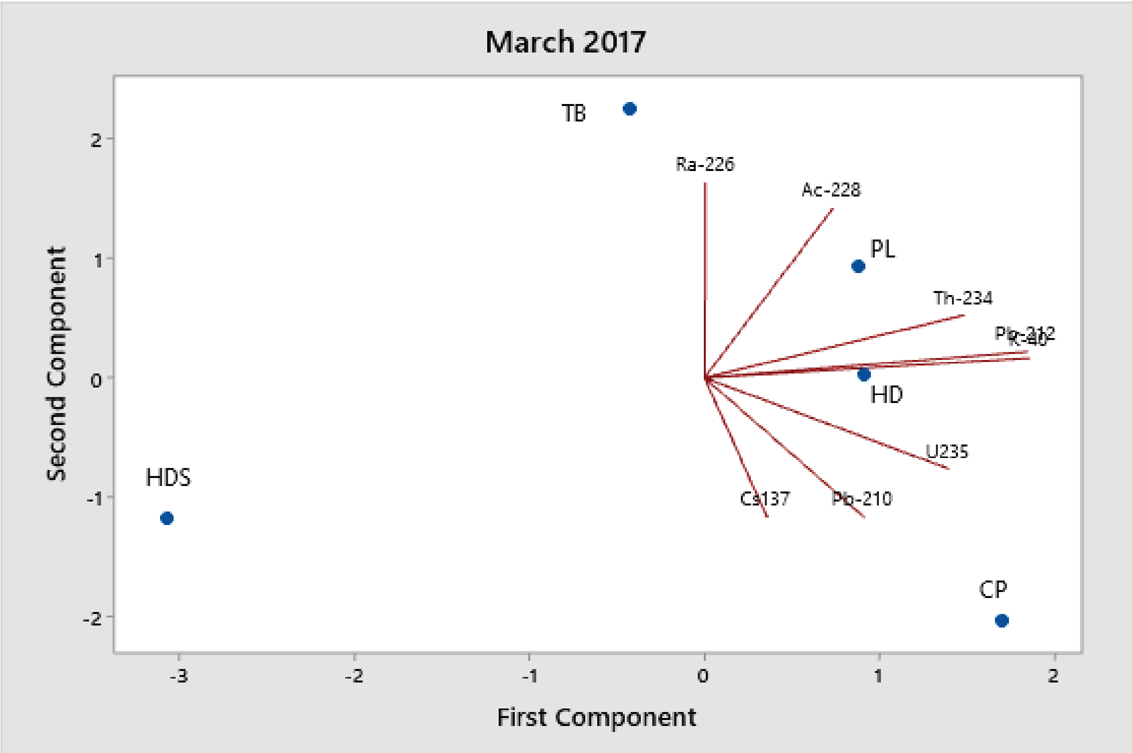

Figure 5.28: PCA biplot of radionuclides and sampler sites March 2017.

^{137}Cs was again observed in HDS (0.68 mBqg^{-1}), HD (1.35 mBqg^{-1}) and at CP (1.01 mBqg^{-1}). HDS had the lowest activities of geogenic radionuclides but given that the swale receives input from RDS this result was to be expected.

October 2017

The data from the samplers in October 2017 are presented in Table 5.19 and the corresponding PCA plot is shown in Figure 5.29.

Table 5.19: October 2017 radionuclide activities in each of the samplers.

	Unsupp Pb-210 mBq g⁻¹	Ra-226 mBq g⁻¹	Cs-137 mBq g⁻¹	Ac-228 mBq g⁻¹	K-40 mBq g⁻¹	Th-234 mBq g⁻¹	U-235 mBq g⁻¹	Pb-212 mBq g⁻¹
CP	620.93	5.803	4.07	6.298	317.18	17.95	1.113	50.97
PL	328.26	41.046	0.00	39.003	379.32	15.74	1.527	46.01
RO	169.93	31.557	0.57	34.323	556.31	33.18	0.000	39.41
HD	174.49	55.065	1.35	44.169	660.92	0.00	1.487	43.40

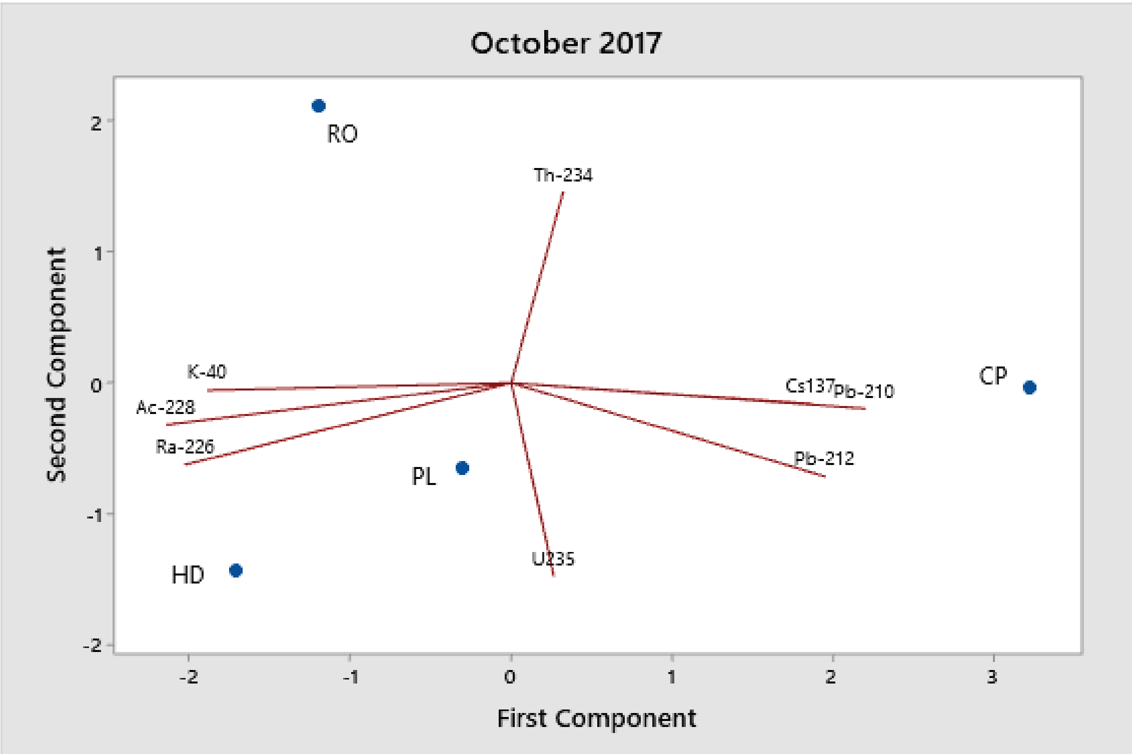

Figure 5.29: PCA Biplot of radionuclides and sampler sites October 2017.

^{137}Cs and $^{210}Pb_{un}$ were dominant within the CP sampler and were also detected in RO and HD. Higher levels of geogenic nuclides were found in the middle of pond 1 (PL and RO).

February 2018

The data from the samplers in February 2018 are presented in Table 5.20 and the corresponding PCA plot is shown in Figure 5.30.

Table 5.20: February 2018 radionuclide activities in each of the samplers.

	Unsupp Pb-210 mBq g^{-1}	Ra-226 mBq g^{-1}	Cs-137 mBq g^{-1}	Ac-228 mBq g^{-1}	K-40 mBq g^{-1}	Th-234 mBq g^{-1}	U-235 mBq g^{-1}	Pb-212 mBq g^{-1}
CP	593.14	36.069	1.03	27.406	451.40	11.44	1.120	31.38
PL	187.02	14.105	0.00	41.891	530.31	9.21	0.000	50.42
RO	190.03	40.750	0.00	30.874	458.99	15.56	2.849	47.83
HD	215.14	36.202	1.33	27.587	465.09	26.00	2.542	39.59
TB	47.10	32.056	1.52	26.439	469.06	20.89	1.361	26.63
FB	527.38	66.840	0.00	33.883	734.05	0.00	0.000	66.11

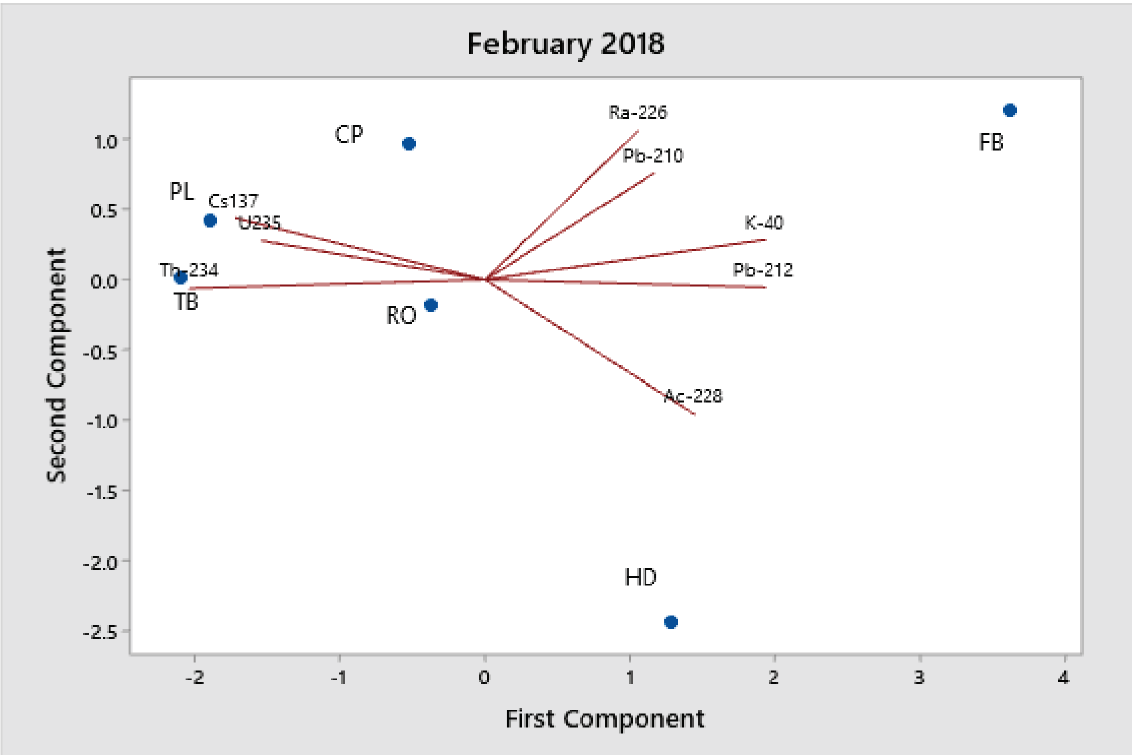

Figure 5.30: PCA Biplot of radionuclides and sampler sites February 2018.

^{137}Cs was measured in CP, HD and TB (for the first time during the sampling period). CP and FB had high activities of $^{210}Pb_{un}$ (Table 5.20). No ^{234}Th or ^{235}U was measured at FB but high levels of ^{40}K were recorded.

July 2018

The data from the samplers in July 2018 are presented in Table 5.21 and the corresponding PCA plot is shown in Figure 5.31.

Table 5.21: July 2018 radionuclide activities in each of the samplers.

	Unsupp Pb-210 mBq g^{-1}	Ra-226 mBq g^{-1}	Cs-137 mBq g^{-1}	Ac-228 mBq g^{-1}	K-40 mBq g^{-1}	Th-234 mBq g^{-1}	U-235 mBq g^{-1}	Pb-212 mBq g^{-1}
CP	346.10	9.941	0.00	37.582	401.94	28.12	4.231	39.79
PL	175.18	37.089	0.00	31.458	370.32	17.15	4.036	41.58
RO	219.54	29.675	0.71	21.324	631.51	25.11	1.590	34.31
HD	178.36	43.489	2.26	35.730	83.48	11.40	3.420	40.48
TB	113.48	40.241	0.00	29.344	487.02	16.05	5.219	40.54
FB	379.55	18.494	2.48	73.737	520.64	13.73	3.507	56.75

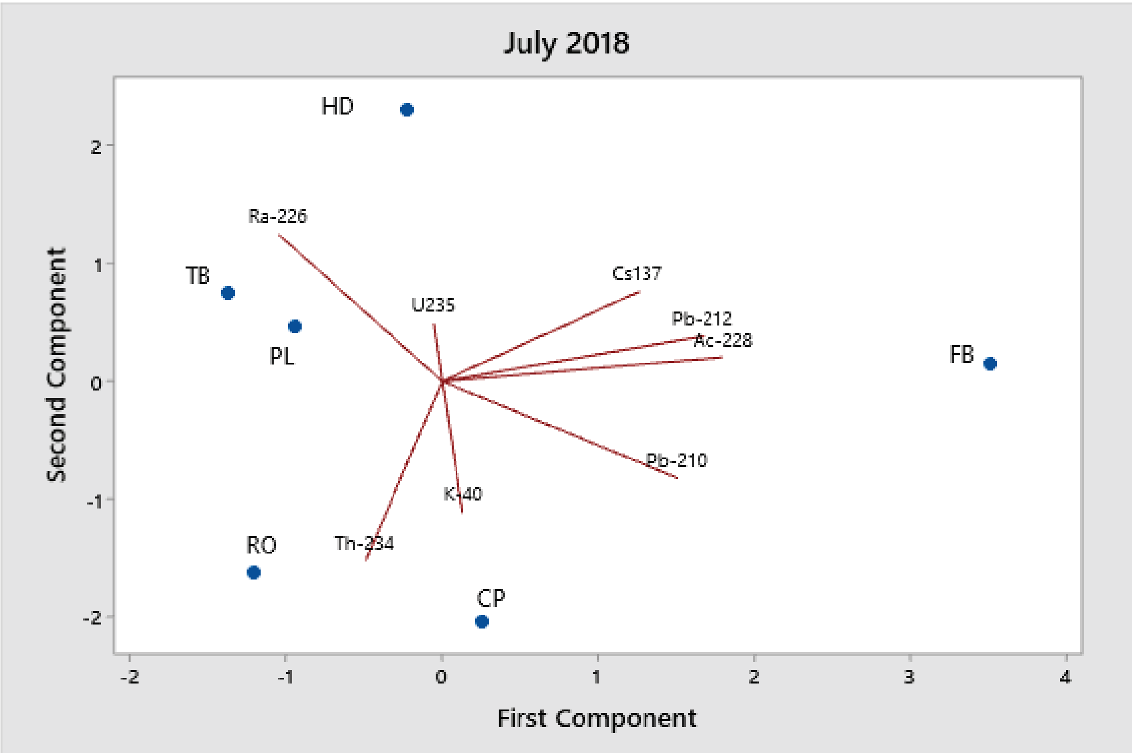

Figure 5.31: PCA biplot of radionuclides and sampler sites July 2018.

^{137}Cs was measured at RO, HD, and FB. Comparatively low levels of ^{40}K were measured in HD but high levels of ^{226}Rn. ^{210}Pb$_{un}$ was higher at CP, RO, and FB.

February 2016- July 2018

PCA was run for the entire data set over the sampling period to look for any trends in the data. The Scree plot (with sites) and the loading plots (with the radionuclides) are shown in Figures 5. 32 and 5.33 respectively.

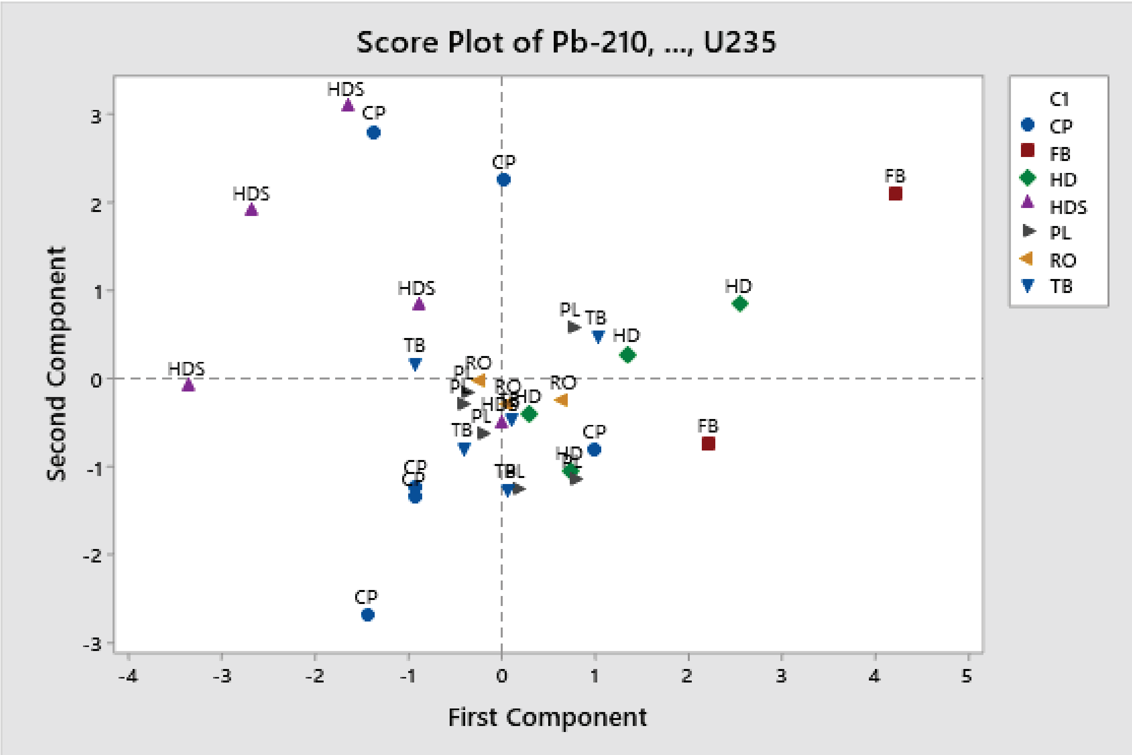

Figure 5.32: PCA score plot for all the sites at all the sampling dates.

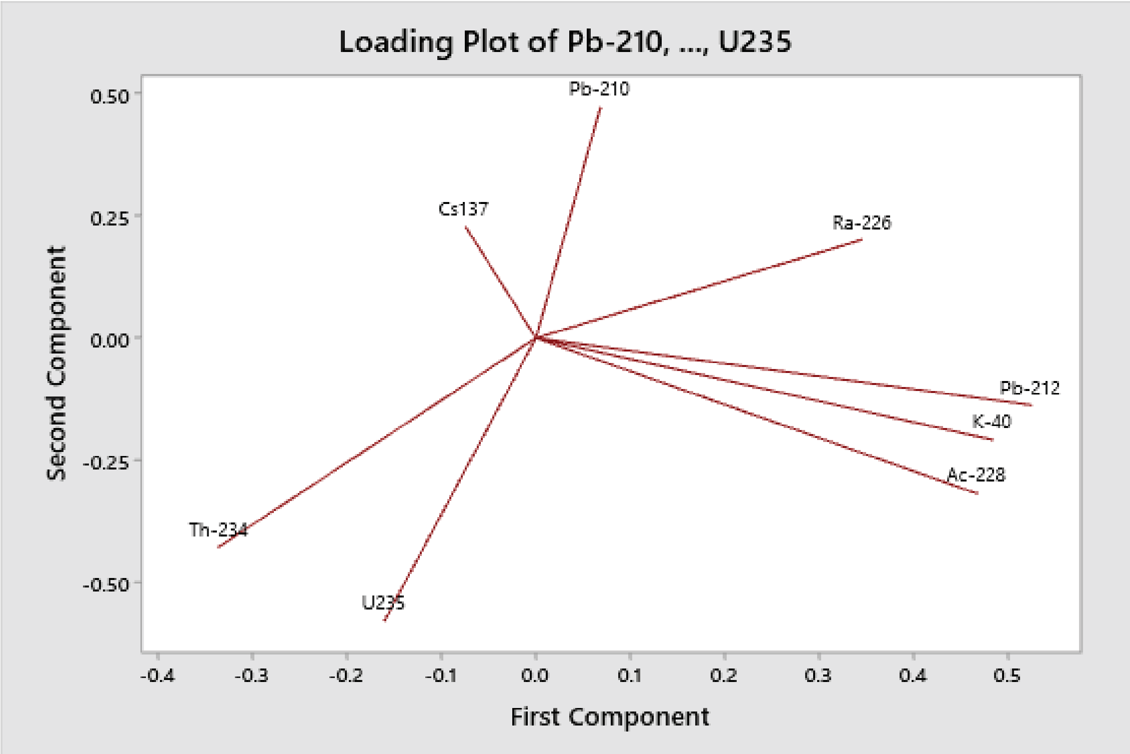

Figure 5.33: PCA loading plot of radionuclides.

Figure 5.33 shows the loading plot of the PCA with the corresponding Figures reported in Table 5.22 and 5.23.

Table 5.22: Eigen analysis of the correlation matrix

Eigenvalue	2.1561	1.7302	1.4523	0.7911	0.6331	0.5174	0.4405	0.2792
Proportion	0.270	0.216	0.182	0.099	0.079	0.065	0.055	0.035
Cumulative	0.270	0.486	0.667	0.766	0.845	0.910	0.965	1.000

Table 5. 23: Principal Component Analysis and eigen vectors of the radionuclides at the study site with regard to all samplers. (Statistically significant correlations, p<0.05, are highlighted in bold).

Variable	PC1	PC2	PC3	PC4
Pb 210	0.069	0.473	-0.458	0.310
Ra-226	0.346	0.201	0.450	-0.353
Cs137	-0.075	0.229	-0.614	-0.469
Ac-228	0.468	-0.319	-0.098	-0.296
K-40	0.485	-0.209	-0.049	0.575
Th-234	-0.337	-0.428	-0.204	0.252
Pb-212	0.526	-0.137	-0.367	-0.063
U235	-0.161	-0.580	-0.150	-0.270

There are four distinct loadings (groups) within the PCA (Table 5.23, Figure 5.33). PC1 is influenced by ^{228}Ac, ^{40}K and ^{212}Pb, PC2 is loaded with the fallout nuclides and was consistently higher in sites CP, HDS and HD (which was the only sampler to record above LOD ^{137}Cs activities over the entire sampling period). The third component was attributed to ^{226}Ra which behaves like other Group 2 alkaline earth metals (Ca, Sr, and Ba) and ion exchange processes play an important role in the movement of ^{226}Ra in soil (Fesenko *et al.*, 2014). Component 4 consists of ^{40}K and ^{234}Th.

The presence of fallout radionuclides is of particular importance as they provide an indication of sources of sediment. ^{137}Cs has already been discussed and there is likely to have been a contribution from topsoil to the sediment in parts of the SUDS, the swale and potentially from the local farm (grazing) which is located in close proximity to the SUDS system. The presence of unsupported ^{210}Pb$_{un}$ is expected in those samplers that are closest to road surfaces or have direct links with road surfaces. as ^{210}Pb$_{un}$ constantly falls out of the atmosphere and accumulates on road surfaces between storm events (Johansson, 2008).

5.6.2 Radionuclides and Heavy Metals

A Spearman's rank correlation between the radionuclides and the heavy metals within the samples was carried out and significant ($p<0.01$; $p<0.05$) correlations are reported in Table 5.24.

Table 5.24: Pairwise Spearman Rank correlations.

Sample 1	Sample 2	N	Correlation	95% CI for ρ	P-Value
Cu	Pb-210	31	0.423	(0.064, 0.684)	0.018
Ni	Pb-210	31	-0.453	(-0.705, -0.100)	0.010
Zn	Pb-210	31	0.389	(0.027, 0.661)	0.030
Pb-212	Cr	31	0.606	(0.291, 0.803)	0.000
Pb-212	Cu	31	0.465	(0.113, 0.713)	0.008
Ra-226	Ni	31	0.466	(0.114, 0.714)	0.008
Pb-212	Pb	31	0.558	(0.227, 0.773)	0.001

Positive significant correlations were observed between the metals Cu and Zn and ^{210}Pb$_{un}$. As the most likely anthropogenic sources of Cu and Zn are traffic emissions (brakes and tyre abrasion respectively) it is possible that $^{210}Pb_{un}$ could be used to identify areas of the SUDS which receive atmospheric input (a component of RDS) which is then washed into the wetland system. $^{210}Pb_{un}$ becomes readily attached to airborne particulate material (Johansson, 2008) from traffic and other emissions, and is then removed from the atmosphere by precipitation or dry deposition. In surface waters $^{210}Pb_{un}$ is rapidly scavenged by iron and manganese oxides and biogenic particles that are then transported downward in the water column by particle settling. $^{210}Pb_{un}$ can be remobilised in significant amounts from lake sediments to the water column under reducing conditions (Benoit and Hermon, 1991) which given the stagnant nature of the ponds during the drier weather could be possible. There is potential for this to occur within the SUDS if pools of water are formed thereby inducing reducing conditions. This may be the case in pond 1 where sediment levels and flow were varied, and, during dry weather spells a series of unconnected pools were formed.

A significant negative correlation was observed between $^{210}Pb_{un}$ and Ni although Ni does have a positive relationship with ^{226}Ra. ^{226}Ra is fairly mobile however, the mobility of ^{226}Ra is dependent on salinity and in rivers and groundwaters radium is strongly adsorbed onto particles (Fesenko *et al*., 2014).

While the significant nature of some of the correlations are reported with regard to ^{212}Pb, it should also be noted that ^{212}Pb has a significant relationship with SSA ($P<0.01$) and therefore some of the assumptions about these relationships may in fact be due to particle size and the specific surface area.

5.7 Environmental Magnetism

As noted in the literature review, many studies have attempted to use several; mineral magnetic signatures as surrogates for heavy metal concentrations (either individually or cumulatively) due to the speed and relatively low cost at which magnetic measurements can be made. In urban systems, the magnetic mineralogy will likely be dominated by fine grained magnetite type minerals which will generally have high values of χ_{LF} and SIRM and will exhibit a soft magnetic remanence with S ratios >0.7.While the main purpose of this analysis was to evaluate the potential for using magnetism as a surrogate for heavy metals, the next section will briefly interpret the magnetic signatures in terms of magnetic mineralogy and magnetic grain size.The magnetic measurements of the sediment samples collected from the tube samplers are given in Table 5.25. Please note the coding of the samplers is as before.

Table 5. 25: Magnetic measurement results of sediment from tube samplers at Upton.

Magnetic parameters		**CP**	**PL**	**RO**	**HD**	**HDS**	**TB**	**FB**
χ_{LF} 10^{-6} m^3 kg^{-1}	Range	0.52-1.83	1.24-3.62	1.92-1.99	0.56-1.08	0.37-0.72	0.49-0.79	0.97-1.46
χ_{FD} 10^{-9} m^3 kg^{-1}	Range	7.19-88.6	62.87-146.62	80.7-106.78	35.55-95.4	8.9-45.5	4.26-28.28	30.86-157.48
$\chi_{FD\%}$	Range	0.9-5.88	4.3-5.55	4.2-5.37	3.35-10.28	2.6-6.77	0.72-5.8	3.37-11.76
ARM 10^{-3} Am^2 kg^{-1}	Range	0.002-0.64	0.09-0.8	0.01-0.50	0.004-0.42	0.16-0.18	0.005-0.36	0.05-0.41
SIRM 10^{-3} Am^2 kg^{-1}	Range	10.37-25.18	17.38-25.22	23.98-24.92	9.95-16.51	6.56-12.63	7.02-12.5	1.15-192
HIRM 10^{-3} Am^2 kg^{-1}	Range	0.76-2.1	1.55-3.6	2.13-2.29	0.79-1.39	0.25-0.71	0.52-1.3	1.15-1.92
χ_{ARM} $10^{-6}$$m^3$ kg^{-1}	Range	0.05-20.7	0.2-25.2	0.36-15.8	0.13-14.67	5.05-5.78	0.16-11.27	1.6-13.14
S ratio	Range	0.75-0.84	0.77-0.82	0.79-0.8	0.74-0.78	0.8-0.88	0.58-0.82	0.73-0.76

Mass specific susceptibility (χ_{LF}) is usually measured to determine the concentration of ferrimagnetic minerals in environmental samples (Thompson and Oldfield, 1986). As a measurement it does not discriminate between ferrimagnetic grain sizes nor mineral types. As well as including primary materials such as titanomagnetites with geological origins, secondary minerals such as magnetite and maghemite can be derived through chemical and bacterial processes or produced during combustion and are therefore associated with polluted dusts (Thompson and Oldfield, 1986). The presence of secondary ferrimagnetic minerals (SFMs) is important within soil and sediment studies and requires a range of magnetic measurements. The main evidence for the presence of such SFMs has been shown as the loss of magnetic susceptibility between two AC frequencies which is assumed to detect the presence of superparamagnetic (SP) grains lying within a small band of grain sizes estimated to lie between 0.018-0.02 μm in diameter (Dearing *et al*., 1996).

Frequency dependent susceptibility approximates to the total concentration of the stable single domain (SSD) and very fine pseudo single domain (PSD) ferrimagnetic grains. Levels of frequency-dependent magnetic susceptibility, χ_{FD}%, vary between and within sites (Table 5.25). Low MD assemblages are common in polluted and urban sediments due to anthropogenic Fe input from combustion and industrial processes. High values represent SP grains derived from natural weathering of parent material (Dearing *et al.*, 1996). Values of χ_{FD}% < 2.0% indicates virtually no SP grains; values between 2.0 and 10.0% are indicative of an admixture of SP and coarser non-SP grains; while values between 10.0 and 14.0% indicates virtually all SP grains (Dearing, 1999). Low values are indicative of predominantly multi-domain (MD) magnetic grain size assemblages. Coarse MD grains contribute significantly to the decrease of high frequency susceptibility, therefore as the percentage χ_{FD}% approaches zero the sample is increasingly dominated by MD assemblages (Figure 5.34). Low MD assemblages are common in polluted and urban sediments due to anthropogenic Fe input from combustion and industrial processes (Crosby *et al.*, 2014) and suggest that the sediment is enriched with ferrimagnetic grains probably because of anthropogenic activities.

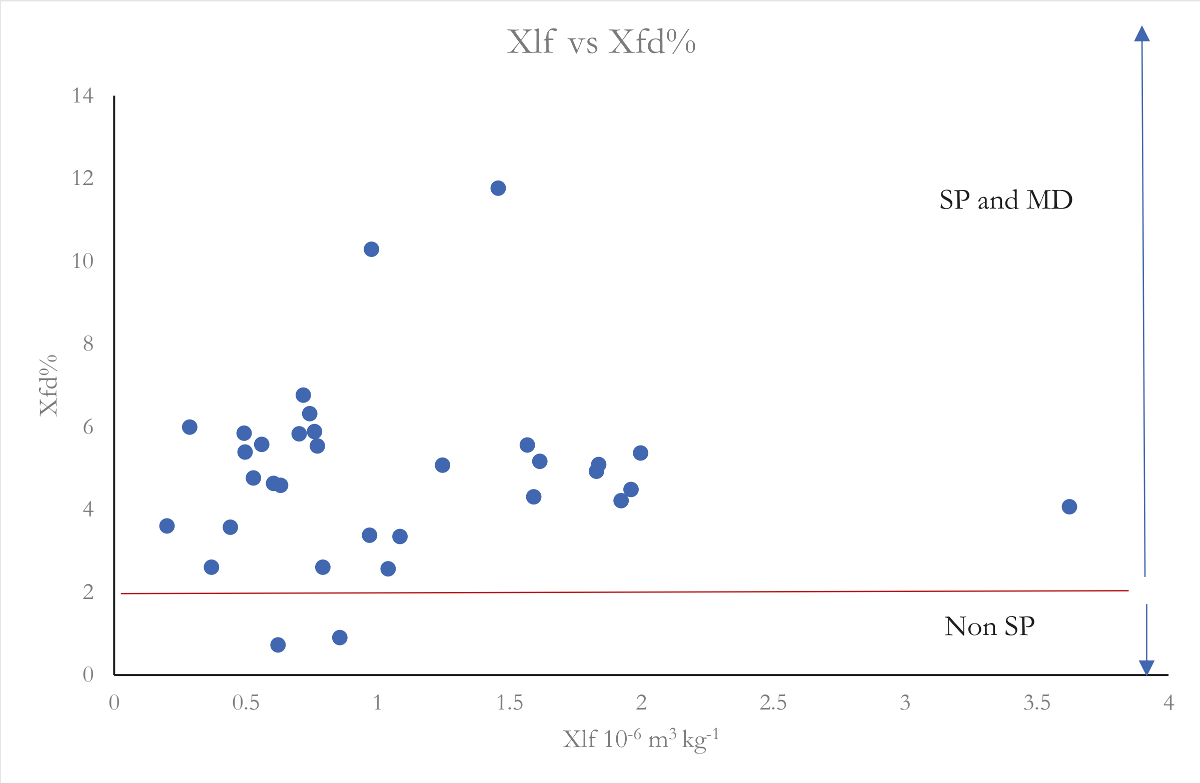

Figure 5.34: χ_{LF} against χ_{FD} %. Non-superparamagnetic, SP and MD boundaries as defined by Walden et al., (1999).

SIRM, HIRM and the S ratio are magnetic mineralogy dependent parameters. The S ratio is a dimensionless parameter that indicates the ratio of ferrimagnetic to antiferromagnetic minerals; values close to 1 correspond to the predominance of ferrimagnetic minerals. The S ratio for the samples is shown in Table 5.25 and varied between 0.58-0.88.

Magnetic minerals in sediments can be identified as soft and hard fractions. The soft fraction has low coercivity and is expected to approximate to the concentration of magnetite (Thompson and Oldfield 1986); the hard fraction has high coercivity and can be used to estimate the total concentration of canted antiferromagnetic minerals (such as hematite) (Oldfield and Richardson 1990). The relatively high values of S ratio and Soft IRM within the samples indicated the presence of ferrimagnetic minerals. The correlation of magnetic susceptibility with SIRM was statistically significant ($p<0.01$; $r= 0.986$) (Figure 5.35). The relatively high correlation indicated the magnetic minerals within the wetland are mainly ferrimagnetic (Gang *et al.*, 2013; Wang *et al.*, 2020).

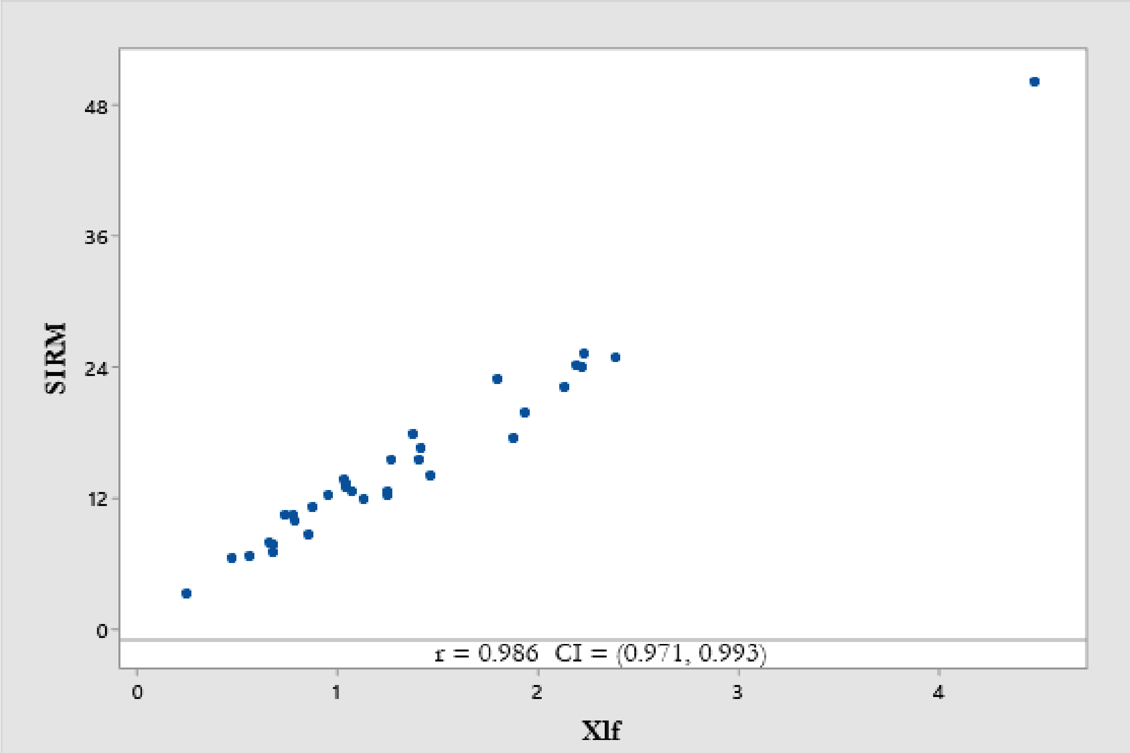

Figure 5.35: Correlation between χ_{LF} vs SIRM , r=0.986 p<0.01.

5.7.1 Magnetism and Metals

Magnetic measurements as a proxy indicator of heavy metal pollution have been reported in the literature (Lu *et al*., 2007; Hanesch *et al*., 2001; Jordanova *et al*., 2004; Strzyszcz and Magiera, 1998). Significant correlations between magnetic susceptibly of contaminated soils and the presence of combustion related pollutants (Pb Zn, Fe) has also been observed (Morris *et al*., 1995; Hay *et al.*, 1997; Heller *et al*., 1998; Jordanova *et al*., 2004; Lu and Bai, 2006; Wang, 2013b; ;). Other studies have also examined other magnetic parameters such as ARM and SIRM (Lu and Bai, 2006; Lu *et al.*, 2007; Yang *et al*., 2009; Wang *et al*., 2012; Zhang *et al*., 2012; Zhu *et al*., 2001) and the corresponding relationships with heavy metals (Cu, Cd, Ni, Pb and Zn). Spearman's rank correlation was performed to investigate any relationships between the heavy metals and magnetic parameters (Table 5.26).

Table 5.26: Correlations between χ_{FD} (%), χ_{LF}, SIRM and ARM.

N=34	χ_{FD} (%)	χ_{LF}	SIRM	ARM	HIRM
Cd	0.294	0.302	0.274	0.259	0.388**
Cr	0.381**	0.320	0.312	0.266	0.324
Cu	0.313	0.349**	0.290	0.286	0.333**
Ni	0.063	0.066	0.072	0.08	0.183
Pb	0.638*	0.608*	0.561*	0.365**	0.596*
Zn	0.363**	0.344**	0.269	0.230	0.313

*significant at $p<0.01$

**significant $p<0.05$

The results show that there are significant correlations between χ_{FD} (%), Cr and Zn and a strong correlation with Pb ($p<0.01$). Regarding magnetic susceptibility, the correlation coefficients for Cu, Pb and Zn were 0.349, 0.608 and 0.344 respectively. Significant relationships have been reported widely in the literature (Lu and Bai, 2006). Pb, Cu and Zn exhibit similar chemical behaviour in solute form and tend to coprecipitate with hydrous oxides of Fe and Mn (Karimi *et al*., 2011) and all appear to be mainly associated with traffic.

Lu and Bai (2006) reported higher correlation coefficients of heavy metal content and ARM and SIRM in relation to Cu, Cd, Pb and Zn but in this study significant correlations were only found between SIRM, ARM and Pb, with a strong correlation ($p<0.01$) between SIRM and Pb.

A Principal Component Analysis (PCA) was carried out on $\chi_{LF,}$ SIRM, ARM, HIRM, Cd, Cr, Cu, Ni, Pb and Zn to observe any groupings between the magnetic parameters and the metals. The Eigen analysis of the correlation matrix is shown in Table 5.27.

Table 5.27: Eigen analysis of the Correlation Matrix.

Eigenvalue	**4.8753**	**1.8090**	1.348	0.7934	0.5953	0.3134	0.3134	0.0671	0.042	0.007
Proportion	**0.488**	**0.181**	0.113	0.079	0.060	0.036	0.031	0.007	0.004	0.001
Cumulative	**0.488**	**0.668**	0.782	0.861	0.921	0.957	0.988	0.995	0.999	1.000

The first two eigen values account for ca. 67% of the total variation in the data and on further inspection of the eigen vectors (Table 5.28) Pb χ_{LF}, SIRM and HIRM (shown in bold) explain 49% of the total variance. The second Principal Component includes Cr, Cu, Zn and Ni (shown in bold). The PCA also reflects the correlation results regarding Pb.

Table 5.28: Eigenvalues relating to PC1 and PC2.

Variable	PC1	PC2
Cr	0.304	**0.373**
Cu	0.294	**0.326**
Pb	**0.395**	0.223
Zn	0.297	**0.336**
χ_{LF}	**0.387**	-0.344
SIRM	**0.376**	-0.383
Cd	0.210	0.078
Ni	0.135	**0.392**
ARM	0.274	-0.289
HIRM	**0.384**	-0.289

5.8 Similarities between Sites

A PCA was used to identify a smaller number of Principal Components from the large data set encompassing all the measured parameters. The aim of the PCA was to explain the variance between sites using the fewest number of principal components. Table 5.29 shows the Eigen values from the correlation matrix. From these seven Principal Components were identified with eigen values greater than 1 (Table 5.30).

Table 5.29: Eigen analysis of the Correlation Matrix.

Eigenvalue	5.7717	2.4575	2.3294	1.9428	1.5302	1.1892	1.0126	0.7912	0.6914	0.5433
Proportion	0.289	0.123	0.116	0.097	0.077	0.059	0.051	0.040	0.035	0.027
Cumulative	0.289	0.411	0.528	0.625	0.702	0.761	0.812	0.851	0.886	0.913
Eigenvalue	0.4280	0.3956	0.3110	0.2213	0.1824	0.0875	0.0587	0.0330	0.0181	0.0051
Proportion	0.021	0.020	0.016	0.011	0.009	0.004	0.003	0.002	0.001	0.000
Cumulative	0.934	0.954	0.970	0.981	0.990	0.994	0.997	0.999	1.000	1.000

Table 5.30: Eigen vectors with strong associations shown in bold.

Variable	PC1	PC2	PC3	PC4	PC5	PC6	PC7
Pb-210	0.045	0.049	**-0.545**	0.008	0.095	0.176	-0.157
Ra-226	0.131	-0.189	0.053	0.456	-0.309	-0.253	-0.118
Cs137	-0.005	-0.076	**-0.349**	-0.311	-0.074	0.043	-0.200
Ac-228	0.146	-0.302	0.128	-0.012	0.334	-0.116	0.351
K-40	0.178	-0.224	0.092	0.185	0.283	0.356	0.284
Th-234	-0.016	0.107	0.286	-0.417	-0.100	0.475	-0.237
Pb-212	0.269	**-0.353**	-0.075	-0.029	0.289	0.070	-0.078
U235	-0.013	0.001	0.309	-0.409	0.303	-0.284	-0.208
Specific surface area	0.037	**-0.422**	0.036	0.038	0.134	0.218	-0.429
Cd	0.168	0.081	0.166	-0.019	-0.380	0.449	0.363
Cr	0.279	-0.116	0.174	-0.257	-0.242	0.006	0.061
Cu	0.251	-0.138	-0.129	-0.331	0.004	-0.176	0.158
Ni	0.093	**-0.303**	0.318	-0.026	-0.377	-0.272	-0.176
Pb	**0.359**	-0.044	-0.189	-0.112	-0.197	-0.027	-0.021
Zn	0.257	-0.122	**-0.369**	-0.158	-0.222	-0.099	0.120
χ_{LF}	**0.344**	0.296	0.033	0.111	0.065	-0.000	-0.204
LOI	-0.303	0.019	-0.094	-0.165	-0.122	-0.163	0.081
SIRM	**0.315**	0.343	0.060	0.085	0.088	-0.053	-0.212
HIRM	**0.354**	0.211	0.084	0.177	0.069	0.006	-0.214
ARM	0.214	0.321	0.041	-0.160	0.176	-0.240	0.290

The first component shows positive associations with χ_{LF}, HIRM, SIRM and Pb (shown in bold), which was demonstrated in Table 5.26 through the significant correlations observed with Pb and these magnetic parameters. The second component has negative associations with Ni, SSA and ^{212}Pb and the third component also has negative associations with ^{210}Pb$_{un}$, ^{137}Cs and Zn.

The first two components are displayed on the loading plot (Figure 5.36).

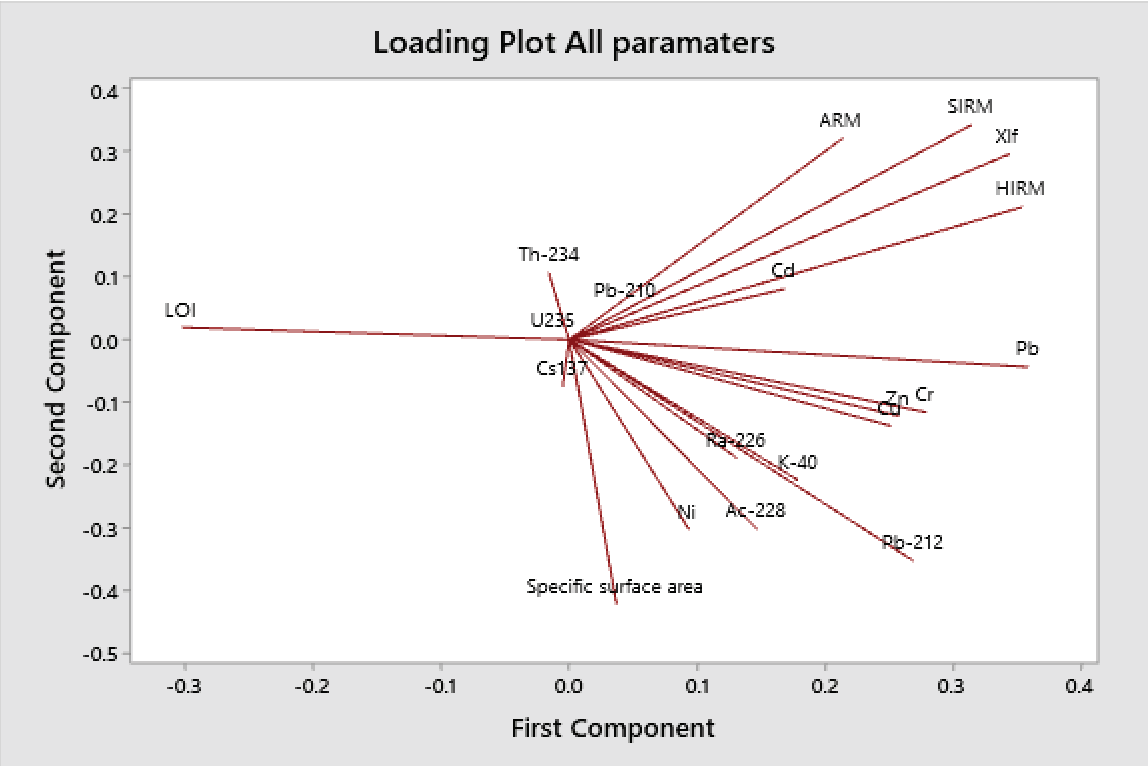

Figure 5.36: PCA loading plot for all the parameters at all the sites.

The PCA indicated that first component was positively associated with Pb, SIRM, HIRM and χ_{LF} . The second component had strong negative associations with ^{212}Pb, SSA and Ni while the third component had negative associations with ^{210}Pb$_{un}$, ^{137}Cs and Zn, which is potentially important in looking at the sites within SUDS which are most affected by atmospheric deposition of particles.

5.9 Discussion

One of the objectives of this chapter was *to* ***investigate the potential sources and connectivity of the sediment-associated contaminants between sites at different times of the year and under different hydrological conditions.*** Each month will be discussed in turn examining potential similarities and connectivity between sites.

5.9.1 February 2016

Organic matter (OM) content of the sediment found within the swale (HDS sampler) was high (43%) in comparison to the other sites CP, HD and TB (OM 25-28%). The swale was designed to only receive input from the immediate area (impermeable surfaces surrounding the swale, including atmospheric deposition) and therefore as well as receiving road deposited sediments the nature of the sediment would also be affected by the surrounding vegetation (both within the swale and surrounding gardens, street trees etc). PL, the sampler nearest the road outlet, had comparatively low OM (9%) and the lowest specific surface area (SSA). The nature of the input to this part of the pond was predominantly characterised by the inlet pipe from the road which conveyed sediment from the road network directly into the pond. There was no relationship between particle size and organic matter content and the lowest d(0.9) was observed at HD (11.35 μm) despite having similar %OM to CP, HD and TB. Little similarity was reported- between sites with regards to metals (Cd, Cr, Cu, Ni, Pb and Zn). HDS had notably lower concentrations of most metals than the rest of the sites except for Cu (40.74 mg kg^{-1}) and Zn (366.85 mg kg^{-1}), sources of which included exhaust fumes and particles relating to the wear of tyres and brake pads and discs (Castanherio *et al.*, 2016). Given that RDS constitutes the main input into the swale such levels would be expected. All sites were found to have Pb (ranging in concentration from 32-51 mg kg^{-1}) and while leaded petrol was banned in 2000, levels of Pb were still found to enrich sediments and actively contribute to road surfaces from surrounding soil erosion (Sutherland and Tolosa, 2000). High levels of $^{210}Pb_{un}$ (one of the "fallout" radionuclides used in tracer studies (Collins and Walling, 2004))were reported in the swale sampler (HDS, 455 mBq g^{-1}). $^{210}Pb_{un}$ was shown to have a significant correlation ($p<0.05$) with Cu and Zn (Table 5.22) and was also found in the HDS sample. ^{137}Cs was found in only one sampler (HD), suggesting that local sediment sources e.g., topsoil, are potentially contributing to the sediment within pond 1, whether as part of RDS or as an independent source transported via erosion by wind. Little connectivity is seen in terms of radionuclides and heavy metals in February 2016 and groupings are shown in Figure 5.12 and Figure 5.26.

5.9.2 March 2016

Higher OM was again observed in HDS along with lower SSA and higher d(0.9) and d(0.1). The sources of input to the swale were discrete from the rest of the system in terms of particle size and OM. Vegetated swales can significantly reduce suspended solids entering the SUDS with high concentrations of metals detected at or near the water arrival point (Deletic, 2001; Napier *et al*., 2009; Jones and Davis, 2013). HD was placed close to the inlet from HDS and sediment within the sampler at HD was shown to have a much higher SSA, lower OM and smaller particle size (in terms of D(0.9), d (0.5) and d(0.1)). A higher concentration of metals was also observed within the HD sediment for March 2016. ^{137}Cs was also found in both HDS and HD (0.85mBqg^{-1} and 1.05mBqg^{-1} respectively) and not within any of the other sediment samplers. The swale was discharging directly into pond 1 at this point and the results from this month indicate that while it is disrupting the connectivity between the impermeable surfaces and the pond (Fryirs *et al*., 2007), the smaller particles of greater concern which have higher SSA, facilitating the sorption of pollutants, $^{210}Pb_{un}$ and ^{137}Cs (Herngren *et al*., 2006), were not being removed in the swale.

In terms of similarity between the sites (Figures 5.14, 5.15 and 5.27), the presence of low concentrations of metals gave rise to the grouping CP, PL, TB and HDS together (64% similarity). In relation to radionuclides, discrete groupings were reported as activities varied in the samplers. It is worth noting that CP and HDS both had higher levels of $^{210}Pb_{un}$ (220.73mBqg^{-1} and 258.09mBqg^{-1} respectively) than the rest of the samplers. It is likely that $^{210}Pb_{un}$ was accumulating on the impermeable surfaces where it bound strongly to particulate material. The chemical stability of the bound $^{210}Pb_{un}$ meant that it was probably being redistributed by erosion and runoff (Mabit *et al.*, 2014) and therefore its presence in HD was most likely directly from the transportation of sediment from the swale.

5.9.3 March 2017

The sediment in PL had the lowest OM (13.8%) with the highest OM observed at HDS (which also had the lowest SSA). The sediment within HD had the highest SSA and smallest particle size, further evidence for the efficiency of the swale in trapping coarser sediments. This is similar in effect to larger reservoirs which have been shown to selectively remove the coarser material from the fluvial transport system immediately downstream (Foster *et al.*, 2019).

The cluster diagram for heavy metals (Figure 5.16) shows that the rest of the sites had no similarity to the swale (HDS). Low concentrations of metals were measured in the swale sediments, with the exception of Zn (403.08 mg kg^{-1}). The 54% similarity in metal content between CP and PL in March 2017 could be a result of good connectivity between the samplers; given their close proximity to each other flow is likely to have connected the two sites. A 54% similarity was also reported between HD and TB but it is unlikely (from the hydrological data) that there was connectivity between these sites. The inflow to both sites were similar, so it is possible that the nature of the sediment reflected similar origins of the inputs rather than good connectivity. The concentrations of Cr, Cu, Ni, Pb and Zn were all slightly lower at TB but considering the position of the sampler it would have been expected that some deposition in pond 2 would have taken place prior to reaching the TB sampler site. The particle sizes within the CP and HD samplers where smaller (d0.9) than PL and TB. Due to cohesiveness of these fine sediments (<63µm) they tend to resist being resuspended, however, at higher velocities they can be remobilised and may remain in suspension for a long period of time and travel long distances (Horowitz, 1991). A significant rainfall event had occurred during the time the samplers were in place (27mm of rain was recorded at Upton on 21/11/2016) so it was possible that flow of water occurred between CP and PL and potentially between HD and FB with some of the <63μm fraction resuspended and transported within the system. This was the first sampling period which indicated potential connectivity between the samplers in terms of heavy metal concentrations. ^{137}Cs was observed in samples from HDS and HD and from CP. CP is near the car park, a site which is used by dog walkers who travel in their vehicles to walk their dogs around the Upton Park. Trafficked soils transported by vehicles have contributed to the sediment at this location in the system. No ^{137}Cs was measured at either PL or TB but, the transport of ^{137}Cs is dependent on sediment composition as sorption to particles is size dependent with a preference for finer particles due to the higher SSA (He and Walling, 1997). The d(0.9) and d(0.5) at both PL and TB is greater than seen at CP and HD indicating ^{137}Cs may not have been resuspended during this time. The potential connectivity between the sites may have only been as a result of the movement of coarser particle size fractions.

5.9.4 October 2017

After the summer of 2017, the sampler within the swale was removed due to the lack of water. The water level in the system was low and no sediment sample had been obtained from TB over this period. A lower OM was seen in HD and an increase in the range of particle sizes was also noted with higher d(0.9), d(0.5) and SSA than had previously been measured (Figure 5.6). Higher concentrations of some of the metals, most notably Cu (100.95 $mgkg^{-1}$) and Pb (376 $mgkg^{-1}$) within HD were measured in October. One potential explanation for the change in these characteristics is that the swale was "managed" over the summer of 2017 by grass cutting and removal of vegetation from the swale, which may have reduced its ability to trap suspended sediment.

This was the first sampling period that a sampler was put immediately in front of the road outlet (RO). During installation it was noted that a channel had been anthropogenically created from the outlet to allow water flow into the centre of the pond. Previous sediment, which had been deposited from directly from the outflow pipe, had remained in situ and therefore was interrupting flow of water into the pond. The excess sediment had not been removed from the system rather just “dug out” to allow flow. High quantities of sediment often occur near inlet points (Jones and Davis, 2013) and provide depositional zones within such systems.

There were few similarities between the sites in October, in terms of heavy metals. High concentrations of Cd (3.96 mg kg^{-1}) were measured from RO sediments, high Pb and Zn in PL (143.81 mg kg^{-1} 738.41 mg kg^{-1} respectively), high levels of Cu and Zn in CP (92.23mg kg^{-1} and 738.41 mg kg^{-1}). In terms of radionuclides the highest levels of $^{210}Pb_{un}$ for the sampling period was observed at CP (620.93 mBq g^{-1}) as well as the highest levels of ^{137}Cs (mBq g^{-1}). ^{137}Cs was also measured in the samples from RO and HD. There was no indication of connectivity during this sampling period, but this may be due to the lack of rainfall over this time.

5.9.5 February 2018

A further sediment sampler was installed in November 2017 at Farm Bridge (FB; Figure 3.4) and was located at the point in the system that both SUDS meet upstream of a channel that leads directly into the River Nene (Figure 3.8). The siting of this sampler was to investigate the nature of the sediment at the distal end of the SUDS. High OM was measured in samples from CP and PL and in FB. The sediment in HD had a smaller d(0.9) than had been measured the previous month possibly indicating that the swale was starting to become hydrologically active again.

The sites were separated into two clusters regarding the metals (Figure 5.20) and two distinct groupings were also seen in terms of the radionuclides (Figure 5.29). An 82% similarity was seen between HD and TB with a 50% similarity between these sites and CP. These three sites had the lowest metal concentrations during this period. However, the concentrations measured from the sediment in CP was much lower than HD and TB. The second grouping comprising PL, RO, and FB, had high levels of Cu, Pb and Zn. The concentration of Zn found at FB was the highest across the site (746.57 $mgkg^{-1}$). This sampler was at the downstream end of the SUDS yet the concentration of Zn was found to be above the Dutch Intervention Levels (Table 2.4). Such high levels of Zn at this location in the SUDS was unexpected. There could be a further input from local sources (potentially from the concreted farmyard immediately adjacent to the sample site). Zn, as well as arising from vehicular sources, can also come from solutions that are used to treat and prevent foot rot in sheep as well as being present in food supplements (Defra, 2003). Lambing occurred at the farm between January and April where the ewes were housed in a barn close the sample site (see Figure 3.13). No further investigations were made as a part of this project, so the source of Zn here currently remains unknown and presents an opportunity for further research.

PL and TB were grouped together in terms of radionuclides (Figure 5.30) both having relatively low levels of $^{210}Pb_{un}$. The highest levels for $^{210}Pb_{un}$ were measured at CP and FB, both areas which received input from large impermeable areas. ^{137}Cs was measured in sediments from CP, HD and TB. HD and TB had similar levels of all the radionuclides (apart from $^{210}Pb_{un}$) and in accordance with the heavy metal data, suggested a high degree of similarity between samples collected from these two sites

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CP, TB, and FB had OM >20% possibly due to those samplers being directly exposed to local inputs whereas PL, RO and HD were situated within the pond. Particle sizes at HD were again smaller than at the other sites and although TB had the greatest d(0.9) 43.5um, the d(0.5) was smaller than that found at RO and the d(0.1) was smaller than that measured at CP, PL, RO and FB.

No similarities were reported between HD and FB this month and they were in distinct clusters in terms of metal content. Connectivity between these sites would only be likely during periods of high rainfall (Chapter 4) so potentially during summer months, with minimal rainfall, they would remain distinct. A 72% similarity was seen between PL and RO which, given their close proximity, would be expected. The similarity between these sites and TB and FB potentially came from the high levels of Zn recorded in these sites as well as the high levels of Cu (152.71mg kg^{-1} and 169.73mg kg^{-1}; Table 5.9). $^{210}Pb_{un}$ was measured at higher levels in CP and FB than at the other sites. ^{137}Cs was measured in three samples: RO, HD, and FB. The presence of ^{137}Cs within HD was consistent across the sampling dates indicating there is probably an input from a local topsoil source at this point in SUDS. The presence pf ^{137}Cs at FB is possibly due to the location of the sampler close to agricultural grasslands.

5.9.7 Conclusion

The metal concentrations over the site varied in different samplers at different times and similarities between sites were only apparent in March 2016, February 2018, and July 2108. A PCA was conducted for all the metals across all the samplers and 2 distinct groups were highlighted based on the heavy metals present. The first group comprised Ni and Cd and the second group comprised Cr, Pb, Zn and Cu. However, it is possible that these groups were influenced by the very low levels of Ni, and Cd recorded at some of the sampling sites.

The <63µm fraction of the sediment was examined within the samplers as metal concentrations can increase in the sediment through an urban area as a function of particle size (Horowitz, 1991). While other factors such as surface area and specific gravity can have substantial effects on trace element concentration, it is the particle size which is the most important and a key factor in determining the transport of heavy metals through urban environments, particularly in consideration of RDS (Horowitz, 1991; Zhao and Li, 2013). The d(0.9), d(0.5) and d(0.1) for each of the samplers at each of the sampling dates was plotted, which, allowed the variation within the <63µm fraction to be examined (potentially further investigation is required into the movement of different fractions). To fully understand the movement of contaminants, and associated tracers through the SUDS, further investigation of the <63µm sediment fraction should be undertaken in the future.

The long-term effects of multiple rainfall and run off events and the resultant long-term sediment deposition and retention in SUDS is not well understood. As such, there is much uncertainty in the nature of long-term maintenance, with regard to sediment and litter removal and vegetation cutting (Allen *et al*., 2015). The aesthetics of SUDS, in particular swales, are important to developments like Upton and residents are now obliged to pay up to ~£170 per annum to the Land Trust to maintain the vegetated areas of the SUDS (Northampton Chronicle and Echo, 2016) after the local authority denied responsibility for the open spaces. During planning and conception, the SUDS was cited as providing an underlying basis of the landscape structure and potentially the long-term requirement for maintenance of the SUDS in terms of performance was not fully understood nor considered. Unless adopted by the local authority or a private company, the landowner remains responsible ensuring the maintenance of the SUDS components. The "maintenance "(which only consisted of vegetation removal) of the swale during the study period has suggested that cutting of vegetation can hinder the performance of the swale and provide direct connectivity between urban sediments and the pond (October 2017) reducing the trapping efficiency and leading to increased metal concentrations (especially Cu and Pb). Like swales, the long-term management of ponds and wetlands within SUDS has not been fully considered. The accumulation of sediments over time has been well documented in the literature (e.g., Heal *et al*., 2006; Napier *et al*., 2009; Tedoli *et al*., 2016) but the long-term performance and the necessity for maintenance is not fully reported. Sediment removal has been suggested (Heal, 1999) and that it should be related to lack of capacity / storage within the system as opposed to sediment contamination. In order to understand the sediment accumulation within SUDS the nature of the sediment and potential contaminants entering the system needs to be ascertained along with the flux and deposition of sediments. The long-term source/ pathway/ sink relationships need further consideration in terms of potential pollutant linkages. This study specifically looked at suspended sediment within the SUDS as opposed to other research which has focussed on accumulated sediment within swales, ponds, or wetland systems. The heavy metals of concern being transported into such systems tend to be those associated with RDS namely Cd, Cr, Cu, Ni, Pb and Zn (e.g., Harrison *et al*., 1981; Sutherland *et al*., 2012; Tedoldi *et al*., 2016). Sources of heavy metals within RDS are noted in Table 2.5. Varying ranges of these metals were measured within the SUDS at Upton (Table 5.2) but in comparison to background samples all of the metals were found in higher concentrations (mg kg^{-1}) over the study period. Due to the variability in areas of deposition the ponds are likely to contain hotspots and potentially non-targeted sampling methods e.g., a herringbone scheme (DoE, 1994), as suggested for contaminated land assessment would be appropriate in future studies to ascertain the potential for pollutant linkages from such sites.

The presence of atmospheric radionuclides within the samplers could provide some indication as to sources of the sediment and the PCA (Table 5.23) showed that while the first two Principal Components were strongly associated with the geogenic radionuclides the third Component is strongly associated with the "fallout" radionuclides ^{210}Pb and ^{137}Cs. Local sediment in addition to RDS was being transported into the system, as characterised by the presence of these fallout radionuclides within some of the samplers. When accounting for sediment transport within the SUDS it is again important that varying sources are considered. It is also worth noting that the loading of the different parts of the systems are not equal. The sediment characteristics observed within the sampler in pond 2 (TB) were often grouped, with regards to similarity to that found in the swale (HDS) and pond 1 (only HD). While there was little potential for connectivity, the hydrology of the system would not necessarily support this suggestion at times other than low frequency high magnitude events (Recurrence Interval; 10 – 20 yr). However, if the inputs to the main components are considered, it could be suggested that pond 2 was more similar to a swale in terms of its sediment flux. The lack of potential connectivity between sites within pond 1 is another potential cause for concern as it is almost certain that there are areas of deposition with little possibility of sediment transport within this one SUDS component.

The second part of the objective for this chapter was to"*……establish whether environmental magnetic measurements could be used as a cheaper alternative to heavy metal analysis for contamination assessment*".

While magnetic measurements as a proxy indicator of heavy metal pollution have been reported in the literature, the only highly significant correlations ($p<0.01$) found in this study were between Pb and χ_{FD} (%) and χ_{LF}, SIRM and HIRM. Other significant correlations ($p<0.05$) were observed between Cr, Zn and χ_{FD} (%), Cu, Zn and χ_{LF}, Pb and ARM, Cd, Cu and HIRM. Higher correlations have been reported (Lu ad Bai, 2006) generally in relation to atmospheric traffic related particulate pollution (Hunt, 1986; Chaparro *et al.*, 2006) and topsoil (Wang, 2013a). Mineral magnetism is advantageous over methods for sediment characterisation as the measurements are relatively quick and it is non-destructive (Bityukova *et al*., 1999). However, with the sediments samples from the Upton SUDS it has a limited application with regard to Cd, Cr, Cu and Zn but has the potential for providing a quick estimation of Pb concentrations within the sediments. Potentially further investigations into the RDS and the sources of sediments is required to explore this in greater detail.

5.10 Chapter Summary

The conceptual model presented in Figure 5.37 shows the connectivity in the Upton SUDS in terms of sediment. However, sediment flux is linked directly with the hydrological responses of the system as shown in the previous chapter. More frequently the sediment flux occurs within individual components of the SUDS, especially with regard to pond 1 and pond 2 rather than between components which only happens during or immediately following very high rainfall. In addition to connectivity, the conceptual model shows the varying inputs and potential sources of contaminants that enter the system with some components receiving a greater variety of contaminants (e.g., swale, pond 1) while others (pond 2 and pond 3) are recipients of more localised sediment flux. SUDS is often reported in the literature as a “treatment train" presenting a range of different stages to not only reduce volumes of water but also to remove sediment and associated contaminants; the “jerky conveyor belt”. This research shows that this is the case at Upton, and the separate components act as largely disconnected sinks, however the levels of metals (mg kg^{-1}) reported in Table 5.2, should be a concern. Building at Site C was completed in 2006/2007 and therefore a majority of the inputs, in terms of sediment have occurred during a 10–12-year period (sampling from the SUDS was between 2016 and 2018). Current guidance on sediment removal is suggested as every 25-30 years and while sedimentation rates were not calculated as part of this research project, the figures presented indicate that this removal rate is based on an overestimation of the capacity of the system. Like hydrological modelling, the need for sediment monitoring is evident from this research. It is important that the management of the SUDS is done at a local level, considering local inputs and periodic, potentially annual monitoring is advised as part of the ongoing management and review of performance and capacity.

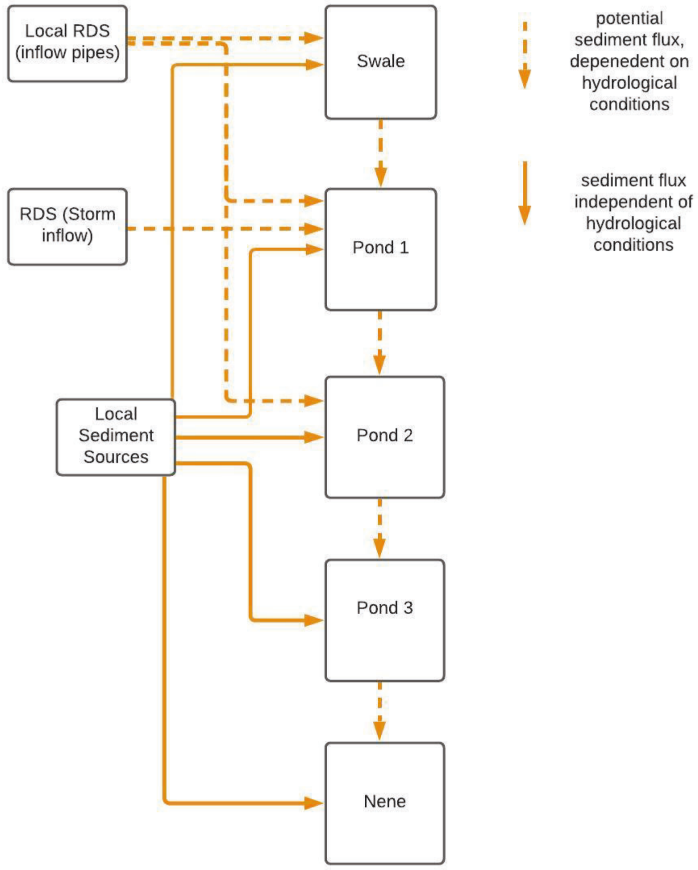

Figure 5.37: *Conceptual diagram representing sediment flux, independent and dependent on hydrological conditions, in the Upton SUDS.*

The conceptual diagram, Figure 5.37, demonstrates the potential for sediment flux within the SUDS, both independent of, and dependent on, hydrological conditions. Within this specific system, pond 1 receives the largest load of sediment from varying different inputs including a major road outflow, local road outflows and potential sediment from the swale, as well as receipt of local sediment (via wind and water erosion). If there is no hydrological connectivity, then deposition and retention of sediments occurs at or very close to the point of entry. In Chapter 4 it was reported that connectivity between pond 1 and pond 2 was rare and therefore rather than manage the system as a whole, management plans should focus attention to individual components of the whole system, including identifying sediment distribution in terms of volume and contaminant levels. While SUDS should be a viewed as a “system” further thought needs to be given at the design stage to the individual roles of each component. When designing management strategies for long term performance, frequent connectivity between the major storage elements should not be assumed. In addition, the potential for pollutant sinks (as reported in this chapter) should also be considered in terms of potentially significant pollutant linkages and future liabilities.

One of the outcomes of this research suggests that periodic monitoring of contamination within SUDS should form an integral part of any annual management plan. At Upton this is especially relevant to pond 1, as this pond probably retains the vast majority of contaminated sediment delivered to it. It is also suggested that the design of SUDS, the efficiency, and limitations of sediment accumulation, are communicated to the managing authority after developers have handed over responsibility to local government. As reported from Upton the management companies employed to maintain SUDS are often unaware of the consequences of their actions e.g., removal of vegetation and decrease in trapping efficiency (Chapter 5.9.7).

While magnetic measurements appear to have limited applications as a proxy in this instance, previous research cites success in using these quick methods as an indicator of heavy metal pollution. The lack of success with these methods during this study may potentially be due to the fact that sampling was undertaken on actively transported sediments in suspension as opposed to bed sediments and RDS. Further investigation into the potential for these methods is required.

Chapter 6 Conclusion

This thesis investigated the hydrological and sedimentological (dis)connectivity within a SUDS, based on high resolution monitoring of a recently installed SUDS system at Upton, Northampton. The research attempted to identify connectivity within and between SUDS components, based on multiple event analysis and long-term monitoring over 4 years. The first part of this concluding chapter will explore the two main objectives and assess the extent to which they have been delivered. The contribution that this study makes to the knowledge of SUDS behaviour and management will also be highlighted. The summary will draw together the main outcomes of this research and raise implications for management and maintenance of such systems which may be of generic importance, and the final section will make recommendations for further research into unresolved specific and general issues that have been established during the course of this research project.

6.1 Research Objectives

The literature review of Chapter 2 identified that there was a relatively poor understanding of the pathways that sediment takes through urban drainage systems to receiving waters (Taylor, 2007) and that the transport of such sediments through SUDS following multiple storm events is neither well researched nor understood. The long-term performance of SUDS was questioned, and the current guidance on their performance and future management is often based on information which is based on single rather than multiple monitored or modelled storm events. This thesis has attempted to fill these gaps in knowledge by undertaking research in a recent SUDS development on the edge of Northampton, UK, which provided a typical example of the range of features normally incorporated into SUDS schemes, e.g., swales, ponds and wetlands that were organised, in the case of Upton, into what has often been described as a “treatment train”. High resolution monitoring of water levels was undertaken at strategic points to establish the degree of hydrological connectivity between and within different parts of the system at a range of rainfall intensities and durations. Additionally, active suspended sediments were collected at a number of locations throughout the study period, in order to establish the degree to which differences in physical characteristics, heavy metal concentrations, gamma-emitting radionuclide activities and mineral magnetic signatures helped to establish which different parts of the system were connected or isolated from other parts of the system. The connectivity of SUDS is pivotal to their effectiveness if the purpose of these systems is to manage flooding and mitigate sediment and potential contaminant transport into surface waters like the River Nene. The design of the project was justified on the basis of this identified knowledge gap in the published academic and grey literature and the research has provided new insights into SUDS behaviour and future management.

Objective 1: To review the long-term data with regard to water levels within the SUDS and assess the impact of rainfall events of different magnitude and frequency on the hydrology and connectivity of the system.

A review of rainfall data and hydrographs showing the response of the swale, pond 1 and pond 2, within the SUDS at Upton, was undertaken using water level data derived from installed pressure transducers. The high resolution of the data, collected at 5-minute intervals, allowed the “flashiness” or “storm transients”, which are defined as the rates of rise and fall in water level (Shuster *et al*., 2008), to be studied in considerable detail. The analysis provided an overview of how quickly flow and water levels within the system changed in response to rainfall inputs. The responses to rainfall in each of the systems measured was variable with the swale generally responding more quickly than the other components, although it was the intensity of the rainfall which dictated the overall hydrological connectivity within the system. Only connectivity between the swale and pond 1 was identified in the examples presented from the study period, with one exception, November 2016. Storm Angus had affected many parts of Southern England and Wales during the 17-22 November, corresponding with the rainfall data reported in Chapter 4, and a total of 45.7mm of rain fell at

Upton over the four days between the 19^{th} and 22^{nd} November. A detailed examination of the hydrographs provided a basis for a quick interpretation of the whole data set which then aided in the identification of further connectivity within the system, between pond 1 and pond 2. The hydrological connectivity between the swale and pond 1 occurred frequently during low intensity rainfall events but the connection between the two components was usually short lived due to a rapid discharge to pond 1, and no connectivity was observed if water levels within the swale remained below 150mm. The quick responsiveness of the swale was demonstrated by an increase in water levels from 0mm to 120mm with as little as 2mm rain over 3 hours (Figure 4.22 Event 1A). The connectivity between pond 2 and pond 3 was also reported, and again occurred frequently, when water levels exceeded 120mm. The discharge from pond 2 was less rapid, in terms of speed of response, than that which occurred from the swale to pond 1, as water levels were more constant and responses to rainfall less "flashy". The observed connection between ponds 1 and 2 during the storm event only lasted for a few minutes, and therefore the flux of sediment between the ponds was assumed to have been minimal.

Objective 2: To investigate potential sources and connectivity of the sediment and sediment-associated contaminants between sampling sites and establish whether environmental magnetic measurements could be used as a cheaper alternative to heavy metals analysis for contamination assessment.

Analysis of the <63μm fraction of the sediment for particle size, organic matter, a variety of metals of concern (Cd, Cr, Cu, Ni and Zn), as well as both geogenic and fallout radionuclides, provided an in-depth evaluation of the nature, and potential sources, of the sediment entering and moving within the SUDS. Heavy metal concentrations varied across the sites and between the sampling dates and provided further insight into the (dis)connectivity within the system and within individual components of the system, such as significant differences between the concentration of metals in samples collected at different locations in pond 1. However, similarities in sediment properties were reported between samplers which were not only physically close together but also those which were situated in different components of the SUDS. The swale sampler was pivotal in understanding the nature of the sediment in the receiving part of pond 1. It was reported that those areas, in direct receipt of inputs, were likely to be sediment and contaminant depositional ‘hot spots’ because sediment was frequently deposited in close proximity to the inflow pipes and was not transported great distances longitudinally along the pond. It was possible to identify potential connectivity between and within the ponds, using the metal concentration data and hydrograph analysis. Environmental magnetic measurements were shown to have the potential to provide an estimate of Pb concentrations in the sediments but had a relatively weak correlation with Cd, Cr, Cu and Zn.

6.2 Research outcomes

The hydrological and sediment data and resultant conceptual diagrams produced in Chapters 4 and 5 (Figures 4.35 and 5.37) have been combined to produce three summary models based on different hydrological scenarios: no rainfall, low intensity rainfall and high intensity rainfall, Figures 6.1 to 6.3.

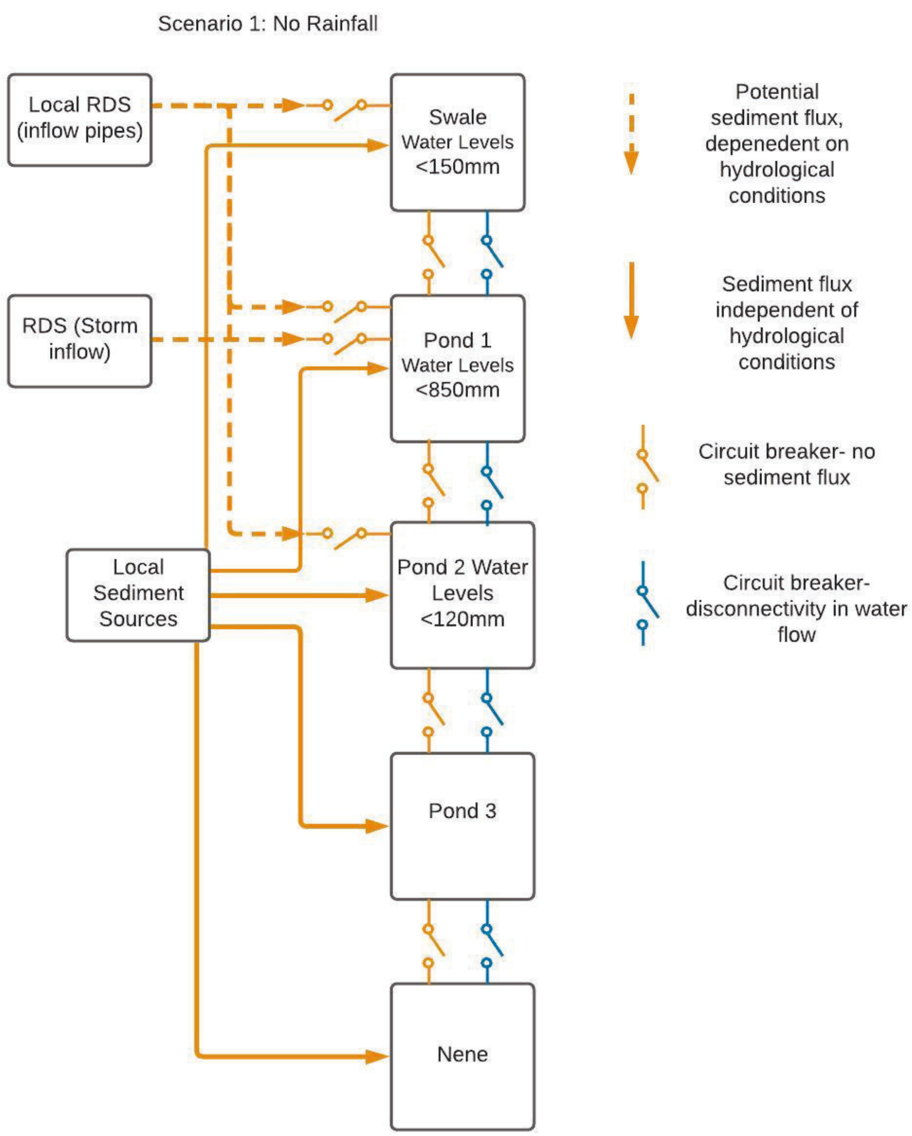

Figure 6.1 Upton SUDS scenario 1: No/ low rainfall.

With no/ low rainfall intensity, e.g., 2mm over 2 hours as seen entering the SUDS in May 2017 (Figure 4.22, the different components are not connected (Figure 6.1). There is no flux of sediment to any of the adjacent or downstream components of the SUDS, however some atmospheric/wind-blown sediment could have been deposited locally. All sediment is retained within the individual ponds and swales under this scenario.

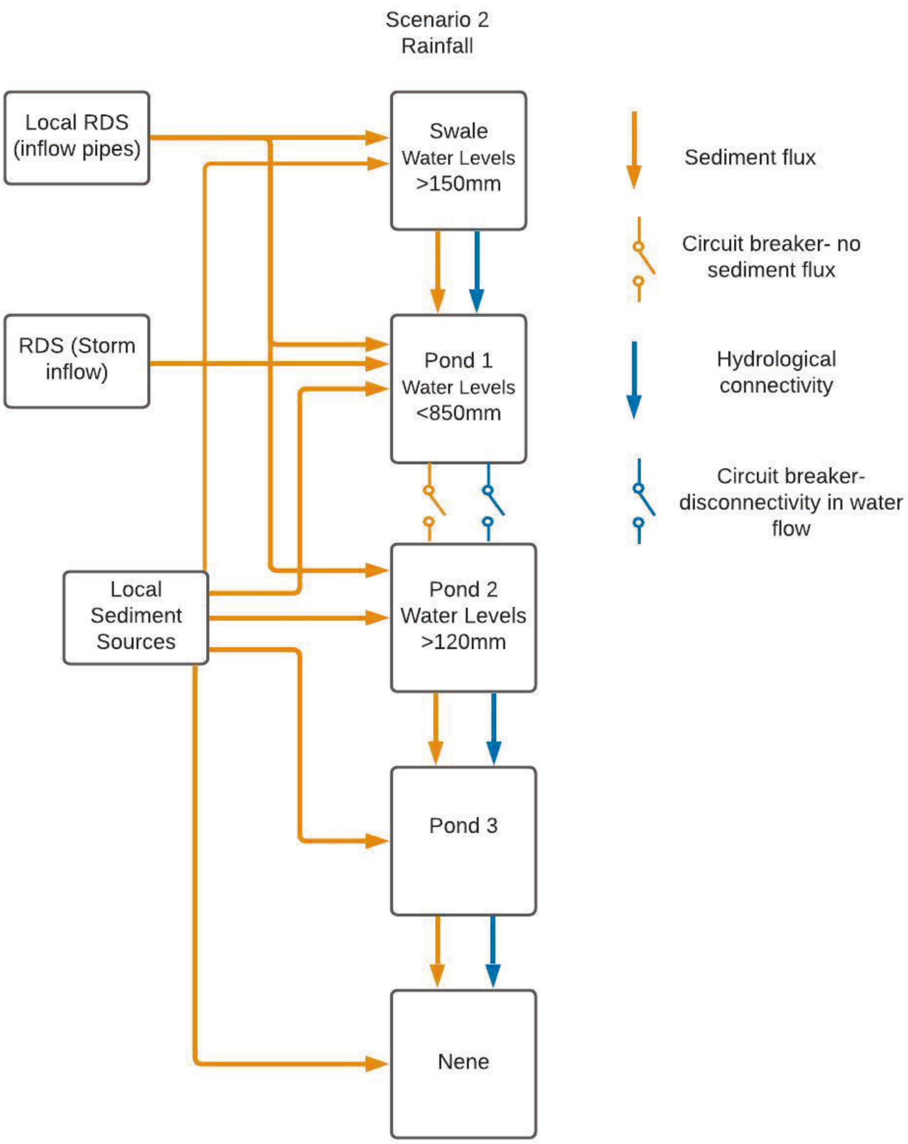

Figure 6.2: Upton SUDS scenario 2: Rainfall (e.g.,>2mm day^{-1} based on Event 1A Figure 4.22).

Scenario 2 (Figure 6.2) represents the rainfall events as seen in Chapter 4 when the swale and pond 1 were connected (e.g., Figure 4.28), and sediment was transferred between the swale and pond 1.

Deposition and retention of sediments was expected in pond 1 as an "end of pipe" solution from a number of inputs due to a lack of connectivity with the rest of the SUDS. Pond 2 connected hydrologically with pond 3 and further downstream but also retained a proportion of sediment delivered from another part of the SUDS but excluding pond 1. Pond 3 behaved similarly to pond 2 and therefore there was continued deposition of sediment in pond 3. The flow of water through the system may not have been sufficient to reach the river and the remaining sediment was therefore deposited.

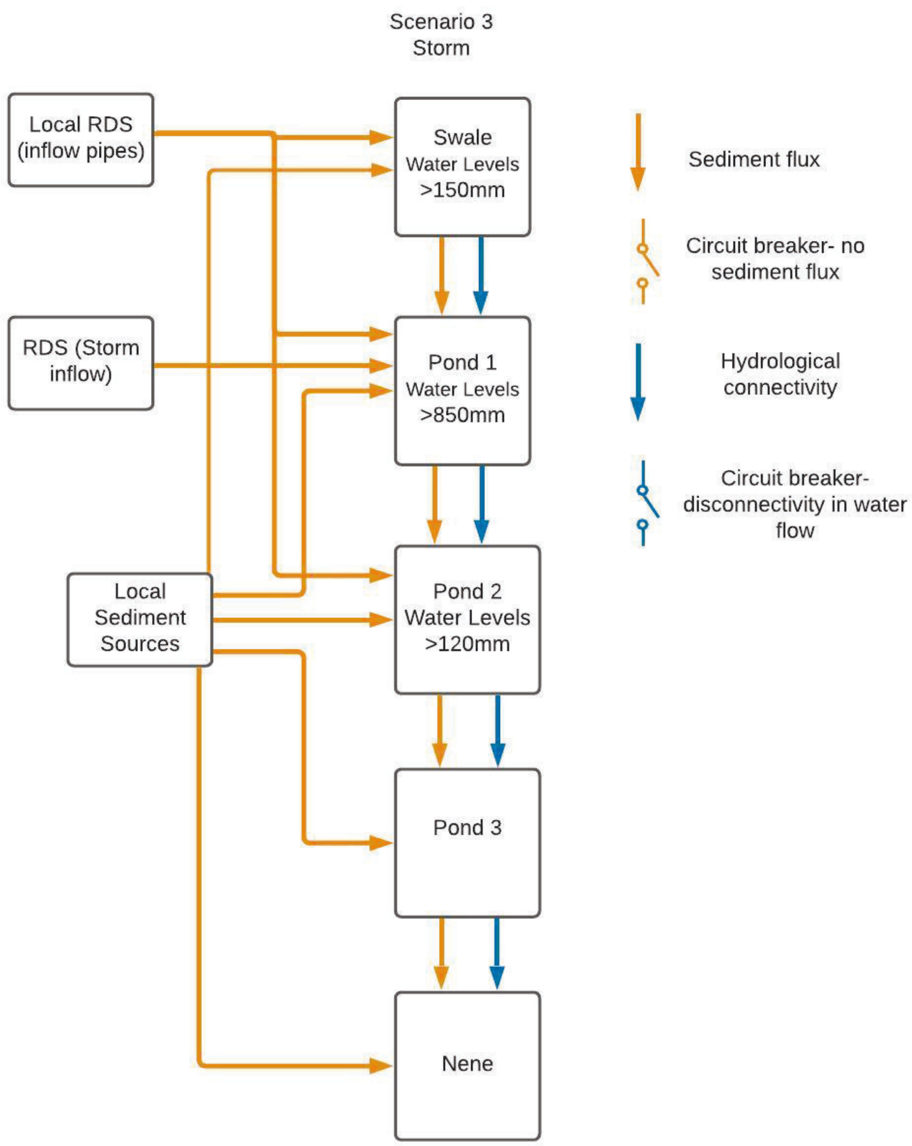

Figure 6.3 Upton SUDS scenario 3: Storm.

Under high intensity conditions e.g., >10mm day^{-1} (as per Figure 4.34) of scenario 3, Figure 6.3 full connectivity of the individual components occurred briefly. Due to its longitudinal design, pond 1 still acted as a predominately depositional environment but some limited transfer of very fine sediment would be expected to pond 2. Pond 2 still received separate inflows but would be temporarily connected to pond 1 and would potentially discharge quickly to pond 3. Some deposition of sediment was likely to occur in pond 2. Again pond 3 will have behaved in a similar manner to pond 2 and flow was potentially sufficient to reach the river Nene with sediment being deposited enroute.

Scenario 3 shows the system doing what it was initially designed to do in terms of a "treatment train" and ultimately controls water velocity and removes pollutants and sediments from receiving surface waters by disconnecting hydrological pathways under these conditions. However, most of the time the SUDS is operating under scenarios 1 and 2 and this fact needs to be considered to inform maintenance and management strategies for the long term. It suggests that options surrounding sediment management, whether removal or remediation, needs to be addressed as the majority of sediment is retained in pond 1 which becomes a major repository for both sediment and contaminants, although their spatial distribution within pond 1 is highly variable due to localised deposition close to inflow points. Suggestions in the published literature of the need for sediment removal every 25-30 years (Heal, 1999) may be an overestimate of the timescales involved, when the average sediment input measured in this study is above background concentrations for many metals. Rates of accumulation, and potential for exceedance of SGVs requires further investigation as well as the potential for pollutant linkages. Evaluation of the toxicity of accumulated sediment will determine whether the excavated material would need to be disposed of as a controlled waste (The Controlled Waste (England and Wales) Regulations 2012).

The design of SUDS needs to be carefully considered in terms of the hydrological responses and flux of sediments and should be based on multiple, potentially high-resolution sets of data detailing the dynamic responses of the system to rainfall of different magnitudes and frequencies. A range of models across a range of flows or the utilisation of research, such as that presented in this thesis, could be used to inform good practice at the design stages. Direct discharges into wetlands and ponds, as observed in pond 1, should be avoided (Susdrain, n.d.c) as the design guidance notes that these “end of pipe” solutions are likely to receive faster run off flows and increased levels of pollution, as observed during this study in pond 1. The design of SUDS requires a balance of different options, with perhaps the need to consider informed risk-based decisions. In terms of Upton, which was intended to “demonstrate good design and development practices for housebuilders” (ADS, 2011, p 5) through Enquiry by Design charettes, the SUDS were an integral part of the sustainable urbanism ethos of the project with the “use of SUDS at scale with urban swale and an emphasis on biodiversity” (Isherwood, 2013, p 3). It was one of the first UK developments which included a plan for “roof to river” surface water management strategies through a total of two hydrological catchments (Figure 3.8) However, the conceptual diagrams of Figures 6.1, 6.2 and 6.3 demonstrate that there are disjunct components in the study area (which represents one of the SUDS catchments) under most rainfall conditions. Therefore, management strategies over the long term, need to consider the entire SUDS system including individual components and different catchments.

As a drainage system SUDS should be regularly monitored and maintained for them to continue to function as designed and such management should be based on a detailed knowledge of the hydrological performance and distribution of contaminants within the system. Due to the uncertainty in and increased frequency of, extreme rainfall events due to climate change, an annual monitoring programme could provide a basis for predictions over the longer term and provide an insight as to the operating function of the system determining fitness for purpose.

In terms of sediment flux, and potential for contamination, the siting of SUDS should consider the nature of the surrounding landscape and land use and ascertain whether potential local inputs could further contribute to the sediment and pollutant loads at different points in the system. The presence of “fallout” radionuclides, $^{210}Pb_{un}$ and ^{137}Cs, could help to identify sources of sediments although little evidence was found regarding the transport of ^{137}Cs through the system, probably due to a lack of connectivity. The potential for using mineral magnetic measurements as a proxy for the identification of pollution was explored and while it has limited application within the context of this SUDS environment it could be used in the future to identify sources of urban sediments/ RDS rather than predict metal concentrations effectively. The accumulation of contaminants in the sediments in this study raises some management concerns. While SUDS are designed to retain sediments, they need to be monitored in terms of exceedance of guideline values. The production of the conceptual models (Figures 6.1-6.3) would aid the development of future management strategies for SUDS. They provide an indication, based on long term monitoring, of dis-connectivity within the systems and the potential areas of concern with regard to “pollutant sinks”. Rather than providing a “treatment train” style solution the SUDS at Upton provides a series of smaller catchments with pond 1 almost presenting an “end of pipe” solution for its inputs. Such designs are not considered “good practice” as they will be more costly and difficult to maintain, requiring removal of potentially contaminated sediment when storage volume has infilled.

6.2.1 Summary

While this research has identified the need for event-based monitoring or event-based modelling of hydrological behaviour and has shown the limited connectivity between potential contaminant stores in a SUDS system, there are several site specific and generic issues that need to be addressed in the future.

- Connectivity within the system should not be assumed and needs to take into account local variability with regard to sediment sources Chapter 5 reported high levels of Zn downstream of the SUDS at the FB sampler which could have arisen from the close proximity of this sampler to the local farm. Modelling of sediment sources in the SUDS using sediment properties as fingerprints could provide useful insight into potential contaminant sources and “hotspots”.
- The design of pond 1 presents an “end of pipe” solution for sediments, potentially creating a “sink” for pollutants. Where possible SUDS should avoid this design.
- Equal consideration should be given to both the hydrological and sedimentological response of the system at the initial design stages. This should in turn be considered at a local level in line with climate change predictions to take into account any potential change in the frequency of extreme rainfall events.
- Local sediment inputs should form part of the sediment budget for SUDS and the cumulative effect of these inputs should be monitored. Movement of contaminants and associated tracers in fine sediments, <63µm, could provide further insight into sediment flux through SUDS.
- Variability in sediment accumulation rates at different locations would pinpoint potential areas where removal of sediment and frequency of removal needs to be addressed in future management plans.
- Contaminant concentrations need to be carefully monitored as they can exceed SGVs and international intervention values within relatively short periods of time (~10 years). Consideration needs to be given to disposal option for such sediments. The high spatial variability in concentrations reported in this study suggests that sampling protocols need to be refined and potentially non targeted sampling methods e.g., a herringbone scheme (DoE, 1994) be adopted, to estimate contaminant concentrations and inform best disposal practice. Decisions on whether the material is a controlled waste or could safely be spread

to land, for example, will have both significant financial and potential liability implications for local authorities.

- Emerging contaminants of concern are now being measured within sediment samples at Upton and elements such as Indium and Gallium are known to be entering urban sediment systems from a variety of potential sources. The presence of these contaminants enhances the risk of environmental exposure and mounting evidence is indicating that these elements can be toxic. Much remains unknown about their natural and anthropogenic cycling; however, evidence is presented that the environmental concentration of both Ga and In is changing and the anthropogenic flux of these metals already appears to be exceeding natural fluxes. Further studies are required to investigate the extent and sources of these elements, as well as other contaminants such as plastic, and look at potential accumulation within SUDS and other parts of urban hydrological systems and receiving water courses.
- The potential for phytoremediation within SUDS has been suggested in some literature but not reported in this thesis. Some pilot studies have been conducted at Upton and bioaccumulation of Cd, Cr, Cu, Ni, Pb and Zn has been observed. Further in-depth studies are required into their effectiveness. The potential for accumulation of such metals within vegetation also has implications for maintenance programmes. If local vegetation is accumulating potentially toxic levels of contaminants, then appropriate disposal needs to be considered.
- Future planning based on real time monitoring should be used to inform both risk mitigation and sediment management practices which should form part of an overall maintenance plan for SUDS from the initial design stages. This future planning will allow Local Authorities to have appropriate management techniques and funding in place when adoption of these schemes passes from developers.

6.3 Concluding remarks

As the global population grows over the next 10-30 years the requirement for the development of housing is crucial and further urbanisation is likely. The impacts of such developments on the structure and function of natural systems, particularly the hydrological system, will be profound. Future design of SUDS needs to address predicted localised climate scenarios especially with the likely increase in the frequency of high magnitude storm events. Monitoring of localised catchments, prior to the design stage of SUDS, can provide useful insights into regional rainfall events and analysis of long-term rainfall and hydrographs can help assess the impact of climate change and increased rainfall intensity on connectivity within these urban environments. Further long-term monitoring of existing SUDS is required to fully understand the processes involved in terms of hydrological and sedimentological connectivity in such systems. This in turn would inform future management of existing systems and address concerns surrounding the management of sediment accumulation and contamination. Previous guidance has suggested that sediment removal within SUDS should be related to lack of capacity / storage in the system but contaminant concentrations, and the presence of emerging contaminants provide cause for concern. Effective management strategies should consider the potential for exceedance of SGVs and intervention values and introduce routine monitoring from the outset. Many SUDS will ultimately fall under Local Authority control, and it is imperative that design / construction companies ensure that effective strategies for both sediment and contaminant management are communicated, and that annual maintenance is carefully planned for the performance of the system rather than the visual aesthetics of the landscape. There are significant financial and potential liability implications for Local Authorities (or assigned management companies) if SUDS are not managed effectively.

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