Sediment, nutrient and runoff management and mitigation in rural catchments

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Declaration

I certify that no part of the material offered in this thesis has been previously submitted by me for a degree or other qualification in this or any other University.

Signed

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Abstract

This thesis is concerned with the quality of surface waters in rural catchments across northern England and the mitigation of Diffuse Water Pollution from Agriculture (DWPA). Runoff Attenuation Features (RAFs) are a range of soft-engineered DWPA *transport* management options, which target hydrological flow pathways for the purpose of slowing, storing and filtering water. This study demonstrates the potential of RAFs to significantly reduce losses of suspended sediment (SS), phosphorus (P) and nitrate (NO_3) in agricultural runoff.

To implement RAFs effectively it is vital to understand how, where and when to best target mitigation efforts. This relies on knowledge of the sediment and nutrient regime and hydrological functioning of a catchment. In response to this a stratified, synchronous grab sampling programme was implemented over two consecutive years in the upper Eden catchment (334 km^2), Cumbria, covering thirteen sub-catchments of multiple scales. No relationship was found between sediment/nutrient yield and catchment area but it was recognised that certain lowland sub-catchments deliver a disproportionate amount of the pollutant load, particularly SS and P, due to increased agricultural activity, and that there were large variations in flux affected by season and hydrological conditions.

One particular sub-catchment dominated by improved grassland, Blind Beck (9 km^2), exhibited both higher nutrient and SS concentrations per unit runoff and higher yields compared with any other sub-catchment. The Blind Beck sub-catchment was selected in which to implement a more detailed investigation of SS and nutrient delivery, which included event sampling. High flows (accounting for 10% of flow duration) contributed 84% of the annual SS load, 76% of the total P and 68% of the soluble reactive P, but just 32% of the NO_3 load. This highlights the acute nature of the SS and P diffuse pollution problem and demonstrates the need to target storm events for effective mitigation.

A number of RAFs were constructed in two established research catchments in Northumberland with a similar mixed land use to the Eden: Belford (15 ha) and Netherton (80 ha). Synchronous inlet and outlet water samples were collected during storm events. Results demonstrate that relatively small RAFs, principally sediment traps, constructed in farm ditches (<1 km^2 catchment area) can reduce mean SS, TP, SRP and NO_3 loads during storm events by 30-49%, 23-37%, 12-27% and 8-14%, respectively. The potential of RAFs designed to reduce DWPA in key locations and at certain scales will be proposed based on the findings of the PhD study.

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List of Abbreviations

ADCP	Acoustic Doppler Current Profiler
AOD	Above Ordinance Datum
AONB	Areas of Outstanding Natural Beauty
AWS	Automatic Weather Station
BFI	Baseflow Index
BOD	Biological Oxygen Demand
BSI	British Standards Institute

C	Carbon
CAP	Common Agricultural Policy
CEH	Centre of Hydrology and Hydrology
CHASM	Catchment Hydrology And Sustainable Management
CoGAP	Code of Good Agricultural Practice
CORINE	Co-ORdination of INformation on the Environment
CSA	Critical Source Area
CSF	Catchment Sensitive Farming
CW	Constructed Wetland
DEFRA	Department for Environment, Food and Rural Affairs
DEM	Digital Elevation Model
DTC	Demonstration Test Catchments
DWPA	Diffuse Water Pollution from Agriculture
EA	Environment Agency
EC	European Community
ECSFDI	England Catchment Sensitive Farming Delivery Initiative
ELS	Entry Level Stewardship
EM	Environmental Measurement
EU	European Union
FDC	Flow Duration Curve
FEH	Flood Estimation Handbook
FYM	Farm Yard Manure
GAEC	Good Agricultural and Environmental Condition
GIS	Geographic Information System
GQA	General Quality Assessment
GWW	Grassed Waterway
H-ADCP	Horizontal-Acoustic Doppler Current Profiler
HLS	Higher Level Stewardship
HOF	Hortonian Overland Flow
HOST	Hydrology Of Soil Type
IACP	Irish Agricultural Catchment Programme
IH	Institute of Hydrology
LCM2000	Land Cover Map 2000
LEAF	Linking Environment and Farming
MOPS	Mitigation Options for Phosphorus and Sediment
N	Nitrogen
NE	Natural England
NFM	Natural Flood Management
NGO	Non-Governmental Organisation
NO_3	Nitrate

NSRI	National Soil Resources Institute
NVZ	Nitrate Vulnerable Zone
OS	Ordinance Survey
P	Phosphorus
PE	Potential Evapotranspiration
PoMs	Programmes of Measures
PP	Particulate Phosphorus
PSYCHIC	Phosphorus and Sediment Yield Characterisation In Catchments
Q	Discharge
RAF	Runoff Attenuation Feature
RBMP	River Basin Management Plan
RDB	River Basin District
RG	Rain Gauge
RP	Reactive Phosphorus
SAAR	Standard-period Average Annual Rainfall
SAC	Special Area of Conservation
SEDEM	Sediment Erosion Model
SMR	Statutory Management Requirement
SOF	Saturation Overland Flow
SOWAP	Soil and Water Protection
SP	Soluble Phosphorus
SPA	Special Protection Area
SPRHOST	Standard Percentage Runoff derived from Hydrology Of Soil Type
SPS	Single Payment System
SRP	Soluble Reactive Phosphorus
SS	Suspended Sediment
SSSI	Site of Special Scientific Interest
SUP	Soluble Unreactive Phosphorus
SY	Specific Yield
TBR	Tipping Bucket Rain gauge
TN	Total Nitrogen
TP	Total Phosphorus
TRP	Total Reactive Phosphorus
TSP	Total Soluble Phosphorus
UKTAG	UK Technical Advisory Group
UP	Unreactive phosphorus
VSA	Variable Source Area
WaTEM	Water and Tillage Erosion Model
WEPP	Water Erosion Prediction Project
WFD	Water Framework Directive

1. Introduction

1.1 Background

The quality of the aquatic environment is of significant importance - good quality waters for bathing, angling and other activities play a substantial role in supporting tourism and recreation; water of good chemical and biological status supports diverse ecosystems. Conversely water of poor quality requires more energy to treat and evokes higher costs to reach a potable standard. The negative impact of excessive sediment and nutrients on the aquatic environment is a recognised catchment management issue. As the dominant land use in the UK, diffuse water pollution from agriculture (DWPA) is a major contributor to excessive suspended sediment (SS), phosphorus (P) and nitrate (NO_3) loads which regularly cause water quality issues in freshwaters, and mitigation of the impacts of DWPA is crucial to meet the requirements of the European Water Framework Directive (WFD).

Sediment and nutrient losses should be prevented at source where possible; however, it is acknowledged that this is not wholly effective, and water quality problems caused by DWPA may result even when applying good agricultural management. Contaminant *transport* management, targeting polluted runoff pathways, has the potential to mitigate a significant proportion of DWPA in a catchment. Runoff Attenuation Features (RAFs) are a range of soft-engineered landscape interventions designed to reduce losses of SS, P and NO_3 in agricultural areas by slowing, temporarily storing and filtering runoff.

To be able to recommend RAFs as viable DWPA mitigation options it is vital that evidence is gathered in the form of quantitative (as well as qualitative) data. This PhD study achieves this by investigating the design, construction and functioning of a number of RAFs in rural catchments across the north of England. Moreover, it addresses the current insufficiency of data associated with DWPA transport management options in the UK and presents design criteria for the future application of RAFs.

As a prerequisite to determining the efficacy of RAFs it is vital to understand the sediment and nutrient regime and hydrological functioning of a catchment to ensure that intervention(s) are targeted effectively. However, the spatio-temporal heterogeneity of catchment processes that control sediment and nutrient fluxes is complex and not fully understood. To address this problem a multiple-scale, nested basin monitoring campaign is employed in this thesis to characterise SS, P and NO_3 regimes in the upper River Eden catchment, Cumbria. The upper Eden catchment provides an excellent case study site: it is representative of the upland catchments common to large areas of northern and western England, Wales and Scotland, as well as being a river of important ecological status in its own right (attributes which have led to its status as one of the Demonstration Test Catchments, selected by Defra). There is an established long-term monitoring network and hydrological dataset on which to base the investigation.

This thesis describes where sediment and nutrients come from in slowly permeable, mixed land use agricultural catchments typically found in the north of England, how and when they are delivered to watercourses, and how to address these losses appropriately. A detailed understanding of dominant runoff and contaminant flow pathways is gained, which demonstrate the importance of storm hydrology and land cover/use as key controlling factors over the export of sediment and nutrients. This allows for the identification of many locations in the rural landscape where RAFs can be implemented.

Runoff Attenuation Features, as relatively low-cost measures, are shown to have the potential to remove significant amounts of SS, P and NO_3 from agricultural runoff, targeting storm events as a priority, and also offer secondary benefits such as flood mitigation and habitat creation.

1.2 Research aims

1. Characterise the SS, P and NO_3 transport regime of the mixed land use upper Eden catchment across a range of scales (1 km^2 - >100 km^2) to inform the targeting of DWPA mitigation efforts; including the determination of contaminant yields and the influence of spatio-temporal scale and controlling factors such as land use and storm events on these yields.

2. Investigate the efficacy of a number of RAFs as DWPA *transport* mitigation options; detailing their capacity to reduce concentrations/loads of SS, P and NO_3, along with important design and construction criteria for future application.

Objectives

1.1 Use an appropriate grab sampling methodology to quantify SS, P and NO_3 concentrations at a range of catchment areas covering three orders of magnitude (micro ≈ 1 Km^2, mini ≈ 10 Km^2 and meso-scale ≈ 100 Km^2) in the upper River Eden catchment, Cumbria. Select an appropriate means to calculate annual SS, P and NO_3 loads and specific yields

1.2 Investigate how determinand concentrations/loads vary with spatial scale, as well as with changes in various controlling processes, such as precipitation/runoff and land cover/use, *inter alia*.

1.3 Compare calculated sediment/nutrient yields with export coefficients from the literature. Determine the representativeness of the data collected in the upper Eden catchment and evaluate the selected methodology for water quality monitoring at the catchment scale.

1.4 Select a sub-catchment within the Upper Eden catchment. Use a spatially intensive sampling campaign to identify pollutant sources within the catchment, and employ automatic storm sampling equipment to determine the importance of storm events on contaminant transfer.

2.1 Describe the design and construction of a number of RAFs in agricultural catchments. Using field measurements, evaluate the efficacy of RAFs to reduce SS, P and NO_3 concentrations/loads in runoff during storm events – measure sedimentation volumes and calculate annual sediment/nutrient removal rates where possible.

2.2 Review the use of RAFs on the larger catchment scale, taking account of lessons learned on sediment/nutrient source pathways, RAF suitability and appropriate spatial scale for implementation. Consider the potential multiple-benefits of RAFs and recommend design criteria for future implementation.

1.3 Thesis outline

Chapter 2 - presents a literature review covering the relevant subject areas, including: hydrological processes and catchment connectivity; the sources, pathways and potential impacts of SS, P and NO_3 on fresh water quality; current legislation governing the management of DWPA, including current thresholds and guideline contaminant concentrations; DWPA mitigation options, including RAFs; and the influence of scale on the operation of runoff, sediment and nutrient regimes and their management.

Chapter 3 – outlines the methodological approach used to meet Aim 1. It provides a description of the upper Eden study catchment (334 km^2) and describes the experimental design used to attain sediment/nutrient concentration and hydro-meteorological data. The laboratory techniques used to determine SS, P and NO_3 concentrations are detailed.

Chapter 4 – reports the hydrological, SS, P and NO_3 regimes of thirteen upper Eden sub-catchments. Annual sediment and nutrient yields for each sub-catchment are calculated and presented. Spatial and temporal patterns in flux are discussed as well as the influence of land use and other catchment variables. Calculated yields are also compared with export coefficients from the literature and export coefficients used to estimate the sediment/nutrient loss from an unmonitored part of the catchment.

Chapter 5 – presents the methods and results of a detailed sub-catchment study (within the upper Eden catchment – Blind Beck, 9 km^2). A spatially high-resolution grab sampling campaign is carried out to identify sediment/nutrients source-pathways, and event-level automatic water sampling to evaluate the importance of storm events in sediment/nutrient fluxes. The Blind Beck catchment was also selected for the implementation of a number of RAFs. However, due to unforeseen circumstances only one intervention was completed and is described here. Further DWPA mitigation experiments were relocated to surrogate sites and this work forms the basis of the two subsequent chapters.

Chapter 6 – is the first of two case studies that focus on the testing of RAFs; this chapter describes investigations carried out in the Belford catchment (5.9 km^2). The first part involves the evaluation of existing flood RAFs and whether they also function to mitigate DWPA. A number of important lessons are learned and a grab sample campaign carried out in a 15 ha sub-catchment provides an insight into the sediment/nutrient regime of the area. Based on these findings a multi-stage stage RAF was constructed in an agricultural drainage ditch to target both sub-surface drain and overland flow pathways. The multi-stage RAF is monitored to determine its ability to reduce SS, P and NO_3 losses during storm events and the results are presented.

Chapter 7 – describes the second case study, carried out in the Netherton Burn catchment (10 km^2), where both flood and DWPA mitigation RAFs were implemented in a real-world project. It details the design and construction of several sediment traps commissioned to treat the runoff from an 80 ha sub-catchment; the design of the features was informed by findings from the Belford case study. The results from event-scale monitoring of the feature are presented and discussed.

Chapter 8 – contains an overall discussion that links the findings from Chapters 4 and 5 with the results from the mitigation experiments (Chapters 6 and 7). The use of RAFs is considered at the wider catchment scale and how they could be successfully integrated into catchment management plans to mitigate for concentrated agricultural runoff pathways such as drainage outfalls, farm ditches and channelled overland flow.

The potential use of RAFs in conjunction with *source* and *mobilisation* mitigation options is evaluated, along with their practicability for farmers and the ability to deliver secondary benefits such as flood attenuation and habitat provision.

Chapter 9 – concludes the thesis with a summary of findings and recommendations for further work.

2. Literature Review

2.1 Introduction

"The unseen threat to water quality" - this is how the Environment Agency describes diffuse pollution. It is also the title of a pivotal document (Environment Agency, 2007) in which the agency list their main concerns regarding the freshwater environment. The list includes:

- High levels of nutrients in rivers, lakes, estuaries and coastal waters, which can cause eutrophication.
- Nitrate contamination of water used for drinking water.
- Pesticides and sheep dip from agriculture entering rivers, lakes and groundwater.
- Oxygen depletion in water due to organic pollution from livestock manure.
- Sediments from soil erosion smothering habitats in rivers, lakes and estuaries.
- Bacteriological contamination of bathing waters and shellfish waters from farm waste and illegally connected sewers.

All of the above can be linked with the agricultural industry. Excess sediment and nutrients in aquatic environments have detrimental impacts, which affect both natural flora and fauna, and also society, which depend on them economically and socially. In this chapter the current state of research on the topics relevant to the study will be examined. Eutrophication is discussed first; specifically how agricultural practices have led to increased sediment and nutrient losses from land to water by increasing soil nutrient concentrations and increasing soil erosion due to hydro-geomorphological impacts. There is a brief review of the hydrological processes responsible for generating runoff, which then focuses on the concept of 'catchment connectivity' as a means of identifying DWPA source areas. Then follow three sections that examine P, NO_3 and SS in turn; evaluating impacts caused by their excessive inputs, how they are mobilised and transferred from the land to water bodies, and what relevant legislation is currently in place to manage agricultural practice.

The next section is concerned with the mitigation of DWPA and looks at the source-pathway-receptor model before providing a critique of the various management options available to farmers and land owners. Runoff attenuation features are then introduced as a type of mitigation that aims to reduce the loss of sediment and nutrients by targeting polluted overland flow pathway; the mitigation experiments conducted in this study are specifically focused on the functioning of RAFs. The final section will discuss the influence of scale (spatial and temporal) on the operation of fluvial geomorphic systems and how this affects both monitoring and management of sediment and nutrient regimes at the catchment scale.

The water quality of our streams and rivers is largely determined by human activity. Regulation of major point sources of pollution such as sewage treatment works and industrial-manufacturing plants, through the introduction of legislative directives such as the Urban Wastewater Treatment Directive (991/271/EEC) and the Industrial Emissions Directive (2010/75/EU) resulted in a significant and relatively rapid improvement in the chemical and biological status of the UK's waterways (Neal and Jarvie, 2005). The reduction in point source pollution has meant that the impacts of diffuse sources of water pollution became increasingly prominent (Foy, 2005; Kronvang *et al.*, 2005).

Diffuse pollution, also known as non-point source pollution, occurs when there is no discrete point of discharge and pollution enters the environment by a multitude of pathways. In comparison with point-source discharges diffuse pollution is more often intermittent and linked to seasonal activities or events such as heavy precipitation, major construction or agricultural tillage. As DWPA often derives from extensive areas of land and is transported to the receiving watercourse via a multitude of pathways, it is difficult to source and quantify and therefore much more difficult to regulate and control than point source pollution (Carpenter *et al.*, 1998). Agricultural activities, such as applying mineral fertiliser and ploughing, *inter alia* give rise to some of the most harmful kinds of diffuse pollution (e.g., Carpenter *et al.*, 1998; Sharpley, 2002; Novotney, 2003; Mainstone *et al.*, 2008; Howarth, 2011) and are discussed in more detail in Section 2.3.

2.2 Eutrophication

Eutrophication is the enrichment of surface waters with nutrients, principally P and NO_3, deriving from human activities (Sharpley, 2002; Sharpley *et al.*, 2003; Neal *et al.*, 2008). This enrichment increases the biological productivity of the water body, i.e., the excessive growth of diatoms, algae and large rooted plants (macrophytes), which in turn stimulates production at higher trophic levels with increases in zooplankton and fish biomass (Foy, 2005; Hilton *et al.*, 2006). Eutrophication can lead to an increase in BOD and the expansion of the anoxic zone; few aquatic macro-invertebrates or vertebrates can survive sustained anoxic conditions (Ferguson, 1996). Agricultural wastes can have particularly high BOD; silage effluent has a BOD value 200-times greater (30000–80000 mg l^{-1}) and cattle slurry 50 times greater (10000–20000 mg l^{-1}) than domestic sewage (Skinner *et al.*, 1997). Table 2.1 lists a number of potential impacts of eutrophication.

Table 2.1: Potential impacts of eutrophication (adapted from Foy, 2005).

Potential impacts of eutrophication
Increase in primary production either as algae or macrophytes
Reduction in water clarity
Replacement of submerged macrophytes by phytoplankton
Increased dominance by blue-green algae which are liable to form surface algal blooms
Increased respiratory demand for dissolved oxygen
Fish kills and reduction in biodiversity at all trophic levels
Increased water treatment costs to remove taste and odour problems
Algal toxins which can threaten public and animal health
Damage to recreational potential and amenity

Hilton *et al.* (2006) estimated the overall annual cost of eutrophication in England and Wales to be approximately £155 million, which includes the cost of water treatment for drinking purposes, loss of biodiversity and amenity value, and also includes the probable cost of remediation. The potential increase in cyanobacteria, or blue-green algae, can have adverse impacts on human and animal health. Toxins have been linked with liver cancer and tumour promotion (Yu, 1995) but perhaps the most dramatic incidence was recorded in 1997 in a Brazilian hospital, where 55 persons died within seven months of exposure to cyanobacterial toxins following routine dialysis treatment (Foy, 2005).

Stevens *et al.* (1999) estimate that if total nitrogen (TN) concentrations in fresh water are at least 0.5 mg l^{-1}, and all other conditions for algal growth are satisfied, total P (TP) concentrations of 0.1 mg l^{-1} (or greater) have a high probability of causing eutrophic conditions and algal blooms. Thus, while both NO_3 and P contribute to eutrophication, there is ample evidence that the main focus for reducing eutrophication should be directed at P (Foy, 2005; Withers and Haygarth, 2007).

2.3 Diffuse water pollution from agriculture

Increased incidence of eutrophication in the UK has largely been attributed to the intensification of agriculture following the Second World War (Lanyon, 1994); specifically the increased use of fertilisers (Kay *et al.*, 2012). Table 2.2 lists a number of key trends associated with post-war agricultural intensification. The Department for the Environment, Food and Rural Affairs - Defra (2005) estimated that agriculture accounts for over 70% of the land area of England and Wales; the geographic extent of the industry means it is inevitable that it will have a significant impact on the environment.

Intensively managed, lowland grasslands, typified by intensive dairy systems (similar to the lowland parts of the upper Eden catchment) occupy 29% of the land area of England and Wales (Defra, 2007b). Activities such as land tillage, spreading of slurry and farm yard manure (FYM), and the use of chemical fertilisers can all give rise to the eutrophication of water supplies. Agriculture is estimated to contribute 60% of NO_3 (Defra, 2009), 25% of P and 75% of SS (Defra, 2007c).

Howarth (2011) reported that between 2004 and 2009 English water companies spent around £189 million removing NO_3, and an unquantifiable amount removing bacterial contamination. On top of this, the EA spent over £140 million on water quality 'issues' in England in 2008-2009 and an estimated £8 million directly on managing DWPA, plus a significant additional expenditure on water quality monitoring. However, with increasing environmental legislation and mounting popular concern for the environment, the importance of good environmental management has now been recognised (Withers and Haygarth, 2007; Defra, 2009). With the introduction of the WFD (discussed in more detail in section 2.8) there is a legislative framework, along with a number of decision support tools and measures, to implement catchment controls over DWPA (Environment Agency, 2007).

Table 2.2: Post World War II agricultural trends.

Post-war agricultural trends
Introduction of winter cropping in the 1960s meant more bare ground at times of maximum rainfall
Cheap mineral fertiliser in the 1970s led to increased amounts spread on the land
Increased use of pesticides to boost productivity and the quality and quantity of food, and for pest control
Switch from unimproved grassland to improved pasture (increased tillage) to feed burgeoning livestock numbers
Increased amounts of farm waste with the need of disposal
Installation of land drains to improve usability and profitability of land
Introduction and expansion of the use of heavy farm machinery led to soil degradation/compaction
Removal of riparian vegetation to create more profitable land

2.3.1 Agricultural practices and water quality

Agricultural activities contribute to water pollution in two principal ways:

- The application of organic and inorganic fertiliser to the land, which increases the amount of nutrients in the soil (available to leaching).
- Hydro-geomorphological impacts, which affect the way that sediment and associated nutrients are mobilised and transferred from land to water.

2.3.1.1 Excess soil nutrients

At the catchment scale, excess inputs of NO_3 and P to agricultural land relative to outputs in produce are closely linked to eutrophication of surface waters (Withers and Lord, 2002; Neal *et al.*, 2008). The application of mineral fertiliser to arable and horticultural crops has, until recently, been very cost-effective. This was, in no small part, a result of the political decision to establish the Common Agricultural Policy (CAP) in 1973 meaning that agricultural subsidies were directly linked to production. Addiscott (2005) provides a detailed review of how the CAP influenced agricultural practices.

One example is of decreasing fertiliser costs as a proportion of profit per unit area of land, to the degree that excess applications of fertiliser were applied to ensure maximum yield rather than maximum efficiency. Applying more fertiliser than is required for the optimum yield greatly increases the opportunity for losses to water bodies, particularly during autumn and winter (Withers and Hodgkinson, 2009). In a 10 ha study catchment in Nigeria, Olarewaju *et al.* (2009) analysed and compared soil and stream water samples throughout a year and reported that high concentrations of NO_3 and P in the topsoil were significantly correlated with high stream water concentrations.

Intensive animal production generally involves the import of feedstocks and the generation of large volumes of animal waste. Disposal problems are comparable to those for raw human sewage, but the regulatory standards for animal waste are generally far less stringent. While nutrients are generally recycled by application to cropland, manure yields from concentrated livestock operations often exceed crop requirements and lead to losses to water (Heathwaite *et al.*, 1998).

2.3.1.2 Hydro-geomorphological impacts

Agricultural activities can have a direct influence on the hydrological functioning of the environment by decreasing the infiltration capacity of the soil and increasing surface runoff (Sharpley, 2002; O'Connell *et al.*, 2005; O' Connell *et al.*, 2007). This also has to be considered alongside a possible increase in soil erosion. Winter cropping means that fields are ploughed and seeded in the autumn, allowing limited crop establishment prior to winter dormancy. The soil is susceptible to erosion as there is limited vegetation cover to protect against detachment processes (Harrod and Theurer, 2002).

Tillage and high stocking densities can lead to soil degradation, compaction, and capping of the surface; reducing the infiltration capability thus generating more surface runoff and erosion (Bilotta *et al.*, 2007b; Withers *et al.*, 2007; Withers and Hodgkinson, 2009). Heavy farm machinery can create 'wheelings', or 'tramlines'. These ubiquitous features reduce local infiltration rates due to soil compaction and can act as a channel for runoff and any diffuse pollutants (Basher and Ross, 2001; Heathwaite *et al.*, 2005). Thus tramlines can increase the hydrological connectivity between distant parts of the catchment and a watercourse and as they often run perpendicular to the hillslope, the potential to act as fast, polluted runoff pathway is significant (Deasy *et al.*, 2009; Deasy *et al.*, 2010).

The degradation of wet areas and riverbanks caused by cattle and sheep has increased in relation to rising livestock numbers (Bilotta *et al.*, 2007a). Poaching is the term used to describe the slurry-like soil conditions that occur on very wet soil when trampled by animals (Drewry, 2006). The majority of damage occurs in wet seasons or during storm events and is when most sediment will be made available for export by surface runoff. The poaching of riverbanks is closely related to the increasing loss of protected riparian zones (where livestock are not excluded). The removal of vegetation diminishes the surface protection layer between animal's hooves and the soil, and the binding effects of roots upon the soil (Bilotta *et al.*, 2007a). It also causes a reduction in the hydraulic roughness and a subsequent increase in flow velocity near the bank (Novotney, 2003). Such situations invariably lead to accelerated channel erosion during periods of high flow.

Perhaps the most significant rural land use factor that changed runoff characteristics of the land was artificial subsurface drainage (Hooda *et al.*, 1999). Withers *et al.* (2000) estimated that 50% of the productive agricultural land in England and Wales has been under drained at some stage. Field drains (also called 'tile' and 'mole' drains) were installed to allow access to fields during wet seasons and to increase the amount of productive land on the farm by reducing surface wetness problems, (Herzon and Helenius, 2008). The impact of drainage on surface water quality may be either positive or negative; as the water table is lowered and infiltration rates increase, the potential for surface runoff, including sediments and associated contaminant losses, are reduced. Conversely, allowing water to by-pass attenuation processes in riparian zones and natural wetlands offsets this potential benefit (Chapman *et al.*, 2003; Sukias and Tanner, 2011). Deasy *et al.* (2009) argue that field drains decrease nutrient sorption/storage by the soil; and connect distant catchment zones directly to the main channel.

A secondary, but highly significant, impact of the increasing use of field drains was the decline in surface drainage ditches on agricultural land. Bradbury and Kirby (2006) argue that there is a widespread need for the reversion of subsurface to surface drainage and the reconciliation of management of ditches for their drainage functions with the support of biodiversity and associated ecosystem services. Herzon and Helenius (2008) concluded that ditches, if managed appropriately, could be used to control water flow and agrochemical transfer as well as provide habitats and increase biodiversity. Management may require periodic de-silting and vegetation removal, the timing of which may have to take into account conservation of certain species and seasonal nutrient uptake.

2.4 Hydrological processes

To understand sediment and nutrient fluxes at the catchment outlet, it is crucial to have an appreciation of the land-water dynamics that contribute to them. The two main preconditions for a chemical element to be transported in a catchment are the availability of material and energy; both are controlled by landscape factors (Gergel *et al.*, 2002). The following section provides a review of the hydrological processes responsible for generating runoff, both surface and subsurface. The level of catchment 'connectivity' is very important when assessing the risk of terrestrial pollutants reaching watercourses and will be described in section 2.4.2. The section will culminate in a review of the critical source area concept, which highlights its importance in the management of DWPA at the catchment scale.

2.4.1 Runoff generation

In the late nineteenth and early twentieth century it was recognised that different parts of the catchment produced different amounts of river flow, both spatially and temporally. This idea of 'lag' in response to rainfall was developed into the unit hydrograph model. This required the hydrograph to be divided into stormflow – from a rainfall event, and baseflow – from groundwater stores. This concept of storage in the catchment proved to be very important as it had vital implications for not only the volume of water, but also the water quality variations and ecological impacts of storm events (Shaw *et al.*, 2011).

Horton (1933) first introduced the concept of runoff generation as the result of rainfall exceeding the infiltration capacity of the soil and producing surface runoff at the hillslope scale. This is referred to as 'infiltration excess', or 'Hortonian' overland flow (depicted as HOF in Figure 2.1 *a*). It was assumed that this occurred uniformly across the catchment and provided a very simple way of back-calculating the amount of infiltration during a storm. However, Betson (1964) brought the Hortonian concept into question saying that infiltration excess overland flow could not occur everywhere, except very rarely during the largest storm events. He demonstrated how small, spatially distributed *areas* within the catchment are responsible for the majority of overland flow as they become saturated more quickly. This will be discussed further below.

In terms of DWPA transport these processes, particularly Hortonian flow, provide little contact time between the soil and flowing water meaning that determinand concentrations will be low when they reach a receiving watercourse. This is sometimes referred to as ‘new water’.

The tailing recessional limb of the hydrograph indicates that not all of the water from the catchment is conveyed through the system at the same speed. Hewlett (1961) observed that storm hydrographs could be recorded in catchments where no overland flow occurred and suggested that the runoff was being transported by subsurface pathways. Weyman (1973) introduced a new mechanism called subsurface stormflow, or ‘throughflow’ that took into account the water table and slope (Figure 2.1 *b*). As the soil profile becomes saturated by water moving though the soil matrix and/or macropores, a perched water table, or ‘saturated wedge’, forms at the foot of the slope. As this water has to percolate through the soil its residence time is greater and its contaminant concentration potentially higher as a result. This concept of displacing stored water in the ground was termed ‘translatory’ or ‘piston’ flow (Hewlett and Hibbert, 1967) and the resulting water referred to as ‘old water’.

In certain situations (e.g., areas of permeable soils) it was believed that subsurface runoff could account for much if not all the storm runoff leaving a catchment (Burt and Pinay, 2005). However, Dunne and Black (1970) demonstrated circumstances when overland flow can be generated on soils with high infiltration capacities; they termed the phrase ‘saturation overland flow’ (depicted as SOF in Figure 2.1 *c*). This process described how areas of the catchment could become saturated by both the downward flow of water within a hillslope and precipitation falling directly onto the area. These saturated areas, also referred to as ‘Variable Source Areas’ (VSAs) (Hewlett, 1961; Ward, 1984) often occur where convergent flow paths meet or where shallow soils overlay an impermeable subsurface layer and can remain close to saturation for prolonged periods of time. In some instances the resulting overland flow could have a component of ‘return flow’, where subsurface water is forced back onto the surface through a seepage face (Shaw *et al.*, 2011). This will result in the mixing of both ‘old’ and ‘new’ water.

New ‘event’ water may reach the channel quickly by Hortonian flow, saturation-excess overland flow or where macropores discharge at the channel banks (not included in Figure 2.1). Water that does infiltrate the soil (recharge) by matrix or macropore flow causes the water table to rise above its pre-event level. This causes flow within the formerly unsaturated zone to become lateral, thus increasing throughflow.

Saturation can also occur above a soil horizon of lower permeability, called a perched water table (Weyman, 1973) and subsurface preferential flow pathways, e.g., pipes in the soil from tree roots, animal burrows, etc., can transfer subsurface flow to the stream in relatively short time periods.

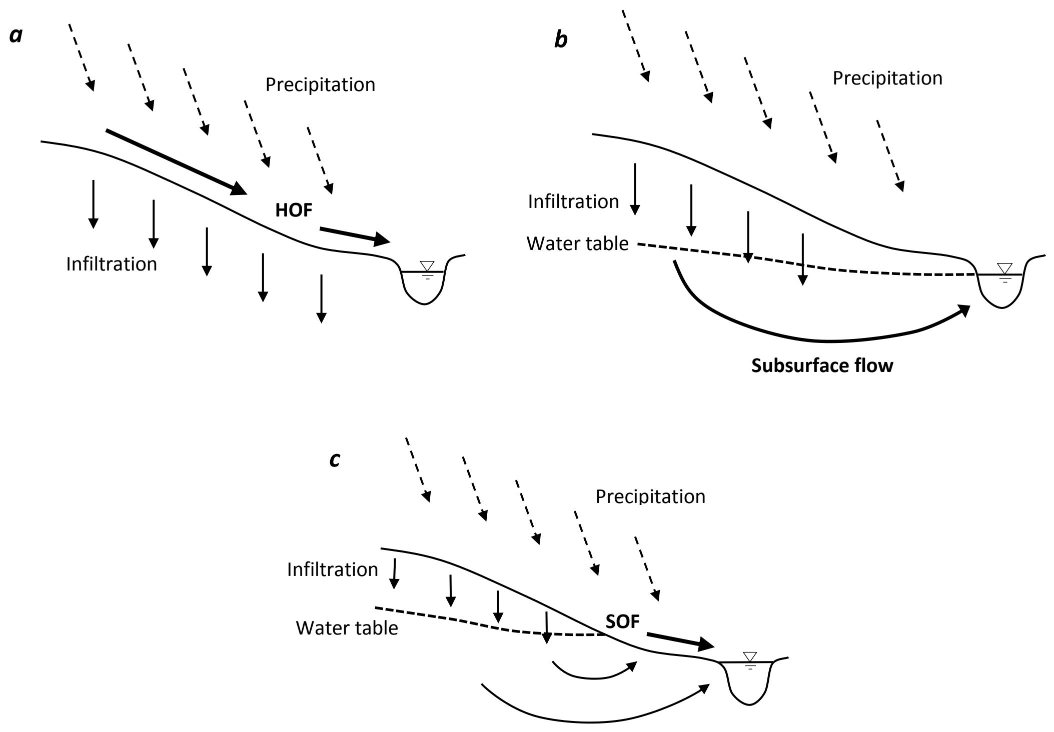


Figure 2.1: Runoff generation processes.

The major concepts of river flow generation are not mutually exclusive; they might all occur in different events in the same catchment, or in the same event in different parts of a catchment (Shaw *et al.*, 2011). Topography, geology and soil type are critical to the geographic distribution of catchment runoff. For example, subsurface runoff will dominate the storm hydrograph where deep permeable soils overlie less permeable soil or bedrock, and where steep hillslopes abut the stream (Anderson and Burt, 1990). Rainfall intensities and prior wetness of the catchment (antecedent conditions) play a vital role in runoff response. A hillslope may generate only subsurface flow during a gentle rainstorm; infiltration-excess surface runoff during a deluge; or subsurface flow alone during a short rainstorm and saturation-excess runoff during a long one (Heathwaite and Dils, 2000). Thus, hydrological processes (and associated contaminant transfer) within catchments reflect a continuum of pathways.

2.4.2 Connectivity and the critical source area concept

The concept of 'connectivity' has evolved from the established runoff processes, i.e., HOF, SOF and VSAs, and encompasses landscape, hydrological and sedimentological systems. It is the coupling of hillslope–channel systems and highlights the important differences in catchment response between hillslopes that are hydrologically 'connected' to a drainage line and those that are disconnected. In terms of sediment and sediment-phase pollutants it is used to describe the potential for sediment transfer from land areas to the water network (Bracken and Croke, 2007). For example, Skinner *et al.* (1997) argued that the coarse fraction of the soil (sand and stones) is likely to be transported short distances only, whereas the finer silt, clay and organic matter can be moved well away from the site with greater potential to reach a water body.

Figure 2.2 depicts the components of a framework that Bracken and Croke (2007) used to conceptualise catchment connectivity. Within each of these components are a number of factors including many complex spatial and temporal patterns, which influence the extent to which a catchment may be regarded as connected. The climate, however, is the key overriding factor; the nature and distribution of rainfall has a significant influence on the runoff regime. These relate to the potential energy required to transport sediment and nutrients from a source to the channel and the likelihood that this energy will be available.

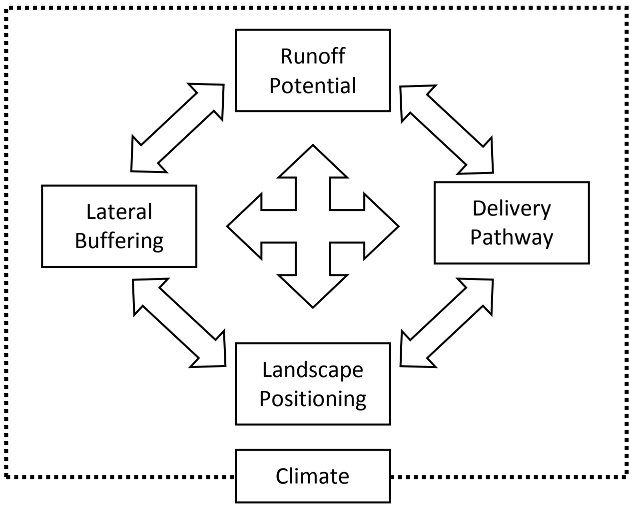


Figure 2.2: The components of catchment connectivity (source: Bracken and Croke, 2007).

In a purely hydrological sense, areas of a catchment that have been influenced by land management and subsequently generate (predominantly) surface runoff are referred to as Critical Source Areas (CSAs) (Pionke *et al.*, 2000; Novotney, 2003; Heathwaite *et al.*, 2005).

Concerning DWPA, CSAs can be further described as specific and identifiable locations within the catchment that are most vulnerable to sediment and nutrient loss in surface runoff, or in subsurface flow when it is an important part of the local hydrology. Consequently, CSAs are often responsible for contributing disproportionate sediment and pollution loads.

The CSA concept is well known as a means of evaluating the spatial variation in the risk of DWPA within a catchment (Gburek *et al.*, 2000). It can be used to predict high pollution risk by combining zones of high soil erosion, high nutrient inputs and/or high soil nutrient concentration (pollution source) with the existence and degree of hydrological connectivity (pollution transport). When attempting to mitigate DWPA, restoration of entire catchments does not make economic sense (Brazier *et al.*, 2005); there is a need to identify and focus on those (relatively small) areas of a catchment that are pivotal in influencing the biological and chemical response of a river system.

Chesapeake Bay is a well-known example of where the CSA concept has been applied. A catchment area of nearly 180,000 km^2 drains into a relatively small area where 76% of NO_3 and 74% of P export was attributed to diffuse sources, of which 58% and 78% of NO_3 and P, respectively, were exported from agricultural sources (Pionke *et al.*, 2000). It became apparent that much of the non-point sources of sediment and nutrients were originating from relatively small and well-defined areas within the catchment, particularly with reference to SS and P, also that most P export originated from relatively few larger storms. Conversely, management of NO_3 depended more on balancing its use across the wider catchment, mainly due to highly diffuse nature of N. Table 2.3 contains a list of system controls for the export of NO_3 and algae available P, based on the Chesapeake dataset. This highlights a potential problem with the targeting of CSAs for mitigation efforts, where solving one water quality problem may lead to the exacerbation of another.

Table 2.3: Systems control on algae available phosphorus and nitrogen export from agricultural land (source: Pionke *et al*., 2000).

Controls	Algae available P	NO_3
Process	Mostly in surface runoff (90%), and in large part dissolved (25-50%)	Mostly in subsurface runoff (70-90%) as NO_3
Spatial	Primary sources of export (90%) are small in area (10%) and predictable	N balance/use distribution (land use distribution)
Temporal	Most export (90%) by stormflow (10%), and mostly (70%) during late winter-spring	None, except most (70%) occurs in winter-spring
Storm	Most export (70%) by largest storms (7 yr^{-1}), with most large storms (5/7) during late winter-spring	Little to none

2.5 Phosphorus

Phosphorus is one of the most important mineral nutrients for biological systems, yet it is also one of the scarcest nutrients in terms of its demand in both terrestrial and freshwater environments. A water body that is deprived of P is often referred to as oligotrophic; this is reported to have an adverse environmental and economic impact by reducing fish populations (Foy, 2005). Conversely, eutrophication has been correlated with high concentrations of P (Carpenter *et al.*, 1998; Sharpley *et al.*, 2003; Withers and Haygarth, 2007).

2.5.1 Phosphorus forms

In the field of water quality chemistry, P is described using several terms and it is vital to have a sound understanding of the terminology. Some of these terms are chemistry based and others are methods based. Orthophosphate is a chemistry-based term that refers to the phosphate molecule all by itself. Soluble reactive P (SRP) is a corresponding method-based term that describes what is actually measured when the test for orthophosphate is performed. Soluble reactive P, sometimes referred to as biologically available P, is considered by many to be almost entirely available for algal growth (Boström *et al.*, 1988; Reynolds and Davies, 2001) and the most important form in terms of eutrophication and its management. As a result, Foy (2005) argues that measures to reduce eutrophication should be targeted at reducing SRP rather than total P (TP). The relationship between P fractions and what constitutes biologically available P for algal growth is complex partly because the precise chemical composition of each P fraction is indeterminate and varies with time and between individual water sources. The operational (methods) defined P fractions are summarised in Figure 2.3.

Total P in water samples is a measure of all the forms of P in the sample (orthophosphate, condensed P, and organic P) and is usually determined by means of the peroxodisulphate oxidation method, as described by Murphy and Riley (1962). Physical P fractioning is based on filtration through a sub-micron filter, for which a pore size of 0.45 µm is most commonly used. Filtration is used to define soluble P (SP) (<0.45 µm) and particulate P (PP) fractions (>0.45 µm).

Some caution has to be taken, however, as Haygarth *et al.* (1998) and Withers and Haygarth (2007) demonstrated this to be not strictly correct. This is because water can contain a continuum of particles below 0.45 µm, referred to as colloids (particles with diameters between 1 nm to 1 µm). Nevertheless, the terminology of SP and PP, as opposed to filtered and unfiltered P, is still seen as the conventional terminology.

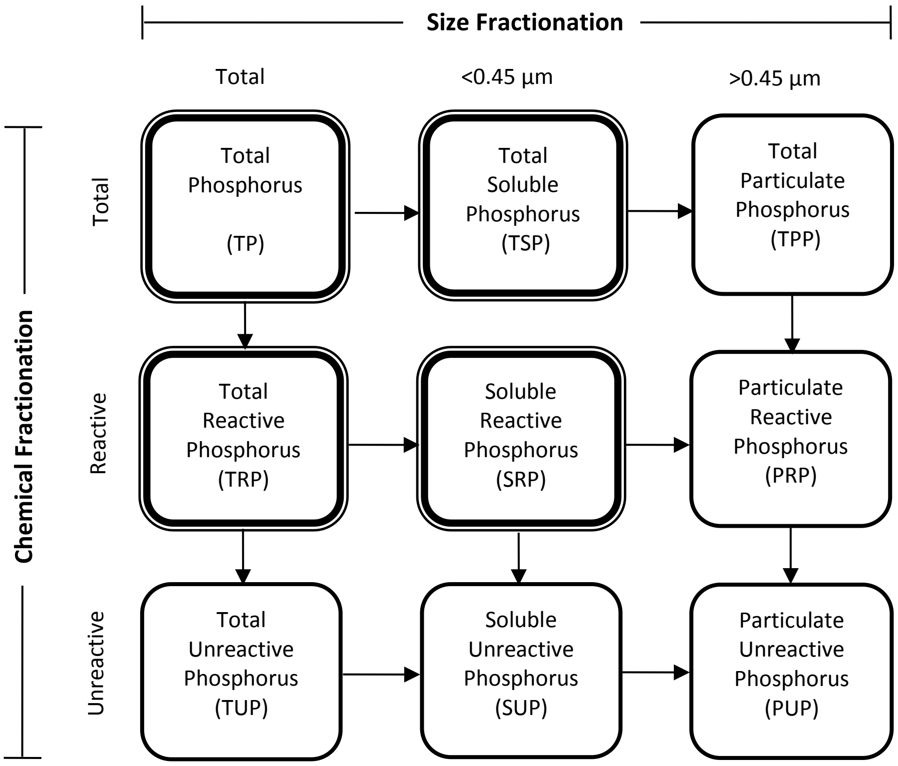


Figure 2.3: Operationally defined phosphorus fractions determined in water (directly determined fractions with bold borders, fractions determined by difference with thin borders) (source: Leinweber *et al.*, 2002).

Chemical fractioning allows the TP (both filtered and unfiltered) to be split into reactive P (RP) and unreactive P (UP). The determination of RP uses the same ascorbic acid/molybdate method as TP but does not require digestion. Although the intention of this method is the determination of soluble inorganic P, the procedure tends to over-estimate concentrations because other forms of P such as labile organic species can also be hydrolysed and subsequently included in the detection, while interference from silica and sample turbidity has also been noted (Condron *et al.*, 2005). Soluble unreactive P (SUP) is often referred to as organic P. However, this is not strictly true, since it can also contain inorganic forms that do not react with molybdate. Therefore, (Leinweber *et al.*, 2002) suggested that it is incorrect to describe RP as inorganic and UP as organic.

2.5.2 Phosphorus in the soil

There has been a perception that fertiliser P is strongly held in the soil matrix in forms unavailable to plants (Baldwin *et al.*, 2002). As a result, fertiliser P input recommendations are generally far above the requirements of plants and have not adequately considered inputs from other sources such as organic matter mineralisation. The P content of many European soils has gradually increased as a consequence forming a reservoir for possible future loss to water (Kronvang, 2007; Ulén *et al.*, 2007). A potential implication of this is that it could take considerable time to see reductions in runoff P concentrations in response to reduced P inputs (as a possible form of water quality mitigation). It has been suggested that 10 years would be needed to see a reduction in SP while a number of decades would be required in order to observe a decline in PP concentrations reaching waters (Withers *et al.*, 2000; Haygarth *et al.*, 2002). Unfortunately, P concentrations in lake waters above 0.02 mg l^{-1}, and river waters above 0.1 mg l^{-1}, are considered to accelerate eutrophication; these values are an order of magnitude lower than plant sustaining soil P concentrations. This disparity between critical soil and water P concentrations highlights the importance of controlling P loss for the terrestrial environment (Sharpley *et al.*, 2003).

Phosphorus is removed from the soil primarily by plant growth followed by crop removal. However, Sharpley *et al.* (2003) suggest that, on average, only 30% of the fertiliser and feed P input to farming systems is output in crops and animal produce. Erosion and surface runoff account for a proportion of P loss, along with leaching to a lesser degree. Phosphorus loss in agricultural runoff is of little agro-economic importance because it typically amounts to only 1-2% of the P applied (Sharpley *et al*., 2003), but as described above, off-site impacts can be much more significant.

Phosphorus cycling in soils is influenced by soil chemistry (e.g., pH, redox potential), soil moisture content, temperature and biological activity - where soils, plants and microorganisms all play an important role. Soil solution P concentrations typically range from <0.01 mg l^{-1} to 1 mg l^{-1} in well fertilised soils but can be as high as 7 to 8 mg l^{-1}. Soil drying and wetting has an important control over microbial P mobilisation which can release substantial amounts of organic P to solution (Condron *et al.*, 2005). Several studies have reported a pronounced seasonal pattern in organic P transfer from soil in a range of environments, with maximum concentrations in the spring and autumn periods. Turner and Haygarth (2000) found organic P concentrations in leachate from cut grassland to be greatest in the spring period; Turner *et al.* (2003) also found that pulses of organic P occurred in first order streams draining UK uplands in the spring. This phenomenon is almost certainly explained by microbial processes (Condron *et al.*, 2005).

In a study that included 22 catchments representative of different agricultural land use practices, Sharpley and Smith (1990) demonstrated how SP concentration decreased with an increase in SS concentration of individual runoff events from unfertilised catchments. Particulate P content decreased as SS concentration of individual runoff events increased. This was attributed to an increased transport of silt-sized (>2µm) particles, of lower P content than the finer clay-sized (<2µm) particles. Larger particles have an increased proportion of primary mineral P (i.e., apatite) that is less bio-available than the P sorbed to clay.

2.5.3 Mobilisation and transport

2.5.3.1 Mobilisation

Before P is transported it has to be mobilised. There are two conceptual processes, described by Haygarth and Jarvis (1999), whereby this occurs:

1. Solubilisation (operationally defined as the P form after < 0.45μm filtration) where the driving mechanism is chemical non-equilibrium
2. Physical detachment of soil particles and colloids with attached P, where the driving mechanism is force exerted by moving water.

The process of solubilisation may be either chemical - resulting from an excess of P in relation to the soils buffering capacity, or biological - resulting in the rapid release of P from organic matter and soil biomass following perturbation (Haygarth *et al.*, 2002). Soil P levels affect chemical solubilisation and thus concentrations of P in drainage. Any increase or decrease in soil P level will broadly be reflected in the export, since there is a general logarithmic relationship between soil P status and solubilisation (Haygarth *et al.*, 2002). Thus, management of P loss by targeting solubilisation has to take this into account and form part of a long-term plan for a catchment.

Detachment is the physical release of soil particles and colloids (>0.45 μm), with P attached. Soil erosion is a selective process, enriching runoff sediment in finer-sized (silt and clay) particles and organic matter (Sharpley and Smith, 1990). As P is strongly sorbed to clay particles (with a larger surface area) (Barrow, 1978; Heathwaite and Dils, 2000) and organic matter contains relatively high concentrations of P, particulate P constitutes the major proportion of P transported in runoff from cultivated land (Kronvang, 1990). Haygarth *et al.* (2002) describe how the detachment process occurs on two tiers. The first involves the steadier, enduring erosion during 'normal' winter rainfall via 'sheet washing', while the second tier involves significant erosion losses during heavy rainfall events. The contact time between runoff water and P source is important in determining the P concentration in runoff. Typical runoff flow velocities are of the order of 0.3 to 15 cm s^{-1} (Dunne, 1983), while typical velocities for pastures are at the low end of this range. Increasing slope length and slow-flowing water will increase contact time and so will be expected to have greater concentrations of P in runoff (Haygarth and Sharpley, 2000; Dougherty *et al.*, 2004). See section 2.7.2 for a more thorough discussion of soil erosion processes.

Preedy *et al.* (2001) and Haygarth *et al.* (2002) also pose a third type of mobilisation, which is concerned with anthropogenic P sources (e.g., manure, fertiliser, etc.), called 'incidental' mobilisation. This is a more direct movement of the P source itself (Gburek *et al.*, 2005) and could include losses from farmyards, hard standings and fields directly after spreading, which coincide with high water flows. Large P applications left on the surface of wet, frozen, compacted, and intensively drained soils are particularly vulnerable to incidental losses. Occurrence depends on the timing and magnitude of runoff events following application. Withers *et al.* (2003) estimate that when incidental P losses occur, they often make the dominant contribution (50-98%) to measured P loads in surface and subsurface runoff.

2.5.3.2 Transport

'Transport' refers to P movement by flowing water once it has been mobilised. 'Transfer' is often used to describe the integration of P mobilisation with the spatial and temporal dynamics of hydrology and resulting transport at the soil and hillslope scale, ultimately to move P to surface waters (Gburek *et al.*, 2005). Beven *et al.* (2005) also used the term 'delivery', usually represented as the ratio of what arrives at a point of interest along a particular transport pathway to the P that was mobilized into that pathway. The term 'loss' is also used and can best describe the transfer of P from one component to another, for example, from a soil or agronomic system to a stream or reservoir. These terms are interchangeable and will all be used hereafter. Pathways of P transfer can be broadly classified into surface and subsurface pathways. The greatest P losses, in association with soil particles, are generally considered to be associated with the former (Haygarth and Sharpley, 2000). Ulén *et al.* (2007) estimated that 40-88% of TP transfer in agricultural catchments is via surface pathways while Pionke *et al.* (2000) reported a 90% loss associated with surface runoff in the Chesapeake Bay catchment.

Surface runoff is an important pathway for P loss but is often spatially limited and temporarily confined to high magnitude, high intensity rainfall events. In a grassland surface runoff experiment in the Pistern Hills, UK, Heathwaite and Dils (2000) measured runoff P losses from top-, mid-, and base-slope plots and recorded TP concentrations of 0.08, 0.11, and 0.15 mg l^{-1} respectively. As well as hillslope position, temporal factors also influence the magnitude of P loss in surface runoff. Higher mean TP concentrations (0.16 – 0.19 mg l^{-1}) were recorded in September and October; autumn storms are often responsible for the greatest losses of P due to high P concentrations in the soil in summer months as a result of fertiliser applications, increased grazing activity and escalated microbial activity due to higher temperatures and soil re-wetting. Conversely, TP concentrations often decrease during the winter months due to source exhaustion, despite increasing larger storms.

Until relatively recently, subsurface pathways have been viewed as fairly insignificant for P transfer owing to the tendency of P to be adsorbed to soil particles. However, research has shown that subsurface pathways can also contribute 12-60% P losses from agricultural fields (Ulén *et al.*, 2007). Groundwater discharge can be an important source of P, particularly organic P, when stream flow is dominated by base flow (a high base flow index – BFI) (Tesoriero *et al.*, 2009).

Preferential flow pathways, particularly soil macropores and field drains, can be important contributors to the overall P load. Again, in the Pistern Hills catchment experiment, Heathwaite and Dils (2000) recorded high P concentrations (mean: 1.2 mg TP l^{-1}) in macropore flow in the upper 0-15 cm of a grassland soil but found that P concentration generally declined with increasing soil depth. Drainflow is a similar mechanism to macropore flow but with significantly greater potential to connect distant parts of the land unit to the stream network; there is also no buffer to ameliorate their impact (Deasy *et al.*, 2009; Sukias and Tanner, 2011).

Figure 2.4 summarises the data gathered by Heathwaite and Dils (2000), showing the variation in magnitude and form of P loss in different hydrological pathways in a grassland catchment.

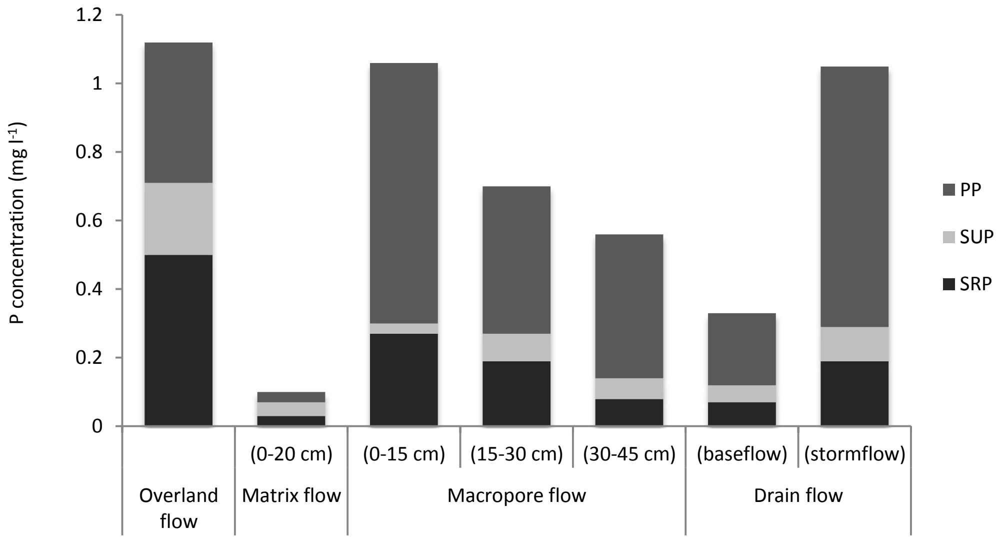


Figure 2.4: Variation in the magnitude and form of P loss in different hydrological pathways for the Pistern Hill catchment, UK (source: Heathwaite and Dils, 2000).

2.5.4 *The influence of land use and storm flow*

Russell *et al.* (1998) reported a P loss range of 160-210 kg km^{-2} yr^{-1} for agricultural catchments in the UK and Jarvie *et al*., (2003) calculated the annual TP export from different sub-catchments in the Herefordshire Wye basin varying between 2 and 90 kg km^{-2} yr^{-1}. Wood *et al.* (2005) working in the predominantly grassland Taw catchment estimated an export of 120 kg km^{-2} yr^{-1}. Ulén *et al.* (2007) also reported that the UK suffered some of the highest P losses in Europe. Thus on average, the typical loss of P to water is estimated at around 100 kg km^{-2} yr^{-1} for grassland farms.

Arable losses are predominantly in particulate form while soluble P is the most important form exported from grasslands and forestry; this is supported by the results of Lemunyon and Gilbert (1993) and McGuckin *et al.* (1999). Grasslands lose most P in soluble form due to their dense vegetative cover, which impedes particulate losses (Haygarth *et al.*, 1998). Investigating two major river catchments in Northern Ireland, the Upper Bann and the Colebrooke, McGuckin *et al.* (1999) found that TP exports from improved and unimproved grasslands to be the same (80 kg km^{-2} yr^{-1}) while SRP exports from unimproved grassland were higher than improved grassland (40-85 kg km^{-2} yr^{-1} and 12-40 kg km^{-2} yr^{-1}, respectively). This is partly attributed to unimproved grassland often being undrained with compacted soils, which can encourage surface runoff. Haygarth *et al*. (1998) suggests that artificial drainage, often associated with improved grasslands, may increase the proportion of PP lost via surface pathways. Drainage (of improved grasslands) can also lead to increased sorption of SRP in the upper soil horizons while opportunity for SRP sorption in unimproved grasslands is limited due to increased amounts of organic carbon (McGuckin *et al.*, 1999).

In a study looking at sediment associated P transport from two intensively farmed catchment areas in Denmark, Kronvang (1990) identified that the annual transport of PP during storm flows was as large, or larger, than the transport during background flow, despite the fact that storm runoff volume comprises only 14-18% of the total runoff. Also that the onset of storms produces high PP transport rates, at least in part as a result of the resuspension of particulate matter accumulated on the bed (possibly during a previous summer drought). The availability of sediment also exerts a strong control on the PP transport; thus reoccurring events in January may give rise to low PP fluxes despite high proportions of storm flow (this issue is discussed in further detail in Section 2.7.4). It was found that 56-66% of the annual P fluxes consisted of PP. 70-90% of the monthly PP fluxes was contributed by short-term storm events; thus, seasonal trends largely reflect changing storm frequencies (Kronvang, 1990). The exhaustion effects (Walling and Webb, 1987) and the resuspension of accumulated sediments further complicate measurement of the P transport.

During a six-month continuous TP monitoring campaign in a 5 km^2 sub-catchment of the Lough Neagh basin (grassland agriculture, impermeable soils and under-drainage), Northern Ireland, Jordan *et al.* (2007) categorised TP transfer into three 'event-types':

Type 1) A Long-term trend associated with baseflow periods where TP concentration was inversely related with stream discharge, thus indicating point sources of P and a concentration effect. Spikes in the TP record were attributed to manure/fertiliser applications and also to small rainfall events that had little or no impact on the stream flow. A diurnal pattern in TP concentrations was caused by either physical/biological interactions that depend on temperature/light, and/or by point source activity during the day.

Type 2) TP transfers driven by storm events, which showed a positive correlation and were typically associated with diffuse runoff from P-rich agricultural soils. Storm-dependent transfer was responsible for the bulk of observed TP; however, contiguous storms demonstrated a decrease in P concentrations indicating a depletion of the P source. This fits with the 'supply limited' model described by Haygarth *et al.* (2004), where hysteresis plots showed a clear 'flushing' effect in early storms where TP concentrations were higher and peaks on the rising limb of the hydrograph. In later storms the TP concentrations closely followed the pattern of the discharge for both rising and falling limbs.

Type 3) Discrete, high magnitude TP transfers unrelated to rainfall or changes in discharge and were caused by pollution incidents.

2.5.5 Phosphorus cycling in water

According to Baldwin *et al.* (2002), P exists in the aquatic environment in one of the following pools:

- Dissolved in the water column.
- Associated with suspended sediment.
- Deposited in bed sediments.
- Incorporated into the biota.

The arrows in Figure 2.5 indicate the exchanges between each of these 'pools' in a conceptual form. Only a brief explanation is provided here, as the chemical cycling and pollutant-sediment interactions associated with P are lengthy and complex. A more detailed explanation can be found in Baldwin *et al.* (2002) and Bowes *et al.* (2003).

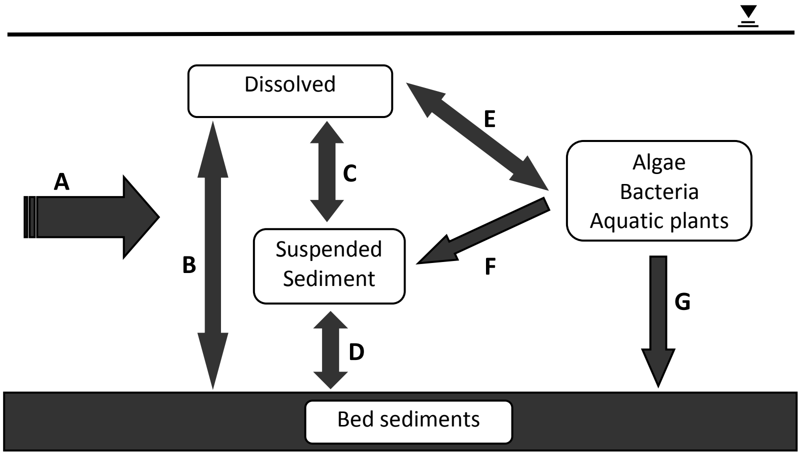


Figure 2.5: A conceptual framework of the phosphorus cycle in aquatic systems**.**

Upstream inputs (A) into a water body can add P into any of the four defined pools. Water and sediment chemistries and biological activity then control the exchange between each of the pools in the water body. Adsorption and desorption exchanges occur between the dissolved pool and the sediment bound pools (B and C). Phosphorus exchanges between the bed sediment and the suspended sediment pools occur through the processes of sedimentation and re-suspension (D) (Baldwin *et al.*, 2002) and P is released from the sediment pools to the dissolved pool by the mineralisation of organic matter present both in the bed and in suspended sediment. Phosphorus is incorporated into the biological pool from the dissolved pool through the growth of algae, bacteria and aquatic plants (E). The release of P from the biological pool back to the dissolved pool occurs either through direct excretion (E) or mineralisation during decomposition following the death of the organism (F to C, and G to B).

Unpolluted freshwaters usually exhibit TP concentrations below 0.025 mg l^{-1}; TP concentrations above 0.05 mg l^{-1} are assumed to be the result of anthropogenic influences. The critical P concentration in water above which eutrophication is likely to be caused is approximately 0.1 mg TP l^{-1}, or 0.03 mg SRP l^{-1} (Leinweber *et al.*, 2002).

2.6 Nitrate

Like P, N is an essential nutrient to both plants and animals, being a vital component of amino acids, proteins and nucleic acids. Although N (atmospheric N: N_2) makes up 78.1% (by volume) of the atmosphere and is a significant component of all soils, it is often a major limitation to the growth of plants (Heathwaite, 1993).

To ensure that plant N availability does not limit crop yields, additional N is often applied in large amounts to agricultural lands as inorganic forms such as nitrate (NO_3) or ammonium

(NH_4) in fertilisers, and in organic forms such as FYM and slurry (Hatch *et al.*, 2002). Figure 2.6 shows a representation of the interaction between crop N uptake and soil mineral N level for arable crops.

However, NO_3 is extremely soluble and can be transferred from terrestrial to aquatic environments with relative ease, causing increasing concentrations in receiving waters (Olarewaju *et al.*, 2009). Historically, concern with elevated NO_3 levels in drinking water stemmed from its potential danger as a cause of methaemoglobinaemia, or blue baby syndrome, in infants (Addiscott *et al.*, 1991). As a result, the EC Drinking Water Directive (98/83/EC) set a maximum limit of 50 mg NO_3 l^{-1} (equivalent to 11.3 mg NO_3-N l^{-1}) and since the 1970s there have been no reported cases in the UK (O'Shea and Wade, 2009).

More recently, the concern with NO_3 pollution arises from its role in the eutrophication of waterways. Seventy per cent of NO_3 entering English waters is estimated to come from agricultural land (Defra, 2007b). Thirty or forty years ago, loss of NO_3 simply implied the loss from the soil of a resource that the farmer would need to replace. Today, however, the concern has shifted from '*loss from'* to '*loss to*', as well as '*where*' has the N gone and in '*what form'* (Addiscott, 2005). The agricultural Nitrates Directive (91/676/EEC) and more recent WFD have targeted NO_3, along with P, as a key nutrient whose concentration/load in aquatic environments should be reduced in order to improve and/or maintain ecological status. These legislative directives will be discussed in Section 2.8.

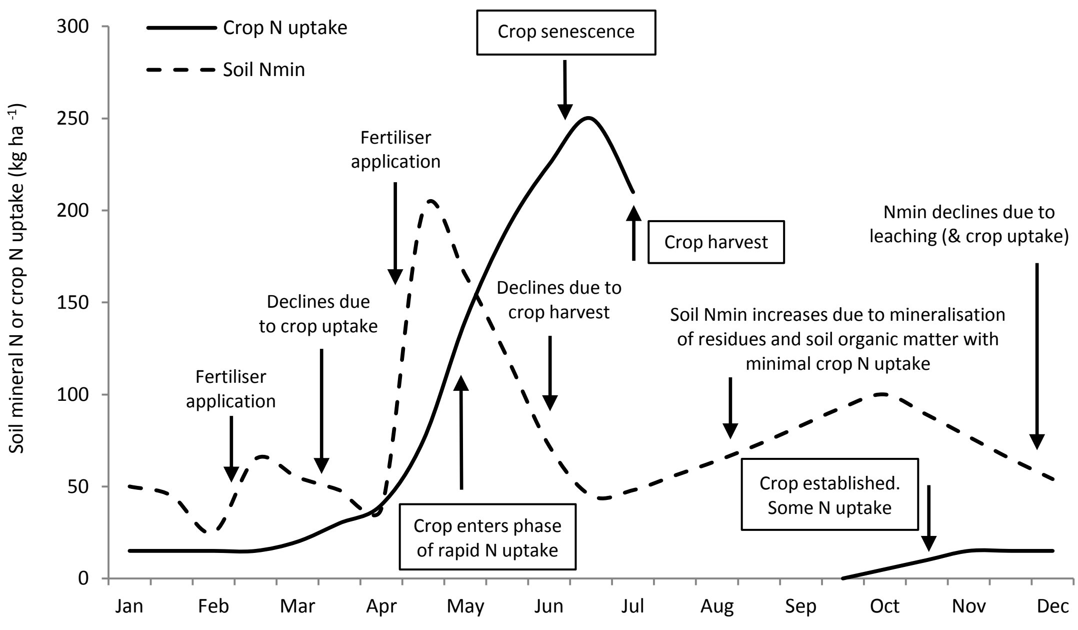


Figure 2.6: Example timeline of nitrogen dynamics showing the leaching risk and synchronicity between N supply from the soil and crop uptake (source: ADAS (2007).

2.6.1 Nitrogen forms and the nitrogen cycle

Forms of N that are of known concern in the context of water pollution are ammonia (NH_3) – which dissolves to form NH_4, nitrite (NO_2) and NO_3. A simplified nitrogen cycle is depicted in Figure 2.7. During the conversion of atmospheric N, cyanobacteria will first convert N_2 into ammonium which is then rapidly nitrified to ammonia by soil micro-organisms and is also held tightly on the negative charges of clay minerals and soil organic matter, and so is relatively immobile and harmless. This is referred to as the nitrogen fixation process. After nitrogen fixation, the NH_3 and NH_4 that is formed will be transferred further, during the nitrification process. Nitrosomonas bacteria first convert NH_3 to NO_2, but as it has a short half-life it does not usually pose a problem to the environment. However, under conditions of high temperature and poor aeration, NH_4 oxidation exceeds NO_2 oxidation and the latter can accumulate. Other factors, including high NO_3 concentrations and pH >7.5, or combinations of these factors, can also lead to nitrite accumulation and subsequent leaching (Hatch *et al.*, 2002). Subsequently, nitrobacter convert NO_2 into NO_3. Plants absorb NH_4 and NO_3 during the assimilation process, after which they are converted into N-containing organic molecules, such as amino acids and DNA.

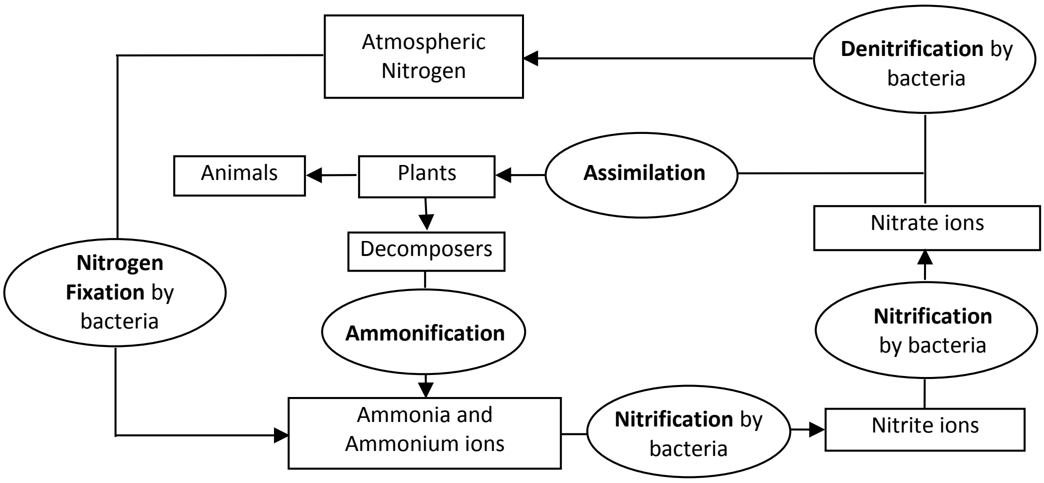


Figure 2.7: A simplified representation of the nitrogen cycle.

Animals cannot absorb nitrates directly; they receive their nutrient supplies by consuming plants or plant-consuming animals. When nitrogen nutrients have served their purpose in plants and animals, specialised decomposing bacteria will start a process called ammonification, to convert them back into ammonia and water-soluble ammonium salts. After the nutrients are converted back into ammonia, anaerobic bacteria will convert them back into nitrogen gas, during a process called denitrification (Vinten and Smith, 1993). Denitrification, as described by Seitzinger *et al.* (2006), is the microbial oxidation of organic matter in which nitrate or nitrite is the terminal electron acceptor. It is a process of anaerobic respiration (suboxic conditions - environments with <0.2 mg O_2 l^{-1}) conducted by bacteria, which can also respire aerobically, and the end product is N_2. Bacteria capable of denitrification are ubiquitous, thus denitrification occurs widely throughout terrestrial, freshwater, and marine systems where the combined conditions of NO_3 and/or NO_2 availability, low oxygen concentrations, and sufficient organic matter occur. Finally, N is released into the atmosphere again. However, nitrous oxide (N_2O) can also be released from the soil during the breakdown of organic matter and nitrogen fertilisers. Nitrous oxide has a global warming potential over 200 times greater than that of CO_2 (Hatch *et al*., 2002).

2.6.2 Nitrate in the soil

Only legume species of plants can obtain N that has been fixed from atmospheric N_2 via a symbiotic relationship with specialised organisms (called Rhizobia) that colonise the roots. Therefore, plants generally take up N as NO_3 or NH_4 from the soil solution.

The NO_3 loss problem occurs because water (rainwater or irrigation water) carries it in solution when it passes through and out of the soil. Nitrification is a key process that mobilises NO_3 and promotes losses to watercourses. Within agricultural soils, the rate of NO_3 production is usually non-limiting, meaning that pool sizes can be considerable. Nitrate is relatively stable, very soluble and does not become fixed on clays or organic matter because of its negative charge; it therefore remains highly mobile. Nitrite is very reactive, toxic to aquatic life, but is usually present in soils and waters in only small quantities. Figure 2.8 depicts the four ultimate fates of NO_3.

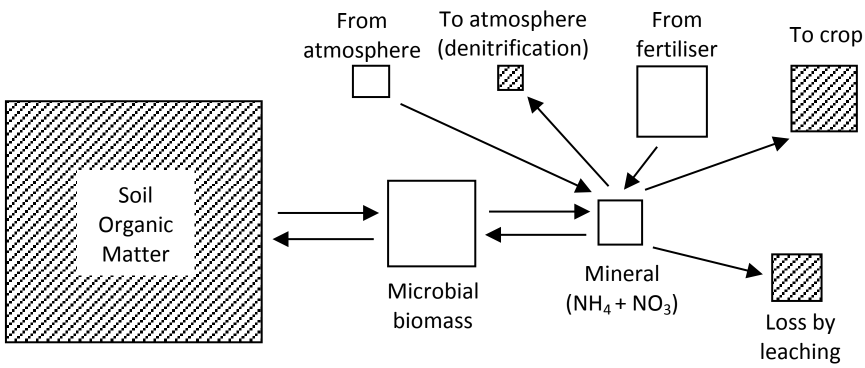


Figure 2.8: The four ultimate fates of nitrate (shaded squares). The size of the square is proportional to the quantity of NO_3 involved (source: Addiscott *et al*., 1991).

2.6.3 Mobilisation and transport

As NO_3 is the most important form of N in terms of water quality impairment and the one used most often in studies of DWPA as a water quality indicator, the rest of the chapter will focus on NO_3 only. Nitrate is found in most natural waters: in rain, rivers, lakes, the sea, and importantly in water stored in porous rocks such as chalk and sandstone.

There are two main general hydraulic pathways by which mobile forms of NO_3 can be transferred from diffuse sources into water bodies:

1. Overland and subsurface lateral runoff.
2. Vertical leaching.

Surface runoff is an important transport mechanism for particulate organic NO_3 and NH_4 adsorbed on to suspended particles in heavily grazed grasslands. Nevertheless, studies have shown that most of the NO_3 lost from both grasslands and crop fields moves through subsurface flow rather than in surface runoff (Heathwaite, 1993; Stevens *et al.*, 1999; Andersen *et al.*, 2001). As a result, Cherry *et al.* (2008) suggested that NO_3 export can only be minimised by limiting inputs and the availability of excess nutrient. Leaching is the major process of NO_3 transfer from hillslope to streams. Downward flow into groundwater is dominant in well-drained soils, while lateral losses predominate in impermeable soils although vertical flow through the soil profile can occur either via bypass flow in large macropores and cracks in heavy textured soils. As a consequence NO_3 is the most common contaminant in aquifer systems (Burkart and Stoner, 2002). Tesoriero *et al.* (2009) found the major source of NO_3 in baseflow-dominated streams was groundwater, while rapid flow pathways were the major source of nitrate in streams with low BFI values.

Specific agricultural activities which contribute substantially to losses of NO_3 include the ploughing of permanent pasture, which releases large amounts of NO_3 through the mineralisation of soil organic matter; leaving land fallow over winter, and application of animal manures or N fertilisers during the autumn when plant uptake is low and over-winter rainfall will increase leaching (Addiscott *et al.*, 1991; Skinner *et al.*, 1997; Addiscott, 2005). Nitrate concentrations in rivers are usually greatest in the autumn, reflecting the first flushes of water from agricultural land. Concentrations in winter often decline during heavy rainfall periods, because a proportion of the flow moves rapidly with limited interaction with the soil and the NO_3 it contains. Sometimes sharp rises in NO_3 concentrations are seen in spring, reflecting applications of fertiliser and organic manures as well as mineralization of readily available soil nitrogen pools (Armstrong and Burt, 1993).

As grassland nearly always has a well-established root system to retrieve NO_3, ten Berge *et al.* (2002) argue that they can be given substantial applications of N fertiliser, up to about 400 kg ha^{-1} yr^{-1}, without appreciable NO_3 losses from the soil. However, this is not to say that NO_3 leaching from grasslands is not a problem. Ryden *et al.* (1984) showed unequivocally that the process occurred because of the non-uniform deposition of urine and dung by cattle and sheep. As more NO_3 is applied to the fields, higher grass yields can support higher stocking densities. However, due to the animals' inefficiency at converting NO_3 into useful products, around 80% consumed by animals is excreted.

2.6.4 A nitrate paradigm shift

In their book 'Farming, Fertilisers and the Nitrate Problem', Addiscott *et al.* (1991) were advocates of the 'nitrate time bomb' and reported both the potential health risks of excess NO_3 to humans and the risks to the environment. However, in his more recent work, Addiscott (2005, pp. 165) stated, "the link between nitrate and stomach cancer is intellectually and administratively dead". Not only that, he also claims, "methaemoglobinaemia is not caused by nitrate in water, but by nitric oxide produced in a defensive reaction against bacterial gastroenteritis". While NO_3 has been shown to cause algal blooms and excessive growth of benthic macroalgae in coastal and estuarine waters, many believe that P not NO_3 is the limiting nutrient for freshwater algal blooms due to the presence of N fixing algal species in these environments. Therefore, the current limit of 50 mg NO_3 l^{-1} has to be questioned. The World Health Organisation (WHO) set European standards for NO_3 in drinking water in 1970, which stated that less than 50 mg l^{-1} was 'satisfactory,' 50-100 mg l^{-1} was 'acceptable' and more than 100 mg l^{-1} was 'not recommended.' The EC adopted 50 mg NO_3 l^{-1} as its upper limit; a seemingly arbitrary decision given that there was little medical evidence of any risk at 100 mg NO_3 l^{-1}. As a consequence of this limit huge sums of money have been spent in trying to reduce NO_3 levels.

However, Addiscott (2005) concluded that despite the possible fallacy that NO_3 was ever a real threat to humans, it inadvertently provided a reason to reform agricultural practices in order to better protect the aquatic environment. Many of these changes have also decreased P and SS exports and without them the state of the impact on the aquatic environment could be a great deal worse.

2.7 Sediment

Soil erosion generates both on-site and off-site impacts and each year millions of tonnes of soil are washed from the Earth's surface into receiving bodies of water. Driven by the demands of the WFD and the need to ensure water quality there has been a shift in emphasis in England and Wales from the on-site impacts of soil erosion (e.g., surface lowering, loss of soil productivity) towards off-site impacts (Brazier *et al.*, 2007). In 2002, the Environment Agency (2002) estimated that soil erosion cost the UK economy around £90 million per annum. Problems arise when the sediment input into streams is increased and accelerated by anthropogenic activities (such as agriculture, forestry, construction and mining) that disturb and expose the land surface and increase erosion rates (Skinner *et al.*, 1997; Novotney, 2003). The main impacts of excess sediment on receiving watercourses are summarised in Table 2.4. However, as this study is concerned with water quality, only the ecological impacts are discussed in more detail in Section 2.7.4.

Table 2.4: Impacts of excess sediment on the aquatic environment.

Potential impacts on watercourses of excess sediment
Increased turbidity cause reduced photosynthetic activity in the stream (Mainstone *et al.*, 2008)
Sediments can damage aquatic habitats such as fish spawning (Harrod & Theurer, 2002)
Deposition can lead to bed aggradation which reduces the conveyance capacity of the channel and may increase flood risk
The high sorption capacity of fine sediment fractions (<0.45µm) means that it is a primary carrier of other pollutants e.g., such as organic components, metals, ammonium ions, phosphates (Owens, et al., 2005; Baldwin et al, 2002)
Economic impacts in terms of increased cost of drinking water purification, the siltation of reservoirs and water abstraction plants, and the loss of income from recreational tourism (Skinner *et al*., 1997)
'Muddy flood' damage to property (Boardman et al., 1994)

The 'transport' of sediment in rivers makes a distinction between SS and bedload sediment. As the transport mechanism of sediments of a given size fraction in water can vary temporally with stream transport capacity, this distinction is not helpful with regard to the issue of excessive agricultural sediment inputs. This study is concerned only with the fine sediment fraction (<63 µm), for which the term 'suspended sediment' is used interchangeably, as the main ecological problems are caused by fine sediment deposition on the bed (rather than when the material is suspended).

The Environment Agency (2007) believe that 23% of rivers in the UK are at risk from excessive inputs of SS. Suspended sediment is also crucial in the transport of sediment-associated pollutants, including particulate nutrients, toxic metals and pathogens, which may bind to fine sediment particles (Edwards and Withers, 2008). For this reason Collins and McGonigle (2008) argued that SS should be given a higher profile in diffuse pollution policy. The rate of SS delivery in a catchment is dependent on the rate of production and the level of connectivity between the source and the channel (as discussed in Section 2.4.1).

2.7.1 Sources

The SS load transported by a watercourse will commonly represent a mixture of sediment derived from different locations and from different source types within the contributing catchment (Walling, 2005). Sediment loss is a naturally occurring process; however, increased water erosion rates on agricultural land occur due to damage to the soil caused by ill-timed, or ill-sited, use. Mostly it happens under wet conditions, by animals' hooves, cultivations, excavations and vehicles travelling on the land (Harrod and Theurer, 2002). All these can take place both on stream banks and more widely throughout the catchment. This section will review the natural processes before considering how they are modified by agricultural activities.

Soil erosion processes within a catchment can initially be divided into either hillslope or channel process. Hillslope processes comprise sheet erosion, rill and gully erosion, and mass movement. Channel processes encompass bed and/or bank erosion. Sheet erosion comprises two processes. The first is raindrop impact (splash) where the loss of kinetic energy as the raindrop hits the soil surface causes the detachment and mobilisation of soil particles. The second is transport of mobilised material via overland flow. Overland flow can also detach particles but in sheet erosion is more important in the transport of soil eroded by the raindrops (Morgan, 2004).

Whereas sheet erosion is normally associated with a uniform degradation of the soils surface, rilling occurs due to a localised concentration of flow and erosive energy. Rills can increase local erosion and the speed in which sediment-laden water can reach the receiving watercourse. Gully erosion is similar to rilling but on a larger scale. It occurs where there is a concentration of flows and forms channels that are too large to be removed by normal agricultural activities. The erosive power of gullying is high on a local scale but often small by comparison with sheet erosion on a basin scale (e.g., 1-5% in agricultural catchments - Harrod and Theurer (2002)). Mass movement is the movement of soil downslope under the influence of gravity. It can either be a slow creep of the soil mass or a rapid collapse of a large soil/rock mass as a landslide.

Channel erosion takes place along the bed and banks of the stream. It involves the natural process of meandering as well as the accelerated erosion due to channel incision and widening. Streams that have increased flow volumes and velocities (possibly due to land use changes in the catchment) are particularly susceptible to increased bed and bank erosion (Novotney, 2003).

The relative importance of each of the above (natural) soil erosion processes and the magnitude of the resulting 'gross' erosion is a function of climate, vegetation, soil type and topography.

Rainfall intensity and droplet size determine the erosive power of the rain. Rainfall seasonal distribution also has an indirect effect on erosion in that it is closely related to vegetation extent and growth. Vegetation is a very effective form of protection against erosion. It shields the soil surface from raindrop impact, binds the soil structure, and improves the infiltration capacity of the soil (thus reducing surface runoff). Conversely, bare soil is the most vulnerable to erosion. Soil hydrology strongly influences the generation and transport of sediment. Soil texture, organic matter content, soil structure and permeability are all factors in soils erodibility (Harrod and Theurer, 2002).

Slope pitch and slope length are the most important topographical factors controlling erosion. Generally the steeper and longer the slope, the greater the risk of erosion due to higher flow velocity and associated higher erosive energy of flowing water (Owens, 2005). Walling (2005) described how a relatively small area of the catchment could contribute most of the SS load at the catchment outlet if it was underlain by an erodible geology and/or susceptible land use; examples include bank collapses or poached areas, which might contribute most of a storm's yield.

Riverbank erosion is the main in-channel source of sediment, the natural occurrence of which depends on the shear stress of the flow causing mechanical failure. There are many variables leading to the spatial variability and rate of bank erosion, which will not be discussed in detail here. However, agricultural activities, particularly the presence of livestock in riparian areas, can accelerate bank erosion through poaching (discussed below). Other sources of sediment in agricultural catchments include drainage ditches, especially when poached by livestock; unpaved roads and farm tracks have also been identified as sediment sources as well as potential transfer pathways (Sheridan and Noske, 2007). Upland afforestation/deforestation has been reported as posing a serious threat to water quality (e.g., Zheng *et al*., 2005).

In an attempt to apportion sediment to different source categories, Collins *et al.* (2009), using the PSYCHIC (Phosphorus and Sediment Yield CHaracterisation In Catchments) model, suggested that the agricultural sector contributed 76% of the total SS load delivered to all rivers across England and Wales (Figure 2.9). It is notable however, that the focus of this assessment is on lowland catchments and does not include assessment of moorland and forestry activities. Although believed to be relatively robust at the national scale, the model outputs should be used conservatively at the catchment scale.

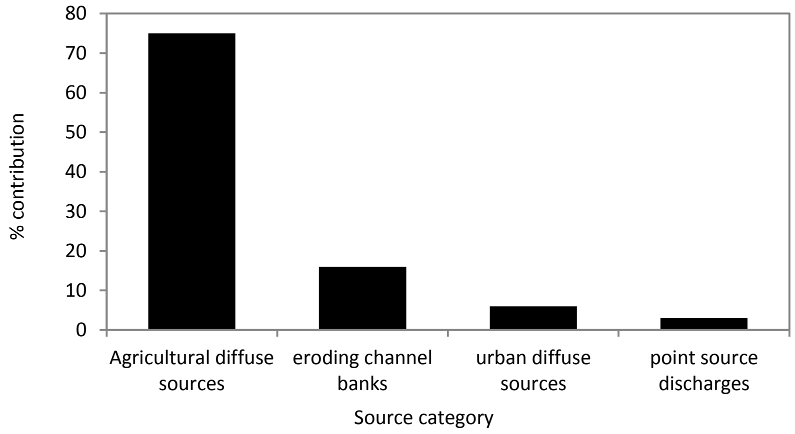


Figure 2.9: National scale sediment source apportionment for England and Wales (year 2000 conditions) (source: Collins *et al*., 2009).

At the catchment scale, Walling *et al.* (1999) conducted a study into the contribution of different types of sediment sources to SS yield in the River Ouse and its tributaries in Yorkshire, UK. Using a 'fingerprinting' method they calculated the relative contributions from uncultivated topsoil, cultivated topsoil and channel banks to the sediment yield to be 25%, 38% and 37% respectively. Thus surface sources, particularly those associated with cultivated areas, dominated in the Ouse catchment.

Also using fingerprinting techniques, Russell *et al.* (2001) investigated sediment sources in two small underdrained agricultural catchments in the UK over two years. One of which (Rosemaund) was dominated by arable land use, and the other (Smisby) was a mixture of arable and dairy pasture. The findings apportioned 10% of SS yield to bank erosion, 55% to field drains and 34% to surface sources in the Rosemaund catchment: and 10% to bank erosion, 30% to field drains and 65% to surface sources in the Smisby catchment. Unlike the Rosemaund catchment, where arable land use was the dominant surface source, pasture areas were considered to be more significant in the Smisby catchment due to the proximity of the pasture areas to the watercourse coupled with high incidence of poaching (also found to be the case in the same catchment by Heathwaite *et al.* (1990)).

2.7.2 Mobilisation and transport

All of the processes described above are influenced, and often exacerbated, by agricultural activities (as introduced in Section 2.3.1). Owens (2005) estimated that human activity might be directly or indirectly responsible for 80-90% of the fluvial sediment delivered to the coastal oceans. Pathways of sediment transport have already been discussed in Section 2.5.4, as P and SS movement is inextricably linked and strong correlations between stream water SS and PP concentrations exist (Kronvang, 2007). Overland flow is the hydrological process that would be implicated in the transport of SS to streams, as the energy associated with the overland flow acts as the driving force for the potential removal of soil particulate matter from the land surface (Scanlon *et al.*, 2004). Walling (2005) found that the highest surface soil contributions were associated with intensively cultivated lowland catchments. This is due to a combination of vulnerable soils, cultivation frequency, timing and method, and lack of crop cover during storm events (Withers *et al.*, 2007).

Semi-permanent tractor wheelings (discussed in Section 2.3.1.2) have the potential to connect distant parts of the catchment to a watercourse and act as fast, polluted runoff pathways. In a survey conducted between 1989 and 1994 by Chambers *et al.* (2000), the presence of tramlines was the major causal factor in 34% of 146 surveyed fields where soil erosion occurred. Bilotta *et al.* (2007b) suggested that overland and/or subsurface flow from grasslands could also contain relatively high concentrations of <0.45 μm sediment particles, both organic and inorganic, especially those in the colloidal size range (0·1–1 μm). This is due to the vegetation cover providing filtration of the coarser particles along with the addition of colloidal-rich material, such as manure and slurry, to grasslands as fertilisers. Pastoral land use often equates to greater stocking densities, relative to arable operations. Animal trampling and poaching can physically detach and mobilise sediment particles; this process is more significant in areas that are well connected to adjacent water bodies. Heathwaite *et al.* (1990) described how heavily grazed and trampled pastureland produced greater surface runoff quantities and also higher SS concentrations, compared with an undisturbed area, in a rural catchment in southwest England.

Relatively recent research has highlighted the potential of field drains to transport such particles. Russell *et al.* (2001) estimated that field drains could be responsible for up to 55% of SS loads in lowland catchments. Chapman *et al.* (2005) recorded concentrations of up to 2600 mg l^{-1}, and Dils and Heathwaite (1999) recorded concentrations of up to 650 mg l^{-1}. Deasy *et al.* (2009) conducted an experiment in the Jubilee catchment, UK, and found that field drains were the dominant pathways for the transfer of runoff and sediment to the stream; surface runoff pathways drained 6.2% of the catchment area and transported around 1% of the catchment sediment load, while subsurface runoff in field drains draining 26.5% of the catchment transported around 24% of the sediment load. Although SS concentrations were found to be higher in surface runoff, the volume of water transported by the drains meant that the overall load was greater.

2.7.3 Sediment in water

In many river systems, most of the suspended load is <2 mm (i.e., sand-sized or less) in size, with much of this being <63 μm (i.e., silt- and clay-sized material) (Baldwin *et al.*, 2002). The <63 μm fraction is of key importance to biochemical fluxes within river systems because the majority of contaminants and nutrients are associated with silt- and clay-sized particles. Conventional water analysis approaches define an arbitrary boundary between 'solute' and 'suspended' load, often at a threshold of either 0·45 μm or 0·7 μm, with particles above these thresholds assumed to travel in suspension and particles below this threshold in solution (Owens, 2005). For the majority of cohesive solids, research has demonstrated that transport frequently occurs in the form of larger aggregates (pedological process), or flocs (waterborne process) (Bilotta and Brazier, 2008). Thus, in a situation where clay would not settle due to low settlement velocity, significant quantities may indeed be deposited if the particles are aggregated with larger particles (Walling, 1990). This has important implications for the mitigation of SS in water bodies (e.g., sediment trap design) and will be discussed in further detail in Section 2.9.3.

2.7.3.1 Ecological Impacts

High concentrations of SS can negatively impact on macrophyte and algal growth, primarily through increased turbidity affecting the amount of light penetrating through the water column, but also by scouring organisms from substrates and by acting as a vector for potentially damaging nutrients, pesticides and herbicides (Bilotta and Brazier, 2008). This can have a direct impact on primary consumers; for example, increased SS concentrations are associated with an increase in invertebrate drift (down- or up-channel migration of organism) and the clogging of feeding structures of filter-feeding invertebrates. Increased turbidity can also impair the vision of many animals relying on sight for catching prey or avoiding predators (Mainstone *et al.*, 2008).

The impact of SS on salmonid fish (trout, whitefish, salmon and grayling) has been extensively researched (e.g., Sear (1993)) mainly because fish are an important economic resource and human food resource. Perhaps the most commonly quoted influence is the deposition of fine material, which can block the pores in the gravel-redd spawning structure and reduce the chance of fish egg survival. Excess sediment can also act as an abrasive to a fish's gills, suppress their immune system and interfere with their natural migration.

The particle size and organic content of the SS also has an important bearing on ecological risk, with organically enriched silts able to exert more oxygen demand and reduce fish egg survival rates (Greig *et al.*, 2005). Heaney *et al.* (2001) found that survival rates of early life stage salmonids declined rapidly as the silt content of spawning gravels increases, particularly over the range from 10 to 15%, mainly due to a decline in oxygen supply to the eggs.

2.7.3.2 Transport

The capacity of rivers to transport SS is very high, although the SS yields of British rivers lie typically in the range 50-100 t km^{-2} yr^{-1} - low by world standards (Walling and Webb, 1987). Upland rivers have higher energy and can transport larger quantities of silt (and more coarse material) than lowland rivers where fine sediment tends to be deposited naturally (Mainstone *et al.*, 2008). This may lead to the conclusion that fine sediment deposition occurs primarily in low-energy lowland rivers; however, Milan *et al.* (2000) argued that considerable amounts of fine sediment can also be deposited in upland rivers during recessional flows and during baseflow conditions, particularly in areas of aggravated bank erosion (i.e., livestock trampling). Unlike sediment from the wider catchment, which enters the river during rainfall events; that from livestock-induced bank erosion enters the river at a time when scouring forces are minimal and is less likely to be transported further down the river system (Mainstone *et al.*, 2008).

Suspended sediment yield per unit runoff is also dependent on between-storm periods and not just the magnitude of the storms. It is common for rapidly consecutive storms to yield lower SS concentrations for the same, or higher, discharges in later storms (Walling and Webb, 1987; Asselman, 1999; Nistor and Church, 2005). In order for concentrations/loads to increase again there has to be a period to allow the replenishment of sediment sources. This will then lead to a relatively high SS export during the subsequent flood event (Walling and Webb, 1987). Suspended sediment yield is also influenced indirectly by the antecedent catchment conditions. For example, a saturated catchment could exhibit connectivity between a critical sediment source which is not connected in previous events and the river (Seeger *et al.*, 2004), meaning there is an increase in yield per unit runoff in consecutive storms. These opposing conditions complicate between-storm SS concentration variability and ultimately, the response depends on the relative influences of catchment wetness and sediment exhaustion on sediment supply (Mills, 2009).

Over a wider temporal scale, seasonality can cause variability in SS concentration and yield due to varying land use and associated vegetation type (Lefrançois *et al.*, 2007), or over an inter-annual scale, the total amount of precipitation.

2.7.3.3 Hysteresis

Fluvial SS transport dynamics are strongly dependent on the dynamics of sediment supply. Walling and Webb (1981) plotted SS concentrations against river discharge and found that SS concentrations increased with discharge and ranged from about 1 mg l^{-1} during low-flows to approximately 1000 mg l^{-1} during extreme flood discharges. However, in other cases the relationship between discharge and SS concentration is often poor (Walling and Webb, 1987; Lenzi and Marchi, 2000), the result of exhaustion of sediment supply. Hysteresis loops are used to describe the non-linear relationship between discharge and SS concentration during storm events and can be highly valuable for making inferences on sediment sources and transport mechanisms (Seeger *et al.*, 2004), although only a brief description will be given here. Hysteresis loops are produced either by a lagged response of one variable, or by an asymmetric response of the two variables, so that a different SS concentration occurs for equivalent discharges in different parts of the storm.

Prowse (1984) described 'true hysteresis' as when the sediment wave lags behind the water wave as a result of energy dissipation in the system; this can only ever cause an anticlockwise loop in the SS/water discharge relationship. Generally, anticlockwise hysteresis indicates that SS concentrations are higher during the latter stages of a hydrograph. This could be the result of water moving through the system faster than the sediment, causing a time lag downstream (Asselman, 1999; Lenzi and Marchi, 2000). A sediment lag could also be caused by a predominance of sediment sources from land areas with relatively high travel time to the catchment outlet.

Arguably, however, the most common type of hysteresis is the clockwise loop direction (Lenzi and Marchi, 2000; Seeger *et al.*, 2004), which may occur for several reasons. Often it is due to the depletion of the sediment supply during the storm event but can also be attributed to in or near channel sources providing easily mobilised sediment on the rising limb, which becomes diluted as water from more distant parts of the catchment begins to contribute to outlet discharge (Jansson, 2002). Asselman (1999) described how different tributaries within a larger catchment contribute different SS concentrations and different travel times to the catchment outlet can result in both clockwise and anticlockwise hysteresis. There are many factors that can complicate the hysteresis type of a catchment and if an area is usually dominated by one type, there is no reason why it could not exhibit another. Multiple or complex loops are usually associated with longer duration/wider coverage storms, which have the potential to mobilise sediment from a greater number of sources and from increasing distances within the catchment. A non-hysteretic response (i.e., where SS concentration is correlated to discharge) indicates that the sediment transport is not supply-limited. This could occur in short-duration storms or in small catchments during extreme events (e.g., Nistor and Church (2005)). Thus, it is important to remember that hysteresis loops alone cannot determine the sediment sources in a catchment due to the large number of factors operating and because a number of scenarios could cause a hysteresis loop of the same shape (Jansson, 2002).

2.8 Current Legislation

2.8.1 The Water Framework Directive

Introduced in 2000 and transposed into UK law in 2003, the EC Water Framework Directive (2000/60/EC) (European Commission, 2000) adopts a comprehensive, integrated catchment management approach whereby land and water are managed as one inter-connected system. The WFD is strongly target-orientated, the overall objective being for all Member States to achieve good ecological and chemical status of water bodies (including groundwater and coastal waters) by 2015 (the first review - the final deadline is 2027). Good status is defined by the UK Technical Advisory Group (UK TAG) on the WFD (2008a p.14) as: "the values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions".

As the 'competent authority' in England and Wales, the EA is responsible for delivering the Directive and where issues have been identified, Programmes of Measures (PoMs) have been devised for each River Basin District (RBD). For the past two decades the EA have used a general quality assessment (GQA) scheme to assess river water quality in terms of chemistry, biology and nutrients. However, the WFD brought about the need for a more sophisticated way of assessing the whole water environment in order to help direct action to where it's most needed. The EA now use a risk-based classification monitoring system where 'poor' individual results drive the overall classification for a water body. It reports on over 30 measures, grouped into ecological and chemical status. Ecological status comprises:

- The condition of biological elements, e.g., fish.
- Concentrations of supporting physico-chemical elements, e.g., N, P, BOD.
- Concentrations of specific pollutants, e.g., copper.
- And for high status, largely undisturbed hydromorphology.

It is recorded on the scale of 'high', 'good', 'moderate', 'poor' or 'bad'. 'High' denotes largely undisturbed conditions and the other classes represent increasing deviation from this natural condition, or 'reference condition' (Environment Agency, 2009). Using the new classification system, results for assessed rivers in England and Wales show that for overall ecological classification 26% of rivers are good or better, 60% are moderate, 12% are poor and 2% are bad (Environment Agency, 2012). The Environment Agency (2009) reported that in the North West river basin district (including the upper Eden catchment), 30% of surface waters met good ecological status or better, meaning that 70% (512 water bodies) did not meet good status. Twenty-two per cent of groundwater bodies were at good overall status with the rest being poor status. The reason for failure in surface waters was chiefly attributed to the invertebrates and fish elements of classification.

2.8.2 Thresholds and guidelines

2.8.2.1 Suspended sediment

The first guideline for SS in rivers was the annual mean concentration of 25 mg l^{-1} cited in the EC Freshwater Fish Directive (78/659/EC), based on the sensitivity of key indicator species such as salmonid fish. This value was later adopted by Natural England before being used in England and Wales under the WFD, in an attempt to realise the goal of 'good ecological status' (although it is not a legal obligation to meet this target). The appropriateness of this annual mean concentration as a threshold target is perhaps questionable considering the general consensus that the majority of the catchment sediment load is transported during a relatively small number of large storm events (meaning the figure is heavily skewed by irregular events and long term averages are likely to differ significantly from annual figures). The UK TAG (2008b) proposed that the guideline standard for SS in the Freshwater Fish Directive should not move directly into the definition of good ecological status under the WFD.

Bilotta and Brazier (2008) also questioned the use of the 25 mg l^{-1} annual mean concentration threshold target and instead proposed an alternative classification based on the duration of exceedance of that threshold. Collins and Anthony (2008) modeled the likelihood of exceedance of the 25 mg l^{-1} target in rivers in England and Wales; they estimated that sediment losses from diffuse agricultural sources need to be reduced by 20% on average in non-compliant catchments, and by as much as 80% in some areas. An alternative classification scheme was proposed by Natural England (Cooper *et al.*, 2008) (Table 2.5), which uses upper- and lower-quartile SSYs as critical thresholds and targets respectively, based on catchment typology as proposed by Walling *et al.* (2008).

Table 2.5: Catchment typology and sediment thresholds (after Cooper *et al*., 2008).

Catchment type (permeability/rainfall/soils)	**Target SS yield ($t\ km^{-2}\ yr^{-1}$)**	**Critical SS yield ($t\ km^{-2}\ yr^{-1}$) (upper quartile)**
High wet and low peat	50	>150
Low wet other	40	>70
Low dry other	20	>50
High wet and high dry other	10	>20
Low dry and low wet Chalk	2	>5

2.8.2.2 Nitrate

The EU Agricultural Nitrates Directive (91/676/EEC), which superseded the voluntary pilot NO_3 scheme under the Water Resources Act in 1991, was introduced to address both the human health issues related to NO_3 contamination and the environmental ones, particularly eutrophication (Withers and Haygarth, 2007; Neal *et al.*, 2008). It was one of the earliest pieces of EU legislation aimed at controlling pollution and improving water quality (Worrall *et al.* (2009) provide a succinct account). While NO_3 levels have stabilised in many member states, the European Union (2010) reported that generally, farming remains responsible for over 50% of the TN discharge into surface waters.

The EC Drinking Water Directive (98/83/EC) provides the basis for national legislation concerning the quality of drinking water. It states that NO_3 concentrations in all public water supplies must be kept below the maximum permissible concentration permitted by the EC of 50 mg l^{-1} (equivalent to 11.3 mg NO_3-N l^{-1}). The recommended limit is 25 mg NO_3 l^{-1}. The Nitrates Directive allows mandatory controls on agricultural activities in areas with high NO_3 levels in the water, areas known as Nitrate Vulnerable Zones (NVZs). In 2009 the area of land in England classed as a NVZ was increased from 55% to 68% (Defra, 2007b).

2.8.2.3 Phosphorus

Setting standards for P has been less clear than for NO_3, where health risks prompted action and a maximum water concentration. A water body with a TP concentration of 0.04 mg l^{-1} is considered to be mesotrophic, which means it has an intermediate level of productivity and a medium level of nutrients. A total P concentration above 0.1 mg l^{-1} means that a water body is considered eutrophic, therefore anything above this should aim to reduce concentrations to below 0.1 mg l^{-1}.

The UK TAG on the WFD originally developed standards for P levels in rivers in 2006 based on analysis carried out on diatoms, as they show greater levels of sensitivity to nutrient pressures than macrophytes. Table 2.6 contains revised standards for SRP (as mean annual concentrations) based on the latest research (UK TAG, 2013). The revised standards were identified using a larger dataset and a new methodology and better match the average biological response to P. The proposed new standards would decrease the proportion of sites with a phosphorus class of good or high from around 80% to 65% (UK TAG, 2013).

Table 2.6: UK Technical Advisory Group standards for phosphorus in rivers (*Lowland* means ≤80 m asl; *Upland* means >80 m asl *Low alkalinity* with a concentration $CaCO_3$ of <50 mg l^{-1}; *High alkalinity* with a concentration $CaCO_3$ of ≥50 mg l^{-1}). The numbers in parentheses are the upper and lower 5th and 95th percentiles (source: UK TAG, 2013).

Type	Annual mean Soluble Reactive Phosphorus (mg l^{-1})			
	Ecological status			
	High	Good	Moderate	Poor
Lowland, low alkalinity	0.019 (0.013-0.026)	0.040 (0.028-0.052)	0.114 (0.087-0.140)	0.842 (0.752-0.918)
Upland, low alkalinity	0.013 (0.013-0.020)	0.028 (0.028-0.041)	0.087 (0.087-0.117)	0.752 (0.752-0.851)
Lowland, high alkalinity	0.036 (0.027-0.050)	0.069 (0.052-0.091)	0.173 (0.141-0.215)	1.003 (0.921-1.098)
Upland, high alkalinity	0.024 (0.018-0.037)	0.048 (0.028-0.070)	0.132 (0.109-0.177)	0.898 (0.829-1.012)

2.9 Mitigation of diffuse water pollution from agriculture

The second half of this thesis is concerned with the mitigation of DWPA. The following section will provide a background to the topic by introducing the source-pathway-receptor conceptual model commonly used in environmental pollution studies and management. It will then provide a review of the various mitigation options available to farmers, which have been developed specifically for reducing DWPA risk. Case studies and examples are provided from the literature along with an introduction to and review of runoff attenuation features, as sediment/nutrient 'transport' mitigation options, as they form the basis of the DWPA mitigation portion of this work.

2.9.1 The source-pathway-receptor concept

The source-pathway-receptor conceptual model is a risk-based approach that provides a framework to manage environmental pollution. The 'source' is the place of origin of a contaminant, or substance, which is located in, on or under the land and has the potential to cause harm to human health, water resources, or the wider environment. The 'pathway' is the means, or route, by which the contaminant can migrate. The receptor is something that could come to harm, including human health, a watercourse or the wider environment, if the contaminant reaches it. In water quality terms, the model seeks to determine what risk, if any, is created by the presence of contaminants, in this case SS, P and/or NO_3, through determining if there are pathways, or 'pollutant linkages' through which the contaminants may impact upon sensitive receptors, and if the risk is acceptable or not. For example, a bare arable field may pose a high risk of soil erosion (i.e., potential sediment source) and there may be a watercourse nearby favoured by salmonid fish for spawning (i.e., sensitive receptor); however, the risk of sediment loss from the terrestrial to the aquatic environments will depend on the level of hydrological connectivity between the two (as discussed in section 2.4.2). In other words, a soil erosion risk does not necessarily equate to a sediment loss risk. Adopting the source-pathway-receptor model for pollution management, specifically the control of DWPA, should help to select the most suitable mitigation option, or suite of options, for a given circumstance.

2.9.2 Mitigation options review

Mitigation options can be conceptually divided in two ways; the first is into 'in-field', 'field margin' or 'in-channel' measures; and the second is into pollutant 'source' (including mobilisation) and 'transport' management options – derived from the source-pathway-receptor model described above. This review will take the form of the latter division, although there is an overlap between the two groupings, where in-field options seek to manage sources and the mobilisation of sediment and nutrients, and field margin and in-channel measures mostly target pollutant transport, i.e., once it has been mobilised from the land. The most appropriate option, or combination of options, will vary according to the DWPA source, pathway, desirability and cost implication from a farming system perspective.

The aim of this section is to provide an account of the nature of the main DWPA mitigation options and also give examples of their success or performance; in this instance, their efficacy in reducing the SS, P and NO_3 losses to water. It is important to note that the mitigation practices, while intended to reduce pollution, should not at the same time decrease farmer income. As this study is primarily interested in 'transport' mitigation options a more in-depth review of these mitigation measures will be given. This will be preceded by an introduction to and critique of 'source' mitigation options.

2.9.2.1 Source management

Conservation agriculture

Conservation agriculture is an in-field management option with three core principles: minimal soil disturbance, permanent vegetation cover and crop rotation. The term is often used as a general collective descriptive for mitigation practices such as no tillage, reduced (or minimal) tillage, residue retention and establishment of cover crops in between successive annual crops (European Conservation Agriculture Federation, 2012). These practices are almost exclusively applied to arable systems meaning that they are not necessarily relevant to the majority of the upper Eden catchment but are more applicable to areas in north-east England. A large literature exists, albeit from plot studies mostly, to support the benefits of these practices in controlling soil erosion (e.g., Fawcett *et al.* (1994); Uri *et al.* (1999)), mainly adopted in the United States. Zhou *et al.* (2009) reported that zero tillage agriculture could reduce sediment loss by over 90% in susceptible catchments on the basis of Water Erosion Prediction Project (WEPP) simulations.

Studies of the effect of conservation principles in the UK are limited, while more exist on the effects on soil erosion than on the effects on sediment loss. In response, the Defra funded Mitigation Options for Phosphorus and Sediment (MOPS) projects (1 and 2) [http://mops2.diffusepollution.info] were designed to investigate the efficacy of different mitigation measures in England and Wales, and evaluate their cost-effectiveness (Deasy *et al.*, 2009b; Deasy *et al.*, 2010). Options tested in MOPS 1 included: crop residue incorporation, contour cultivation, minimum tillage, beetle banks on the contour, and tramline modifications. MOPS 2 examines transport management options and is discussed later.

Mitigation trials were undertaken at the hillslope-scale over three years. Crop residue incorporation (in this case cereal straw) results suggested it can be just as effective at preventing soil erosion on poorly-structured sandy soils as cover cropping, with overwinter SS, TP and TN losses reduced by 40%, 35-50%, and 40-55%, respectively (Deasy *et al.*, 2009b). A reported downside to this option was the potential long-term release of soluble P as the straw decomposed. Contour cultivation resulted in SS reductions of 40-43% and was found to be more effective for ploughed clay soils; however, it was concluded that the practice might increase sediment and nutrient losses if cultivation does not take place exactly on the contour (Deasy *et al*., 2009b) . Beetle banks were combined with contour cultivation (on gentle slopes only) to enhance the buffering effect; results showed a reduction in runoff, SS, TP and TN losses by a further 9-97 %. Minimum tillage was found to be the only option to have net cost savings for the farmer (all the others were roughly neutral) but didn’t take account of possible farm yield decrease; it resulted in a 45-79% reduction in SS loss. In all cases there was a similar reduction in runoff (Deasy *et al.*, 2009b), which indicated that a large proportion of the benefit was in maintaining infiltration rates to prevent HOF.

Stevens *et al.* (2009) tested the effect of shallow depth disk cultivation, in comparison with conventional ploughing methods, and reported no significant reduction in SS losses. It is believed that minimal soil disturbance measures have to be used in conjunction with residue retention in order to reduce surface runoff generation and soil mobilisation (e.g., Blanco-Canqui *et al.* (2009)). Tramline management was believed to offer the greatest mitigation potential for reducing sediment loss in MOPS 1 (Deasy *et al.*, 2010); a simple tine/disc was run through wheelings to reduce near-surface compaction and thus, increase infiltration capacity (other tramline management techniques are discussed by Silgram *et al.* (2010)). Results demonstrated overwinter runoff, SS, TP and TN reductions of 70-99%, although it should be noted that the trial took place in sandy and silty soils only. Withers and Hodgkinson (2009) consider tramline management to be relatively cost effective at the farm scale but that effectiveness depends on soil type.

It is apparent that the adoption of conservation agriculture is dependent upon economic justification, i.e., it is only cost-effective when there are high rates of soil erosion to mitigate, as this has to offset a potential loss of farm productivity. Where soil erosion rates are lower the benefits are more limited (e.g., Leys *et al.* (2007)).

Land use change

Targeted land use change usually involves the reversion of an area of farmed land from erosion-prone, high-risk cultivation activities such as vegetables or maize, to lower risk activities such as cereals or grass. Land use change on a large scale is not particularly viable but when targeted at specific areas on a farm or in a catchment (i.e., individual fields) may make sense environmentally and economically (in terms of making changes to the farming system). Grassland reversion is more commonplace in the United States, particularly in severe soil erosion areas. For example, Kuhnle *et al.* (2008) reported a reduction in annual sediment yield of greater than 60% following a 20% conversion of the Goodwin Creek watershed in Mississippi from cropland to permanent cover between 1982 and 2005. Under arable land use, a change from winter to spring sown cereals may have the potential to reduce soil erosion risk as it reduces the period of time when the soil is vulnerable to erosion by heavy rainfall. This relies on the maintenance of ground cover, be it grass, residue retention or cover cropping, during this period.

Although no data is available for UK studies, experiments in Norway have yielded significant reductions in stream SS and TP loads (e.g., Bechmann and Stålnacke (2005)) and Lundekvam and Skoien (1998) reported a reduction in soil loss of 90% as a result of changing from autumn to spring tillage. Boardman *et al.* (2009) assessed available options for sediment loss mitigation in regard to prevention of muddy floods in the South Downs, UK, and pointed out that while reversion to grass is effective, it is not a farming system that farmers would consider in the area, due to local circumstances. Research has implied that farmers would be more willing to make land use changes if benefits were on-site (i.e., resource protection on their farm) as opposed to off-site (i.e., elsewhere in the catchment) (e.g., Posthumus *et al.* (2011)). Incentives may be required if the latter were to be the case.

Livestock management

Research has suggested that in livestock dominated catchments bank erosion can be the primary sediment (and associated nutrients) source and that it can reach problematic levels with regards to aquatic ecosystem health (e.g., Walling *et al.* (2003)). In a catchment in southwest England, where bank erosion had been shown to be the biggest contributor of sediment to the river, Collins *et al.* (2010) inferred that bank fencing (ten years after installation) was responsible for a dramatic reduction in sediment loss from the banks. Owens *et al.* (1996) investigated a 26 ha catchment in Ohio for 7 years where cattle were allowed access to the stream after which the watercourse was fenced off. Monitoring during the subsequent 5 year period revealed that the annual sediment concentration decreased by more than 50% and the amount of soil lost decreased by 40%.

Perhaps more obvious are the benefits of excluding stock from the watercourse itself in order to prevent animal excreta entering the channel. This may involve the re-routing of stock (and/or vehicle) crossing points over watercourses using a hard crossing point or culvert. Related to this is to (re)site feeding rings and drinking troughs on flat ground as far away as is practical from watercourses to minimise dung and urine gaining direct access to the water.

Intensive grazing can reduce vegetation cover and lead to compaction of near-surface soil. These effects can cause an increase in runoff (Bilotta *et al.*, 2007a), especially where animal movement is concentrated, e.g. around feeding and drinking troughs, in gateways and along paths (Heathwaite *et al.*, 1990; Cuttle *et al.*, 2007). While it is very difficult to avoid topsoil compaction, tillage and natural processes can re-loosen the topsoil. Subsoil compaction is much more persistent and difficult to remove. Artificial loosening of the subsoil with subsoiling equipment (used to break up compacted layers and return the soil's structure to a more natural state) is available but no data on its effectiveness could be found at the time of writing. Thus, it may be preferable that subsoil compaction should be prevented instead of being repaired or compensated and reducing livestock density could be one way of achieving this.

Evans (2005) provided evidence gathered from long term studies in the Peak District, which indicated that erosion caused by overstocking slows rapidly once livestock densities are reduced. This effect may be more prominent in upland areas where topographic and climatic controls mean that relatively high sediment losses can result from vegetation damage.

On a positive note, Posthumus and Morris (2010) allege that livestock farming enterprises are likely to become more extensive (as opposed to intensive) due to the CAP reform and changing economic circumstances; they believe that extensification will lead to a reduction in erosion (and runoff) pressures.

Nutrient management

Nutrient management, or budgeting, can help a farmer save money and reduce DWPA by controlling excess nutrients in the system (Lanyon, 1994). Appropriate timing of fertiliser applications is important to optimise plant uptake and avoid losses due to rainfall-induced runoff or leaching. It is vital to establish nutrient levels in soil, manure and slurry in order to assess the appropriate level of fertiliser applications. By siting field (manure) heaps away from sandy or gravely sites, recently drained land, at least 10m away from any watercourse, and 50m away from a spring, well or borehole will also help reduces nutrient losses to water. In a study covering eight agricultural catchments in Norway, Bechmann *et al.* (2008) found that decreasing trends in nutrient application may have contributed to decreasing trends in N and P losses, but for P the process is generally more long-term because of the build-up of soil P (Kronvang, 2007).

Farmyard management

Although a general awareness exists regarding farmyards as potential contributors of contaminants, especially pesticides and nutrients to surface and groundwater, few actual measurements of runoff composition or fluxes exist. However, Dunne *et al.* (2005) and Edwards *et al.* (2008) both studied runoff from dairy farms and suggested that farmyard dirty water contains considerable amounts of nutrients and contaminants.

Although the composition of runoff varied between individual farms generally there were no seasonal variation in volumes and concentrations of contaminants, and all studied yards showed a potential for a dynamic and rapid linkage with adjacent surface waters, even during relatively light rainfall events. A number of options are available to farmers to help reduce incidental losses of sediment and nutrients from farm buildings, hard standings and tracks.

These options include roofing of manure storage, slurry storage, stock gathering areas and silage stores, roof water should then be directed away into a clean water drain; yard works for clean and dirty water separation; relocation of gateways; stock tracks constructed using wood chippings and hardcore to reduce the amount of poaching and runoff. A full list of options can be found in Defra (2007a). While it is difficult to quantify the benefits of such actions on water quality in terms of numerical evidence, it is assumed that they have a positive impact. Their application should be considered on a case-by-case basis and administered where certain potential pollution issues are identified.

2.9.2.2 Transport management

Buffer strips

A buffer strip, or zone, or sometimes a riparian buffer, can be considered a permanently vegetated area of land usually 2-10 metres in width, most likely but not exclusively adjacent to a watercourse and managed separately from the rest of the field or catchment. The aim of a buffer strip is to reduce the connection between a potential pollution source (most often a cultivated field) and a receiving water body (Muscutt *et al.*, 1993). They are designed to function as a biochemical and physical barrier against pollution. The low-cost and general simplicity of buffer strips have made them attractive mitigation options to farmers; their efficacy in erosion and pollution control has been the subject of numerous studies, which generally show a positive effect on reducing the transfer of SS, pesticides, and nutrients to surface waters.

As runoff water reaches a buffer strip it is forced to slow down due to the increased surface roughness of vegetation, which may promote sedimentation; this vegetation also increases the infiltration capacity of the soil with the presence of its root system. Nitrate retention in a grass strip is dependent on the level of denitrification, degradation and decomposition (Dorioz *et al.*, 2006). Particulate P retention is related to the trapping of fine sediment; however, the finest particles are not always deposited (Uusi-Kämppä *et al.*, 1997); the process of infiltration retains soluble P. The rate of sediment and nutrient retention is also controlled by the concentration, or loading, in the runoff entering the buffer strip, slope width of the buffer, vegetation type and management, source area, and the ratio of a buffer area to a source area (Hoffmann *et al.*, 2009).

Based on a number of experiments, Dorioz *et al*. (2006) reported sediment retention ranging from 40 to 100%, with more than 50% reduction in more than 95% of the cases. The same range of variation was found for PP, with a reduction rate ranging from 50 to 97%; however, they report a very different situation for the soluble forms of P, whose retention percentage varied from -83 to +95, with the most common values being around 20–30%. This means that the load of soluble P can actually increase during transfer across the grass buffer strip due to processes such as reductive dissolution of ferric hydroxides carrying P under anaerobic conditions (Shenker *et al*., 2005), release from organic P and microbial pools (Dorioz *et al*., 2006), and drying/wetting, freezing/thawing and associated microbial activity (Perrott *et al*., 1990). Hoffmann *et al*. (2009) reported that sedimentation was the main physical process in buffer strips and may account for P retention rates of up to 128 kg P ha^{-1} yr^{-1}. Leeds-Harrison *et al.* (1999) investigated the effectiveness of buffer strips to reduce losses of NO_3-N in paired buffered and unbuffered headwater catchments at three sites (with conditions representative of much of the agricultural land in England and Wales) and showed that they did not substantially reduce concentrations entering the streams.

A number of findings suggest that grassed riparian buffer strips may not be effective in controlling DWPA unless the hydrology of the strip allows for a suitable environment. Owens *et al.* (2007) used Astroturf mats located at various points across the width of six buffer strips and found that the majority that collected sediment were at the front of the buffers and that most of the collected sediment was sand-sized (>63 µm). As a consequence they believed that a significant amount of fine sediment (enriched in P) may have passed straight through the buffers. Both Liu *et al.* (2008) and Yuan *et al.* (2009) in their detailed review of the evidence reported that concentrated flow significantly compromises the effectiveness of riparian buffer strips because much of the sediment/nutrients transported by these higher energy pathways is not effectively 'treated'.

The issue of flow concentration, leading to 'break-points', indicates that the effectiveness of buffers is negatively correlated with the degree of runoff occurring along concentrated pathways. As during large events these pathways may account for the majority of runoff (Qiu, 2009), the effectiveness of the buffers may be regarded as much lower than plot derived figures would suggest. Overall, the use of buffer strips appears to provide useful short-term functions in the reduction of SS and P transport to surface waters. However, Dorioz *et al.* (2006) argues that the long-term benefits remain questionable given the relatively short-term use of the approach and the lack of long-term experimental results.

Grassed waterways, bunds, fences and hedges

Grassed waterways (GWWs), or 'swales', are permanent uncultivated strips which follow recurrent flow pathways, particularly in valley bottoms. They operate to reduce DWPA using similar processes as those described for buffer strips: permanent vegetation acts to slow runoff velocity while increasing the infiltration capacity of the soil. As runoff conduits in the agricultural landscape, when compared to cultivated land and/or open ditches, they also offer increased levels of erosion and sediment remobilisation resistance. Evidence has indicated that they may be highly effective in reducing sediment losses.

In a seven year study in Germany, runoff and sediment delivery were measured in paired catchments with and without GWWs; runoff volume was reduced by 90% and 10%, respectively, and sediment delivery by 97% and 77%, respectively (Fiener and Auerswald, 2003). The reductions were attributed to increased infiltration and a reduction in flow energy; they also suggested that GWW efficiency was improved when the channel width was doubled with a flat-bottom, as opposed to being v-shaped. Grain sizes >50 µm were settled due to gravity in both GWWs while smaller grain sizes were primarily removed due to infiltration, which increased with a more effective runoff reduction. Zhou *et al.* (2009) also found that GWWs could potentially reduce SS losses in agricultural catchments but were only economically viable options in high-risk erosion areas, or in the absence of other measures, such as reduced tillage; based on WEPP simulations for arable catchments in Iowa, they calculated a reduction in SS yield from 5.09 to 2.67 t ha^{-1}. However, Evrard *et al.* (2008) observed no infiltration into GWWs and for their study in Belgium suggested that grass strips have a lower infiltration rate and higher runoff coefficient (62–73%) than most cultivated soils. This means that soluble P and NO_3 mitigation potential would be vastly reduced and that SS reduction would rely on filtration by vegetation and reduction in flow energy alone.

The use of field-edge structures, such as small dams, has the advantage that no major changes to land management are required; they have been used specifically to address problems of ephemeral gully erosion (e.g., Boardman (2003)) but also have the potential to reduce peak flows if structures are carefully designed. Fiener *et al.* (2005) monitored four field-edge detention ponds for 8 years and found that they trapped 54-80% of the incoming sediment; lowered peak runoff during heavy rains by a factor of three and lowered peak concentrations of agrochemicals by a factor of two. Chow *et al.* (1999) evaluated the use of terraces and GWWs to reduce sediment losses from potato farms in Canada; they reported a reduction in runoff volume as well as a reduction in soil loss from 2000 to 100 t km^{-2} yr^{-1}. However, Boardman (2003) judged the use of dams as unsuccessful in the South Downs due to their small size and lack of storage capacity during large events. Although the removal of hedgerows and field boundaries during agricultural intensification is an often cited cause of accelerated erosion in the UK in particular (Boardman, 2002), evidence regarding the effectiveness of traditional or modified field boundary structures in mitigating sediment and nutrient losses is lacking.

Sediment traps and basins

Sediment traps are generally excavations (deepening and/or widening) in the bed of a small watercourse and/or small dams constructed across the channel, designed to limit the downstream movement of sediment from upstream sources. Fiener *et al.* (2005) reported numerous positive effects of such features, which included the sediment trapping from upslope, the enrichment of major nutrients in the trapped and delivered sediments, the amount of runoff retained temporarily, the amount of runoff reduced by infiltration, the decrease in peak runoff rate and the decrease in peak concentrations of agrochemicals due to the mixing of different volumes of water within the mitigation features. By confining sediment (and associated nutrient) deposition to a confined area of channel, ditch management costs can be significantly reduced. In a case where a distinct pollution vector is identified, for example runoff from farm hard standings, along tracks and in small ditches, sediment traps may offer an effective solution, although they should be viewed as a mitigation option to be used alongside other (source) management options.

A certain amount of cross-over exists in the literature between sediment traps and small constructed wetlands (CWs). Generally, sediment traps are smaller (thus require significantly less land take) and often deeper with a lower retention time, while CWs are more extensive and are designed to retain runoff for longer time periods. To reduce sediment and nutrient concentrations in runoff, sediment traps rely predominantly on particle settlement (for sedimentation of solids down to coarse- and medium-silt) whereas wetlands utilise other physical, chemical and biological processes (to remove the fine sediment, and dissolved and finely dispersed contaminants). See below for more details on wetlands. A coarse sediment trap is often required as the upstream component of a constructed wetland system in order to prevent the wetland from becoming 'choked up' with coarse sediment; the sediment trap should be quicker/easier to empty and cause less disruption to the surrounding environment, in comparison to a wetland.

The impact of farm dams/ponds was simulated using the WATEM/SEDEM model by Verstraeten and Prosser (2008), who estimated a 47% reduction in SS delivery to the rivers in the Murrumbidgee catchment (New South Wales, Australia). Boix - Fayos *et al.* (2008), using the same model, predicted that check dams (without any other land use changes) would have reduced sediment yield by 77% between 1956 and 1997 in a 47 km^2 catchment in southeast Spain. On a smaller scale, Wang *et al.* (2009) reported that the use of over 200 gulley plugs (check-dams along eroding channels) contributed to a 52% reduction in runoff and an 86% reduction in sediment loss in a 22.5 km^2 Texan catchment. Xiang-zhou *et al.* (2004) estimated the amount of sediment retained by check-dam systems was the largest of all methods trialled in the Loess Plateau, China. They also welcomed the formation of productive farmlands (with enriched fertile soil and ample water), increased flood control and water storage for irrigation. In England, the MOPS 2 project (on-going at the time of writing) is investigating the role of ponds and constructed wetlands as potential mitigation options. Based on two years' worth of data, collected after the construction of ten unlined ponds, sediment trapping rates of 1–7 t km^{-2} yr^{-1} at a clay soil site, 2–40 t km^{-2} yr^{-1} at a silt soil site and >50 t km^{-2} yr^{-1} at a sandy soil site have been reported (Ockenden *et al.*, 2012). Phosphorus retention was also found to be highest at the sandy soil site, with P trapping rates ranging from 0.6 – 100 kg km^{-2} yr^{-1} across all ten sites in the first year.

As trapping efficiency is a function of basin size and related detention time (Braskerud, 2002b; Braskerud, 2002a; Braskerud *et al.*, 2005), this can severely reduce their sediment and nutrient trapping potential (e.g. Owens *et al.* (2007); Boardman *et al.* (2009)) and in some circumstances turn features into pollutant sources. This occurs when previously deposited, easily eroded material (deposited during residual flow conditions and small storm events) is remobilised by high discharges. As storm events are predicted to increase in both frequency and magnitude in the UK as a result of climate change, the need to control emissions during high runoff periods will be increasingly important (Mainstone *et al.*, 2008).

Evidence suggests that sediment traps can be highly effective in reducing sediment/nutrient losses to watercourses if constructed in the correct location and where pollutant concentrations are relatively high. Their ability to attenuate runoff peaks makes them attractive options particularly where DWPA risk is combined with flood risk. There is an important implication for long-term management as traps need to be periodically emptied to remain effective and to help reduce remobilisation of previously trapped material during storm events. Certain design criteria should be adhered to in order to maximise the trapping potential of a sediment trap feature; these principles also apply to wetlands and thus are described below.

Constructed wetlands

Mitsch and Gosselink (2007) describe wetlands as 'the kidneys of the catchment' as they have the capacity to attenuate water flows and improve water quality. The CW concept is based on the holistic use of land to control water quality (Scholz *et al.*, 2007), and when positioned strategically within a farmscape can intercept and 'filter' agricultural runoff (Kadlec *et al.*, 2000; Woltemade, 2000; Zedler, 2003). Wetlands also provide numerous secondary benefits, which include the provision of flood storage, increased groundwater recharge, new wildlife habitat and improved aesthetic value (Díaz *et al.*, 2012). Constructed wetlands are traditionally used to 'treat' regulated inflows from industrial sources; however, less information is available on their performance when supplied by unregulated event-driven inflows.

Within a wetland, the dominant retention processes include: physical filtration of suspended solids; settling of particulate matter; uptake, transformation and breakdown of nutrients, hydrocarbons and pesticides by biomass, plants and microbes; accumulation and decomposition of organic matter; microbial mediated processes such as nitrification and denitrification; and chemical precipitation and sorption of nutrients such as P by soil (Reddy *et al.*, 1999; Koskiaho *et al.*, 2003; Carty *et al.*, 2008). The use of wetlands for the effective treatment of minewater was reported by Younger (2000) and Jarvis and Younger (2000), but their use for the mitigation of DWPA has been relatively limited in the UK to date. However, research carried out in countries such as Norway and Sweden provides strong evidence that CWs have the potential to deliver cost-effective water quality amelioration, although a significant variation in performance is reported across the literature.

Johannesson *et al.* (2011) studied SS and P retention in a 2.1 ha (2% of the catchment area) CW in Sweden using in- and out-flow sampling over 4 years; results revealed a P retention of 17% (280 kg km^{-2} yr^{-1}). They also discovered the sediment thickness was over four-times higher at the inlet and that the sediment P corresponded to almost 80% of the P load; this suggested an efficient settling of PP, also reported by Braskerud (2003). Despite the efficient removal of inflowing PP, Johannesson *et al.* (2011) also highlighted periods of net P release from the wetland. These periods occurred during cold months and were related to relatively high flows, an effect observed by other authors (e.g., Braskerud *et al.* (2005)).

Fisher and Acreman (2004) collated the results of 57 wetland studies from around the world and concluded that 80% of the wetlands reduced NO_3 loading while 84% reduced P loadings in the water flowing through them. Mitsch and Gosselink (2007) also reviewed the results from a number of wetland studies and reported a NO_3 retention range of 40%-95%, while for P, a much greater variation ranging from 0% (in some cases a net loss was recorded) to 99% retention. Based on case studies carried out in the United States, Woltemade (2000) reported removal rates of up to 68% for NO_3-N and 43% for P, but that these values were highly variable, mainly as a result of varying flows affecting retention times.

Carty *et al.* (2008) provides a comprehensive guide to the design, operation and maintenance of CWs and an extensive list of principles have been collated for the design of an effective wetland; these include: increasing storage volume to catchment area ratio (CW/CA ratio) in order to increase residence time (Kadlec *et al.*, 2000; Koskiaho *et al.*, 2003); the use of vegetation and obstructions to slow the velocity of the runoff and avoid preferential flow (Braskerud, 2001) to help promote particle settlement (Uusitalo *et al.*, 2003) and mitigate the resuspension of sediment (Braskerud, 2001); and the provision of a support structure for microbial colonies to develop. The lower the CW/CA ratio, the greater the significance of a hydraulically efficient design; the geometry and arrangement of wetland cells has an important role governing the flow patterns and hydraulic efficiency of the feature (e.g., Koskiaho *et al.* (2003). On balance, supported by a growing source of data, CWs used in agricultural landscapes can result in significant improvement of runoff quality.

Woodchip filters

The use of a woodchip filter, or 'bioreactor', is a relatively novel method for removing NO_3 from agricultural runoff. An existing drainage ditch or tile drain is dug out to approximately 1.5 metres deep and up to a metre wide, lined with a water-tight barrier, back-filled with woodchips and finally sealed under a soil cap. Under anaerobic conditions bacteria on the wood chips metabolize available oxygen, feed on the carbon, and denitrify NO_3 from the runoff water. They are designed to treat much smaller areas than CWs and work best as field-edge features, thus requiring less land take.

Studies in the upper Midwestern US and have shown them to effectively reduce NO_3 levels by 33% on average, but up to 100% during certain conditions (Woli *et al.*, 2010). Greenan *et al.* (2009) and Saliling *et al.* (2007) report positive results from a series of laboratory experiments using woodchips. The Minnesota Department of Agriculture (MDA) monitored a site with a contributing area of 26 acres with a bioreactor 70 m long, 1.8 m deep, and 1 m wide. Results revealed an overall reduction in NO_3 of 28%, which was lower in the winter due to colder air and water temperatures. Although originally designed to reduce specifically the amount of NO_3, they reported that TP average load was also reduced by 79% (Minnesota Department of Agriculture, 2013). They suggested that the longevity of the woodchips is related to the continuous presence of water in the bioreactor, which helps keep the carbon to nitrogen (C/N) ratio high.

The cost effectiveness and practicality of bioreactors depends considerably on the topography and cost of digging the trench and obtaining woodchips. Based on an experiment carried out by the University of Minnesota, the operators estimated that the cost would be £600 ha^{-1} (which included flow control structures, trenching and woodchip costs) (Minnesota Department of Agriculture, 2013).

2.9.3 Runoff Attenuation Features

Runoff Attenuation Features, or RAFs, are soft-engineered landscape interventions developed by the Proactive team at Newcastle University [http://research.ncl.ac.uk/proactive/belford/]. They are designed to intercept or modify a hydrological flow pathway for the purpose of mitigating the impacts of diffuse pollution and flooding. Key design attributes include: easy accommodation into the landscape with little/no impact on farming practices; relatively small size (<500m^2); low cost construction from local materials wherever possible; potential multiple benefits, for example, reduce sediment/nutrient losses (Jonczyk *et al.*, 2008; Barber and Quinn, 2012; Wilkinson *et al.*, in press); provide Natural Flood Management (NFM) (Wilkinson *et al.*, 2010; Nicholson *et al.*, 2012); and habitat creation (Shaw *et al.*, 2011).

There are several RAF types, including bunds, ponds, traps, leaky dams, physical filters and small CWs - all of which seek to '*slow, store and filter*' runoff from agricultural land.

A number of RAFs were trialled at Nafferton Farm (294 ha), Northumberland, to establish their ability to reduce SS, P and NO_3 losses in agricultural runoff. Quinn *et al.* (2007) reported a reduction in TP concentrations of approximately 40% from a combined sediment and P trap during a number of average-sized storms. However, directly following the installation of the P trap (ochre pellets were used to chemically bind soluble P) high SS (> 90%) and TP removal (> 80%) rates were recorded. This was attributed to the physical filtering performed by the ochre pellets and not by chemical processes for which it was originally intended. Negligible removal of NO_3 was attributed to the short residence time in the feature. Field corner scrapes/bunds designed to temporarily store overland flow and farm track runoff and increase infiltration were found to retain approximately 50 m^3 of sediment per annum. A 25 m long in-stream sedge wetland yielded a wide range of results ranging from net losses of SS and TP, during large storms, up to 43% removal of TP during a small event recorded (Jonczyk *et al.*, 2008). The wetland was found to have little impact on NO_3 concentrations; this was attributed mainly to the cold temperatures during the study period.

The RAF approach advocates the use of many (small) features located throughout the landscape, with the benefits accrued by the network of features rather than one large scale/dominant intervention. They are not to be used as a one-stop solution but instead should be applied alongside source/mobilisation options in a holistic, integrated manner.

2.9.4 Legislation and initiatives

This section has described a number of DWPA mitigation options available to farmers and land owners but it is also important to know how these measures are administered and who is responsible for coordinating it. The WFD is the most important, over-riding piece of legislation concerning the improvement and protection of fresh water quality in England and Wales. While the WFD has made the abatement of DWPA a priority, there are several government initiatives that focus more directly on the matter. These are not separate from the WFD but operate alongside and within its framework; they represent the combined efforts of Defra, the EA and Natural England (NE).

Following the CAP reform in 2003, farmers continued to receive direct income payments to maintain income stability, known as the Single Payment System (SPS), but the link to production was severed (Defra, 2005). In addition, farmers now have to respect environmental, food safety and animal welfare standards by complying with a set of Statutory Management Requirements (SMRs) and demonstrate that land is kept in Good Agricultural and

Environmental Condition (GAEC) by following a Code of Good Agricultural Practice (CoGAP) for farmers (Europa, 2010). The England Catchment Sensitive Farming Delivery Initiative (ECSFDI), or Catchment Sensitive Farming (CSF) for brevity, is an agri-environmental scheme which raises awareness of DWPA issues and is responsible for the administration of appropriate mitigation options through Environmental Stewardship schemes, of which there is an entry level and a higher level. By entering into an Entry Level Stewardship (ELS) scheme a farm is committed to adhering to a number of environmental criteria for which it receives an annual payment. Higher Level Stewardship (HLS) aims to deliver significant environmental benefits in priority areas with agreements lasting for ten years. Under the HLS scheme there are a number of options specifically designed for resource protection and protection of fresh water bodies and funding is available on a case-by-case basis. Both ELS and HLS schemes are administered by NE and adopted by farmers on a voluntary basis. There are also a number of schemes such as LEAF (Linking Environment and Farming) and SOWAP (Soil and Water Protection) which provide advice to farmers but are also voluntary.

At present a lot of conjecture is used in predicting the effectiveness of these measures and their impacts on water quality at the catchment scale is uncertain (Kay *et al.*, 2009). Both Bechmann *et al.* (2008) and Kay *et al.* (2009) call for further research into combinations of different mitigation options at the catchment scale, which is of course crucial; however, Deasy *et al.* (2010) highlight that few studies into mitigation effectiveness have yet been trialled even at the field scale. It is thus extremely important to establish the physical effectiveness of these different measures, but also their practicability for farmers.

2.9.5 Summary of mitigation options

This section of the review has indicated that all the available management options described have the potential to mitigate DWPA and thus aid in the improvement and protection of freshwater quality. Conservation measures appear effective in reducing the mobilisation of sediment and nutrients but are more appropriate in arable farming systems or in high risk areas where sediment/nutrient losses threaten sustainability. Better nutrient management, farmyard operations and livestock handling should be promoted as general best practice despite the lack of numerical evidence proving their effectiveness.

In mixed agricultural landscapes, the prevention of animals from entering watercourses in areas where there is a water quality concern would appear to be a simple solution. The fencing off of streams will allow riparian areas to establish and provide natural buffering capacity, will reduce the incidence of bank collapse, and stop animal waste directly entering the water. Other infrastructure improvements related to this issue are improved crossing points (e.g., bridges, armoured crossings) and new water feeders. The use of buffer strips has arguably received the greatest amount of research attention, possibly due to their positive results in reducing sediment and nutrient losses, relative low-cost, and ability to deliver multiple benefits. However, caution should be exercised when using them as they are likely to function less well in areas with concentrated runoff and also where subsurface drains are present. Field-edge bunds and GWWs have a narrower range of application potential as they require specific landscape forms and hydrological regimes in order to target concentrated overland flow pathways. This may entail higher costs and management restrictions but these features have been shown to return positive results in terms of DWPA mitigation.

Sediment traps, CWs and woodchip filters (and RAFs) have all exhibited the ability to retain significant amounts of sediment and nutrients in agricultural runoff. However, they offer more specialised solutions at higher capital costs (particularly CWs and woodchip filters) and their application is more justified in areas of concentrated pollutant-rich runoff and where flood mitigation is also desirable. When wetlands, sediment traps and buffer strips are combined they can also offer conservation benefits. Policy makers, catchment managers and stakeholders must be aware that despite increased acceptance of mitigation methods by farmers due to various drivers, downstream water quality may not demonstrate a trend of improvement, at least in the short-term (Bechmann *et al.*, 2008). Comprehensive management changes are required and more long-term monitoring programmes are clearly needed to assist analyses of catchment response to mitigation.

2.10 The influence of scale on the operation of fluvial geomorphic systems

The influence of scale (spatial and temporal) on process understanding in fluvial geomorphological systems is well recognised (Klemeš, 1983; Bloshchl, 2001). Environmentalists are interested in spatial patterns because they are essential in scaling-up from localised measurements to provide assessments of pollutant losses, and mitigation impacts, at the catchment, regional or national scale for policy purposes. A particular difficulty is that a change in scale often results in a change in a range of factors such as hydrological pathways, sources, and viable measurement methods (Dougherty *et al.*, 2004). Temporal patterns are particularly important with regards to the monitoring of DWPA mitigation efforts, as there can be a significant time lag between the implementation of an intervention and its impact at the catchment outlet; for example, NO_3, as leaching pathways between soils, groundwater and rivers are generally long and complex (Collins and McGonigle, 2008). The main components of lag time include the time required for an installed practice to produce an effect, the time required for the effect to be delivered to the water resource, the time required for the water body to respond to the effect, and the effectiveness of the monitoring program to measure the response (Meals *et al.*, 2009).

The WFD stipulates that the management of rivers and their catchments, including DWPA mitigation, should be holistic in its approach and undertaken at the catchment scale. However, for practical reasons the majority of studies on sediment and nutrient sources, mobilisation and transportation, and mitigation measures are conducted at a smaller scale (lysimeters/plot/hillslope - <1km^2) (Brazier *et al.*, 2005). Therefore, while there is a relatively good understanding of the relevant processes at these scales, there is considerable uncertainty in the application of such findings at the catchment scale.

Slaymaker (2006) described how geomorphic systems are more than the sum of their parts; the problem of discontinuity (Addiscott *et al.*, 1991) means catchment modellers cannot simply 'multiply up' from small-scale measurements. For example, Parsons *et al.* (2006) demonstrated how soil erosion rates estimated from field scale measurements cannot be multiplied to give catchment scale sediment yields as these estimates ignored the fact that travel distances to receptors increase with scale; so proportionally less sediment reaches the outlet as the area of measurement increased. That is not to say that these smaller-scale processes are not occurring, just that they are operating in combination with each other and that new processes become apparent at larger scales.

These are referred to as 'emergent properties' (Klemeš, 1983; Cammeraat, 2002; Slaymaker, 2006). This means that processes that are dominant at the plot/hillslope scale are not dominant at the catchment scale (Sidle, 2006); there are in fact different process domains at different scales within a catchment (Slaymaker, 2006). For example, in terms of sediment (and associated nutrient) loss, sheet and rill erosion may dominate at the small (hillslope-field scale), while at larger scales, geomorphic processes such as gully and channel erosion, and mass movement account for much larger transfers.

Slaymaker (2006) refers to this problem as the 'scale linkage problem' (see Figure 2.10), which begs the question: 'to what extent is it possible to transfer findings from one scale of investigation to another'? Soulsby (2006) suggested that the hydrological dynamics at the catchment scale represent an averaging of the smaller scale heterogeneous processes; therefore experiments to understand large-scale catchment function must be designed to be representative of the overall system, rather than being site-specific.

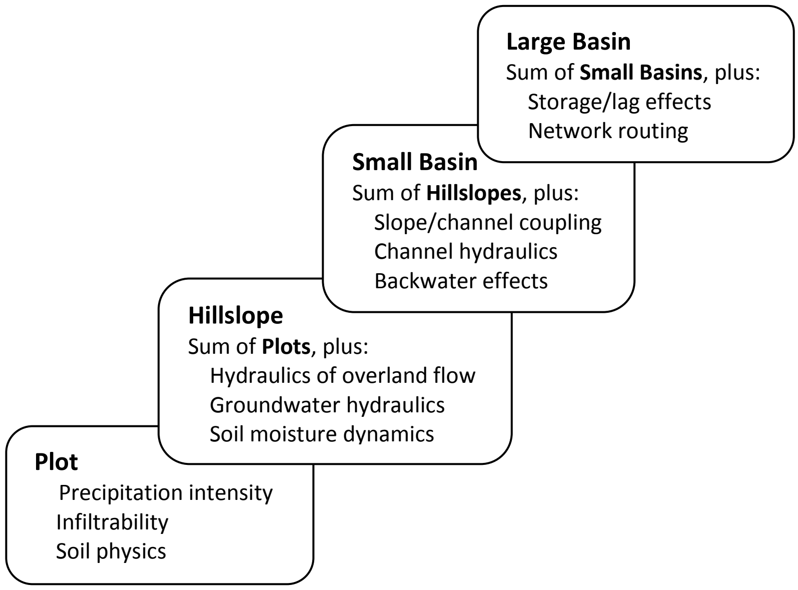


Figure 2.10: The scale linkage problem (source: Slaymaker, 2006).

Nested catchment studies, such as the Catchment Hydrology And Sustainable Management (CHASM) programme (see Chapter 3) and the Demonstration Test Catchments [DTC - http://www.demonstratingcatchmentmanagement.net/], represent an attempt to understand how processes in catchments function at different scales and how these relate to each other. For convenience, such studies have often been arbitrarily divided on the basis of the most likely processes occurring at each scale (Dougherty *et al.*, 2004); they can be categorised as laboratory-, profile-, plot-, field-, and catchment-scale studies.

2.10.1 Relationships between scale and the transfer of sediment and nutrients

The specific yield of a catchment, be it SS, NO_3 or P, is the total load of the river divided by the catchment area and is therefore an expression of the mean rate of production per unit area per unit time (usually in t km^{-2} yr^{-1} for SS, or sometimes kg km^{-2} yr^{-1} for P and NO_3). Verstraeten and Poesen (2001) found no significant correlations between SS yield and seemingly important factors such as mean slope, soil erodibility, and proportion of the catchment under agricultural production. Catchment area was found to be the dominant control on SS yield, with this particular parameter masking a number of relevant factors thought to control sediment transport. They observed a decrease in yield with increasing catchment area (an inverse relationship) and attributed it to the fact that the proportion of the catchment acting as a sediment sink increases with increasing catchment area.

The negative relationship between SS yield and catchment area is widely recognised (Schumm, 1977; Walling, 1983) and assumes that hillslopes are the main sediment source and that sources are uniformly distributed. It occurs if hillslopes or gullies are the main sediment supply because erosion is greatest in the headwaters, while sediment deposition increases with distance downstream (Church *et al.*, 1999). The downstream decrease is accentuated if, as is often the case, rainfall (and thence erosive energy) is also higher in the headwaters than in the lower catchment (Birkinshaw and Bathurst, 2006).

However, a positive (or direct) relationship can occur if channel bank erosion is the main sediment source as channel bank height generally increases downstream and therefore has greater potential to collapse and supply sediment. Also, elevation controlled non-uniformity of land use (e.g., moorland or forestry at higher elevations, arable at lower elevations) can produce an upward trend (Birkinshaw and Bathurst, 2006). Bull *et al.* (1995) found the contribution of channel sources increased the SS yield downstream in the upper River Severn, UK, which more than compensated for the decreasing contribution of sediment from other sources. Church *et al.* (1999) and Dedkov (2004) showed that sediment yield tended to increase with catchment area in undisturbed catchments but that the relationship was negative in disturbed (e.g., cultivated) catchments.

Concerning P, Wood *et al.* (2005) carried out an experiment in the Taw catchment, UK, and recorded significantly lower P concentrations at the catchment scale than plot-scale losses, which suggested dilution from other lower yielding catchment areas. The major source of this dilution was likely to be water from the high-rainfall upland areas of the Taw catchment, known to be low in P. However, it was concluded that although this dilution meant P impacts on aquatic ecology were not of immediate concern in the catchment, in the neighbouring Torridge and Tamar catchments, where there was no dilution from upland headwaters, algal blooms resulting from eutrophication had been observed. Uncertainty still exists over the intensity of agricultural diffuse pollution required within the catchment system to cause significant eutrophication response (Jarvie *et al.*, 2008b). For example, excessive P loads can be measured in runoff from an individual field due to poor management of manures, but only a few hundred metres downstream they may have dissipated due to the effects of dilution from 'good' quality water.

It is well known that detecting trends in this type of data is difficult because the natural variation is large; the retention of nutrients in the system is high, causing delays in the effect; and there is large inherent uncertainty in the determination of agricultural contribution because it is often estimated as the residual when all other contributors are subtracted from the measured total load (Kronvang, 2007). Soil P decline was estimated by Schulte *et al*. (2010)

using data from Irish plot-scale experiments; a model was developed to predict the time required to move from excessive (Index 4) to the upper boundary of the optimum (Index 3) soil P concentration range. For worst-case scenarios, average time to the boundary was estimated at 7–15 years. Maguire *et al.* (2009) report that despite more than 25 years of efforts to decrease agricultural P surpluses in the soil in Denmark, a surplus of 11 kg P ha^{-1} remains. Concerning NO_3, Howden *et al*. (2010) find no system recovery in the last 40 years working on a 140 year fluvial NO_3 concentration record for the River Thames, while Lord *et al.* (2007) calculate a 58-131 year delay before borehole NO_3 concentrations fall to 50% of their initial values. This suggests that conclusions drawn from short-term monitoring could be erroneous and that such *delays* are difficult to quantify. The potential extended life span make conduction of experiments demanding (Meals *et al.*, 2009) and time frames may be substantially greater than that for achieving good ecological status under the WFD.

Haygarth (2010) described the considerable lag time between the occurrence of the initial starting event in the landscape and the water quality impact at the catchment outlet as extremely important. This means that changes made at source, for example, the application of DWPA mitigation measures, may take considerable time to register a signal at the catchment outlet due to catchment buffering and long transit times (>50 years) (Cherry *et al*., 2008). Thus, it is unlikely that responses to intervention will be observed by 2015 in many water bodies. Realistic timescales for achievement of good status for groundwaters or groundwater dominated surface waters must be based on estimates of catchment specific time lags (e.g., Fenton *et al*., 2011).

2.11 Summary

This chapter has provided information on the main subjects of the thesis with references and examples from the relevant literature. It has described the importance of both improving and protecting freshwater bodies for social, economic and environmental benefits. Diffuse Water Pollution for Agriculture has been identified as posing a significant threat to the WFD goal of achieving good ecological status by 2015. A number of DWPA mitigation options have been deliberated, all of which have merits and limitations. However, the focus of this study is on the use of RAFs as a form of contaminant 'transport' management. It is considered that interventions such as RAFs have received relatively less research attention and their scope to deliver multiple-benefits, by targeting polluted flow pathways, could make them desirable to both farmers and funding bodies.

To be justifiable for wider application, any mitigation measure, including RAFs, requires quantitative evidence confirming its effectiveness. Further research is required to understand the integrated impact of DWPA mitigation measures on the water quality of larger scale catchments. However, the heterogeneity of hydrological processes and controlling factors between catchments, which cause environmental 'noise' at the larger catchment scale, makes it more difficult to detect a change in pollutant signal due to intervention. The influence of spatial and temporal scale only confounds this issue further due to, for example, the problem of response delay. So while the gathering of data at larger scales is of high importance the pressure of the WFD requires more imminent action. Thus, it is thus apparent that delivering on the WFD requires coordination that transcends a continuum of scales (e.g., Winter *et al*., 2011). In other words, the appropriate scale for monitoring catchments differs from that for testing, and management at the field- and farm-scale remains crucial to water quality outcomes.

As such this thesis will assess the efficacy of RAFs at the local scale, using input/output measuring techniques. It will also involve catchment-scale monitoring to describe, with sufficient accuracy, the relevant catchment processes leading to the export of DWPA and how they behave temporally by using a data-driven approach. This will allow better understanding of the influence of spatial scale, land cover and management, and other catchment properties on runoff, sediment and nutrient regimes. The outcomes of this will help identify ailing sub-catchments in which to target mitigation efforts, and provide representation of the dominant pollutant source-pathways in order to select the most appropriate type of mitigation intervention.

3. General methodology

3.1 Introduction

The aim of this chapter is to describe the methods used to ascertain SS, P and NO_3 concentration and hydrometeorological data for a number of sub-catchments, which are then examined in detail in Chapter 4. The first part describes the study area, including topographic, geological, pedological and climatic characteristics. This is followed by the methodological approach, which includes the experimental design adopted in the project. The experimental design provides details on the location of water quality monitoring sites, how samples were collected and the laboratory analyses used to determine concentrations of SS, P and NO_3. It also describes the collection and handling of hydrometeorological data, including precipitation, river discharge and evapotranspiration.

3.2 The Study Area

The River Eden catchment (Figure 3.1) in Cumbria covers an area of 2288 km^2 and drains parts of the Lake District and Pennine Hills of northwest England. The area includes 90 Sites of Special Scientific Interest (SSSI), two Areas of Outstanding Natural Beauty (AONB), parts of two National Parks (The Yorkshire Dales and Lake District), and Hadrian's Wall World Heritage Site.

The River Eden itself rises on the limestone hills of Mallerstang Common (675 m AOD – above ordinance datum), on the border of Cumbria and Yorkshire, and flows 130 kilometres in a north-westerly direction before discharging into the Irish Sea, via the Solway Firth. The river and its tributaries are a Special Area of Conservation (SAC). They are excellent for salmon fishing, support a sea trout run (as well as many other species of fish) otters and native crayfish (Walsh, 2004). A high diversity of breeding birds is also supported. The Eden is also host to an unusual and exceptionally rich aquatic flora with 183 plant species recorded, the highest of all rivers in Britain, apparently due to the degree of variation in the physical and chemical character of the river, resulting from the variety in underlying geology (Eden Rivers Trust, 2011).

The upper Eden catchment study area is highlighted in black outline in Figure 3.1.

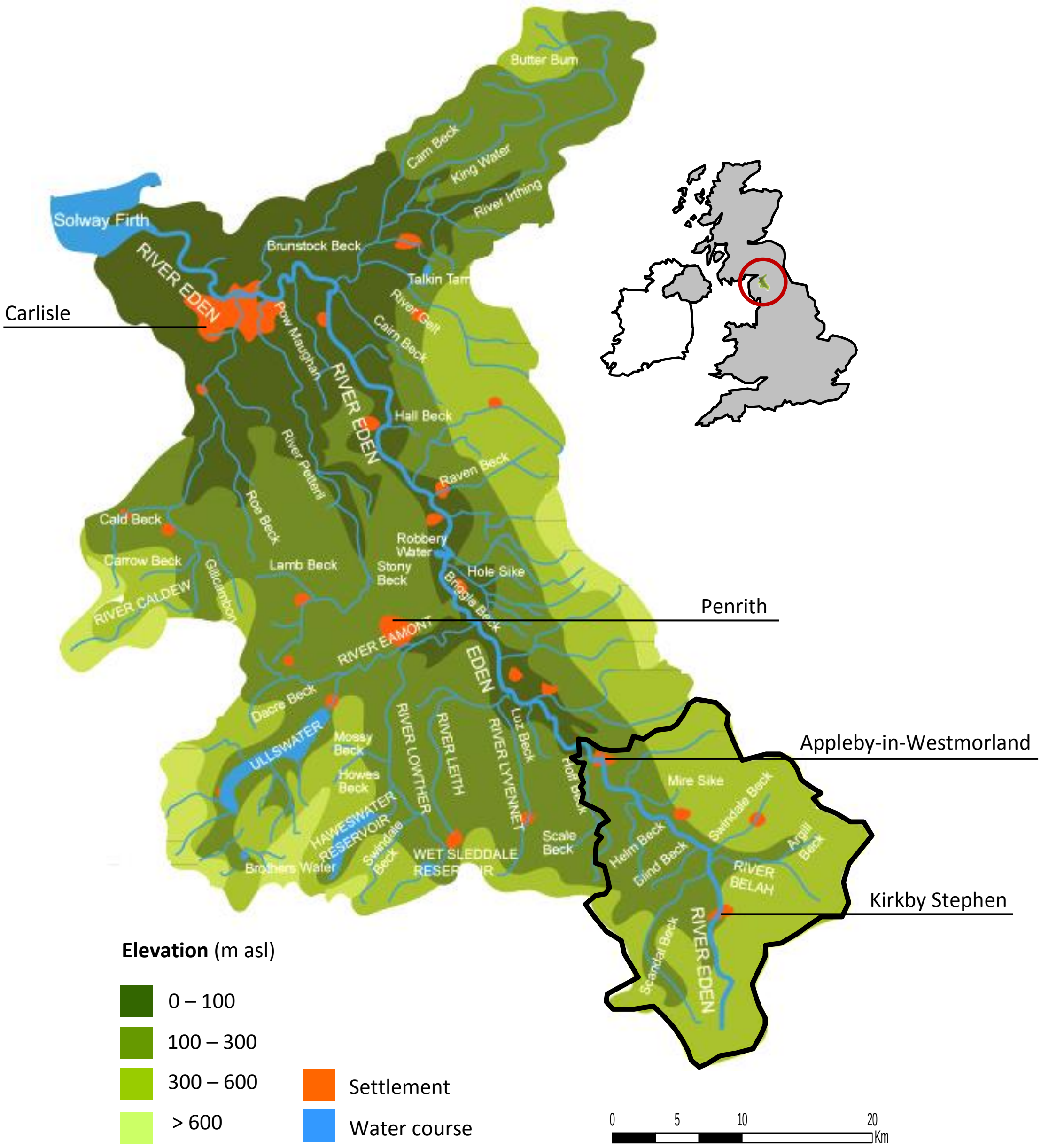


Figure 3.1: The River Eden catchment. The upper Eden study catchment is indicated by the black outline (adapted from the Eden Rivers Trust, 2011).

3.2.1 The CHASM project

The upper Eden catchment has been the subject of dense, nested instrumentation since 2003 as a result of the Catchment Hydrology And Sustainable Management (CHASM) project. The principal aim of the project was to investigate many of the fundamental issues in catchment hydrology and management (e.g., land use, flooding, abstractions, water quality and ecology) and to bridge the gap between micro (~1 km^2), mini (~10km^2), and meso catchment (~100 km^2)

scale response, using nested, multi-scale monitoring networks (Quinn *et al.*, 2000; Mayes *et al.*, 2006). The Eden was one of four catchments investigated by the CHASM program, the others being: the upper Severn (Wales), the Oona (Northern Ireland) and the Feshie (Scotland). A number of PhD studies have been conducted in the upper Eden as a result of the CHASM program. These include the simulation and analysis of flow regimes (Walsh, 2004), the investigation of spatial behaviour of rainfall and flooding (Wilkinson, 2009), groundwater and recharge processes (Fragalà, 2009), and the study of scale dependency of sediment yield (Mills, 2009).

3.2.2 Topography

The Pennine Hills and Howgill Fells dominate the eastern and southern parts of the upper Eden catchment respectively, reaching altitudes of over 700 m AOD (Figure 3.2). The central part of the study catchment, from Kirkby Stephen to Appleby-in-Westmorland, consists of lowland topography and lies between 175 and 123 m AOD. Devensian ice flows left behind a gentle, undulating topography in the basin bottom along with an extensive field of drumlins (Allen *et al.*, 2010). The river to the town of Appleby drains a catchment of 334 km^2 and will be the focus for this study.

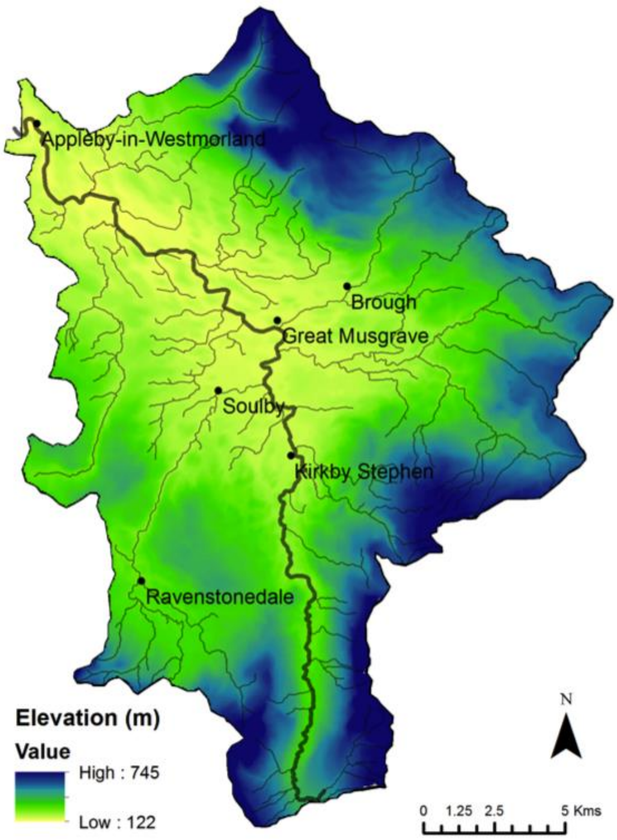


Figure 3.2: Digital Elevation Model of the upper Eden catchment (50 m resolution) (source: EDINA Digimap Ordnance Survey Service).

3.2.3 Geology

The Eden valley is aligned approximately southeast-northwest (in the direction of flow), is 56 km long and varies between 5 and 15 km in width. The valley sits between two upland areas: the Pennines to the east and the Lakeland Fells to the west, separated by the Pennine Fault. The valley is an *inter-montane basin* in which a thick Permo-Triassic sedimentary sequence accumulated from the late Carboniferous period. This sequence includes two important sandstone hydrostratigraphic units: the Permian Penrith Sandstone and the Triassic St. Bees Sandstone (Younger and Milne, 1997). Both of these are significant aquifers; the Penrith Sandstone formation supplies large quantities of groundwater for public supply in the northern part of the catchment (Butcher *et al.*, 2006) and also provides the baseflow component to the River Eden and its tributaries. Eden Shales separate the Penrith Sandstone from the St. Bees sandstone and Carboniferous Limestone and Millstone Grit surround, and to a large extent, underlie the Eden Valley. Millstone grit (mudstones, sandstones, thin limestones and thin coal seams) overlies the Lower Carboniferous limestones and form the Cumbrian Fells to the east. The solid geology of the upper Eden catchment is depicted in Figure 3.3.

Superficial deposits (Figure 3.4) cover the valley floor with a thickness up to 30 metres. The southern part of the valley is dominated by glacial till with small areas of post-glacial Holocene fluvial deposits; forming terraces adjacent to the River Eden. The till, comprising chaotically interbedded clays, silts, sands and gravels (Younger and Milne, 1997), forms hummocky moraines and drumlins, which may act locally to prevent recharge to the underlying aquifers (Fragalà and Parkin, 2010). Conversely, high permeability fluvial deposits will promote recharge where they directly overlay deeper aquifers (Butcher *et al.*, 2006).

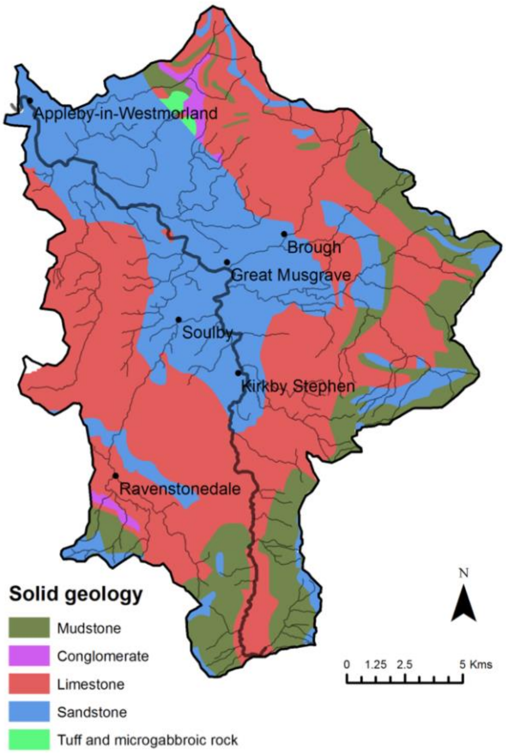


Figure 3.3: Catchment **s**olid geology map.

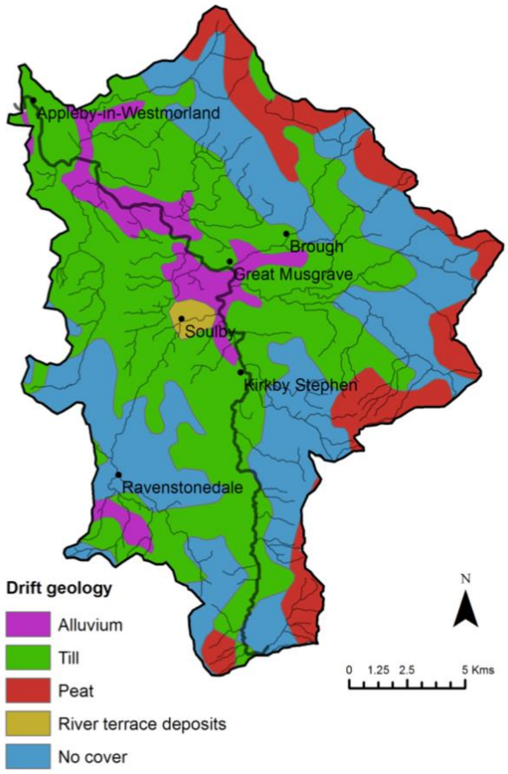
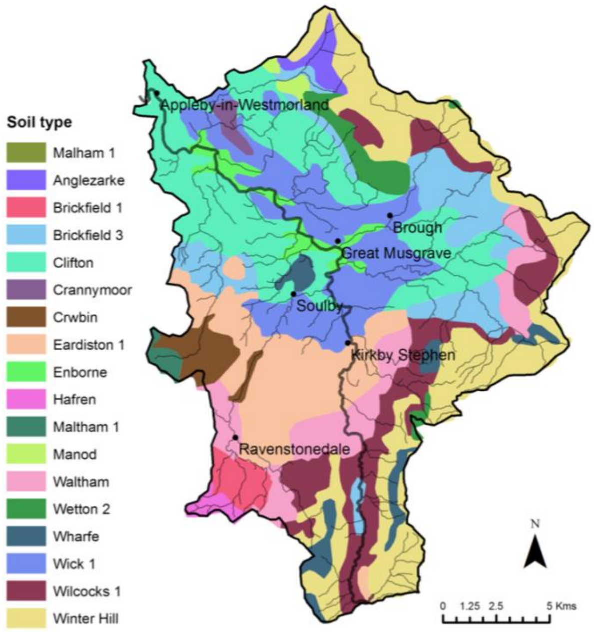


Figure 3.4: Catchment drift geology map.

3.2.4 Soils

Soil Association and Hydrology of Soil Type (HOST) maps are depicted in Figure 3.5 and 3.6, respectively. The uplands are dominated by Winter Hill Association blanket peat and peaty gley soils. Due to almost permanent waterlogging (Wetness Class VI) the soils do not absorb excess rainwater meaning runoff is rapid. On the valley sides and upland valley bottoms are Wilcocks 1 Association: slowly permeable, wet, and very acid upland soils with a peaty surface. In the lowlands, from Kirkby Stephen to Appleby, there are two main soil types that are roughly divided by the River Eden. To the northeast of the river lies Wick 1 Association: coarse loamy soils formed in sands and gravels. It has a HOST classification of 5, meaning that it is free draining (Wetness Class I). To the south-west of the river lies Clifton Association: mainly slowly permeable fine textured soils formed in glacial till. This Association has a HOST classification of 24, meaning it is a non-calcareous mineral gley with a wetness class III or IV. Perforating these dominant soils is a small area of Wharfe Association to the southwest of Great Musgrave: a free-draining alluvial soil with a wetness class of I, and small stretches of Enborne Association: a loamy and clayey, poorly drained alluvial soil with naturally high groundwater (wetness class IV) that follows the river channel. In a transitional zone to the south and southwest of Kirkby Stephen, between the uplands and lowlands, lies Eardiston 1 Association. This is a reddish, well-drained, coarse loamy and fine silty brown earth with a wetness class of I.



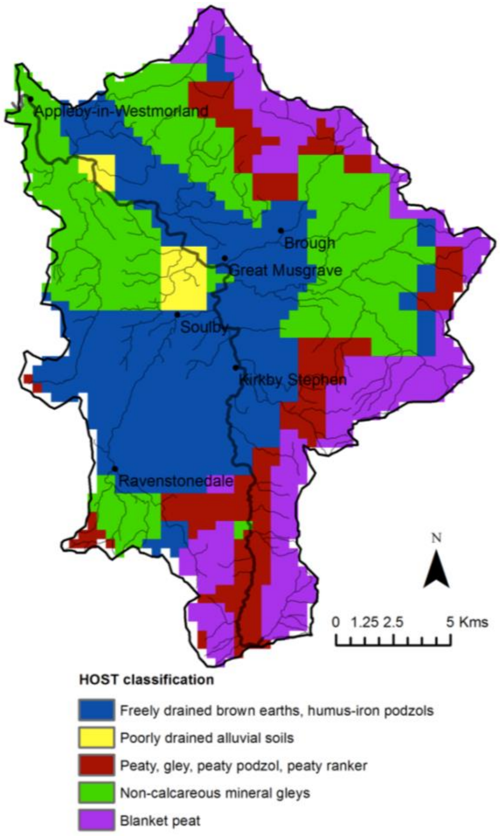


Figure 3.5: Catchment soil association map.

Figure 3.6: Catchment HOST classification map.

3.2.5 Land cover and land use

The upper Eden is largely rural, as depicted in Figure 3.7 - the Centre of Ecology and Hydrology (CEH) Land Cover Map 2000 (LCM2000) for the upper Eden catchment. Permanent pasture and moorland account for 48% and 40% of the land cover respectively, while a small amount of cultivated land (7.5%) occupies the lowlands and small pockets of deciduous woodland (2.5%) exist usually in steep-sided tributary valleys. Farming is the main land use in the catchment. Stocking densities of cattle and sheep are higher in the lowlands due to the better quality of grass (Wilkinson, 2009). The main urban areas in the catchment are Appleby-in-Westmorland (population circa. 2500) and Kirkby Stephen (population circa. 1900), although they occupy only 0.4% of the total catchment area. A proportion of the northern part of the catchment (mainly moorland) is an infantry training ground owned by the Ministry of Defence, thus has restricted access. The upland areas in the south and east are dominated by natural vegetation such as moorland and marsh grass as well as peat bog and a small amount of limestone pavement. Low-density sheep grazing is the principal agricultural activity in the moorland area. Grouse shooting takes place between August and December.

Figure 3.7: Land use map of the upper Eden catchment (source: Centre for Ecology and Hydrology, 2000).

3.2.6 Climate

The Eden catchment is located in the northwest of England, the wettest region in the country due to frontal systems bringing moist air from the Atlantic Ocean. Average annual rainfall in the valley is approximately 1000 mm yr^{-1} with in excess of 2000 mm yr^{-1} on higher ground (Wilkinson, 2009). This large variation is caused mainly by differences in elevation, but may also be influenced by rain shadow effects, with higher rainfall occurring on the leeward side of the Lakeland Fells. Temperature in the winter months can fall to -15°C and in the summer months can reach 30°C. There are 0-5 snow days on average in the lowlands but up to 15-30 days in the uplands (>300 m).

3.2.7 River Characteristics

The River Eden is predominantly a gravel bed river, comprising riffle and pool sequences with bed material ranging from fine sand to coarse gravel. The river flows northwards from the

Carboniferous Limestone fells of Mallerstang Common down a steep sided valley to Kirkby Stephen (circa. 20 km), where the catchment area is 69 km^2. Between Kirkby Stephen and Great Musgrave three monitored sub-catchments enter the Eden. The Eden reaches Great Musgrave Bridge at 26 km where the catchment area is 233 km^2. Around this point the river enters its lowland zone with a much wider floodplain present where agricultural activities become more intensive with increased stocking densities, more improved pasture, larger farm steadings and some arable land use. The river also becomes much wider with large meanders as it flows towards Appleby where only small tributaries enter the Eden, which are not monitored in this study. The Appleby monitoring site defines the outfall of the study catchment (334 km^2).

Storm hydrograph response in the Eden is flashy, owing particularly to the steep slopes and thin and/or poorly drained soils in the uplands (Mills, 2009), combined with relatively frequent intensive rainfall events (Wilkinson, 2009). The Base Flow Index (BFI) for the Eden at Kirkby Stephen is 0.26. The mean annual flow at Kirkby Stephen is 2.59 $m^3\ s^{-1}$ and the catchment has a SPRHOST (standard percentage runoff – based on HOST classification) value of 46%. Other selected hydrometric statistics are listed in Table 3.1. The annual hydrograph for the River Eden at Kirkby Stephen from 2010 is shown in Figure 3.8, the relatively low flow period between April and July being particularly notable.

Table 3.1: Hydrometric data for the Kirkby Stephen catchment. BFI = Base Flow Index; SAAR = Seasonally Adjusted Annual Rate (source: National River Flow Archive (Centre for Ecology and Hydrology, 2012)).

Descriptor	Value	Period of record
Mean annual flow	2.59 $m^3\ s^{-1}$	1971-2010
Q95	0.166 $m^3\ s^{-1}$	1971-2010
Q50	1.015 $m^3\ s^{-1}$	1971-2010
Q10	6.52 $m^3\ s^{-1}$	1971-2010
BFI	0.26	1961-1990
SAAR	1483 mm	1961-1990
SPRHOST	46 %	1961-1990

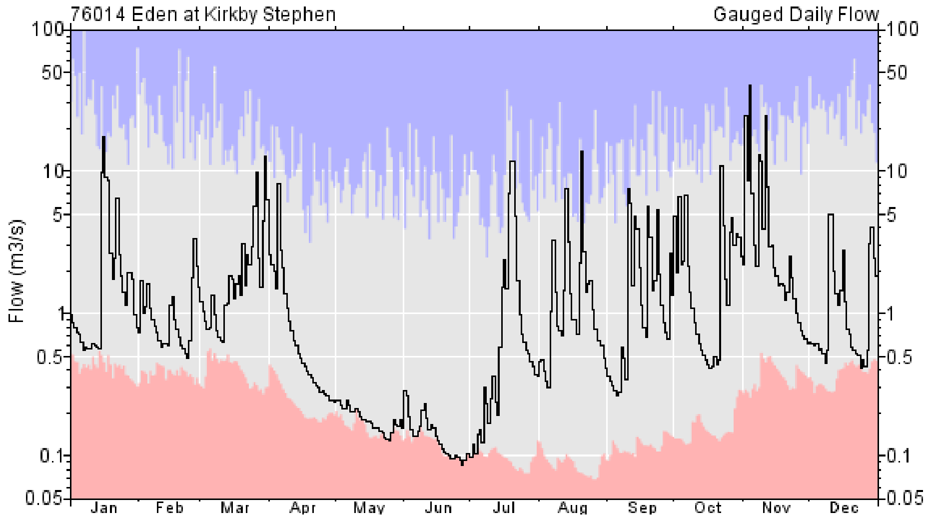


Figure 3.8: Hydrograph of gauged daily flows on the River Eden at Kirkby Stephen during 2010, with maximum and minimum daily mean flows from 1971 to 2010 (source: Centre for Ecology and Hydrology, 2012).

3.3 Experimental design, network and monitoring

The aim of the first section of this thesis is to investigate sediment and nutrient regimes across the upper Eden catchment and how they are affected by spatial and temporal scale. To allow this, data (both hydrometeorological and sediment/nutrient) need to be collected across a range of spatial scales, physical catchment characteristics and land uses. Thus, an experimental design that is fit for purpose is vital. The selected monitoring regime incorporates a multi-scale, nested approach where stratified water sampling is used to provide sediment/nutrient fluxes representative of a wide range of flow conditions and seasonal patterns.

3.3.1 Monitoring locations

Thirteen sub-catchments were selected to monitor sediment and nutrient concentrations and yields within the upper Eden catchment (Figure 3.9) with the experimental design strongly informed by the original CHASM programme and associated follow-up projects. This study uses two nested catchments - one along the main River Eden (sites 1-4, Figure 3.9) and the other that incorporates Scandal Beck (sites 5-7) and River Eden sites 3 and 4. A further six sub-catchments outside the nested system (sites 8-13, Figure 3.9) were also selected. The sub-catchments cover the full range of spatial scales set out under the CHASM project, ranging in area from 1.1 to 334 km^2. The purpose of the nested system is to investigate how SS, P and NO_3 yield varied with increasing catchment area along a continuous stretch of river. The six non-nested catchments allowed the effects of spatial variability in catchment characteristics (e.g., elevation – upland and lowland, land use, geology, etc.) on yields to be taken into account. Figure 3.10 provides a simplified schematic of the monitored river network.

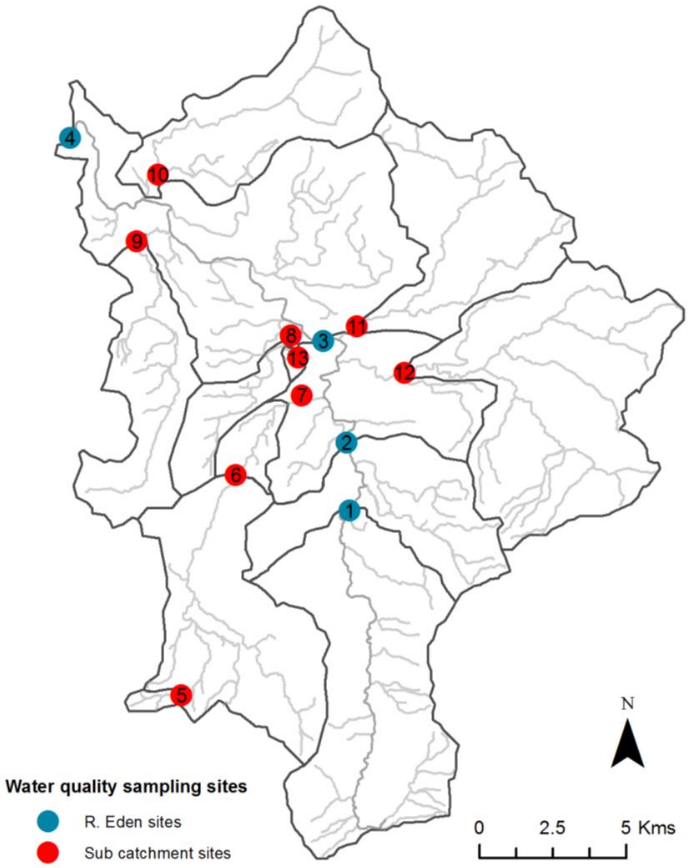


Figure 3.9: Location of water quality monitoring sites in the upper Eden catchment (see Table 3.2 for site information).

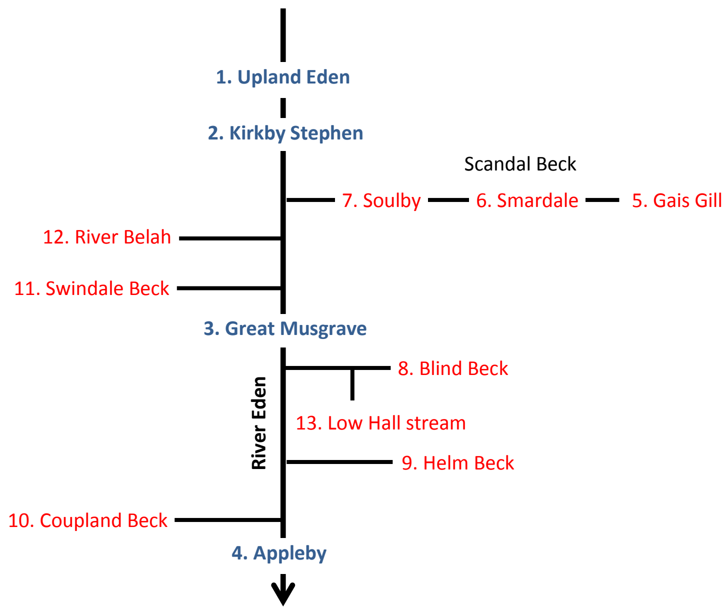


Figure 3.10. Schematic of upper Eden catchment monitored river network (not to scale). Numbers and colours correspond to Figure 3.9.

Table 3.2 provides summary information for the monitoring sites depicted in Figure 3.9. Site 1 was chosen to represent the River Eden that exclusively drains upland areas with very little urban settlement. It also marks the start of the first nested catchments along the main River Eden. Site 2 is downstream of the town of Kirkby Stephen and coincides with an EA river gauging station (installed in 1971). The catchment is high relief and drains Carboniferous Limestone, which forms most of the watershed, with the middle reaches underlain by Permian Sandstone. There is a variable boulder clay cover in the lowlands. The uplands are dominated by hill peat and moorland land use. Some livestock pasture can be found in the valley bottom.

Site 3 at Great Musgrave is also an EA river gauging station, installed more recently in 2000. Site 4 is just downstream of the town of Appleby and marks the outfall of the 344 km^2 study catchment. Site 5 at Gais Gill is the first site on the nested sampling system on Scandal Beck, which rises on the Howgill Fells in the south of the catchment. It was selected in the CHASM project as a micro (~1 km^2) catchment that represents the uplands. Scandal Beck flows through Ravenstonedale (population circa. 600), a narrow gorge at Smardale where site 6 is located, and finally Soulby (population circa. 200), where Site 7 is located downstream of the village. Sites 8-13 are outside the nested system. Site 8 is at the outfall of Blind Beck, a lowland mini (~10 km^2) catchment dominated by improved grassland and pastoral land use. Site 9 is on Helm Beck, a lowland sub catchment with relatively high levels of agricultural activity, including dairy farming.

Site 10 is on Coupland Beck, which drains the Hilton and Murton Fells to the north. There are two small hamlets, Hilton and Murton, located in the catchment and the land use is low-intensity sheep grazing. This catchment was previously unmonitored by the CHASM project but it was felt that its relatively large 27.5 km^2 catchment area was too large to omit from this study. Swindale Beck drains the north east of the catchment, which contains upland peat bogs and moorland on the Pennine plateaux, before flowing through the village of Brough (population circa. 700). There is some improved pasture in the lower reaches upstream of the monitoring point at site 11.

Site 12 is on the River Belah, which also contains uplands on the Pennine plateaux. The catchment hosts only low-density sheep grazing. Sites 11 and 12 were not part of the original CHASM network but were added in 2007 by Mills (2009) as part of a sediment yield investigation. Site 13 is at the outfall of another micro (~1 km^2) catchment, selected by the CHASM project to be a comparison to site 5 at Gais Gill. The Low Hall stream, which flows into Blind Beck just upstream of site 8, represents the lowlands with a relatively low relief and increased agricultural activity; a large dairy operation exists in the catchment.

Table 3.2: Sub catchment characteristics calculated using ArcMAP 10 GIS software.

No. on map (Figure 3.9)	Name	Catchment area (km^2)	Mean elevation (m)	Maximum elevation (m)
1	Upland Eden	48	442	707
2	Eden at Kirkby Stephen	69	385	707
3	Eden at Great Musgrave	223	345	707
4	Eden at Appleby	334	307	745
5	Gais Gill	1.1	470	602
6	Scandal Beck at Smardale	37	331	707
7	Scandal Beck at Soulby	40	316	602
8	Blind Beck	9	220	376
9	Helm Beck	18	252	374
10	Coupland Beck	28	434	745
11	Swindale Beck	32	410	650
12	River Belah	53	377	660
13	Low Hall stream	1.25	154	162

3.3.2 Hydrometeorological data collection

Hydrology plays an essential role in the transfer of sediment and nutrients from land to water and also their movement through the fluvial system. Thus, in order to characterise this movement accurately it is vital to have records of the main inputs to and outputs from the catchment; principally precipitation and discharge.

The author was responsible (unless otherwise indicated) for maintaining and downloading the CHASM hydrometric network, as well as archiving the data, during the study period (November 2009 to December 2011, inclusive).

Water samples were collected in order to determine SS, P and NO_3 concentrations at the Newcastle University laboratory, also carried out by the author.

3.3.2.1 Flow

Discharge was not measured directly at any site in the Eden. Instead, stage was recorded at 15 minute resolution, which was converted to flow by way of an existing stage-discharge relationship, or rating curve. Figure 3.11*a* depicts the locations of the stage gauges in the upper Eden catchment. Eight sites were equipped with CHASM owned stage recorders, which included: three OTT Thalimedes float and counterweight shaft encoders (referred to as '*Thalimedes*' for brevity), five pressure transducers (referred to as 'divers' – produced by Van Essen Instruments), and one Horizontal Acoustic Doppler Current Profiler (H-ADCP) at Appleby. Barometric pressure was measured using barometers at various locations in the catchment. These data are needed to 'correct' those recorded by the divers to ensure the latter showed only changes in water level, not atmospheric pressure. Stage gauges (and barometers) were downloaded every two months on average along with manual stage measurements, which were used to check the data from the corresponding instrument (after 'correction'); thus ensuring a more accurate stage time series.

The EA monitor river stage at several fixed structure sites along the River Eden; stage data is converted by the EA to 15 minute flow records by way of a rating curve. Data from three EA stations were used in this study; from upstream to downstream the sites are: Kirkby Stephen, Great Musgrave and Temple Sowerby. Table 3.3 contains a summary of the upper Eden flow monitoring instrumentation. Further information on the EA gauging stations can be found at http://www.environment-agency.gov.uk/hiflows/91727.aspx. Flow data from Temple Sowerby were not used to calculate sediment and nutrient loads as the site is outside the study catchment; however, the data provided a means by which to compare runoff values to help improve calculated catchment water balances.

Rating curves for the Eden sites were established during the CHASM project (2003-2005) and updated by Mills (2009) in 2007-2008. In order to construct the stage-discharge relationships discharge was gauged at a range of flows using an impeller flow meter operated whilst wading. Where flow was too deep to allow safe wading, a float-mounted ADCP was used. It was decided that updating the rating curves was beyond the remit of this study due to time constraints. Thus, sites with fewer gaugings and less reliable rating curves should be treated with a greater degree of uncertainty. The only site that was flow gauged during the study was Blind Beck (site 8); the highest gauged stage as a percentage of the highest recorded stage was increased from 71 to 84%. Rating curve coefficients can be found in Appendix A1.

Three sites were not instrumented with stage gauges: Scandal Beck at Soulby, Coupland Beck and the upper Eden. Discharges for the upland Eden and Scandal Beck at Soulby were down- and up-scaled, respectively, from the Eden at Kirkby Stephen and Scandal Beck at Smardale, respectively, using catchment area. Discharge at Coupland Beck was estimated by down-scaling the Kirkby Stephen discharge record (as above) followed by the application of a multiplier value, based on the SPR HOST values for each site, to take into account soil and runoff property differences between the two catchments. More details of this can be found in Chapter 4.2.3.

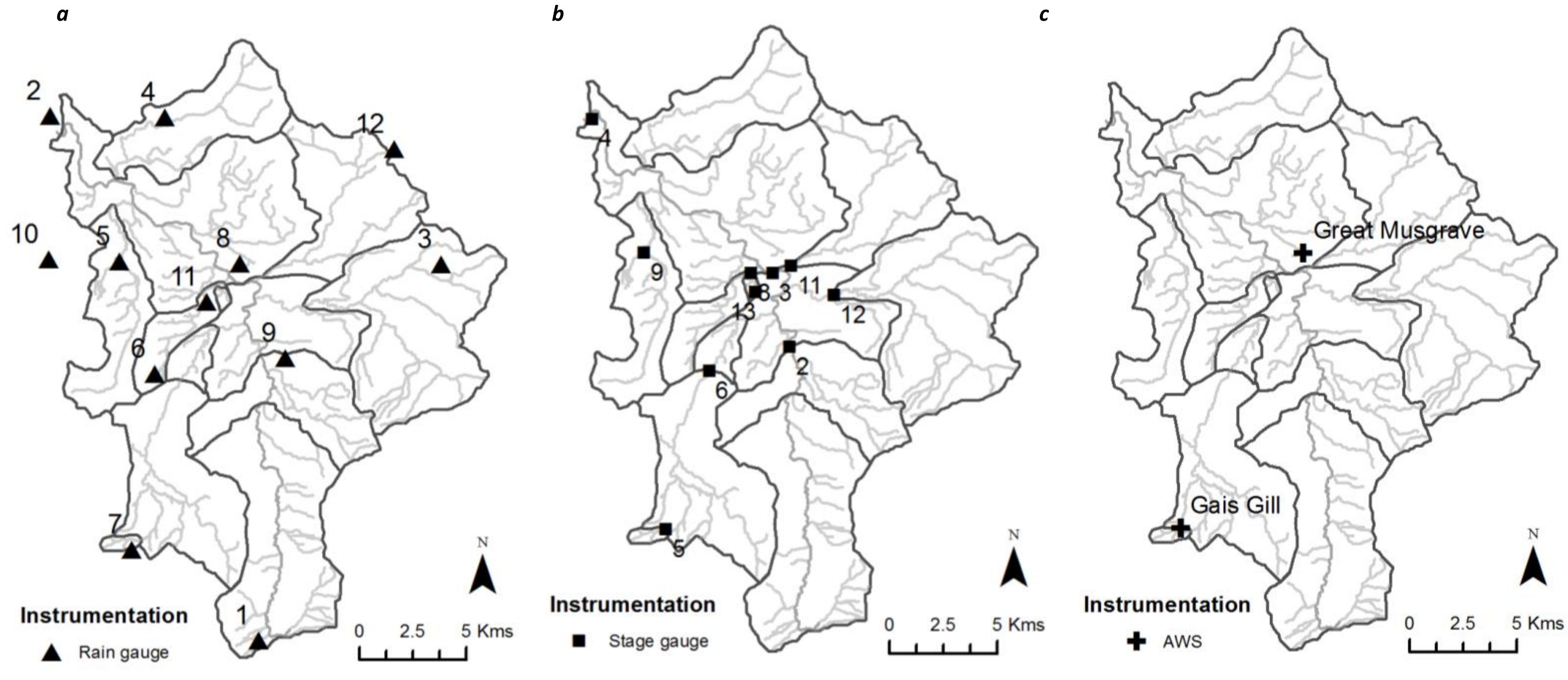


Figure 3.11: Hydrometric instrumentation maps of the upper Eden catchment: *a* = stage gauges; *b* = rain gauges; *c* = AWSs.

Table 3.3: Details of flow monitoring sites in the upper Eden catchment.

No.	Site	Catchment area (km^2)	Length of record (years)	Site manager	Instruments	Highest gauged stage as % of max. recorded	R2 value of stage-discharge relationship
1	Upland Eden	48	-	CHASM	None	Ungauged	-
2	Eden at Kirkby Stephen	69	29	EA	Float and counterweight	95	N/A
3	Eden at Great Musgrave	223	11	EA	Float and counterweight	95	N/A
4	Eden at Appleby	334	7	CHASM	Pressure transducer, H-ADCP	58	0.92
5	Gais Gill	1.1	8	CHASM	Pressure transducer	61	0.94
6	Scandal Beck at Smardale	37	9	CHASM	Thalimedes	42	0.99
7	Scandal Beck at Soulby	40	-	CHASM	None	Ungauged	-
8	Blind Beck	9	8	CHASM	Thalimedes	84	0.96
9	Helm Beck	18	9	CHASM	Thalimedes	51	0.96
10	Coupland Beck	28	-	CHASM	None	Ungauged	-
11	Swindale Beck	32	4	CHASM	Pressure transducer	29	0.74
12	River Belah	53	4	CHASM	Pressure transducer	19	0.52
13	Low Hall stream	1.25	5	CHASM*	Pressure transducer	57	0.93 and 0.76**

*maintained by Ockenden (2010) **two part rating curve (No. refers to the location in Figure 3.11 A).

3.3.2.2 Precipitation

Rainfall was monitored using a network of twelve rain gauges (Figure 3.11*b*): five CHASM owned ARG100 tipping bucket gauges and seven EA operated gauges; Table 3.4 provides details of the rain gauges. The original CHASM network consisted of 25 rain gauges (including EA gauges); this was necessary, as spatial variability in rainfall was being investigated as part of the project (Wilkinson, 2009). However, since then many gauges have been damaged or had logger problems, and subsequently been removed. It was also deemed unnecessary in this study to conserve such a large number of gauges as maintaining them would be very time consuming, especially those in hard to reach, distant locations. It was felt that the reduced network was sufficient to give a reliable estimation of areal rainfall across the catchment in order to calculate catchment water balances.

Table 3.4: Details of rain gauges in the upper Eden catchment used in this study (No. refers to the location in Figure 3.11*b*).

No.	Rain gauge	Type	Resolution	Site manager	Elevation	OS grid reference
1	Aisgill	TBR	15 minute	EA	360	NY 778963
2	Appleby Castle	TBR	Daily	EA	148	NY 684200
3	Barras	TBR	15 minute	EA	343	NY 845121
4	Brackenber	TBR	15 minute	EA	176	NY 722195
5	Brakes Hall	TBR	15 minute	CHASM	175	NY 701139
6	Crosby Garrett	TBR	Daily	EA	198	NY 728097
7	Gais Gill	TBR	15 minute	CHASM	390	NY 714009
8	Great Musgrave	TBR	15 minute	CHASM	155	NY 757138
9	Kirkby Stephen	TBR	Daily	EA	183	NY 772078
10	Scalebeck	TBR	15 minute	EA	183	NY 673144
11	Sykeside	TBR	15 minute	CHASM	180	NY 747122
12	West Clove Hill	TBR	15 minute	CHASM	510	NY 835194

Thiessen polygons

Areal rainfall for each sub-catchment was interpolated from spot measurements using the Thiessen Polygon method in ArcGIS. This method assigns weights to each gauge station in proportion to the catchment area that is closest to that gauge. Figure 3.12 depicts the Thiessen polygons created in ArcGIS for the Appleby catchment, i.e., the entire study area.

Figure 3.12: Thiessen polygon map of the upper Eden catchment.

Wilkinson (2009) highlighted a potential issue with the method when applied to the Kirkby Stephen sub-catchment. As the catchment contains only two rain gauges the Thiessen polygon has a bias towards the Kirkby Stephen raingauge with that polygon covering 57% of the catchment and the Aisgill polygon accounting for only 43% (Figure 3.13). As the Kirkby Stephen rain gauge is at a relatively low elevation, and therefore with lower storm rainfall totals, this would reduce the actual areal average rainfall totals. A method was applied that used an elevation-weighted SAAR map (produced by Wilkinson, 2009) to synthesise rainfall for four additional rain gauges that used to operate in the catchment: Angerholme Potts, Lunds Fell, Nateby Common and Outhgill.

A multiplier value was derived from the map for each additional gauge, which was proportional to the increase/decrease in elevation with reference to either the Kirkby Stephen or Aisgill gauge (whichever was closest); this multiplier value was then applied to the corresponding 'actual' dataset to synthesise a new one. A new Thiessen polygon was then created using all six gauges (Figure 3.14), which meant that the Kirkby Stephen raingauge accounted for only 31% of the catchment rainfall.

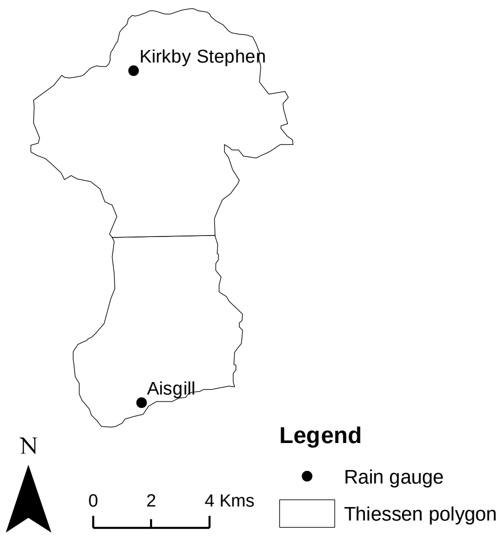


Figure 3.13: Kirkby Stephen Thiessen polygon map.

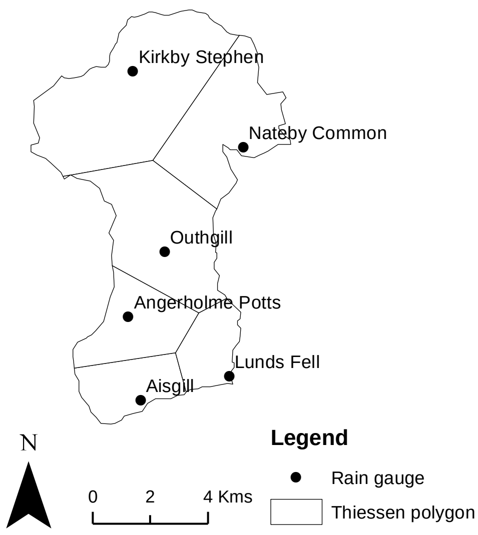


Figure 3.14: Kirkby Stephen Thiessen polygon map using synthetic rain gauges.

3.3.2.3 Evapotranspiration

Two Environmental Measurement (EM)-Ltd Automatic Weather Stations (AWSs) are located in the upper Eden catchment; one in the uplands at Gais Gill (440 m AOD), and the other in the lowlands at Great Musgrave (148 m AOD) (Figure 3.11 *c*). The AWS measure maximum and average wind speed, wind direction, air temperature, relative humidity and net radiation; these parameters are used to calculate evapotranspiration. The AWS also have a raingauge attached to them and form part of the raingauge network. Each parameter is averaged and stored every 15 minutes and these data are stored on a Campbell's CR10X logger, which can store up to 50 days' worth of data. Fifteen-minute potential evapotranspiration (PE) was calculated using a modified version of the Penman-Monteith equation (parameterised for the upper Eden catchment by Wilkinson (2009)) using air temperature, relative humidity and net radiation data collected by the Great Musgrave AWS.

3.3.2.4 Data archiving

Hydrolog 4, a database manufactured by *HydroLogic* was used to archive all stage, rainfall and meteorological data. This software uses a GIS interface to store and access data, as well as facilitating some data analysis. Stage-discharge relationship coefficients are stored in the program therefore allowing it to calculate flow time series. *Diver Office*, software specific to the diver instruments, was used to correct for barometric pressure and also put the information in the database.

3.3.3 Water sample collection

Water samples were collected from the thirteen sites (Figure 3.9) using a synchronous grab sample method. Although not taken at exactly the same time, significant effort was made to collect samples from all sites in the shortest time period possible in order to provide a multi-scale snap shot of the sediment and nutrient concentrations while the catchment was under the same, or similar, flow conditions. Where possible the sample was taken from the centre of the watercourse using a 1 litre acid washed polyethylene bottle. When wading was not possible due to fast/deep river flow samples were collected from the bank using a bottle attached to a pole. Acid washing of the bottles was necessary to ensure no phosphate residue from detergent was left in the bottle after washing. All sample bottles were stored in a cool, dark plastic carrier to be returned to the laboratory where they were stored below 4 degrees centigrade.

Monitoring took place between November 2009 and December 2011 (inclusive), across as wide a range of flow conditions as possible. On average, samples were collected at fortnightly intervals during 'residual' flow periods. Additional targeted sampling was made during high discharges in order to produce representative pollutant rating curves and allow accurate estimation of yields (e.g., Horowitz (2003) – this issue is discussed in more detail in Chapter 4.3.1 and 4.4.1). This was achieved by regularly monitoring the weather forecast. In total, grab samples were collected on 49 separate occasions.

In addition to the grab sample campaign, Teledyne ISCO 3700 automatic water samplers were deployed in the Blind Beck sub-catchment, at Sykeside Farm. These were triggered during high flows and collected up to 24 x 1-litre water samples. More detail on this can be found in Chapter 5.2.4. Samples were treated as above and subjected to the same laboratory analyses.

3.3.4 Laboratory analysis

All laboratory analysis was carried out at Newcastle University by the author. To reduce the opportunity for changes in nutrient concentrations due to chemical transformations, reactive P concentrations were determined within 24 hours of collection. Concentrations of NO_3 and other P fractions (requiring digestion) were determined within 48 hours whenever possible. On five occasions NO_3 determination was not possible due to equipment failure (e.g., Dionex 100 Ion Chromatograph – described below). Suspended sediment analysis could take place at any time following collection but was usually conducted alongside nutrient analysis.

A QC/QA program usually incorporates the collection and analysis of blank, duplicate, replicate and/or spiked samples, reference materials to ensure the integrity of the analyses, and regular inspection of the equipment to ensure it is operating properly. Due to the time-consuming nature of the laboratory analysis and associated high costs, duplicate analysis could not be conducted in this study. However, certain measures were taken to mitigate for any inaccuracies and thus ensure the precision and ultimately the representativeness of the data. Nitrate analysis was carried out on a Dionex 100 ion chromatography machine that was calibrated and maintained to a very high standard by the Newcastle University laboratory technicians. Every sample run was carried out with deionised water blank and standard solutions. Phosphorus was also determined alongside a deionised water blank and five standard solutions of known P concentration, which were made up every time analysis was performed.

3.3.4.1 Suspended Sediment

Suspended sediment was determined using standard gravimetric filter analysis. Filter papers (Whatman GF/C 70 mm) were pre-dried in the oven at 105°C for 20 minutes before being weighed. A 200 ml (whenever possible) aliquot was passed through the filter before the filters were dried in oven at 105 °C for 2 hours. Finally, the filters were cooled in a desiccator, and then re-weighed. The suspended sediment concentration is obtained using:

$$c_{ss} = \frac{w_B - w_A}{V_f} \qquad \textbf{(Equation 3.1)}$$

where c_{ss} is the suspended sediment concentration (mg l^{-1}), w_B is the weight of filter paper before filtration (mg), w_A is the weight of filter paper after filtration (mg), and V_f is the volume of sample filtered (l).

3.3.4.2 Phosphorus

Phosphorus concentration fractions measured in this study include: SRP - a measure of the inorganic monomeric P and easily-hydrolysable P in the less than 0.45µm fraction; TSP - the combination of SRP and SUP, released by potassium peroxodisulphate digestion on a filtered sample; and TP - the fraction released by potassium peroxodisulphate digestion (as described by Murphy and Riley (1962)) on an unfiltered sample.

Phosphorus concentration was determined colourimetrically by UV spectrometry (at a wavelength of 880 nm) on all P fractions according to British Standard methods (BS 6068: 2.28: 1986, ISO 6867/1 1986). Total P was determined on unfiltered samples and SRP and TSP after filtering through a 0.45µm cellulose acetate membrane filter.

3.3.4.3 Nitrate

Nitrate concentrations were determined using a Dionex 100 Ion Chromatograph. Prior to analysis samples were filtered through 0.45µm cellulose acetate membrane filters; dilution was unnecessary due to the relatively low NO_3 concentrations encountered in this study.

3.4 Summary

This chapter has introduced the upper Eden catchment and emphasised its importance as a highly valued ecological area. The catchment comprises a variety of land uses, which are linked to its topographic, geological, pedological and climatic characteristics. The experimental design has been selected to best complement the aim of the next chapter, which is to characterise the sediment and nutrient regimes of the upper Eden catchment across a range of spatial and temporal scales.

Thirteen sub-catchments have been selected that cover a range of land uses, which will allow examination of the effects of varying physical attributes on sediment/nutrient dynamics. To enable the calculation of sediment/nutrient loads and yields that are representative of actual in-situ conditions, and to understand the patterns, processes and magnitudes observed, a synchronous, multi-scale grab sampling campaign is used. Reliable precipitation, discharge and evapotranspiration data are also vital and details of their measurement have been described. In the next chapter the methods used for the quantification of yields will be described and the results obtained using these methods presented.

4. Characterising the nutrient and sediment regimes of the upper Eden catchment

4.1 Introduction

The main aim of this chapter is to characterise the SS, P and NO_3 regime of the upper Eden catchment in 2010 and 2011, and to identify the important factors controlling water quality variability. This includes the examination of spatio-temporal variability in nutrient and sediment concentrations, and in total and specific yields. Export coefficients are derived for a number of sub-catchments within the River Eden and used to investigate the effect of spatial and temporal scale, as well as sub-catchment specifics such as land use. Sources of uncertainty in the measurement and calculation of nutrient/sediment budgets are discussed.

4.2 Methodology

4.2.1 Nutrient and sediment concentration characterisation

Water quality monitoring to assess non-point source pollution is particularly difficult because there is a large degree of natural variability to account for, including spatial and temporal variability in weather, hydrological conditions, and land use and management practice. The best way to account for this variability is to monitor pollutant concentrations on a continuous (or near-continuous) basis. However, this requires highly specialised and (often) expensive on-site automatic sampling and analysing equipment, which is normally feasible only in relatively small-scale, funded research projects (Johnes, 2007; Rozemeijer *et al.*, 2010; Cassidy and Jordan, 2011).

Traditionally, water quality monitoring has relied heavily upon the collection of grab samples, including the EA's GQA scheme, which provide 'snapshots' of concentrations. However, to capture the dynamic behaviour of surface water quality and fully represent the 'actual' pollutant flux in a river over a longer time period (e.g., a month or a year) a large number of samples are needed (Webb *et al.*, 1997). Increasing sampling frequency is associated with increased field sampling, sample transport and laboratory procedures, which are laborious and expensive. As a consequence, surface water quality monitoring will continue to rely predominantly on (relatively) low-frequency grab sampling data (Rozemeijer *et al.*, 2010).

Marking a new era of environmental measurements, the UK Demonstration Test Catchments (DTC) [http://www.demonstratingcatchmentmanagement.net] and Irish Agricultural Catchments Programme (IACP) [http://www.teagasc.ie/agcatchments] are two large-scale, government funded projects, which have invested in state of the art automatic monitoring equipment capable of recording a multitude of water quality parameters on a near-continuous basis (Owen *et al.*, 2012). While this level of investment is beyond the remit of the majority of monitoring programmes, such as the EA's GQA scheme, one of the goals of the DTC and IACP is to develop and determine surrogates for pollution such as turbidity (e.g., suspended sediment – Gray and Gartner (2009) and Minella *et al.* (2007), and total P – Nairn and Mitsch (1999)), which would be more economically viable for widespread use.

4.2.1.1 Method selection and sources of uncertainty

For this study, monitoring at a single site (the stream outlet) was believed to be adequate to broadly characterise the water quality of a particular catchment. With this approach, the effect of the point and non-point source pollution processes occurring throughout the catchment is integrated. A strategy which takes samples at regular but infrequent intervals (e.g. weekly or fortnightly) is unlikely to sample over a full range of discharge and sediment and nutrient concentration values and is thence liable to under- or over-predict average concentrations and loads (Walling *et al.*, 1992; Phillips *et al.*, 1999; Horowitz, 2003; Johnes, 2007; Cassidy and Jordan, 2011). In order to address this, an effort was made to collect samples during high flow events.

During a sample collection campaign, a significant effort was made to visit all the monitoring sites within a few hours (particularly during or shortly after a storm event) to provide a multi-scale, synchronous snapshot of sediment and nutrient concentrations across the entire study catchment. This assumes some behavioral similarity between sites and reduces random errors caused by sampling on different days

Taking a point measurement (in this case the collection of a grab sample but also relevant to automatic water samplers that draw water from a fixed point in the river) is making an assumption that it is representative of the entire river cross-section at that instantaneous moment in time. However, particularly during high flows, vertical and horizontal velocity gradients will exist within the cross-section and changes in sediment/nutrient concentrations are directly related to these velocity variations (Ingram *et al.*, 1991). It is suggested that multiple depth-integrated samples should be taken at intervals across the river cross-section to take account of this. However, this is often not practical due to time/resource constraints, the size of the river, or the depth of flow (especially during high-flow events). A study carried out in the Yorkshire Ouse catchment showed very little SS concentration variation in samples taken at 0.5 m vertical intervals.

However, it did reveal that depth-integrated samples taken across river sections exhibit a systematic increase in concentration towards the centre of the channel (Evans *et al.*, 1997; Wass and Leeks, 1999).

The use of stage-discharge rating curves, especially when extrapolated above measured flows, is a particular problem as high discharges are often related to the largest transfers of sediment/nutrients. Even if a site has a reliable rating curve (gauged to a high percentage of the maximum stage, i.e., bank full) it cannot account for out of bank flows. Summer weed growth and physical changes can also affect a river cross-section, which can subsequently alter the stage-discharge relationship.

4.2.2 Quantifying sediment and nutrient yield

4.2.2.1 Method selection and sources of uncertainty

Knowledge of a catchment's sediment/nutrient yield (also referred to as export, load or flux) is important for detecting trends in annual transport rates and assessing the effects of measures taken to reduce their export. However, calculation of an accurate yield value from non-continuous data is problematic due to the many sources of uncertainty involved. This uncertainty derives from both the sampling strategy (continuous, regular or stratified – as previously described) and the method of yield calculation employed (interpolation or regression/extrapolation – discussed below).

A number of studies on the reliability of mass load estimates indicate that no individual estimation method is superior and that poor accuracy can be obtained using individual estimation methods on a specific stream for a given determinand and year. Refer to Dolan *et al.* (1981), Kronvang and Bruhn (1996) and Johnes (2007) for a comprehensive explanation of the available methods.

In its most basic form, an *interpolation* method takes the known (measured) determinand concentration and multiplies it by the corresponding instantaneous flow, or the mean of the instantaneous flow observations for a given time interval. This is then integrated over a specific time period (e.g. a year). A correction factor can also be applied (e.g., *Beales' Ratio Estimator*). This can provide reasonable TP yield estimates for streams with a high BFI but large errors for low BFI, 'flashy' streams. Thus, catchments with a lower BFI tend to return a wider range of load estimates. As the upper Eden catchment has a medium (in lowland catchments) to low (in upland catchments) BFI (see Table 4.6 and Table 4.7) it was deemed inappropriate to use an interpolation calculation method, especially as sampling frequency was relatively low (fortnightly on average).

Johnes (2007) found that an *extrapolation* method using a log-log rating produced underestimates of annual TP yield – even when based on daily samples, and Walling *et al.* (1992) demonstrated that SS load estimates based on rating curves could be associated with substantial underestimation (in the order of 75%). The greatest bias is associated with sites where there is a poor discharge-concentration correlation. However, Johnes (2007) also found that extrapolation methods returned the most accurate yield estimates when stratified sampling was employed. Webb *et al.* (1997) found that extrapolation using a rating curve produced the most accurate SS load estimates for rivers in eastern England and Jordan *et al.* (2007) argued that annual P fluxes calculated from relationships between discharge and concentration are acceptable if the data set is of sufficient size and covers all flow percentiles.

As described previously, water quality grab sampling did not include sampling at or near to peak flows at all sites (Table 4.9), meaning some ratings will have to be applied with significant extrapolation, which may be a source of load over/underestimation. This should be acknowledged when analysing the calculated yields of SS, TP, SRP and NO_3; they may be of value when comparing with other annual exports but may be limited when comparing short-term or event scale fluxes, e.g., following mitigation measures. They may also not be reliable enough to allow comparison with catchments outside the study area.

4.2.2.2 Sediment and nutrient rating curve development

In order to calculate sediment and nutrient yield, a rating curve is used to complete the SS, TP, SRP and NO_3 concentration dataset at times when only discharge records exist (i.e., infrequently sampled water chemistry data at a site where continuous discharge is monitored).

The standard model is the equation:

$$\boldsymbol{C = aQ^b} \qquad \textbf{(Equation 4.1)}$$

where *C* is the concentration, *Q* is the discharge and *a* and *b* are empirical constants. Linear regression is often used for water quality data analysis. However, the application of linear regression requires normally distributed data. If data are not normally distributed, then one must use data transformation techniques or non-linear models. Anderson-Darling normality tests confirmed that all concentration data in this study were not normally distributed ($p >$ 0.005). The most common method of rating curve construction in this instance is to log-transform the discharge and concentration data (Cooke *et al.*, 2005) and then perform linear least squares regression (Phillips *et al.*, 1999; Asselman, 2000), whereby:

$$\boldsymbol{\log_{10}C = a + b\log_{10}} \qquad \textbf{(Equation 4.2)}$$

This method assumes that the concentration data are log-normally distributed, that there is a log-linear relationship between *C* and *Q* and that the residuals are log-normally distributed. The regression is then back-transformed to give a prediction of *C* in the form:

$$\boldsymbol{C = 10^{a}Q^{b}} \qquad \textbf{(Equation 4.3)}$$

Asselman (2000) found that the use of a power function between SS *C* and *Q*, fitted using non-linear regression, gave the most accurate results. Thus, in this study the *C* and *Q* data for each site were log-transformed and rating curves calculated in the form of Equation 4.2.

4.2.2.3 Calculation of yield

Rating coefficients were applied to each site's discharge record (2010 and 2011) to provide an estimation of continuous (15 minute interval) SS, TP, SRP and NO_3 concentrations. With corresponding continuous values of discharge and sediment/nutrient concentration the total yield of a determinand transported in the river over a given time period is given by:

$$\boldsymbol{Y = \int_{t_{start}}^{t_{end}} Q(t)C(t)dt} \qquad \textbf{(Equation 4.4)}$$

where *Y* is the total yield over the sampling period (t_{start} to t_{end}), *Q* is the discharge and *C* the concentration at sample time *t*. The specific yield (yield per unit area) is simply the total yield divided by the catchment area. For clarity, from this point forth *specific yield* will be referred to simply as *yield*, and *total yield* will be referred to as *load*.

4.2.3 Comparison of yields with long term estimates - flow duration curve method

In order to investigate the inter-annual differences in calculated determinand exports, long-term yields were calculated for Kirkby Stephen (as it has the longest available discharge dataset and is representative of the rest of the catchment). Eleven years of 15 minute discharge data (2000-2011) were used to construct a FDC.

Following the methodology described by Julien (1998), discharges at 15 flow duration intervals were extracted from the FDC (Table 4.17). Intervals are more closely spaced at the higher discharges as they account for a greater proportion of the overall sediment/nutrient transport and at a higher discharge there is greater sensitivity of concentration to a given change in discharge (particularly for SS and P). Equation 4.1 was used to calculate the SS, TP, SRP and NO_3 concentration for each discharge value; the annual load is then calculated as the sum of the product of each paired discharge and concentration value, together with the time occupied (in seconds) by each discharge interval.

4.2.4 *Reconciliation of total loads with export coefficients from the literature*

The export coefficient modeling approach was initially developed to determine the origin of increased nutrients to North American lakes (Beaulac and Reckhow, 1982) and is used to predict the sediment and/or nutrient loading of a catchment as a function of the export from each individual source within that catchment (Johnes, 1996). Coefficients are used for delineating pollutant loads to different land use types. Values are determined by monitoring land uses, such as forest, arable or urban and are expressed as mass of pollutant per unit area per year (e.g., kg km^{-2} yr^{-1}) (Reckhow *et al.*, 1980).

Although relatively simple, the model can provide an inexpensive approach with minimal data requirements (McGuckin *et al.*, 1999), which has been used to predict TP loadings from a number of catchments in England to within 5% of observed loads (Johnes, 1996). It also allows scaling up from plot scale to larger catchment size (Hanrahan *et al.*, 2001). The annual load of SS, TP, SRP and NO_3 for a catchment (L) can be estimated using export coefficients of each land cover type (i) using:

$$\boldsymbol{L = \sum_i (A_i E_i)} \qquad \textbf{(Equation 4.5)}$$

where A_i is the area (km^2) of the ith land cover type with an annual export of E_i (t *or* kg km^2 yr^{-1}). For diffuse sources, losses of sediment and nutrients are often assumed to be proportional to discharge in the river. Equation 4.6 is also used to predict the export of a determinand in a month (E_m) while taking account of the discharge of water from the catchment during that month (Q_m) and that derived from baseflow (B_m) (both in m^3), as follows:

$$\boldsymbol{E_m = B_m c_b + \frac{Q_m - B_m}{Q_t - B_t} \sum_i A_i E_i} \qquad \textbf{(Equation 4.6)}$$

where Q_t is the total annual discharge of water (m^3) and B_t is the total annual contribution of baseflow water (m^3) with a fixed determinand concentration of c_b. Annual load is the sum of the calculated monthly outputs. This method (adapted from May *et al*., 2001) assumes that there are no point source inputs of SS, TP, SRP and NO_3 in the catchment.

This is justified in the upper Eden due to the low population density, although it is accepted that there are a number of sewage treatment works that are unaccounted for. Domestic septic tanks are also a known source of P and NO_3, particularly during low flow conditions (Withers *et al.*, 2012) but they are a notoriously difficult point source to locate and quantify and are also omitted from these analyses. Without point inflows of nutrients, river waters are expected to be closer to a dynamic equilibrium with respect to natural internal exchange processes (e.g., uptake/release by sediments, plants and algae; losses/gains due to deposition/re-suspension of sediment); therefore no further formulae are necessary. Traditionally, TP has been used in most model calculations due to the operational problems associated with measuring SRP caused by its rapid exchange with particulate matter (Hanrahan *et al.*, 2001). Here, TP is used for the export coefficient model and the exchange between phosphorus fractions during export is not considered. To allow comparison with the literature NO_3 has been converted to total N - by applying a conversion factor of 62/14.

Land classification for the upper Eden catchment (refer to Chapter 3.2.5 for map) was further simplified into four groups: urban and rural development, unimproved grassland, improved grassland, and tilled land (Figure 4.1). Table 4.1 contains the areal extent of each land class for each sub-catchment. Suspended sediment, TP and TN export coefficients were selected from the literature (summarised in Table 4.2 - comprehensive lists of coefficients for each water quality determinand can be found in Appendix E3) and combined with areal extents using Equation 4.4 to calculate pollutant loads for each sub-catchment. Total P coefficients were selected from a number of studies conducted by McGuckin *et al.* (1999), Jordan *et al.* (2000), May *et al.* (2001) and Johnes (1996); the former two having defined them based on the Co-ORdination of INformation on the Environment (CORINE) classification and the latter two using the LCM2000.

Table 4.1: Sub-catchment land classification by areal extent.

Sub-catchment	**Catchment Area (km^2)**				
	Urban and rural development	**Unimproved grassland**	**Improved grassland**	**Tilled land**	**Total**
Upland Eden	1	31	14	2	48
Eden at Kirkby Stephen	2	38	25	4	69
Eden at Great Musgrave	9	100	96	18	223
Eden at Appleby	13	137	160	24	334
Gais Gill	0	1.1	0	0	1.1
Scandal Beck at Smardale	1	23	11	2	37
Scandal Beck at Soulby	1.5	22	14	2.5	40
Blind Beck	0.5	1	7	0.5	9
Helm Beck	0.5	5.5	10.5	1.5	18
Coupland Beck	1	20	6	1	28
Swindale Beck	1	19	10	2	32
River Belah	1	32	18	2	53
Low Hall stream	0	0	1	0.25	1.25

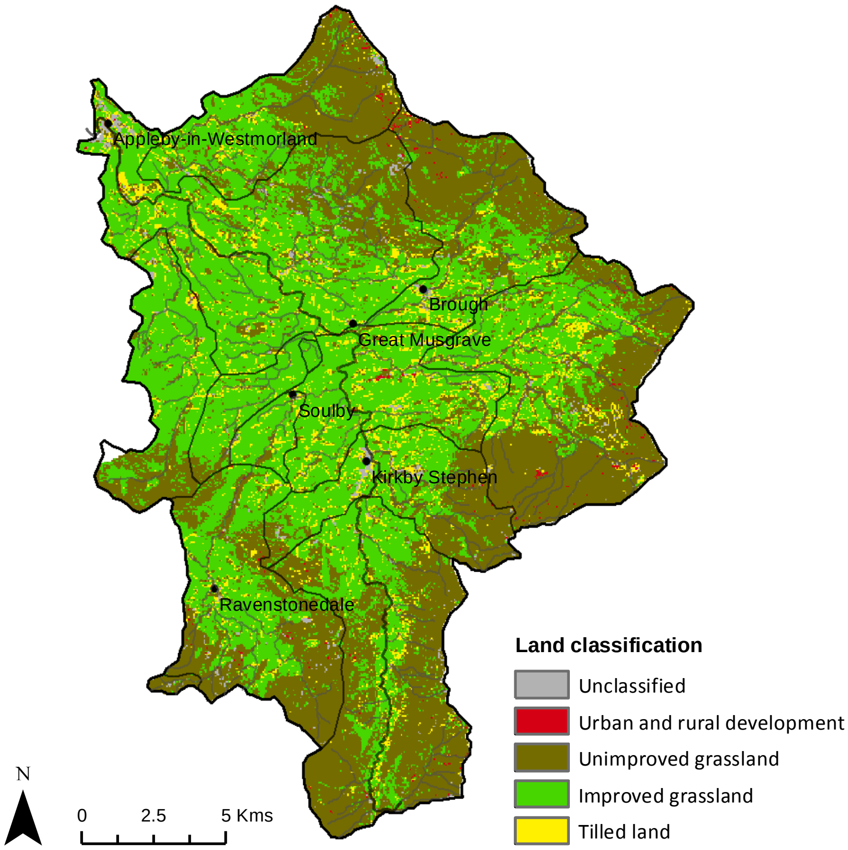


Figure 4.1: Reclassified land classification map (please refer to Figure 3.7 for original LCM 2000 map).

Table 4.2: SS, TP and TN export coefficients selected for simplified land cover types in the upper Eden catchment.

Land classification	Export coefficient ($km^{-2}\ yr^{-1}$)		
	SS (t)	TP (kg)	TN (kg)
Urban and rural development	20	83	2000
Unimproved grassland	15	10	400
Improved grassland	25	30	1300
Tilled	60	66	3500

4.3 Results

4.3.1 Hydrological characterisation

To put the sediment/nutrient fluxes calculated in this study into a wider context, it is crucial to understand the hydrological conditions of the study period in comparison with long-term averages. To assess the accuracy of precipitation and discharge data, evapotranspiration is estimated in order to calculate catchment water balances. Accurate discharge records are important for calculating reliable pollutant yields and understanding the dominant hydrological controls.

Precipitation

Rainfall totals were calculated for each sub-catchment using Thiessen polygons. The two study years, 2010 and 2011, can be considered dryer and wetter than average, respectively, in comparison to the corresponding SAAR values for the period 1961-1990 (Table 4.3). The SAAR value for the entire study catchment (Appleby - 334 km^2) is 1188 mm, while 856 mm (28% less) was recorded in 2010 and 1336 mm (12% more) in 2011. On average 40% more rainfall was recorded in 2011 than in 2010.

There is a positive correlation between rainfall and elevation (R^2 = 0.57 (SAAR), 0.53 (2010), 0.31 (2011) - Figure 4.2) with an approximate rainfall-elevation gradient of 2 mm m^{-1}, which is in the range of, but lower than, the value of 3.1 mm m^{-1} reported for the Lake District by Brunsdon *et al.* (2001). Walsh (2004) showed that total annual rainfall in the upper Eden catchment could be linearly related to elevation. The Gais Gill catchment has the highest average elevation and the highest corresponding SAAR value.

However, in both 2010 and 2011 the rainfall at Gais Gill was lower than the precipitation in the upper Eden and Kirkby Stephen catchments. This is most likely attributed to wind affected undercatch at the CHASM operated Gais Gill rain gauge (also reported by Wilkinson (2009)).

Table 4.3: Upper Eden catchment precipitation parameters (see Figure 3.9 for locations).

Site	Catchment area (km^2)	Mean catchment elevation (m)	SAAR (mm) (1961-1990)	Measured precipitation (mm)	
				2010	2011
Upland Eden	48	413	1610	1293	2178
Eden at Kirkby Stephen	69	395	1492	1070	1913
Eden at Great Musgrave	223	351	1270	913	1404
Eden at Appleby	334	319	1188	856	1336
Gais Gill	1.1	478	1906	1013	1740
Scandal Beck at Smardale	37	336	1515	985	1670
Scandal Beck at Soulby	40	322	1456	985	1670
Blind Beck	9	214	1018	779	1429
Helm Beck	18	252	1159	785	1450
Coupland Beck	28	371	1169	858	1431
Swindale Beck	32	393	1132	1014	1788
River Belah	53	385	1116	856	1066
Low Hall stream	1.25	153	854	719	1131

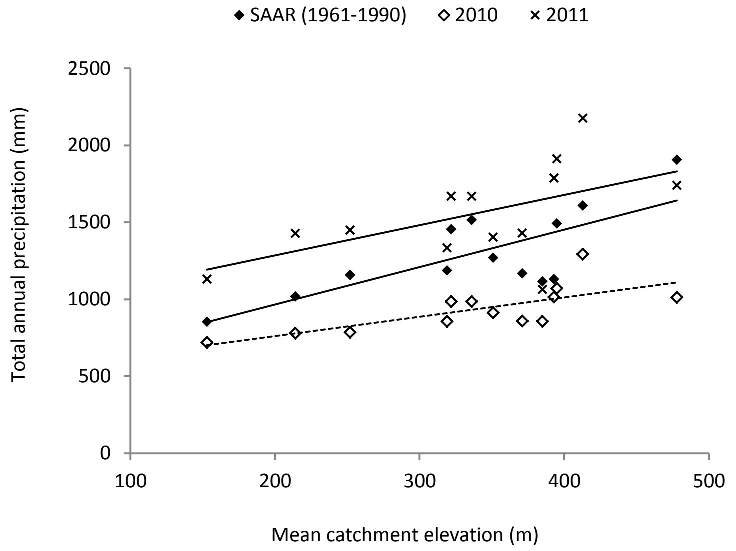


Figure 4.2: Total annual precipitation (2010 and 2011) compared with Standard-period Average Annual Rainfall (SAAR 1961-1990) against mean elevation for 13 study catchments.

Discharge

The River Eden at Kirkby Stephen has been selected to describe the hydrological regime for the entire upper Eden study catchment for the two-year study period. It was chosen as it has a long-term continuous discharge record (monitored by the EA), derived from a well established stage-discharge rating curve (gauged to 94% of maximum recorded stage – CEH, 2000). The catchment has representative land cover/use and is used as a 'control' with which to compare and quality assure the rest of the data collected in the study programme. Figure 4.3 depicts the daily precipitation and discharge hydrograph for the entire study period, as well as the days when samples were collected for sediment and nutrient analysis. The first half of 2010 was relatively dry and only 295 mm of rainfall was recorded in the Kirkby Stephen catchment, 28% of the annual total. In the second half of the year 27% of the total annual flow occurred in a series of storms between October 22nd and November 13th.

Figure 4.4 shows cumulative precipitation and runoff at Kirkby Stephen for 2010 and 2011.The low runoff accumulation between April and September 2011, despite a steadily increasing cumulative precipitation line, indicates increased soil moisture recharge in the summer, hence a decrease in rainfall-runoff ratio. Significant increases in runoff in January 2010, November 2010 and January 2011, are the result of snow melt events. As the two study years run from January to December, greater runoff than rainfall in the first half of both 2010 and 2011 is attributable to base flow time lag. Twenty years of mean daily discharge data (1991-2010) were used to produce a flow duration curve (FDC) for the River Eden at Kirkby Stephen (Figure 4.5) and accompanying flow statistics (Table 4.4).

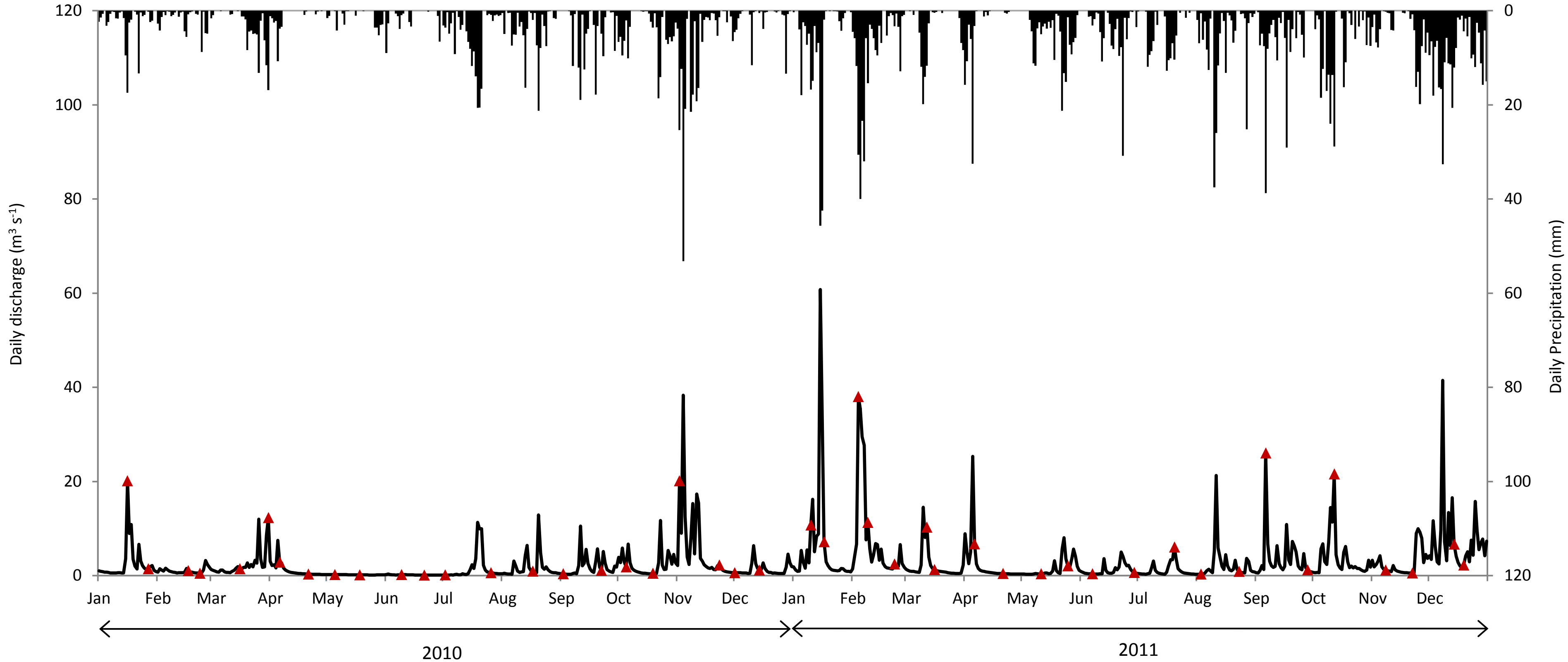


Figure 4.3: River Eden at Kirkby Stephen daily discharge and precipitation record 2010-2011. Markers indicate water quality sampling dates.

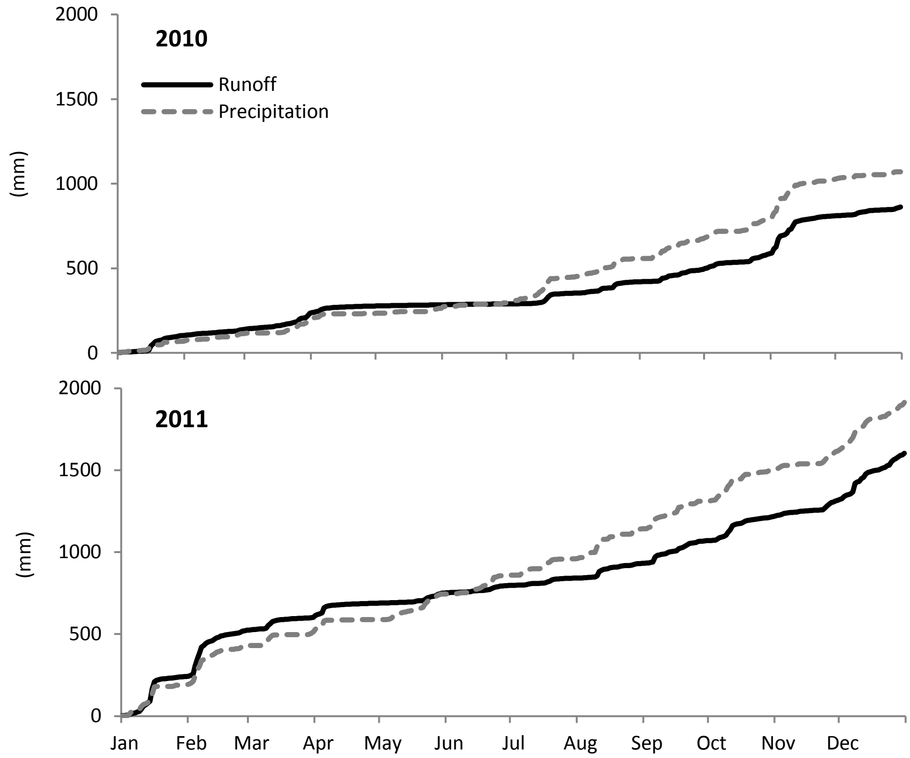


Figure 4.4: River Eden at Kirkby Stephen cumulative runoff and precipitation (mm) 2010-2011.

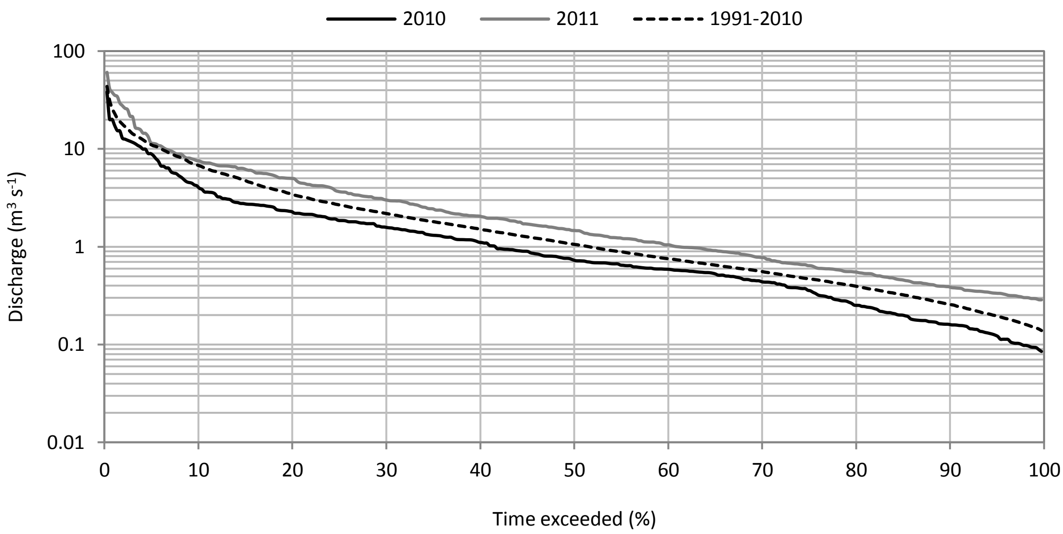


Figure 4.5: River Eden at Kirkby Stephen flow duration curve (based on mean daily discharge) comparing 2010 and 2011 with the 20-year (1991-2010) average.

Table 4.4: Flow statistics for the River Eden at Kirkby Stephen (based on mean daily discharge).

Period	**Daily discharge ($m^3 s^{-1}$)**			
	Maximum	**Q10**	**Q50**	**Q95**
2010	38.3	4.2	0.74	0.12
2011	60.8	7.6	1.47	0.34
1991-2010 (average)	43.6	6.9	1.07	0.20

Missing discharge data

As described in Chapter 3, discharge was not recorded at every site. The Eden at Kirkby Stephen and Great Musgrave were monitored by the EA and the majority of other sites were gauged by the author (as part of the CHASM program). There were three sites that were ungauged (for neither stage nor discharge). Besides the two EA monitored sites, complete records exist for the Eden at Appleby, Blind Beck and Scandal Beck at Smardale (see Table 4.5). Discharges for the ungauged sites of the upland Eden and Scandal Beck at Soulby were down- and up-scaled, respectively, from the Eden at Kirkby Stephen and Scandal Beck at Smardale, respectively, using catchment area. This was applied to discharge values recorded every 15 minutes. Discharges at the third unmonitored site, Coupland Beck, were estimated by down-scaling the Kirkby Stephen discharge record (as above) and then applying a multiplier based on the SPR HOST values for each site, to take into account soil and runoff property differences between the two catchments. For example, the multiplier was calculated as *Coupland Beck SPR HOST value* divided by *Kirkby Stephen SPR HOST value*; in this case 38.74/45.76 = 0.847. SPR HOST values were obtained for each sub-catchment using the Flood Estimation Handbook (Institute of Hydrology, 1999) CD_ROM and can be found, along with other FEH catchment descriptors, in Appendix C1.

The Helm Beck, River Belah and Low Hall sites all suffered instrumentation malfunctions. At Helm Beck this was discovered when quality assuring the data later in the study period meaning that 2011's data were disregarded. Blind Beck data were used to estimate the missing discharge values as the two catchments are the most similar in terms of physical characteristics; they are in relatively close proximity and are of approximate catchment areas. The data were up-scaled using catchment area and adjusted according to the SPR HOST values (as described above).

Following detailed analysis the River Belah stage record was also deemed unreliable, particularly during recessional and low flows as a result of inappropriate instrument siting. Thus the entire two year record was disregarded and a surrogate discharge hydrograph was scaled and adjusted from the Eden at Kirkby Stephen. The Low Hall discharge record contained a number of erroneous spikes, particularly during low flow conditions. The data were plotted alongside that of Blind Beck (as the two catchments are located next to each other they are assumed to exhibit similar hydrological responses to precipitation) in order to identify and remove spikes believed to be due to instrumental error. Subsequent data gaps were in-filled by interpolation.

At the Swindale Beck site, a large section of bank collapsed and buried the stream gauge in May 2011. A new stream gauge was installed but insufficient time meant that a stage-discharge rating relationship could not be established for the site and the change in river cross-section meant that the old rating equation was no longer applicable. The discharge record from Kirkby Stephen was scaled and adjusted to match the 2010 record. Linear regression between the two site's overlapping data had a R^2 value of 0.73. The stream gauge at Gais Gill was found to be missing in September 2011 (last downloaded in June 2011). Similarly to Swindale Beck, a new gauge was installed but a revised rating curve was not developed due to time constraints. To in-fill the missing discharge record data from a downstream stream gauge at Artlegarth Beck (2.9 km^2) was downscaled according to catchment area. Linear regression between the two site's overlapping data had a R^2 value of 0.96.

The Kirkby Stephen discharge record is thus vital. It is preferential to have local stream gauges at all sites but due to instrumentation malfunction and error, it is assumed that Kirkby Stephen provides a reasonable indicator of the hydrological catchment response.

Table 4.6 and Table 4.7 contain calculated hydrological statistics based on the finalised discharge and precipitation time series for each monitoring site for 2010 and 2011, respectively. As a means of checking consistency between discharge data normalised annual FDCs were constructed for each site (for each study year) and can be found in Appendix D1.

Table 4.5: Upper Eden catchment discharge records for 2010 and 2011.

Site	Source	Complete/Issue	Comment
Upland Eden	No gauge	-	Discharge down-scaled from Kirkby Stephen
Eden at Kirkby Stephen	EA flow	Complete	-
Eden at Great Musgrave	EA flow	Complete	-
Eden at Appleby	CHASM rating curve	Complete	-
Gais Gill	CHASM rating curve	Missing stream gauge June 2011	2011 discharge down-scaled from Artlegarth Beck
Scandal Beck at Smardale	CHASM rating	Complete	-
Scandal Beck at Soulby	No gauge	-	Discharge up-scaled up from Smardale
Blind Beck	CHASM rating	Complete	-
Helm Beck	CHASM rating	Gauge failure May 2011	2011 discharge adjusted from Blind Beck
Coupland Beck	No gauge	-	Discharge adjusted from Kirkby Stephen
Swindale Beck	CHASM rating	Bank collapsed on gauge October 2010	New gauge installed but no stage-discharge rating curve. Discharge adjusted from Kirkby Stephen
River Belah	CHASM rating	Gauge untrustworthy	Gauge recorded flow peaks but recession and base flow inaccurate. Discharge adjusted from Kirkby Stephen
Low Hall stream	CHASM rating	Gauge malfunction 2011	Gauge recorded flow peaks but recession and base flow inaccurate. 2011 discharge missing

Table 4.6: Upper Eden catchment flow parameters 2010. *calculated using Institute of Hydrology method (Gustard *et al.*, 1992).

Site	Catchment area (km^2)	Mean daily flow ($m^3 s^{-1}$)	Max hourly flow ($m^3 s^{-1}$)	Q10 ($m^3 s^{-1}$)	Q50 ($m^3 s^{-1}$)	Q95 ($m^3 s^{-1}$)	Q10/Q95 ratio	BFI*	Annual precipitation (mm)	Annual runoff (mm)	Percentage runoff
Upland Eden	48	1.26	69	2.33	0.47	0.083	28	0.25	1293	897	69
Eden at Kirkby Stephen	69	1.89	103	3.5	0.71	0.125	28	0.38	1070	862	81
Eden at Great Musgrave	223	5.27	247	10.8	2.39	0.620	17	0.31	913	714	78
Eden at Appleby	334	6.92	322	12.5	3.30	1.194	10	0.33	856	653	76
Gais Gill	1.1	0.033	1.12	0.051	0.023	0.014	4	0.56	1013	942	93
Scandal Beck at Smardale	37	0.838	41.23	1.458	0.418	0.189	8	0.38	985	714	72
Scandal Beck at Soulby	40	0.881	43.37	1.534	0.440	0.199	8	0.38	985	695	71
Blind Beck	9	0.164	3.46	0.248	0.117	0.071	3	0.61	779	576	74
Helm Beck	18	0.31	6.58	0.471	0.223	0.134	4	0.42	785	532	68
Coupland Beck	28	0.636	34.84	1.181	0.239	0.042	28	0.25	858	646	75
Swindale Beck	32	0.884	43.55	1.479	0.299	0.053	28	0.25	1014	889	88
River Belah	53	1.361	75.81	2.527	0.510	0.090	28	0.25	1016	810	80
Low Hall stream	1.25	0.018	0.184	0.035	0.011	0.008	4	0.63	719	460	64

Table 4.7: Upper Eden catchment flow parameters 2011.

Site	Catchment area (km^2)	Mean daily flow ($m^3 s^{-1}$)	Max hourly flow ($m^3 s^{-1}$)	Q10 ($m^3 s^{-1}$)	Q50 ($m^3 s^{-1}$)	Q95 ($m^3 s^{-1}$)	Q10/Q95 ratio	BFI*	Annual precipitation (mm)	Annual runoff (mm)	Percentage runoff
Upland Eden	48	2.34	81	5.0	0.90	0.223	22	0.21	2178	1668	77
Eden at Kirkby Stephen	69	3.5	122	7.5	1.35	0.335	22	0.21	1913	1604	84
Eden at Great Musgrave	223	9.03	275	20.1	4.05	1.140	18	0.26	1404	1222	87
Eden at Appleby	334	12.3	355	24.1	5.94	1.559	15	0.27	1336	1164	87
Gais Gill	1.1	0.058	1.58	0.087	0.035	0.018	5	0.44	1740	1649	95
Scandal Beck at Smardale	37	1.469	62.46	2.440	0.575	0.290	8	0.27	1670	1252	75
Scandal Beck at Soulby	40	1.545	65.69	2.566	0.605	0.305	8	0.27	1670	1218	73
Blind Beck	9	0.295	3.50	0.580	0.142	0.071	8	0.39	1429	1035	72
Helm Beck	18	0.87	6.65	1.102	0.269	0.135	8	0.25	1450	983	68
Coupland Beck	28	1.184	41.42	2.531	0.456	0.113	22	0.21	1431	1202	84
Swindale Beck	32	1.563	51.88	3.170	0.571	0.142	22	0.21	1788	1475	82
River Belah	53	2.451	87.434	5.415	0.975	0.242	22	0.21	1548	1491	96
Low Hall stream	1.25	0.028	0.190	0.073	0.021	0.009	8	0.52	1131	902	80

Water Balances

A catchment water balance is a simple method of checking the accuracy of hydrometeorological data for a certain catchment area, over a given period of time. Assuming that there are no losses from the catchment (e.g., to groundwater) and thus negligible changes in storage, the water balance is defined as:

$$\boldsymbol{Q = P - E} \qquad \textbf{(Equation 4.7)}$$

where *Q* is discharge, *E* is actual evapotranspiration and *P* is precipitation (all in mm – after discharge has been divided by catchment area). Potential evapotranspiration was calculated using a modified version of the Penman-Monteith equation, parameterised for the upper Eden catchment by Wilkinson (2009). Inputs for the calculation were hourly measurements of relative humidity, net radiation and temperature – all recorded by the Great Musgrave AWS. The actual evapotranspiration is here assumed to be equal to the potential evapotranspiration. For the headwaters this may be more acceptable than for the lower lying sub-catchments, as the water table of upland peat is near the surface for most of the year (Evans *et al.*, 1999). Annual water balances were calculated for all thirteen study catchments (Appendix D2).

The water balance is regarded as being sufficiently accurate when recorded *Q* is within +/- 10% of calculated *P-E* (Table 4.8); a positive value indicates that *Q* < *P-E* and a negative value indicates that *Q* > *P-E*. From this check, 15 out of 26 values are within +/- 10% and the remaining 11 are relatively close (apart from Upland Eden 2010). Thus, it is concluded that the error in the hydrometric data is acceptable considering the simplifying assumptions made for the water balance.

Possible error could to be attributed to an underestimation of total precipitation (undercatch by the raingauge in strong winds) or an error in the calculation of evapotranspiration (assuming that the runoff total is correct). An alternative explanation could be discharge over- or underestimated as a result of rating curve extrapolation. This may be an issue at CHASM sites but is assumed not to be a problem at the EA sites due to largely complete stage-discharge rating curves (i.e., gauged to a high percentage of the maximum recorded stage – Kirkby Stephen = 94%).

Table 4.8: Kirkby Stephen catchment annual water balances.

Site	% diff. between Q and P-E	
	2010	2011
Upland Eden	19	12
Eden at Kirkby Stephen	3	2
Eden at Great Musgrave	2	-9
Eden at Appleby	3	-10
Gais Gill	-14	-13
Scandal Beck at Smardale	11	10
Scandal Beck at Soulby	13	12
Blind Beck	3	10
Helm Beck	11	16
Coupland Beck	4	-4
Swindale Beck	-7	2
River Belah	3	-18
Low Hall stream	14	-6

4.3.2 Concentration characterisation

Table 4.9 contains information on the maximum discharge (peak 15 minute discharge value) recorded at each site during the study period along with the maximum discharge at which a water sample was collected (to the nearest 15 minute interval). This is expressed as a percentage of the maximum discharge and ranged from 29-92% - the lowest percentage is associated with Coupland Beck. Due to its position in the catchment Coupland Beck was often the last to be sampled during a sample collection day, meaning that it was perhaps further into recessional flow if the sampling campaign was in response to an earlier high-flow event.

Despite an effort to collect grab samples during high flow events only two out of the top twelve discharge events were captured. Table 4.10 contains the date and time of the largest flow peaks recorded at the Kirkby Stephen gauging site and shows whether the event was sampled. The largest sampled event was on 04/02/2011, which was the fifth largest discharge event overall. The only other significant event to be sampled was on 12/10/2011 but the flow peaked at 08:00 am meaning that sampling occurred on the recession. This highlights another important issue as to whether samples are taken on the rising or falling limb of a storm and is discussed further in section 4.3.3.1. Table 4.10 shows that half of the twelve largest high flow events occurred during night time hours and a further two over the weekend, both of which were impractical to sample.

Table 4.9: Percentage of maximum discharge sampled for water quality analysis.

Site	Max discharge recorded ($m^3 s^{-1}$)	Max discharge sampled ($m^3 s^{-1}$)	Percentage of max discharge sampled
Upland Eden	82	42	51
Eden at Kirkby Stephen	123	60	49
Eden at Great Musgrave	281	98	35
Eden at Appleby	358	127	36
Gais Gill	1.71	0.74	43
Scandal Beck at Smardale	64	23	36
Scandal Beck at Soulby	69	21	30
Blind Beck	3.5	3.1	89
Helm Beck	12.3	10.1	82
Coupland Beck	49	14	29
Swindale Beck	43	15	35
River Belah	71	31	44
Low Hall stream	0.182	0.167	92

Table 4.10: Time/date of 12 largest flow peaks (defined as the largest 15 minute discharge value) at the Eden at Kirkby Stephen (2010-2011) and whether they were sampled.

Event rank	Date/time	Peak discharge ($m^3 s^{-1}$)	Sampled collected	Comment
1	08/12/2011 11:30	123	No	Not available
2	15/01/2011 14:15	109	No	Saturday
3	04/11/2010 18:45	105	No	Not available
4	06/02/2011 14:15	77	No	Sunday
5	04/02/2011 12:30	69	Yes	Largest event sampled
6	06/09/2011 00:15	68	No	Time
7	05/04/2011 05:15	63	No	Time
8	12/10/2011 08:00	57	Yes	Sampled on recession
9	02/11/2010 20:00	56	No	Time
10	13/12/2011 03:15	46	No	Time
11	23/10/2010 01:00	42	No	Time
12	10/01/2011 22:15	39	No	Time

Figure 4.6 summarises diffuse pollutant concentrations within the upper Eden.

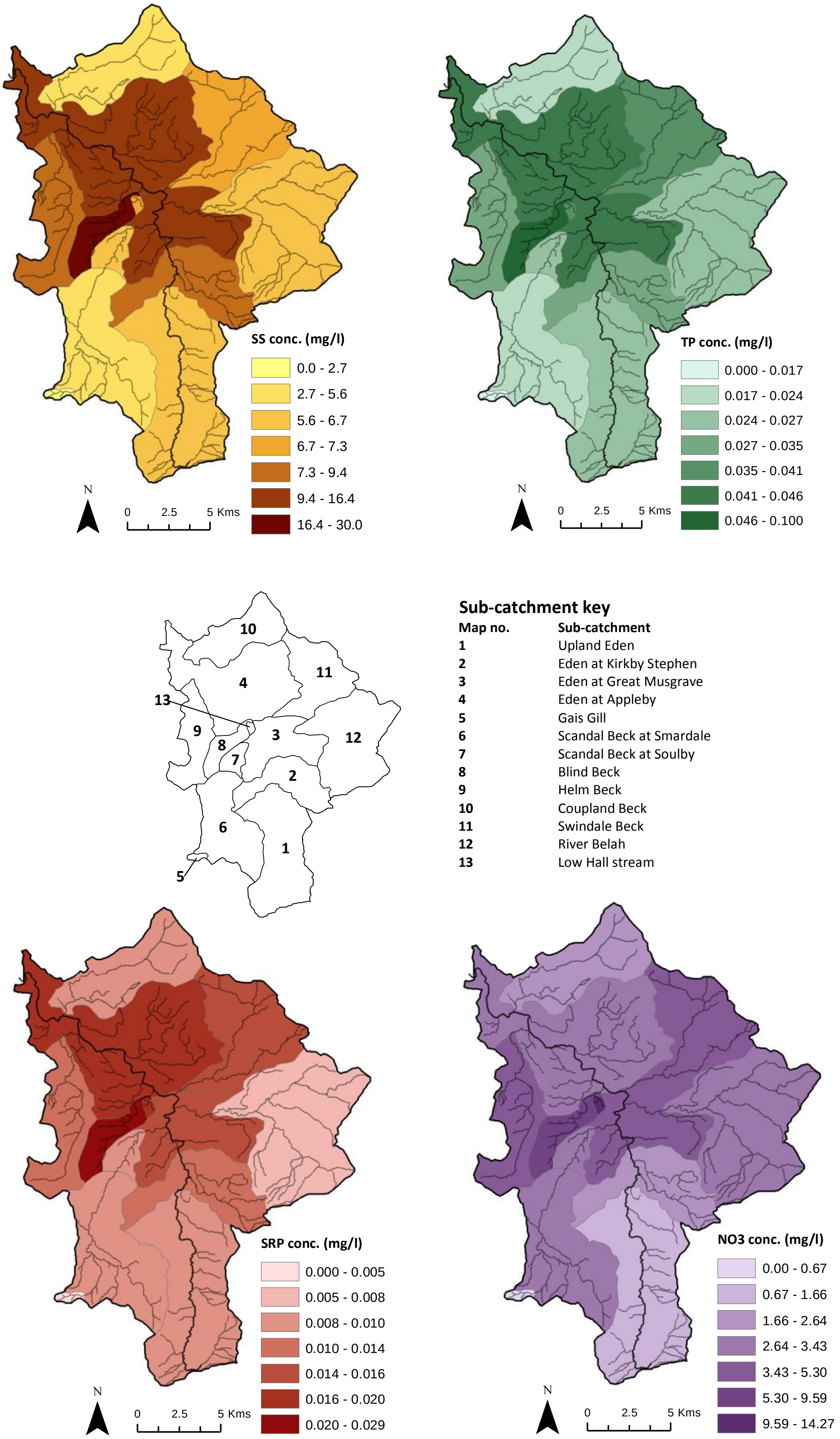


Figure 4.6: Maps depicting mean SS, TP, SRP and NO_3 concentrations. Concentration is proportional to the darkness of the colour (i.e., light = low concentration, dark = high concentration).

Blind Beck shows up as having the highest mean concentrations of all water quality determinands, with the exception of NO_3 where the Low Hall stream (a tributary of Blind Beck) is greater. Higher SS concentrations (relative to the rest of the catchment) are found in central areas. These catchments include Gt. Musgrave and Appleby and to a slightly lesser extent Helm Beck and Kirkby Stephen. Gais Gill, Smardale and Coupland Beck exhibit the lowest SS concentrations. Total P concentrations are relatively high in the Gt. Musgrave and Appleby catchments and a similar pattern can be seen for SRP, with the addition of Swindale Beck. Gais Gill has the lowest mean P concentration for both fractions. The highest NO_3 concentrations were found in the Low Hall stream and Blind Beck sub-catchment with the Gt. Musgrave, Helm Beck and Swindale Beck catchments also having elevated concentrations relative to the rest of the catchment. Again, Gais Gill stands out as having the lowest overall mean concentration.

Suspended sediment, TP, SRP and NO_3 concentration data are summarised in turn by boxplots in the following section; sites have been ordered considering Figure 3.10 to better visualise how the concentrations vary along the river system (from upstream to downstream). Tabulated concentration and discharge data can be found in raw form in Appendix E1. Suspended sediment concentration data is summarised in Figure 4.7. Mean SS concentration increases site-by-site along the River Eden continuum (upstream to downstream), with 6.7 mg l^{-1} at the upland Eden and 16.4 mg l^{-1} at the Eden at Appleby site. Despite the increase in mean SS concentration between Kirkby Stephen and Gt. Musgrave, all the sub-catchments that enter the main river between these points have lower mean concentrations. Gais Gill has the lowest mean concentration. The River Belah and Swindale Beck have comparable mean and maximum SS concentrations along with a similar standard deviation.

Blind Beck has a higher mean SS concentration (29.2 mg l^{-1}) than both Gt. Musgrave (t = 2.70; p = 0.009) and the Eden at Appleby (t = 2.48; p = 0.017). In fact Blind Beck had the highest mean concentration of all the study sites as well as the highest maximum SS concentration of 276.5 mg SS l^{-1}. After Blind Beck, Helm Beck has the second highest mean SS concentration of the sub-catchment sites, albeit by a small amount.

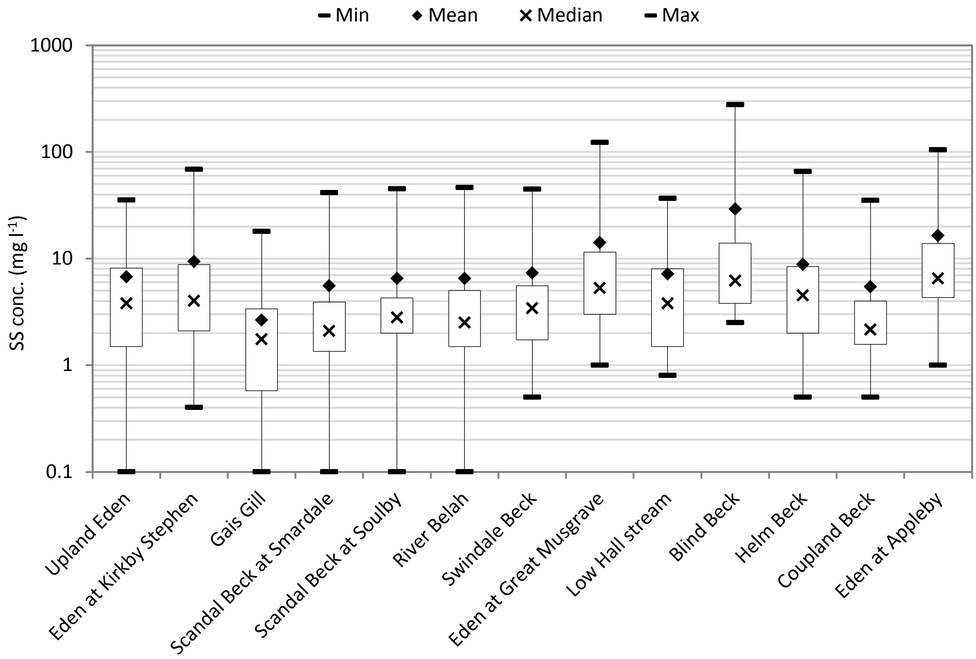


Figure 4.7: Box and whisker plot summarising suspended sediment concentrations across all upper Eden sites.

Total phosphorus concentration data is summarised in Figure 4.8. Mean TP concentrations exhibit a similar pattern to those of SS. Figure 4.9 shows a positive correlation between SS and TP (Pearson's R = 0.493; p = 0.001) using corresponding data from all 13 study sites. There is a significant increase along the River Eden continuum from 0.027 mg TP l^{-1} at upper Eden to 0.046 mg TP l^{-1} at Appleby (t = -7.20; p < 0.001). Blind Beck has the highest mean TP concentration of 0.098 mg l^{-1}, significantly greater than the second highest at Appleby (t = 3.13; p = 0.003). Gais Gill and Coupland Beck have the lowest mean concentrations of 0.017 and 0.024 mg TP l^{-1} respectively. Along the Scandal Beck nested system, there is a significant increase in mean TP concentration between Gais Gill and Scandal Beck (0.017 – 0.024 mg TP l^{-1} – t = -4.95; p < 0.001). Mean concentrations from the outfalls of Scandal Beck, at Soulby, and the River Belah were both lower than that at Kirkby Stephen, while Swindale and Helm Beck were both higher.

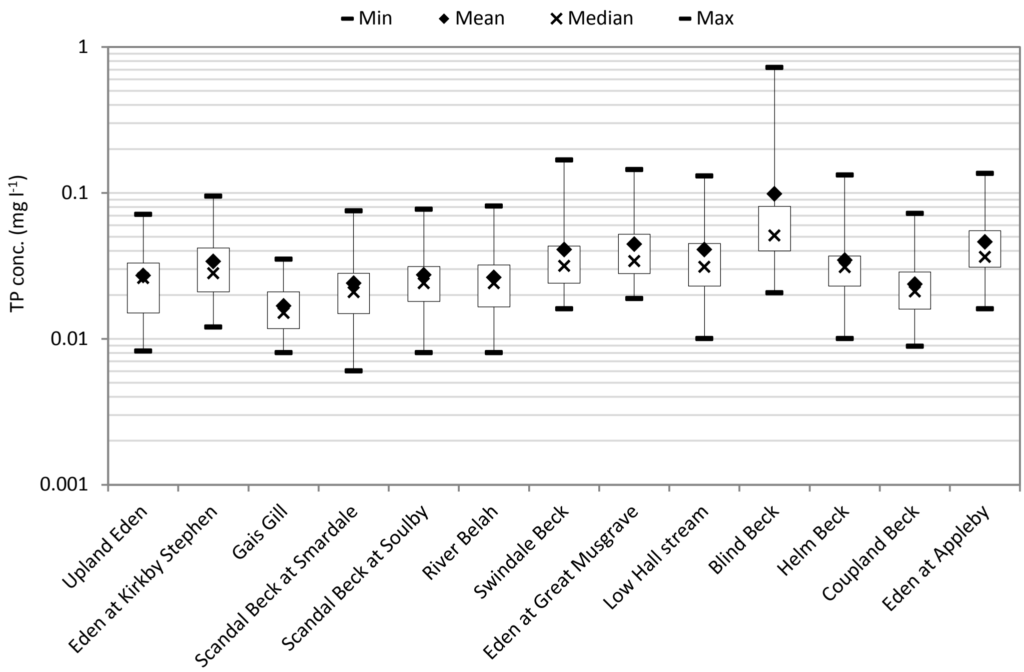


Figure 4.8: Box and whisker plot summarising total phosphorus concentrations across all upper Eden sites.

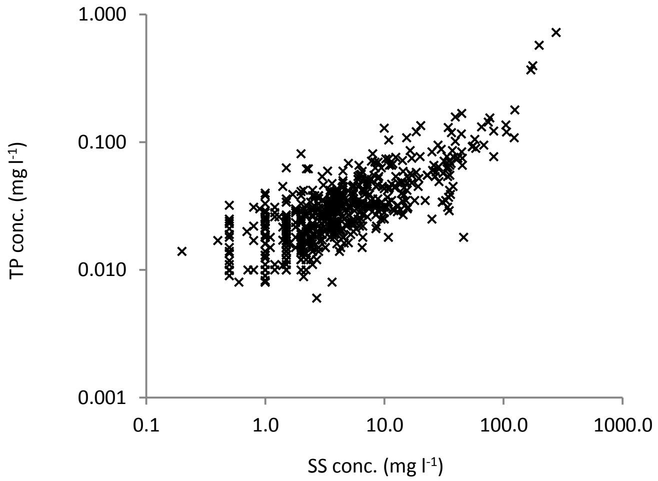


Figure 4.9: Correlation between SS and TP concentrations from all sites
(zero values not included on the log-log plot).

The pattern of SRP mean concentrations also follows that of TP. The correlation between TP and SRP concentrations (across all sites – see Figure 4.11) is strong and positive (Pearson's R = 0.755; p < 0.001). Figure 4.10 summarises the SRP data. The greatest mean of 0.029 mg l^{-1} and the greatest maximum of 0.159 mg l^{-1} were both recorded at Blind Beck. The Low Hall stream has the second highest mean concentration followed by the Eden at Appleby and Helm Beck.

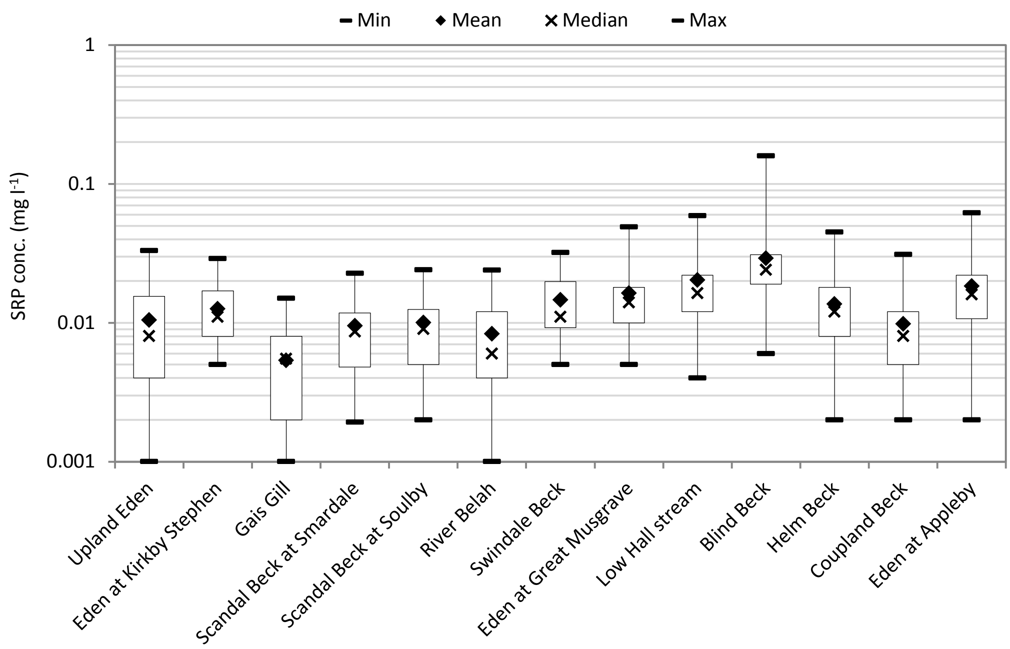


Figure 4.10: Box and whisker plot summarising soluble reactive phosphorus concentrations across all upper Eden sites.

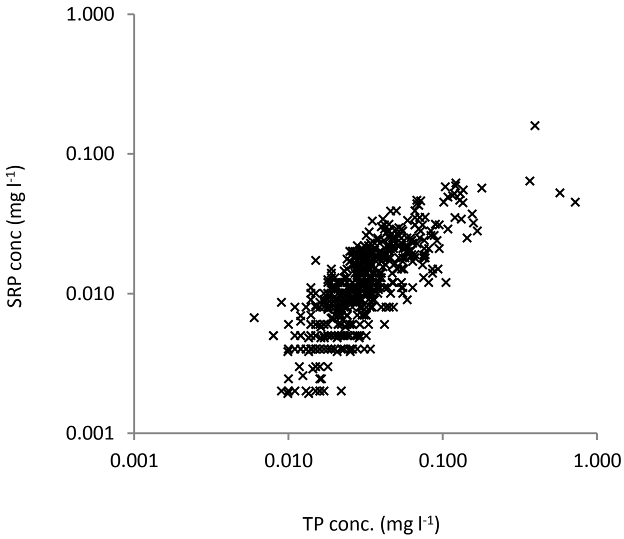


Figure 4.11: Correlation between TP and SRP concentrations from all sites (zero values not included on the log-log plot).

Mean NO_3 concentrations also steadily increase along the River Eden continuum but on the whole remain low (Figure 4.12). A mean of 1.66 mg NO_3 l^{-1} (with a maximum of 3.55) was recorded at the upper Eden site and a mean of 3.24 mg NO_3 l^{-1} (with a maximum of 6.6) was registered at Appleby over the two-year period. Gais Gill has the lowest mean concentration and the Low Hall stream has the highest, at 14.13 mg NO_3 l^{-1}. The highest single concentration was also recorded at Low Hall stream: 24.5 mg l^{-1}. Blind Beck has the highest mean NO_3

concentration of all the sub-catchments that discharge directly into the River Eden (Significantly greater than the second highest, Helm Beck (t = 10.67; p < 0.001)). This is likely due to the influence of the Low Hall stream.

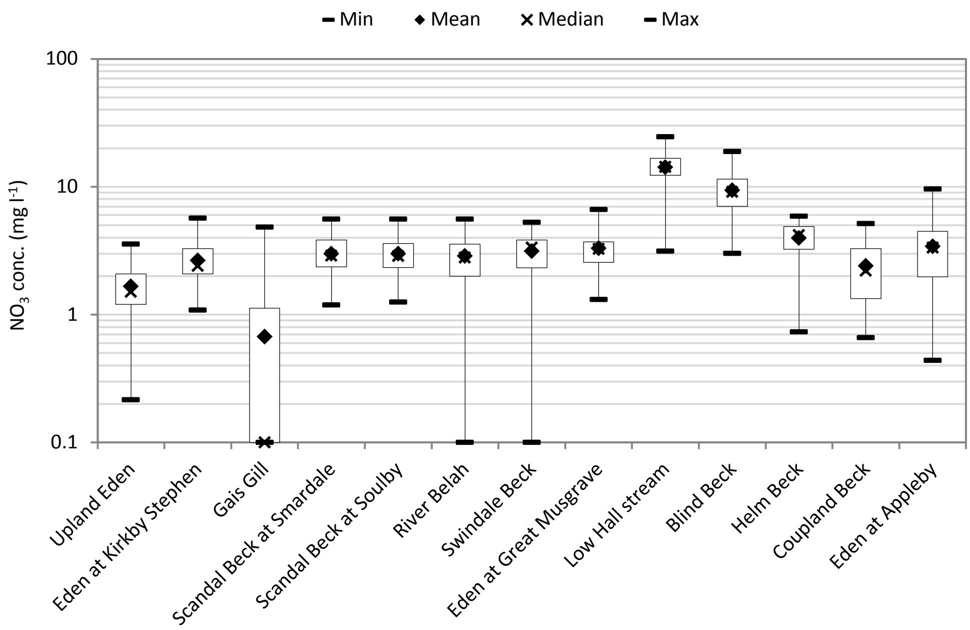


Figure 4.12: Box and whisker plot summarising nitrate concentrations across all upper Eden sites.

4.3.2.1 Seasonality

Figure 4.13 presents SS, TP, SRP and NO_3 concentration data arranged by month, recorded at Kirkby Stephen. The plots exhibit a very general pattern that is subject to variability in storm events and droughts; however, SS displays a peak between October and February with an intervening trough centred on June. A similar pattern is observed for TP and SRP. For NO_3, there is a slight increase in winter and early spring but the range is only between 2 and 5 mg NO_3 l^{-1}.

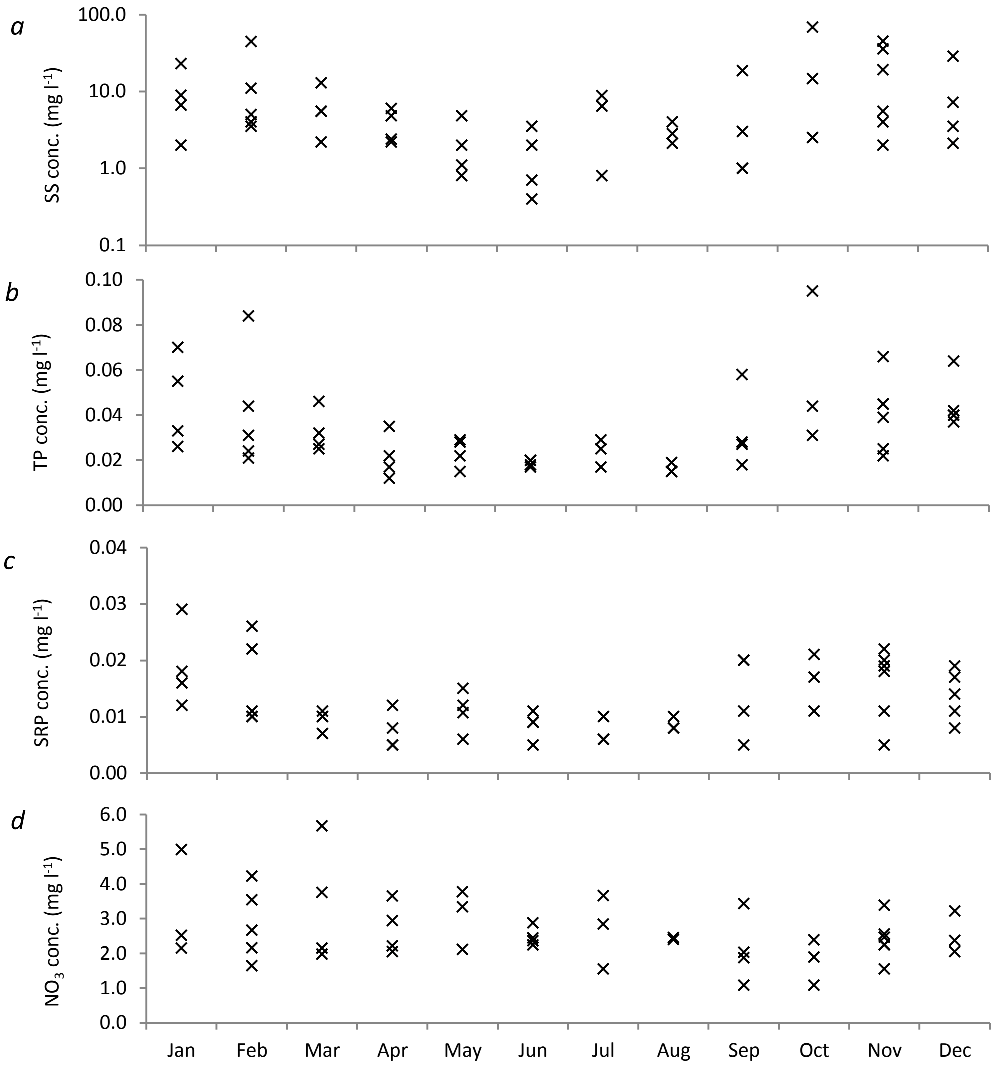


Figure 4.13: Monthly a) SS, b) TP, c) SRP and d) NO_3 concentrations recorded at Kirkby Stephen.

4.3.3 Sediment and nutrient rating curve development

Correlations of instantaneous SS, TP, SRP and NO_3 concentrations with discharge for the River Eden at Kirkby Stephen are depicted in Figure 4.14. Determinand concentration-discharge rating coefficients for each site can be found in Appendix E2. A certain amount of scatter can be attributed to measurement error but is mainly due to the complexity and time-varying nature of sediment and nutrient supply and delivery rates, which confound the discharge-concentration relationship. This was discussed in the Chapter 3.

Variability in concentration/instantaneous load at low discharges is not seen as overly problematic, however, as the effect on estimation of total exports is minimal. Pearson's correlation coefficients for all water quality determinands at all sites can be found in Table 4.11. All the SS, TP and SRP rating curves are statistically significant (at a minimum of the 5% level), but only NO_3 ratings from Blind Beck, Low Hall and Appleby are statistically significant.

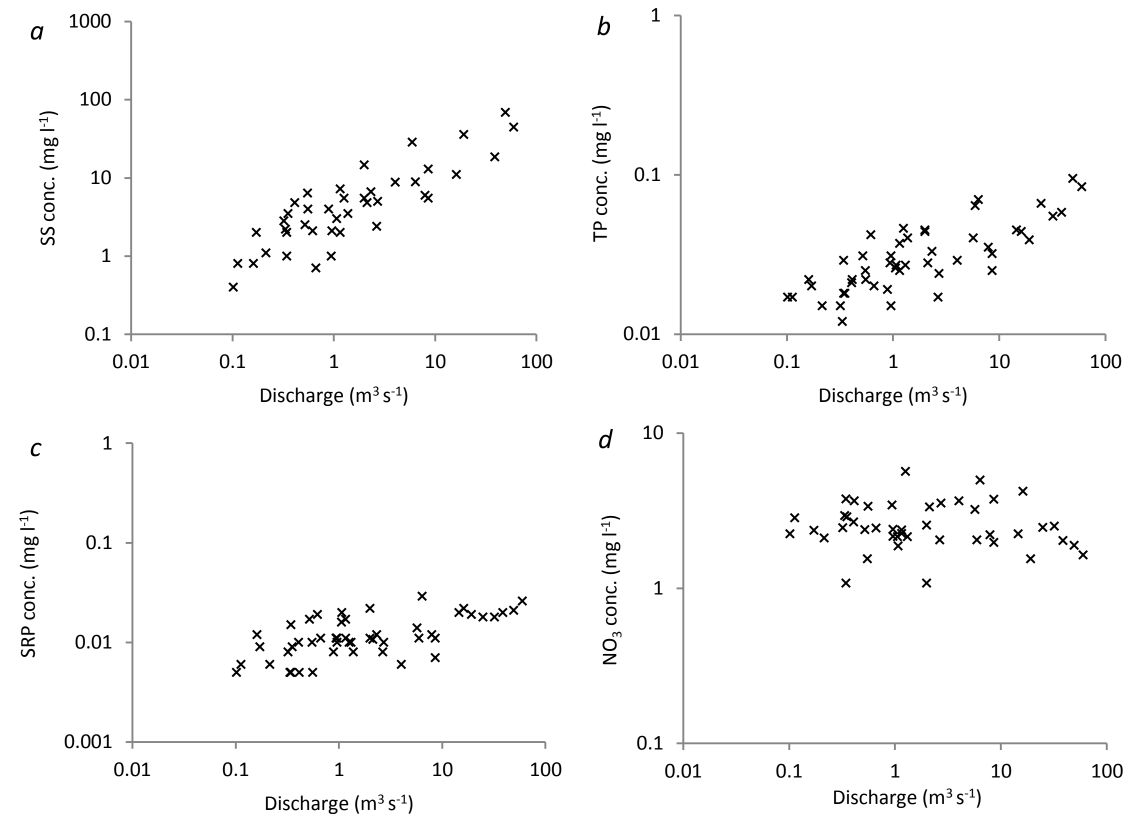


Figure 4.14: Relationship between a) SS; b) TP; c) SRP; d) NO_3 and discharge for the River Eden at Kirkby Stephen.

Table 4.11: Pearson's correlation coefficients for SS, TP, SRP and NO_3 against discharge.

Site	SS	TP	SRP	NO_3
Upland Eden	0.729***	0.790***	0.585***	-0.274
Eden at Kirkby Stephen	0.856***	0.793***	0.589***	-0.202
Gais Gill	0.933***	0.596***	0.474**	0.089
Scandal Beck at Smardale	0.860***	0.840***	0.577***	-0.268
Scandal Beck at Soulby	0.911***	0.828***	0.617***	-0.233
River Belah	0.904***	0.743***	0.315*	-0.332
Swindale Beck	0.912***	0.896***	0.576***	-0.289
Eden at Great Musgrave	0.888***	0.859***	0.673***	-0.426
Low Hall stream	0.827***	0.862***	0.762***	-0.737***
Blind Beck	0.927***	0.888***	0.728***	-0.605***
Helm Beck	0.931***	0.891***	0.743***	-0.156
Coupland Beck	0.921***	0.819***	0.767***	-0.306
Eden at Appleby	0.913***	0.829***	0.726***	-0.439**

Significance levels: *p<0.05 **p<0.01 ***p<0.001

4.3.3.1 *The influence of sample collection timing on rating curves*

The effect of season and the position on the hydrograph (i.e., whether the sample was collected on the rising or falling limb) on the discharge-concentration relationship will be considered. The number of samples collected for each site ranged between 35 and 50, which limits the level of analysis that can be carried out. It is clear from the data that the majority of samples were collected on the falling limb (80-85% - depending on the number of samples collected at the individual site). This is mainly due to the time needed to collect samples following the onset of a storm event. Owing to the nature of the hydrographs in the upper Eden catchment (see example from Kirkby Stephen - Figure 4.15, where the time to peak is 3 hours 45 minutes while the recessional lasts for over 24 hours), rising limbs occupy much less time than falling limbs, thus it was much more difficult to collect samples on the rising limb of storm events. This effect has already being highlighted as an issue at Coupland Beck in particular, which had the lowest discharge sampled (as a percentage of the maximum recorded discharge).

Figure 4.16 shows scatterplots of SS, TP, SRP and NO_3 against discharge from Kirkby Stephen, but with the data split between rising and falling limb. Suspended sediment concentrations are marginally higher on the rising limb, for a given value of discharge, when comparing the linear regression lines. There is no visual difference for TP and SRP, and although there is no correlation between discharge and NO_3 concentration, concentrations appear marginally higher on the falling limb than on the rising limb.

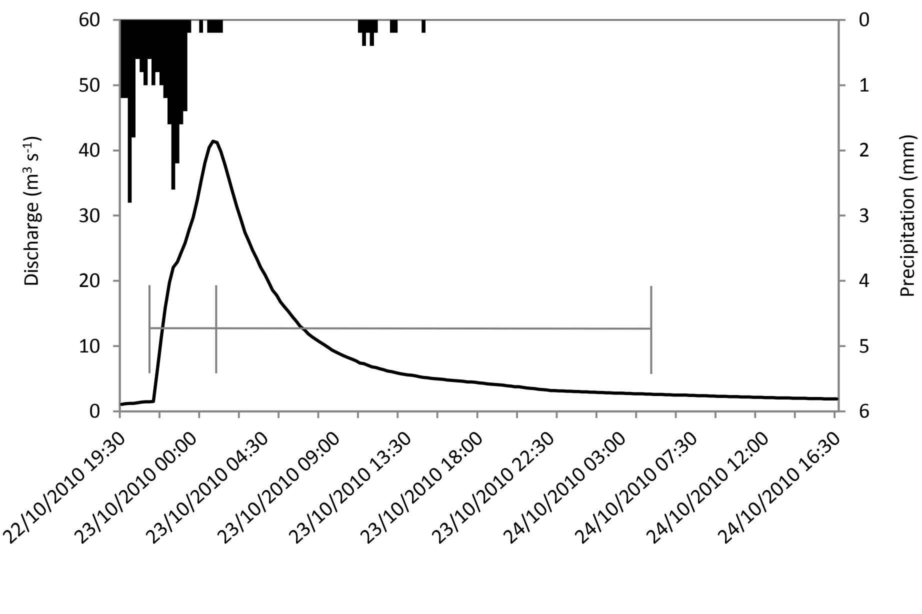


Figure 4.15: Typical River Eden hydrograph recorded at Kirkby Stephen.

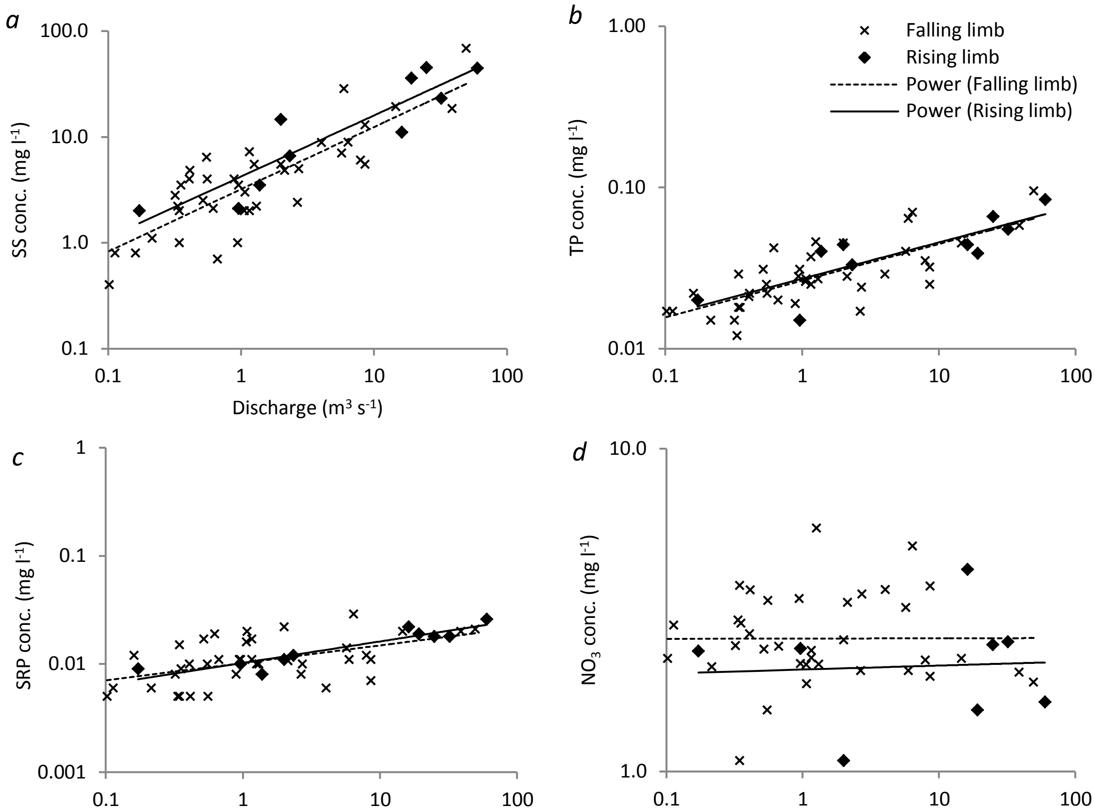


Figure 4.16: Scatter plots of a) SS; b) TP; c) SRP and d) NO_3 concentration against discharge; divided into rising and falling limb for Kirkby Stephen.

Data from each site is grouped according to season. May to September constitute the summer, and October to April the winter. Separate linear regression lines between SS, TP, SRP and NO_3 concentration and discharge were plotted for the summer and winter periods; below are examples from Kirkby Stephen (Figure 4.17). Visual examination of the regression lines reveals that SS and TP concentrations are higher in the winter, for a given discharge value; there is no discernible difference for SRP; and that in winter NO_3 concentrations have no correlation with discharge while there is a slight negative correlation in the summer. However, there is insufficient data and too much scatter to have any significant confidence in this pattern. Given the low numbers of samples in this study it was believed that separating the rating curves would not significantly improve the accuracy of the yields based on them.

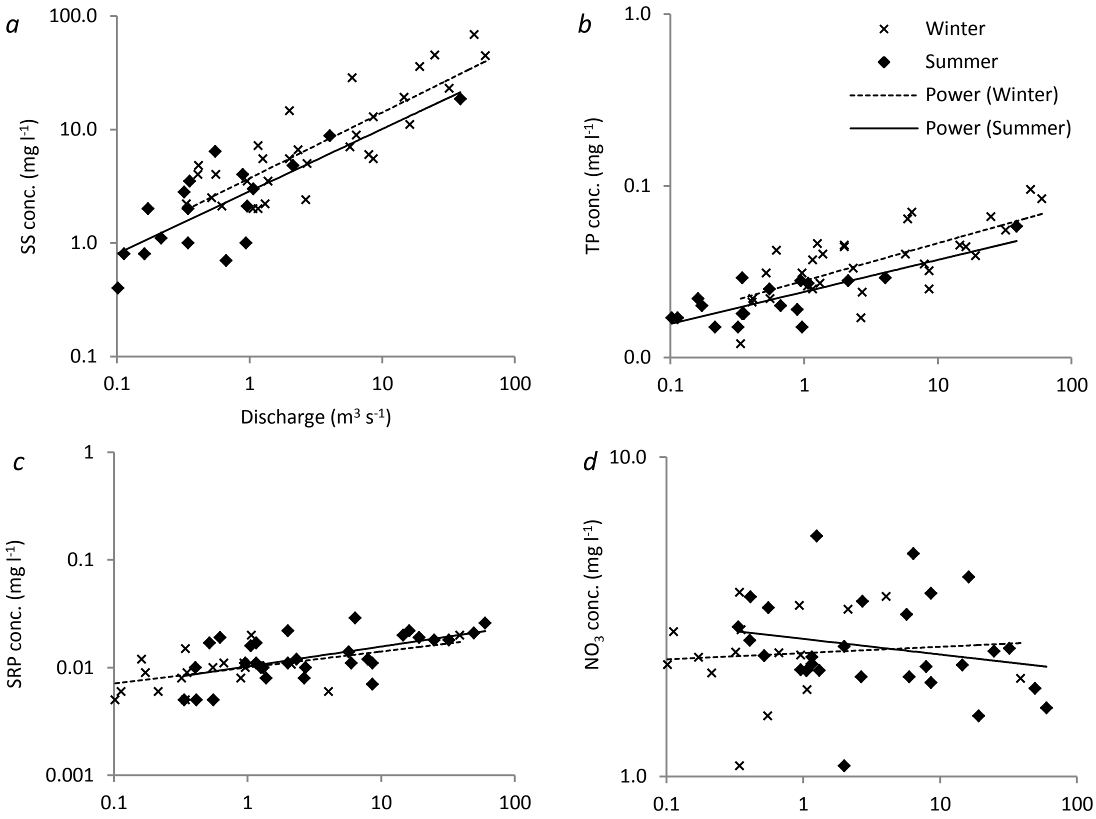


Figure 4.17: Scatter plots of SS, TP, SRP and NO_3 concentration against discharge; divided into summer and winter for Kirkby Stephen.

4.3.4 Yield characterisation

Figure 4.18 summarises diffuse pollutant yields within the upper Eden.

The maps are based on 2011 data, which has higher yields than 2010, but the relative difference between the sub-catchments is very similar. Blind Beck has the highest SS yield, with Gt. Musgrave, Kirkby Stephen and the River Belah also having higher yields than the other catchments. Gais Gill, Low Hall and Coupland Beck are the sub-catchments with the lowest yields. Blind Beck also stands out as having the highest TP yield and is again followed by Gt. Musgrave and Kirkby Stephen. However, unlike SS, Swindale Beck and Helm Beck stand out from the remaining sub-catchments as having elevated yields. Gais Gill and Low Hall have the lowest.

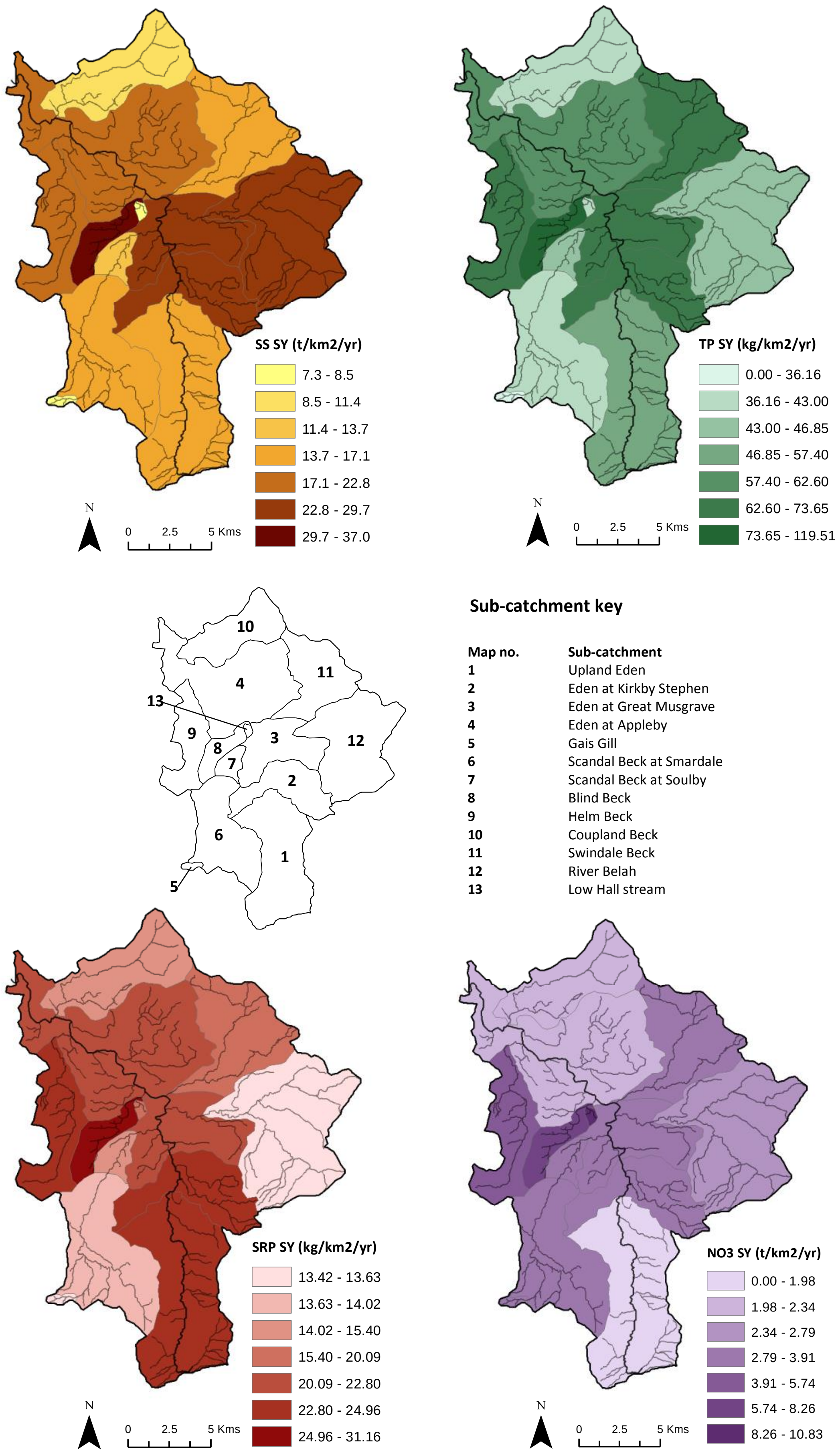


Figure 4.18: Maps depicting mean SS, TP, SRP and NO_3 specific yield. Yield is proportional to the darkness of the colour (i.e., light = low yield, dark = high yield).

Blind Beck has the highest SRP yield but the pattern for the other catchments is slightly different to that of TP. The Kirkby Stephen catchment is again highlighted as having a higher yield but also along with the upper Eden. Helm Beck is also in the same classification. SRP yield is lowest at Gais Gill and the River Belah while the Low Hall stream is showing to have a higher SRP yield relative to its TP yield. Nitrate yield is highest for the Low Hall stream and lowest for the upper Eden. Blind Beck also shows up as having elevated NO_3 yield, possible as a result of the input from the Low Hall stream, and Helm Beck is also has levels elevated above the remaining sub-catchments.

Table 4.12, Table 4.15, Table 4.14 and Table 4.16 contain the calculated loads and yields of SS, TP, SRP and NO_3, respectively, for each sub-catchment for 2010 and 2011. For all water quality determinands, yield is significantly higher in 2011 than 2010 ($p < 0.001$). The outlet to the study catchment (the River Eden at Appleby) had the greatest total annual loads, as expected, as it is the largest catchment. There is a strong positive correlation between SS load and catchment area (Pearson's $R = 0.983$ (2010) and 0.987 (2011); $p < 0.001$). The SS load at the River Eden at Great Musgrave is over three-times greater than that at Kirkby Stephen in both years, although this is roughly proportional to catchment area.

There is, however, a discrepancy when the difference in loads is compared with the sum of the loads from the sub-catchments that enter the main river between these points. There is an increase of 2105 tonnes and 4435 tonnes of sediment between Kirkby Stephen and Gt. Musgrave in 2010 and 2011, respectively. The sum of the totals from the contributing sub-catchments between Kirkby Stephen and Great Musgrave (Scandal Beck, River Belah and Swindale Beck) is 783 t and 2649 t for 2010 and 2011; leaving a discrepancy of 1322 and 1786 tonnes, respectively. This could be attributed to an area of approximately 11 km^2 that lies between Kirkby Stephen and Great Musgrave and is not part of the three sub-catchments. This issue is considered in more detail in section 4.3.6.

Table 4.12: Calculated SS yields and loads for 2010 and 2011.

Site	Catchment area (km^2)	SS yield ($t\ km^{-2}\ y^{-1}$)		SS load ($t\ yr^{-1}$)	
		2010	2011	2010	2011
Upland Eden	48	6.77	17.10	325	821
Eden at Kirkby Stephen	69	10.53	27.84	727	1921
Gais Gill	1.1	3.10	8.50	3.1	8.5
Scandal Beck at Smardale	37	4.70	15.60	174	577
Scandal Beck at Soulby	40	4.64	14.90	186	596
River Belah	53	7.40	29.70	392	1574
Swindale Beck	32	6.47	16.70	205	529
Eden at Great Musgrave	223	12.70	28.50	2832	6356
Low Hall stream	1.25	2.33	7.34	2.9	9.2
Blind Beck	9	8.73	35.35	79	318
Helm Beck	18	4.04	22.36	73	402
Coupland Beck	28	4.54	11.35	125	312
Eden at Appleby	334	9.60	22.80	3206	7615

Figure 4.19 shows cumulative SS load for the two study years at Kirkby Stephen. This provides a good example of the seasonal distribution of sediment delivery throughout the year and is applicable to all of the study sites. As continuous SS concentration is calculated as a function of discharge, the temporal pattern is very much controlled by the hydrology. It should be noted that significant snowmelt events occurred in January and November 2010 and January 2011. As a result there is a very strong positive correlation between cumulative discharge and cumulative SS load (R = 0.982, P <0.001 (2010) and R = 0.979, P <0.001 (2011). There is a marked difference between 2010 and 2011 in terms of total export but there are similarities in the timing of significant sediment fluxes.

Suspended sediment yields are compared with those calculated for the upper Eden catchment by Mills (2009), where yield values represent a long-term average (derived from 6-years' worth of discharge data) (Table 4.13). On the whole, the yields calculated in this study are low, particularly 2010 values. However, 2011 yields are much more in agreement; Kirkby Stephen, Great Musgrave, Scandal Beck at Smardale and Soulby, and the River Belah are all similar. However, Mills (2009) reports higher yields for Swindale Beck, Helm Beck, Blind Beck, Gais Gill and Appleby.

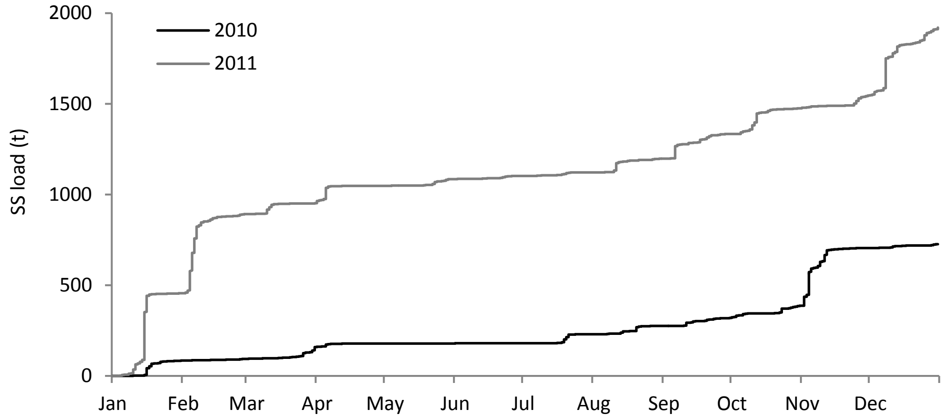


Figure 4.19: Cumulative SS load for 2010 and 2011 at Kirkby Stephen.

Table 4.13: Long-term SS yields calculated by Mills (2009).

Site	Catchment area (km^2)	SS yield (t km^{-2} y^{-1})	Highest discharge sampled as % of max discharge
Eden at Kirkby Stephen	69	26	61
Gais Gill	1.1	23*	29
Scandal Beck at Smardale	37	12	47
Scandal Beck at Soulby	40	11	47
River Belah	53	35	35
Swindale Beck	32	26	53
Eden at Gt. Musgrave	223	22	73
Blind Beck	9	73*	67
Helm Beck	18	46	34
Eden at Appleby	334	46*	100

*sites equipped with automatic storm sampling equipment.

Total P yield and load data is summarised in Table 4.15. The highest yield is for Blind Beck (119.5 kg km^{-2} yr^{-1} in 2011), significantly higher than all the other sub-catchments. Gais Gill has the lowest yield of all the study sites while Coupland Beck, River Belah and Scandal Beck have the lowest of the sub-catchment that discharge directly into the River Eden. Total P yield increases along the main River Eden (highest at Gt. Musgrave in 2010 and Kirkby Stephen in 2011), but then decreases at Appleby. This is despite the relatively high yield from Blind Beck, which in absolute terms is only supplying a small proportion of the TP load due its small catchment size.

Swindale Beck and Helm Beck have yields elevated above the rest of the sub-catchments, but not as high as for Blind Beck. Soluble reactive P yield data is summarised in Table 4.14. Similarly to TP, SRP yield generally increases along the main River Eden (peaking at Gt. Musgrave) before decreasing slightly at Appleby. Blind Beck has the highest yield in both 2010 and 2011: 13.4 and 31.2 kg km^{-2} yr^{-1}, respectively. All the other sub-catchments have relatively low SRP yields of around 7 kg km^{-2} yr^{-1} in 2010 and 15 kg km^{-2} yr^{-1} in 2011, with the River Belah being the lowest and Swindale Beck being slightly higher. The value for Helm Beck appears relatively elevated in 2011.

Table 4.14: Calculated SRP yields and loads for 2010 and 2011.

Site	Catchment area (km^2)	SRP yield (kg km^{-2} yr^{-1})		SRP load (kg yr^{-1})	
		2010	2011	2010	2011
Upland Eden	48	11.1	24.2	531	1159
Eden at Kirkby Stephen	69	12.1	25.0	835	1723
Gais Gill	1.1	6.7	13.6	7	14
Scandal Beck at Smardale	37	6.7	14.0	246	519
Scandal Beck at Soulby	40	6.9	14.7	277	589
River Belah	53	4.2	13.4	221	711
Swindale Beck	32	9.9	19.5	314	617
Eden at Great Musgrave	223	12.1	22.8	2825	5313
Low Hall stream	1.25	7.7	20.1	10	25
Blind Beck	9	13.4	31.2	120	280
Helm Beck	18	6.8	24.5	122	440
Coupland Beck	28	7.3	15.4	201	423
Eden at Appleby	334	11.2	22.4	3751	7489

Table 4.15: Calculated TP yields and loads for 2010 and 2011.

Site	Catchment area (km^2)	TP yield (kg km^{-2} yr^{-1})		TP load (kg yr^{-1})	
		2010	2011	2010	2011
Upland Eden	48	26.8	57.4	1288	2755
Eden at Kirkby Stephen	69	34.6	73.7	2387	5082
Gais Gill	1.1	17.8	36.2	18	36
Scandal Beck at Smardale	37	18.8	42.9	695	1585
Scandal Beck at Soulby	40	21.2	46.6	846	1863
River Belah	53	14.3	46.9	756	2483
Swindale Beck	32	33.4	70.9	1059	2246
Eden at Great Musgrave	223	37.2	72.8	8669	16966
Low Hall stream	1.25	15.8	43.0	20	54
Blind Beck	9	38.3	119.5	345	1076
Helm Beck	18	18.7	68.8	337	1239
Coupland Beck	28	19.8	41.4	544	1138
Eden at Appleby	334	31.0	62.6	10353	20920

Nitrate yield data is summarised in Table 4.16. The highest NO_3 yield (6.36 and 10.83 t km^{-2} yr^{-1} for 2010 and 2011, respectively) is from the Low Hall catchment. The lowest NO_3 yields were recorded at the upland Eden site, the River Belah and Coupland Beck with the other catchments being marginally higher. There is a slight increase in yield along the main River Eden continuum but unlike the SS and P, NO_3 does not decrease between Gt. Musgrave and Appleby.

Figure 4.20 describes cumulative NO_3 load at Kirkby Stephen. Although NO_3 export is clearly driven by discharge events, when compared with cumulative SS (Figure 4.19) the accumulation is 'smoother' with less distinct responses to high flow events.

Table 4.16. Calculated NO_3 yields and loads for 2010 and 2011.

Site	Catchment area (km^2)	Specific NO_3 yield (t km^{-2} yr^{-1})		NO_3 load (t yr^{-1})	
		2010	2011	2010	2011
Upland Eden	48	1.11	1.98	53.5	95.0
Eden at Kirkby Stephen	69	2.12	3.91	146.3	270.1
Gais Gill	1.1	2.24	3.51	2.2	3.5
Scandal Beck at Smardale	37	1.93	3.23	71.5	119.3
Scandal Beck at Soulby	40	1.94	3.24	77.5	129.6
River Belah	53	0.98	2.79	51.7	147.7
Swindale Beck	32	1.80	3.22	57.0	102.1
Eden at Great Musgrave	223	2.07	3.38	483.4	786.8
Low Hall stream	1.25	6.36	10.83	8.0	13.5
Blind Beck	9	5.65	8.26	50.9	74.3
Helm Beck	18	2.03	5.74	36.6	103.4
Coupland Beck	28	1.35	2.34	37.2	64.3
Eden at Appleby	334	2.09	3.36	699.6	1121.5

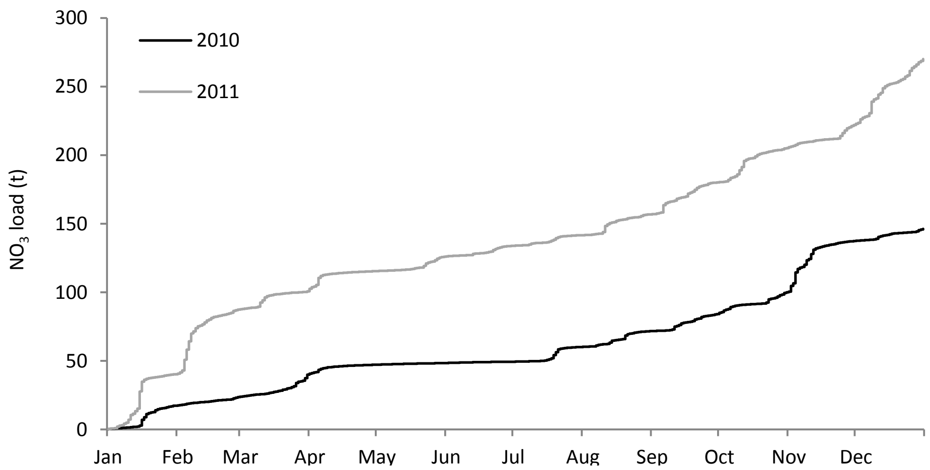


Figure 4.20: Cumulative NO_3 load for 2010 and 2011 at Kirkby Stephen.

4.3.4.1 Cumulative load exceedance

Figure 4.21 depicts cumulative SS loads (as percentages) for 2010 and 2011 for Kirkby Stephen. Ninety per-cent of the SS load was delivered in 14 % of the time in 2010 and in just 6.5 % of the time in 2011.

Very similar values are returned for TP, SRP and NO_3, apart from being 5.5% of the time in 2011. These low values indicate the importance of infrequent, high magnitude events in the transport of sediment and nutrients in the upper Eden catchment.

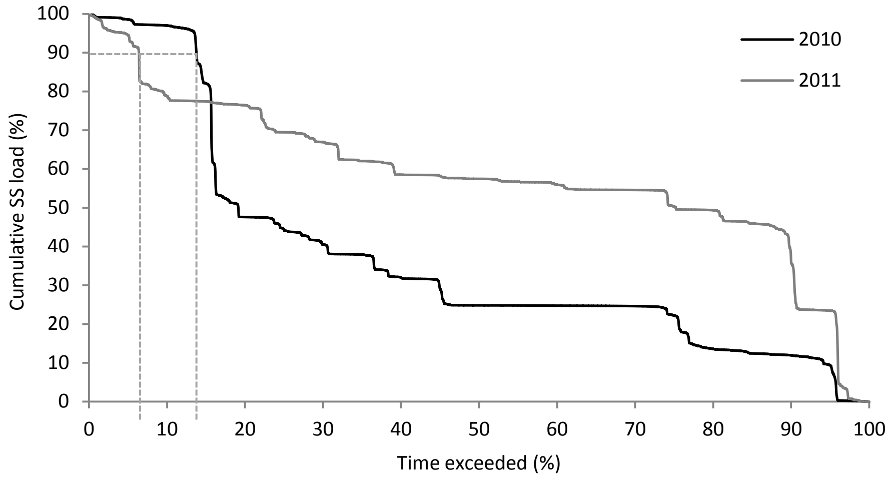


Figure 4.21: Cumulative SS load (as percentage) exceedance at Kirkby Stephen.

Cumulative TP load exceedance for Blind Beck is depicted in Figure 4.22. Even for a different sub-catchment (with a totally separate discharge record) a similar pattern to Figure 4.21 is observed. Ninety per-cent of the total TP load was accounted for by 13 % and 5.5 % of the time in 2010 and 2011, respectively. Similar values are found for TP and SRP while 90 % of NO_3 is delivered in just 10 % of the time in 2010 and 5% in 2011.

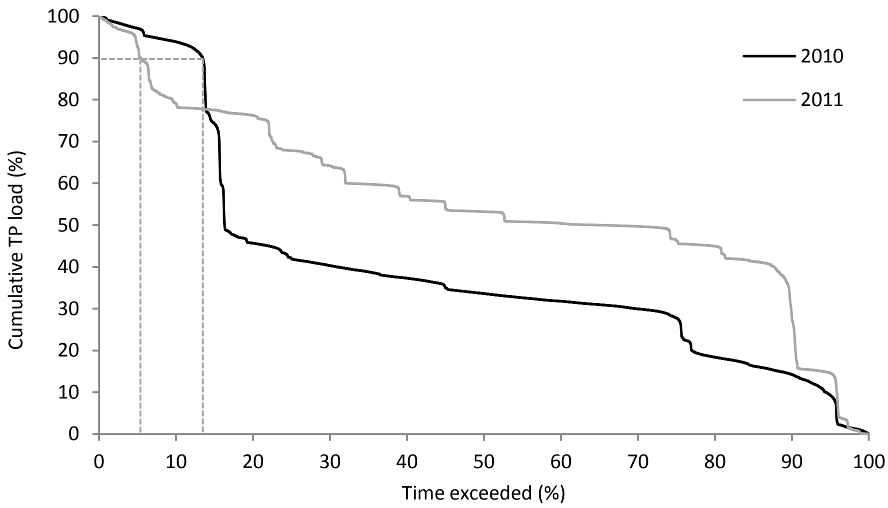


Figure 4.22: Cumulative TP load (as percentage) exceedance at Blind Beck.

4.3.4.2 *Flow duration curve method*

Results for SS only are shown in Table 4.17. All determinand loads and yields calculated using the 11 year FDC are displayed in Table 4.18. They are taken to be the best long-term yield estimates and because long term variability in flow is likely to be similar at each sub-catchment due to the relatively small catchment size, it is apparent that 2010 produced lower yields and 2011 produced higher yields than the 11 year, long-term average.

Table 4.17: Flow duration curve intervals and discharges with corresponding SS concentrations and loads for Kirkby Stephen.

Start - end (%)	Interval	Discharge ($m^3 s^{-1}$)	SSC ($mg l^{-1}$)	Load (t)
0 - 0.002	0.0002	120.6	60.2	45.8
0.002 - 0.1	0.0008	93.4	51.6	121.7
0.1 - 0.5	0.004	52.6	36.5	242.7
0.5 - 1.5	0.01	29.1	25.6	234.7
1.5 - 5	0.035	15.1	17.2	287.0
5 - 15	0.1	6.40	10.3	207.1
15 - 25	0.1	2.96	6.45	60.2
25 - 35	0.1	1.99	5.07	31.8
35 - 45	0.1	1.35	4.02	17.1
45 - 55	0.1	0.97	3.29	10.0
55 - 65	0.1	0.72	2.75	6.23
65 - 75	0.1	0.56	2.35	4.13
75 - 85	0.1	0.40	1.94	2.48
85 - 95	0.1	0.26	1.50	1.24
95 - 98.5	0.035	0.16	1.10	0.19
			Total (t)	**1272**
			Specific yield ($t\ km^{-2}\ yr^{-1}$)	**18.4**

Table 4.18: 2010 and 2011 yields and loads compared with those calculated using the FDC method for Kirkby Stephen.

	Annual yield (per km^2)			**Annual load**		
	FDC	2010	2011	FDC	2010	2011
SS (t)	18.4	10.53	27.84	1272	727	1921
TP (kg)	53.6	34.6	73.65	3697	2387	5082
SRP (kg)	18.4	12.1	24.96	1269	835	1723
NO_3 (t)	3	2.12	3.91	208	146	270

4.3.5 Relationships between load/yield and spatial scale, land use and precipitation

A principal aim of this study was to investigate the influence of spatial scale on sediment and nutrient yields. There is a significantly strong positive relationship (*P*<0.001) between catchment scale and SS, TP, SRP and NO_3 loads (combined total over the two-year study period) (Figure 4.23).

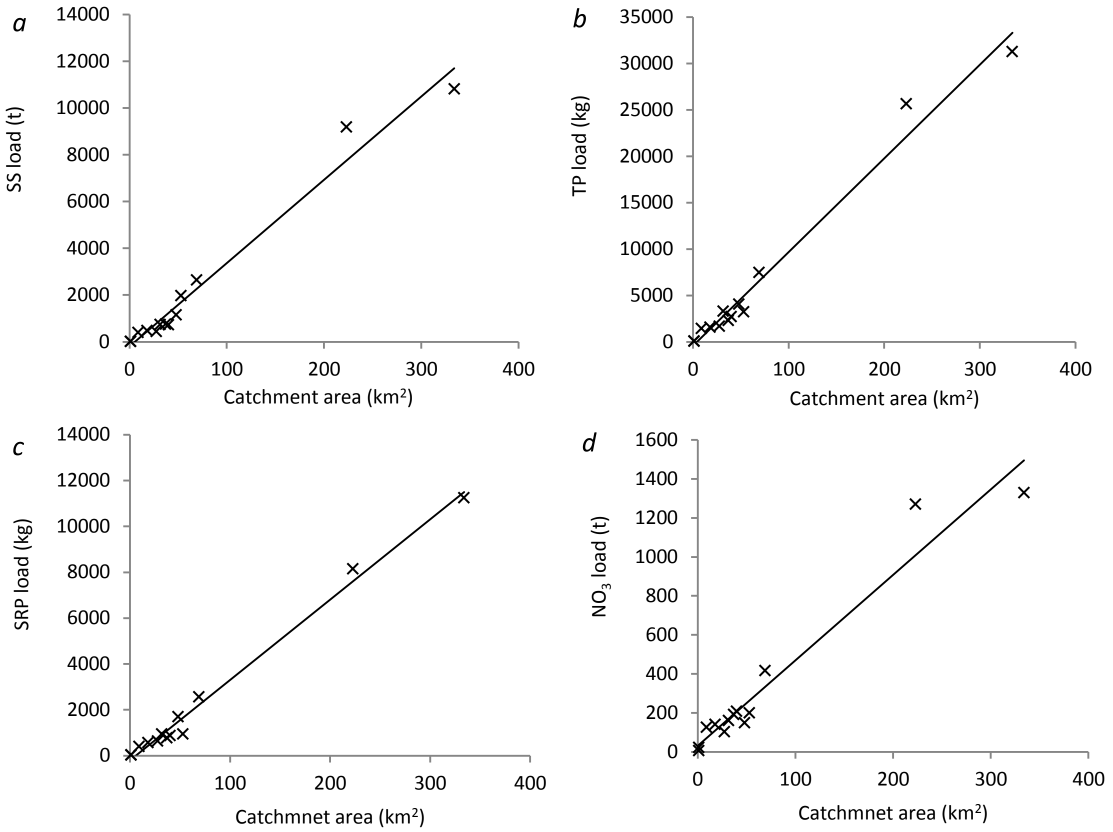


Figure 4.23: Correlation between *a*) SS; *b*) TP; *c*) SRP and *d*) NO_3 load and catchment area.

When correlations of SS, TP, SRP and NO_3 yield against catchment area were plotted (Figure 4.24), there is no clear trend and this is reinforced by Pearson's correlation coefficients (Table 4.19); *P* values indicate that there is a higher degree of scatter in 2011, the wetter of the two study years. This may mean that whatever processes/factors are responsible for controlling sediment and nutrient export, they are being positively influenced by increased levels of hydrologic activity (i.e., more rainfall, total runoff and possibly different flow pathways). Results presented in section 4.4.3 suggested that SS, TP and SRP yields increased along the main River Eden but then started to decrease again at the catchment outfall at Appleby, while NO_3 yield remain relatively constant.

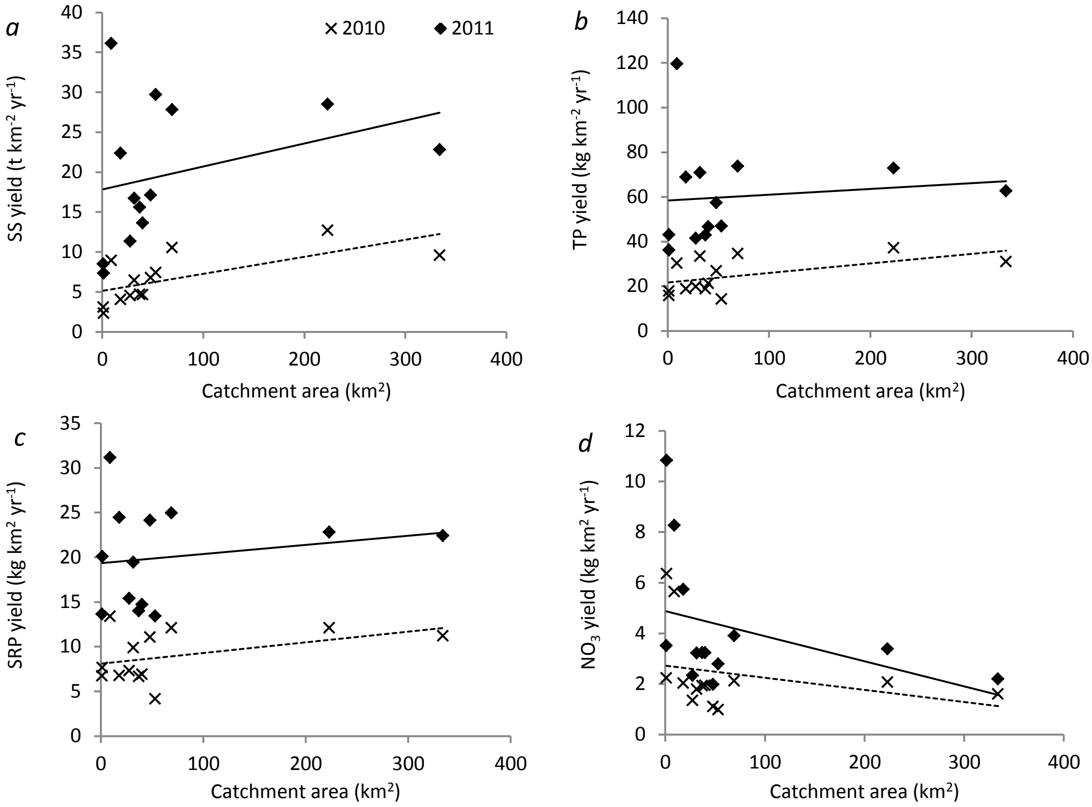


Figure 4.24: Correlation between *a*) SS; *b*) TP; *c*) SRP and *d*) NO_3 yield and catchment area.

Table 4.19: Pearson correlation coefficients and *P*-values for correlations between sediment/nutrient yield and catchment area.

Determinand	Year	Pearson's *R*	*P*-value
SS	2010	0.668	0.013*
	2011	0.318	0.290
TP	2010	0.531	0.062
	2011	0.114	0.711
SRP	2010	0.411	0.164
	2011	0.180	0.557
NO_3	2010	-0.284	0.347
	2011	-0.373	0.210

*significant correlations (P<0.05)

The data suggest that catchment area does not have a significant role in determining the sediment and nutrient yield of a catchment; it is more likely an effect of position in the catchment and/or the local characteristics. The relationship between land class and SS, TP, SRP and NO_3 yield is examined by plotting yields against the percentage of *improved agriculture* (improved grassland and tilled land combined). An assumption is made that this land class is more intensively farmed (e.g., a higher stocking density, involve arable rotations and/or silage cutting and reseeding, receive higher nutrient loading, etc.) hence potentially more likely to export higher quantities of sediment/nutrients.

When all study sites were included in the analysis no significant correlation was found for any water quality determinand ($P > 0.05$ in all cases). The analysis was repeated on the non-nested sub-catchments only (Figure 4.25) (i.e., omitting the River Eden sites) but with Low Hall removed as it is an anomaly - 100% improved agricultural land but low SS, TP and SRP yields. However, Low Hall was included in the NO_3 plot; this site will be discussed in section 4.6. With the above exclusions considered, the scatterplots suggest that there is a positive correlation between the percentages of improved agriculture and yield, for all determinands. These correlations are all significant (at the 5% level) apart from for SS, TP and SRP in 2010 (Table 4.20).

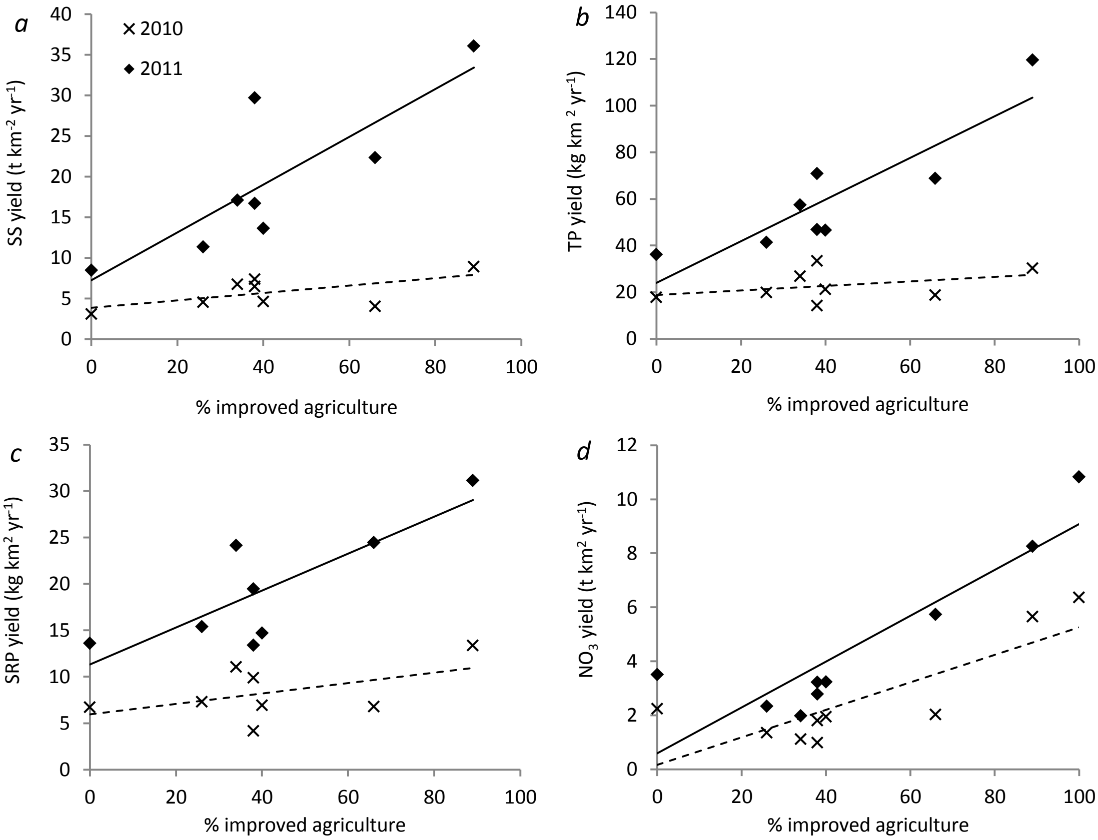


Figure 4.25: Correlation between *a*) SS; *b*) TP; *c*) SRP and *d*) NO_3 yield and percentage of improved grassland and tilled land.

Table 4.20: Pearson correlation coefficients and *P*-values for correlations between sediment/nutrient yield and percentage of improved grassland and tilled land.

Determinand	Year	Pearson's *R*	*P*-value
SS	2010	0.614	0.105
	2011	0.825	0.012*
TP	2010	0.385	0.347
	2011	0.882	0.004*
SRP	2010	0.504	0.203
	2011	0.814	0.014*
NO_3	2010	0.811	0.008*
	2011	0.88	0.002*

*significant correlations ($P<0.05$)

The export of SS, TP, SRP and NO_3 is closely linked to precipitation in the upper Eden catchment. Figure 4.26 *a* and *b* shows the relationship between total monthly precipitation and SS and NO_3 load, respectively. There is a significant positive correlation for all water quality determinands (Table 4.21). Thus, in order to accurately predict sediment and nutrient exports a good hydrological/land use based index is vital. Export coefficients are examined below.

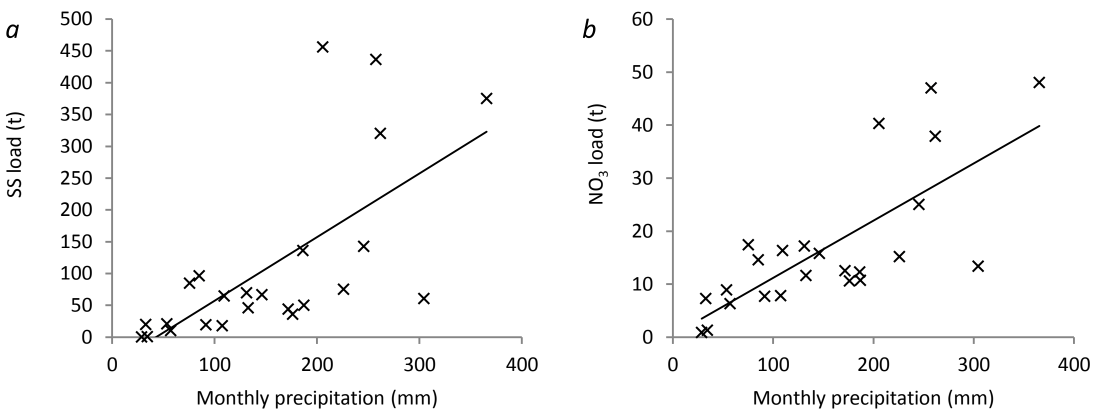


Figure 4.26: Correlation between *a*) SS; *b*) NO_3 load and total monthly precipitation.

Table 4.21: Pearson correlation coefficients and *P*-values for the relationship between sediment/nutrient load and total monthly precipitation.

Determinand	Pearson's *R*	*P*-value
SS	0.666	<0.001
TP	0.725	<0.001
SRP	0.734	<0.001
NO_3	0.743	<0.001

4.3.6 Reconciliation with published export coefficients

Suspended sediment, TP, SRP and NO_3 loads have been calculated for thirteen sub-catchments in the upper Eden catchment. Correlation analysis has indicated that there is a general trend of increasing sediment/nutrient yield with increasing percentage of improved agricultural land in a catchment, although this effect is not applicable across all sites. The following section attempts to reconcile these loads with those calculated using simple export coefficient models.

Table 4.22 contains a summary of diffuse pollutant loads calculated from grab samples collected in this study (2010 and 2011 observed loads - see section 4.3.4) and loads predicted using Equation 4.5. Suspended sediment predictions are within an acceptable range (general mixture of over- and under-predictions) but much closer to 2011 values than 2010 ones. The method over-predicts SS loads for Gais Gill and Low Hall and under-predicts the 2011 SS load in Blind Beck, Kirkby Stephen and Gt. Musgrave. The prediction for Appleby is within 10% of the 2011 'actual' load.

The majority of TP predictions are within the ranges calculated for 2010 and 2011, although they are in much more agreement with 2011 values. The model under-predicts for Gais Gill, Upland Eden, Swindale Beck, Blind Beck and most significantly so for Kirkby Stephen; and over-predicts for Scandal Beck, River Belah, Helm Beck and Low Hall. The catchment outfall at Appleby is also over predicted (compared to 2011) but by 15% only. Predictions for Coupland Beck and Great Musgrave are in very close agreement. Total N predictions are within the same magnitude with a mixture of over- and under-predictions, although within an acceptable range for the majority of sites. However, the River Eden at Gt. Musgrave and Appleby are significantly over-predicted, while Low Hall, Blind Beck and Helm Beck are all under-predicted.

Table 4.22: Observed (2010 and 2011) and predicted SS, TP and TN loads.

	SS load (t)			TP load (kg)			TN* load (kg)		
	2010	**2011**	**Predicted**	**2010**	**2011**	**Predicted**	**2010**	**2011**	**Predicted**
Upland Eden	325	821	955	1288	2755	2297	12081	21452	39600
Eden at Kirkby Stephen	727	1921	1475	2387	5082	4156	33035	60990	65700
Gais Gill	3.1	8.5	16.5	18	36	11	497	790	440
Scandal Beck at Smardale	174	577	760	695	1585	2019	16145	26939	32500
Scandal Beck at Soulby	186	596	860	846	1863	2494	17500	29265	38750
River Belah	392	1574	1070	756	2483	2571	11674	33352	45200
Swindale Beck	205	529	675	1059	2246	1913	12871	23055	29600
Eden at Great Musgrave	2832	6356	5160	8669	16966	16903	109155	177665	245800
Low Hall stream	2.9	9.2	40	20	54	189	1806	3048	2175
Blind Beck	79	318	230	345	1076	759	11494	16777	12250
Helm Beck	73	402	445	337	1239	1525	8265	23348	2210
Coupland Beck	125	312	522	544	1138	1136	8400	14519	20500
Eden at Appleby	3206	7615	7755	10353	20920	24769	157974	253242	372800

*Nitrate converted to total nitrogen by applying a factor of 14/62.

To add a hydrological component to the calculation of catchment loads, and to disaggregate them into monthly totals, Equation 4.6 was used. Fixed baseflow pollutant concentrations for each site were defined by the concentration data collected in section 4.3.2. The effect of adding in a discharge and baseflow component to the equation (which acts as a multiplier of the *L* value calculated by Equation 4.6) was to increase the loads, compared with those calculated using Equation 4.5. On average, SS and TP predictions were increased by 5-10% but NO^3 predictions significantly more so. The greatest increase was in 2011.

Predicted monthly loads (2010 and 2011) for Kirkby Stephen are compared with observed loads calculated in section 4.3.4 (Figure 4.27). For SS and TP the model appears to over-predict loads in the winter months (November – February) in 2010 and under-predict them in 2011. Over predictions occur in the majority of months compared to 2010 load values (SS and TP), which is accountable to either measured loads being too low or the model not being able to match the true values, or both. Total N predictions are very high in comparison to observed loads for both years, particularly in winter months, perhaps overemphasizing the influence of increase rainfall and runoff.

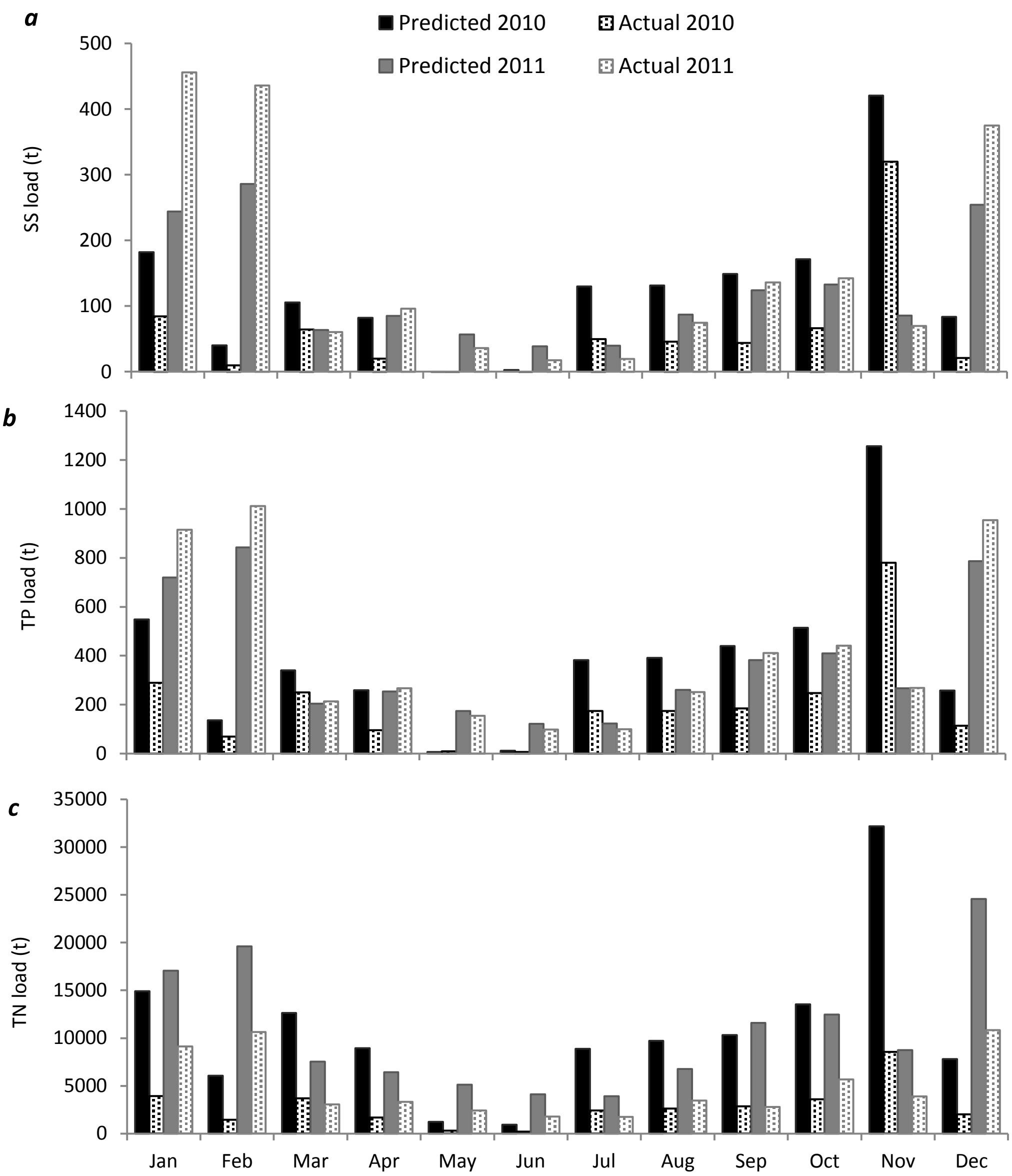


Figure 4.27: Monthly observed and predicted *a*) SS; *b*) TP, and *c*) TN loads for the River Eden at Kirkby Stephen.

4.3.6.1 Use of export coefficient to predict unaccounted for losses

In Section 4.3.4 a significant discrepancy was identified between the sum of the total yields (of all determinands) from the headwater sub-catchments (Scandal Beck, River Belah and Swindale Beck) and the increase in loads between Kirkby Stephen and Great Musgrave. A similar disparity was also reported by Vogel (2003), who used a mixing equation to determine SS and TP concentrations at Gt. Musgrave, only to find that they were significantly lower than the observed concentrations. Bathurst *et al.* (2005) described how during low flow occasions in summer 2004, the combined average SS input of the four headwater catchments (including Kirkby Stephen) was similar to the average output from Great Musgrave. However, for a (single) high flow event, the Great Musgrave output was three times greater than the combined headwater input. Figure 4.28 depicts a map of the area in question and highlights a 25 km^2 lowland zone that is not part of the three monitored headwater sub-catchments. The discrepancy in load is illustrated by Figure 4.29, which depicts the annual SS budget for the study catchment, based on 2011 data, where 'other sources' account for the largest SS contribution between Kirkby Stephen and Great Musgrave.

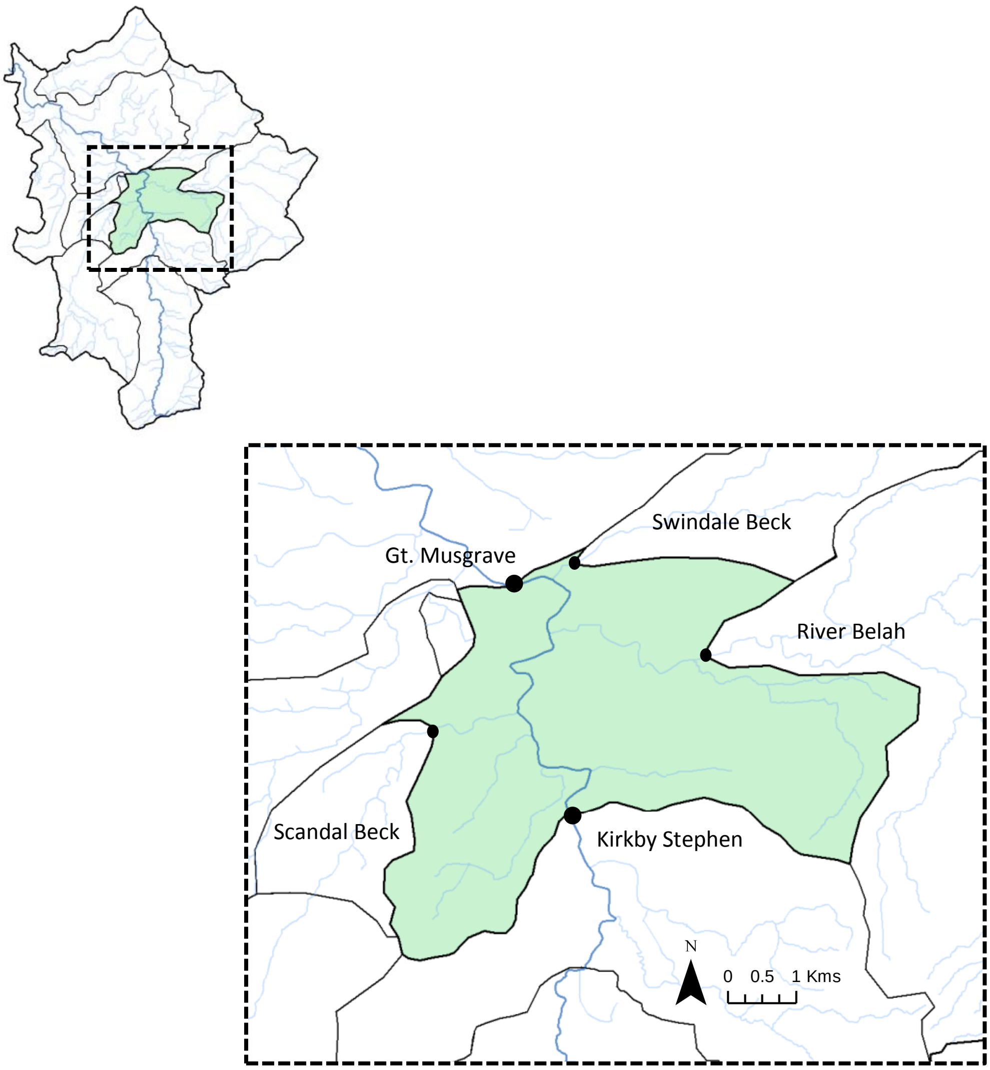


Figure 4.28: Map indicating the 25 km^2 lowland area between Kirkby Stephen and Gt. Musgrave unaccounted for by monitored upland sub-catchments.

According to Figure 4.1 this 25 km^2 area contains 74% improved grassland, 15.5% tilled land, 10% unimproved grassland and 0.5% urban and rural development. Equation 4.7 was used to see if the simple export coefficient model could account for the SS, TP and TN loads unaccounted for by the headwater sub-catchments (Table 4.23). The model only predicts 55% of the unaccounted for load in 2010 and 41% in 2011, while TP predictions are improved: 81% and 55%, respectively. Conversely, NO_3 is over-predicted by the model: 115% of the unaccounted for load in 2010 and 126% in 2011.

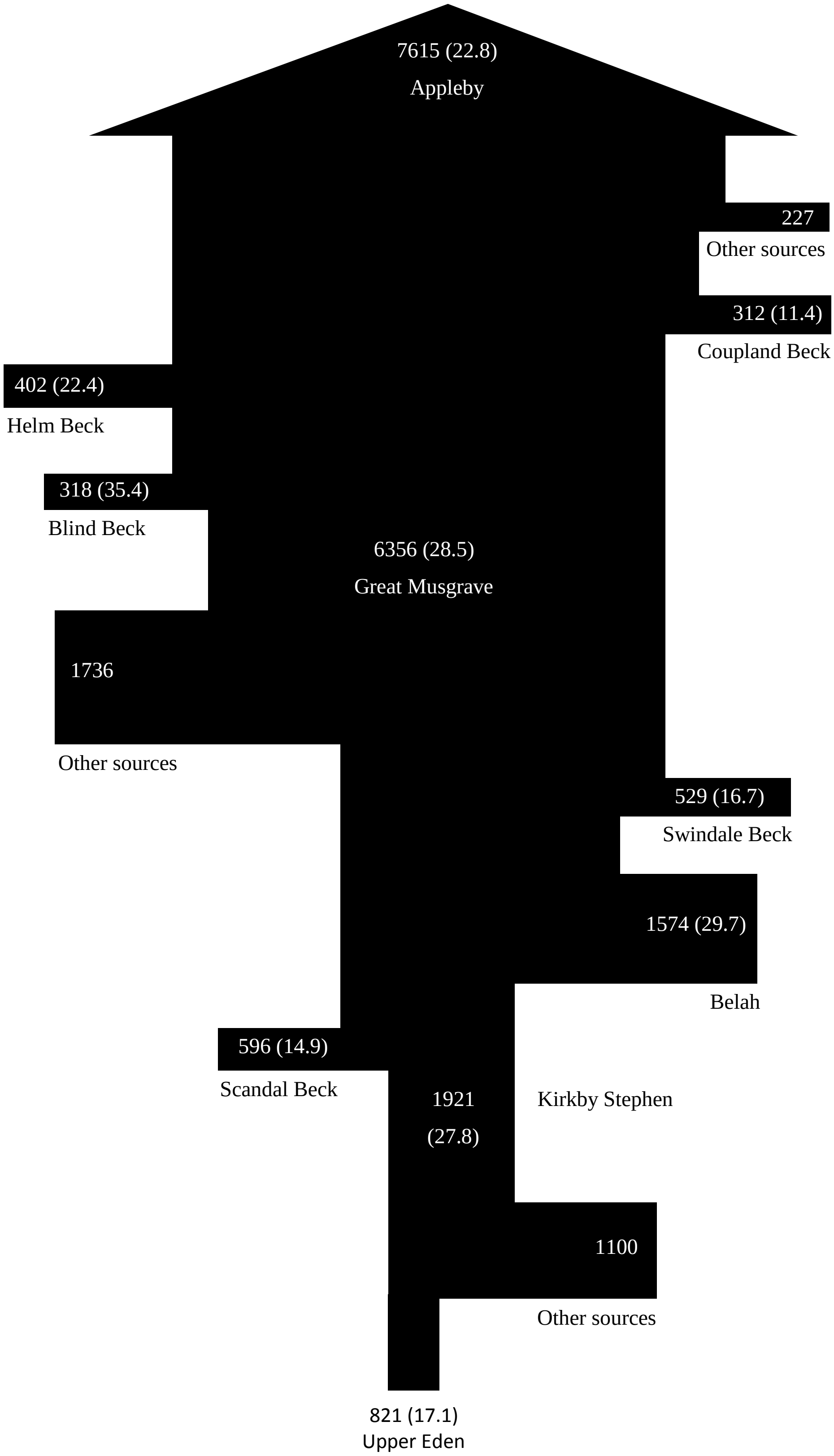


Figure 4.29: Suspended sediment budget schematic for the upper Eden catchment based on 2011 data where bar width is proportional to the magnitude of the sub-catchment's annual SS load. Numbers indicate total annual load (t yr^{-1}) and numbers in parentheses indicate the specific yield (t km^{-2} yr^{-1}).

Table 4.23: Predicted load for 25 km^2 of land between Kirkby Stephen and Gt. Musgrave.

Determinand	Load predicted using export coefficient model	Load unaccounted for by sub-catchments		Percentage of unaccounted load predicted by model	
		2010	2011	2010	2011
SS (t)	730	1322	1786	55	41
TP (kg)	2922	3621	5292	81	55
TN (kg)	39100	34075	31003	115	126

4.4 Discussion of findings

4.4.1 Discharge and precipitation

The upper River Eden shows large annual variability in pollutant concentrations and yields, which appear to be attributable to highly variable precipitation. Data collected in this study show that the average monthly precipitation patterns described by Wilkinson (2009) are not always withheld. The second lowest monthly total in 2010 occurred in December while the third and fourth highest totals in 2011 were recorded in August and May, respectively. Rainfall totals recorded for 2010 and 2011 are dryer and wetter, respectively, than the long-term average and this is reflected in the annual FDCs. A statistically significant correlation exists between total monthly rainfall and yield for all measured water quality determinands and generally a positive correlation between discharge and pollutant concentration.

Annual runoff ratios calculated in this study are circa 0.7 for the majority of catchments. It is expected to find between 0.7 and 0.8 for upland peat land catchments in the UK (Holden and Burt, 2003), while Ward (1981) reported ratios between 0.5 and 0.75, in general, for catchments in northwest England. Data collected in this study showed that rainfall-runoff ratios can be very high in some storm events (up to 100%) in the Eden catchment - highest in the winter and particularly in smaller, upland sub-catchments, but also in larger catchments (Gt. Musgrave and Appleby) during long-duration precipitation events.

4.4.2 Sediment and nutrients

4.4.2.1 Sediment

Mean sub-catchment SS concentrations measured in this study ranged between 1.8 (Gais Gill) and 6.2 mg l^{-1} (Blind Beck). The average concentration for the other non-nested sub-catchments was circa 5 mg l^{-1}, which may reflect their closeness in geographical location, land cover and land use. This concentration is within the range quoted by Walling and Webb (1987) for other UK catchments and by Bronsdon and Naden (2000) for sites on the Rivers Tweed and

Teviot (6.6 to 8.0 mg l^{-1}). After Blind Beck, Helm Beck has the second highest mean SS concentration of the sub-catchment sites, albeit by a small amount. This is a reflection of having a larger proportion of lowland area, therefore an increased likelihood of agricultural activity (e.g., greater livestock density - Bathurst *et al.*, 2005), which increases the existence of both sources and pathways of diffuse pollutants in the catchment. Coupland Beck and Scandal Beck exhibited relatively low mean SS concentrations, possibly due to their low population density and high proportion of upland area. As the SS concentrations in Scandal Beck are notably lower than in the Eden at Gt. Musgrave, Scandal Beck is not the main source of the SS in the Eden.

Using Kirkby Stephen as a representative, SS concentrations in excess of 25 mg l^{-1} (the Guideline Standard for SS in the Freshwater Fish Directive; although this annual mean is not directly associated with failure of good ecological status) only occur for 4% of the time. This would suggest that levels of SS on the upper Eden are not a serious concern. Even at Blind Beck, which has the highest mean and maximum SS concentrations and highest SS yields, 25 mg SS l^{-1} is only exceeded 4.4% of the time. However, despite these relatively low values it is clear that the number of larger storms is vital to calculating the export as this is when the highest concentrations occur.

The annual SS yields measured in the upper Eden catchment ranged from 35.3 t km^{-2} yr^{-1} (Blind Beck, 2011) to just 2.3 t km^{-2} yr^{-1} (Low Hall, 2010). These values are likely to be underestimates of the true yield, at least for the smaller basins; Walling and Webb (1981) quote measured yields of up to 250 t km^2 $year^{-1}$ for small north Pennine basins. Although Labadz *et al.* (1991) presented SS yield estimates for upland catchments in the UK (areas between 42 ha and 7.7 km^2) of between 0.7 and 66 t km^{-2} yr^{-1}, and Bronsdon and Naden (2000) calculated yields (over the 3 years) of 17.3 t km^{-2} yr^{-1} for the Upper Tweed and 19.7 t km^{-2} yr^{-1} for the Teviot catchments in northeast England.

Suspended sediment yields were compared with those calculated for the upper Eden catchment by Mills (2009) (whose values represented long-term estimations); on average values in this study are lower. 2010 values were significantly lower but there was a general agreement between 2011 values and the long-term estimates. At sites where significant underestimations were apparent, this can be largely attributed to the difference in sampling methodology used in the two studies. Automatic water samplers were utilised by Mills (2009) at Gais Gill, Blind Beck and Appleby to collect water samples during high-discharge events.

This would have a profound effect on the discharge-sediment rating curves and it is somewhat unrepresentative to compare these sites directly with those that weren't subjected to the same sampling regime.

Accepting that calculated export values may be underestimates of the actual annual loads, the values represent upper and lower ends of the range in SS yield likely to be experienced in this catchment as they represent relatively dry (2010) and wet (2011) years. In most cases the highest yields come from catchments with the highest percentage of improved agricultural land (improved grassland and tilled land) and these areas tend to be located in the lowlands. This relationship was also found by Wass and Leeks (1999), who suggested that this land type has a higher erosion rate than uncultivated, or unimproved, grassland.

There is a significant correlation between SS yield and catchment area in 2010 but not in 2011, the fact that there is one in 2010 is possibly due to the low amount of precipitation that would otherwise place a stronger emphasis on the land use of the catchments through increased soil erosion and the operation of alternative flow pathways (e.g., overland flow). The data highlight temporal variations in SS yield, both on an annual and sub-annual timescale; for example the SS yield for Kirkby Stephen varied between 10.5-27.8 t km^2 yr^{-1} in successive years, with similar variability observed at all sites. At sub-annual timescales, SS transport was limited primarily to the winter months with 59% (2010) and 69% (2011) of the load being discharged during the months November-February at Kirkby Stephen.

4.4.2.2 Phosphorus

Mean SRP concentrations ranged between 0.005 mg l^{-1} (Gais Gill) and 0.029 mg l^{-1} (Blind Beck). Gais Gill had the lowest mean P concentrations as was expected due to the low population density and dominance of unimproved grassland in the catchment. Along the Scandal Beck nested system, there is a significant increase in mean TP concentration between Gais Gill and Scandal Beck (0.017–0.024 mg TP l^{-1} – t = -4.95; $p < 0.001$), possibly resulting from Smardale's position downstream of the Ravenstonedale settlement. Soluble reactive P yields at the upland Eden site and the study catchment outfall at Appleby were surprisingly similar. Naturally, it may be expected that the influence of increased urban and agricultural activity would mean that the upland Eden would have a lower yield than further downstream; this appears not to be the case.

The average SRP concentration for the non-nested sub-catchments is circa 0.015 mg l^{-1}. The SRP values in this study are relatively low compared with the long-term UK average (Table 4.24). Blind Beck, the sub-catchment with the highest P concentrations, only has TP concentrations greater than 0.1 mg l^{-1} for 5.3% of the time. The greatest mean and maximum SRP concentrations of 0.029 mg l^{-1} and 0.159 mg l^{-1}, respectively, were both recorded at Blind Beck. Low Hall has the second highest mean concentration followed by the Eden at Appleby and Helm Beck. All of these catchments are predominantly in the lowlands. Data shows that the pattern of annual P accumulation is similar to the SS. This suggests that either P moves in association with SS or flow, or both, and that times of greatest export are mainly in winter months.

Haygarth *et al*. (2012) report a TP loss of 60 kg km^{-2} yr^{-1} from a 22 ha freely draining, mixed grassland catchment but a loss of 600 kg km^{-2} yr^{-1} from 48 ha slowly permeable catchment with a mixture of grassland and arable management, both in the Southwest of England. Jarvie *et al*. (2003) report a TP loss range of 2 - 90 kg km^{-2} yr^{-1} for unimproved and improved grasslands, and Wood *et al*. (2005) measured a mean TP loss of 120 kg km^{-2} yr^{-1} for the 1242 km^2, predominantly slowly permeable, mixed agriculture Taw catchment. The overall mean TP yield of 42.4 kg km^{-2} yr^{-1} for the upper Eden catchment calculated in this study is at the lower end of this range. McGuckin *et al.* (1999) report SRP yields of 40-85 kg km^{-2} yr^{-1} (improved grassland) and 12-40 kg km^{-2} yr^{-1} (unimproved grassland), thus the overall mean SRP yield of 14.5 kg km^{-2} yr^{-1} for the upper Eden catchment also falls at the lower end of the range.

Table 4.24: UK annual average concentrations of nitrate and orthophosphate (SRP) by landscape type 1980 to 2011 (from data.gov.uk: http://www.defra.gov.uk/statistics/environment/inland-water/).

Land type	Mean concentration (mg l^{-1})	
	SRP	NO_3
Lowland arable	0.434	23.58
Lowland pastoral	0.298	15.44
Uplands	0.053	5.68

4.4.2.3 Nitrate

Mean NO_3 concentrations in the upper Eden ranged from 0.67 (Gais Gill) to 14.13 mg l^{-1} (Low Hall) with the range of maximum concentrations varying from 3.55 (Upper Eden) and 4.81 (Gais Gill) to 24.5 mg l^{-1} (Low Hall). Compared with the long-term average data from the UK (Table 4.24), concentrations in the Eden are relatively low, with the exception of the Low Hall and the Blind Beck catchments, which are in a similar range to the value for *lowland pastoral*. Nitrate loads for the upper Eden catchment ranged from 251 – 2447 kg km^{-2} yr^{-1} for the upper Eden and Low Hall, respectively. Blind Beck had relatively high yields compared with the other sub-catchments (based on 2011 data); 1866 kg km^{-2} yr^{-1} compared with around 600 – 700 kg km^{-2} yr^{-1}, with the exception of Helm Beck that had a yield of 1296 kg km^2 yr^{-1}. The high yield from Low Hall explains why Blind Beck has a yield greater than the other non-nested sub-catchment. Analysis of major and minor ions and trace elements in the Low Hall and Blind Beck sub-catchments by Ockenden (2010) showed that concentrations of most ions were higher by 7-26% in the former, but that NO_3 was approximately twice as high. It was hypothesised that the Low Hall stream has an input of water from the sandstone aquifer in the bottom of the Eden Valley, which is reported to have a rising NO_3 concentration (Butcher *et al.*, 2006). Although NO_3 export is clearly driven by discharge events, when compared with annual accumulation of SS and P, the accumulation is 'smoother' with less distinct responses to high flow events. This is due to the weak correlation between discharge and NO_3 concentration. Despite NO_3 concentrations not significantly increasing as a result of high flow (or in some instances decreasing), the load increases simply due to the greater volume of water, probably tied to the leachate and/or deeper runoff pathways (e.g., Tesoriero *et al*. (2009)).

In order to contextualise the NO_3 yields they have been converted to TN, as this is how the majority of studies report N exports. Total N yield values from the literature ranged from 200 - 8000 kg km^{-2} yr^{-1} with means ranging between 500 and 1650 kg km^{-2} yr^{-1}, although these values are all from the USA and are for improved pasture, arable and mixed agriculture. Kyllmar *et al.* (2006) reported TN yields ranging between 200 and 4100 kg km^{-2} yr^{-1} from a study in Sweden covering a wide extent of catchment scales and land uses. These values largely agree with those found in this study.

4.4.3 Spatial and temporal variability

Sediment and nutrient data collected in this study exhibit high spatial variability in both concentrations and yields. However, the data show that no relationship exists between SS, P and NO_3 yield and catchment area. Strong positive correlations between percentage improved agriculture and yield in 2011 suggest that land use has a stronger influence over yield in wetter conditions and this compliments the findings from the yield–catchment area correlation analysis. Conversely, in relatively dry years the lack of rainfall/runoff may mean that sediment and nutrients are not being transferred to the river system as they would in a wet year, assuming that the agricultural activity (pollutant source) was the same. It could be argued that the larger River Eden catchments have a heterogeneous mixture of land uses therefore any obvious trend is lost (above a certain spatial scale). The Low Hall catchment also opposes the general trend as despite consisting of 100% improved agricultural land it only exhibits a high NO_3 yield, while the other determinands are relatively low.

The greater variability in exports is associated with the non-nested sub-catchments. Blind Beck has a relatively high yield for all determinands. The two smallest sub-catchments, Gais Gill and

Low Hall, both have correspondingly low yields of SS and P; however, Low Hall has the highest NO_3 yield of all the study sites. This indicates that the greater heterogeneity of landscape characteristics in larger catchments has a buffering or 'averaging' effect on the yields. Smaller catchments are likely to be more homogeneous, therefore have greater potential to produce extreme values of sediment/nutrient yield. Larger catchments have larger annual loads but tend to exhibit lower specific yields compared with small catchments. This is attributed to the increasing input of 'clean' water from low intensity agriculture headwater sub-catchments and/or increasing deposition of sediment and assimilation of nutrients in the lowlands.

4.4.3.1 Dilution

The majority of sub-catchments in the upper Eden have a significant upland zone meaning that they are more likely to exhibit lower sediment and nutrient concentrations/loads due to the dilution effect of the clean water originating from these parts. If this source of 'clean' water is absent then the dilution effect is reduced, resulting in higher pollutant yields. Does this mean that the main Eden gets polluted downstream, or are the problems contained within specific sub-catchments, such as Blind Beck? The data collected in this study suggest that the upper Eden catchment has a potential chronic pollution problem but is mitigated by the runoff originating from the uplands. However, this system is vulnerable to low rainfall as it would equate in reduced dilution.

Wood *et al.* (2005) report a similar effect on P in the Taw catchment, UK, where it was concluded that although dilution meant that P impacts on aquatic ecology were not of immediate importance, in neighbouring catchments where there was no dilution from upland headwaters, algal blooms resulting from eutrophication had been observed. In the upper Eden, Mannix (2005) suggests that groundwater NO_3 concentrations in the lowland portion of the catchment were significant to the point that they could cause eutrophication in the main river under low flow conditions. This idea was supported in summer 2005 when significant algal growth occurred in the River Eden following a prolonged dry spell (Plate 4.1) and gives insight into what may happen if dryer/warmer summers become more common as a result of climate change, and/or the quality of water from the uplands is compromised, perhaps due to agricultural intensification.

Plate 4.1: The River Eden at Gt. Musgrave with significant algal growth (date: 22/07/2005 - source: Mannix, 2005).

4.4.3.2 Seasonal trends

Seasonal trends are exhibited by SS and P concentration with a less obvious pattern observed for NO_3. Higher SS concentrations generally occur during the autumn as erosion rates increase due to increased precipitation and also because of first flushing of any material deposited in-steam during summer low flows. There is also evidence of sediment exhaustion in contiguous storms (this is discussed in more detail for Blind Beck in the subsequent chapter). A similar pattern is observed for TP, probably due to its propensity to move in association with fine-sediment, and also SRP. This is perhaps more unusual for SRP as higher discharges (most often in the winter) can lead to a reduction in concentration due to the dilution, especially if the origin of SRP is a point source (e.g., a sewage outflow, farm slurry store). However, considering that the majority of the upper Eden catchment has relatively low urbanisation and low intensity agricultural activity, there are relatively few potential point sources.

In many agricultural catchments, concentrations are often greatest in early winter, reflecting the first flushes of water from farm land, but this is not evident in the upper Eden (there is a slight increase in winter and early spring but the range is only between 2 and 5 mg NO_3 l^{-1}). Nitrate concentration exhibits little seasonal variation, as its movement is associated with more continuous base flow. Also sharp rises in NO_3 concentrations are sometimes seen in spring reflecting applications of fertiliser but this effect is not immediately clear in this instance either, possibly due to low nutrient loading in the catchment.

4.4.4 Sources of uncertainty

Catchments that were ungauged or suffered data loss due to equipment loss/failure had discharge records derived from a surrogate catchment. This is assumed to provide an acceptable estimate of the discharge volume but doesn't necessarily take account of spatio-temporal variation due to inconsistency in rainfall distribution across the entire catchment.

The limitations of using a grab sample campaign to estimate sediment/nutrient annual exports has been discussed earlier in this chapter but perhaps the main source of error is from the sediment/nutrient rating curves. Errors are likely to be greater where the curve is based on fewer samples and where the curve is extrapolated to predict concentrations for high discharges. Due to relatively low numbers of samples collected at high discharges it is highly likely that the loads/yields presented in this chapter underestimate the true values. The use of rating curves also hides any hysteretic effects; limb analysis demonstrated that concentrations of SS were generally higher on the rising limb and NO_3 was higher on the falling limb, but no significant effect was identified. No clear pattern was observed for TP or SRP.

Despite these methodological limitations, the difference within the results has significance beyond this, and the data are sufficient to characterise the sediment and nutrient regime of the upper Eden catchment in order to identify ailing sub-catchments. Relationships between DWPA flux and land use, and the reliance on storm events for contaminant transfer, have also been determined. Automatic water samples are utilised in Chapter 5 (alongside grab samples) to allow the collection of water samples during high-flow events; the effect of this on the calculation of annual determinand exports and on hysteresis will be investigated.

4.4.5 Export coefficients

The basic export coefficient model shows potential for predicting sediment and nutrient loads even in catchments with mixed land uses, especially when taking into account its simplicity. The selection of export coefficients for each land cover/use is crucial to the performance of the model but generally it predicted SS, TP and TN exports that lie between measured 2010 and 2011 values, which is acceptable as these years are dryer and wetter than average, respectively. The use of the basic model could be considered sufficiently accurate for general management in the upper Eden catchment as it captures the upland/lowland split (based on land use) and the dilution effects of scale. The model performs well for the study catchment outlet at Appleby (the largest catchment area measured), which suggests there is a need for an areal distribution of land use for a good export estimate.

When using the extra hydrological component, the model is not very sensitive to hydrological extremes (i.e., wet and dry months) and the difference between monthly SS and TP loads in 2010 (dry) and 2011 (wet) are much greater for measured values compared with those predicted using export coefficients. Moreover, the model over-predicts in dry periods and under-predicts in wet ones – this effect is more prominent in the winter months. Total N predictions are significantly increased using the extra baseflow component and consistently over-predict the measured loads as a result. Overall, export coefficient modelling offers a reasonable method for general farming intensity but lacks local detail; however, this is inherent of the method. The Eden may be very atypical due to density of dairy cows and slurry (on heavy soils), and some pristine, non-agricultural land in a complex mix. The Low Hall sub-catchment, which has 100% improved agricultural land, exhibits very low SS and TP yields but the highest TN export and could be considered a local phenomenon in the Eden. The Low Hall catchment is considered in more detail in the following chapter.

In a practical application where the model was used to predict sediment/nutrient losses from a 25 km^2 area of lowland (with relatively high livestock density) between Kirkby Stephen and Gt. Musgrave, predicted SS and TP exports were low while TN was slightly high, compared with the actual discrepancy derived from the measured data. It is hypothecated that at low flows there is little net supply or deposition along the reach but during higher transport events the middle reaches were contributing a proportionally higher SS and P load, with possible sources being ditches and minor streams, bank erosion and the channel bed. Thence, in a risk-based system precipitation, runoff and land use is a good start but representation of what happens within a storm event is vital to accurately predict losses. There is a need for the export coefficient model to include a flow pathway component, with both baseflow and storm events represented.

4.5 Summary

The principle aim of this chapter was to gain understanding of the spatio-temporal patterns of sediment and nutrient flux as a means of focusing future mitigation efforts in the upper Eden catchment. The process of quantifying SS, P and NO_3 yields of sub-catchments of varying spatial area has been described, including the calculations used to estimate yields. Final estimates are shown and potential sources of error acknowledged, as accurate quantification is highly dependent on the quality of the sampling regime.

Calculated sediment and nutrient yields are likely to be underestimated due to lack of water samples collected during high-discharge events. A small degree of scatter exhibited in the determinand concentration-discharge rating curves is attributed to seasonal and limb effects but the remaining unaccounted for scatter serves to demonstrate the complexity of the sediment/nutrient supply and delivery process operating in the catchment. All determinand yields did not show a clear trend with catchment area suggesting that the influence of other variables is more important. A number of relationships between sediment/nutrient yield and rainfall amount, and land use were found, but are all subject to localised noise. Thus, the general methodology is deemed sufficient to broadly characterise the upper Eden catchment but has important limitations considering its intended purpose.

Comparison of calculated annual yields with published sediment/nutrient export coefficients showed good general agreement, whereby the model was capable of representing different agricultural land uses with sufficient accuracy. However, the export coefficient model was less reliable on a monthly basis where extremes in rainfall, both wet and dry, occurred - particularly during the winter.

The data collected in this chapter demonstrate how many local factors in space and time dominate the actual export rates. Firstly, there is a need to target intense agricultural sub-catchments, such as Blind Beck. However, it has identified the need for a more fundamental insight into the cause of sediment/nutrient loss in order to target mitigation efforts. The measurement of wet and dry years indicated that a better representation of the influence of storm hydrology is required, including the number of storms, their size and timing, the operation of different flow pathways, and antecedent conditions. This necessity forms the basis of the subsequent chapter, which will focus on the Blind Beck sub-catchment and employ a spatial-intensive sampling regime and storm event monitoring.

5. Blind Beck sub-catchment study

5.1 Introduction

Results presented in Chapter 4 demonstrate that Blind Beck has consistently higher nutrient and sediment concentrations per unit runoff as well as higher total yields per unit area than any other monitored sub-catchment in the upper Eden. However, the results did not help determine where or indeed what the pollutant source-pathways are within the catchment. This knowledge is crucial in ensuring that efforts to mitigate sediment and nutrient pollution are administered effectively.

Since eutrophication (chiefly P) and excessive fine sediment are the main pollution pressures affecting water quality in the Upper Eden, Blind Beck was selected for 1) a detailed evaluation of nutrient and sediment regimes and 2) as a case study catchment within which to carry out mitigation experiments. The former comprises the increase of grab sampling locations along the river system (as opposed to just the catchment outlet) and also the use of automatic water samplers to collect samples during storm events. These data will allow the identification of pollutant source areas within the catchment, as well as the examination of sediment and nutrient behaviour during storm events and the importance of high-flows in the determination of annual fluxes.

While it is prescribed under the WFD that rivers are best managed (in terms of water quality, flooding, etc.) at the catchment scale, it can be argued that the farm is the most logical unit for the administration of actions to tackle DWPA. One reason is that the landowner could potentially be held responsible for the quality of water leaving their farmed areas, if an issue is identified. Previous studies (Gravier, 2004; Barber, 2008; Mills, 2009) have highlighted one particular farm in the Blind Beck catchment - Sykeside Farm (located at the downstream end of the catchment) as being not only a potential source of sediment/nutrients, but also a suitable location to deploy a suite of RAFs in an attempt to mitigate DWPA for the entire 9 km^2 catchment.

A well-established working relationship with the farmer at Sykeside Farm existed from previous work in the area (the CHASM project). The farmer granted permission for the construction of a number of RAFs on the grounds that they could be integrated into a HLS funding scheme. The RAFs were to deliver water quality benefits as well as flood mitigation and ecological gains. However, following the part-completion of a wetland RAF all future HLS funding in England was temporarily withheld by Natural England (in 2010) and the full mitigation plan could not be executed.

As a result, the decision was made to transfer the mitigation trials to other existing instrumented study catchments in northeast England. The results of these experiments are presented in Chapters 6 and 7.

In the absence of experimental testing of mitigation features, the objectives of this chapter are threefold:

1. Investigate temporal patterns and source-pathways of sediment and nutrients using a spatially high-resolution grab sampling campaign.
2. Examine the influence of different water quality monitoring regimes (i.e., event sampling vs. non-event sampling) on the estimation of annual sediment/nutrient loads.
3. Evaluate the success of a single, large-scale installed mitigation feature.

5.2 Methodology

5.2.1 The study area

Although described with the rest of the upper Eden catchment (see Chapter 3), the Blind Beck catchment will be considered here in more detail. The location of Blind Beck in relation to the wider study catchment is depicted in Figure 5.1. Blind Beck rises on a limestone pavement situated on the fells to the southwest of Crosby Garrett (population circa 120); catchment elevation ranges between 142-376 m AOD (Figure 5.2). The catchment is situated predominantly in the lowlands with only a small proportion accounted for by the fells (uplands) to the southwest. The Beck flows for approximately 8 km in a north-easterly direction and joins the River Eden at Little Musgrave (OS grid reference: NY375513), just downstream of the EA river gauging station at Great Musgrave. The Beck falls approximately 140 m in the first kilometre but then just 90 m between Crosby Garrett and the confluence with the River Eden.

The Beck has two main tributaries: Wygill Beck (C.A. 1.1 km^2), which joins from the west approximately mid-way along its length; and Low Hall stream (C.A. 1.25 km^2), which drains part of the lowlands to the north of the catchment and enters the main Beck just before the outlet monitoring site. Blind Beck has a catchment area of 9 km^2 with approximately 7 km^2 lying upstream of Sykeside Farm. A railway embankment bisects the southwest third of the catchment (see Figure 5.1).

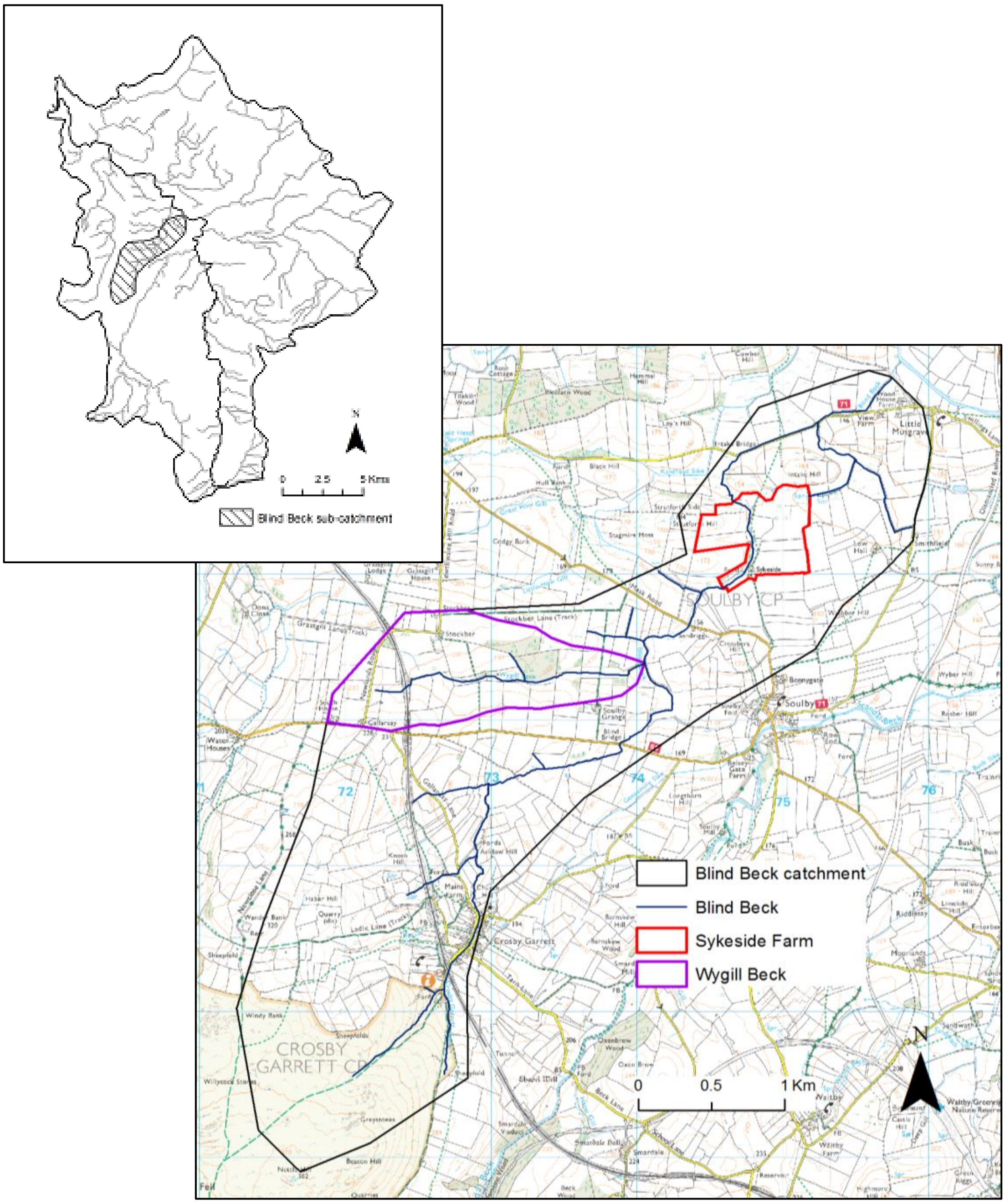


Figure 5.1: Blind Beck catchment map, also showing the extent of Sykeside Farm (source: Ordinance Survey).

5.2.1.1 Geology and soils

Solid geology consists of *Carboniferous Limestone* in the headwaters and *Permian Penrith Sandstone* further downstream (see Appendix F for geology and drift geology maps). The Vale of Eden was the location of a major ice flow in the late Devension (Mitchell and Clark, 1994); there is a significant thickness (10-20 m) of quaternary drift along the valley floor, with hummocky moraines and drumlins forming a mix of diamicts, clays, sands and gravels (Younger and Milne, 1997).

Soils are described below following the path of the river (from upstream to downstream) and are depicted in Figure 5.3. *Crwbin Association* soils, shallow and well drained, overlay the limestone pavement. The Beck then flows over loamy and fine silty soils formed over sandstone of the *Eardiston 1 Association* in the area around Crosby Garrett; this is followed by an area of *Wick 1 Association* soils: deep, well drained coarse loamy and sandy soils formed in coarse textured glacio-fluvial drift (Jarvis *et al.*, 1984).

After about 3 km the river begins to flow around the southern perimeter of a drumlin, which divides the soil type on each side of the river. To the west, overlying the flanks of the drumlin is the *Clifton Association* – a seasonally waterlogged soil developed in slowly permeable fine loamy till and thin overlying glaciofluvial deposits. To the east of the present channel *Wharfe Association* - mainly deep, well-drained, fine loamy alluvial soils are recorded on the level floodplain. The sub-catchment of the Wygill Beck tributary which joins from the north consists mainly of *Brickfield 3 Association*: slowly permeable seasonally waterlogged fine loamy over clayey soils formed in glacial till, and *Clifton Association* soils (Jarvis *et al.*, 1984).

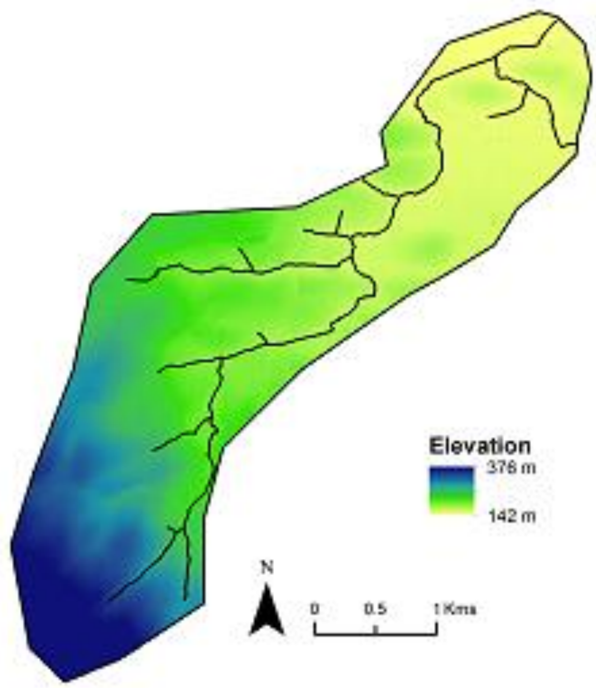


Figure 5.2: Blind Beck catchment DEM.

5.2.1.2 Land cover/use

Figure 5.4 shows land cover of the catchment, based on the CEH LCM2000, and summarised in Table 5.1. However, catchment walkovers revealed some inconsistency in the LCM2000 map, particularly the amount of tilled land (which is over-estimated), especially around Wygill Beck. An updated detailed land use map was created based on real time observations (Figure 5.5). Tilled land includes any field parcel that was ploughed during the study period.

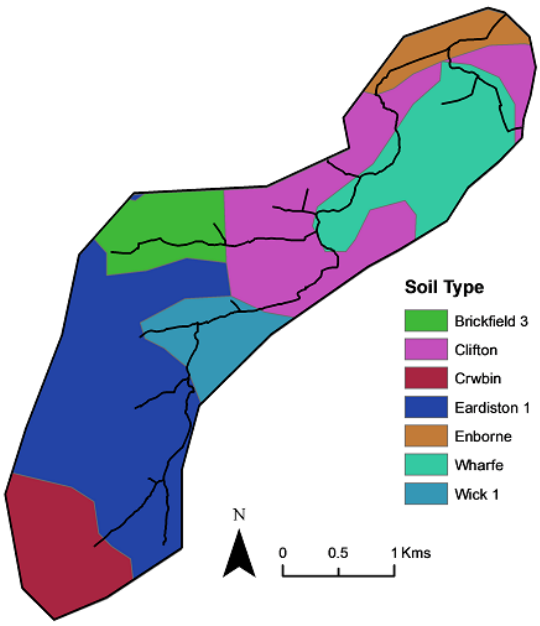


Figure 5.3: Blind Beck catchment Soil Association map.

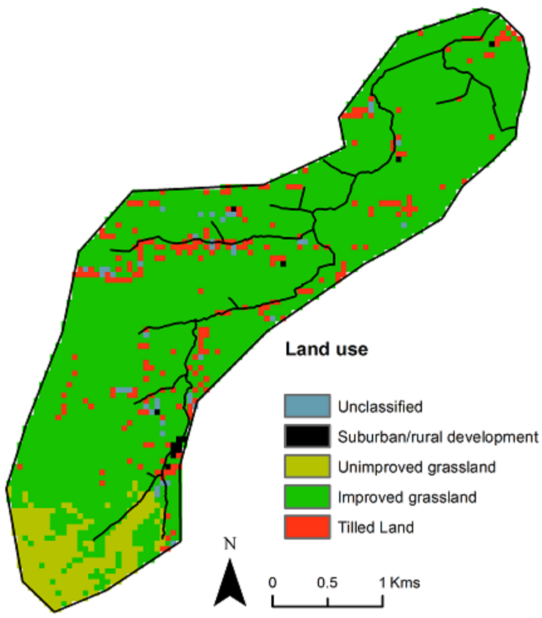


Figure 5.4: Blind Beck catchment land use map (derived from the LCM2000; CEH, 2011).

Table 5.1: Blind Beck catchment land use percentage.

Land use class	Percentage cover
Unclassified	0.3
Urban and rural development, woodland	1.2
Unimproved pasture	11.2
Improved pasture	80.8
Tilled land	6.6

The upland area upstream of Crosby Garrett is moorland and is used for extensive sheep grazing. Crosby Garrett is the only settlement. There are nine farms located in the catchment:

- Two small holdings in Crosby Garrett, both sheep and beef cattle farms.
- Soulby Grange Farm, a dairy and sheep operation (located near sample site 3).

- Stockbar Farm (at the top of the Wygill Beck catchment), a small beef cattle farm.
- Bonnygate Farm (near the village of Soulby), a sheep and beef cattle farm.
- Low Hall Farm (in the Low Hall catchment), an intensive dairy operation.
- Sykeside Farm, a relatively small sheep and beef cattle farm.
- Wood House Farm and View Farm (both located in Little Musgrave - downstream of the Blind Beck monitoring site but have some sheep grazing some of the fields upstream of the outlet).

As a general rule the stocking density in the catchment increases further downstream. So to does the number of fields used for silage/hay production and areas that are periodically ploughed/reseeded. Soulby Grange and Low Hall are the most significant farms in terms of animal numbers as they are both dairy operations. Sheep and beef cattle density is also relatively high around Sykeside Farm as Bonnygate use this land for grazing. Blind Beck is not fenced and animals have free access to the river along its length.

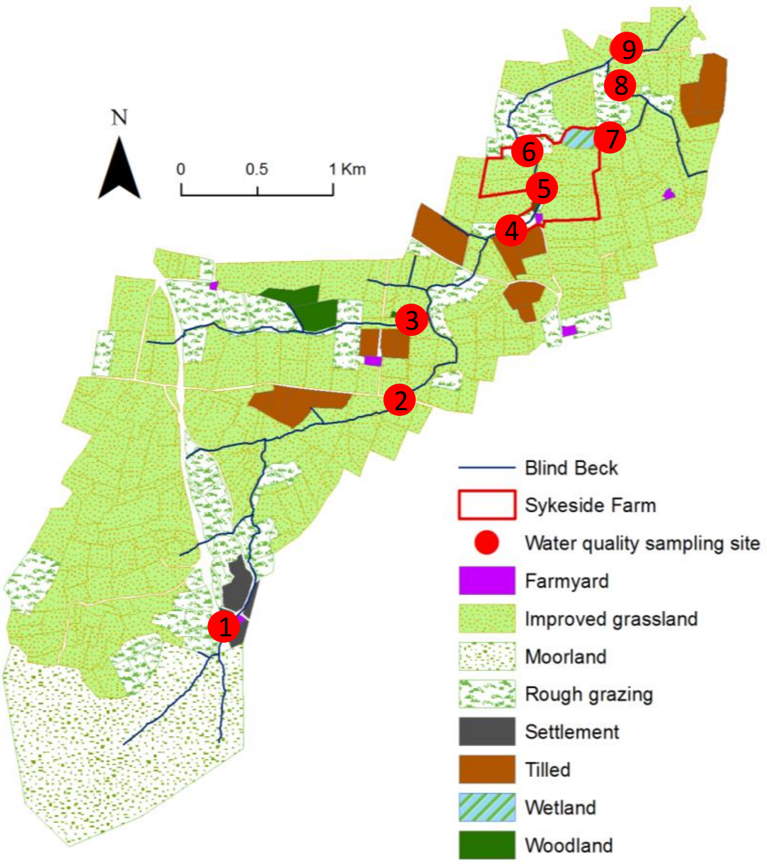


Figure 5.5: Blind Beck custom land use map from catchment walkovers. Refer to Table 5.2 for sampling location key.

Table 5.2: Blind Beck sampling location names and description.

No. on map (Figure)	Name	Catchment area (km^2)	Stream length (km)
1	BB - Crosby Garrett	1.08	0.8
2	BB - Blind Bridge	4.46	3.14
3	Wygill Beck	1.1	7.85
4	BB - Sykeside Farm in	7.05	4.93
5	BB - Downstream Sykeside Farm	7.57	5.31
6	BB - Sykeside Farm out	7.78	5.61
7	Wetland out	0.01	0
8	Low Hall stream	1.25	1.05
9	BB - out	9	6.7

5.2.1.3 Sykeside Farm

Sykeside Farm occupies 0.5 km^2, circa 5.5% of the Blind Beck catchment. The area is predominantly of low relief apart from Strutforth Hill to the west (Figure 5.6). Land cover is almost exclusively improved grassland with the exception of a small woodland area and a few rough grazing fields. One of the rough grazing fields became the site for the wetland RAF (see below). The principal land use is sheep and beef cattle grazing pasture with a small number of fields used for silage/hay production.

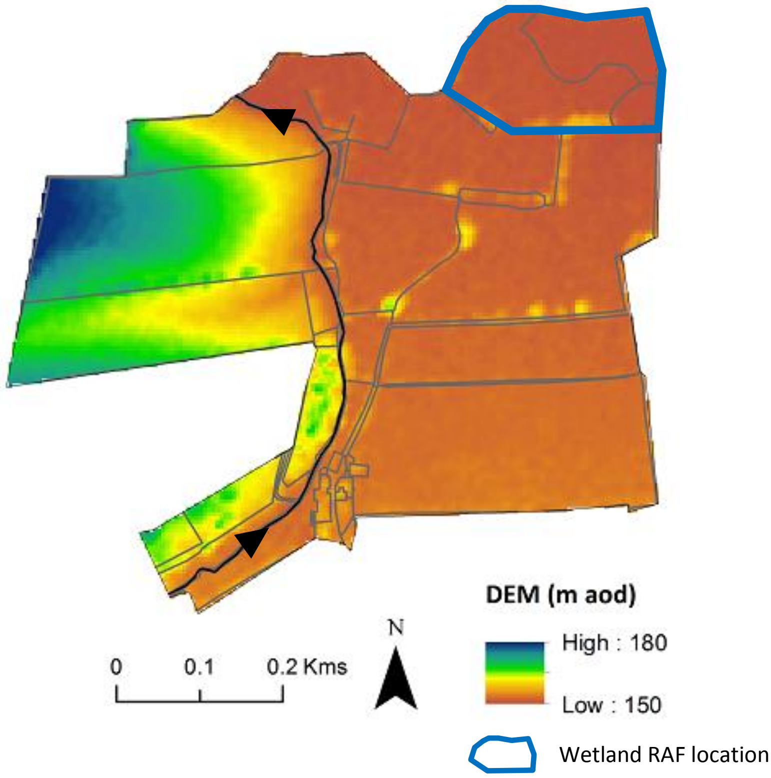


Figure 5.6: Sykeside Farm DEM (resolution 5 m) (see Figure 5.5 for location).

5.2.1.4 Monitoring a modified wetland RAF for DWPA mitigation

Prior to this study, an area of rough grazing land (circa 2 ha) on Sykeside Farm prone to surface water ponding had been selected by the Proactive group from Newcastle University and the farmer as the location to construct a wetland (Figure 5.6 for location). The final wetland design and location were not decided as part of this study but the feature was included in the monitoring campaign in order to determine its impact on the sediment and nutrient regime of Blind Beck. Wetlands are widely reported as multifaceted landscape features with numerous social, economic and environmental benefits associated with them (e.g., Mitsch and Gosselink (2007)).

The rationale behind the creation of the feature was three-fold; 1) there was a need stated by the farmer to better control the movement of water across this area of the farm (S Wharton, pers. comm.); Figure 5.7 depicts a number of issues, which include flooding of a gateway and a neighbouring farm's fields. It was believed that water from Blind Beck itself was conveyed into the area via a ditch network during high flow events and that the wetland would provide temporary water storage. 2) by temporarily storing a portion of the (high) flow from the main river the wetland would reduce concentrations of sediment and nutrients through sedimentation and other natural attenuation processes (please refer to Chapter 2.9.2), and 3) the feature would provide some flood peak temporary storage/attenuation therefore helping to reduce flood risk. Notwithstanding the potential of the wetland to deliver multiple benefits, this study is concerned with sediment and nutrient regimes and their management; therefore the feature is evaluated for water quality purposes only.

A local contractor constructed the wetland using locally sourced earth and rubble. A 1 m high (at the highest point) bund was built around the field perimeter, to the south and east, to provide approximately 3000 m^3 storage capacity. The gateway previously in the southeast corner of the field was relocated further along the fence line and the operation of the sluice was improved (see Figure 5.8). A timber dam was keyed into the ground and surrounding bund, where a v-notch weir was installed. A pressure transducer was located next to the weir to monitor water stage thus allowing the calculation of continuous (15 minute interval) discharge (the V-notch weir specification and discharge calculation equation can be found in Appendix B1). An auto-sampler was also deployed at the outfall to take water samples during high flow events; results are presented in section 5.3.6. Plate 5.1 shows a photograph of the completed wetland bund and outlet structure.

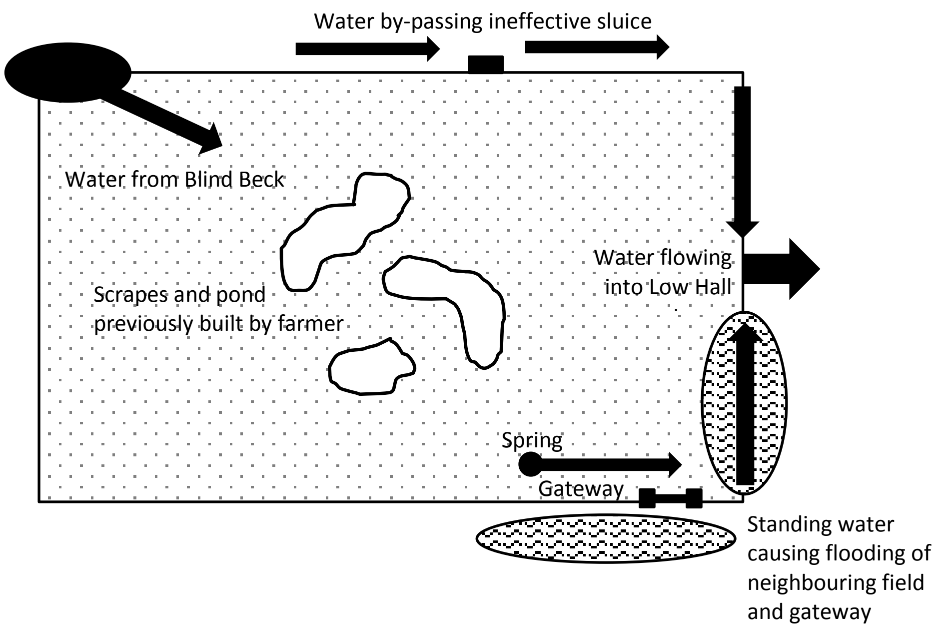


Figure 5.7: Schematic of wetland area prior to modification with issues highlighted (not to scale) (see Figure 5.6 for location).

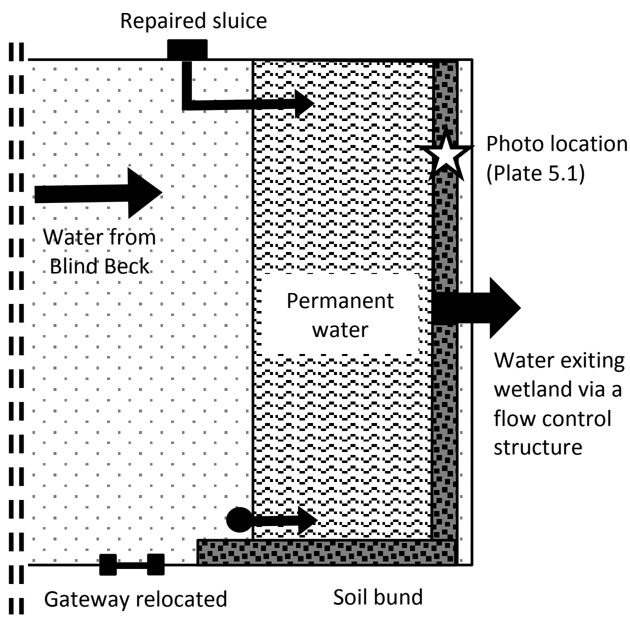


Figure 5.8: Schematic of wetland after modification (not to scale).

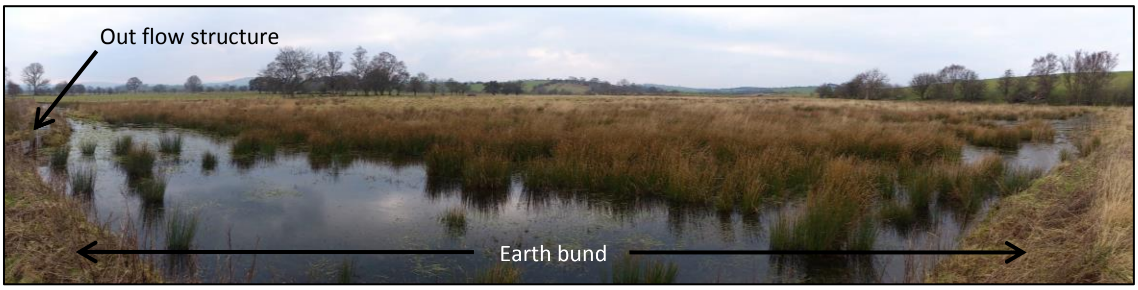


Plate 5.1: Wetland RAF after modification with the outflow structure and monitoring equipment visible on the left.

5.2.2 Hydrometeorological data collection

Figure 5.9 shows the instrumentation on Sykeside Farm, which includes a rain gauge (a second EA-operated rain gauge is located in Crosby Garrett – see Table 3.6 for locations), stream gauges, piezometers and two automatic water samplers. Catchment rainfall was calculated using Thiessen polygons and discharge using stage recorded at 15 minute intervals by a Thalimedes float and counterweight shaft encoder, and stage-discharge rating curve, as described in Chapter 3.3.1.

A transect of four shallow water table (<3 m) piezometers were installed at Sykeside Farm (see Figure 5.9) to monitor the soil water level between the stream and the wetland. Holes were sunk using a soil auger until solid gravel was reached, then lined with PVC tubing. The tubes were sealed at the bottom, 5 mm holes drilled in the side to allow water to enter the well, and the hole surrounding the top of the tube was sealed using bentonite clay. Pressure transducers were installed in each piezometer to log at 15 minute intervals.

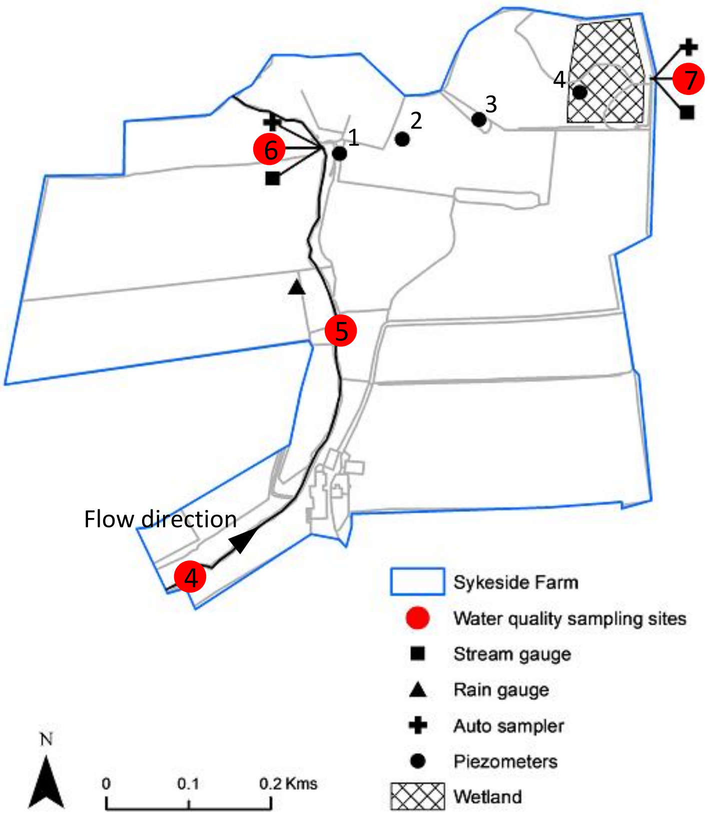


Figure 5.9: Sykeside Farm instrumentation map.

5.2.3 Sediment and nutrient concentration characterisation

5.2.3.1 Spatially intensive grab sampling

Nine grab sampling locations were selected (including the catchment outlet used in the previous chapter – previously referred to as *Blind Beck*, now noted as site number 9). Figure 5.5 shows the location of the sampling sites and Table 5.2 includes site names and additional information. Site 1 is upstream of the Crosby Garrett hamlet; site 2 is at 4.46 km^2 – so represents approximately half of the total catchment area (although a good proportion of this is on the west of the railway embankment). Site 3 is on Wygill Beck, a 1.1 km^2 sub-catchment with relatively steep valley sides to the north. Site 4 is where Blind Beck enters Sykeside farm; Sykeside Farm house and hard standings are situated alongside the river, between sites 4 and 5, with the nearest farm building less than 10 m away from the channel. Site 6 is where Blind Beck leaves the Sykeside Farm area before it turns to the north-east and flows alongside the road towards Little Musgrave. Site 7 is located at the outfall of the modified wetland feature on Sykeside Farm; this also marks the start of the Low Hall stream. Site 8 is the same as the 'Low Hall' site in the previous chapter, which drains the land owned by the Low Hall dairy farm; and site 9 is the Blind Beck catchment outfall.

Grab samples for sediment/nutrient analysis were collected on the same dates as those in Chapter 4 (see Appendix G1 for sample dates) using the same methodology outlined in Chapter 3. The laboratory methods described in Chapter 3.3.4 were used for the determination of SS, TP, SRP and NO_3 concentrations.

5.2.3.2 Event sampling

Automatic water samplers were deployed to take hourly samples during high-flows from Blind Beck and the modified wetland outlet. A float switch installed in the stream/wetland (located next to pressure transducers to record water stage) activated the sampler program. Upon initialisation a maximum of 24 x 1 litre samples could be taken as long as the water depth remained above the chosen stage threshold. The wetland sampler drew water from the outlet (defined by the V-notch weir) thus enabling the calculation of sediment/nutrient loads (using corresponding discharge data). The Blind Beck sampler was located at manual sample site 6 and not at the catchment outfall (site 9). This was because site 9 is next to a public road and the security of the monitoring equipment could not be guaranteed. Discharge for site 6 was estimated by down-scaling the discharge from site 9, using catchment area, thus allowing the calculation of pollutant loads.

5.2.3.3 Comparison between actual and predicted concentrations

Water samples were collected on a daily basis (at 12:00) from Site 6 over a 36 day period (28/09/2011 – 02/10/2011) using a programmed auto-sampler. Samples were analysed for SS and TP concentrations. These *actual* concentration values are then compared with *predicted* concentrations calculated using the developed sediment/nutrient – discharge rating curves, in order to test the accuracy of the method.

5.2.4 Revised annual load/yield estimation

Sediment and nutrient loads and yields for the blind beck catchment are calculated according to the method outlined in Chapter 4.2.2.3, using the revised rating curves. The revised loads/yields are then compared with those presented in Chapter 4 to examine the influence of event sampling on the estimation of determinand export.

5.2.5 Comparison of interpolation and extrapolation methods for calculating loads

The daily *actual* SS and TP concentration data (see 5.2.3.3 above) are used to compare two methods of load calculation – extrapolation (i.e., the rating curve method used throughout this study) and interpolation (please refer to Section 4.2.2.1 for more information).

The standard interpolation methodology used for load estimation where daily data sets are available (see for example Kronvang and Bruhn (1996), Webb *et al.* (1997) and Johnes (2007)) uses the following method:

$$\boldsymbol{L = \frac{k\sum_{i=1}^{n}(C_i Q_i)}{\sum_{i=1}^{n} Q_i}\overline{Q_r}}$$

Equation 5.1

where: L is load, C_i is instant concentration, Q_i is instant discharge, Q_r is mean discharge for period and K is the conversion factor for the time period.

5.3 Results

5.3.1 Hydrological characterisation

Flow parameters for Blind Beck are reported in Chapter 4. Figure 5.13 depicts the precipitation and discharge hydrograph for the two-year study period; red markers indicate days when grab samples were collected and blue markers indicate when the auto-samplers were operational. The hydrograph suggests a flashy runoff regime. Ockenden (2010) described how the time constant at Blind Beck (10.9 hours) was most similar to the River Eden at Temple Sowerby (catchment area 616 km^2). This was attributed to similar geology (38% sandstone and 62% limestone in Blind Beck, and 28% sandstone and 69% limestone in Temple Sowerby). Eleven separate high-flow events (A - J) were captured with the Blind Beck auto-sampler, the precipitation and discharge statistics of which are summarised in Table 5.4.

According to the Flood Estimation Handbook (Institute of Hydrology, 1999) (summary statistics for Blind beck can be found in Appendix C), the Standard Percentage Runoff derived from HOST (SPR HOST), defined as the percentage of rainfall that causes a short-term increase in flow, is 35% for the Blind Beck catchment; the Base Flow Index derived from HOST (BFI HOST), the long-term average of flow that occurs as base flow, is 0.56; and the Standard Period (1961-1990) average annual rainfall is 1018 mm. Based on the data collected during this study, the average runoff percentage for Blind Beck was calculated as 74% and 72% for 2010 and 2011 respectively (compared to 81% and 84% for Kirkby Stephen – reported in Tables 4.5 and 4.6, section 4.2), and 100% runoff is possible during storm events; for example, storm event 14/01/2011 lasted 41 hours, 42.8 mm of precipitation fell and 43.7 mm of runoff was recorded. Daily precipitation totals exceeding 10 mm occurred on average 33 days per year (Figure 5.10); Q_{50} is 0.125 m^3 s^{-1} and Q_5 is 0.7 m^3 s^{-1} (Figure 5.11).

High discharges occur predominantly in the winter months; especially in 2010 when relatively little rainfall fell between May and October. This pattern is less apparent in 2011 as a number of significant storms occurred in the summer; annual cumulative precipitation and runoff plots depict the difference between the two study years (Figure 5.12).

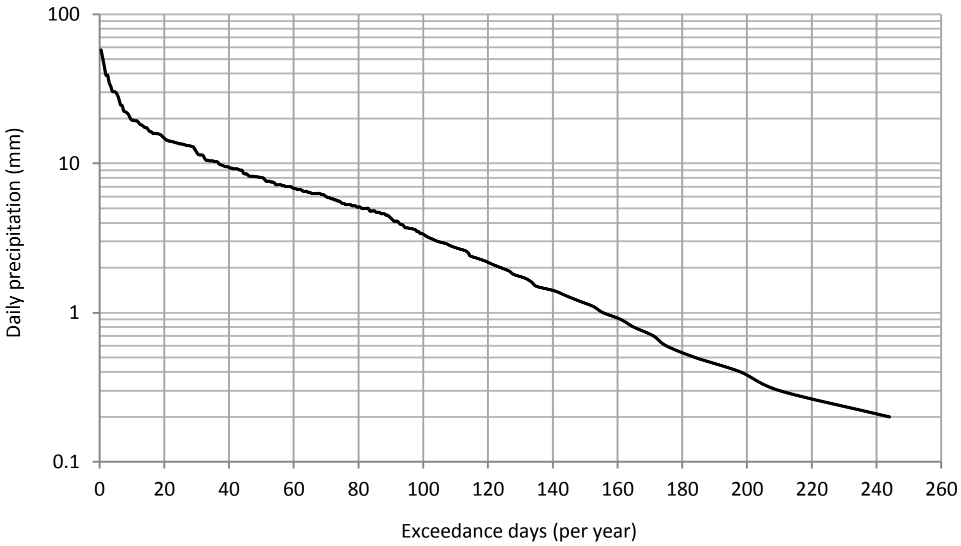


Figure 5.10: Blind Beck daily rainfall exceedance frequency (2010-2011).

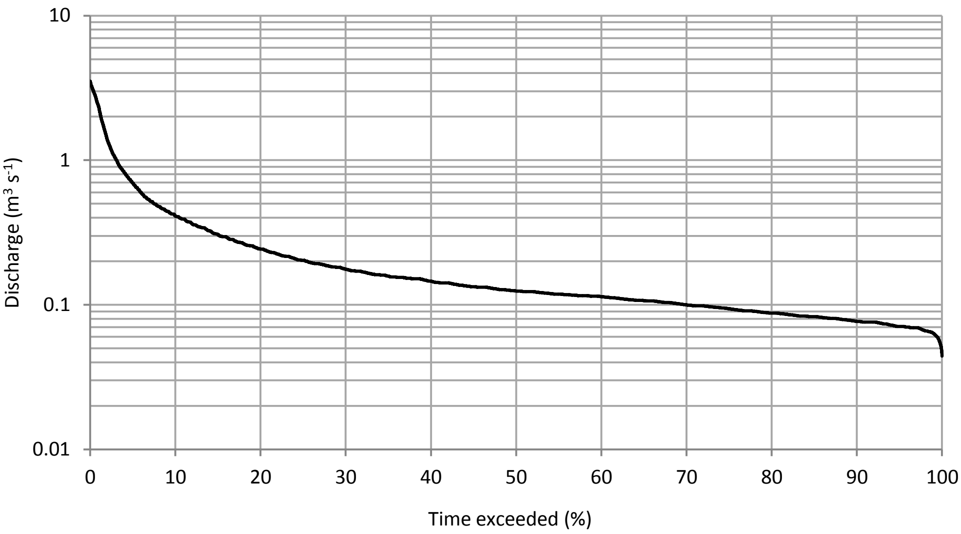


Figure 5.11: Blind Beck 15 minute flow duration curve (2010-2011).

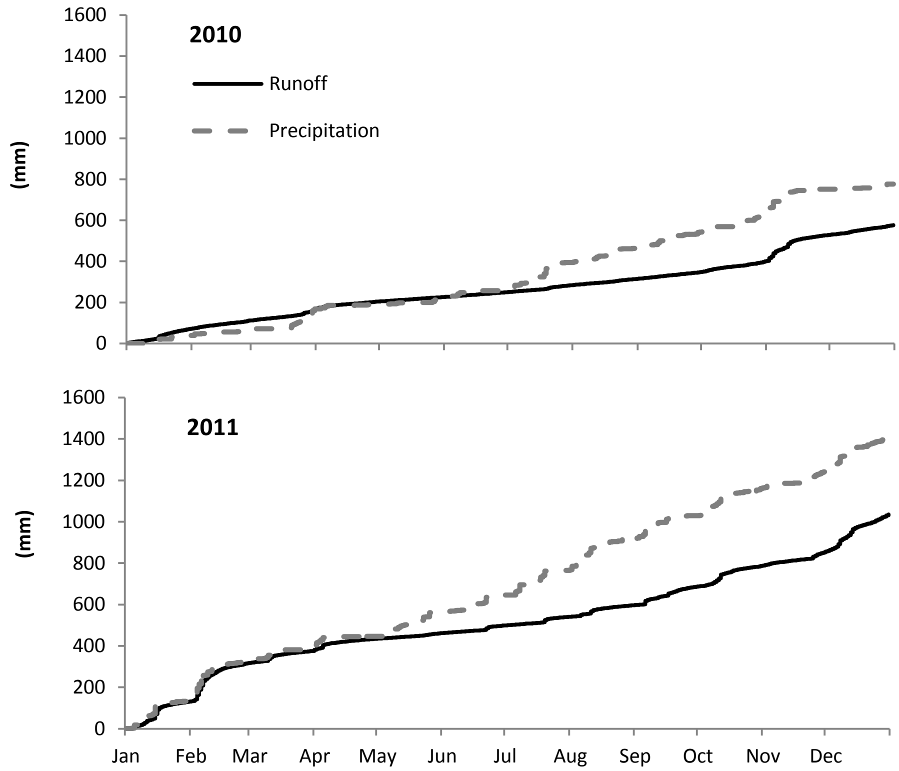


Figure 5.12: Blind Beck cumulative runoff and precipitation (2010-2011).

5.3.1.1 Catchment water balance

Water balances were calculated for both study years according to the method described in Chapter 4.2 – Equation 4.1. The water balance is acceptable for both years as recorded *Q* is within +/- 10% of calculated *P-E*, but recorded Q in 2011 is relatively low, compared with calculated *P-E* (Table 5.3). Some possible error could to be attributed to an underestimation of discharge resulting from extrapolation of the stage-discharge rating curve; Blind Beck is flow-gauged at 84% of the maximum recorded stage.

Table 5.3: Blind Beck catchment annual water balances.

Year	*P* (mm)	*Q* (mm)	*E* (mm)	*P-E (mm)*
2010	779	576	184	595
2011	1429	1035	280	1149

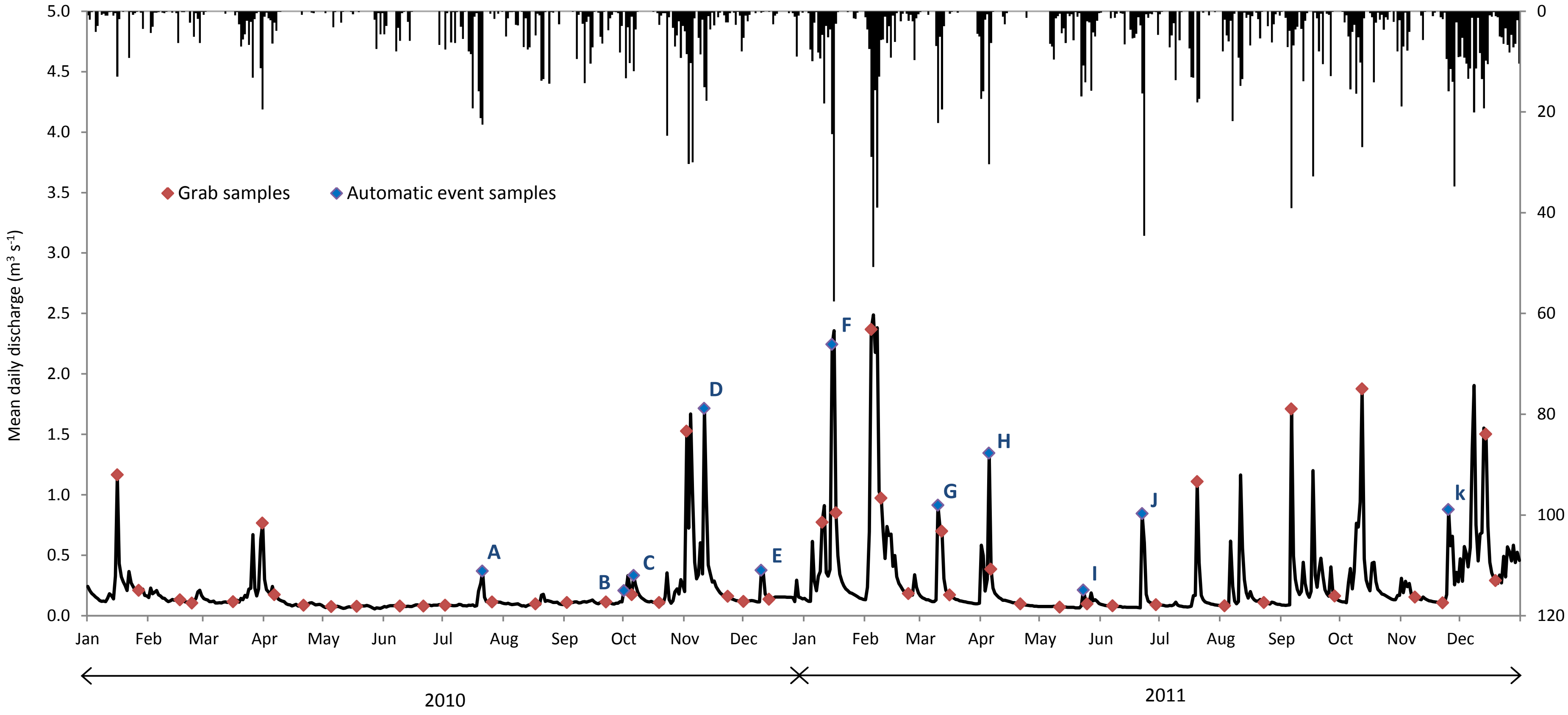


Figure 5.13: Blind Beck daily discharge and precipitation record 2010-2011. Markers indicate water quality sampling dates.

5.3.1.2 Soil water level and wetland discharge

Soil water level and discharge from the modified wetland is depicted in Figure 5.14. Significant wetland discharge is restricted to winter months only. Soil water levels remain low until November 2010, when they suddenly increase and the wetland produces a small amount of discharge (04/11/2010). Heavy precipitation on 15-16/01/2011 and 04-05/02/2011 causes soil water levels at site 3 and 4 to reach the ground surface, which coincide with discharges in the wetland of over 80 l s^{-1}. The wetland ceases to flow after April and apart from a small number of low discharges in response to heavy summer storms, does not flow again until a period of heavy precipitation at the beginning of October. The wetland discharges into the Low Hall stream.

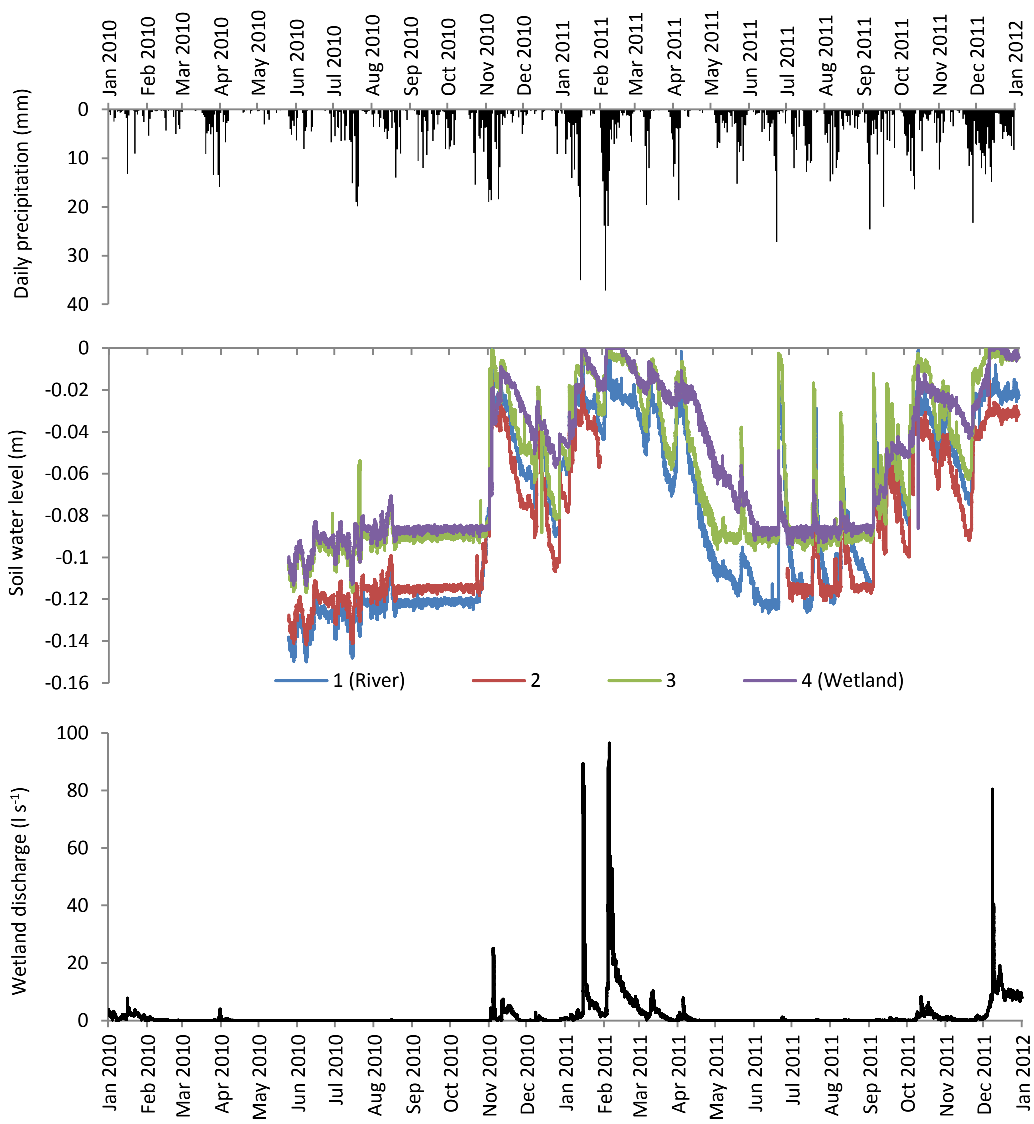


Figure 5.14: Soil water level and wetland response to precipitation at Sykeside Farm, Blind Beck 2010-2011 (see Figure 5.9 for instrument locations – 1, 2, 3 and 4).

5.3.2 Sediment and nutrient concentration characterisation

5.3.2.1 Variability along the river network

The following section presents SS, TP, SRP and NO_3 concentration data from grab samples collected along the Blind Beck river network (Figure 5.15 - logarithmic scales are used for plots *a* and *b* to account for the large variation between minimum and maximum SS and TP concentrations, respectively). Mean SS concentration generally increases along Blind Beck reaching a maximum at site 5 (downstream of Sykeside Farm – 42.5 mg l^{-1}) and then decreases to 29.2 mg l^{-1} at the catchment outlet. The maximum-recorded SS concentration of 342.5 mg l^{-1} was recorded at Site 6 – as Blind Beck leaves Sykeside Farm. The lowest SS concentrations are found at Site 1 (6.1 mg l^{-1}), closely followed by Low Hall and Wygill Beck.

Total P and SRP both have elevated mean concentrations at Site 1, which then attenuate slightly downstream until reaching Site 6 where mean concentrations peak (TP = 0.117 mg l^{-1}, SRP = 0.038 mg l^{-1}). Wygill Beck and Low Hall stream have the lowest mean P concentrations. Mean NO_3 concentrations increase between Sites 1 and 6 (2.66 and 4.81 mg l^{-1}, respectively). Low Hall has the highest mean concentration of 14.1 mg l^{-1} (and the highest recorded maximum concentration of 24.5 mg l^{-1}). Blind Beck outfall mean NO_3 concentration is significantly greater (t = -1055; p < 0.001) than that at Site 6.

Grab sample data tables including date, contaminant concentration and discharge can be found in Appendix G1.

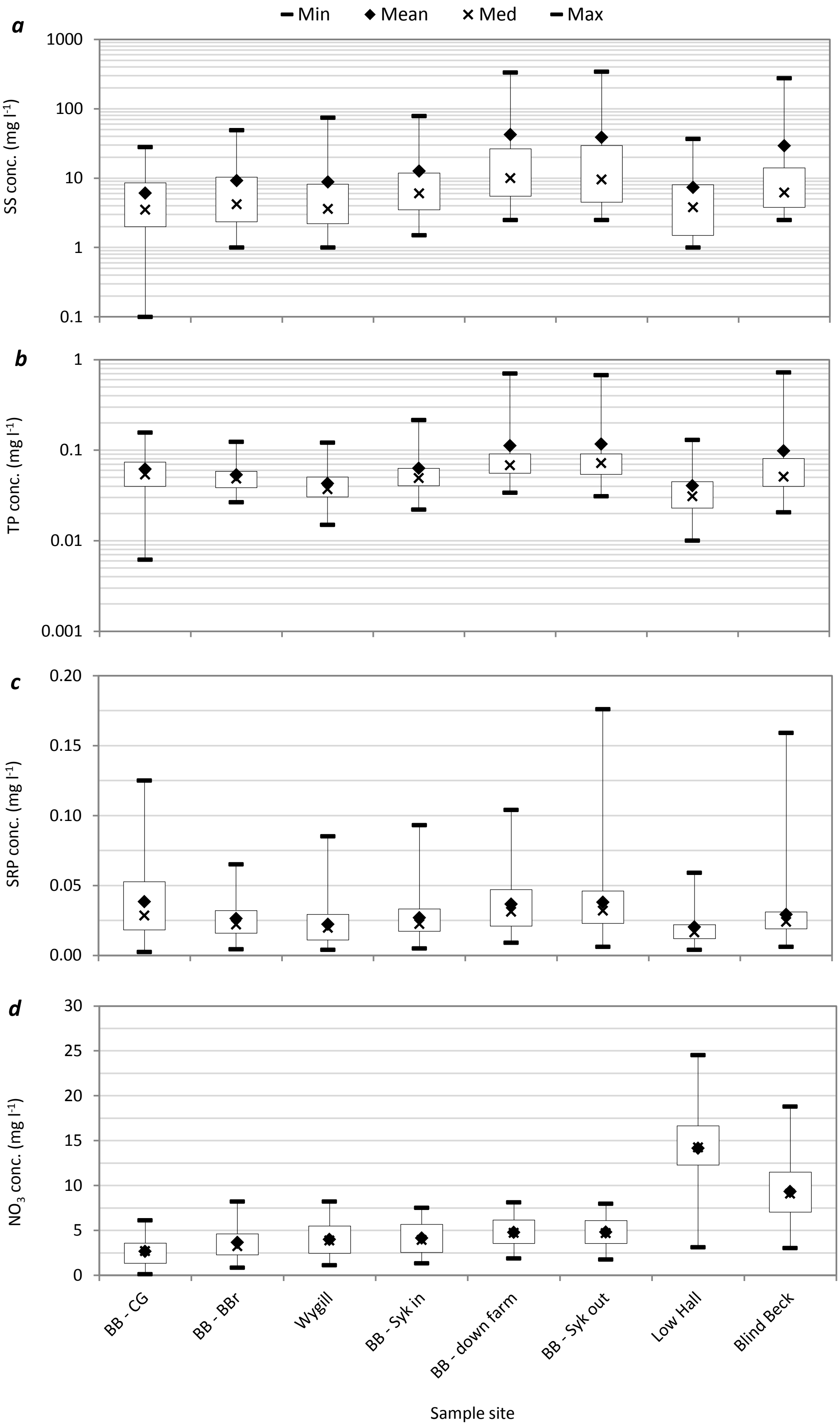


Figure 5.15: Concentrations of *a*) SS, *b*) TP, *c*) SRP and *d*) NO_3 along the Blind Beck river network (see Table 5.2 for sample site explanation).

5.3.2.2 Storm event data

Data are selected to illustrate sediment/nutrient losses in relation to events of different magnitude and temporal occurrence (event data not reported graphically in the following section can be found in Appendix G2). Samples were collected across the whole discharge range (Figure 5.13) meaning that concentrations and derived loads (see section 5.3.4) are a good representation of the full range of hydrological conditions. Table 5.4 contains discharge and precipitation summary data for all eleven sampled events (raw data can be found in Appendix G3). The maximum-recorded discharge during the monitoring period was 3.52 $m^3 s^{-1}$ and the maximum discharge at which a sample was taken was 3.25 $m^3 s^{-1}$, which equates to 92.3% of the maximum recorded.

Table 5.4: Blind Beck precipitation, discharge and sampling event summary 2010-2011.

Date	Event	Duration sampled (hrs)	Total precipitation (mm)	Peak hourly precipitation (mm)	Peak discharge ($m^3 s^{-1}$)	Peak runoff (mm hr^{-1})
20/07/2010	A*	10	20	2.8	0.704	0.241
01/10/2010	B	12	10.2	2.4	0.418	0.142
06/10/2010	C	13	23.8	9.2	0.499	0.169
11/11/2010	D*	24	24.6	3.6	2.736	0.944
10/12/2010	E	14	14	4.2	0.92	0.314
14/01/2011	F*	20	42.4	2.4	3.245	1.115
09/03/2011	G*	21	9.4	2.8	1.715	0.616
04/04/2011	H	12	9.4	3.4	2.861	0.985
23/05/2011	I*	10	12	3.8	0.791	0.260
22/06/2011	J	19	17.4	10.4	3.171	1.099
24/11/2011	K	18	13.2	3.4	2.399	0.832

*Event analysed in text.

Event A took place in July 2010 following a prolonged dry period. It is a relatively short duration, low magnitude event that exhibits a significant hydrograph lag behind peak precipitation intensity (Figure 5.16). Samples were collected for 10 hours on both the rising and falling limb but only examined for SS and TP concentrations due to a time delay between collection and laboratory analysis. Suspended sediment and TP concentrations peak at 375 and 0.53 mg l^{-1}, respectively. Peak TP concentration corresponds with peak discharge but peak SS is one hour previous; falling limb measurements correlate well with the hydrograph.

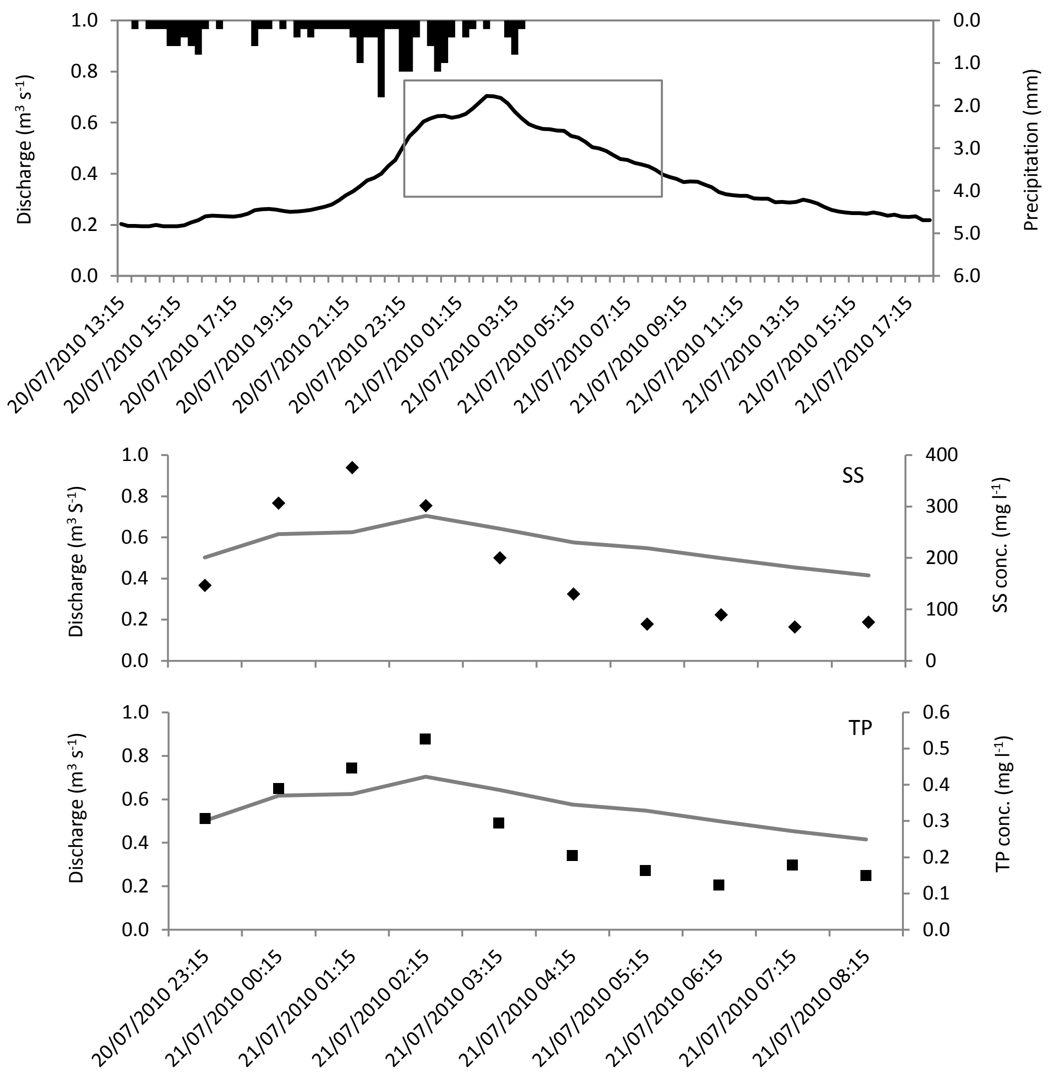


Figure 5.16: Discharge, SS and TP concentration record – Event A.

Event D (Figure 5.17) occurred in November 2010, it is a relatively high magnitude, multi-peaked event that followed two similar sized contiguous storms. Twenty-four samples were taken in total. Suspended sediment concentration peaked (at 546 mg l^{-1}) after four hours, one hour after the first discharge peak and circa one hour following peak intensity rainfall. Total P (the total height of the stacked columns) reaches 1 mg l^{-1} after seven hours, three hours later than the SS peak concentration. Particulate P accounts for circa 80% of the peak TP concentration and SRP makes up the majority of the soluble fraction. Soluble P (SRP plus SUP) peaks in the ninth sample hour. Both SS and P concentrations generally decline despite the second and third discharge peaks, which elicit a far more limited response. The second and third peaks in discharge occur in response to lower intensity rainfall (higher catchment runoff coefficient). Nitrate concentrations demonstrate very little variation over the 24-hour period, rising from 4.3 to 5.1 mg l^{-1} (peaks at 15 hours) before declining slightly.

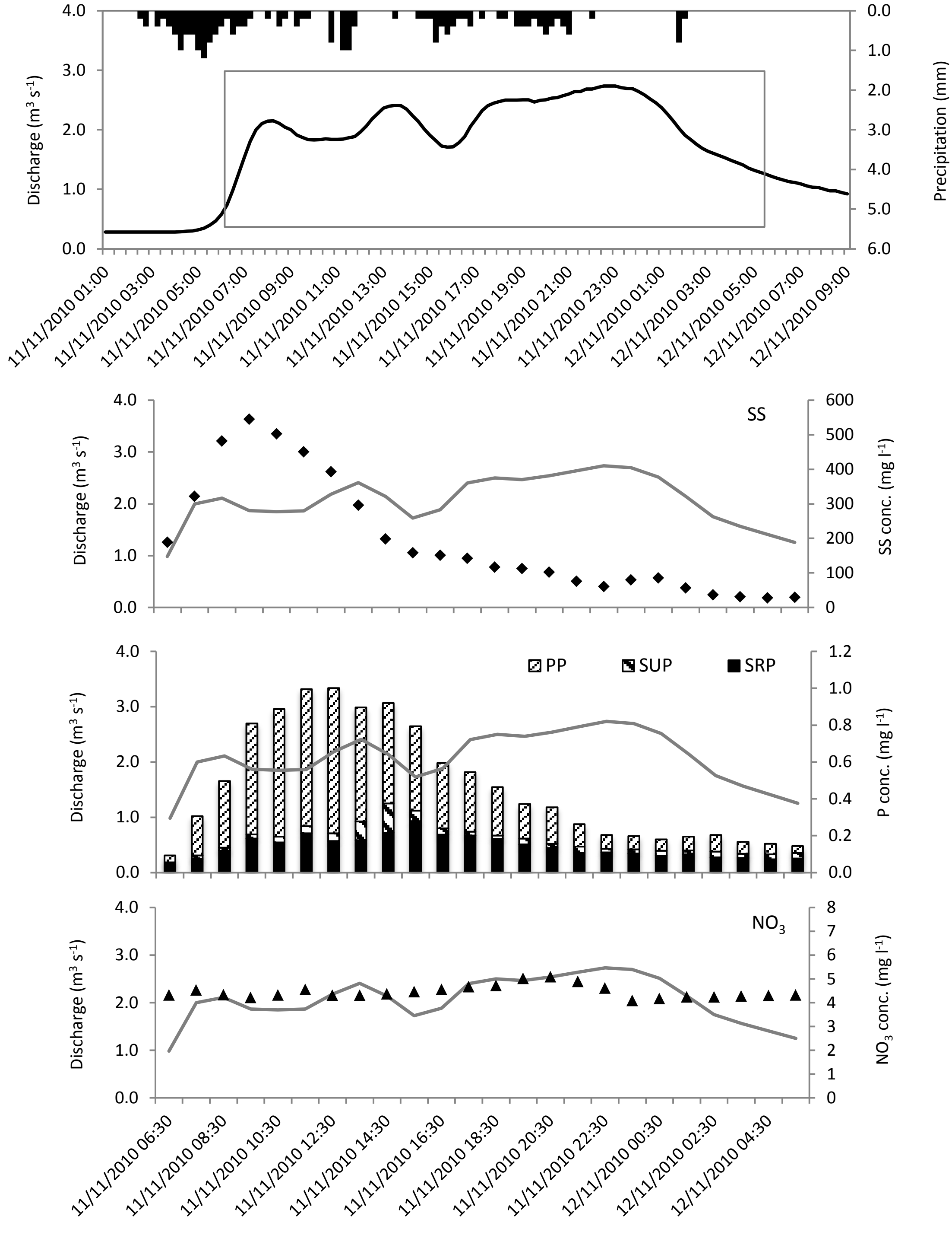


Figure 5.17: Discharge, SS and P and NO_3 concentration record – Event D.

Event F (Figure 5.18), which took place in January 2011, is a high-magnitude (second-highest discharge recorded during the study period) event in response to relatively low intensity, high magnitude precipitation (42.4 mm in 20 hours), where peak discharge is maintained for circa nine hours. Runoff greater than 1 mm hr^{-1} was recorded at the peak, which is relatively rare in the catchment. Peak SS (387 mg l^{-1}) and TP (0.63 mg l^{-1}) concentrations occur 2-3 hours before peak discharge. Circa 80% of peak TP concentration consists of PP, while soluble P (80%

reactive) peaks at seven hours and accounts for 62% of TP. Similarly to Event D, NO_3 demonstrates very little overall variation throughout the storm but in this instance does show a slight decrease (negatively correlated with discharge) reaching a minimum concentration (3.75 mg l^{-1}) at seven hours, and then increases to a maximum of 5.27 mg l^{-1} at 14 hours.

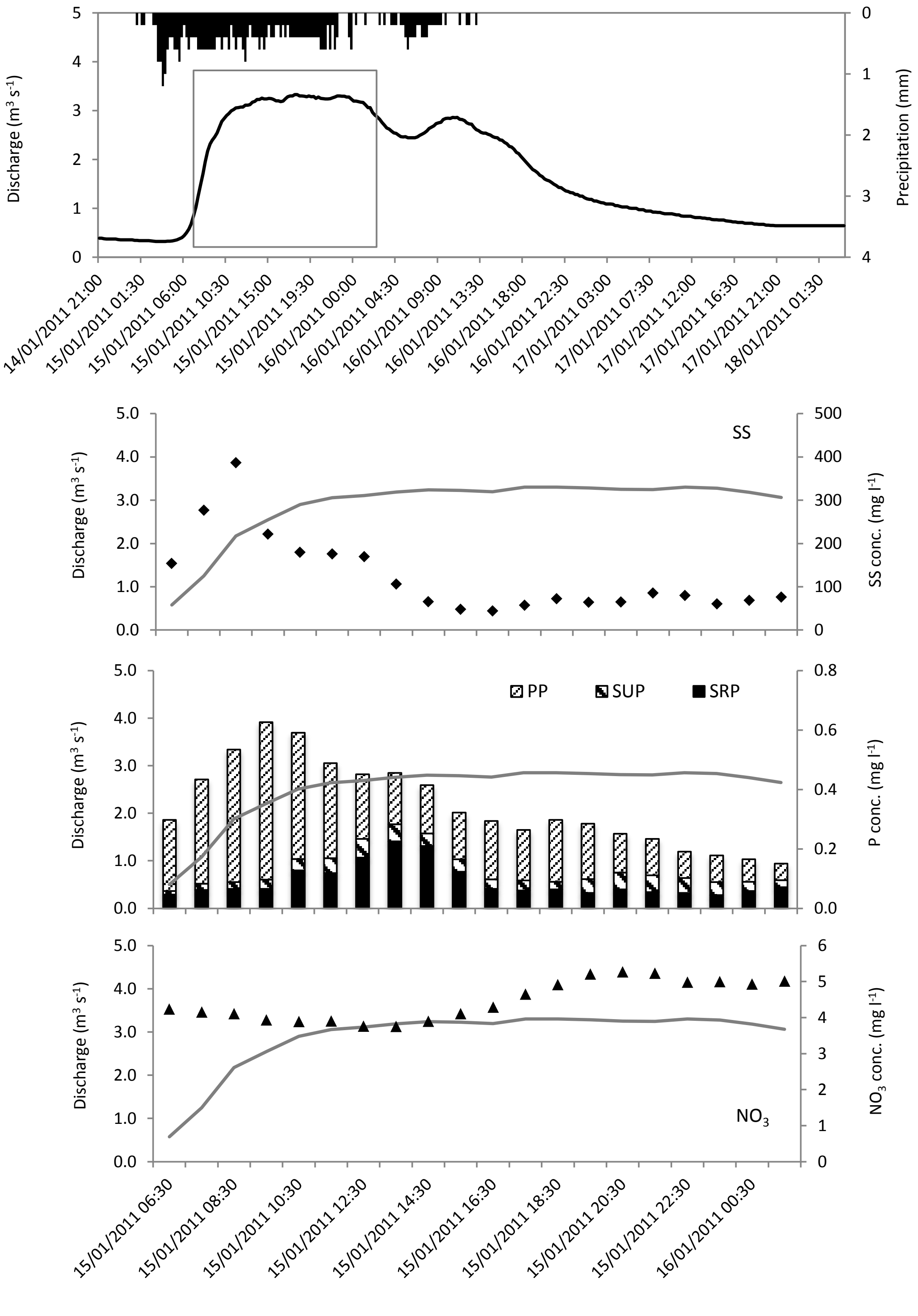


Figure 5.18: Discharge, SS and P and NO_3 concentration record – Event F.

Event I (Figure 5.19) occurred in May 2011. It is a relatively intense, short duration summer event following a month of very little rainfall. There is significant hydrograph lag behind high peak intensity rainfall.

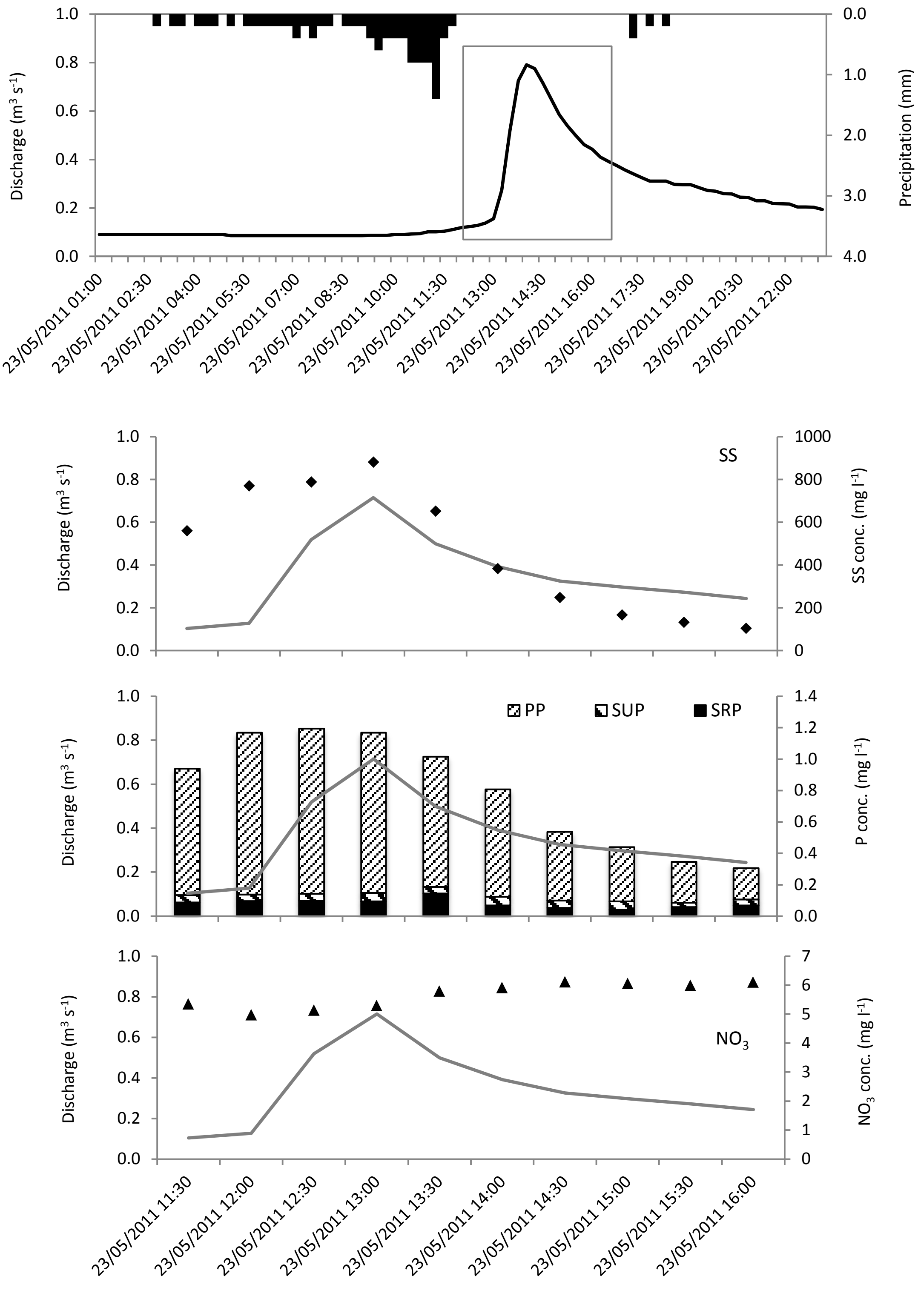


Figure 5.19: Discharge, SS and P and NO_3 concentration record – Event I.

Although discharges less than 1 $m^3 s^{-1}$ were recorded, this event exhibits the highest recorded SS concentration (881 mg l^{-1}), which coincides with peak discharge, and relatively high TP. Particulate P accounts for circa 88% of TP. Both SS and TP (and PP) decrease with strong correlation with the falling limb. Nitrate exhibits a similar pattern to previous events whereby concentration decreases on the rising limb before increasing again on the recession. Maximum NO_3 concentration (6.1 mg l^{-1}) is recorded three hours after peak discharge.

5.3.2.3 Hysteretic behaviour

Hysteretic behaviour of sediment/nutrient concentrations in Blind Beck is described using Event G as an example; a medium magnitude event recorded in March 2011 (Figure 5.20). For SS and TP concentrations, hysteresis loops were predominantly clockwise (for 9 out of 11 and 8 out of 11 recorded events, respectively). A low magnitude event that was mostly sampled on the falling limb exhibited anticlockwise loops, and a multi-peaked event resulted in no clearly defined loop direction for both determinands. Another anticlockwise loop was recorded for TP and was the result of increasing SRP concentrations during the latter part of the monitored event. SRP concentrations exhibited anticlockwise loops on 4 out of 7 occasions, while the other three demonstrated no clear patterns. NO_3 concentrations showed no pattern during 3 out of 5 events and anticlockwise loops during the other two. Hysteresis plots for all determinands from all recorded events can be found in Appendix G4.

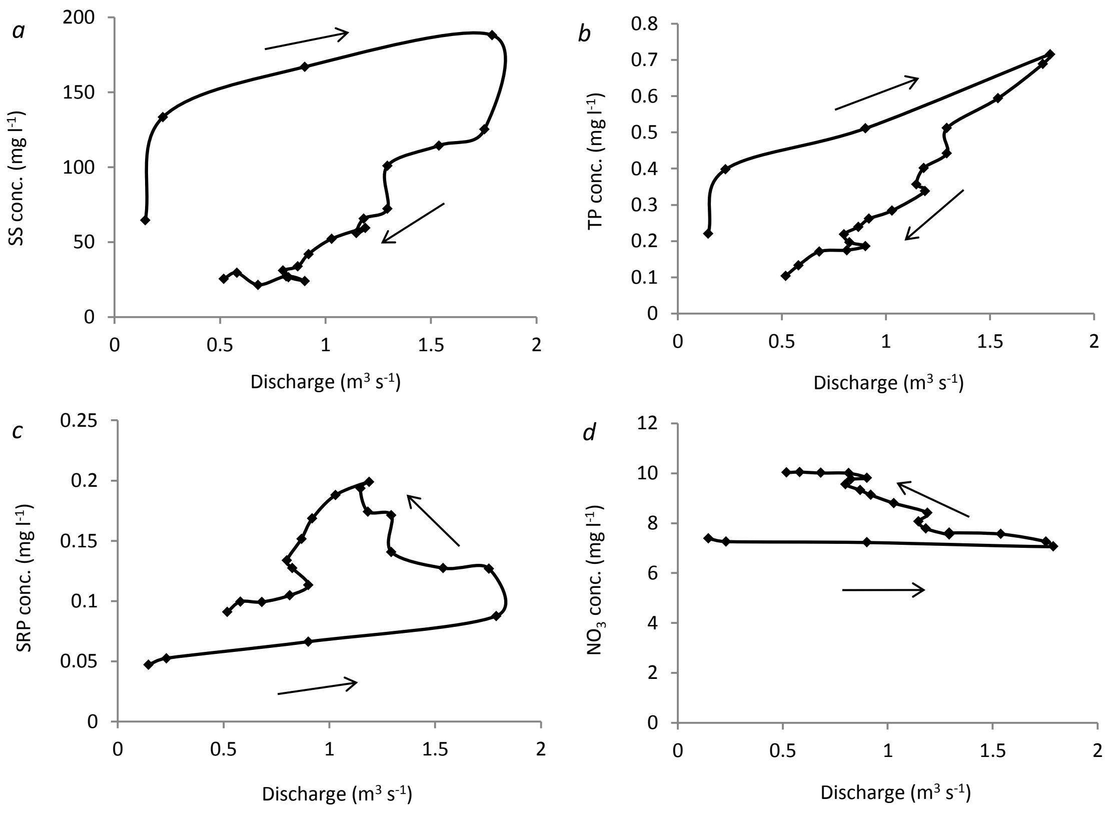


Figure 5.20: Hysteresis at Blind Beck *a)* SS, *b)* TP, *c)* SRP, and *d)* NO_3 – Event G.

Predominance of clockwise hysteresis of SS and TP concentrations during storm events in Blind Beck means that concentrations peak before peak discharge. Conversely, anticlockwise hysteresis of SRP and NO_3 concentrations indicates that there is a lag of peak concentration behind peak discharge.

5.3.2.4 Correlation between water quality determinands

Figure 5.21 depicts the correlation between SS and TP concentrations recorded at the Blind Beck outlet; a strong positive relationship exists with a Pearson's *R*-value of 0.817. Table 5.5 contains Pearson's correlation coefficients for all the measured water quality constituents; all correlations are significant (p<0.001).

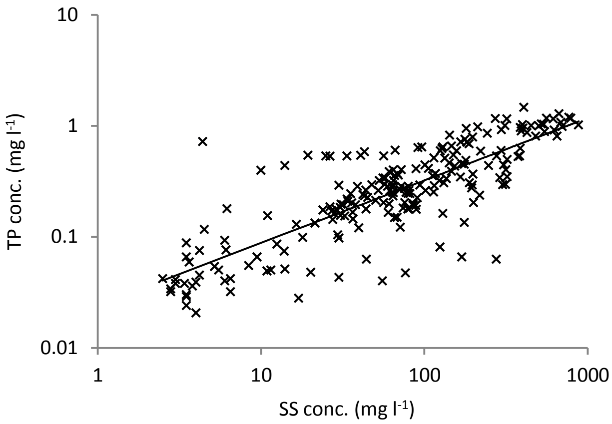


Figure 5.21: Correlation between SS and TP concentrations in Blind Beck.

Table 5.5: Pearson's correlation coefficients to describe relationship between water quality determinand concentrations.

	TP	SRP	PP	NO_3
SS	0.817	0.286	0.912	-0.462
TP	-	0.532	0.974	-0.471
SRP	-	-	0.417	-0.379
PP	-	-	-	-0.430

5.3.3 Revised sediment/nutrient rating curves

Concentration data collected by the Blind Beck auto-sampler were combined with grab sample data (from the catchment outlet) to produce revised sediment/nutrient-discharge rating curves for Blind Beck (Figure 5.22). All correlations are significant (p < 0.001 - Table 5.6).

Considerable scatter is shown, particularly for SS and TP and especially at higher discharges. This may be partly attributed to the hysteresis effect, which is discussed in section 5.5. Rating coefficients can be found in Appendix G5.

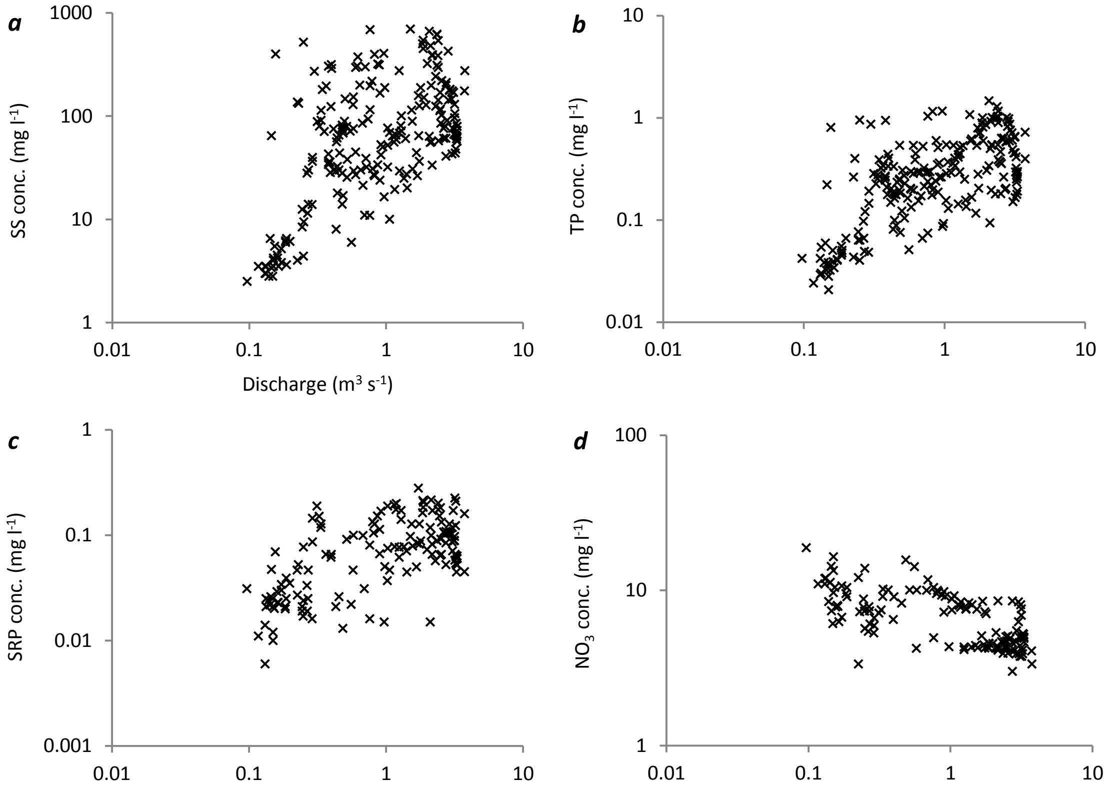


Figure 5.22: Relationship between *a*) SS; *b*) TP; *c*) SRP; *d*) NO_3 and discharge for the River Eden at Blind Beck.

Table 5.6: Pearson correlation coefficients and *P*-values for correlations between sediment/nutrient concentrations and discharge at Blind Beck.

Constituent	Pearson's *R*	*P*-value
SS	0.258	<0.001
TP	0.472	<0.001
SRP	0.411	<0.001
NO_3	-0.628	<0.001

The effect of using event-collected samples, as well as grab samples, is to increase the mean and maximum concentrations of SS, TP and SRP for the same location over the sample period, compared with using just grab samples (Table 5.7 and Table 5.8 contain constituent concentration data for both grab and event samples, and just grab samples, respectively).

Mean NO_3 is decreased by 21%. Mean SS concentration is increased by over 300%, mean TP by 260% and SRP by 167%. A greater standard deviation is also associated with the event-sampled data, but not for NO_3.

Table 5.7: Blind Beck determinand concentrations (grab and event samples).

Constituent	n	Concentration (mg l^{-1})				σ
		min	max	mean	median	
SS	218	2.5	686.4	118.4	65.7	140.8
TP	218	0.021	1.46	0.36	0.27	0.30
SRP	145	0.006	0.28	0.08	0.07	0.06
NO_3	139	3.0	18.8	7.1	6.9	3.0

Table 5.8: Blind Beck determinand concentrations (grab samples).

Constituent	n	Concentration (mg l^{-1})				σ
		min	max	mean	median	
SS	49	2.5	276.5	29.2	6.2	58.4
TP	49	0.02	0.72	0.10	0.05	0.14
SRP	49	0.006	0.16	0.03	0.02	0.02
NO_3	38	1.1	18.8	9.6	9.4	3.8

5.3.3.1 Comparison between actual and predicted concentrations

Daily SS and TP concentrations (recorded at 12:00) are presented alongside corresponding concentrations predicted using the sediment/nutrient rating curves (Figure 5.23).

Figure 5.23 *a* shows a strong agreement between actual and predicted SS concentrations during residual flow conditions. There is one small rainfall event that occurs on the 02-04/10/2011, which causes a slight increase in actual SS concentration, but as this doesn't elicit a response in discharge there is also no response in predicted SS concentration. During storm events where discharge peaks occur there is a trend for the predicted SS concentrations to be greater than the actual ones, varying between circa 10 and 25 mg l^{-1}, but on the whole the representativeness of the predicted values is good. However, the discharge peak on 17/10/2011 has no corresponding peak in SS concentration as it occurred overnight (sampling time was 12:00 pm), which highlights the major issue with samples collected on a daily basis.

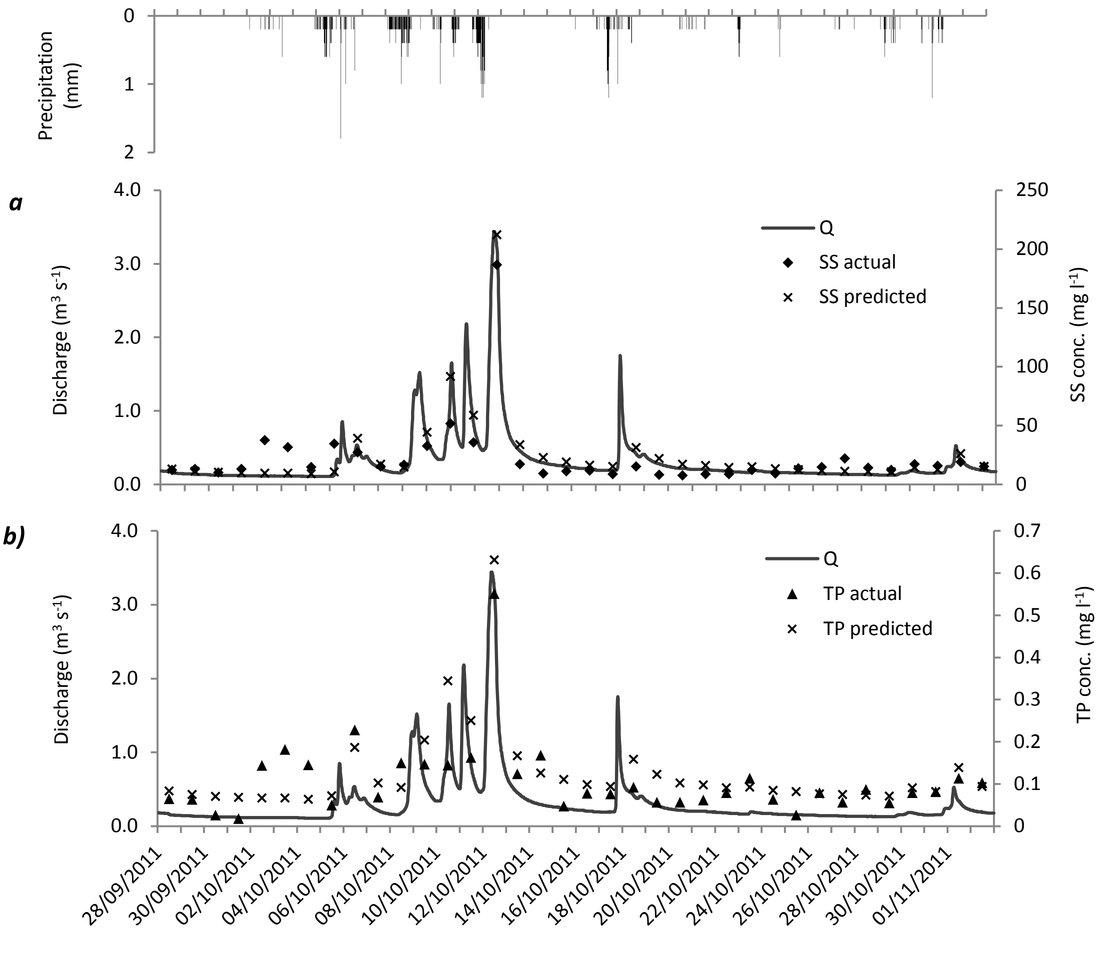


Figure 5.23: comparison of instantaneous *a*) SS and *b*) TP concentrations from daily collected samples and rating curve predictions.

Figure 5.23 *b* indicates that actual TP concentrations during residual flow conditions exhibit more fluctuation compared with SS concentrations. The same concentration increase due to the rainfall event near the start of the monitoring period is seen as it was for SS. Predicted TP concentrations during the series of storm events are also high in comparison with the actual values, in one instance over 100% greater but there are also occasions where the actual TP concentration is greater than the predicted one.

Over the 36 day period the sum of predicted SS concentrations is 12% greater than the sum of the actual concentrations, and predicted TP is 16.5% greater. Generally there are strong positive correlations between the two for both SS and TP (Figure 5.24) although this may be, in part, due to the single 'high' concentration recorded on 12/10/2011.

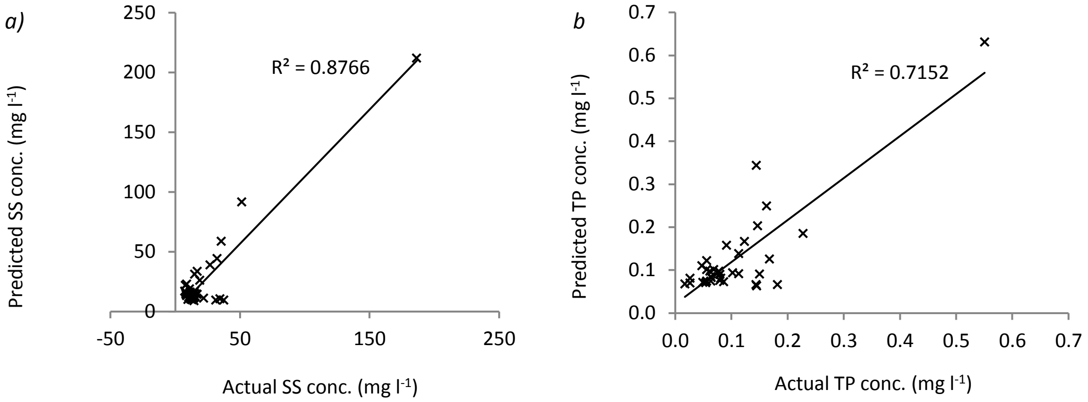


Figure 5.24: Correlation between predicted (using rating curves) and actual concentrations of *a)* SS and *b)* TP.

5.3.4 Revised load/yield estimation

5.3.4.1 Calculation of annual yields

Annual yields of SS, TP, SRP and NO_3 are presented in Table 5.9. Figure 5.13 indicates that sampling took place over the full range of discharges during the study period and can therefore be assumed to be representative. For comparison, Table 5.10 contains loads and yields calculated for Blind Beck based on *grab samples only* (as presented in Chapter 4).

Table 5.9: Blind Beck sediment/nutrient loads and yields (event + grab samples).

Constituent	**Load (yr^{-1})**		**Yield (km^{-2} yr^{-1})**	
	2010	**2011**	**2010**	**2011**
SS (t)	170	631	18.9	70.1
TP (kg)	746	2345	82.9	260.6
SRP (kg)	207	555	23	61.6
NO3 (t)	49.5	72.2	5.50	8.00

Table 5.10: Blind Beck sediment/nutrient loads and yields (grab samples only).

Constituent	**Load (yr^{-1})**		**Yield (km^{-2} yr^{-1})**	
	2010	**2011**	**2010**	**2011**
SS (t)	79	318	8.73	35.35
TP (kg)	345	1076	38.3	119.5
SRP (kg)	120	280	13.4	31.2
NO3 (t)	50.9	74.3	5.65	8.26

Based on the revised rating curves SS yield in 2011 (70.1 t km^2 yr^{-1}) is very close to that proposed by Mills (2009) as a long-term estimate for the catchment (73 t km^2 yr^{-1}); although the 2010 value is considerably lower. However, the effect of the *event + grab sampling* method on SS load/yield is greater in 2010 (116% increase) than 2011 (98% increase), compared to *grab sample only*. The addition of event sampling increases TP yields by 116% and 118%, and SRP yields by 72% and 97% in 2010 and 2011, respectively. There is no discernible difference for NO_3.

5.3.4.2 Load exceedance

High, residual (medium) and low (base) flow threshold values were defined for Blind Beck by analysis of the discharge hydrograph (Figure 5.13) and FDC (Figure 5.11). Daily base flow was calculated using the IH method (Gustard *et al.*, 1992) and an average value of 0.1 m^3 s^{-1} was taken; which is the flow range less than the discharge that is exceeded 70% of the time and accounts for 11% of the total annual discharge (Table 5.11). The high flow threshold was taken as the average value of troughs that lay between the largest contiguous discharge peaks; this value of 0.4 m^3 s^{-1} defines the flow range greater than the discharge that is exceeded 10% of the time, and accounted for 40% of the total discharge. By this definition 25 high flow events occurred during the two-year study period: 7 in 2010 and 18 in 2011. By deduction, residual flow occurred for 60% of the flow period ((but is not the flow exceeded for 60% of the time) and contributed 49% of the overall discharge.

Cumulative exports of SS, TP, SRP and NO_3 (based on estimated continuous loads) demonstrate that the majority of SS, TP and SRP were derived from high-flow conditions (Figure 5.25 and Table 5.11). The analyses show that 84% of SS is exported during high flow events; 76% of TP and 68% of SRP. Conversely, low flow conditions contribute very small proportions of SS and P loads. The majority of NO_3 (54%) is transferred under residual flow conditions and high flows only contribute 32% of the total flux. Low flows account for the lowest proportion of NO_3 export but significantly more than for SS and P.

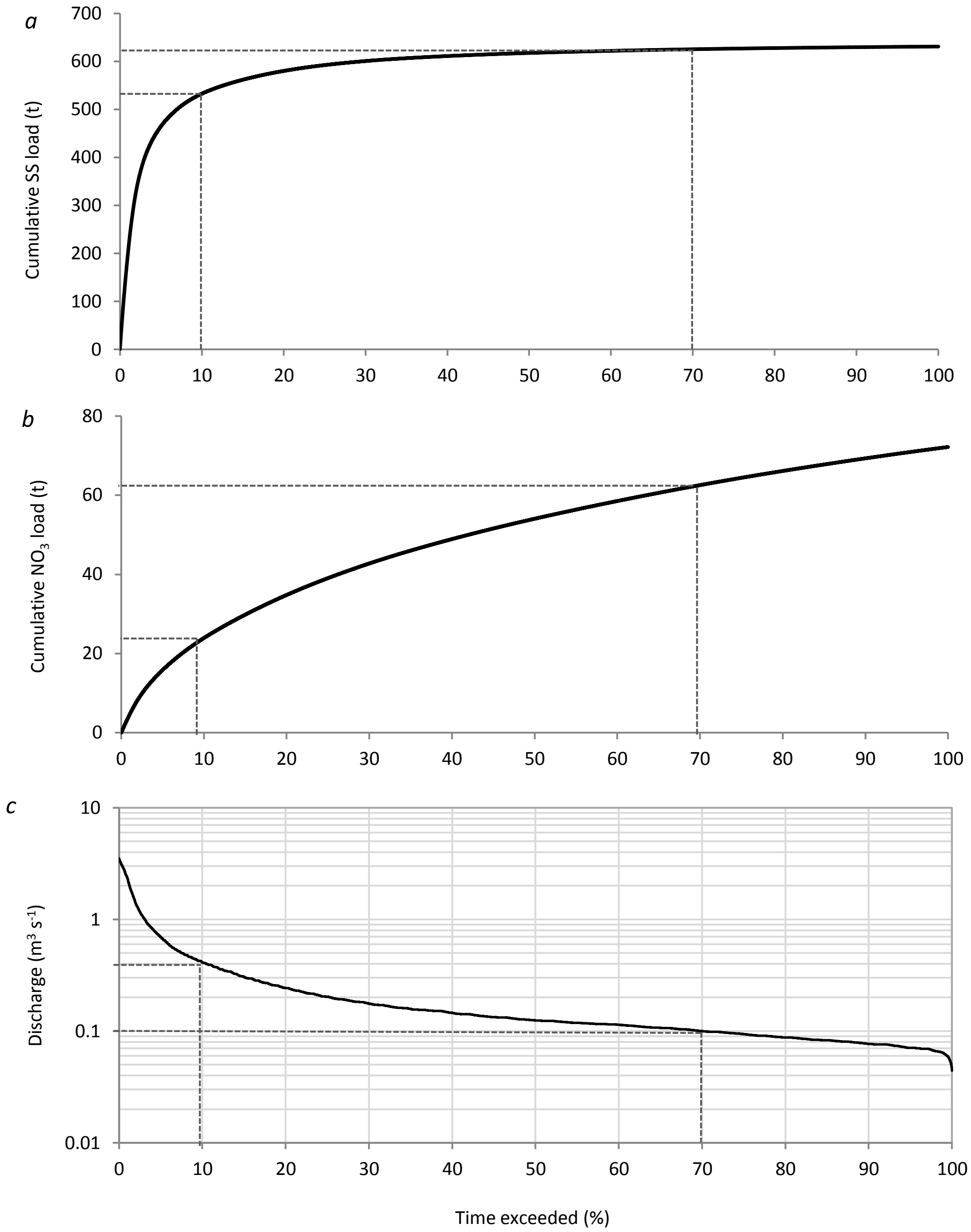


Figure 5.25: Cumulative *a*) SS, *b*) NO_3 export (2011) and *c*) FDC for Blind Beck.

Table 5.11: Contribution of different flow conditions to the export of SS, TP, SRP and NO_3 at the Blind Beck outlet.

Flow condition	Time exceeded (%)	Discharge threshold (m^3 s^{-1})	Percentage contribution				
			Discharge	SS load	TP load	SRP load	NO_3 load
High	10	>0.4	40	84	76	68	32
Residual		0.1><0.4	49	15	22	28	54
Low	70	<0.1	11	1	2	4	14

5.3.4.3 Storm event load and transfer rate

The load and transfer rate of sediment/nutrients transported during storm events in Blind Beck varies greatly between individual events (see Table 5.12), although there is a propensity for the largest loads/highest rates to be in the winter months; for example Events D, F, H and K. However, Events B, C and E all exhibited relatively small loads/low export rates and they too occur in the winter. Event D (November 2010) exhibits the highest values for SS and also TP, but the highest TP transfer rate was in Event H. The largest NO_3 load was transported in Event F, although only approximately half of all sampled events included NO_3 analysis. Eleven events out of approximately 25-30 definable storm events were sampled during 2010 and 2011. The total loads of SS, TP and NO_3 measured during these events account for 13%, 10% and 2%, respectively, of the total estimated overall loads for 2010 and 2011 combined (Table 5.12).

Table 5.12: Summary of SS, TP and NO_3 loads recorded during 11 sampled storm events (2010-2011).

Sampled event	Duration (hrs)	Load			Transfer rate (kg hr^{-1})		
		SS (t)	TP (kg)	NO_3 (kg)	SS	TP	NO_3
A (July 2010)	9	3.8	6.3		423	0.7	
B (Oct 2010)	11	1.8	3.5	103	165	0.3	9.4
C (Oct 2010)	12	1.1	5.3		92	0.4	
D (Nov 2010)	23	34.1	84.6	801	1482	3.7	34.8
E (Dec 2010)	13	5.4	10.4		417	0.8	
F (Jan 2011)	19	22.7	69.8	935	1194	3.7	49.2
G (Mar 2011)	20	5.8	29.3	613	291	1.5	30.7
H (Apr 2011)	11	11.2	57.1		1021	5.2	
I (May 2011)	4.5	3.4	5.4	36	744	1.2	8.0
J (June 2011)	9	0.5	6.7	91	57	0.7	10.1
K (Nov 2011)	17.5	13.5	41.3		770	2.4	
	Total event load	103	320	2579			
	Total overall load (2010 and 2011)	801	3091	121700			
	Percentage of total overall load	13	10	2			

5.3.5 Comparison of interpolation and extrapolation methods for calculating loads

Section 0 presented SS and TP instantaneous concentration data collected on a daily basis from Site 6 over a 36 day period. These *actual* concentrations were compared with corresponding *predicted* values derived from the revised rating curves, where it was found that the sum of predicted SS concentrations was 12% greater than the sum of the actual concentrations and predicted TP was 16.5% greater.

The calculated SS and TP loads for the 36 day period using the extrapolation and interpolation methods are presented in Table 5.13. As the rating curves predicted moderately higher concentrations (compared with the measured concentrations) it could be expected that the extrapolation method would estimate higher loads also. However, the interpolation method gave the highest values for both SS and TP - 24% and 11% greater, respectively.

Table 5.13: Comparison of SS and TP loads calculated using extrapolation and interpolation (Equation 5.1) methods.

Methodology	**Load (kg)**	
	SS	**TP**
Extrapolation	56874	221
Interpolation	70603	245

5.3.6 Modified wetland RAF water quality analysis

Wetland discharge was mainly restricted to the winter (Figure 5.26) and all grab samples except two were collected during the months of October to March. Suspended sediment, TP, SRP and NO_3 data are summarised in Table 5.14. Values for all determinands are relatively low and all mean values are significantly lower ($p < 0.05$) than those recorded in Blind Beck at Site 6 - the area where water was believed to leave Blind Beck and flow towards the wetland during high flow events.

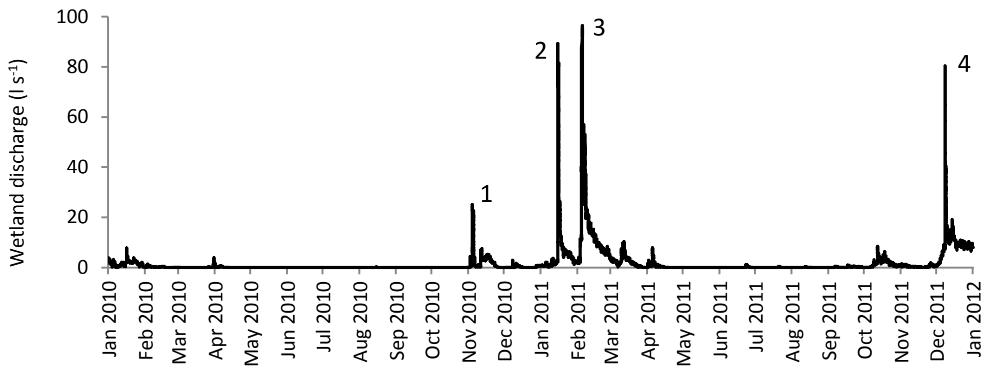


Figure 5.26: Modified wetland discharge hydrograph; markers indicate dates when auto-samplers were operational.

Table 5.14: Modified wetland grab sample SS, TP, SRP and NO_3 concentration summary.

Constituent	**n**	**Concentration (mg l^{-1})**				**σ**
		min	**max**	**mean**	**med**	
SS	24	2.0	44.0	8.8	6.6	9.0
TP	23	0.010	0.075	0.038	0.031	0.020
SRP	22	0.005	0.041	0.017	0.015	0.011
NO_3	21	0.1	11.4	3.2	2.2	2.6

Four high flow events were sampled using an auto-sampler at the wetland outlet (indicated in Figure 5.26). Event 2 (Figure 5.27) takes place in January 2011 and corresponds with Event F in Blind Beck (Figure 5.18), which was the second-highest discharge recorded during the study period. Sampling was initiated in the Beck at 15/01/2011 06:30 but not in the wetland until 12:15. The piezometer located in the wetland area indicates that the soil water table reached the ground surface at circa 11:30 (see Figure 5.14), suggesting that the entire catchment would have been at or near to saturation during this storm event. This is reinforced by the runoff value of greater than 1 mm hr^{-1}, recorded at peak flow (in Blind Beck). Wetland discharge peaked at 89.5 l s^{-1} at 17:00, six hours later than the first defined peak in the Beck.

In Event 2, SS and TP concentrations increase and decrease in strong correlation with discharge with SS peaking at approximately the same time, and TP circa two hours later. Maximum SS and TP concentrations of 47 and 0.08 mg l^{-1} were recorded, respectively. NO_3 exhibits a more delayed response, increasing gradually, then dipping during peak flow, and peaking after 12 hours. Maximum-recorded NO_3 concentration was 7.2 mg l^{-1}. During the January 2011 event (Event F in Blind Beck and Event 2 in the wetland) 22.7 t of SS, 69.8 kg of TP and 935 kg of NO_3 were exported from the beck over a 19 hour sampling period. By comparison, in the wetland 0.1 t of SS, 0.2 kg of TP and 30.5 kg of NO_3 were recorded over a 24 hour sampling period (Table 5.15).

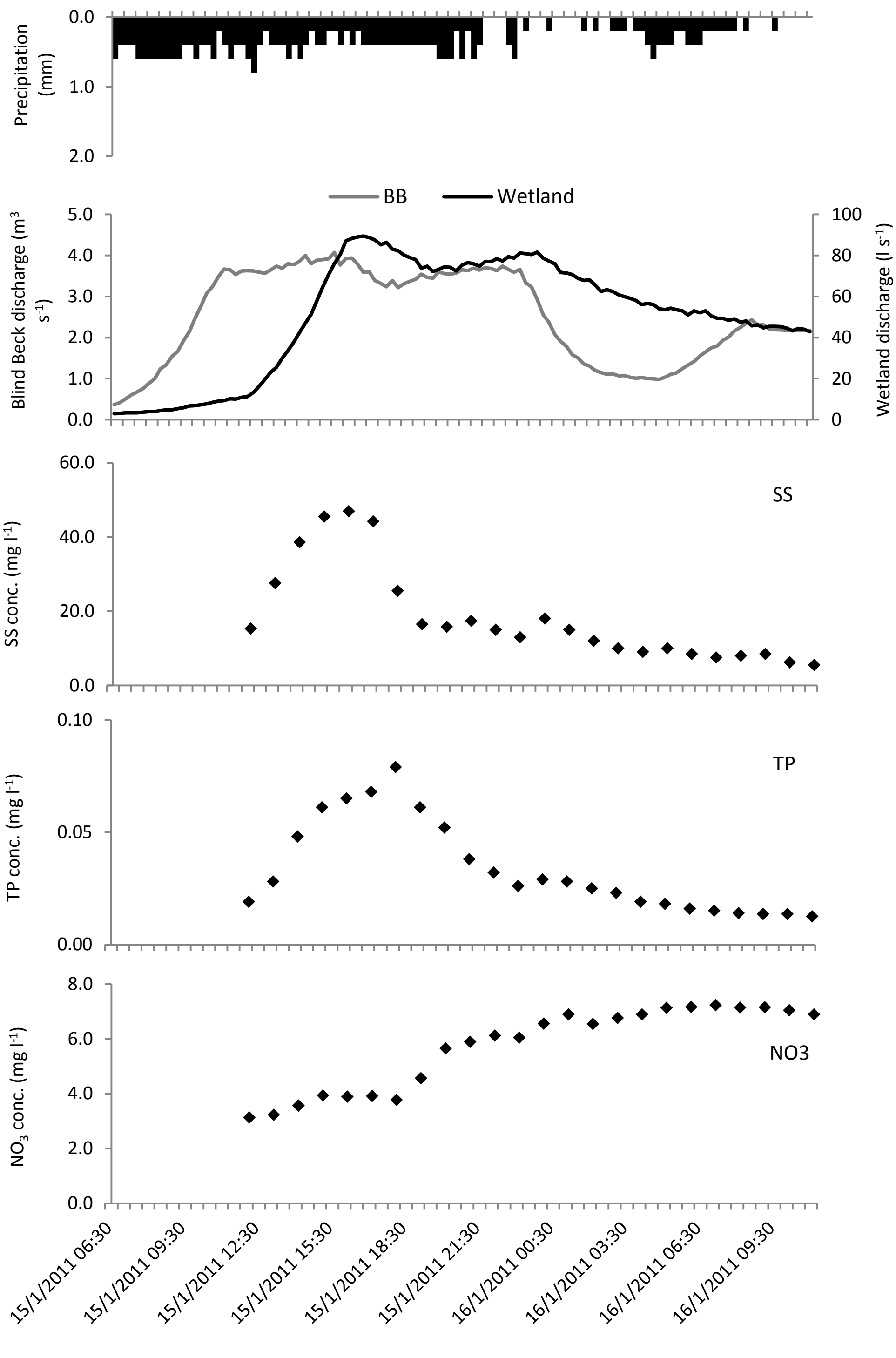


Figure 5.27: Discharge, SS, TP and NO_3 concentration record – Event 2.

Event 3 (Figure 5.28), the highest discharge event, occurs in February 2011, 19 days after Event 2. The initial discharge peak in Blind Beck (at 12:15 – event not sampled in Blind Beck) is reflected in the wetland but to a lesser extent and circa 3 hours later. A second discharge peak occurs in the Beck 12 hours later, but this time the wetland response to the precipitation is faster and outlet V-notch reaches capacity flow (circa 100 l s^{-1}). The SS and TP concentration response is markedly different from that in Event 2 as they both decreased as discharge increased. Suspended sediment concentration fell to circa 7 mg l^{-1} and remained at that level for the duration of the storm, showing very little reaction to the main discharge peak. TP concentration exhibited a small peak, which appears to be related to the initial modest discharge rise, before falling again to circa 0.02 mg l^{-1}. As there is no accompanying rise in SS at hours 5-7 it is assumed that the increase in TP is accounted for by soluble forms of P, which were not measured in this instance. Nitrate response is also rather different as the concentration increases relatively sharply, compared with Event 2. The rise in concentration was circa 3 hours before the main discharge peak, and approximately 3 hours after the initial, smaller peak. Following peak NO_3 concentration of 7.6 mg l^{-1} at sampling hour 11, levels decline gradually.

Total SS, TP and NO_3 event loads of 41.1 kg, 93.4 g and 26.0 kg, respectively, were recorded during Event 3, which are significantly lower than the exports in Event 2 (Table 5.15).

Table 5.15: Summary of modified wetland event SS, TP and NO_3 loads.

Event	**Date/time**	**Hours sampled**	**Event load**			**Total discharge (m^3)**
			SS (kg)	**TP (g)**	**NO_3 (kg)**	
1	04/11/2010 19:15	24	13.8	26.5	5.0	4880
2	15/01/2011 12:15	24	101.1	204.0	30.5	20935
3	04/02/2011 10:45	24	41.1	93.4	26.0	18774
4	08/12/2011 11:30	24	40.0	122.0	16.5	15898

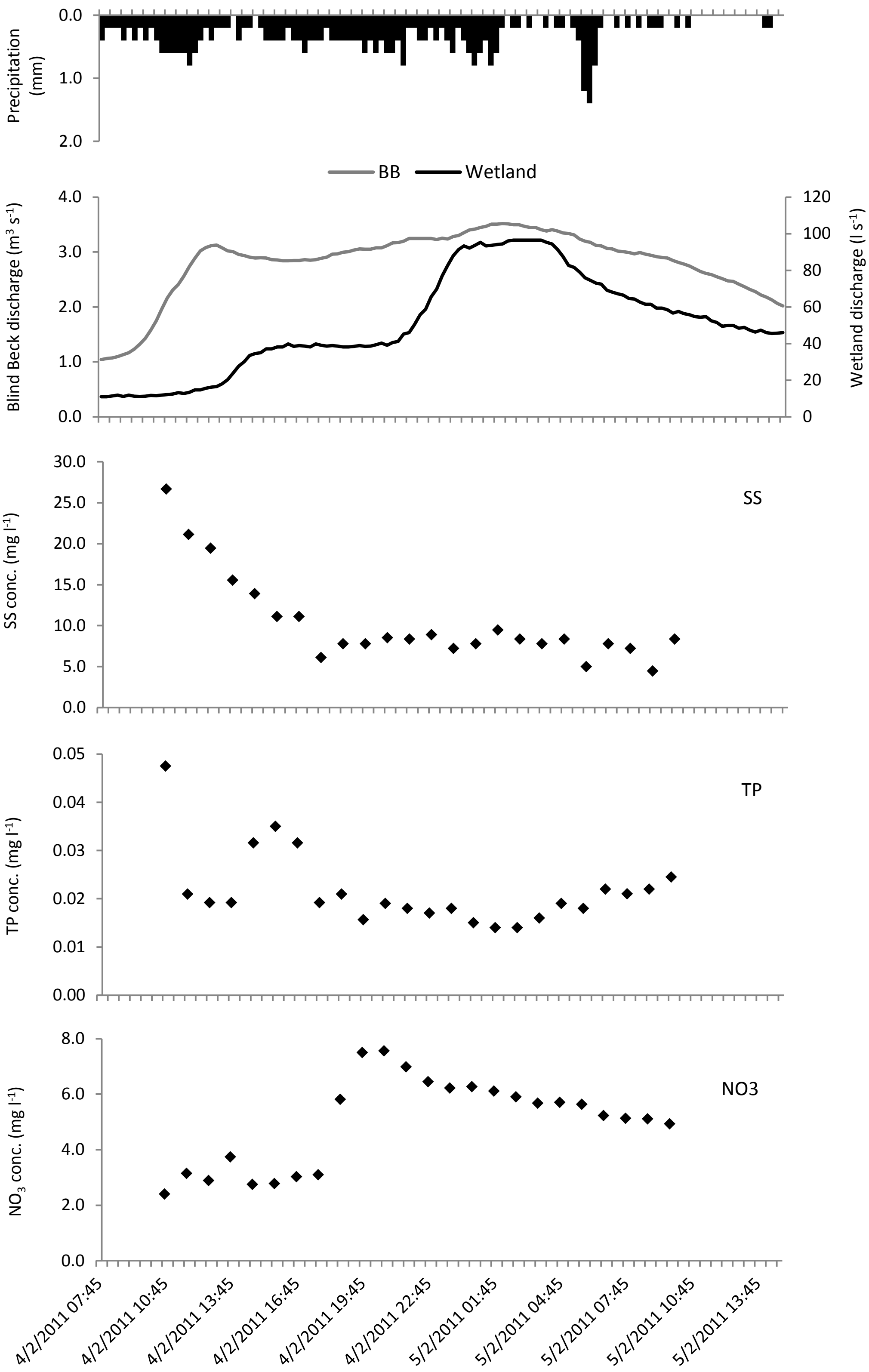


Figure 5.28: Discharge, SS, TP and NO_3 concentration record – Event 3.

5.4 Discussion of findings

5.4.1 Hydrological regime

As Blind Beck is part of the upper Eden catchment, much of the description of annual and seasonal patterns of precipitation stated previously in Chapter 4 is withheld here. Blind Beck has a higher BFI than the rest of the sub-catchments in the upper Eden according to FEH descriptive statistics and values calculated in Chapter 4. Ockenden (2010) confirmed this by demonstrating that Blind Beck had a time constant similar to the much larger Temple Sowerby catchment (on the main Eden – 616 km^2) - longer than the other upper Eden sub-catchments. This was mainly attributed to the larger proportion of sandstone (circa 38%) underlying the catchment and that circa 46% of discharge was estimated to move via a slower flow pathway, i.e., subsurface flow. Despite this, overland flow (generated by saturation excess) was observed in the catchment on several occasions (Plate 5.2) and can be expected to deliver large quantities of sediment and sediment-phase nutrients to the stream. Overland flow is also capable of connecting more distant parts of the catchment with the stream. However, as the majority of the catchment is of low relief and has a high percentage of permanent grass cover, the erosive potential of overland flow is relatively low.

Plate 5.2: Overland flow near Crosby Garrett (04/02/2011 - largest recorded discharge during this event).

Plate 5.3: Water from Blind Beck flowing along the adjacent road, just upstream of the outlet monitoring station.

Calculated water balances for the Blind Beck catchment were acceptable (runoff within +/- 10% of *P-E*) for both study years, but in 2011 discharge (in mm) was lower relative to 2010. This could be attributed to an underestimation of discharge resulting from out-of-channel flow during high-flow events, which runs along the adjacent road (Plate 5.3). As peak discharges are potentially underestimated, this means that calculated sediment and nutrient loads would also be underestimated as a consequence.

5.4.2 Source-pathways and temporal patterns of sediment and nutrient losses

Managing agricultural diffuse pollution is notoriously troublesome as by definition, the source is often dispersed across an extensive area. However, in many cases the reality involves trying to identify ‘distributed point sources’, which can vary in both space and time, within a given catchment. Mean SS concentrations were relatively low upstream of Sykeside Farm, after which they significantly increased (from 12.7 to 42.5 mg l^{-1} between Sites 4 and 5), particularly during higher discharges. The highest P concentrations were also found downstream of Sykeside Farm (increase from 0.06 to 0.11 mg l^{-1}) although SRP and subsequently TP concentrations were often found to be relatively high at Site 1. This was likely due to a point source of soluble P, for example, discharge from a farmyard or a septic tank leak, and the lack of dilution by ‘clean’ water in this small catchment area meant that concentrations were high. Subsequent dilution at downstream sites was apparent until the increase downstream of Sykeside Farm.

This suggests that SS and P sources must exist between Sites 4 and 5, which may lie outside and/or inside the channel. In-channel P sources refer to bed and channel bank sediments, on which P may be bound and from which it can be re-released. A heavily silted streambed was commonplace in the lower reaches of Blind Beck during low flow periods, particularly during the summer (Plate 5.4). Both outside and inside channel sources are conceivable in Blind Beck. Nitrate concentrations also increased along the river network but the magnitude and range was generally low, increasing from circa 3 to 5 mg l^{-1} at Site 1 and 6, respectively. The Low Hall catchment consistently exported significantly higher concentrations of NO_3 (mean 14 mg l^{-1}), which will be discussed later in this section.

Suspended sediment and P concentrations decrease between Site 6 and 8 (Blind Beck outlet). This is likely due to either dilution by the Low Hall stream (which exhibits significantly lower SS and P concentrations) or in-stream deposition upstream of Site 8. This particular stretch of river is fenced along the field side and has the road on the other (see Plate 5.3) and becomes very overgrown with vegetation, particularly in the summer. This may act to reduce SS concentrations by reducing the energy in the flow at certain points, which may lead to sedimentation and some filtering effect (e.g., Jones *et al.* (2012)). Wygill Beck was found to have low SS, P and NO_3 concentrations on average and loads could not be calculated, as discharge was not measured at this site. However, the data collected are sufficient to confidently rule out the 1.1 km^2 sub-catchment as a sediment/nutrient ‘hotspot’.

5.4.2.1 *Pollutant sources*

The Sykeside farmyard and hard standings are located in very close proximity to Blind Beck (Plate 5.5) and are perhaps the most obvious source of pollutants. With mere metres to transfer sediment/nutrients to the watercourse there is a permanent high level of connectivity, even small precipitation events would be adequate to mobilise and wash contaminants from the yard and into the stream. However, no apparent sediment input point was identified during the time spent at the site, nor was an obvious SS plume ever witnessed in the stream at this location, although this is not to say that it didn't occur. However, the same argument cannot be applied to the transfer of soluble P and NO_3, as there would be no visual indication of their movement.

Plate 5.4: Heavily silted bed and algal growth (02/06/2010).

Plate 5.5: Sykeside Farm hard standings in close proximity to Blind Beck.

While the land cover is generally the same around Sykeside Farm as further upstream (improved grassland), the stocking density of both sheep and cattle increases as the river moves downstream through the catchment. Official stock numbers are unknown but the pattern was obvious from time spent in the catchment. Vogel (2003), Gravier (2004) and Mills (2009) also reported increased stocking densities in the Sykeside and Low Hall areas relative to upstream areas. Blind Beck has no defined riparian area along its entire course and none of the stream is fenced off apart from a small stretch along the road near Little Musgrave. Although not an issue in itself, when combined with elevated numbers of animals the risk of poaching and stream bank degradation is high. Poaching and bank degradation appears to be a major problem in the lower parts of Blind Beck, particularly on and around Sykeside Farm (Plate 5.6 and Plate 5.7).

Plate 5.6: Poaching near Site 5, Sykeside Farm (05/05/2010).

Plate 5.7: Poaching and animal excrement in Blind Beck near Site 6, Sykeside farm (07/06/2011).

Soil texture is arguably the most important soil property affecting soil erosion. To the west of Sykeside Farm, overlying Strutforth Hill is Clifton Association; a soil developed in fine loamy till with high clay content and slow permeability. This means that it has a propensity to produce surface runoff (particularly if it's on a slope) and also to poach easily when soils are at or above field capacity (National Soil Resources Institute (NSRI), 2013). Poaching on sloping sites over poorly drained soils is one possible explanation for increases in sediment derived from fields adjoining the watercourse. Bank collapse is also linked to nature of bank-forming material, although heavily influenced by degree of stock access and stocking density. Fine textured poorly drained bank material (e.g., boulder clay) may also be more susceptible than coarse textured well drained material (e.g., alluvium) to loss of stabilising vegetation due to poaching, resulting in greater incidence of back collapse, despite the fact that coarser textured banks are naturally less cohesive. It may be that boulder clay areas have steeper/higher banks than alluvial areas making catastrophic collapses more frequent, or possibly a combination of destabilised banks and flow accumulation, which means that critical shear strength is exceeded by the stream.

Bank collapses are a common sight in the lower half of the Blind Beck catchment (Plate 5.8) as well as instances of large-scale slumping at the foot of slopes (Plate 5.9). Although they both are naturally occurring geomorphological processes, accelerated erosion can provide a huge input of sediment and sediment-phase nutrients, which can remain in the channel to be remobilised and transported in a subsequent storm. Research has suggested that in livestock dominated catchments bank erosion can be the primary sediment (and associated nutrients) source and that it can reach problematic levels with regards to aquatic ecosystem health (e.g., Walling *et al.* (2003)). Large amounts of sediment retained within the stream network may also cause water eutrophication problems through resuspension and/or the release of dissolved nutrients (Owens and Walling, 2002; Stutter *et al.*, 2007; Jones *et al.*, 2012). It is believed that a bank collapse was responsible for what was observed during Event I, which was sampled by the auto-sampler at Site 6 in May 2011 (see Section 5.4.3). This was a relatively low magnitude event following a period of very little precipitation. A short, high intensity storm caused a short, sharp peak in discharge that exhibited the highest SS and TP concentrations recorded throughout the entire study.

Plate 5.8: Bank collapse near Site 4, Sykeside Farm (February, 2011).

Plate 5.9: Bank slumping/mass movement at the base of Strutforth Hill, Sykeside Farm (April, 2010).

5.4.2.2 Timing of pollution delivery

The majority of SS and TP (principally PP) was exported during short time periods, almost exclusively associated with discharge peaks. Walling and Webb (1987) stated that 60% of the overall sediment load was transported in 2% of the time in a review of the discharge of contaminants to the sea in the River Exe catchment. Eleven separate storm events were sampled during the study period, which covered a good range of hydrological conditions; including short duration, high intensity summer events and high magnitude, long duration, low intensity winter events. As expected the winter events were responsible for the largest exports of SS, P and NO_3, partly due to their longer duration and greater discharge values.

The greatest transfer rates were also exhibited in the winter with 1.5 t hr^{-1} of SS lost during a November storm and 5.2 kg hr^{-1} of TP during a March event. However, Jarvie *et al.* (2006) argued that although PP may form a significant proportion of the P load to rivers, it may have little impact on river eutrophication. They showed that agriculturally derived bed sediments (to which PP is attached) actually have the potential to reduce SRP concentrations from overlying river water during low flow/summer conditions (where SRP is already at elevated levels). Conversely, Jarvie *et al.* (2006) also acknowledge that agricultural fine sediment is harmful to riverine ecology if concentrations are excessive and thus needs to be controlled.

Relationships between intra-storm exports and (seasonal) land use and hydrological conditions are difficult to recognise given the length of monitoring period/number of events captured. However, sources of inter-event variability can include seasonal differences in land use activities/timings of stock grazing and cultivations; bank collapses; the effect of antecedent conditions; the remobilisation of fine bed material deposited during low/recessional flows; and the occurrence of contiguous events in quick succession leading to exhaustion.

Despite these complications a number of patterns were exhibited by the monitored water quality determinands during storm events:

- Discharge and SS concentration rose in response to rainfall, and P increased in line with sediment concentration.
- SS and P concentrations peaked rapidly either with or just before peak discharge.
- NO_3 initially fell due to dilution or stayed fairly constant, but increased during event recessions as slower flow pathways reached the stream (and rapid pathways receded).

There was no discernible difference between summer and winter responses, either as a result of the complications described above or perhaps due to the number of storms captured.

5.4.2.3 Hysteresis

Clockwise hysteresis dominated the response of SS and TP concentrations during the majority of the storm events monitored (82% and 73% of the time, respectively). This indicates that sediment supplies are abundant at the beginning of events but cannot be sustained (i.e., source limited), resulting in a concentration peak before peak discharge. Alternatively, these sources may become diluted as water from more distant parts of the catchment contributes to outlet discharge (Jansson, 2002). The widest loops are exhibited by SS meaning that there is a greater difference between concentrations on the rising and falling limb (less well correlated with discharge). This suggests that sources are either close to the channel, or within the channel itself (or both) thence allowing for rapid mobilisation. Mills (2009) also found that Blind Beck exhibited clockwise hysteresis for turbidity (which was used as a proxy for SS concentration) in 75% of events. TP was seen to respond quickly to an increase in discharge and generally peaked along with maximum flow (as with SS). Like sediment, the P (mainly PP) is most likely to originate from within the channel (bed sediments) and from the riverbanks, particularly in areas where there is poaching (e.g., Bowes *et al.* (2005)).

Anticlockwise hysteresis of SRP concentrations occurred during 57% of monitored events and indicates that there was a lag of peak concentration behind peak discharge. This could be due to a more distant pollutant source or transfer pathway and subsequent travel time to the catchment outlet. The peak in SRP after peak discharge could be explained by the subsurface transport of soluble and potentially colloidal P, which has a delay in reaching the stream. The majority of NO_3 concentration responses to discharge exhibited no hysteretic relationships but anticlockwise loops did occur for 40% of monitored events. The shape of the loops were characterised by a flat line (sometimes downwards sloping) where concentrations fall slightly as discharge increases, before concentrations steadily increase as discharge decreases. This is indicative of dilution of NO_3 concentration during the early stage of the event. Some studies have found that shallow groundwater can contribute more NO_3 to stream water during the recession period, after the rise of the saturation zone towards upper soil layers enriched by the accumulated nitrate pool (Rozemeijer and Broers, 2007; Oeurng *et al.*, 2010).

5.4.2.4 Low Hall sub-catchment

The Low Hall sub-catchment is an anomaly in the Blind Beck catchment as it consistently exhibits low SS, TP and SRP concentrations but relatively high levels of NO_3. It is the contribution of runoff from this 1.25 km^2 area of farmland that causes the NO^3 concentrations in Blind Beck to suddenly increase between Sites 6 and 9 (mean concentrations of 6.1 and 9.1 mg l^{-1}, respectively). The mean NO_3 concentration recorded at Low Hall (Site 8) was 14.3 mg l^{-1}, with a maximum of 24.5 mg l^{-1}.

Ockenden (2010) carried out several different types of investigation (a series of chemical tests and rainfall-discharge model output analyses) in the Blind Beck catchment, specifically focusing on the Low Hall sub-catchment. All the studies suggested that a significant proportion of water in the Low Hall stream, which is entirely on the *Permian Penrith Sandstone* bedrock, was from a groundwater source. The Low Hall stream had a higher specific conductivity than Blind Beck, which was attributed to higher concentrations of calcium carbonate, thus suggesting that the water in Low Hall stream spent longer in the ground, with longer contact with the rock. Continuous monitoring of stream water temperature showed that the Low Hall stream was warmer than Blind Beck in winter but colder in summer, also suggesting that a significant input from a deeper source (the temperature of the rock deep below the surface remains relatively constant throughout the year compared to the air temperature at the surface). Finally, End Member Mixing Analysis (EMMA), using specific conductivity, revealed that that 69% ± 10% of the water in the Low Hall stream was '*old*' water (slow pathway), compared with 46% ± 8% in Blind Beck. Previous to the work of Ockenden (2010), an investigation carried out by Mannix (2005) of two boreholes (one shallow – 6m and one deep – 211m) at Sykeside Farm and also of soil cores taken on the farm revealed 'extremely' high concentrations of NO_3 in the soil and shallow borehole and relatively high concentrations in the deep borehole. This comprehensive combination of analyses explains why the Low Hall stream is characterised by relatively high NO_3 concentrations.

5.4.3 The effect of event sampling on load estimation.

One of the aims of this chapter was to investigate the influence of event sampling (to complement the stratified grab sampling regime carried out in Chapter 4) on the calculation of SS, P and NO_3 annual loads. Loads calculated in this study, using the revised rating curves, are greater than those calculated in Chapter 4. Suspended sediment yield ranges between 18.9 and 70.1 t km^2 yr^{-1} for 2010 and 2011 respectively; TP between 82.9 and 260.6 kg km^2 yr^{-1}, SRP between 23 and 61.6 kg km^2 yr^{-1}, and NO_3 between 5.5 and 8.0 t km^2 yr^{-1}. Similarly to Chapter 4, the differences in export magnitudes between the two years are chiefly attributed to the difference in runoff volume - with 2011 being the wettest. This emphasises the importance of hydrology as the main driver of DWPA losses from rural catchments.

Russell *et al.* (1998) reported a TP loss range of 160-210 kg km^2 yr^{-1} for agricultural catchments in the UK and Jarvie *et al.* (2003) calculated the annual TP export from different sub-catchments in the Herefordshire Wye basin, which varied between 2 and 90 kg km^2 yr^{-1}. Wood *et al.* (2005) working in the predominantly grassland Taw catchment, estimated an export of 120 kg TP km^2 yr^{-1}. Thus TP exports calculated in this study fall well within ranges quoted in the literature. Labadz *et al.* (1991) reported SS yield estimates for upland catchments in the UK (areas between 42 ha and 7.7 km^2) of between 0.7 and 66 t km^{-2} yr^{-1}, and Bronsdon and Naden (2000) calculated yields (over 3-years) of 17.3 and 19.7 t km^{-2} yr^{-1} [1] for the Upper Tweed and Teviot catchments in northeast England, respectively. The yields calculated in this study are in general agreement with these values. Cooper *et al.* (2008) produced a SS yield classification system, which uses upper- and lower-quartile yields as critical thresholds and targets respectively, based on catchment typology (described in section 2.8.2). According to this classification, a catchment such as Blind Beck should have a target SS yield of 40 t km^2 yr^{-1} and a critical yield of 70 t km^2 yr^{-1}. Therefore, in a wet year (such as 2011) Blind Beck could be considered as yielding critical loads of SS, especially if the yields calculated in this study are underestimates of the true export.

Intra-storm sampling using automatic water samplers meant that concentrations of SS, P and NO_3 transported by short-duration peak flows could be measured. As expected, higher concentrations of SS and P were found to be associated with these higher discharges (e.g., Jordan *et al.* (2007)), mainly due to the higher erosion and transportation capacity of the runoff. The impact of these auxiliary data on concentrations measured at the Blind Beck outfall, when compared with *grab samples only*, was to increase mean SS concentration by over 300%, mean TP by 260% and SRP by 167%. However, mean NO_3 was decreased by 21%.

Based on the revised rating curves the effect of the event sampling on calculated SS load/yield is a 116% increase in 2010 and a 98% increase in 2011, compared with *grab sample only* values. TP yields are increased by 116% and 118%, and SRP yields by 72% and 97% in 2010 and 2011, respectively. There is no discernible difference for NO_3.

The 36 day mini-study where daily water samples were collected in order to compare *actual* SS and TP concentrations with values predicted by the revised rating curves showed that there was a good general agreement between the two. On average the rating curves predicted moderately higher SS and TP concentrations (compared with the measured concentrations), which mainly stemmed from over-predictions during high-flow events. The short experiment indicated that actual TP concentrations during residual flow conditions exhibit more fluctuation compared with SS concentrations. This may be due to soluble fractions of P affecting the TP concentrations whereas SS concentrations stay more stable during residual flow conditions as there is insufficient energy to transport particulate substances in suspension.

The shortcoming of the extrapolation method is that the response of sediment/nutrients is dependent on discharge. This was shown to not always be the case as a small two-day rainfall event, which caused an increase in SS and TP concentrations (based on *actual* measurements), did not elicit the same response in predicted concentrations, as there was no increase in discharge, despite the rainfall. This effect is stronger in the summer or when soil moisture deficit is high as rainfall-runoff ratios will be lower (meaning less or no increase in discharge) but contaminants may still be washed into the river. This is particularly important for nutrients as they can take soluble form and don't rely on physical detachment to become mobilised, thus can be transferred more easily. Domestic septic tanks have been identified as posing a significant threat to water quality in rural areas (Macintosh *et al.*, 2011; Withers *et al.*, 2012) and discharges from them would largely be ignored using the extrapolation method, as their impact will be greater during low flow conditions.

As the rating curves predicted higher SS and TP concentrations, on average, it could be expected that the extrapolation method would estimate higher loads also. However, the interpolation method gave the highest values for both SS and TP - 24% and 11% greater, respectively. Without measuring water quality determinand concentrations continuously (circa 15 minutes is most commonly used where such monitoring equipment is deployed (e.g., Cassidy and Jordan (2011), Johnes (2007), Owen *et al*. (2012)), along with corresponding discharge, it is impossible to determine whether low-resolution data, using interpolation or extrapolation calculation methods, over- or under-predict loads.

A stratified grab sampling regime, such as the one employed in Chapter 4, where efforts are made to collect samples during high discharge events (as opposed to fixed-time interval sampling, e.g., weekly, monthly, etc.) can provide sufficient data to estimate sediment and nutrient loads. However, it should be understood that these estimates would likely be underestimates, particularly for SS and sediment-phase nutrients due to the bias towards low and residual flow sampling. This Chapter has shown the value of including event-level sampling, using automatic-water samplers, in conjunction with a stratified grab sampling. Samples were taken at discharges up to 93% of the maximum-recorded discharge at the catchment outlet, compared with 89% in Chapter 4. Although this appears to be a small increase it is because a grab sample was collected from Blind Beck during high flows; however this was not the case for the majority of the other sub-catchments monitored in Chapter 4 (with percentages ranging between 30 and 50). If circa 90% could be achieved in all catchments then this would help improve the representativeness of the sediment/nutrient–discharge rating curves and thus increase the accuracy of the estimated pollutant loads.

5.4.4 Evaluation of a constructed mitigation feature

As previously stated, the wetland design and location were not decided as part of this study but the feature was included in the monitoring campaign in order to determine its impact of the sediment and nutrient regime of Blind Beck. Data collected at the wetland outfall suggested that SS, P and NO_3 concentrations and loads were significantly lower than those measured in Blind Beck (at Site 6). However, as there was no clearly identifiable surface water source/pathway to the wetland it is very difficult to put the wetland data into context, and it is perhaps unrepresentative to compare it to concentrations/loads recorded in the main river. The wetland appears to be its own source of sediment and nutrients as contiguous storms lead to exhaustion of SS and P. Sheep graze the area and it is likely that soil disturbance occurs in inter-storm periods, which provide rejuvenated sediment and associated nutrient sources. Nitrate concentrations were relatively low and responses to discharge were more attenuated in comparison, which indicates that NO_3 is being leached upwards through the soil as it becomes saturated during storm events. Leaching requires a sufficient flow of water to mobilise any available NO_3 and the presence of a lag between peak flow and an increase in surface water concentration may indicate the build-up of storm water in the soil as a prerequisite. On the grounds of water quality, specifically the ability of the wetland to reduce losses of SS, P and NO_3 from the Blind Beck catchment, the wetland cannot be deemed a success.

In the absence of a draw-off swale connecting Blind Beck to the wetland, the input of water to the RAF was attributed to a combination of direct precipitation, outflow from a spring that was believed to be a (possibly broken) field drain, other unidentified springs, and return flow from groundwater (as indicated by the piezometer data). After constructing and monitoring the behavior of the wetland for some time it was realised that significant intervention, with the potential for high financial cost, was needed to convey water from the beck to enter the feature. The wetland was just one of a number of planned RAFs for the Sykeside Farm area. A farm runoff management plan had been complied to complement a Higher Level Stewardship application that the farmer was undertaking at the time. However, due to unforeseen circumstances the HLS schemes lost financial backing and the plan to construct and monitor a suite of different RAFs on Sykeside Farm had to be halted. This forced mitigation experiments to be transferred to surrogate catchments, which provide the basis of Chapters 6 and 7.

5.4.4.1 Lessons learned

The grab sampling campaign carried out along the length of Blind Beck indicated that Sykeside Farm was the main source area for SS and P by a significant increase in concentrations between the sampling locations up- and downstream of the farm. A number of possible explanations have been given for why this may be the case but the main issues have been identified as accelerated bank collapses and poaching. As RAFs are designed to intercept polluted surface pathways during storm events, in order to slow, store and filter runoff, they are not the best solution to this particular problem. Where possible it is best to keep nutrients and sediment on the farm, or source area, they are derived from. Bank collapses and poached areas are obvious problems and a more appropriate strategy would to be to deal with the visible critical sources individually and the diffuse chronic ones with RAFs. There is a strong argument for the use of catchment walkovers in order to identify obvious critical pollution sources and the mitigation of such sources can be seen as an '*easy win*'.

A befitting choice of mitigation option(s) should rely on their source-pathway suitability. In this instance the most suitable strategy would be to stop animals entering Blind Beck (particularly on Sykeside Farm) to lower the occurrence of poaching and to help stabilise the banks. The most effective and cost-efficient way of doing this would be to introduce stock fencing. Owens *et al*. (1996) and Collins *et al*. (2010) both reported reductions in sediment losses from bank sources due to bank fencing. However, the farmer at Sykeside Farm was against this idea. This raises an interesting discussion point about the amount of conclusive evidence with which to force a landowner into action he doesn't like; i.e., why should a farmer be allowed to jeopardise the water quality of the stream with stock watering if it's a WFD issue?

The wetland experiment at Sykeside Farm also highlighted the importance of spatial scale for the application and functioning of RAFs to reduce losses of sediment and nutrients. Attempting to mitigate DWPA for the entire 9 km^2 Blind Beck catchment at Sykeside Farm is unrealistic due to the amount of runoff conveyed by the main channel at this scale. It would be more feasible to target smaller contributing areas, perhaps in the order <1 km^2. Larger features could perhaps be justified if they are part of a flood defense scheme. Despite the shortcomings of the wetland RAF to deliver water quality benefits it has solved a number of runoff related problems for the farmer by constraining surface water to one field (thus preventing the inundation of a neighbouring farm's field). It has also added temporary flood water storage capacity to the catchment and has created a new wetland habitat, although these gains are not quantified in this study.

No attempt was made in this study to mitigate NO_3 losses from the Low Hall sub-catchment, which was identified as the main contributor to export at the catchment outlet. It is highly likely, based on the findings of Ockenden (2010) that the majority of the NO_3 measured at the Low Hall outfall is derived from a groundwater source. Options available for reducing NO_3 concentrations would be: 1) to reduce inputs to the system at source, and/or 2) the treatment of the runoff in the channel itself. The first option would require significant investigation to identify the source of the NO_3 as a major proportion of the runoff in the catchment is believed to move via slow, subsurface pathways. Reducing NO_3 concentrations in the channel could be achieved using constructed wetlands and/or woodchip filters to provide conditions suitable for denitrification (please refer to Chapter 2.9.2 for details). However, these are both relatively high-cost options and may not be justifiable in this instance as the NO_3 input from the Low Hall catchment is diluted firstly by Blind Beck and then by the main River Eden; thus its impact is likely to be minimal under present hydrological conditions.

5.5 Summary

A grab sampling campaign carried out along the Blind Beck stream network allowed the source area of SS and P to be identified as principally Sykeside Farm. Although specific sources proved difficult to identify, site investigations revealed the existence of a significant number of in-channel and near-channel sources in the lower catchment, suggesting the destabilising effect of stock access to the stream.

Event-scale sampling at the catchment outlet allowed the examination of sediment and nutrient storm dynamics and thresholds of activation**.** Suspended sediment and TP was activated by high flows, most likely due to bank erosion. Nitrate and SRP appear to be mobilised by throughflow (subsurface) pathways. The employment of an auto-sampler meant that water samples were taken across a wide range of discharges and produced discharge-contaminant rating curves that are more representative (compared with those Chapter 4), of the sediment and nutrient regime of the catchment; although more scatter was added to the higher-value end of the rating curves. As a result of the revised rating curves, both mean concentrations and annual loads/yields of SS and P for Blind Beck were increased when compared with the values calculated in Chapter 4. There was no discernible difference in NO_3 concentrations/loads.

A 36 day long experiment allowed the comparison between daily sampling (actual concentrations of SS and TP) with corresponding concentrations estimated from discharge using the appropriate rating curve. An interpolation method was used with the former and an extrapolation method the latter to calculate total loads from the experimental period. Despite the extrapolation method slightly over-estimating concentrations on average, the interpolation method gave the highest total load estimates.

The plan to deploy a number of RAFs in the Blind Beck catchment, at Sykeside Farm, could not be fulfilled. A single modified wetland RAF was completed but the absence of a clearly defined inflow point made it difficult to contextualize the sediment and nutrient concentrations measured at the wetland outfall.

The wetland had two main reasons for being unsuccessful at meeting its water quality amelioration aims, besides not being the most appropriate way of treating the SS and P source-pathways identified in the catchment (a more appropriate strategy would to be to deal with the visible critical sources individually). Firstly it was too far away from the main Blind Beck channel (to allow the hydrologic connection to be made between Blind Beck and the feature a large swale is needed to convey high flows across the land surface). Secondly it was administered at an unsuitable scale (i.e., too far downstream); the size of feature needed to treat runoff from a 9 km^2 catchment is unfeasibly large. A contributing area of no greater than 1 km^2 is recommended, but this issue will require further investigation.

However, the wetland functions well as surface water management feature, reducing the inundation of surrounding fields, and also as a wetland habitat. Importantly many vital positive lessons have been learned during this study and several RAF design aspects improved, for example the use many small RAFs close to runoff/pollution sources as opposed to one large downstream feature. The aim of the Chapters 6 and 7 is to apply these to ensure an appropriate, rigorous design is implemented.

6. DWPA mitigation case study one: The Belford catchment

6.1 Introduction

The two successive chapters are concerned with the mitigation of DWPA. Moreover, they provide case studies of where RAFs have been used to target overland flow pathways during storm events in order to reduce losses of SS, P and NO_3 in agricultural runoff.

The Belford Burn catchment was selected due to the presence of existing RAFs constructed to lower flood risk in the town of Belford, Northumberland. However, as the catchment eventually discharges into a highly valued ecological zone (Budle Bay), which was experiencing macro-algal blooms consistent with eutrophication, there was already a move to assess whether existing RAFs were having an effect on sediment/nutrient losses. This chapter has two components; the first part (Section 6.3) characterises the SS, P and NO_3 concentrations in a sub-catchment of the Belford Burn catchment and assesses the impact of two existing flood-related RAFs on the sediment/nutrient loss regime. The second part (Section 6.4) describes the design, construction and performance of a multi-stage RAF built specifically for water quality amelioration – with the design informed by the findings of the first work component.

The aims of this chapter are thus:

1. To characterise the sediment/nutrient regime of an intensive agricultural sub-catchment and evaluate the effectiveness of RAFs designed primarily for flood alleviation purposes to reduce losses of SS, P and NO_3.

2. To construct and monitor trial RAFs specifically designed to mitigate DWPA.

6.2 Materials and methods

6.2.1 Site description

The Belford Burn catchment (5.9 km^2) is located upstream of the village of Belford (OS Grid Reference NU-339107), in northeast England (Figure 6.1). Catchment elevations range from 53 to 208 m AOD, with the majority below 200m (lowlands). Bedrock is chiefly Alston formation, a mix of Carboniferous limestone and sandstone. The western upper slopes are formed of hard impermeable sandstone of the Fell Sandstone formation and an outcrop of the dolerite Whin Sill exists in the east, to the north of the town. Superficial geology is dominated by Devensian till, which covers the majority of the catchment, excluding the elevated Fell Sandstone and Whin Sill outcrops. The dominant (circa 95% coverage) soils are of the Dunkeswick Association

- deep slowly permeable fine loamy and clayey soils formed in glacial drift (Jarvis *et al.*, 1984). This has resulted in extensive artificial under-drainage of the catchment using tile drains. Some small areas of peat occur in the upper catchment. Land use is predominantly agricultural, split between arable in the eastern (lower) half and pastoral in the western (upper) half of the catchment. The small portions of steeper slopes host rough grazing and coniferous plantations and there are areas of mixed deciduous and coniferous woodland mainly along the stream corridor (Figure 6.2). Mean annual rainfall is 695 mm (Wilkinson *et al.*, 2010).

Hydrometeorological data

River stage was recorded at 15 minute resolution using pressure transducers at several locations (R1-R4, Figure 6.1) in the Belford catchment (as part of another study – Nicholson (2013)) and discharge was calculated using a hydraulic model. The model was calibrated using manual discharge measurements and constrained with channel gradient and cross-sectional measurements to extrapolate discharges beyond bank full levels and improve estimates for high flows. Discharge recorded at R3 is used in this study to demonstrate the prevailing hydrological conditions during sampling, as continuous discharge is not measured in the Lady's Well sub-catchment for the entire study period. The R3 gauge was selected as it was the most complete and reliable record. Rainfall is recorded using a tipping bucket rain gauge at the head of the catchment (Figure 6.1).

The Lady's Well sub-catchment (34 ha) located in the northeast of the Belford catchment is highlighted in Figure 6.1; it provides the only permanent surface tributary to Belford Burn. The sub-catchment was selected in which to test a number of RAFs for their effectiveness in reducing DWPA (further details are given below). The catchment has previously been reported to have a high capacity to absorb precipitation, but a rapid (short lag time), high amplitude discharge response once soil storage capacity is exceeded, particularly in the winter when it is more likely that soil saturation conditions are reached (Palmer, 2012). It could be considered to be high risk in terms of SS (and associated nutrients) losses due to being a predominantly lowland catchment (susceptible land use) with slowly permeable soils. As the Belford catchment discharges into a sensitive, low-energy receptor (Budle Bay - more details below) where there is potentially a long retention time, the impact of sediment (and associated nutrients) may be highly important.

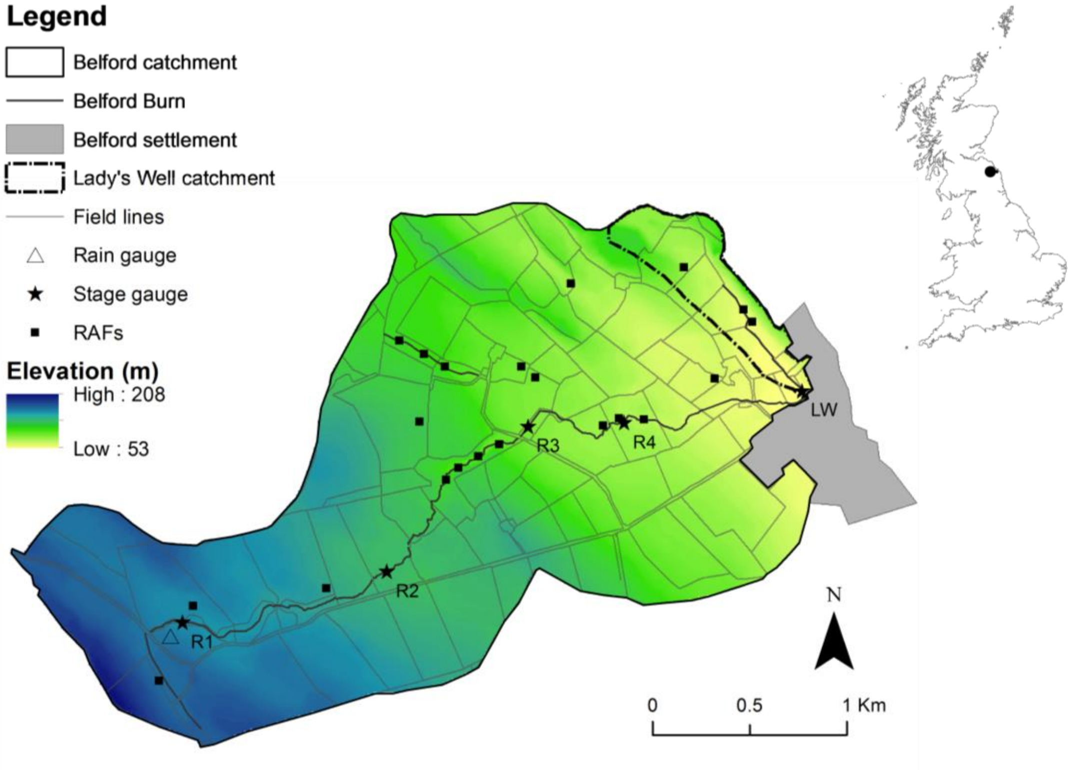


Figure 6.1: Map of Belford Burn catchment and DEM.

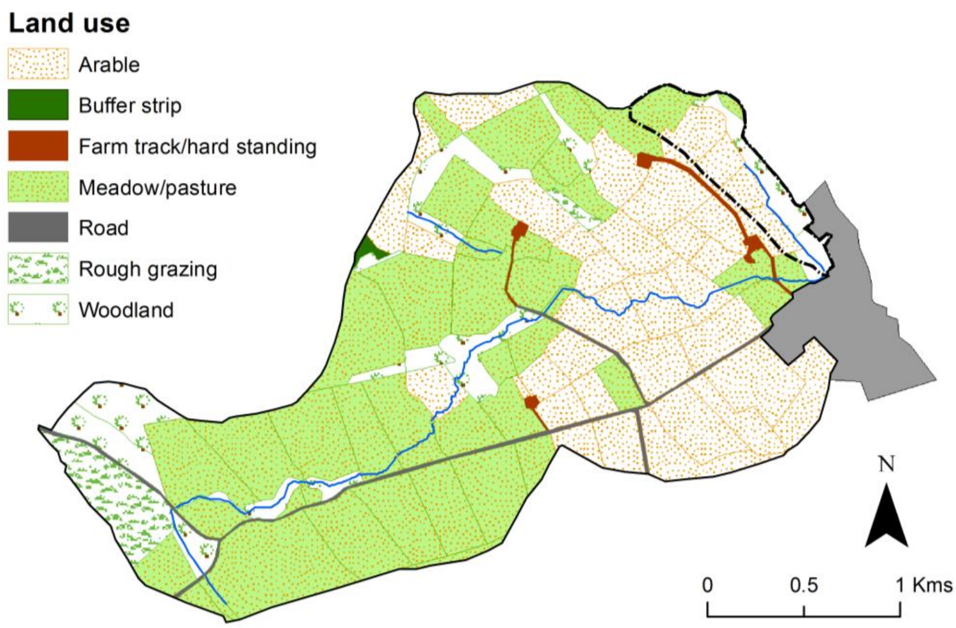


Figure 6.2: Land use map of the Belford catchment (as of spring 2011).

6.2.2 Catchment issues

Flooding

Flooding in Belford presents a risk to thirty one properties, a caravan park and two major transport links; the A1 and the East Coast Mainline (Nicholson *et al.*, 2012). Recent flood events were recorded in 1997, 2002, 2005 and 2007 and all caused damage to properties, infrastructure and/or local businesses. Factors increasing the risk of flooding in the town include the constriction of the main river channel through the centre of the town by walls and bridges, and increased runoff due to relatively intensive agricultural land use and associated soil compaction/infiltration reduction effects (Wilkinson *et al.*, 2010). Traditional flood defences were not available for Belford due to the town failing to meet the criteria for Grant-in-Aid funding. As a result an alternative scheme was proposed and funded by the EA's Local Flood Levy to construct a number of RAFs to temporarily store and attenuate flood peak runoff in the catchment upstream of the town. The scheme was designed and implemented by the Proactive team at Newcastle University in collaboration with the EA (Wilkinson *et al.*, 2010). Since the project began in August 2008 approximately 35 RAFs have been constructed in the catchment. A detailed account of many of the features is provided at [http://research.ncl.ac.uk/proactive/belford/] and by Nicholson *et al.* (2012).

Water quality

Belford Burn flows in a roughly easterly direction and joins two other streams east of Belford Town to form Ross Low. These watercourses along with the Waren Burn to the south are collectively known as the 'Lindisfarne Coastal Streams' and they discharge into Budle Bay (OS Grid Reference NU-150356) on the northeast coast of England (Figure 6.3). Budle Bay is located within a SAC under the Habitats Directive (92/43/EEC) and forms part of the Lindisfarne Special Protection Area (SPA) protected under the Birds Directive (79/409/EEC). It is also designated a Natura 2000 site and Ramsar wetland. The intertidal mudflats and sand flats of the ecosystem support a diverse infauna, which in turn supports internationally important populations of waterfowl (Palmer, 2012). The bay occupies an area of approximately 315 ha and has a total catchment area of circa 94 km^2. The town of Belford is the only sizable settlement in the Budle Bay catchment with a population of circa 1000 persons.

Palmer (2012) described how over recent decades increasing summer blooms of macro-phytic algae have occurred in Budle Bay, particularly in the area of the bay surrounding the mouth of the Lindisfarne Streams. The algae forms thick mats, which can inhibit the growth of the native sea grass and has the potential to negatively impact the food source for wading birds. This has resulted in the ecological status of Budle Bay being graded as 'moderate' according to the WFD status indicators for England and Wales (UK TAG, 2008). Following the designation of the bay as a SAC in 2000, a Habitat Management Plan drawn up by NE acknowledged that there was indeed a freshwater eutrophication problem and that DWPA was the main concern. The River Basin Management Plan (RBMP) for Northumbria River Basin District (Environment Agency, 2009) reported that the Budle Bay SPA was not achieving its environmental objectives and attributed this to a water quality problem caused by agricultural runoff.

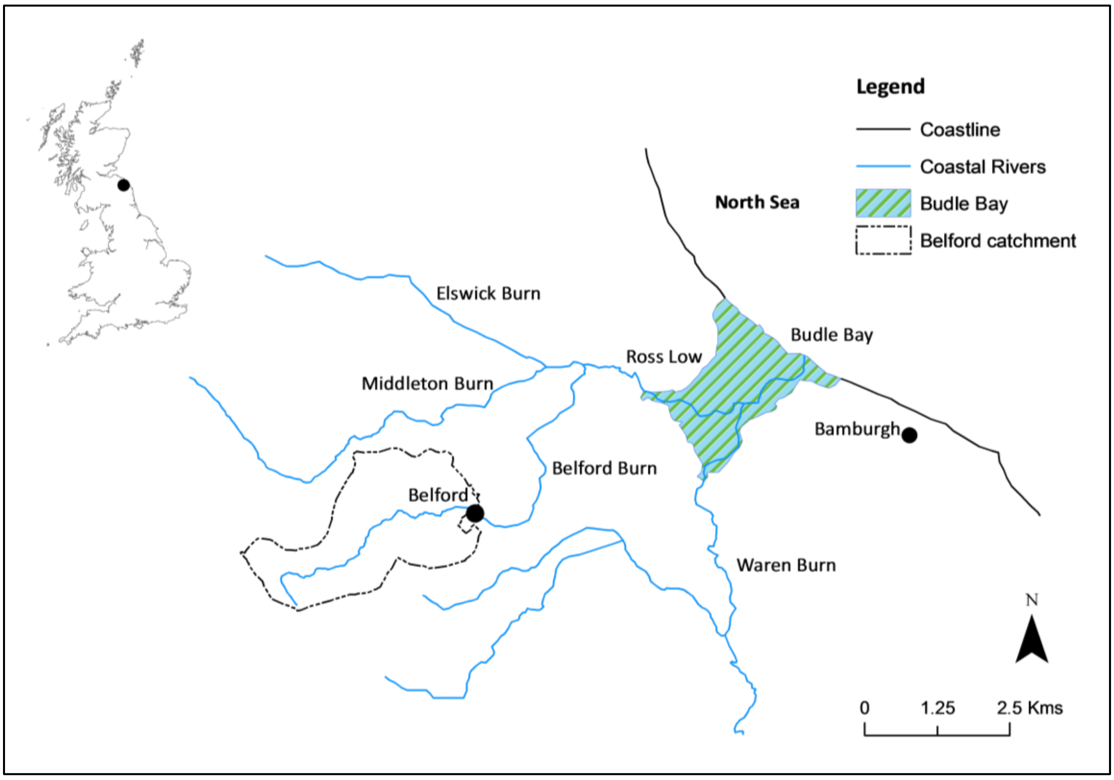


Figure 6.3: Map of Budle Bay catchment.

6.2.3 The study area: Lady's Well sub-catchment

The Lady's Well sub-catchment (34 ha) located in the northeast of the Belford catchment (Figure 6.4*a*) was selected in which to test a number of RAFs for their effectiveness in reducing DWPA. This involved two separate components; firstly a grab sampling campaign to characterise the sediment and nutrient concentrations in the catchment alongside evaluation of the capacity of two existing RAF (designed and constructed principally for flood alleviation purposes) to reduce SS, P and NO_3 losses. One RAF was evaluated by Palmer (2012) for its ability to retain sediment and will also be referred to here. The second component involved the construction and monitoring of a new multi-stage RAF designed specifically for DWPA mitigation.

Elevation ranges from 53 m AOD in the southeast to 111 m AOD in the northwest of the Lady's Well catchment (Figure 6.4*b*). Steep slopes running along the north-eastern side of the catchment represent the Whin Sill outcrop. Similarly to the rest of the Belford catchment artificial drainage has been installed, as a result of the clay-rich soil subsurface horizons. This consists of a herringbone system where collector drains (6'' clay ware tiles) are aligned along the main slope and smaller diameter lateral drains run perpendicular to them (Figure 6.4*b* and *c* show the location of the collector drains). The Lady's Well stream is fed chiefly by one of these drains and emerges in a surface drainage ditch approximately halfway along the length of the catchment, with a contributing area of 15 ha. The stream has an average slope of 4.3%, falling 55 m over a distance of 1270 m (Figure 6.4*b*). Land use in the catchment is predominantly arable rotation – mainly cereals with some fields used for cattle and sheep grazing.

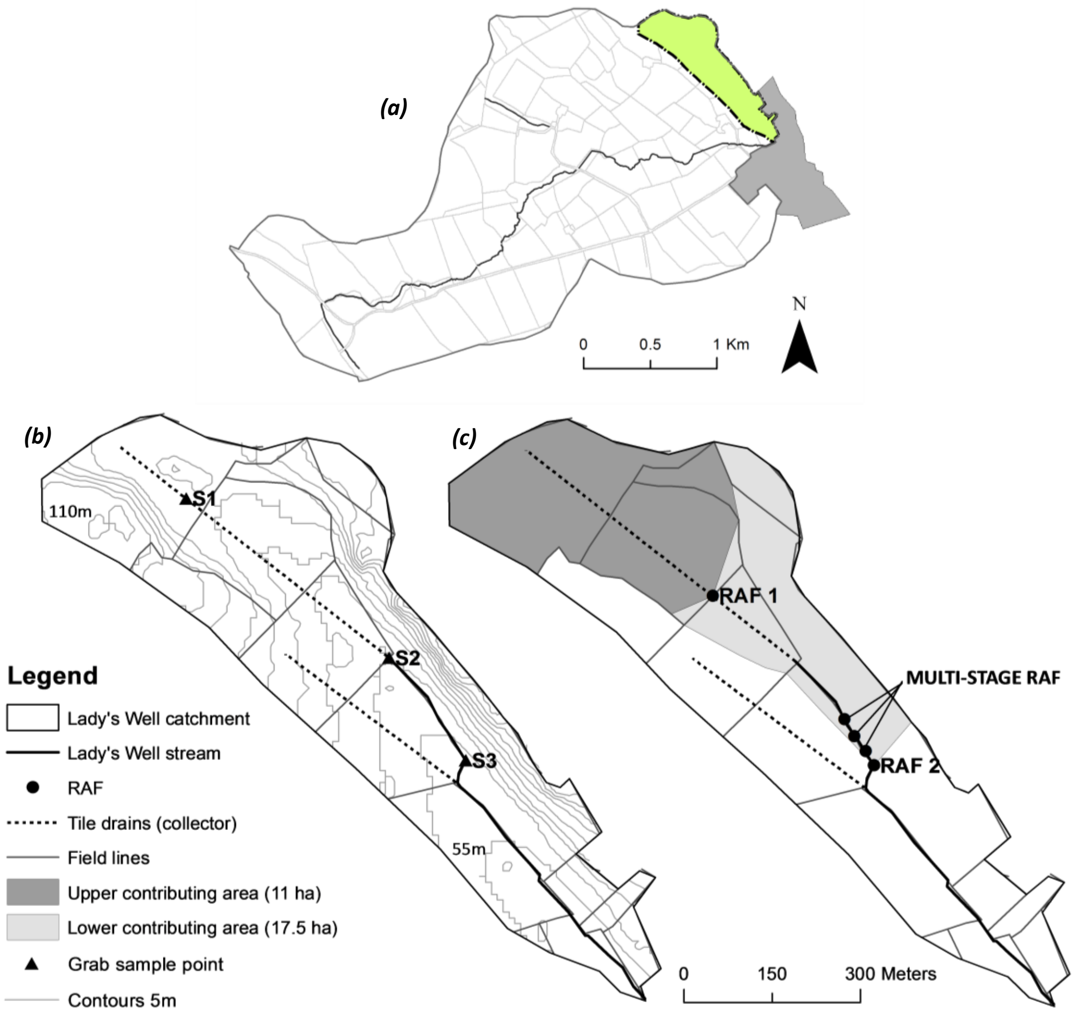


Figure 6.4: Maps of the Lady's Well sub-catchment: *a*) location in the Belford catchment, *b*) grab sample locations and 5m contours, and *c*) RAF locations.

6.3 Sediment/nutrient characterisation and evaluation of existing RAFs

6.3.1 Grab sampling campaign

The water quality in Lady's Well was characterised between January 2010 and February 2013 using a low-intensity synchronous grab sampling regime. Samples were collected from three locations (S1, S2 and S3 - Figure 6.4*b*). S1 is a drain inspection point at the top of the catchment; S2 is where the subsurface tile drain discharges into the surface ditch, and S3 is from the ditch 300 metres downstream of S2 – just downstream of RAF 2 (Figure 6.4*c*). The sampling and laboratory techniques described in Chapter 3.3 were used to determine SS, P and NO_3 concentrations. Samples from all three locations were collected within 20 minutes of each other, therefore can be considered to provide a representative 'snapshot' of the determinand concentrations across the catchment. At this time no flow measurement equipment was installed in the sub-catchment so the data only exist as concentrations. Section 6.3.3 contains a discharge (measured at R3) and precipitation record for the Belford catchment, which provides a representation of the hydrological conditions on sample collection dates. Few samples were collected during the summer, as there was no observed drain flow for the majority of the time.

6.3.2 Flood management RAFs evaluated for DWPA mitigation

RAF 1 (see Figure 6.4*c* for location), constructed in November 2009, is a within-field retention bund. It is built across the main valley thalweg (the line following the lowest part of the valley) of an arable field (4.1 ha) and is designed to intercept and temporarily store overland flow during storm events (Plate 6.1). The field has an average gradient of approximately 4 degrees and its land cover (predominantly winter wheat during the study period) makes it susceptible to soil erosion. The bund (maximum of 1 m high) provides a storage capacity of approximately 500 m^3. The bund has a 220 mm diameter outlet pipe installed at mid-height to help prevent over-topping and possible erosion of the bund. It also allows the feature to drain in several hours; this is important in the event of a second flood peak. The RAF also doubles as a raised farm track, which prevents the farmer trafficking this previously waterlogged area. Construction was carried out by the farmer using locally-sourced materials, thus incurred relatively low cost. While its ability to retain overland flow is obvious (Plate 6.1) its sediment/nutrient trapping capabilities were more difficult to quantify. A pressure transducer was installed in RAF 1 on 28/10/2011 (by Nicholson (2013)) to record when (and the depth of) water was held behind the bund during a storm event; this data provides an insight into when overland flow is occurring in the Lady's Well catchment, as this is the principle pathway by which water enters the feature.

Palmer (2012) was able to calculate the mass of sediment retained by RAF 1 following a storm event in January 2011 by quantifying the sediment fan left behind the retention bund (Plate 6.2), It was estimated that 0.99 tonnes of sediment, mainly silt/clay and fine-sand, was captured. Trapped sediment becomes re-incorporated back into the topsoil during annual ploughing. However, it was acknowledged that a proportion of fine sediment was lost via the feature's outlet pipe and also bypassed by subsurface drains.

Plate 6.1: RAF 1, a within-field retention bund storing overland flow during a storm event (17/10/2012). Dashed arrow indicates direction of overland flow.

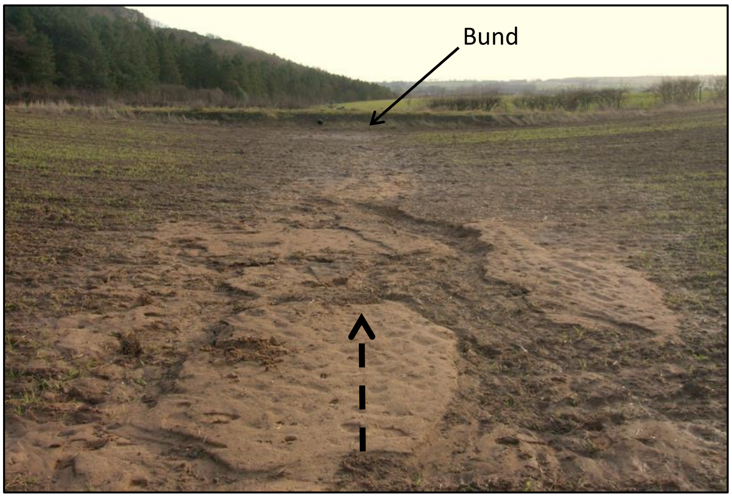


Plate 6.2: Sediment retained in RAF 1 following January 2011 storm (photograph taken on 19/01/11). Dashed arrow indicates direction of overland flow.

RAF 2, constructed in November 2009, is a flood RAF but was designed to double as a sediment trapping pond. It has two components (Figure 6.5 - see Figure 6.4*c* for location): a permanent on-line pond feature to retain sediment (Plate 6.3) and a higher level separate crescent-shaped pond to store flood water once the pond over-tops. The sediment-trapping pond has a capacity of approximately 200 m^3. After construction the pond quickly began to fill with sediment and a delta could be seen developing at the inlet. A pressure transducer was installed in the pond to record stage on 26/02/2010. Paired automatic samplers were deployed at the inlet and outlet of the feature to determine whether it was retaining SS, P and NO_3 during storm events. The samplers were programmed to take a sample every hour and a float switch located next to the pressure transducer initiated sampling.

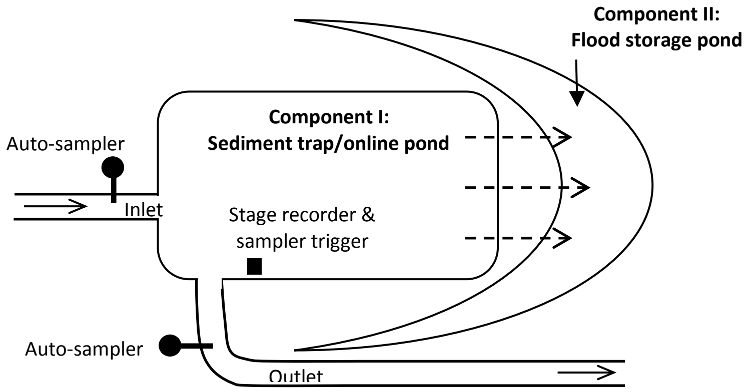


Figure 6.5: Schematic of RAF 2 (not to scale).

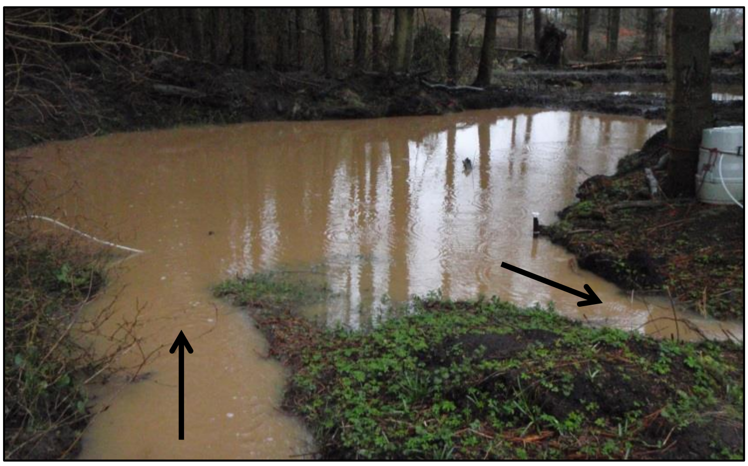


Plate 6.3: RAF 2 – sediment trap/online pond component during a storm event (sampled with auto-samplers – see data in 6.3.3). Black arrows indicate direction of flow.

6.3.3 Results

Hydrology

The flow regime of Belford Burn (measured at R3) displays considerable inter-annual variability in yield (Figure 6.6; Table 6.1). This may be explained by a more limited runoff response to precipitation during the growing season (April-September) than in winter months. During 2010, runoff events are limited to the winter period, which coincides with the greatest rainfall (proportionally) while in 2012 the largest events are in response to extreme daily rainfall events occurring in summer months. This is reflected in the higher percentage runoff value in 2010 compared with 2012 (70% and 64%, respectively). 2011 yielded the lowest total runoff, which was in response to the lowest total precipitation. Proportionally the greatest amount of precipitation fell in July-August but no obvious runoff response was observed (apart from on 11/08/2011), probably due to the effect described above. The 2011/2012 winter period received relatively little precipitation with no significant discharge events occurring between September 2011 and April 2012 (Figure 6.7).

Table 6.1: Belford Burn annual runoff and precipitation parameters 2010-2012.

Year	Annual runoff (mm)	Annual precipitation (mm)	Percentage runoff
2010	617	883	70
2011	352	646	54
2012	669	1049	64

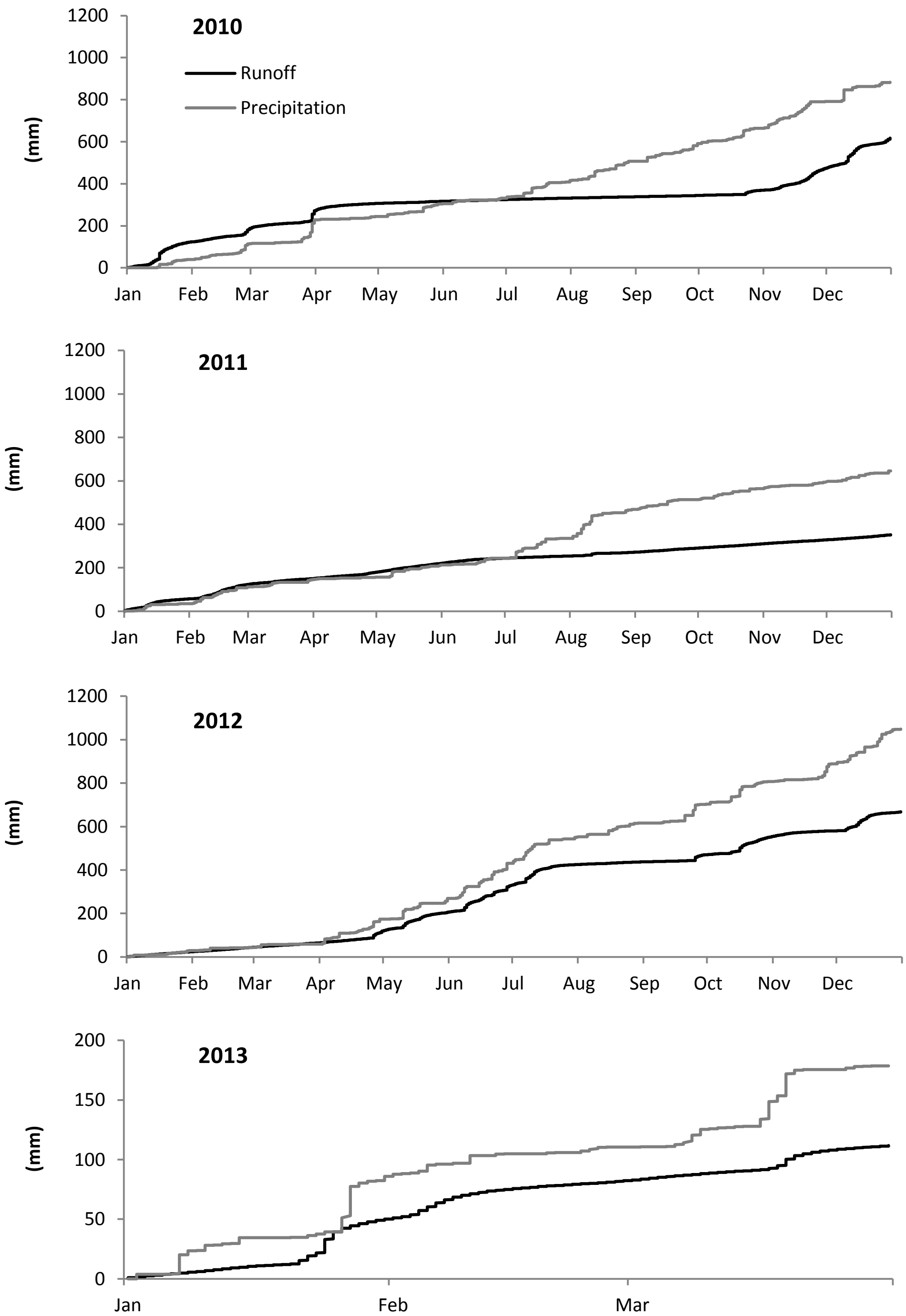


Figure 6.6: Belford Burn cumulative runoff and precipitation (mm) 2010-2013.

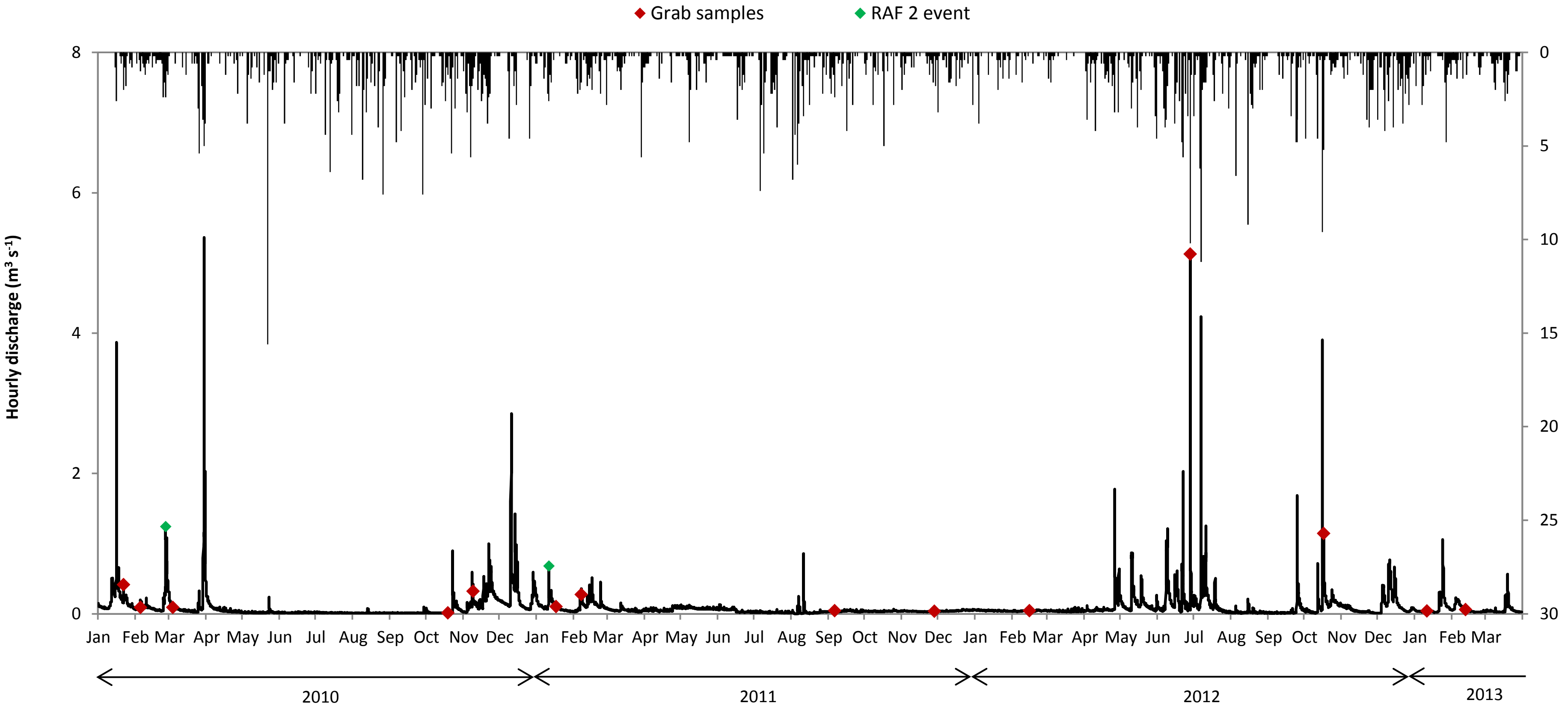


Figure 6.7: Discharge (measured at R3) and precipitation record for 2010-2012/13, with water quality sampling dates marked.

Water stage recorded in RAF 1 is depicted in Figure 6.8 and can be used to signify when significant overland flow occurred in the wider catchment (e.g., Plate 6.4). This indicates that there were no major incidents of overland flow during the 2011-2012 winter period. The first major storm event in 2012, following the prolonged dry spell, occurred on 26th April. The unusually wet summer of 2012 caused the feature to fill on several occasions, most significantly in June and July. The 2012-2013 winter period displays behaviour in line with what would be expected under normal conditions with numerous occurrences of overland flow in response to proportionally lesser precipitation (in comparison with the summer), in marked contrast to the 2011-2012 winter.

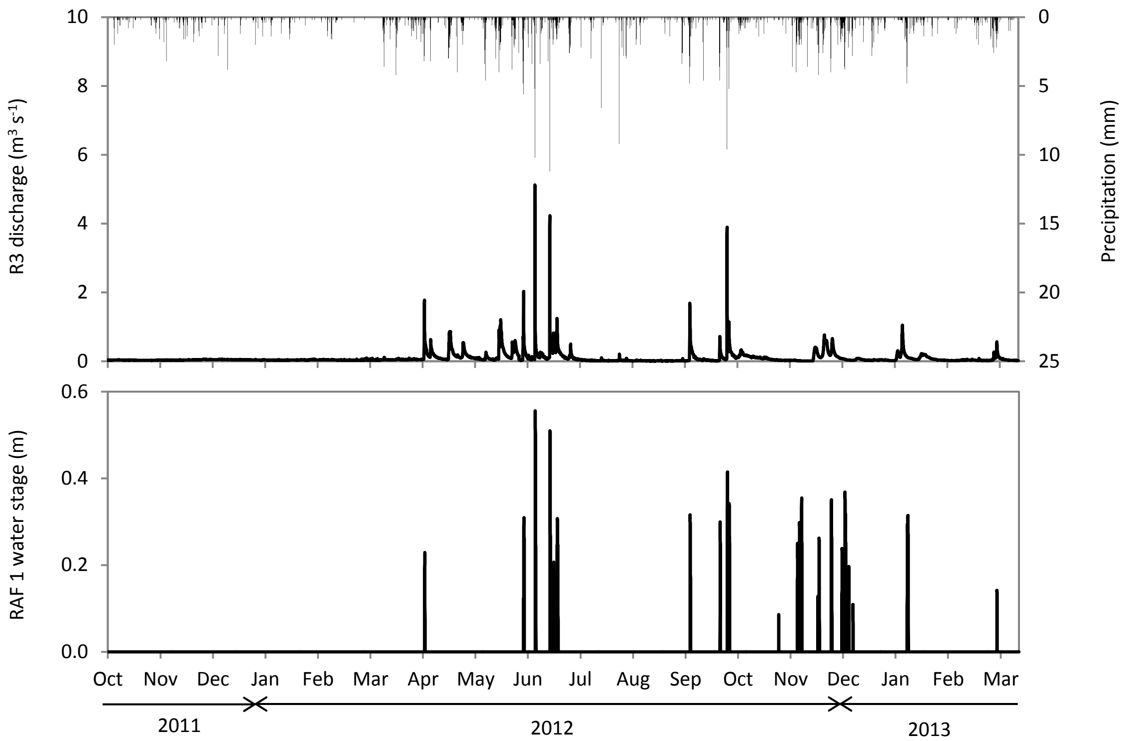


Figure 6.8: RAF 1 water stage (bottom panel) for October 2010-March 2013, compared with precipitation and discharge response recorded at R3 (top panel).

Plate 6.4: Overland flow in the Lady's Well catchment; photograph taken on 17/10/2012 at S2 looking northwest towards RAF 1. Dashed arrows indicate direction of overland flow.

Grab sample campaign

Table 6.4 contains data from the grab sampling survey, collected on fifteen separate occasions between January 2010 and February 2013 (see Figure 6.7). Samples cover a wide range of hydrological conditions ranging from days of no rainfall and dry antecedent conditions, to heavy rainfall preceded by wet antecedent conditions. On every date all determinand concentrations increased significantly between S1 and S2 (Table 6.2 – for sample locations refer to Figure 6.4 *b*). The highest SS and P (TP and SRP) concentrations (and largest increase over S1 concentrations) were recorded at S2 on the days of greatest precipitation, most notably 28/06/2012 and 17/10/2012. On 17/10/2012 14.6 mm of rain fell, which followed 54.4 mm of rain over the previous five days; SS and TP concentrations of 1315 mg l^{-1} and 1.34 mg l^{-1} were recorded, respectively – the highest overall. Overland flow occurred in the catchment on this date (see Plate 6.4). The highest SRP concentration of 0.45 mg l^{-1} was measured on 28/06/2012. SS concentrations decreased on average between S2 and S3 but on approximately half of the sample dates TP and SRP concentrations increased slightly. The highest NO_3 concentrations did not correspond with those of SS and P, but occurred on days of little rainfall that followed a prolonged wet period, e.g., 19/10/2010 and 28/06/2012. The highest NO_3 concentration of 44.6 mg l^{-1} was recorded at S3 on 19/10/2010. Similarly to P, NO_3 concentrations were found to be higher at S3 than S2 on approximately half of the sample dates.

To establish if there was a seasonal difference in determinand concentrations, the data were divided according to whether samples were collected in the autumn or the winter (Table 6.4). As the majority of samples were collected over the winter period (as the drains seldom flowed in the summer) autumn was taken as samples collected in September, October and November (as well as one sample collected in June 2012). For all determinands across all three sample locations, average concentrations were higher in the samples collected in the autumn months. However, based on data collected at S2, only the differences between autumn and winter SS concentrations were significant (at the 95% confidence level - Table 6.3).

Table 6.2: Paired T-test results for water quality data recorded at S1 and S2.

Determinand	n	*T-value*	*P-value*
SS	13	-3.21	0.007
TP	13	-4.35	0.001
SRP	11	-3.57	0.005
NO_3	12	-3.38	0.006

Table 6.3: Two-sample T-test results for autumn and winter data recorded at S2.

Determinand	*T-value*	*P-value*
SS	2.47	0.029
TP	1.74	0.107
SRP	1.17	0.265
NO_3	2.04	0.066

Table 6.4: Grab sample data.

Date	Rainfall on day/antecedent conditions	Determinand concentration (mg l^{-1})											
		SS			TP			SRP			NO_3		
		S1	S2	S3	S1	S2	S3	S1	S2	S3	S1	S2	S3
22/01/2010	8.8 mm rain/following 19.6 mm over 7 days	12.4	297.5	284.0	0.054	0.661	0.621	0.021	0.256	0.215	6.8	10.5	9.2
05/02/2010	5.6 mm rain/following 4 mm over 3 days		275.0	244.0		0.392	0.375		0.125	0.103		14.9	14.7
04/03/2010	No rain/4 days after large storm	5.6	40.0	32.0	0.031	0.192	0.221	0.027	0.121	0.146	8.5	12.0	10.9
19/10/2010	1.2 mm rain/following week of steady rain	44.0	400.8	283.6	0.062	0.249	0.329	0.054	0.110	0.132	14.7	40.8	44.6
09/11/2010	2 mm rain/following 39.8 mm over 7 days	10.6	475.0	302.2	0.065	0.488	0.449	0.038	0.122	0.143	5.6	8.8	9.4
18/11/2010	3.6 mm rain/following 26.2 mm over 5 days	17.5	112.7	80.6	0.060	0.200	0.175	0.016	0.069	0.073	7.4	12.4	13.6
17/01/2011	No rain/4 days after storm	10.7	64.0	75.0	0.056	0.286	0.168	0.033	0.054	0.054	13.7	15.8	16.3
07/02/2011	6.8 mm rain/following 20.6 mm over 5 days	12.5	312.2	305.0	0.042	0.388	0.394	0.028	0.187	0.195	6.6	9.5	9.9
06/09/2011	5 mm rain/following week of steady rain	34.8	748.8	722.0	0.061	0.812	0.830						
28/11/2011	0.2 mm rain/following week of steady rain	12.0	137.0	131.0	0.052	0.446	0.395	0.019	0.184	0.171	12.0	19.1	19.0
15/02/2012	No rain/following relatively dry period	5.0	37.0	29.0	0.044	0.303	0.285	0.024	0.103	0.096	6.1	11.2	10.5
28/06/2012	27.4 mm rain/following 45 mm over 7 days	42.0	1188.0	1032.0	0.109	1.178	1.211	0.041	0.454	0.461	10.7	23.7	25.2
17/10/2012	14.6 mm rain/following 54.4 over 5 days	52.5	1315.0	1044.0	0.066	1.336	1.201		0.316	0.227		7.3	6.9
11/01/2013	No rain/following 4.4 mm previous day		24.0	17.0		0.089	0.072		0.023	0.018	7.7	9.4	9.1
12/02/2013	No rain/following 6.4 mm two days previous	6.0	19.0	24.0	0.035	0.075	0.059	0.011	0.029	0.032	5.5	12.0	10.9
	n	13	15	15	13	15	15	11	14	14	12	14	14
	Mean	**20.4**	**247.2**	**200.8**	**0.053**	**0.357**	**0.341**	**0.031**	**0.131**	**0.133**	**9.0**	**15.6**	**16.1**
	σ	16.6	415.0	348.4	0.019	0.376	0.366	0.012	0.120	0.111	3.2	8.7	9.8
		Autumn											
	n	6	6	6	6	6	6	5	5	5	5	5	5
	Mean	**26.8**	**510.4**	**425.2**	**0.068**	**0.562**	**0.565**	**0.034**	**0.188**	**0.196**	**10.1**	**21.0**	**22.4**
	σ	15.2	406.6	373.3	0.021	0.371	0.384	0.016	0.154	0.152	3.6	12.5	13.8
		Winter											
	n	6	8	8	6	8	8	6	8	8	7	8	8
	Mean	**8.7**	**133.6**	**126.3**	**0.044**	**0.298**	**0.274**	**0.024**	**0.112**	**0.107**	**7.8**	**11.9**	**11.4**
	σ	3.5	134.6	127.6	0.010	0.190	0.188	0.008	0.080	0.073	2.8	2.3	2.6

RAF 2 event sampling

Two separate storm events were sampled at both the inlet and outlet of RAF 2 with the auto-samplers during the monitoring period; Table 6.5 contains summary data for the two events (26/02/2010 and 10/01/2011). Figure 6.9 shows data recorded in a 19 hour event on 26th February 2010. Antecedent conditions were wet following steady rainfall (22 mm over the previous 7 days) before 21 mm of rain fell on the 26th, with a peak rainfall intensity of 3 mm hr^{1}. At the onset of sampling both SS and TP concentrations were slightly higher at the inlet than the outlet but only for the first 3 hours. After this, concentrations were higher at the outlet. SS peaks at 698 mg l^{-1} (at the outlet), which coincides with the maximum pond stage after 4 hours. Total P concentration peaks just after the maximum pond stage at 1.22 mg l^{-1} (inlet) but the highest TP concentration of 1.24 mg l^{-1} was recorded 2 hours later at the outlet, at the start of the recession. Suspended sediment concentrations remain higher at the outlet until the 11th hourly sample, after which higher concentrations are recorded at the inlet. A similar pattern happens with TP.

Nitrate concentrations differ only very slightly between the inlet and outlet for the entire sampling duration. The overall pattern is a slight reduction during the rising limb followed by a steady increase during the recession. The NO_3 concentration of 11 mg l^{-1} may not represent the peak as higher concentrations were recorded in the grab sample campaign (Table 6.4). In this instance the sampling sequence was stopped due to a fall in pond stage, therefore less of the recession was recorded. Overall, during this event, there was a net loss of SS and NO_3, 2.3 and 2.5% respectively, and a small 1.6 % net retention of TP; these percentages are based on concentrations alone as loads could not be calculated.

A 17 hour event was recorded on 10th January 2011; Palmer (2012) measured the mass of sediment retained behind RAF 1 (described above) following this same event. The synchronous data collected at the pond inlet and outlet demonstrated a very similar pattern as described above, whereby no significant net retention of sediment/nutrients occurred. At peak pond stage the highest concentrations of SS and TP were recorded: 1011 and 1.702 mg l^{-1}, respectively.

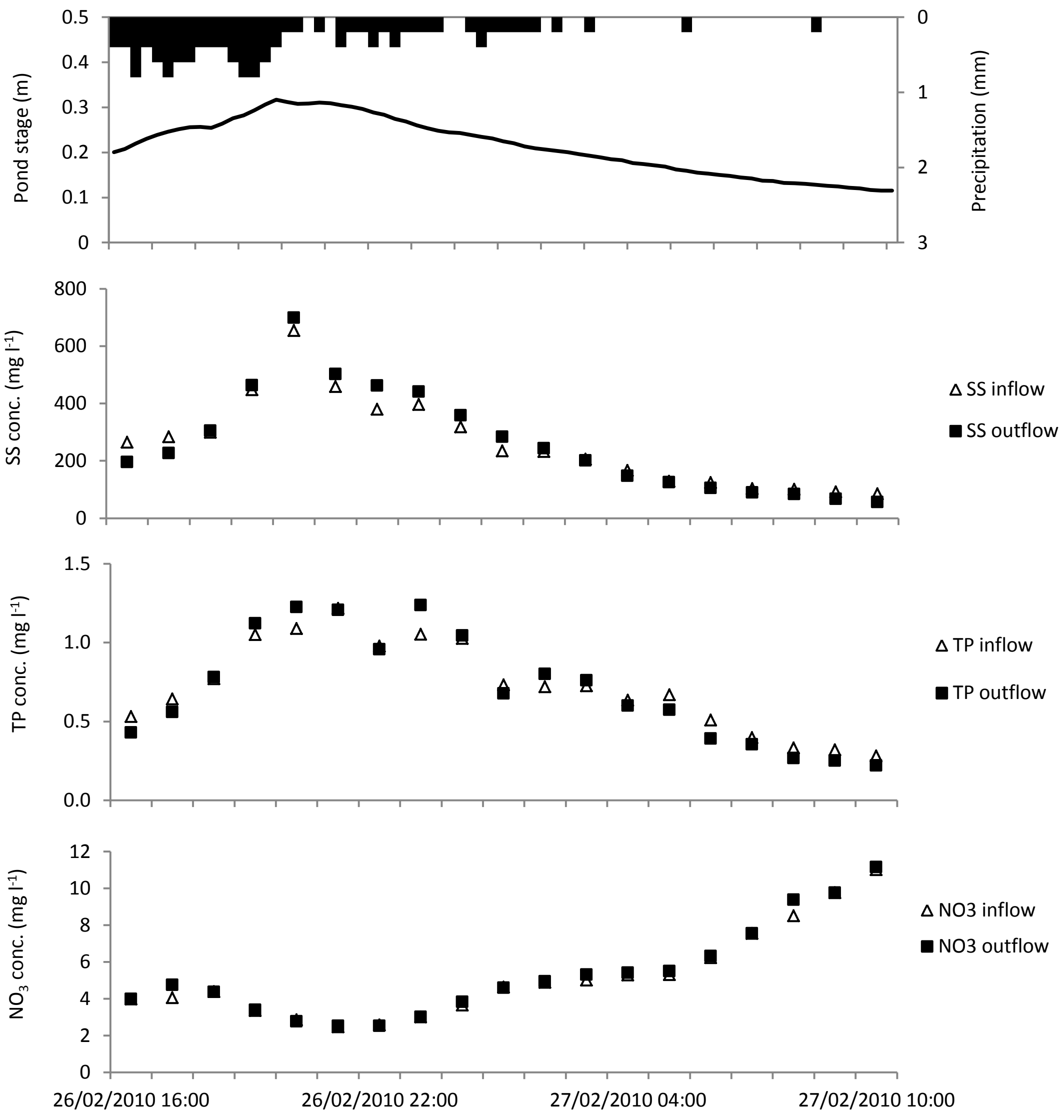


Figure 6.9: RAF 2 inflow and outflow SS, TP and NO_3 concentrations during a storm.

Table 6.5: Summary of results recorded at RAF 2

Date/time	Duration (hrs)	Determinand retention (%)					
		SS		TP		NO_3	
		Range	Mean	Range	Mean	Range	Mean
26/02/2010 16:00	19	-21.6 - 34.1	-2.3	-17.6 - 23.3	1.6	-17 - 3.5	-2.5
10/01/2011 23:00	17	-11.4 - 15.8	0.3	-11.8 - 21.9	-1.2	-3.3 - 5.2	-1.7

6.3.4 Discussion

Sediment and nutrient loss – patterns and processes

The grab sample and auto-sampler results provide evidence of the different processes contributing to the export of sediment and nutrients from the Lady's Well catchment. The grab sample data demonstrate that relatively high concentrations of SS, P and NO_3 occur in the catchment throughout the year (apart from the summer when the drains/ditch often stops flowing). On nearly every occasion TP and SRP concentrations exceeded the EA recommended maximum concentrations of 0.1 mg l^{-1}, and SS concentrations as high as 400 mg l^{-1} were recorded that significantly surpass the 25 mg l^{-1} acceptable threshold prescribed under the Freshwater Fish Directive (2006/44/EC). An average NO_3 concentration of 16.1 mg l^{-1} was recorded, with a maximum of 44.6 mg l^{-1}. Although below the 50 mg l^{-1} prescribed under the Drinking Water Directive (98/83/EC), Skinner *et al.* (2003) and Hickey and Martin (2009) argue that such concentrations are potentially of ecological significance.

The results of the grab sampling campaign suggest that mean SS, P and NO_3 concentrations are higher in autumn than winter. Other studies have reported a similar seasonal effect on sediment and nutrient loss in surface runoff from agricultural land; Heathwaite and Dils (2000) recorded highest mean concentrations (0.16 – 0.19 mg TP l^{-1}) in September and October. This is attributed to autumn storms, due to high P concentrations in the soil in summer months as a result of fertiliser applications, increased grazing activity and escalated microbial activity due to higher temperatures and soil re-wetting. Pollutant concentrations often decrease during the winter months due to source exhaustion.

The export of diffuse pollution from the Lady's Well sub-catchment (and most likely in the wider Belford catchment) takes two forms; 'chronic' export, which takes place during residual flow conditions where drainflow and shallow subsurface flow are the dominant flow pathways; and 'acute' export, which occurs in larger storms and where overland flow is the major conduit. The significant increase in all determinand concentrations between S1 and S2 strongly suggests that the field drains are responsible for the transfer of a significant proportion of polluted runoff. This is because the ditch (at S2) is principally drain-fed and there was no known overland flow in the catchment during the majority of sample collections (28/06/2012 and 17/10/2012 being the exceptions). Data collected during storm events at RAF 2 show how concentrations of SS and P increase in correlation with stage (assumed to be positively related to discharge) therefore at times of elevated flow exponentially greater loads of SS and P are being exported from the catchment. The same effect was reported by Deasy *et al.* (2009) in the Jubilee catchment (with slowly permeable soils) and also by Deasy *et al.* (2008) in the clay soil Hampshire Avon. Large surface runoff events are relatively infrequent but are likely to account for the majority of soil erosion and related sediment (and associated nutrient) losses from the catchment.

Due to the level of sediment/nutrient monitoring and lack of discharge measurements in this particular study it is not possible to estimate percentages losses according to different flow pathways. However, Palmer (2012) predicted that on average 92% of the annual runoff in the Lady's Well catchment was delivered by drainflow. Research has shown how subsurface pathways, including field drains, can play an important role in sediment/nutrient export. Chapman *et al.* (2001) showed that large quantities of sediment were being lost via subsurface flow from agricultural land during storm events and that the source of this sediment is usually within the top 35 cm of the soil profile. They suggest that as a consequence PP losses can also be substantial. The proposed mechanism for this was macropore flow, which can be especially efficient at the end of summer dry periods, before the soil becomes re-wetted. This could account for the elevated P concentrations recorded in the Lady's Well grab sample campaign. Ulén *et al.* (2007) reported that subsurface pathways could contribute 12-60% P losses from agricultural fields and Deasy *et al.* (2009) argued that in the Jubilee catchment, subsurface runoff pathways had higher discharges on average and flow for a greater proportion of the time than surface runoff pathways. Hence they have the capacity to transfer larger sediment/nutrient loads (for example, on an annual basis); event durations for drainflow are also longer than for surface runoff.

Mitigation potential

It was clear from visual observations that sediment was accumulating in RAF 2. Plate 6.5 and Plate 6.6 provide an example of an online pond RAF (also in the Belford catchment on the main channel, but not measured as part of this study) that became silted up over a period of four years and nine months. Plate 6.7 shows the same RAF after it was cleaned out by the farmer (as the photographs were taken from different angles, the arrows indicate the direction of water flow though the pond – from inlet to outlet). Plate 6.6 shows how the bulk of the sedimentation occurred at the inlet of the feature and the same was seen to happening in RAF 2 (albeit to a lesser extent as it was not on the main stream channel). Johannesson *et al.* (2011) also found that the majority of sediment was deposited near the inlet of a monitored wetland, and that the P content of this sediment corresponded to almost 80% of the P load. However, despite this evidence, the synchronous samples collected at the inlet and outlet of RAF 2 during storm events indicate otherwise. Higher SS and TP concentrations were recorded at the outlet in comparison with the corresponding inlet sample during the early high flow component and peak of the event. Thus it is apparent that the feature (and probably other online pond RAFs) may be functioning to reduce chronic losses of SS (and sediment-phase nutrients), but are largly ineffective in storm events. This is likely attributed to the reduction in residence time during high flows. It is also strongly suspected that remobilisation of previously deposited material is the main reason for failure.

To improve pollutant retention without having to increase the overall size of the RAF (as volume is positively correlated with residence time) the features could be modified by adding baffles, or introducing vegetation (as reported by Braskerud (2001)). These would help prevent flow from 'shortcircuting' the system and would reduce remobilisation. Also, in order to maximise the lifespan and water storage capacity of the online pond RAFs it would be favourable to construct upstream sediment traps to attenute the sedimentation rate in the main ponds.

Plate 6.5: Online pond RAF after construction (September 2008).

Plate 6.6: Online pond RAF filled with sediment (June 2013).

Plate 6.7: Online pond RAF after sediment removed by farmer (June 2013).

Visual observations suggetsed that RAF 1 was functioning to capture sediment duirng storm events. Palmer (2012) was able to quantify this on a single occasion and estimated that approximately 1 tonne of sediment was retained. Despite this positive finding, it is important to note that this type of feature will only function in events that generate surface runoff, although this is arguably when the largest pollutant loads are exported (e.g., Haygarth *et al.* (2005)). However, in this particular location it is apparent from the grab sampling campaign that high concentrations of pollutants are lost via the subsurface field drain (Ulén *et al.*, 2007; Deasy *et al.*, 2009) and therefore by-pass the feature. In a situation such as this where field drains have been identified as an important transfer route for sediment and sediment-phase nutrients, the number of mitigation options available are limited.

Perhaps the most effective would be the reversion of land use from arable back to grass, but this is unfavourable to the farmer. Another option, which would allow for the continuation of arable land use, would be to 'treat' the runoff as it leaves the drains and enters the ditch network as has been attempted with RAF 2.

6.4 The need for a bespoke ditch-based RAF

6.4.1 Multi-stage treatment RAF

A new RAF (see Figure 6.4*c* for location) was constructed in February 2011 in the 150 m length of ditch directly upstream of RAF 2 and approximately 500 m down the catchment from RAF 1. Prior to construction the ditch was cleaned out by the farmer, effectively removing all accumulated sediment. The design represents the culmination of experience gained from work described above and has the aim of mitigating polluted tile drain flow by achieving the following objectives:

- reduce sediment/nutrient losses during residual flow and small storm events (with no surface runoff).
- target specific locations for sedimentation and reduce the remobilisation of previously settled material (and associated nutrients) during storm events.

To meet the above criteria the design was a multi-stage treatment RAF, which comprised an upstream sediment trap, followed by a filtering system consisting of leaky willow barriers and brash screens, and a final wood chip barrier/filter (Figure 6.10).

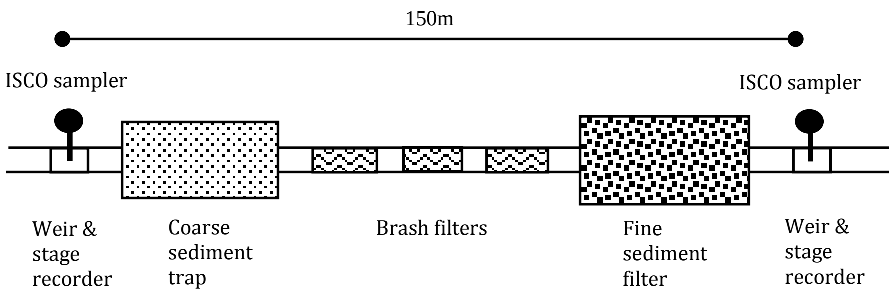


Figure 6.10: Schematic of the multi-stage RAF constructed in Lady's Well (not to scale).

RAF design and construction

Initial findings in Belford suggested that it was important to create more sediment traps, especially upstream of online ponds to help reduce their sedimentation (with coarse sediment - >125 µm), thus prolonging maximum flood storage capacity. Sediment traps also help to determine where in the system material is stored. It is important the design is simple but appropriate to allow quick construction and straightforward maintenance (i.e., sediment removal).

Sediment trap design

The flow rate in the ditch was estimated based on the pipe size of the subsurface field drain which feeds the surface ditch at S2 using:

$$\boldsymbol{Flow\ rate = \frac{1}{4}\,\pi\,(pipe\ diameter)^2\ velocity} \quad \textbf{Equation 6.1}$$

giving a value of 17.7 l s^{-1} using an assumed velocity of 1 m s^{-1} (this value is in close agreement with the 16 l s^{-1} flow capacity calculated by Palmer (2012) for the Lady's Well subsurface field drain).

Sediment traps rely predominantly on particle settlement to remove sediment from runoff. One of the key factors that influence the ability of a trap to retain material is the residence time. Sediment trap residence time is given by:

$$\boldsymbol{T = V/q} \quad \textbf{Equation 6.2}$$

where T is residence time, V is volume of system, q is flow into system. In order to improve the design of effective sediment traps (i.e. to ensure that sufficient residence time is provided) particle settling times for a range of particle sizes (Table 6.6) have been estimated based on Stoke's Law for settling velocity (Equation 6.3).

$$v_s = \frac{2}{9}\frac{(\rho_\rho - \rho_f)}{\mu} g\, R^2$$ **Equation 6.3**

where v_s is the particle's settling velocity (m s^{-1}) (vertically downwards if $\rho_p > \rho_f$, upwards if $\rho_p < \rho_f$), g is the gravitational acceleration (m s^{-2}), ρ_p is the mass density of the particles (kg m^{-3}), and ρ_f is the mass density of the fluid (kg m^{-3}). The following assumptions were made: water temperature = 10°C; particle density = 2800 kg m^{-3} (for silt/clay); fluid density = 1000 kg m^{-3}. It should be acknowledged that Stoke's Law assumes steady state flow, which may not always be the case in RAFs.

Table 6.6: Particle size classification

Aggregate class	Size range
Coarse sand	0.5–1 mm
Medium sand	0.25–0.5 mm
Fine sand	125–250 µm
Very fine sand	62.5–125 µm
Silt	3.9–62.5 µm
Clay	< 3.9 µm

According to Figure 6.11, in order to settle a particle of 3.9 µm (clay) a distance of 0.5 m (a value taken to represent the depth below the outflow pipe assumed sufficient for settlement from the water column) it would take circa 9 hours 20 minutes; a particle of 30 µm (median silt sized) circa 9 minutes 30 seconds; and a 63 µm particle (fine sand) circa 2 minutes 30 seconds. Thus, with a target particle size for settlement of 30 µm (it is not feasible to target clay) and a flow rate of 17.7 l s^{-1} a sediment trap with 10.1 m^3 storage capacity is required.

The ditch was both widened and deepened and a rock and earth bund constructed to dam the water (Plate 6.8); the feature has a total storage capacity of circa 10 m^3 (6.25*2 .0*0.8 – (m) l*w*d). A 150 mm diameter riser pipe (Plate 6.9) was installed to drain the feature from the surface in order to help minimise the remobilisation of previously deposited material. The riser pipe orifice was situated at approximately 60% of the total depth to allow extra water storage capacity during high flows (circa 4 m^3). Concrete slabs (600 mm^2) were used to partially line the bottom of the trap in order to provide a solid bottom to aid sediment recovery by mechanical digger and also enable the measurement of sediment accumulation depth.

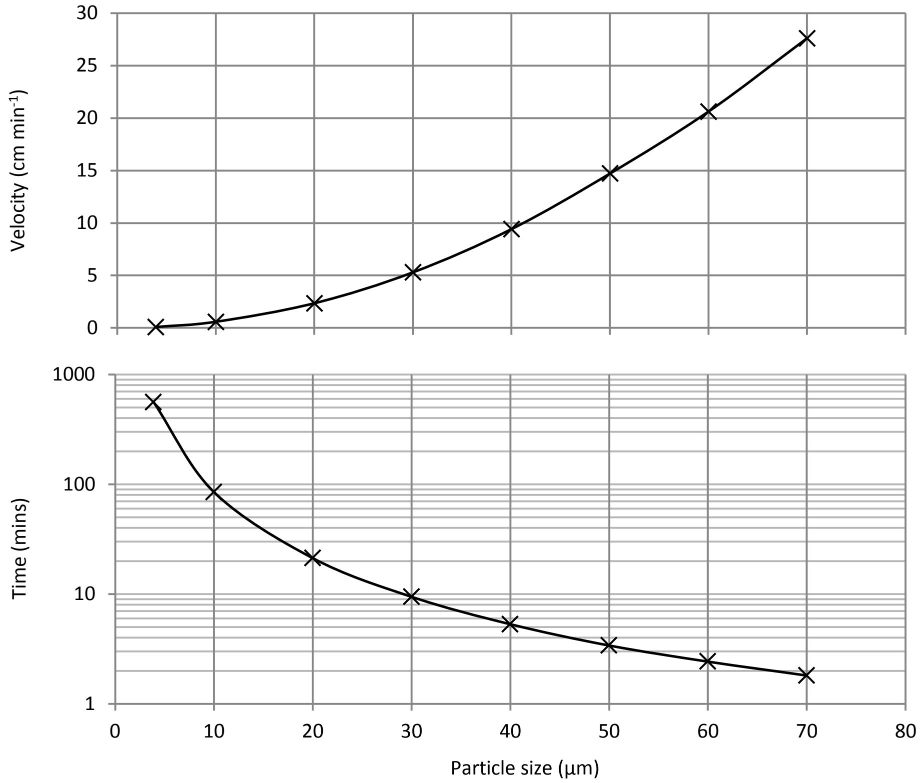


Figure 6.11: Steady state *a*) particle settling velocity, and *b*) time for particle to settle to 0.5 m, based on Stoke's Law (Equation 6.3).

Plate 6.8: Sediment trap after construction.

Plate 6.9: Rock and earth dam with outlet pipe.

Three woven willow check dams (Plate 6.10) were installed in the channel downstream of the sediment trap with brash screens placed upstream and pinned into place to prevent them from being washed away. After only a few months the willow canes had taken root and had sprouted leaves (Plate 6.11); a living dam should have a much longer lifespan compared with one constructed from dead timber, which would decay. The rationale of this feature was to reduce the velocity of the water in a particularly steep section of ditch and to have a partial damming effect on the flow, causing it to back up and promote sedimentation.

Also once pockets of larger sediment particles are trapped in the brash it is believed that these may have a flocculation effect on the fine clay material.

Plate 6.10: Woven willow dam after construction (taken 23/03/2011).

Plate 6.11: Living willow dam circa two months after construction (taken 02/06/2011).

The fine filter was designed to remove fine sediment (particles <125 μm) and associated nutrients. To achieve a high level of filtration whilst ensuring that water can pass through relatively easily, wood chippings were used as a cost-efficient filter media. The channel was deepened and widened to accommodate around 6 m^3 of chippings, which were held in place by a timber pen lined with wire mesh and supported by wooden stakes (Plate 6.12). The use of a wood chip bioreactor is a method for removing NO_3 from drainage water by denitrification (in which NO_3 is converted to nitrous oxide and nitrogen gas). Bioreactors have been studied and have been shown to effectively reduce NO_3 concentrations in agricultural runoff through denitrification (e.g., Saliling *et al.* (2007); Greenan *et al.* (2009); Woli *et al.* (2010) – see Chapter 2.9.2). The filter medium will require periodic renewal as it degrades, and can be spread to land following removal. A spillway was created around the side of the feature to allow the bypass of water during large runoff events, preventing overtopping and damage.

Cost

The overall construction cost of the multi-stage RAF was circa £1000. This included contract digger hire ≈ £250, wood chippings ≈ £150, timber and wire ≈ £100, and the cost of one day's work for two people ≈ £500.

Plate 6.12: Fine filter feature containing circa 6 m^3 of wood chippings.

Instrumentation

The feature was originally instrumented with a pressure transducer and a rectangular flume at the upstream end. To determine the performance of the RAF synchronous water samples were taken during storm events using two auto-samplers located up and downstream of the feature. Samples were analysed for concentrations of SS, P and NO_3 according to the methodology described in Chapter 3.3. The samplers were set to a single float switch located adjacent to the upstream pressure transducer; twenty-four x 1 litre samples were collected hourly from the point of programme initiation. However, due to the flashy nature of the ditch it was difficult to measure the discharge accurately over a range of flows using the rectangular flume. Thus, to allow the calculation of flow-weighted concentrations and pollutant loads, v-notch weirs and accompanying pressure transducers were installed at both the up and downstream monitoring points at a later date (Figure 6.10). Discharge was calculated using the appropriate weir equation (see Appendix B1 for details).

Measurement of sediment accumulation

The sediment trap was divided into three zones (inlet, central and outlet) and the volume of retained sediment calculated by multiplying the surface area with the mean sediment thickness of a single zone. Sediment thickness was determined by inserting a measuring rule through the sediment until it hit the solid concrete slab (slab location was marked by wooden posts); the depth was then read from the rule. Due care was taken when entering the trap so as not to disturb the area of measurement. The total volume of trapped sediment is given by the sum of the three-zone volumes.

To convert sediment volume into mass it is necessary to measure the bulk density of the sample. Bulk density is defined as the ratio of dry sediment mass to bulk sediment volume (including pore spaces) given by:

$$\rho_d = M_s/V \qquad \text{Equation 6.4}$$

where ρ_d is the bulk density, M_s is the mass of the dry sample, and V is the total volume (of the wet sample). The two most common methods used for obtaining a soil sample of known volume are the core method (a cylindrical coring tool of known volume is driven into the soil to a desired depth) and the excavation method (level soil surface and dig a hole to desired depth; line hole with plastic, then fill it with measured volume of water). However, as the sediment trap is partially filled with water the collection of an undisturbed sediment sample is far less straightforward. Samples was dug up from the bottom of the trap using a spade and carefully lifted through the water to keep disturbance to a minimum. Sample were then placed into containers of known volume and mass and sealed before taking them to the laboratory where the wet samples were weighed and then oven dried.

The available P content of the sediment was determined using British Standards Institute method BS 775-3.6:1995 (BSI, 1995). Each sample was air-dried and ground to pass through a 2.0 mm mesh. The P content was extracted using the sodium hydrogen carbonate (NaHCO3) extraction method developed by Olsen et al. (1954) (on a volume basis where a 5 ml of soil was extracted with 50 ml of 0.5 M NaHCO3). The P concentration in the extract was determined by the phospho-molybdate method proposed by Watanabe and Olsen (1965) and Murphy and Riley (1962). The absorbance of the phospho-molybdate complex was measured on a UV-Vis Spectrophotometer after calibration with P standard solutions at a wavelength of 825 nm.

6.4.2 Results

Hydrology

Data recorded by the pressure transducer in the original upstream flume was unreliable, due to low water depths that occurred for a large proportion of the time, so was disregarded. As a surrogate the pond stage from RAF 2 (directly downstream of the multi-stage RAF) is used as it spans the entire monitoring period (Figure 6.12). The stage record shows how 2011 was relatively dry and that only one significant high-discharge event occurred in August. This event was not sampled. The 2011/2012 winter was also unseasonably dry and the first major runoff event was not recorded until April 2012; for the period April 2011-March 2012 total precipitation was 557 mm and for the corresponding 2012-2013 period precipitation was 1169.

Three events were sampled before the V-notch weirs were installed; nine events were sampled in total.

Table 6.7 summarises the hydrological conditions for the monitored events. Figure 6.13 depicts the stage record measured at the upstream V-notch weir. Discharge response is very flashy in relation to precipitation and a maximum discharge of 56 l s^{-1} was recorded. Discharges above 20 l s^{-1} only account for 5% of the flow duration and Q_{50} is 3.02 l s^{-1} (Figure 6.14). Manual measurements of water stage were taken at both V-notch weirs whenever equipment was downloaded to allow validation of the pressure transducer readings and increase accuracy.

Overall there was a very good agreement in recorded stage (and therefore discharge) records in terms of shape, timing and magnitude between the upstream and downstream weirs. When analysing the upstream/downstream discharge responses during monitored storm events (see next section) it is apparent that the downstream hydrograph is less 'spikey', showing signs of a buffered response. Examination of rising and falling limbs indicates a very slight attenuation of flow at the downstream weir on rising limbs and steeper, shorter recessional limbs at the upstream weir (Figure 6.16 – top panel).

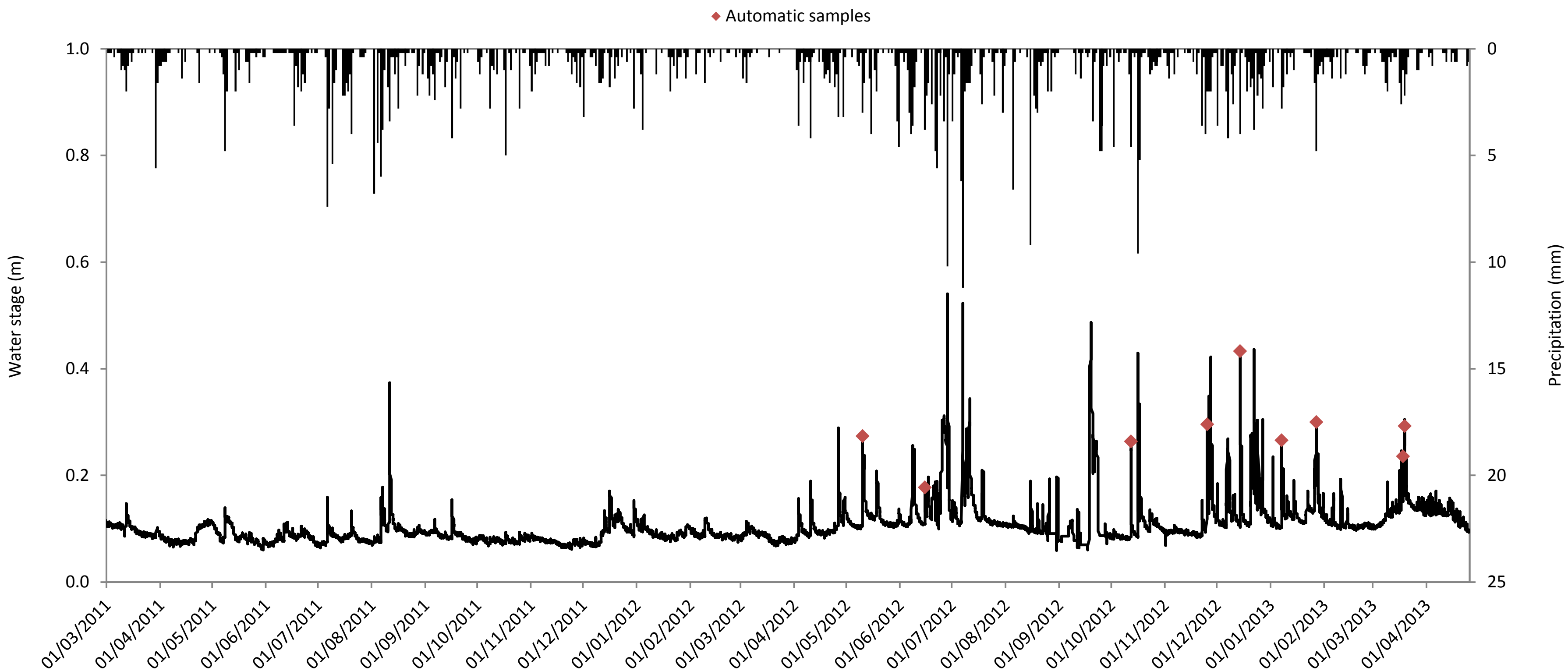


Figure 6.12: Stage recorded in RAF 2. Red markers indicate occasions when the auto-samplers were operational.

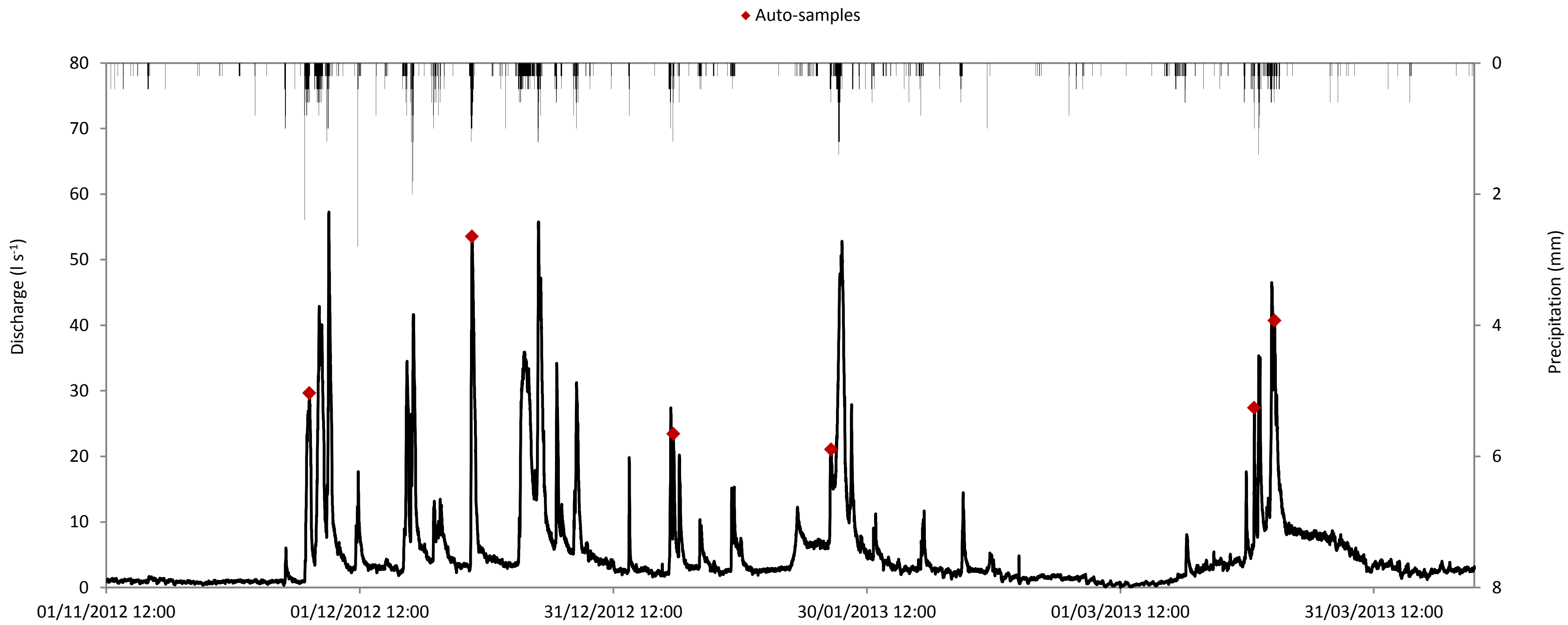


Figure 6.13: Lady's Well stage recorded at the upstream V-notch weir after installation on 01/11/2012. Red markers indicate occasions when the auto-samplers were operational.

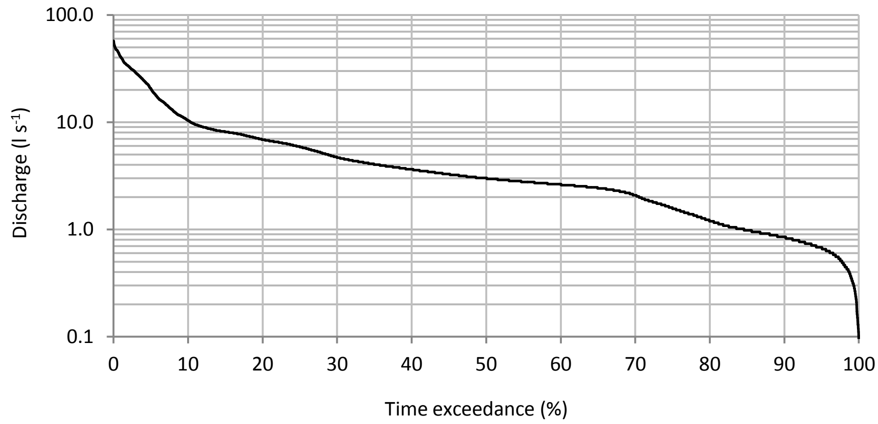


Figure 6.14: Lady's Well flow duration curve based inlet V-notch weir data.

Table 6.7: Lady's Well precipitation, discharge and sampling event summary 2012-2013.

Event	Start time/date	Duration sampled (hrs)	Total precipitation (mm)	Peak hourly precipitation (mm)	Peak discharge (l s^{-1})
1	10/05/2012 00:00	24	33.6	2.8	-
2	17/06/2012 13:30	12	5.4	1.6	-
3	12/10/2012 01:30	14	24	5	-
4	25/11/2012 02:15	18	24.2	4.4	29.7
5	14/12/2012 15:30	14	24	3.6	53.5
6	07/01/2013 04:30	15	16.4	2.8	27.4
7	26/01/2013 21:15	24	31.8	4.8	52.8
8	17/03/2013 08:00	22	14.4	2.8	35.3
9	19/03/2013 05:45	24	18.8	2.2	46.5

Water Quality

Paired samples were collected from the inlet and outlet of the multi-stage RAF during nine storm events (Figure 6.12, Figure 6.13 and Table 6.7). Tables containing raw discharge and concentration data can be found in Appendix H1.

Event 4 (the first captured following the installation of the V-notch weirs) is depicted in Figure 6.15. It is a medium-magnitude discharge event in response to a large storm event following a prolonged period of little precipitation. Suspended sediment concentration peaks early at the RAF inlet relative to peak discharge; this is also when maximum SS reduction occurs (51.8%). There are several points after peak concentration has passed through the RAF that outlet concentration is slightly higher than inlet concentration. On average SS concentration is reduced by 25% over the duration of the event. When combined with discharge, SS load is reduced by a mean of 32%. Peak TP concentration (at the inlet) is recorded three hours after peak SS but also decreases prior to peak discharge. Over the course of the event TP concentration is reduced by a mean of 15.2% (max. 36%) and load by 21%. Similarly to SS, there are some occurrences where TP concentrations are slightly higher at the outlet compared with the inlet, as overall concentrations fall. Nitrate concentrations exhibit no correlation with discharge and increase gradually (at both the inlet and outlet) over the duration of the event. A slight reduction in NO_3 is recorded on the rising limb (max. 19%) with a mean of 8% (concentration) and 9% (load) calculated for the entire event.

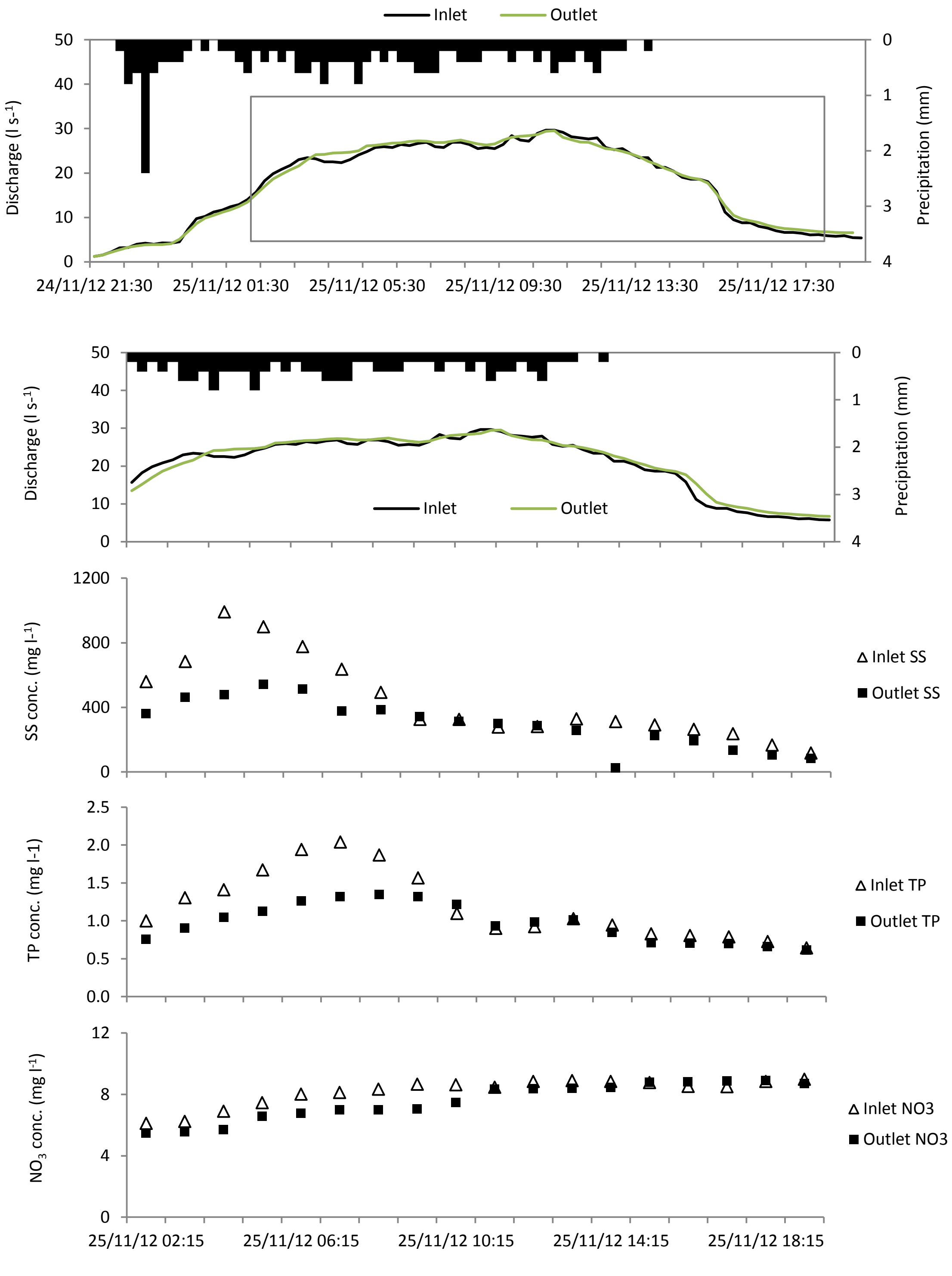


Figure 6.15: Discharge and SS, TP and NO_3 concentration record – Event 4.

Figure 6.16 depicts SS, TP, SRP and NO_3 concentrations recorded at the inlet and outlet during Event 9, a relatively high magnitude, double peaked event in response to moderate but prolonged rainfall. Generally SS and TP are closely related to discharge and peak conjointly. Soluble reactive P concentration also increases with discharge but exhibits a more attenuated and buffered response, peaking circa two hours after peak discharge. Nitrate shows little to no relation to discharge but rather displays a slow and steady increase throughout the first 20 hours of the 24 hour sampling period. Suspended sediment, TP and SRP concentrations recorded at the outlet sample site are generally lower than corresponding samples collected at the inlet. In this instance SS, TP and SRP concentrations are reduced by an average of 26.2%, 13.9% and 15.2% over the 24 hour event, while loads are loads are reduced by 29.5%, 20.2% and 14.7%, respectively. Nitrate reduction is negligible with a 3.9% decrease in concentration and a 5.3% decrease in load recorded.

Time series charts for all monitored events can be found in Appendix H2. Summary tables of RAF performance, including ranges and means of both concentrations and loads, can be found in Appendix H3; observations indicate positive but variable effectiveness.

Some caution is required in the interpretation of sediment trap efficiency at low discharges, when the residence time within the trap means samples taken at the same time are not paired. However, over all the monitored events, maximum reduction in peak SS was 87.9% (concentration) and 65.4% (load); maximum TP decrease was 89.9% (concentration) and 63.4% (load); maximum SRP decrease was 62.0% (concentration) and 58.3% (load); maximum NO_3 decrease was 48.9% (concentration) and 52.1% (load). However, it should be noted that the highest concentration reductions (for all determinands) were recorded during Event 1 and that discharge values are not available for this occasion. The largest reduction in SS and TP occurred during the rising limb and at peak discharge.

Sediment/nutrient losses from the RAF (higher concentration recorded at the outlet compared with the corresponding inlet value) were recorded during some hourly time steps. For SS, TP and SRP this occurred predominantly during the recession. The event pattern for NO_3 showed no discernible retention/net loss occurring during the rising limb. This was often followed by limited retention on the recession and a switchover point following which inlet concentrations decreased whereas outlet concentrations either stayed constant or increased slightly.

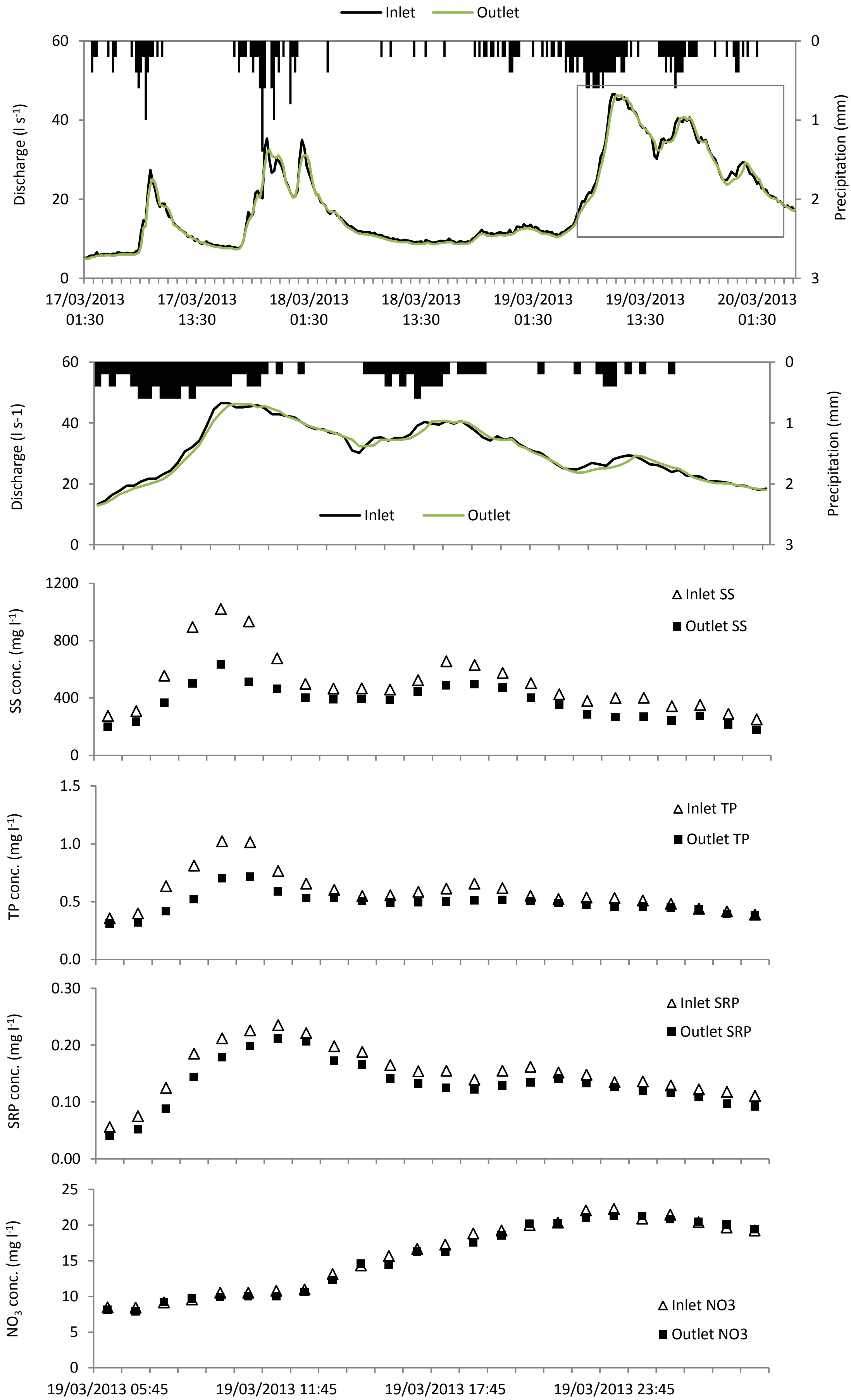


Figure 6.16: Discharge and SS, TP, SRP and NO_3 concentration record – Event 9.

Overall RAF performance is summarised in terms of inlet and outlet determinand concentrations in Table 6.8, mean concentration reduction percentage and significance in Table 6.9, and mean load removal percentage and significance in Table 6.10. For SS and TP better efficiency is found in the reduction of loads and for SRP and NO_3 better efficiency is found in the reduction of concentrations. A mean significant decrease in both concentration and load reduction was found for all determinands.

Table 6.8: Summary statistics of RAF inlet and outlet determinand concentrations.

	SS conc. (mg l^{-1})		TP conc. (mg l^{-1})		SRP conc. (mg l^{-1})		NO_3 conc. (mg l^{-1})	
	Inlet	**Outlet**	**Inlet**	**Outlet**	**Inlet**	**Outlet**	**Inlet**	**Outlet**
Minimum (mg l^{-1})	27.0	11.0	0.055	0.030	0.009	0.007	5.9	5.5
Maximum (mg l^{-1})	1068.0	822.0	2.040	1.346	0.408	0.299	49.5	48.5
Mean (mg l^{-1})	391.6	280.9	0.716	0.564	0.190	0.149	14.8	13.3
Median (mg l^{-1})	329.0	245.0	0.635	0.521	0.184	0.141	9.5	8.8
σ (mg l^{-1})	227.9	154.4	0.350	0.251	0.102	0.081	12.2	10.8
n	165	165	165	165	62	62	143	143

Table 6.9: Mean percentage concentration reduction and significance.

Determinand	n	Mean % reduction	Paired T-Test	
			***T*-value**	***P*-value**
SS	165	25.7	14.67	<0.001
TP	165	19.6	12.50	<0.001
SRP	62	18.9	9.86	<0.001
NO_3	143	9.0	5.90	<0.001

Table 6.10: Summary of pollutant loads, mean percentage reduction and significance.

Determinand	n	Mean load (instantaneous)		Mean % reduction	Paired T-Test	
		Inlet	**Outlet**		***T*-value**	***P*-value**
SS (g)	117	12.7	9.0	30.1	9.35	<0.001
TP (mg)	117	22.62	17.53	23.3	8.65	<0.001
SRP (mg)	38	5.550	4.84	12.4	4.87	<0.001
NO_3 (mg)	95	326.00	304.7	7.6	5.26	<0.001

Concerning SS, the greatest reductions occurred during events with high inlet concentrations (Figure 6.17 *a*) but there was no clear correlation between SS reduction (RAF efficiency) and discharge (Figure 6.17 *b*). Table 6.11 contains Pearson's correlation coefficients for RAF efficiency vs. both inlet concentrations and discharge, for all determinands.

A weak positive correlation occurred between SRP inlet concentration and reduction, and a weak negative correlation between discharge and reduction percentage. No correlation was observed between NO_3 concentration and reduction, or between discharge and reduction percentage.

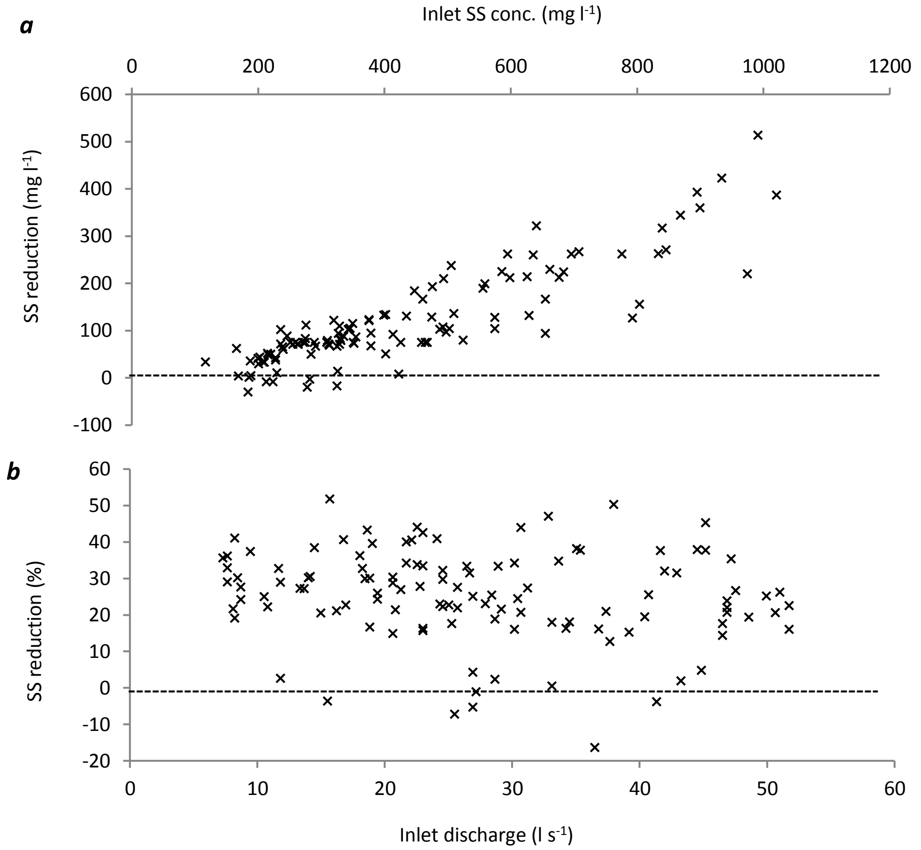


Figure 6.17: The relationship between SS reduction and *a*) inlet concentration (measured at the RAF inlet), and *b*) discharge.

Table 6.11: Pearson's correlation coefficients for RAF efficiency.

Determinand	**Inlet conc. vs. reduction (mg l^{-1})**		**Inlet discharge vs. reduction (%)**	
	Pearson's *R*	***P*-value**	**Pearson's *R***	***P*-value**
SS	0.876	0.000	-0.179	0.054
TP	0.803	0.000	0.075	0.419
SRP	0.339	0.105	-0.411	0.046
NO_3	-0.015	0.884	0.192	0.062

Figure 6.18 shows RAF performance (determinand concentration reduction percentage) over the duration of the monitoring period. The analysis indicates that the feature's ability to reduce concentrations of sediment and nutrients was variable, with the scatter attributed to the different magnitude and type of event, but importantly did not significantly decrease over time.

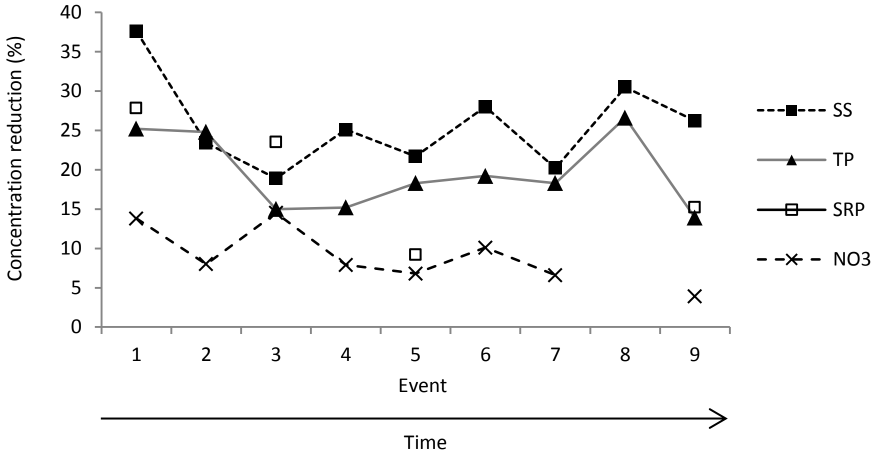


Figure 6.18: RAF performance over time (see Table 6.7 for event dates).

Sediment trap sediment accumulation

In the sediment trap component of the RAF, accumulated sediment depth increased along the length of the feature (from inlet to outlet - Figure 6.19). The total volume of retained sediment was measured as 2.21 m^3, a total dry sediment mass of 2.01 tonnes.

Sediment P concentration was measured as 52.2 mg kg^{-1} at the inlet zone and 32.7 mg kg^{-1} at the outlet zone. This equates to a P trapping rate of 0.004-0.007 kg ha^{-1}.

Sediment trap zone

	Inlet	**Middle**	**Outlet**	**Total**
Area (m^2)	3.45	3.45	3.45	10.35
Sediment depth (m)	0.17	0.22	0.25	
Sediment volume (m^3)	0.59	0.76	0.86	2.21
Bulk density (g cm^{-3})	0.941	0.915	0.888	
Sediment mass (t)	0.56	0.70	0.76	2.01

Direction of flow

Figure 6.19: Sediment trap accumulated sediment.

6.4.3 Discussion

Hydrology

The installation of V-notch weirs at the inlet and outlet of the RAF made the measurement of discharge significantly more accurate and allowed the subsequent calculation of sediment and nutrient loads. Comparison of the inlet and outlet discharge records during storm events indicates that a small amount of flow attenuation was occurring at the downstream monitoring location, this was evident on the rising limb and at peak discharge. However, as the pressure transducers were logging at 15 minute intervals it is difficult to quantify the delay as the small size of the ditch and short length of reach mean that any delay in time to peak is going to be a maximum of 15 minutes or likely less. Logging discharge on a shorter time interval would provide more insight into this matter but the evidence suggests that the ditch interventions were acting to back up runoff and attenuate peak flows, despite the small size of the features. V-notch weirs were selected for their ability to accurately measure low flows as well as high flows. However, they have some disadvantages for this type of application that are important to divulge. It may not be possible to install a V-notch weir large enough to convey the largest discharges if the ditch is small and/or shallow. The V-notch weir also creates an upstream backwater zone, which can act as a mini sediment trap. Rectangular weirs may be more appropriate; the measurement of low flows will be less accurate compared with V-notches, but the quantification of high-flows is arguably more important.

Sediment and Nutrient retention

The multi-stage ditch RAF consistently reduced concentrations and loads of sediment and nutrients at the event scale. Overall, SS, TP, SRP and NO_3 concentrations were reduced by 26%, 20%, 19% and 9%, respectively, while respective loads were reduced by 30%, 23%, 12% and 8%. The greater reduction in SS and TP loads, compared with concentrations, is attributed to the positive correlation between SS (and associated P) and discharge, meaning higher concentrations at higher discharges and subsequently greater load reduction. The opposite effect occurs for SRP and especially NO_3 as an increase in discharge exhibits either an attenuated response in concentration or even a reduction.

The largest overall SS and TP reductions occurred during events with high inlet concentrations (higher loadings), also reported by Kadlec and Knight (1996) and Mitsch and Gosselink (2007). RAF performance appears not to be severely negatively affected by high discharges but there is an indication that SS and SRP removal efficiency is slightly lower at the highest of flows. The majority of SS and TP removal occurs during the rising limb of storm events and at peak discharge (e.g., Figure 6.16 and Appendix H2), which explains why load retention percentages are higher for these two determinands. The increase in PP retention at higher discharges is likely due to the input of coarser PP while an increase in the SRP:TP ratio would see a decrease in TP retention as less PP would settle (e.g. Braskerud *et al.* (2005)). The reduction in SRP losses is perhaps more difficult to explain, although Braskerud *et al.* (2005) reported how SRP retention was higher in young wetlands than older ones, and attributed this to a high iron content in the water, which facilitated P retention by forming P-Fe complexes. This chemical 'stabilisation' process then becomes exhausted over time Kadlec (2005). The results in this study are promising as one of the main design criterions was to reduce losses during high flows – partly by preventing the remobilisation of previously deposited fine sediment (this was in response to the event data recorded at RAF 2, reported in Section 6.3.3m - Figure 6.9).

Fine- and coarse-filter features

The inlet and outlet sampling regime using automatic water samplers has helped to evaluate the multi-stage RAF as one feature. As the RAF consists of three separate components, which target different sediment/nutrient removal processes, it would be desirable to monitor each element individually. Thus it is difficult to identify which parts of the system were functioning to reduce DWPA and possibly which were not.

However, visual evidence suggests that all three components were having some impact on sediment removal (clearly it is not possible to see trapped nutrients). The woven willow barriers/brash '*coarse*' filters were observed during several high-flow events and were seen to act as 'leaky dams', which created a back water zone. Overtime the brash filters became increasing bound up with more brash and there was evidence of coarse sediment deposition. The willow barriers have continued to grow and will require cutting back in the future but the fact that they are living means that they will help provide stability to the steep ditch banks and may help reduce in-channel erosion. The installation of woven willow barriers as leaky dams is a low-cost approach to reducing flow velocities in ditch systems and can provide localized sedimentation zones. A member of the Northumberland Rivers Trust commented that such features could be very effective and importantly could be easily taken up by farmers (G. Dodds, pers. comm.).

The wood chip fine-filter allowed the passage of low and residual flows through the filter media but higher flows were forced to back-up in the ditch, and during large runoff events water bypassed the feature via the spillway channel. The physical filtering of ditch runoff during small-medium sized storms was almost certainly having a positive impact on the SS and attached P concentrations. Periodically the wood chips were dug up to inspect their condition and it was clear that significant amounts of fine sediment were being incorporated within the chippings. However, the feature was not assessed for its biological and/or chemical impact on nutrient cycling. Research suggests that denitrification could be occurring during low flow conditions but it is highly unlikely that this would happen in storm events due to the low contact time. However, some NO_3 retention was recorded on the rising limbs of some storm events but this may due to subsurface pathways contributing NO_3 to different parts of the ditch at different times as the soil wets up. It is also argued that NO_3 export is of less importance during high flows due to dilution. There is an issue of maintenance related to this feature, specifically that the chippings would require periodic renewal. The regularity of this would depend largely on the sediment/nutrient loading of the filter and the frequency of storm events.

To attain the level of denitrification needed to significantly reduce NO_3 concentrations the wood chip filter would have to occupy a significant length of the ditch. This approach (described in Chapter 2.9.2) is best suited to controlled discharges as it relies on the slow passage of runoff through the system, thus it is not best suited to catchments with flashy flow regimes capable of yielding high discharges. However, as NO_3 export is not strongly correlated with discharge then the targeting of low and residual flows is justified. If it was decided that the NO_3 export from the Lady's Well catchment was of ecological concern and required action then a wood chipping bioreactor of appropriate size could be installed in the circa 150 m ditch upstream of the multi-stage RAF (directly downstream of the field drain outlet). Higher discharges that would otherwise overwhelm the filter could be bypassed around the feature using a relief channel. Burt and Pinay (2005) described how land drainage results in drier soils, enhanced nitrogen turnover and reduced denitrification; also that field drains transfer leached NO_3 rapidly to the surface water network, reducing the potential for riparian zone denitrification. This argument would also provide justification for the use of an in-ditch bioreactor using wood chippings. However, considering the relatively high cost of installation and potential high level of maintenance (thus incurring more costs) of the wood chip filter feature is not recommended for situations such as this where relatively high, flashy discharges are common.

Sediment trap feature

One component of the multi-stage RAF that can be part-evaluated individually is the sediment trap. The feature functions to target sedimentation to a defined location as opposed to it occurring along the entire length of the ditch. This should make maintenance quicker and thus more cost-effective, although it would require more regular attention. In order to settle out fine sediment the trap has to provide sufficient residence time as fine particles have a slower settling velocity. The settling of silt and clay is important as these fractions are associated with the degradation of river beds (e.g., Kemp *et al.* (2011)) and nutrient losses. Phosphorus concentrations of the trapped sediment were greater at the inlet zone compared with the outlet zone, which is consistent with other research findings (e.g., Johannesson *et al.* (2011)). It is probable that a large proportion of the fine particle load will move as part of larger aggregates (e.g., Braskerud (2003)), meaning that the sedimentation of finer particles may be more likely than primary particle diameter would otherwise suggest.

It is estimated that circa 2 tonnes of sediment were trapped in the feature between its construction at the end of February 2011 and the end of monitoring in April 2013 – 25 months later. Although it should be remembered that 2011 and the 2011-2012 winter was very dry in comparison with the same periods one year later. If the assumption is made that the majority of the sediment captured in the Lady's Well sediment trap ocured in the second year it is reasonable to estimate a trapping rate of 0.1 t ha^{-1} yr^{-1}, based on a total of 1.5 tonnes. This value falls well within the ranges described by Ockenden (2012) for sites with a similar soil type (Palmer (2012) reported that the sediment retained in RAF 1 was predominantly clay and silt).

Although in this particular study SS reductions were not measured for the sediment trap on its own, the 26% (concentration) and 30% (load) averages recorded for the entire multi-stage RAF put it in the same range as other studies. Ockenden *et al.* (2012) reported SS concentration reductions of circa 20% for a clay site and up to 60% for a silty loam site; the latter was a large feature that represented 0.1% of the contributing catchment area. These reductions are within the same range as the clay particle retention reported by Braskerud (2003) for two small constructed wetlands on arable land in Norway (57% and 22%). Considering that the sediment trap component of the RAF is relatively small in comparison to those described by Ockenden *et al.* (2012) and Braskerud (2002), this suggests that the feature as a whole is fairly affective at reducing losses of sediment and nutrients. Despite the problem that sediment traps don't remove 100% of the sediment from runoff, storing sediment even temporarily increases the possibility of nutrient uptake by vegetation, thus lowering overall nutrient losses to the wider riverine environment.

Sediment P concentration was measured as 52.2 mg kg^{-1} at the inlet zone and 32.7 mg kg^{-1} at the outlet zone. By applying the same assumptions as described above this equates to a P trapping rate of 0.004 – 0.007 kg ha^{-1} yr^{-1}. Phosphorus retention rates ranging from 0.006 – 1 kg ha^{-1} yr^{-1} across ten sites were recorded in the first year of monitoring in the MOPS project (Ockenden *et al.*, 2012), with P trapping rates also found to be highest at the sandy soil site.

Analysis suggested (Figure 6.18) that RAF performance did not decrease over the course of the monitoring period; Braskerud *et al.* (2005) also reported that increasing wetland age did not negatively influence particulate P retention. However, sediment depth measurement showed that the sediment trap had been reduced in capacity by circa 20% over the two year study period, which would have a knock-on effect on the residence time afforded by the feature, reducing its effectiveness in the longer term and increasing the risk of remobilisation. Remobilisation of trapped sediment should be prevented by design and regular removal. Some of the design principles used in constructed wetlands could be considered to improve sediment and nutrient retention potential.

Sediment trap design

Despite providing a 30% reduction in sediment loss, the sediment trap was designed according to an estimated inflow of 17.7 l s^{-1}, which was of sufficient size to provide the residence time to allow the settlement of particles >30 µm. However, discharges up to 56 l s^{-1} were recorded at the inlet V-notch weir that would provide circa 3 minutes of residence time, compared with the estimated 9 minutes 30 seconds. Analysis shows that 10.4 l s^{-1} is exceeded for 10% of the flow duration (assumed to be when the majority of SS and P is transferred), meaning that the sediment trap would be providing a residence time of at least 16 minutes for 90% of the flow period in autumn and winter.

Thus, according to the peak discharge measured in this study, if a new sediment trap was to be installed in Lady's Well, or the existing one dug out then increased in capacity, to target the removal of medium silt particles (and coarser – requiring circa 9 minutes 30 seconds residence time) a trap of 32m^3 would be required to function as designed during peak flow conditions.

(570 seconds*56 l s^{-1} = 31,920/1000 = 32 m^3)

This size of feature is not unfeasible in this environment, or alternatively the trap could be divided into a number of separate smaller cells if it was more appropriate. Shallow cells have shorter settling distances, meaning they may be more effective than deeper cells for trapping sediment (Reinhardt *et al.*, 2005) but will require more regular maintenance. Braskerud (2001) suggests the use of vegetation and obstructions such as timber baffles to slow the velocity of the runoff and stop short-circuiting of the flow. These could promote particle settlement and resist remobilisation (Uusitalo *et al.*, 2003). Based on the experiences gained in this study it is suggested that sediment traps should be made as large as possible and woven willow leaky dams could be constructed in the traps instead of as separate features.

6.5 Summary

Concentrations of SS, P and NO_3 recorded in the Lady's Well sub-catchment are potentially of ecological significance in relation to WFD thresholds. Losses of all determinands were found to increase during storm events. The main source of flow for the majority of the flow duration was subsurface drainflow, particularly during smaller rainfall events and the rising limbs and recessions of larger events. Therefore, the presence of artificial drainage could be increasing the duration of exposure to ecologically significant diffuse pollutant concentrations. Large storm events that generate overland flow are responsible for significantly elevated SS and P exports but operate for much shorter time periods. However, as the study catchment eventually discharges into a sensitive, low-energy environment receptor where large volumes of fine sediment and associated nutrients may be retained and have a long term negative ecological impact, having a mitigation plan which also targets surface runoff pathways (i.e., larger runoff events) appears appropriate.

Two RAFs installed in the Lady's Well catchment for flood attenuation purposes were evaluated for their ability to reduce losses of sediment and nutrients. An edge of field retention bund RAF, which intercepts a concentrated surface flow pathway, demonstrated the capacity to retain significant amounts of SS (and attached nutrients) but only functions during large storm events that generate overland flow, thus addressing acute export. An online pond RAF appeared to retain sediment during residual and low-flow conditions (adressing chronic export) but displayed little-to-no sediment/nutrient retention capibility during high-flows. The remobilisation of previously deposited fine material is most likely the principal reason that outlet concentrations were often greater than inflow concentrations recorded on the rising limb and at peak discharge.

A multi-stage RAF was constructed at a cost of circa £1000 in a 50 m length of ditch, with design principles informed by the previous RAF studies in the catchment and in light of the identified DWPA export regime. The feature, made up of a sediment trap, a series of coarse filters (woven willow leaky dams), and fine filter system (using wood chippings), consistently reduced concentrations and loads of sediment (up to 30%) and nutrients (up to 21%) at both the small/medium and large event scale.

While wood chipping filters cannot be recommended due to relatively high costs and potential high maintenance regime, woven willow dams, as well as being ecologically and aesthetically pleasing, could be added to sediment traps as a component to improve trapping efficiency. Sediment traps work effectively if appropriately designed and offer a simple, cost-effective mitigation option to farmers, particularly if the identified sediment loss pathway is the surface ditch network (and/or the subsurface drainage network that feeds it). Ditches have to be managed by farmers to maintain levels of land drainage. By adopting them to an optimum design there is real potential to reduce ditch management costs by concentrating sedimentation to localised ‘zones’, at the same time reducing losses of sediment and nutrients to the wider aquatic environment.

7. DWPA mitigation case study two: The Netherton catchment

7.1 Introduction

This chapter reports the outcomes of a case study implementation of RAFs in the Netherton Burn catchment, Northumberland. The work was commissioned by Cheviot Futures, a non-governmental organisation (NGO) working with land managers to help rural communities located around the Cheviot Hills to improve their resilience to future climate change. This includes providing practical solutions to flooding, droughts and water quality issues. Sediment trap RAFs were developed as part of this study and form the second DWPA mitigation case study. The design, construction and efficacy of the RAFs at reducing SS, P and NO_3 losses in agricultural runoff are described in the following chapter.

7.2 Materials and methods

7.2.1 Catchment description

The Netherton Burn catchment (10 km^2) is located upstream of the village of Netherton (OS Grid Reference NT-081980) in Northumberland, northeast England (Figure 7.1). The Netherton Burn, a tributary of Wreigh Burn, rises on the Cheviot Hills and flows into the River Coquet upstream of the town of Rothbury. Catchment elevations range between 544 and 147 m AOD (Figure 7.1). Biddlestone Burn rises on the fells to the north of the catchment and flows in a southerly direction down a steep incised valley where it meets another headwater stream (at Biddlestone) to form Netherton Burn.

The majority of the catchment is formed in lowland terrain (<300 m AOD). The upland northern slopes are formed of early Devonian Cheviot Andesite of the Cheviot Volcanic Formation, while Carboniferous sandstone, siltstone and limestone of the Ballagan Formation underlie the lowlands. Devensian till dominates lowland superficial geology, excluding a line of alluvium along the river corridor and associated river terrace deposits. Soils in the valley bottom are recorded as typical stagnogley seasonally waterlogged soils developed in loamy till which have slowly permeable clay-enriched subsoils (Payton and Palmer, 1990). This has resulted in artificial under-drainage of the catchment using tile drains. The lower slopes are recorded as freely draining loamy, typical brown podzolic soils and the higher elevations as soils with acid-peaty topsoils with little or no water storage capacity during the wet season (Payton and Palmer, 1990).

Land use is predominantly agricultural, with sheep grazing in the uplands and a mixture of pasture and arable in the lowlands. Arable rotations are found chiefly, but not exclusively, to the south of Netherton Burn where the land is of lower relief. There are relatively small areas of mixed deciduous and coniferous woodland throughout the catchment. Six different landowners own the farmland in the catchment but this particular study is focused on land owned by Elilaw Farm, to the eastern-side of the catchment (outlined in red in Figure 7.1). There is a quarry at Biddlestone where a red mica-porphyrite, called 'Biddlestone Red' is extracted; it is particularly suitable for specialised use in road surfaces.

The SAAR value for the Coquet at Bygate (59 km^2 – circa 15 km west of Netherton) is 1020 mm and the value for Usway Burn at Shillmoor (21 km^2 – circa 10 km west of Netherton is 1056 mm (source: Environment Agency, Hiflows, 2013). Therefore the SAAR for the Netherton catchment is assumed to be circa 1000 mm as maximum catchment elevation is slightly lower.

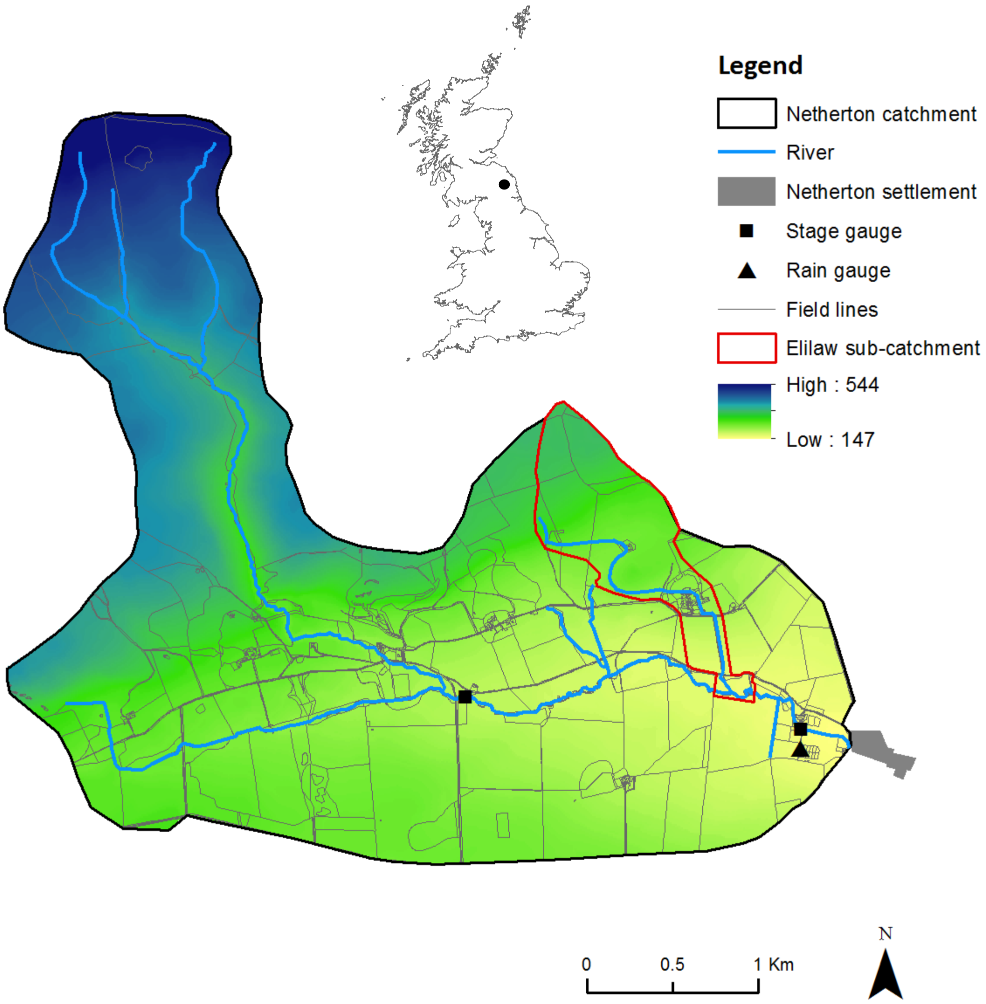


Figure 7.1: Map of Netherton Burn catchment and DEM.

Hydrometeorological data

A rain gauge and three stage gauges are installed in the Netherton catchment (Figure 7.1). At the downstream location (circa 250 m upstream of the village) two pressure transducers are installed side-by-side in the stream, one acting as a back-up. The main gauge is a vented pressure transducer while the back-up gauge is not, thus requires a corresponding barometer to correct for atmospheric pressure. All instruments are logging at 15 minute intervals.

7.2.2 Flood and DWPA mitigation scheme

Netherton village is prone to flooding (Plate 7.1 and Plate 7.2) and a number of phases of work using the local floodplain to implement Natural Flood Management (NFM) were carried out under the Netherton Project. Led by Cheviot Futures [http://www.cheviotfutures.co.uk/], in close collaboration with the Proactive group from Newcastle University, the project relied on a successful partnership and stakeholder engagement in order to ensure uptake and delivery. As well as addressing the problem of flooding in Netherton, the project also aimed to tackle water quality issues; thus becoming a multiple-benefit scheme. This chapter describes Phase One of the project, specifically the design and implementation of the DWPA mitigation RAFs.

Plate 7.1: Netherton Burn in flood (September 2008).

Plate 7.2: Netherton village flooded (September 2008).

7.2.3 Mitigation site description

The 80 ha Elilaw sub-catchment (Figure 7.2 - see Figure 7.1 for wider location) is of relatively steep relief, falling from 326 to 158 m AOD over a distance of 2 km. The surface drainage regime has been heavily modified in the past by a mill race constructed to supply water to Netherton Mill. The mill race was designed to collect runoff from the hill to the north of Elilaw Farm, and a channel constructed along the contour of the hill directs water into a pond located at the Farm. Water is piped under the settlement and then flows in a surface ditch down the hill to the mitigation site. The farm pond and piped drain both act to regulate the flow of the watercourse (P. Stott, landowner, pers. comm.). Over time the mill race channel has degraded so that the majority of the flow enters into the Netherton Burn instead of taking its intended course towards the mill (Figure 7.3). Besides the mill race, a number of subsurface drains contribute water to the site, most of which enter the old mill race before spilling into the Burn (Figure 7.3). Prior to RAF construction the mitigation site was used for rough grazing.

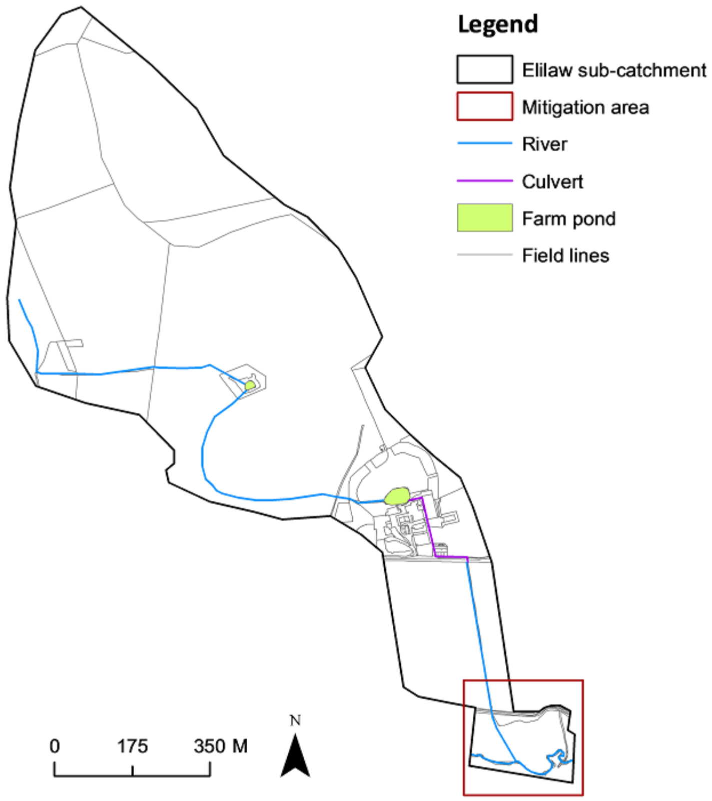


Figure 7.2: Map of Elilaw sub-catchment; red square indicates mitigation area (Figure 7.4).

Figure 7.4 depicts the mitigation site post intervention. The channel from Elilaw Farm has been re-directed to a large floodwater storage RAF via a three-tier sediment trap feature (contributing catchment area of circa 70 ha). To the north of the site a single sediment trap was constructed through which water from the mill race was re-directed before being channelled to the main storage feature (contributing catchment area of circa 10 ha). Details of the individual RAFs are given below.

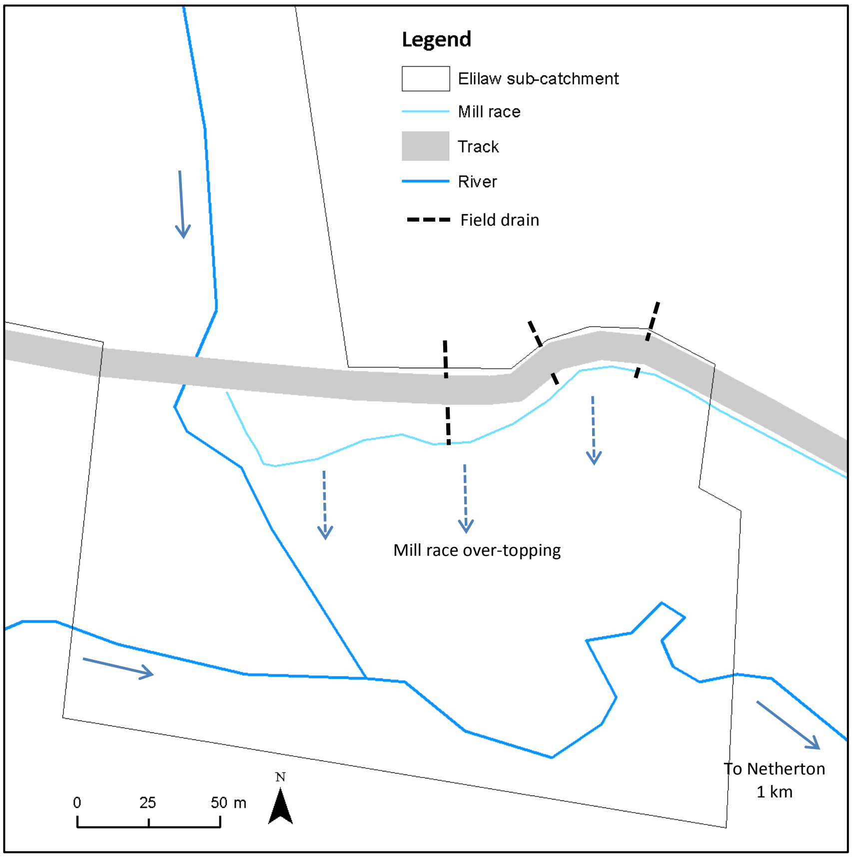


Figure 7.3: Elilaw mitigation site – before intervention.

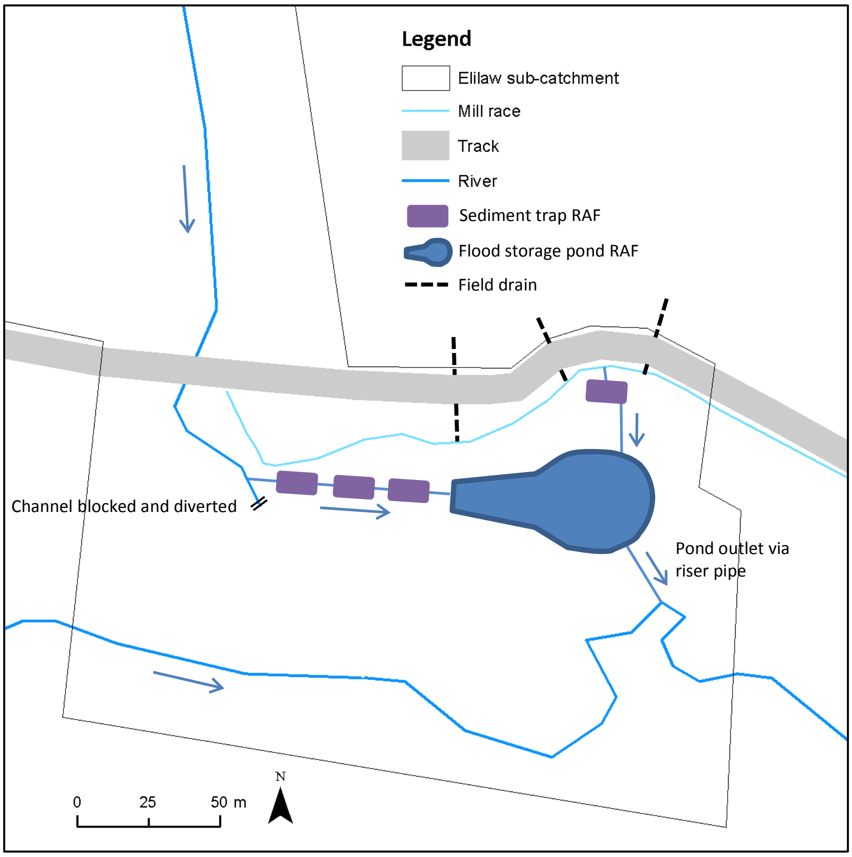


Figure 7.4: Elilaw mitigation site – after intervention.

7.2.4 RAF design and construction

Flood storage pond

Cheviot Futures and the landowner agreed to construct a flood storage RAF through which to re-direct and temporarily store runoff from the entire Elilaw catchment, before it discharges into the Netherton Burn.

The RAF was excavated and the removed soil used to construct a bund (1 m max.); excess material was moved to another part of the farm to be used in Phase Two of the project. An island was built in the centre of the pond to provide new habitat (e.g., for wetland birds) and to create a more natural appearance (Plate 7.3). The pond is drained by a riser pipe that discharges into the Netherton Burn and holds approximately 1400 m^3 of water under normal flow conditions. One-metre of freeboard was created to allow for an additional 1000 m^3 of floodwater storage. An armoured spillway ensures that the bund is not overtopped and subsequently eroded.

Plate 7.3: Main pond RAF after construction (Feb 2012).

Sediment traps

Based on the experiences gained from the Lady's Well experiments at Belford (Chapter 6), it was decided to construct a number of sediment traps upstream of the flood storage pond. The sediment traps offer a multiple level of attenuation that will improve water quality by reducing DWPA, will allow for easier removal of sediment by the farmer when needed as well as slowing the sedimentation rate in the main flood attenuation pond.

To estimate sediment trap volume for the multi-cell feature a model inflow was estimated based on the discharge from a prolonged 1 mm hr^{-1} storm falling across the 70 ha catchment.

Assuming 100% runoff this would produce 700 m^3 hr^{-1}, 0.19 m^3 s^{-1}, or 194 l s^{-1}. Using Equation 6.2 ($T = V/q$) it was estimated to take circa 9 minutes and 30 seconds to settle a 30 µm (median silt) particle 50 cm. Therefore, to retain medium silt particles (and larger) with an inflow of 194 l s^{-1}, a storage volume of circa 110 m^3 is required (using Equation 6.2 - Chapter 6.4.1). This was assumed to be a conservative estimate of inflow but as runoff from the catchment is partially controlled by the mill race this is justified in this case.

A three-cell, terraced design (Figure 7.5) was selected for a number of reasons: dividing the sediment trap into multiple cells (as opposed to one large pond) using bunds helps to reduce short-cutting of flow during high discharge events, the raised outlet pipes also help to reduce remobilisation by skimming clean water from the surface; smaller cells were deemed to be easier to maintain as access by a mechanical digger is more straightforward (5-6 m is the maximum reach of an averaged-sized machine); the terracing was necessary as the ground between the inlet channel and the main pond was sloping, the area was surveyed in order to provide the digger operator with long profile designs detailing excavation depths, bund heights and outflow pipe heights.

Each cell was designed to measure 10*5*1 (m – l*w*d) in dimension to give a combined storage volume of 150 m^3, this equates to a sediment trap to catchment area ratio of circa 0.02%. As straight sides would be unsuitable due to erosion risk, the cells were excavated with 45 degree sloping banks, thus producing trapezoidal tank shapes (Plate 7.4). Following construction the cells were surveyed and total storage volumes of 52.5, 44.5 and 59.2 m^3 (in an upstream to downstream order) were calculated, giving a combined total storage volume of 156.2 m^3. Outlet pipes of 150 mm (internal diameter) were laid through the bunds at a height of 0.5 m (from the trap base to the bottom of the pipe) and at an angle of 1 degree. This means that the combined minimum volume of stored water will be 74.9 m^3.

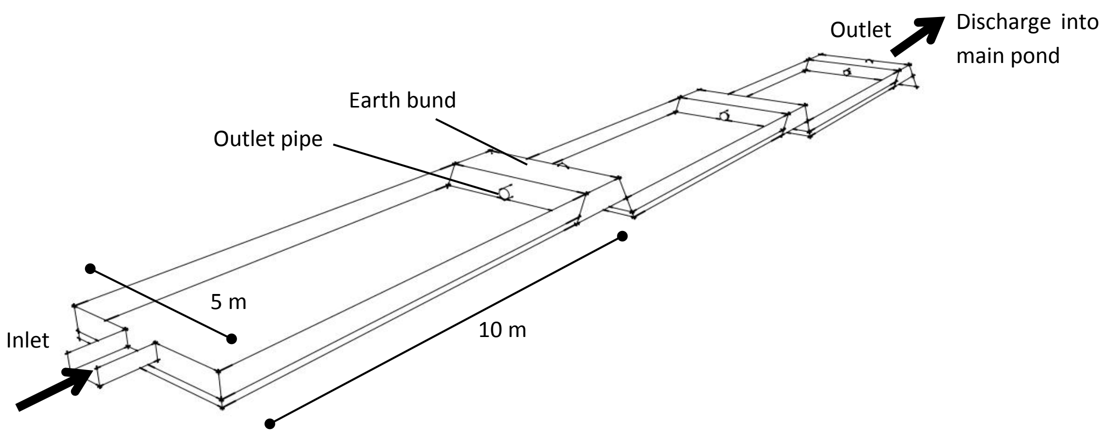


Figure 7.5: Schematic of the three-cell RAF (not to scale).

The maximum flow capacity of the outlet pipes is estimated using Equation 6.1 (Chapter 6.4.1) as 17.7 l s^{-1}. Under normal, steady state flow conditions it is estimated that between 2 and 5 l s^{-1} of water will discharge through the pipes. Residence time (calculated using Equation 6.2) is thus between 10.4 and 4.2 hours, respectively. The freeboard above the outlet pipes provides an additional 81.3 m^3 of storage capacity for storm runoff before water is released via constructed spillway points so as not to overtop the bunds and potentially erode and weaken them. Any excess water is directed alongside the feature and into the main pond. The calculation of residence time during storm conditions is difficult due to a constantly changing inflow discharge and storage volumes in each cell.

A second sediment trap RAF located to the north of the flood storage pond is a single-celled, trapezoidal design with a storage capacity of 40 m^3 (Plate 7.5). Runoff entering this feature is mainly from field drains, which discharge into the old mill race (which was modified to spill into the new RAF) and from run-on from the farm track (Plate 7.6 and Plate 7.7). This feature was not monitored as part of this study.

Plate 7.8 shows an overview of the three-cell sediment trap with the flood storage pond RAF in the background (taken from the same location as Plate 7.4) taken circa seven months later, by which time the site had fully re-vegetated – mainly naturally, with some re-seeding of the earth bunds. Phase One of the project was completed in February 2012; the site was fenced off to prevent animals entering and trees were planted by the Forestry Commission. The overall construction cost for Phase One was £10,000. The cost of the two sediment trap features was circa £1500, which includes a cost of £50/hour for a contractor and £50 per 6 m length of drainage pipe.

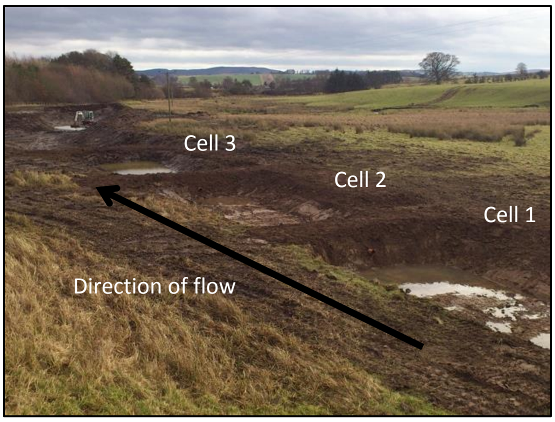


Plate 7.4: Three-cell RAF with main pond in background after construction (Feb 2012).

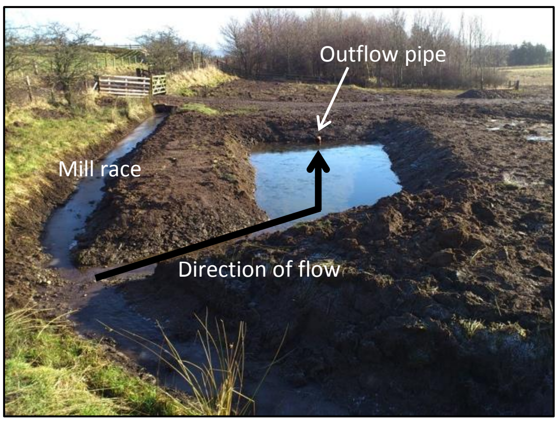


Plate 7.5: Single-cell RAF after construction (Feb 2012).

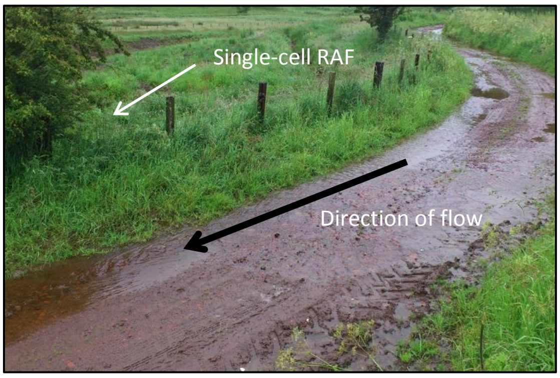


Plate 7.6: Farm track conveying runoff directly to Netherton Burn during a storm.

Plate 7.7: Inlet made to allow track runoff to enter single-cell RAF.

Plate 7.8: Overview of multi-cell RAF and main pond after re-vegetation (taken 11/09/2012).

7.2.5 RAF monitoring

The three-cell sediment trap RAF was monitored as part of this study and so all details from henceforth will concern that feature only.

Hydrology

A trapezoidal flume was constructed in the RAF inlet channel to monitor inflow discharge using a pressure transducer (and flume equation). The flume proved to be inaccurate and was replaced by a V-notch weir in November 2012. RAF outflow discharge was first estimated using a pressure transducer to measure the depth of water at the outlet pipe, which was to be converted to discharge using a hydrostatic equation for discharge through an orifice of known size (see Nicholson (2013) for details).

This method also proved to be inaccurate so a V-notch weir was also retrofitted to the downstream bund of the last sediment trap cell in November 2012, also equipped with a pressure transducer (Plate 7.9). Details of the V-notch weirs can be found in Appendix B. Water stage was recorded at 15 minute intervals and manual stage measurements were taken at each V-notch weir on every instrument download occasion. Manual measurements helped to ensure the accuracy of the pressure transducer data.

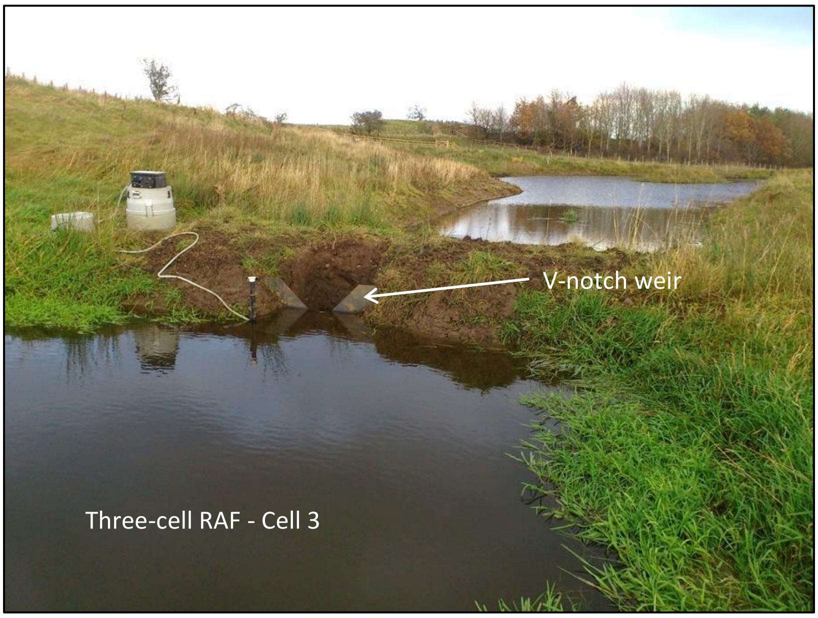


Plate 7.9: V-notch weir and automatic water sampler at the outlet of the three-cell RAF with main pond in background (taken 16/11/2012).

Water quality

To assess the impact of the RAF on sediment/nutrient loss, two automatic water samplers were deployed at the inlet and outlet. The samplers were triggered simultaneously during storm events by a float switch located next to the inlet pressure transducer. Suspended sediment, P and NO_3 concentrations were determined in the laboratory using the methods described in Chapter 3.3.4. Where available, determinand concentrations were combined with corresponding discharges to give pollutant loads.

Sediment accumulation plates were placed at four points along the bed of each cell, effectively dividing them into four zones, to allow sediment depth measurement. Sediment thickness was determined by carefully inserting a measuring rule through the sediment until it hit the sedimentation plate; the depth was then read from the rule. The volume of trapped sediment in a cell was calculated as the sum of the volumes in each of the four zones, which is given as the zone's cross-sectional area multiplied by the average sediment depth. Total trapped sediment for the RAF is given simply as the integration of the three sediment trap cells. Recovered sediment was analysed for bulk density to allow the estimation of sediment mass. The available-P content of the sediment was determined using British Standards Institute method for Olsen-P - BS 775-3.6:1995 (BSI, 1995) (see Chapter 6.4.1 for methods).

7.3 Results

7.3.1 Hydrology

Discharge data for the three-cell RAF are only available from 01/11/2012 following the installation of the V-notch weirs. As a surrogate the stage record from the downstream monitoring point in the Netherton Burn (see Figure 7.1 for location) is used to show the prevailing catchment hydrological conditions for the entire monitoring period (Figure 7.8), as well as the times when water samples were collected from the RAF by the auto-samplers. The back-up pressure transducer was required to infill missing data when the main instrument was broken; the correlation between the two instruments (R^2 = 0.99 - Figure 7.6) validates its use.

Precipitation data is available until 07/01/2013 after which there was a fault with the rain gauge. A total of 1142 mm of rainfall was recorded in 11 months between 01/02/2012 and 01/01/2013; this is significantly higher than the estimated SAAR of 1000 mm, particularly as January 2013 is not included. Figure 7.8 indicates that 2012 was relatively wet with numerous high-flow events occurring throughout the year, including the summer. The hydrograph response is 'flashy' with steep rising limbs and relatively short recessions. According to the Netherton Burn stage record (Figure 7.8) the three largest runoff events (assuming that stage and discharge are positively correlated at the site) during the monitoring period occurred before the installation of the V-notch weirs. The largest single event was recorded on 24-25/09/2012 where a stage of 1.36 m was measured in the river. This provides some context to the magnitude of the events captured at the RAF with corresponding discharge data.

Figure 7.9 depicts the RAF inlet and outlet discharges recorded at the V-notch weirs. There is a strong agreement (R^2 = 0.89 - Figure 7.7) between the two records during low and residual flow conditions, using corresponding data points (from the periods 01/11/2012-21/11/2012 and 18/02/2013-14/03/2013). There appears to be a threshold stage in the feature above which the outlet discharge plateaus (circa 40 l s^{-1} - indicated by blue ellipses in Figure 7.9). This occurs on 26/11/2012, 20, 21, 27/12/2012, and 27/01/2013, and is attributed to water exiting the feature via the spillways. Samples were collected during the 20/12/2012 storm (Event 9) and the 27/01/2013 storm (Event 10); in order to correct for this loss of water and provide more realistic estimate of input/output determinand loads, an assumption is made that outflow discharge equals inflow discharge. Discharge and precipitation summary statistics are provided for the eleven sampled events in Table 7.1.

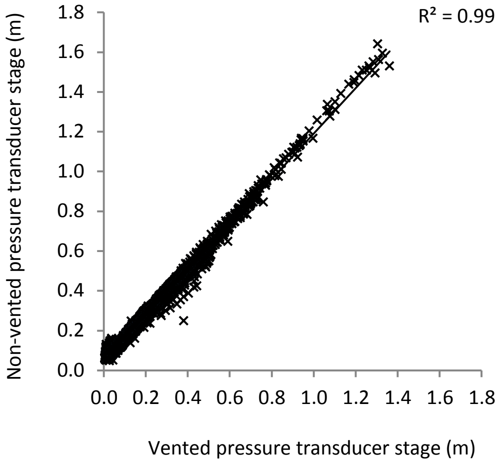


Figure 7.6: Vented vs. non-vented pressure transducer stage gauges.

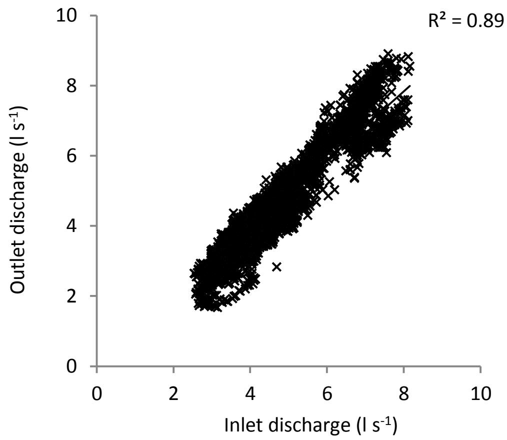


Figure 7.7: V-notch weir inlet vs. outlet discharge during low flow conditions.

Table 7.1: Elilaw RAF precipitation, discharge and sampling event summary 2012-2013.

Event	Start date/time	Duration sampled (hrs)	Total precip. (mm)	Peak hourly precip. (mm)	Peak stage* (m)	Peak discharge** ($l\ s^{-1}$)
1	20/04/2012 11:45	11	14.4	6.2	0.53	-
2	26/04/2012 09:15	21	28.4	5	0.98	-
3	10/05/2012 04:45	15	31.4	3	0.39	-
4	06/07/2012 22:30	12	36	5.6	1.02	-
5	05/08/2012 10:15	9	26	21	0.78	-
6	24/09/2012 11:00	24	78.2	6.6	1.36	-
7	12/10/2012 03:00	13	24	4.2	0.61	-
8	22/11/2012 16:00	9	10	3.8	0.38	15.3
9	20/12/2012 09:00	20	28.4	1.6	0.61	59.6
10	27/01/2013 09:30	24	-	-	0.55	67.3
11	18/03/2013 00:30	16	-	-	0.54	22.3

* Recorded at Netherton Burn
** Recorded at RAF V-notch weirs

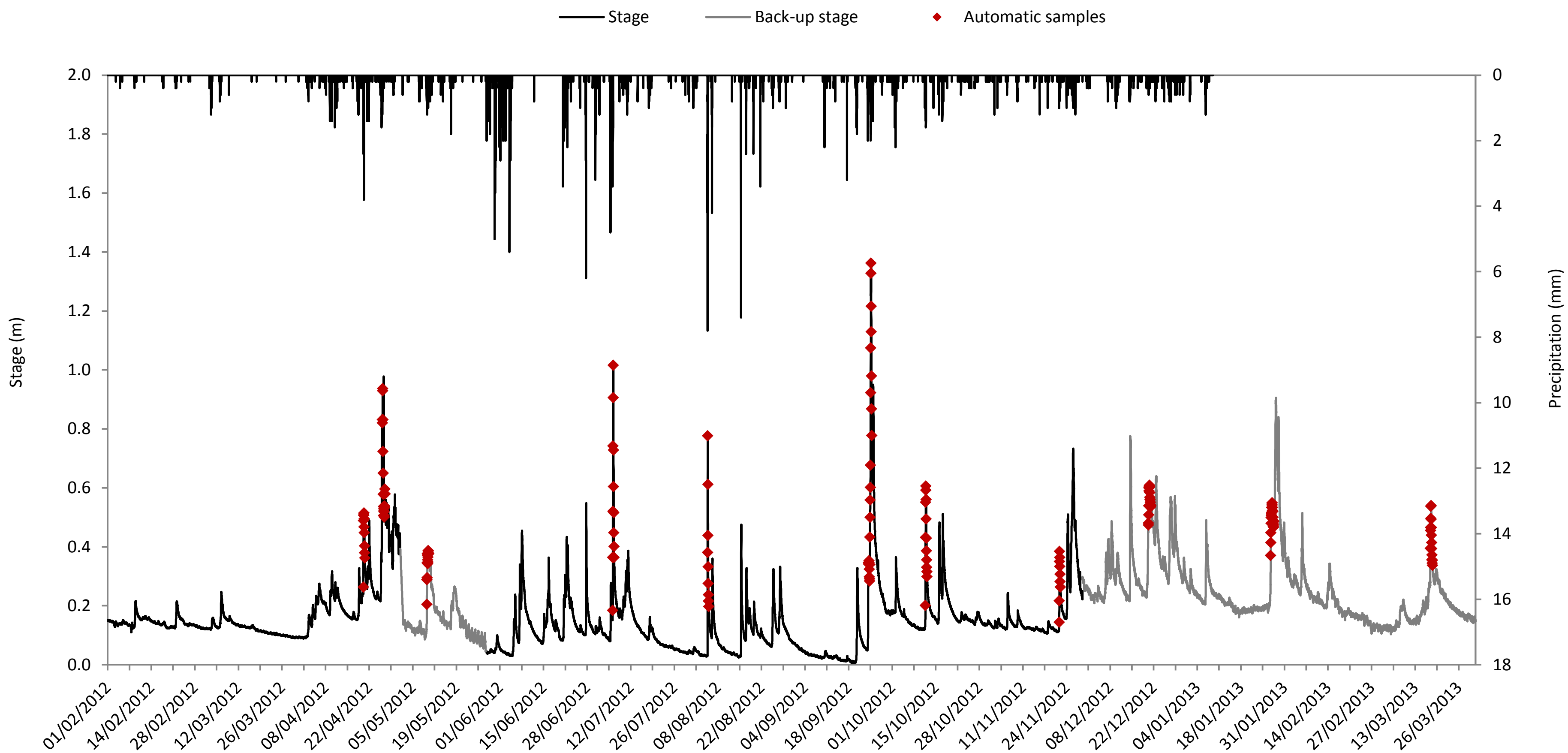


Figure 7.8: Netherton Burn stage and precipitation record. Red markers indicate the collection of automatic water samples from the RAF.

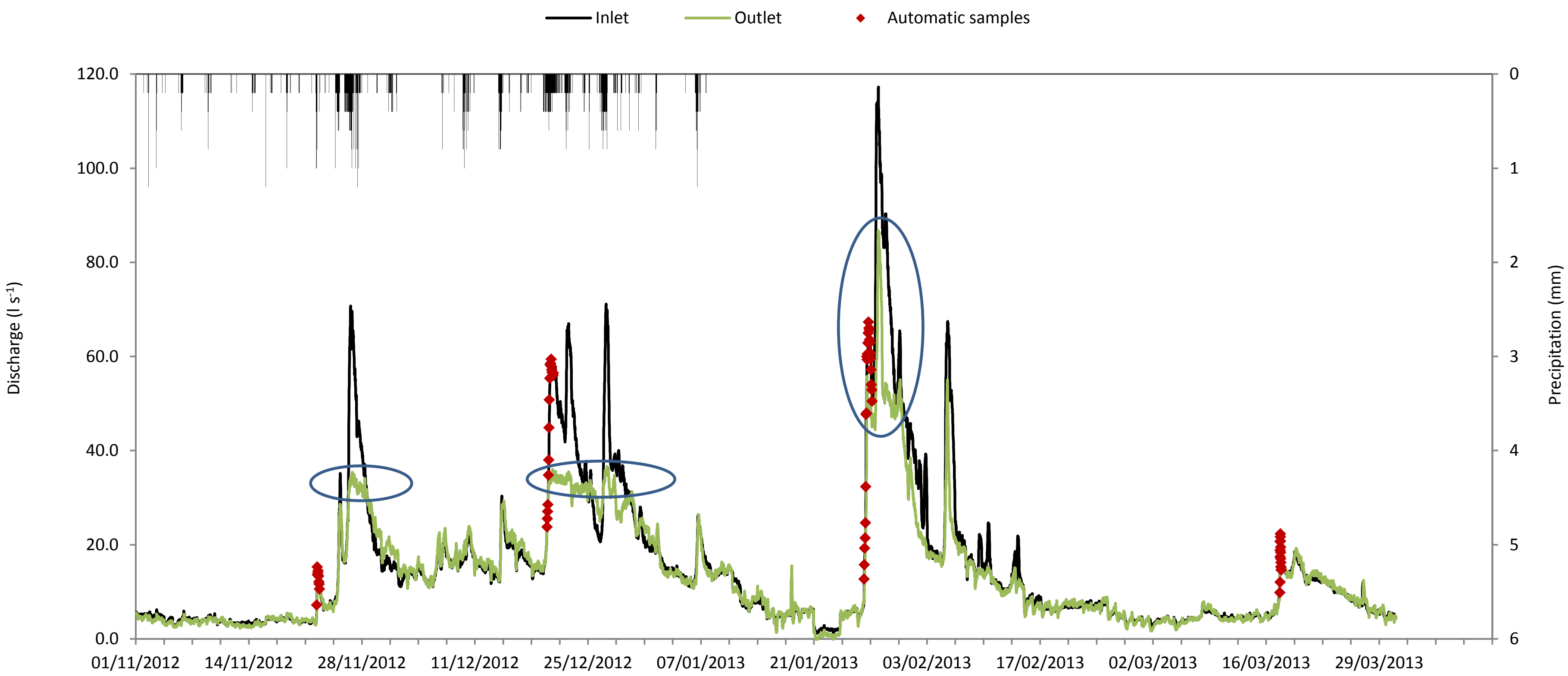


Figure 7.9: RAF inlet and outlet discharge and precipitation record. Red markers indicate the collection of automatic water samples; blue ellipses show when the spillways were in operation.

7.3.2 Water quality

Up to 174 paired samples were collected and analysed from the three-cell RAF in total over the monitoring period (less for SRP and NO_3 due to not bing able to analyse the samples in sufficient time, and unavailability of the Dionex machine, respectively). Inlet and outlet concentrations for all determinands are summarised in Table 7.2 (raw concentration and discharge data (where available) can be found in Appendix I1). Maximum concentrations of SS, TP, SRP and NO_3 of 599, 1.5, 0.26 and 9.9 mg l^{-1}, respectively, were recorded at the RAF inlet. Mean percentage concentration reductions of 42%, 26%, 15% and 5%, were calculated, respectively, all of which were significant at the 1% level (Table 7.3).

Loads were derived for Events 8, 9 10 and 11 and included 69 paired samples (49 for NO_3). As highlighted previously, water was lost from the feature via the spillways during Events 9 and 10. Therefore loads calculated using discharges recorded at the outlet V-notch weir during these events would produce under estimated determinand loads and thus over estimated load reductions. In these instances the outflow discharges are assumed to equal those recorded at the inlet V-notch weir. With this taken into account, mean percentage reductions for SS, TP, SRP and NO_3 of 43%, 30%, 19% and 14% were calculated, respectively. All reductions were significant at the 1% level (Table 7.4).

Table 7.2: Summary statistics of RAF inlet and outlet determinand concentrations for all monitored events.

	SS conc. (mg l^{-1})		**TP conc. (mg l^{-1})**		**SRP conc. (mg l^{-1})**		**NO_3 conc. (mg l^{-1})**	
	Inlet	**Outlet**	**Inlet**	**Outlet**	**Inlet**	**Outlet**	**Inlet**	**Outlet**
Minimum (mg l^{-1})	15.5	7.5	0.060	0.044	0.015	0.011	2.14	1.87
Maximum (mg l^{-1})	598.7	208.3	1.497	1.028	0.256	0.197	9.90	8.99
Mean (mg l^{-1})	116.1	61.9	0.529	0.382	0.085	0.071	5.75	5.47
Median (mg l^{-1})	85.3	44.6	0.505	0.368	0.072	0.060	6.29	5.80
n	174	174	174	174	122	122	108	108

Table 7.3: Mean percentage concentration reduction and significance.

Determinand	**n**	**Mean % reduction**	**Paired T-Test**	
			***T*-value**	***P*-value**
SS	174	42.4	11.53	<0.001
TP	174	25.7	13.88	<0.001
SRP	122	14.9	7.22	<0.001
NO_3	108	5.2	7.01	<0.001

Table 7.4: Summary of pollutant loads, mean percentage reduction and significance.

Determinand	n	Mean load (instantaneous)		Mean % reduction	Paired T-Test	
		Inlet	Outlet		T-value	P-value
SS (g)	69	3.30	1.96	42.6	10.20	<0.001
TP (mg)	69	20.58	15.5	29.5	9.69	<0.001
SRP (mg)	69	3.53	3.07	19.2	9.49	<0.001
NO_3 (mg)	49	188.8	172.2	13.9	6.95	<0.001

7.3.2.1 Storm event data

Paired water samples were collected from the inlet and outlet of the three-cell RAF during eleven storm events (Table 7.1). Time series charts for all events can be found in Appendix I2.

Event 6

This event occurred in September 2012 and is depicted in Figure 7.10; 24 samples were collected. Antecedent conditions were relatively dry for the week before the event, apart from a medium-sized storm three days previous. Stage data recorded at the inlet flume are shown as the V-notch weirs were yet to be installed.

Both SS and TP inlet concentrations peak a considerable time before peak stage (circa 8 hours) with maximum TP measured one hour later than SS. Peak SS and TP concentrations are recorded at the outlet one and two hours later, respectively, and are 44% and 31% lower, respectively, than peak inlet concentrations. A reduction in determinand concentration was measured at every time step, with SS ranging between 14-61% retention and TP between 13-41%; mean concentration reductions for the event of 40% and 25%, respectively, were recorded (Appendix I3 contains determinand concentration/load reduction summary tables for each event).

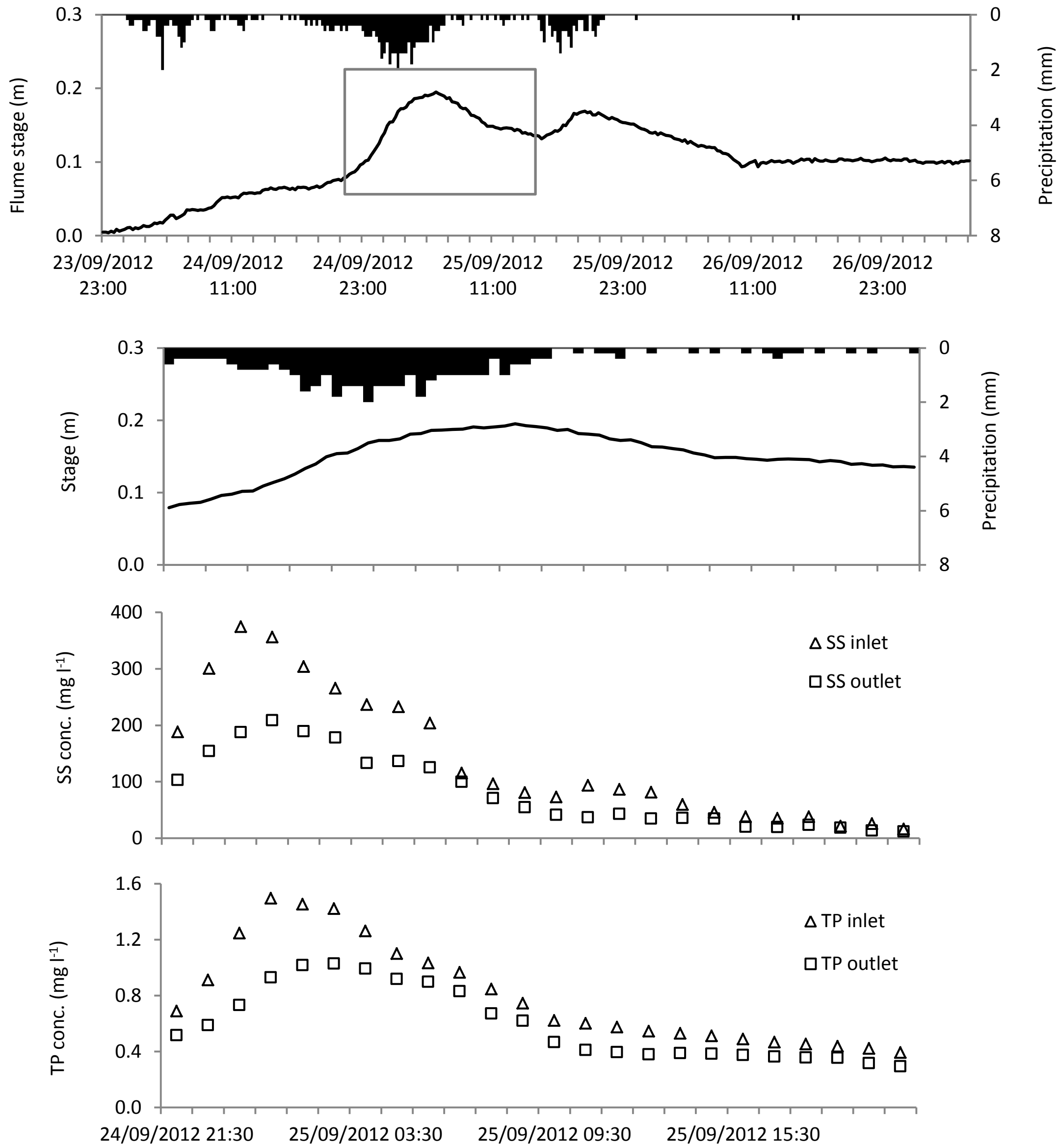


Figure 7.10: Discharge, SS and TP concentration record – Event 6.

Event 8

This storm event occurred in November 2012. It was a relatively small storm but yielded the largest mean reduction in SS concentration and in load for all determinands (Figure 7.11). The inflow hydrograph exhibits a more 'spikey' response to precipitation while the out flow response is more 'smoothed' and shows some evidence of attenuation. There is, however, increased discharge recorded at the RAF outlet after peak discharge at the inlet for some time as the recession is slower.

Samples were collected for nine hours and analysed for all determinands. A maximum SS concentration reduction of 88% was recorded with a mean of 55% across the whole event;

total SS load was reduced by 83%. The greatest reductions occurred on the rising limb and at peak discharge. Total P reduction exhibited a similar pattern to SS with a mean reduction of 27%, no net losses, and a total load reduction of 67%. These values are greater than corresponding ones for SRP (15% and 63% respectively) and the time to peak concentration was observed to be shorter (Figure 7.11). There were occurances where outlet concentrations of SRP were higher than corresponding inlet concentrations (up to 8%), which occurred at the tail-end of the recession. Nitrate concentrations did not peak in relation with discharge but instead increased gradually throughout the duration of the storm, although maximum reduction was recorded at the same time as peak discharge. Concentration reduction ranged between -0.8% and 24% (mean 6.8%) but a considerable load reduction of 57% was measured, significantly greater than in any other event.

Event 10

This event yielded the highest discharge measured by the V-notch weirs with a peak discharge of 67.3 l s^{-1} (at the inlet - Figure 7.12). The loss of discharge via the feature's spillways is clearly evident. However, this is believed to have no significant impact on determinand concentration, unlike load. Precipitation data are unavailable for this event due to rain gauge failure.

Peak SS and TP concentrations at the RAF inlet occur simultaneously circa one hour before the first, smaller discharge peak, which is recorded at circa seven hours. The most significant reductions in SS and P are on the rising limb and at the first discharge peak. The SS and TP response at the outlet is both reduced and attenuated; SS concentrations are lowered by a mean of 35% and TP is reduced by 25%. No hourly losses are recorded. Soluble reactive P demonstrates a more 'smoothed' response to discharge at both the inlet and outlet with relatively less retention. Peak SRP concentrations are recorded 2-3 hours after TP and coincide more with the second larger discharge spike. Soluble reactive P concentration retention ranges between -18% and 49% with a mean of 13%. Nitrate concentration exhibits a non-response to the first minor discharge peak and only starts to increase following the second larger peak. Concentrations then rise steadily from circa 4 mg l^{-1} to circa 7 mg l^{-1} and plateau after 20 hours. Concentration reduction ranges between -10% and 22% with a mean of 5%.

Loads calculated using matching inflow and outflow discharges (to overcome water loss via the spillway) give SS, TP, SRP and NO_3 reductions of 32.6%, 22%, 10% and 4%, respectively. This is in contrast to the values calculated using the 'actual' discharge data recorded at the V-notch weir, where reductions of 45.2%, 37.2%, 27% and 22%, respectively, are considerably higher.

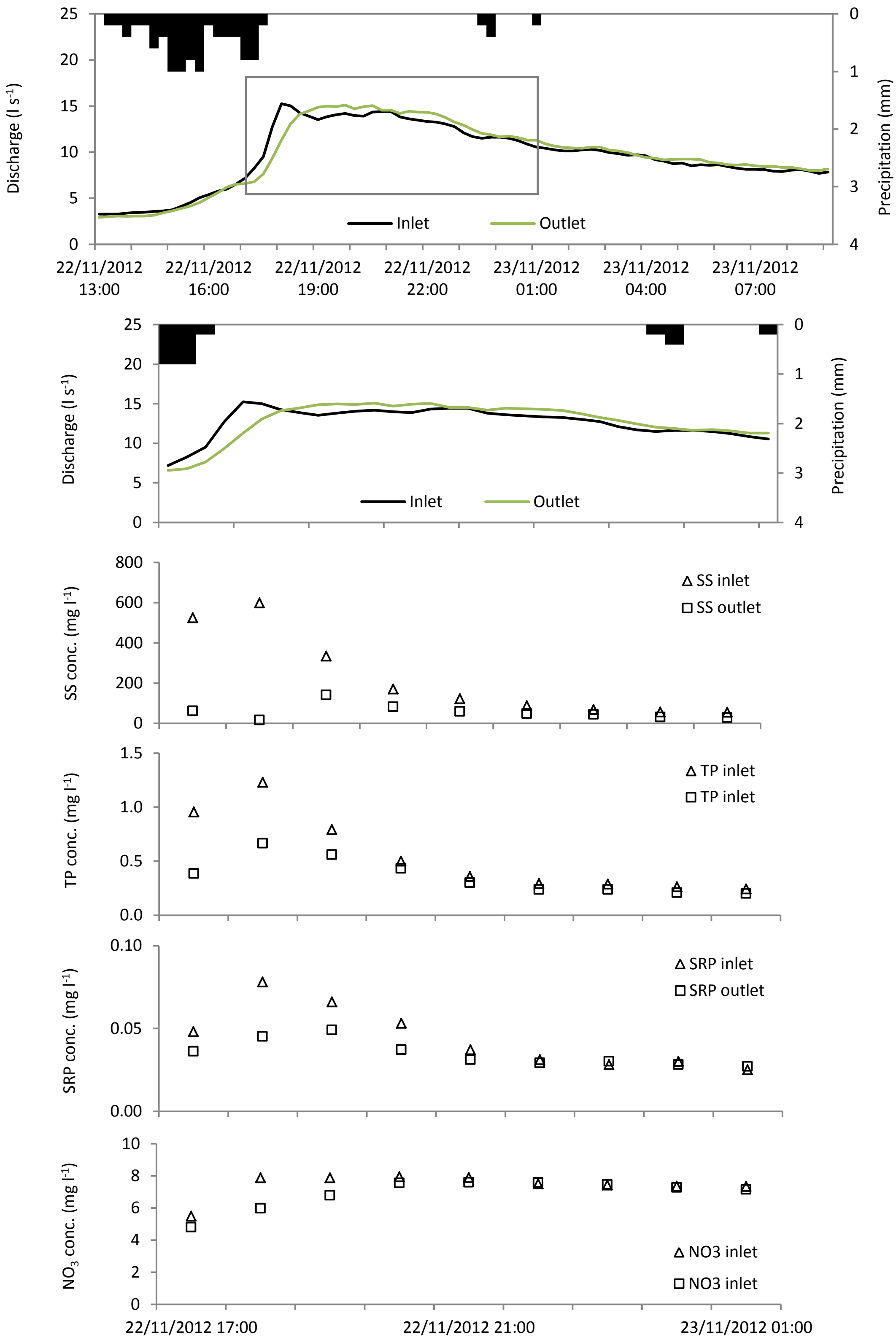


Figure 7.11: Discharge and SS, TP, SRP and NO_3 concentration record – Event 8.

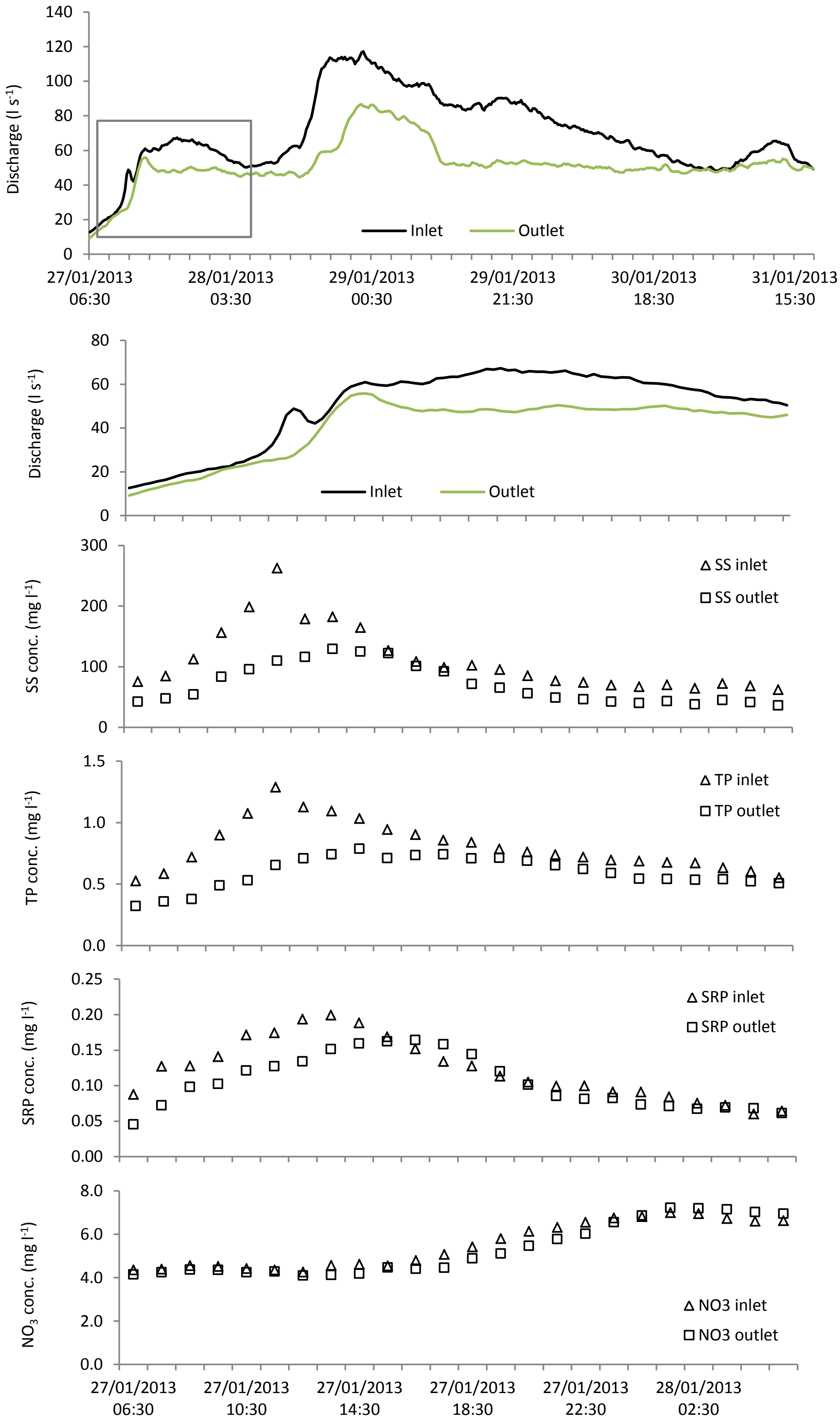


Figure 7.12: Discharge and SS, TP, SRP and NO_3 concentration record – Event 10.

7.3.2.2 RAF efficiency in relation to determinand concentration and discharge

A strong positive correlation exists between inlet SS concentration and removal (Figure 7.13 *a*). This correlation is also observed for TP and SRP, but with lower Pearson's *R* values (Table 7.5); all correlations are significant at the 1% level. No correlation exists for NO_3.

Conversely, RAF efficiency is negatively correlated with inlet discharge for SS (Figure 7.13 *b*) and TP (both significant at the 1% level - Table 7.5). Soluble reactive P exhibits a weak, negative correlation and discharge appears to have no influence on NO_3 removal.

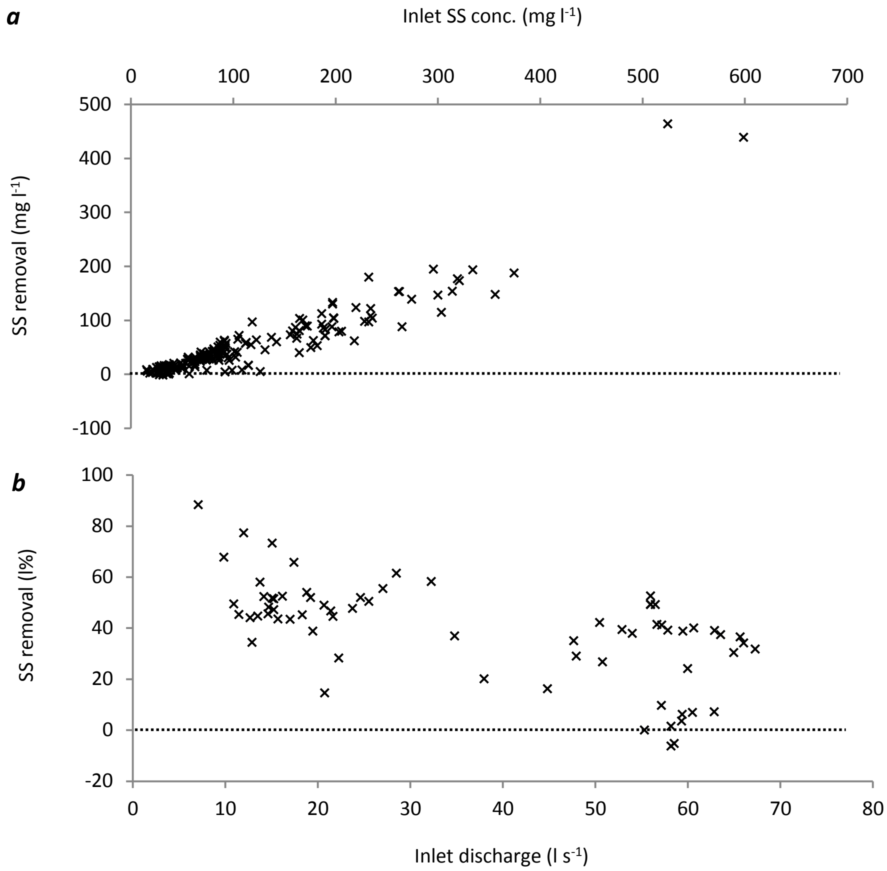


Figure 7.13: The relationship between SS reduction and *a*) inlet concentration (measured at the RAF inlet) and *b*) discharge.

Table 7.5: Pearson's correlation coefficients for RAF efficiency.

Determinand	**Conc. vs. reduction (mg l^{-1})**		**Discharge vs. reduction (%)**	
	Pearson's *R*	***P*-value**	**Pearson's *R***	***P*-value**
SS	0.930	<0.001	-0.608	<0.001
TP	0.821	<0.001	-0.566	<0.001
SRP	0.601	<0.001	-0.306	0.011
NO_3	0.179	0.063	0.009	0.949

Total P concentration during storm events appears to exhibit similar patterns to SS but sometimes with a slightly delayed response. SS and TP concentrations (Figure 7.14) measured at the RAF inlet and outlet exhibit a weak positive correlation with relatively few data points at the high end of the concentrations scales. Figure 7.15 depicts the proportion of TP accounted for by SRP at both the RAF inlet and outlet, the inverse of which can be used as a proxy for the proportion of particulate P. On average SRP accounts for 21% of TP at the inlet and 28% at the outlet.

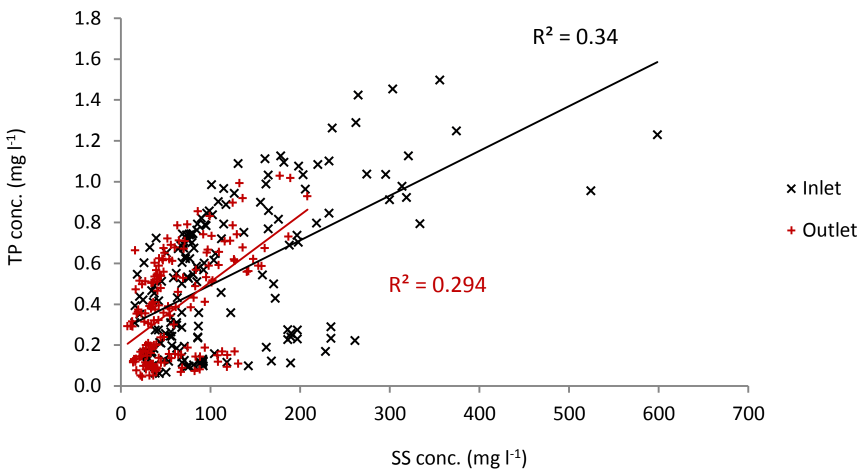


Figure 7.14: Relationship between SS and TP concentrations.

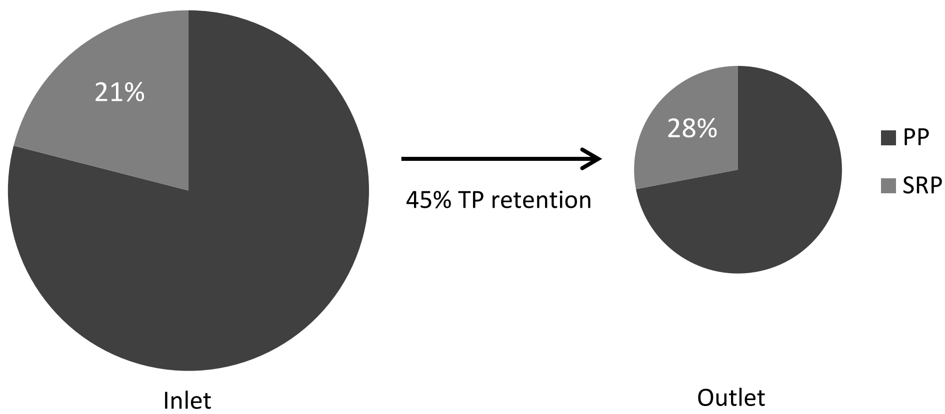


Figure 7.15: Percentage TP made up of SRP - comparison between RAF inlet and outlet (n = 122 for each).

7.3.2.3 Sediment trap sediment accumulation

The depth of accumulated sediment was measured in April 2013; Plate 7.10 depicts cell one, where the height of the deposited material had reached the bottom of the outlet pipe. The total volume of captured (wet) material is 22.25 m^3 (Figure 7.16). On average, sediment depth was greatest in cell one (upstream) and lowest in cell three (downstream). Within cells one and two sediment depth increased between the inlet and the outlet while the greatest depth in cell three was found in the central zone.

The bulk density of the recovered sediment is highest at the inlet of cell one (1.105 g cm^{-3}) and lowest at the outlet of cell three (0.859 g cm^{-3}). Combined with corresponding sediment volumes this gives an estimated total of 22.02 tonnes of sediment (dry mass) retained between March 2012 and April 2013, which equates to a trapping rate of circa 0.31 t ha^{-1} yr^{-1}.

If average SS retention is 43% and 0.31 t ha^{-1} was retained in the RAF, this suggests that the unmitigated sediment loss rate for the catchment would be circa 0.72 t ha^{-1} yr^{-1}, or 72 t km^{-2} yr^{-1}. This value falls above the range of SS export coefficients reported in Chapter 4, of 25 and 60 t km^{-2} yr^{-1} for improved agricultural land and tilled land, respectively. Natural England proposed critical and target SS yields based on catchment typology. For a lowland impermeable catchment, such as Netherton, the critical SS yield is 50 t km^{-2} yr^{-1} and the target is 20 t km^{-2} yr^{-1} (Cooper *et al*., 2008). Based on the trapping values for the three-cell RAF, catchment sediment loss was reduced from 72 t km^{-2} yr^{-1} to 41 t km^{-2} yr^{-1}; although not reaching the target export, if repeated over long term it suggests a much healthier sediment yield according to the NE classification.

Sediment available-P concentrations of 5.17, 7.3 and 5.05 mg l^{-1} were recorded in cells one, two and three, respectively, which equates to 51.7, 73.0 and 50.5 mg kg^{-1} sediment. This gives a P mass of 1.26 kg for the entire feature, 0.018 kg ha^{-1} or 1.8 kg km^{-2} yr^{-1}. Cell one yielded the highest P trapping rate of 0.01 kg ha^{-1} yr^{-1}, followed by 0.006 kg ha^{-1} yr^{-1} in cell two and 0.002 kg ha^{-1} yr^{-1} in cell three.

Plate 7.10: Cell one of the three-cell RAF contain 1 circa one year's worth of sediment (taken April 2013).

Sediment trap zone

Cell 1	Inlet	Centre		Outlet	Total
Area (m^2)	12.10	13.64	12.40	10.20	48.34
Sediment depth (m)	0.19	0.27	0.29	0.31	
Sediment volume (m^3)	2.30	3.68	3.60	3.16	12.74
Bulk density (g cm^{-3})	1.105	1.105	0.992	0.992	
Sediment mass (t)	2.54	4.07	3.57	3.14	13.31

Cell 2					
Area (m^2)	8.40	10.07	12.78	6.93	38.18
Sediment depth (m)	0.12	0.15	0.19	0.19	
Sediment volume (m^3)	1.01	1.51	2.43	1.32	6.26
Bulk density (g cm^{-3})	0.953	0.953	0.932	0.932	
Sediment mass (t)	0.96	1.44	2.26	1.23	5.89

Cell 3					
Area (m^2)	12.60	14.35	14.88	11.10	52.93
Sediment depth (m)	0.06	0.07	0.07	0.04	
Sediment volume (m^3)	0.76	1.00	1.04	0.44	3.25
Bulk density (g cm^{-3})	0.878	0.878	0.859	0.859	
Sediment mass (t)	0.66	0.88	0.89	0.38	2.82

Total sediment mass (t) **22.02**

Direction of flow

Figure 7.16: Three-cell RAF accumulated sediment.

7.4 Discussion

7.4.1 Hydrology

As hydrometeorological monitoring only took place in the Netherton catchment for circa one year it is difficult to extrapolate the hydrological conditions during this study with reference to long term patterns. However, circa 150 mm more rainfall than the estimated long term average (SAAR) fell in just 11 months meaning that the study year was wetter than average. There were also numerous medium-large storm events during the summer, which could be classed as unseasonal. Thus, it should be acknowledged that the collected data, thence the performance of the RAF, are only representative of the monitored period.

The installation of V-notch weirs at the inlet and outlet of the RAF made the measurement of discharge significantly more accurate and allowed the subsequent calculation of sediment and nutrient loads. Comparison of the inflow/outflow hydrographs demonstrated that the weirs (and the pressure transducers) were in acceptable agreement during residual flow conditions, thus giving confidence in their measurements during other flow conditions. Despite the relatively long logging interval of the stage gauges, it is apparent that the three-cell RAF caused some attenuation of rising limbs and peak flow between the inlet and outlet. This attenuation effect is inversely proportional to increasing discharge, as an increase in flow will result in a decrease in residence time, given a constant storage volume.

On average, event peak discharge during the monitoring period was approximately 70 l s^{-1}, which equates to an estimated residence time of 37 minutes for the given storage volume of the three- cell RAF. The feature would provide a residence time of circa 23 minutes during the maximum recorded discharge of 115 l s^{-1}. Without intervention along this 60 metre length of ditch it is estimated that the time of travel for flow would be between one and five minutes. According to the calculations carried out in Chapter 6 using Stoke's law for particle settling velocity, 23 minutes would be sufficient time to settle a 20 µm particle (medium-fine silt) to a depth of 50 cm. Braskerud (2003) reported how SS (principally clay particles) retention in small wetlands often exceeded expectations based on calculations such as Stoke's Law. Experimental results showed how clay particles behaved as fine silt and medium silt (~20 µm) particles, thus indicating strong aggregation. This will be discussed in more detail below.

The fact that a portion of inflow is lost from the RAF via the spillways during the highest discharge events (Plate 7.11) has been taken into account when calculating determinand load reductions. Otherwise in events where paired samples were collected and water was lost via the spillways the calculated loads will be an underestimate of the true values. Figure 7.12 suggests that at discharges greater than 60-70 l s^{-1} flow is lost from the RAF via the spillways;

higher discharges than this were recorded at the outlet but quickly return to this ‘plateau’ value. However, it is not feasible to construct sediment traps large enough to accommodate the full range of discharges due to size restrictions imposed by the landscape and neighbouring farming practices. In this instance flow lost via the spillways is directed into the flood storage pond over rough grass, thus is not entering the main burn untreated.

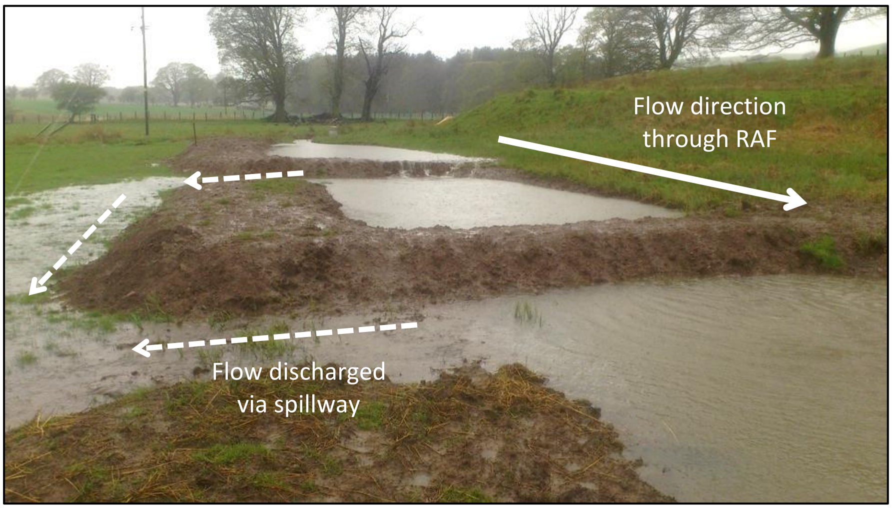


Plate 7.11: Three-cell RAF during a flood (taken 26/04/2012).

7.4.2 Sediment and Nutrient retention

Over a period of circa one year, based on four storm events of differing season, magnitude and duration, the three-cell RAF reduced SS, TP, SRP and NO_3 loads by 43%, 30%, 19% and 14%, respectively. These sediment and nutrient reduction values are within the ranges reported in the literature: Fiener *et al.* (2005) described how small detention ponds in Germany trapped 54-85% of the incoming sediment load; Reinhardt *et al.* (2005) reported a TP load reduction of 23% in a small agricultural wetland in Switzerland but it is notable that SRP made up the majority of TP in this instance, which may explain relatively poor performance. Uusi-Kämppä *et al.* (2000) report that ponds and constructed wetlands reduced TP loads by 17 and 41% respectively, and Moreno *et al.* (2007) reported 24-43% removal of TN using natural and constructed wetlands of between 50 and 800 m^2.

7.4.2.1 Retention efficiency

Suspended sediment, TP and SRP all demonstrated greater retention with increasing inlet concentrations. As the concentrations recorded at this site are relatively low in comparison to some other study sites (including Lady’s Well in Belford and Blind Beck in the upper Eden catchment) RAF performance may be expected to be higher at more polluted sites. Other constructed wetland and detention pond studies have found that sediment retention often increases with increasing inlet discharge (e.g., Braskerud (2003)); however, this was not the case at Netherton. Data collected at the RAF show that sediment and nutrient peak concentrations are often delivered before peak discharge (maximum SS and TP reduction also occurred on the early stages of the rising limb), suggesting that pollutants are quickly mobilised in the upstream channel. Suspended sediment and P concentrations are then seen to decrease before and at peak discharge, which is likely due to dilution with cleaner water from other parts of the catchment. Retention efficiency of SS and P also decreases at peak discharge, thus is appears that the trap is efficient on the rising limb before residence time gets too short

On average SRP accounts for 21% of TP measured at the inlet and 28% at the outlet. This indicates that the retention of PP exceeded that of SRP and suggests that sedimentation was a more important retention process than others such as uptake by algae and macrophytes. However, although cell one trapped considerably more (126%) sediment than cell two, the highest sediment available-P concentration (73.0 mg kg^{-1}) was found in cell two of the RAF, compared with circa 50 mg kg^{-1} in cells one and three. This is likely the result of the settling of PP, which is associated with fine-texture sediment (i.e., cell one was filled with a higher proportion of coarse sand and gravel, especially near the inlet). The overall P trapping rate for the three-cell RAF was estimated at 0.018 kg ha^{-1} yr^{-1}, compared with 0.004 – 0.007 kg ha^{-1} yr^{-1} measured at Belford. The greater efficiency is most likely attributed to its increased sediment trap to catchment area ratio (0.02% compared with 0.007%) providing more residence time for the settlement of PP. All of these trapping rates fall within the range recorded across ten sites in the first year of monitoring in the MOPS project of 0.006 – 1 kg ha^{-1} yr^{-1} (Ockenden *et al.*, 2012).

The variations in retained sediment thicknesses in the three cells can be explained primarily by the different hydrologic conditions of the ponds and secondly by the variation in sediment texture between the deposits. Analysis of retained sediment revealed that bulk density decreased from the inlet to the outlet (across each separate cell and across the RAF as a whole), as one would expect due to the effects of settling velocities and residence time. When considering the bulk density of pond sediments it is important to acknowledge the effects of compactions by overlying material. Coarse sediments (sand and gravel) settle down with relatively little available pore space and have an initial dry sediment bulk density close to their final value. Fine sediments settle down with a lot of water occupying the space between the grains, resulting in a low initial bulk density. Over time compaction (by overlying sediment) causes the particles to move closer together and water to be squeezed out, both of which reduce the sediment volume (Verstraeten and Poesen, 2001). Thus, the difference between initial and final bulk density for fine sediment can be significant.

Exposure of retained sediment to air (e.g., cell one, as depicted in Plate 7.10) can also have a significant impact of the final bulk density (Verstraeten and Poesen, 2001). However, as this occurred only in cell one, which was filled with relatively coarse material, the process is less important due to the close proximity of their initial and final bulk density values. The finer-grained sediment in cells two and three were not known to be exposed to the air during the study period.

According to Walling (1990) and Slattery and Burt (1997) the runoff from agricultural catchments produces sediment in aggregated form and Slattery and Burt (1997) reported that the proportion of fine primary particles increased, and the proportion of sand decreased, with increasing discharge. The fact that the concentration of P was greatest in cell two supports these findings. Condron *et al.* (2005) reported that soil solution P concentrations typically range between <0.01-1 mg l^{-1} but can be as high as 7-8 mg l^{-1} in well fertilised soils. Sediment available P concentrations measured in this study are at the upper-end of this range. It is probable that sediment enrichment in fine particles relative to topsoil explains the relatively high P concentrations. The three-cell sediment trap captured circa 22 tonnes of sediment during the monitoring period (one year), and cell one (containing circa 13 tonnes) required emptying. The landowner was surprised at the rate at which the feature had filled and upon learning the potential fertilising value of the trapped material, said that he would spread it back to land once it was removed from the RAF (P. Stott, landowner, pers. comm.).

7.4.3 RAF design

An advantage of dividing the RAF into smaller cells, compared with one large feature is that the upstream cell can be emptied of retained sediment with relative ease while the others can be left undisturbed for longer. Sediment trap size vs. regularity of sediment recovery is a trade-off. The regularity of maintenance will also vary according to the individual site but an assumption is made that cell one at Netherton will need clearing out on an annual basis; cells two and three on a bi-annual basis. Such a management regime does not appear too onerous a task for the farmer to undertake. Another advantage of a multiple-celled RAF is that downstream cells will help to mitigate the impact of emptying upstream ones, which will inevitably cause some re-mobilisation of sediment and associated nutrients.

In the case of Netherton, where flood management is paramount, the sediment trap features are all helping to extend the lifespan of the main floodwater attenuation feature, which would be very arduous and expensive to dredge due to its surface area and depth. If the 22 m^3 of sediment retained in the traps were to reach the main pond at this rate it would reduce its water storage capacity by circa 1% per annum.

7.5 Summary

Approximately 22 tonnes of sediment was captured in a three-cell sediment trap RAF, with a surface area to catchment area ratio of 0.02%, in a single year. On average SS, TP, SRP and NO_3 loads were reduced by 43%, 30%, 19% and 14%, respectively, across the full-range of flow conditions experienced during the monitoring period.

Dividing the sediment trap RAF into separate cells is a more effective design at retaining sediment and nutrients compared with a single-cell feature. This is because flow cannot 'short-circuit' the system as easily (in a way that vegetation of baffles may function), particularly during high flow events with increased velocities. Maintenance of the sediment trap RAFs will include periodic emptying. Based on one-year of monitoring it is estimated that the upstream cell of a three-cell trap will require annual attention and downstream cells biannually.

This relatively low-cost intervention (along with others at the same site) will serve to not only improve water quality by mitigating sediment and nutrient losses to the main Netherton Burn, but also will increase the lifespan of a large flood attenuation RAF by reducing sedimentation. Thus, as a suite of RAFs can be considered a truly multi-functional scheme that is treating runoff (and associated DWPA) delivered via a number of flow pathways from the 80 ha catchment.

The subsequent chapter will provide an overall summary that draws together the sediment and nutrient regime studies carried out in the upper Eden catchment and the DWPA mitigation experiments conducted in the Belford and Netherton catchments. It will describe the lessons learned throughout the study and provide better understanding of how to reduce losses of SS, P and NO_3 in agricultural runoff using RAFs to target various runoff/pollutant pathways at a range of spatial scales.

8. Overall summary

This thesis has two principal aims that can be summarised by the following questions:

- *How, where and when is sediment/nutrient pollution generated?*
- *How, where and when can DWPA mitigation be targeted best?*

The following chapter will provide answers to these questions by compiling the outcomes of the sediment/nutrient regime characterisation studies carried out across the upper Eden catchment, and in the Blind Beck sub-catchment, with the results from the DWPA mitigation case studies implemented in the Belford and Netherton catchments.

8.1 Sediment and nutrient regimes

This study has provided sediment and nutrient data across a range of catchment scales. The upper Eden catchment sediment/nutrient regime has been characterised using a multi-scale, stratified, synchronous grab sampling campaign that clearly identifies spatio-temporal variability in SS, P and NO_3 fluxes; however, no relationship was found between sediment/nutrient yield and catchment area. There appears to be more heterogeneity between smaller sub-catchments, with an increased likelihood of extreme values, but a more 'averaged' pattern at the larger scale. It was recognised that certain lowland sub-catchments deliver a disproportionate amount of the pollutant load, due to increased agricultural activity, and that there were large variations in flux affected by season and hydrological conditions. Dilution of potentially polluting lowland sub-catchments by relatively 'clean' water inputs from headwater areas means that the upper Eden catchment (at larger catchment scales) is currently on course to meet WFD targets.

The Blind Beck sub-catchment was singled out for more detailed study as it exhibited higher nutrient and SS concentrations per unit runoff as well as higher sediment and nutrient yields per unit area than any other sub-catchment. However it was recognised that a more intensive grab sampling campaign was required to identify the main sources of sediment and nutrients within the catchment. Blind Beck, particularly in the lower reaches (near Sykeside Farm) was characterised by a high availability of SS and associated P, which was related to higher agricultural intensity and a greater extent of superficial sediment deposits. These gave rise to *near-channel* contaminant sources, such as bank collapses and areas of poaching, both of which can be linked with livestock management.

The use of automatic water samplers to collect storm event data demonstrated that sediment and P losses were dominated by high flow events, with minimal inputs under dry weather conditions. High flows (accounting for 10% of flow duration) contributed 84% of the annual SS load, 76% of the total P and 68% of the soluble reactive P (SRP), but just 32% of the NO_3 load. This highlights the acute nature of the SS and P diffuse pollution problem in the upper Eden. Hysteresis analysis largely confirmed that sources were in relatively close proximity to the watercourse, and that the source was either quickly exhausted or that subsequent dilution was occurring, or both. Intra-storm variability in export was reliant on storm size, antecedent conditions, and also the occurrence of contiguous events in quick succession leading to exhaustion of *near-channel* sources. It was found that sediment/nutrient mean concentrations and load estimates were significantly higher (for all determinands) when derived from both event samples and grab samples. Thus the exports based solely on grab samples represent significant underestimates.

The construction of a wetland RAF at Sykeside Farm in the Blind Beck sub-catchment, and subsequent monitoring, highlighted the importance of selecting DWPA mitigation options that are appropriate to the dominant pollutant source-pathways. The identified SS and P sources in the catchment would be best managed using stream bank protection measures such as stock fencing and riparian buffer strips, and not RAFs. This reinforces the need for an approach that involves initial evaluation of whether this particular method is viable.

8.2 Appropriate choice of mitigation option

The source-pathway for sediment and nutrients in agricultural catchments is critical to the identification of appropriate mitigation strategies. In catchments with slowly permeable soils, such as Belford, sub-surface (tile) drainage networks appear often to be a significant sediment/nutrient loss pathway, particularly as a result of permanent and high-level connectivity between distant parts of the catchment and the watercourse. This scenario, especially combined with a susceptible land use (e.g., arable), will lead to an increase in the 'chronic' loss of sediment and nutrients as transfers during low-medium magnitude events will be greater in comparison with catchments without artificial drainage. However, landscape processes such as concentrated overland flow also pose a significant threat to soil erosion and/or sediment/nutrient loss. Monitoring at the event scale in this thesis has shown that SS, P and NO_3 concentrations and loads are, to varying degrees, all positively correlated with discharge, thus equating to significantly increased pollutant transfers during large storm events. Although these events are often short in duration, the potential impact of the delivery of high levels of sediment and nutrients will depend on the nature of the receptor. For example, if the polluted discharge from a sub-catchment is significantly diluted by the input of downstream ‘clean’ sub-catchments (as appears to be the case in the upper Eden catchment) then the need to target ‘acute’ events may be less. Conversely, in catchments such as Belford, which discharge into ecologically sensitive ecosystems, the targeting of large events synonymous with concentrated surface runoff is justified.

The literature review chapter gave an account of the many DWPA mitigation options available to farmers and landowners. It appears that ‘source-mobilisation’ mitigation is arguably the most effective measure, particularly conservation measures such as minimum tillage. For arable systems reversion to grassland is perhaps the option most guaranteed to yield the best results, but is only feasible in limited very high risk areas where high levels of sediment/nutrient could severely threaten water quality. In a situation where population growth, food shortage and potential climate change are serious pressures, the need for farmers to produce food is ever increasing and it may not be wise to reduce the amount of productive land. While many in-field mitigation options are available, , the use of RAFs (as an example of DWPA ‘transport’ management, or ‘edge of field’ or ‘in-channel’ options) can be justified on the level that their impact on normal farm operations is very small.

Riparian and field-edge buffer strips have received a relatively large amount of research attention and real world uptake due to their perceived effectiveness, low cost and multi-functionality. As a means of preventing the degradation of riparian areas and river banks, buffer strips could be considered as adequate. However, they appear to be largely ineffective where concentrated overland flow or subsurface field drains are the dominant contaminant pathways. RAFs, including the ones described in this study and others including grassed water ways, have received less attention, particularly in the UK. They are generally more specialised and necessitate a degree of ‘design’ to ensure their suitability, which is associated with higher costs and a need for on-going management. Considerate planning and/or a good knowledge of the catchment’s hydrological functioning is expedient in order to best locate mitigation features to insure maximum efficiency. The desired impact may be on water quality or flood risk, or indeed both, as has been demonstrated in the Belford and Netherton catchments.

The multi-functionality of RAFs may go some way to offsetting their relatively high financial cost and although not investigated in this study, they can potentially add to/enhance the buffering capacity (in terms of flooding and DWPA) of catchments, which could help mitigate for future agricultural intensification and/or climate change. In terms of uptake, a farmer would be more willing to have RAFs constructed on their land, for example, as a form of natural flood management (to lower flood risk in a downstream settlement) if they were simultaneously receiving the on-site benefits of sediment and nutrient retention. The funding of RAFs is also more justifiable as more 'win-wins' are attainable; including habitat creation, carbon sequestration and other ecosystem services.

Runoff Attenuation Features appear most viable used alongside 'source-mobilisation' options and not instead of them. In the case of the Blind Beck catchment and Sykeside Farm (and other livestock systems common in the upper Eden catchment), the prevention of animal access to stream banks is the most simple and effective solution. Stock fencing can also be beneficial to the animals themselves, particularly during flood events. In specific areas where sediment/nutrient pollution risk has been identified infrastructure improvements such as new bridges, armoured river crossing points and water feeders should be employed. To complement these actions, RAFS could be used for the treatment of distributed point sources such as field drains, small ditches and areas of concentrated overland flow.

8.3 Mitigation of DWPA using RAFs

The mitigation experiments in this study have all been monitored at the local scale using inflow/outflow measurements of discharge and contaminant concentration during storm events. This was considered the most accurate method in the absence of in-situ, continuous water quality monitoring equipment; without which it would be very difficult to detect changes in sediment/nutrient concentrations at the larger catchment, or even the farm scale. Measuring the impact of DWPA mitigation interventions at the catchment scale is the ultimate goal under the WFD; however, any change in contaminant signal (due to intervention) at this scale is likely to be confounded by a multitude of natural variables and, in the case of NO_3 particularly, system recovery is likely to take decades.

The evidence from this study demonstrates that relatively small RAFs, principally sediment traps, constructed in farm ditches (<1 km^2 catchment area) can reduce mean SS, TP, SRP and NO_3 loads during storm events by 30-43%, 23-30%, 12-19% and 8-14%, respectively. The preeminent process of pollutant reduction is the settlement of SS and associated nutrients (mainly PP), which is affected by the feature size to catchment area ratio (i.e., residence time) and individual design criteria, (e.g., the number of wetland cells). Results suggest that retention of finer sediment than would be predicted by particle settlement velocities occurred and is likely attributed to the aggregation of soil particles and/or the flocculation of waterborne sediment. Importantly, this finding suggests that even relatively small features can result in a significant reduction in catchment diffuse pollutant load.

Although the ability of RAFs to reduce losses of SS, P and NO_3 in agricultural runoff has been shown to be promising at the local scale, the significant challenge of providing such evidence at the catchment scale still remains and there exists an insufficiency of data. It is hoped that government-backed research programmes such as the Defra DTC and the IACP can help tackle this by using state-of-the-art monitoring equipment that can provide continuous water quality determinand data. One of the goals of projects such as these is to develop and determine surrogates for pollution such as turbidity, which would be more economically viable for widespread use. However, a question mark exits as to whether these projects can survive for sufficient time to detect any change in sediment/nutrient signal at the catchment outlet as the result of the 'multiplier effect' of a number of farm-scale interventions. This unknown makes the local scale monitoring of RAFs (and other mitigation options) viable as they can be recommended as 'no regrets' measures that will not have adverse impacts on the environment without fully understanding their collective impact in the future, this is particularly important in light of the urgent requirements of the WFD.

8.4 RAF maintenance

Remobilisation of fine sediment and associated nutrients has been identified as a problem associated with in-channel features. It is highly important that RAFs do not become a net source of sediment/nutrients and thus require management to ensure their long-term effectiveness. Sediment traps installed in agricultural ditches will require periodic emptying to reduce the risk of future release of sediment and P. Due to nutrient enrichment, the trapped fine-sediment has fertilizing properties that make it worthwhile spreading back to land. As trapped volumes of sediment are relatively small, it may be appropriate to stockpile material, perhaps mixing it in with FYM. However, it is argued that this would be more time-efficient than cleaning out significant lengths of ditch, as traps function to localise sedimentation. Harvesting emergent vegetation would also reduce the risk for redox-induced release of soluble P. Other RAF management duties could potentially include coppicing willow barriers, removal and renewal of brash screens, ensuring outlet pipes are clear and repairing damaged structures such as earth bunds. The fulfilment of these tasks is most important following a large storm event.

Rules concerning the maintenance of RAFs are still under consideration; questions such as 'who should remove the material', 'at what rate' and 'at what cost' are important to their uptake. It would be pertinent for farmers themselves to take ownership of features and integrate upkeep with regular farm activities. However, this will require a compensation or subsidy mechanism to be established. Low maintenance RAFs that require the least attention from farmers will be the most successful in terms of uptake. One of the key components of durability is appropriate design and this study has provided a wealth of experience.

8.5 Lessons learned

It is strongly recommended that in-channel interventions should be located relatively close to source (<2 km^2 catchment area) so as not to be overwhelmed by large discharge volumes. The flooding of surrounding farmland, nearby roads, etc. is not acceptable. Building RAFs of sufficient capacity to cope with the most extreme discharges is not feasible due to limited space, increasingly complicated construction methods and inherent costs. The installation of a network of smaller RAFs can achieve a desired storage volume without the need for a large feature such as that installed at Netherton.

Spillways or overflow channels are a necessary feature to ensure that bunds, etc. are not overtopped by uncontrolled overflow, which could lead to damage or failure. In the case of the Netherton three-cell RAF, it became apparent that water was being spilled too early from the final cell as levels rose. One solution would be to increase the height of the spillway, but perhaps the best solution would be the installation of a larger diameter discharge pipe to increase controlled discharge. For future applications it is highly recommended that initially a large diameter pipe size be installed as it easier to reduce size at a later date by using a collar, than to retrofit a larger pipe, which would be disruptive and costly.

Where large RAFs are desirable, such as treatment wetlands or flood storage ponds, it is recommended that sediment trap features (including willow/brash dams) be installed directly upstream. This will increase the lifespan of larger features (large features are extremely difficult to dredge). A network of relatively small RAFs are less intrusive in the landscape, simpler and cheaper to construct, easier to maintain by the farmer, and pose less risk in the event of failure.

The use of wood chippings as a fine-filter media is not recommended for future use in farm ditch applications. Besides being relatively expensive, they are time consuming to install due to the need of a retaining structure, and also require regular renewal. The use of wood chippings as bioreactors in sub-surface systems appears to be promising, but this method should be reserved for situations where the financial investment can be justified (i.e., where good ecological status is being compromised by high levels of NO_3). In-channel willow dams and brash screens, however, were deemed a success in that they were observed to attenuate high flows within the ditch and overtime the brash became bound up with sediment. The willow was locally sourced at no cost and installation was very simple, although some periodic management will be required to ensure the ditch does not get blocked. There is the potential to combine the willow barriers with sediment traps to help reduce flow velocities within the feature and reduce the incidence of system 'short circuiting'.

8.6 A hypothetical catchment management plan using RAFs

This study has demonstrated how high intensity agriculture practices can yield significant losses of SS, P and NO_3, and that storm events drive pollutant transfers via a number of hydrological pathways. It has also identified where and how best to target these pathways using RAFs, which have shown the potential to significantly reduce sediment and nutrient losses at the local scale.

To incorporate all of the experiences gained during this study a hypothetical catchment management plan using RAFs is proposed (Figure 8.2). This highlights the key catchment scales at which different RAFs should be implemented and how a suite of RAFs can be used to manage a plethora of flow pathways.

Key RAF design and implementation criteria are suggested:

- Overland flow interception bunds should be located in field margins across concentrated runoff pathways (<1 km^2). Bunds should be 1 m high (max.) and have an outlet pipe installed to allow post-storm drainage and prevent storage capacity being reached at low runoff discharges, which would limit effectiveness and damage crops.
- Ditch management RAFs (e.g., ditch widening, willow dams/brash, sediment traps – see Figure 8.1) should be targeted slightly downstream of where sub-surface drains, overland flow pathways, etc. emerge but before the main channel (<2 km^2). All flow should be contained within the ditch or directed through a spillway during high-flows.
    - Lengths of drainage ditch should be widened (1.5 m max.) and given a flat base to reduce flow velocities and promote sedimentation.
    - Sediment traps should be constructed in the widened ditch sections by excavating deep 'cells'. Traps to be a maximum of 1 m deep, which includes ≥50% freeboard to accommodate high flows.
    - Earth and/or rock bunds can be used as check dams to create within-ditch pools; or they can be combined with sediment traps to create more storage capacity. Appropriately sized outlet pipes should be installed above base level to reduce remobilisation of previously settled sediment.
    - An armoured spillway or high-flow channel should be constructed on the top/around the side of any earth/rock bunds to allow the release of excess

runoff (within the confines of the ditch) so that the bund structure is not eroded and compromised.

  - Woven willow screens and accompanying brash filters can function as check-dams to reduce flow velocities and promote sedimentation. They can also be used in conjunction with sediment traps, either upstream of them, or within them to create separate cells.

- Larger flood storage RAFs should be situated adjacent to the main channel (<5 km^2) and connected using a swale, or grassed waterway, that only operates during high-flows. Sediment trap(s) should be installed directly upstream. Outflow from the flood pond is via a pipe and a spillway is essential.
- Wetland RAFs can be in-channel (<3 km^2) but adjacent to channel is preferable (<5 km^2). Features should be designed to receive runoff at all times but high-flows should be by-passed around the feature. Sediment trap(s) should be installed directly upstream.
- Riparian buffer strips and stock fencing should be used along all main channels and indeed all channels where livestock are farmed.

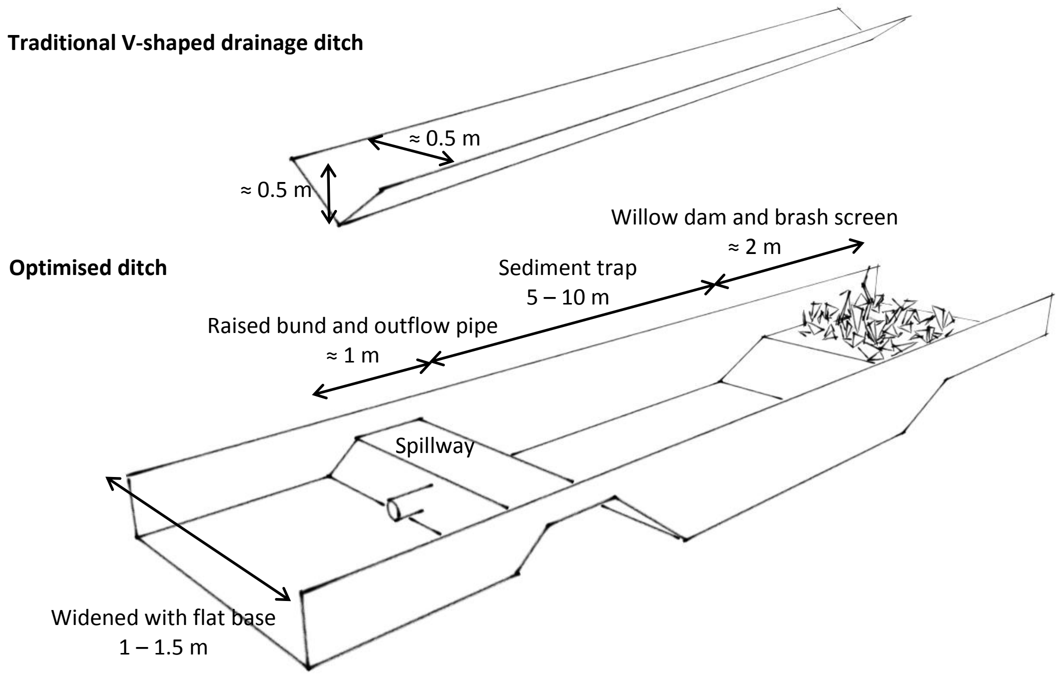


Figure 8.1: Optimised ditch schematic (not to scale).

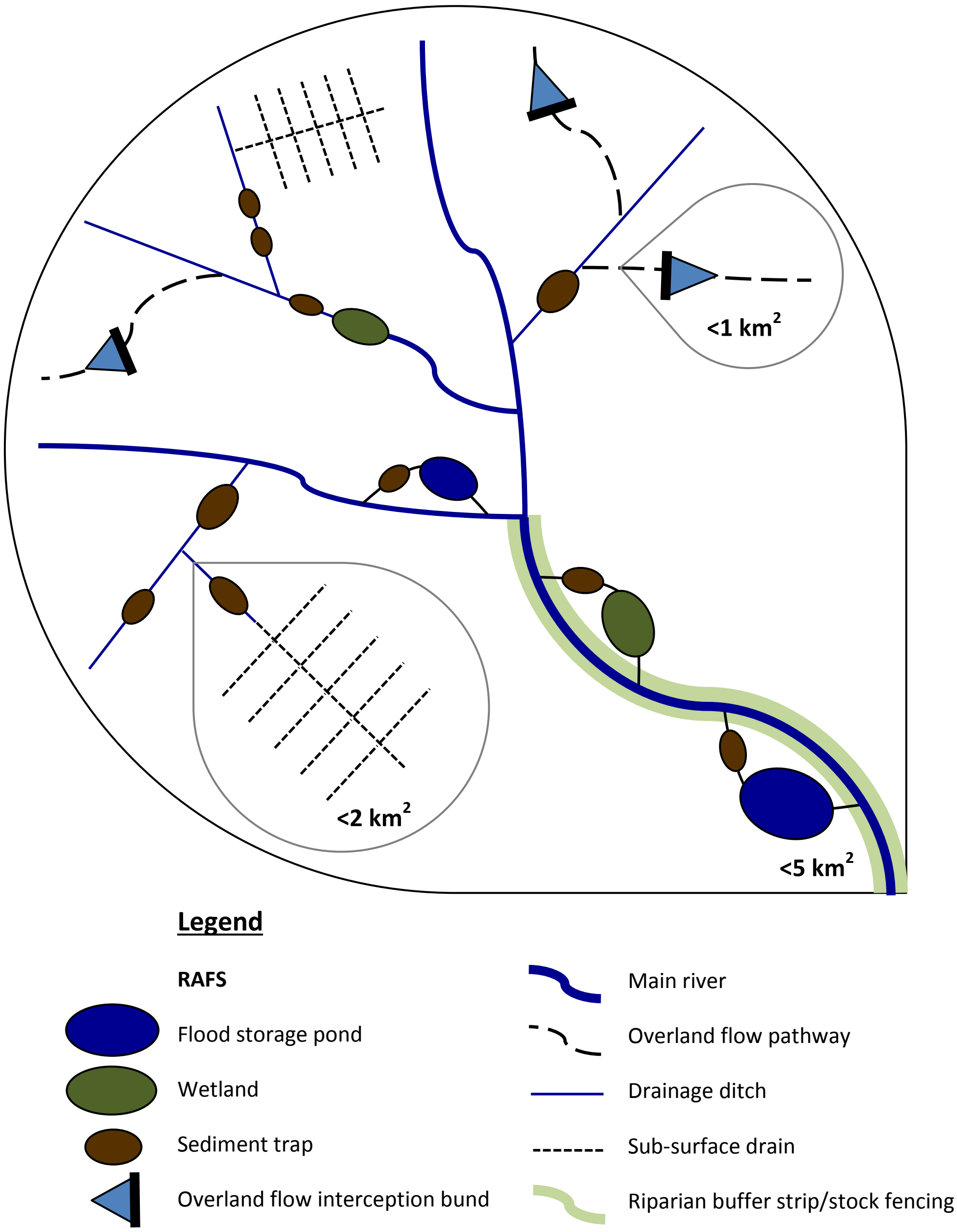


Figure 8.2: Schematic of a hypothetical catchment management plan using RAFs (not to scale).

9. Conclusions and recommendations

Conclusions

This chapter will put forward the conclusions of the thesis in relation to the aims and objectives, as stated in Chapter 1. Finally, recommendations for future research are provided.

The principal aims of the thesis were to *characterise the sediment and nutrient transport regime of the upper Eden catchment to inform the targeting of DWPA mitigation efforts*, and to *investigate the efficacy of a number of RAFs to reduce concentrations/loads of SS, P and NO_3 in agricultural runoff*. The conclusions arising from each of the research objectives are stated below.

1.1 *Use an appropriate grab sampling methodology to quantify SS, P and NO_3 concentrations at a range of catchment areas covering three orders of magnitude (micro ≈ 1 Km^2, mini ≈ 10 Km^2 and meso-scale ≈ 100 Km^2) in the upper River Eden catchment, Cumbria. Select an appropriate means to calculate annual SS, P and NO_3 loads and specific yields*

Suspended sediment, P and NO_3 concentrations were measured at thirteen sub-catchments at an unusually high spatial resolution for a catchment of its size. Stratified sampling meant that efforts were made to collect water samples at high discharges (during storm events) in order to reduce the bias associated with fixed-period grab sampling campaigns, which are synonymous with being unrepresentative of actual concentration fluxes. By collecting samples within a few hours of each other a representative ‘snapshot’ of sediment/nutrient concentrations is taken while the catchment is under the same hydrological conditions. An extrapolation method (using rating curves) was employed to estimate continuous (15 minute) SS, P and NO_3 concentrations, which were subsequently used to calculate annual contaminant loads/yields. It was deemed inappropriate to use an interpolation calculation method due to relatively low sampling frequency and the flashy nature (low-medium BFI) of the upper Eden catchment.

1.2 *Investigate how determinand concentrations/loads vary with spatial scale, as well as with changes in various controlling processes, such as precipitation/runoff and land cover/use, inter alia.*

Suspended sediment, P and NO_3 loads/yields were calculated for all sub-catchments but no relationship was found between yield (of all determinands) and catchment area. Due to the relative lack of samples collected at high discharges it is believed that loads/yields are likely to be underestimates of the true exports. However, the difference between the results is seen to be more important than their level of uncertainty. As samples were collected within a short timeframe (on the same sampling day) it can be assumed that the uncertainty is within the same range for all the sites. The data suggest that greater variation in yield exists between smaller sub-catchments and reach more similar values as the catchments scale increases, reflecting an averaging effect at larger scales.

A strong positive relationship exists between total precipitation and export of SS, P and NO_3. The two study years – 2010 and 2011 were dryer and wetter than average, respectively, and calculated loads of all determinands were significantly greater in 2011 compared with 2010. The largest increases were for SS and TP. Higher concentrations of SS and P were recorded at higher discharges, which equate to increasing loads. Although NO_3 export is clearly driven by discharge events, loads are smaller relative to those of SS and P due to the weak correlation between discharge and NO_3 concentration. Nitrate load increases simply due to the greater volume of water, probably tied to the leachate and/or deeper runoff pathways.

The greater variability in exports is associated with the non-nested sub-catchments due to increased heterogeneity of land use. Strong positive correlations between percentage improved agriculture and yield in 2011 suggest that land use has a strong influence over yield in wetter conditions, which is attributed to increasing mobilisation and transfer of contaminants. The Low Hall catchment opposes the general trend as despite consisting of 100% improved agricultural land it only exhibits a high NO_3 yield, while other determinands are relatively low.

Seasonal trends are exhibited by SS and P with greater concentrations recorded in the autumn, which is linked to increased precipitation and the first-flushing of contaminants from agricultural land or the remobilisation of fine sediment deposited in the channel during summer low flows. Nitrate concentration exhibits little seasonal variation, as its movement is associated with more continuous base flow.

There is evidence of dilution in the upper Eden catchment as lowland sub-catchments with increased intensity agriculture such as Blind Beck and Helm Beck, which yield relatively high SS and P average concentrations and yields, appear to have no negative impact on the main river Eden. Clean water from upland sub-catchments such as Scandal Beck and the River Belah, is sufficient to dilute potentially polluting lowland areas. However, this system is vulnerable to low rainfall, particularly if climate change results in dryer summers and warmer temperatures.

1.3 *Compare calculated sediment/nutrient yields with export coefficients from the literature. Determine the representativeness of the data collected in the upper Eden catchment and evaluate the selected methodology for water quality monitoring at the catchment scale.*

Calculated exports for the upper Eden are relatively low in comparison with other studies but fall within the range of export coefficients reported in the literature. A simple export coefficient model was used to predict annual SS, TP and TN yields for the thirteen upper Eden sub-catchments based on a land cover type and the areal extent of that land cover. The model results were acceptable with the majority of yield estimates falling between 2010 (dry year) and 2011 (wet year) calculated exports. The model appears to perform best for the larger catchments that have a more heterogeneous land cover (e.g., predictions for the Gt. Musgrave and Appleby sub-catchments were in close agreement with measured exports).

A hydrological component was added to the model to take account of seasonal variation in discharge and that derived from baseflow, which allowed monthly estimates of sediment/nutrient yield to be made. The model over-predicted SS and TP in dry periods and under-predicted in wet ones, with the effect being more prominent in the winter months. The extra baseflow component served to significantly increase predicted TN export above measured values.

It is understood that annual contaminant load/yield estimates based on grab samples only are likely to be underestimates of true values. This is more prominent for SS and sediment-phase nutrients due to the bias towards low and residual flow sampling. The employment of an auto-sampler at the Blind Beck outlet meant that water samples were taken across a much wider range of discharges and produced discharge-contaminant rating curves that are more representative of actual conditions.

Accurate load estimations (and the high costs associated with measuring them) are important for geomorphological purposes, but here it is questioned whether they are necessary for the identification of potential DWPA mitigation locations. It is proposed that the use of an appropriately designed grab sample regime along with automatic water sampling equipment is sufficient to characterise the runoff, sediment and nutrient regime and identify the dominant pollutant source-pathways to an extent that suitable DWPA mitigation options can be selected.

1.4 Select a sub-catchment within the Upper Eden catchment. Use a spatially intensive sampling campaign to identify pollutant sources within the catchment, and employ automatic storm sampling equipment to determine the importance of storm events on contaminant transfer.

Blind Beck was identified as consistently exhibiting the highest nutrient and sediment concentrations per unit runoff as well as the highest total yields per unit area than any other monitored sub-catchment in the upper Eden. The collected SS, P and NO_3 concentration data located the principal source of SS and P as the area around Sykeside Farm. Nitrate concentrations increased gradually along the river corridor but did not change significantly until the confluence with the Low Hall stream, which consistently yielded the highest NO_3 concentrations recorded in the upper Eden catchment during this study. Sources of SS and P within the Sykeside Farm area were identified as the farmyard itself, as it is in very close proximity to the stream; increased stocking densities of both cattle and sheep; increased incidence of stream bank erosion/collapse and poaching. Bank degradation is linked to the lack of stock fencing and riparian buffer zone along the length of Blind Beck. It is also associated with the nature of bank-forming material (fine-textured, poorly drained boulder clay), which despite being relatively cohesive is more susceptible to loss of stabilising vegetation and more prone to bank collapse if higher/steeper banks are present.

Water samples collected during high-flow storm events revealed that the majority of SS and TP (principally PP) was exported during short time periods, almost exclusively associated with discharge peaks. Nitrate concentrations initially fell due to dilution or stayed fairly constant, but increased during event recessions as slower flow pathways reached the stream (and rapid pathways receded). Analyses of storm data revealed that discharge and SS concentration rose in response to rainfall, and P increased in line with sediment concentration; and SS and P concentrations peaked rapidly either with or just before peak discharge.

Contaminant-discharge rating curves were updated to include storm samples (along with grab samples) for the Blind Beck outlet. Annual loads/yields were derived and compared with those calculated from grab samples only. Suspended sediment yield was increased by circa. 110%, TP increased by circa. 117%, SRP by circa. 80%, and there was no discernible difference for NO_3. Calculated SS and P yields were in agreement with values from the literature for catchments in the north of England and southern Scotland. However, according to the Natural England sediment loss thresholds (Cooper *et al*. 2008), Blind Beck exceeded the critical yield of 70 t km^2 yr^{-1} in 2011.

2.1 *Describe the design and construct of a number of RAFs in agricultural catchments. Using field measurements, evaluate the efficacy of RAFs to reduce SS, P and NO_3 concentrations/loads in runoff during storm events – measure sedimentation volumes and calculate annual sediment/nutrient removal rates where possible.*

A wetland RAF was constructed at Sykeside Farm in the Blind Beck sub-catchment and monitored as part of this study. A number of storm events were captured using automatic sampling equipment but the ability of the wetland to retain sediment/nutrients could not be examined, as there was no clearly identifiable inflow point to the feature. There is need of a draw-off swale to convey a proportion of flow from Blind Beck to the wetland, but this was not constructed due to lack of funding. Lessons learned from the catchment-wide monitoring in Blind Beck and from observing the wetland RAF showed that a befitting choice of mitigation option(s) should rely on the nature of the DWPA problem. In this instance where near-channel sources appear to be dominant the use of RAFs is not ideal. The most effective and cost-efficient way of solving the majority of the issues would be to introduce stock fencing. It was also learned that RAFs should have smaller contributing areas than the wetland at Sykeside, to ensure that runoff quantities are more easily manageable.

A number of RAFs were also examined in the Belford catchment, Northumberland. First, two RAFs designed to reduce flood risk in the town of Belford were examined to establish whether they also had the capacity to mitigate DWPA. An on-line pond RAF was visually accumulating sediment but inlet/out samples collected during storm events showed that little-to-no SS, P or NO_3 retention was occurring at higher discharges. This was attributed to reduced residence time in the feature and the remobilisation of previously deposited material. An overland flow interception bund constructed in a field margin was shown to be effective at trapping sediment (and associated nutrients) during storm events that generated surface runoff. However, the feature only functions during large storms and the presence of artificial sub-surface drains in the catchment meant that significant losses of sediment/nutrients were occurring during moderate-sized events (that don't result in overland flow).

Based on the above realisations, there was a need to design and construct a RAF in the Belford catchment that functioned in all hydrological conditions, treated runoff from sub-surface drains, and reduced the amount of remobilisation occurring during high-flow events. The ideal location was in the surface ditch drainage network as it was downstream of field drain discharge points but upstream of the on-line pond RAF. The ditch was dredged prior to the construction of a multi-stage RAF. The feature consisted of an upstream sediment trap with a rock dam and riser pipe, a series of woven willow leaky-dams along with brash filter screens, and a downstream fine-filter consisting of wood chippings. The RAF was monitored during storm events using auto-samplers; simultaneous inlet/outlet samples were collected along with discharges were possible.

The multi-stage ditch RAF at Belford consistently reduced concentrations and loads of sediment and nutrients at the event scale. Overall, SS, TP, SRP and NO_3 concentrations were reduced by 26%, 20%, 19% and 9%, respectively, while respective loads were reduced by 30%, 23%, 12% and 8%. Greatest SS and P retention was recorded on the rising limb, which was attributed to the settlement of relatively coarse sediment before residence time in the feature was reduced by peak discharge. The sediment trap component yielded a trapping rate of circa. 0.1 t ha^{-1} yr^{-1}, which is promising considering the relatively small size of the trap. Willow dams and brash screens can be recommended due to their low cost and ability to reduce flow velocities and promote sedimentation. The use of wood chippings as a fine filter cannot be recommended for future use due to relatively high installation costs and the need to renew the filter media on a semi-regualr basis.

The Netherton Project, commissioned by Cheviot Futures, involved the implementation of a number of flood-related RAFs in conjunction with DWPA mitigation features. This study was responsible for the design and construction of a series of sediment traps, situated in agricultural ditches upstream of flood storage ponds; a three-cell tiered feature was subjected to monitoring. The aim of the sediment traps was to reduce SS and P losses to the Netherton Burn (with NO_3 reductions being an added bonus), but also to slow the sedimentation rate of the flood storage ponds to maximise their lifespan.

The three-cell sediment trap at Netherton reduced SS, TP, SRP and NO_3 loads by 43%, 30%, 19% and 14%, respectively. Sediment was trapped at a rate of circa. 0.31 t ha^{-1} yr^{-1} with greatest retention recorded in the upstream cell. The bulk density of trapped material was greatest in the upstream cell and lowest in the downstream one; the P concentration of the sediment was greatest in the middle cell. The division of the sediment trap at Netherton into three separate terraced cells improved RAF efficiency as it reduces 'short-circuiting' of the feature during high-flows. The ability of sediment traps to capture fine sediment of particle sizes smaller than what would be expected due to their slow settlement velocities is explained by the aggregation of soil particles and/or the flocculation of water-borne sediment particles.

2.2 *Review the use of RAFs on the larger catchment scale, taking account of lessons learned on sediment/nutrient source pathways, RAF suitability and appropriate spatial scale for implementation. Consider the potential multiple-benefits of RAFs and recommend design criteria for future implementation.*

Based on the outcomes of the Eden studies, including the more intensive monitoring carried out in the Blind Beck sub-catchment, it is suggested that pollutant *source* and *mobilisation* mitigation options would be more approriate than *transport* management ones, such as RAFs. This is because the high-level of catchment connectivity in the Belford and Netherton catchment is not present in the Eden and the ‘distributed point sources’ such as concentrated overland flow and discharges from field drains are much less frequent. Based on the findings of this study the most befitting management approach for Blind Beck would include stock fencing and riparian zone protection. These would help prevent animals from entering the watercourses thus reducing river bank degradation and poaching, and reduce the incidence of bank collapse.

The use of RAFs to mitigate DWPA is best suited to relatively small catchments with concentrated (polluted) runoff pathways and relatively low rainfall. The export of diffuse pollution from the Lady’s Well sub-catchment (and most likely in the wider Belford catchment) appears to take two forms: ‘chronic’ export, which takes place during residual flow conditions where drainflow and shallow sub-surface flow are the dominant flow pathways; and ‘acute’ export, which occurs in larger storms and where overland flow is the major conduit. As this catchment shows high levels of connectivity, as well as a sensitive downstream receptor (Budle Bay) the use of RAFs is a viable option. The approach has also been shown to be suitable for the runoff/DWPA regime at Netherton.

The multi-functionality of RAFs has not been explicitly tested in this thesis but their ability to provide secondary benefits is clear. These include flood attenuation, increased biodiversity, potential carbon sequestration, added amenity value, *inter alia*.

Throughout the course of this research a number of important lessons have been learned concerning the use of RAFs as DWPA mitigation options. Firstly it is crucial to determine the nature of the sediment/nutrient problem in a catchment before deciding whether the implementation of the approach is appropriate. Understanding of the catchment’s hydrological regime, including the dominant flow pathways, is necessary to be able to effectively target RAFs.

Experience from the Belford and Netherton catchments has shown that agricultural drainage ditches can be optimised to significantly reduce losses of SS and P and also positively impact on the export of NO_3. Not only will this have a water quality benefit the collateral impact on the surrounding farm will be minimal.

Suitable spatial scales for different types of RAFs have been provided; for example, overland flow should be intercepted close to source and ditch RAFs should be located downstream of field drain discharge points, but upstream of the main channel. Where required, larger RAFs such as flood storage ponds and treatment wetlands should be accompanied by upstream sediment traps. A number of important RAF design criteria have been reported and maintenance schedules highlighted.

The number of RAFs required to ‘treat’ a catchment depends on many variables, including the level of sediment/nutrient pollution, land use, soil type, precipitation regime, presence of sensitive receptor, and many more. And while it is difficult to scale-up the findings from the mitigation experiments to the larger catchment scale due to the heterogeneity and complex nature of controlling processes, by proving the ability of RAF to significantly reduce losses of sediment and nutrients at the local scale they can be recommended as ‘no-regrets’ options to be used in conjunction with other DWPA mitigation approaches.

Recommendation for future study

Additional sampling at the thirteen upper Eden sub-catchments would improve the contaminant concentration-discharge rating curves and thus reduce the errors associated with the sediment/nutrient yield estimates. The use of auto-samples would allow high discharges to be sampled that otherwise prove very difficult to measure using grab sampling alone. This would allow stronger conclusions to be drawn about the spatial variability in DWPA yield and the relationships between yield and catchment characteristics. Improved data could also be used to create concentration-duration-frequency curves.

Further work to investigate the finding that the majority of fine sediment is derived from riparian sources in Blind Beck would be of value. A small-scale detailed study could determine bank erosion rates and quantify their relative contribution. At the larger scale sediment fingerprinting techniques could be employed.

More RAFs of all type should be tested at the local scale to add to the evidence base. It would be highly beneficial to carry out a larger-scale investigation, possibly at the farm scale (e.g., <5 km^2) where a large number of features could be installed. The use of intermittent grab sampling and storm sampling is valid but will be limited to a relatively small number of individual features.

To monitor impacts at a larger spatial scale the use of in-situ turbidity monitoring equipment is recommended. Once a correlation between SS and turbidity (and P and turbidity) has been established the need for manual sampling is reduced. It is also important to establish the cost-effectiveness of RAFs in order to evaluate them against other DWPA mitigation options.

The development of a simple, robust method to measure the accumulation of sediment in a sediment trap would be valuable, as would a means of removing the trapped sediment that would cause minimal disturbance to enable more representative analysis of dry bulk density. It would also be beneficial to measure the particle-size distribution of the sediment to better determine the settlement time and the sorption capacity of the SS. Finer particles tend to have a higher sorption capacity due to a large surface area to volume ratio and surface charges. Ultimately, this would determine the capacity of the SS loads to act as a vector of contaminants.

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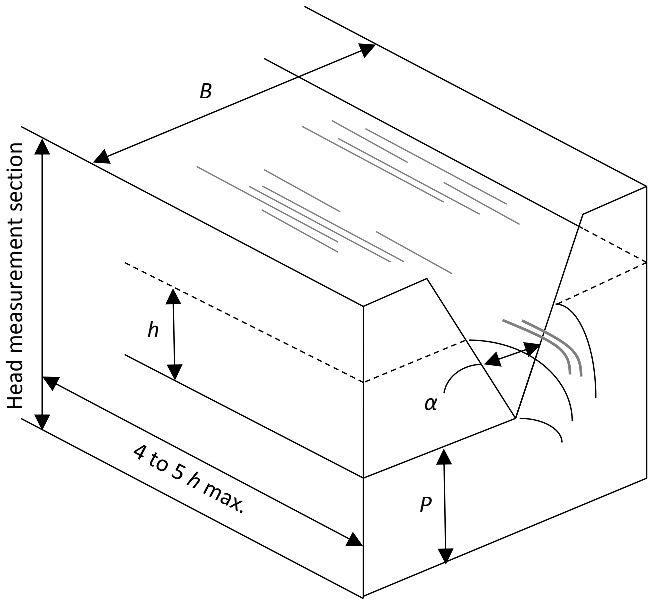
Appendix

Appendix A - Upper Eden catchment stage-discharge rating coefficients.

Site	*h* min	*h* max	*a*	*b*	Note
Appleby			21.98	2.103	
River Belah	0.00	0.75	48.25	2.395	Estimated from FDC*
	0.75	1.40	34.75	1.022	
Blind Beck			2.135	2.296	
Gais Gill			6.244	2.321	
Helm Beck	0.00	0.58	6.429	3.881	Estimated from FDC*
	0.58	1.40	7.767	3.491	
Low Hall stream**	0.00	0.25	0.375	1.327	
	0.25	0.70	0.137	0.027	
Ravenstonedale	0.00	0.60	4.297	1.717	Estimated from FDC*
	0.60	1.40	11.45	3.511	
Smardale Beck			41.86	2.557	
Swindale Beck	0.00	0.37	13.73	3.059	Estimated from FDC*
	0.37	1.40	11.80	2.486	

*Established by Mills (2009)
**Established by Ockenden (2010)

Appendix B – V-notch weir specifications.



Schematic representation of a v-notch weir (from BSI (1981)) not to scale.

Discharge Q (m^3/s) is calculated from stage h (m) according to BS368: Part 4A: 1981 using the Kindsvater-Shen formula:

$$Q = C_e \frac{8}{15} tan\frac{\alpha}{2}\sqrt{2g_n}h^{5/2}$$

Where: C_e is the coefficient of discharge (non-dimensional), g_n is the acceleration due to gravity (m/s^2), α is the notch angle (degrees) and *h* is the head measurement (m).

The limits of application of the Kindsvater-Shen formula for V-notch weirs are:

1. The ratio h/p should be equal to or less than 1.2.
2. The ratio h/B should be equal to or less than 0.4.
3. The head over the vertex of the notch should not be less than 0.05 m nor more than 0.60 m.
4. The height of the vertex of the notch above the bed of the approach channel should not be less than 0.10 m.
5. The width of the rectangular approach channel should exceed 0.60 m.
6. The notch angle of a fully contracted weir may range between 25 and 100 degrees.
7. The tailwater level should remain below the vertex of the notch.

B1. Sykeside wetland RAF V-notch weir specification.

Dimension	Value
Width	>1 m
Height	0.83 m
Nappe of V to top	0.6 m
Plate thickness	3 mm
P	0.23 m
α	¼ 90°
C_e	0.587

B2. Belford RAF V-notch weir specification.

Dimension	Value
Width	0.7 m
Height	0.5 m
Nappe of V to top	0.3 m
Plate thickness	3 mm
P	0.2 m
α	½ 90°
C_e	0.578

B3. Netherton RAF V-notch weir specification.

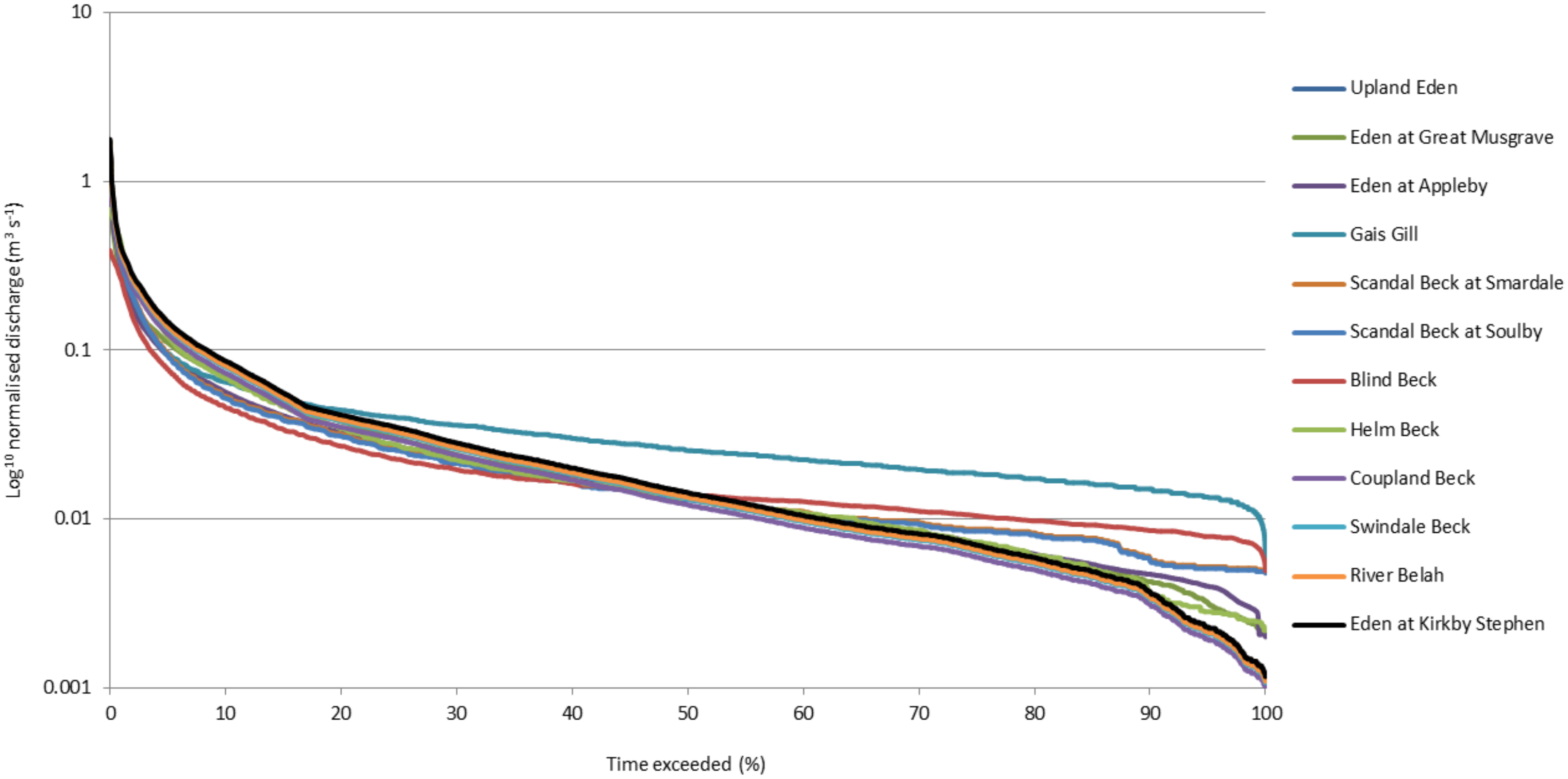
Dimension	Value
Width	0.7 m
Height	0.55m
Nappe of V to top	0.35 m
Plate thickness	3 mm
P	0.2 m
α	90°
C_e	0.579

Appendix C – Upper Eden catchment FEH catchment descriptors

Descriptor	Explanation (unit of measurement)	Upland Eden	Eden at Kirkby Stephen	Eden at Great Musgrave	Eden at Appleby	Gais Gill	Scandal Beck at Smardale	Scandal Beck at Soulby	Blind Beck	Helm Beck	Coupland Beck	Swindale Beck	River Belah	Low Hall stream
OS GRIDREF	Ordinance Survey grid reference	NY 77200 07350	NY 77300 09700	NY 76500 13100	NY 68100 20450	NY 71800 01150	NY 73900 09100	NY 75050 10950	NY 75250 13050	NY 70950 14900	NY 70550 18350	NY 77550 13550	NY 79550 12150	NY 75250 13000
AREA	(km^2)	48	69	223	334	1.1	37	40	9	18	27.5	31.7	53	0.9
ALTBAR	mean catchment altitude (m AOD)	413	395	351	319	478	336	322	214	252	371	393	385	153
ASPBAR	dominant aspect (north = 0)	330	309	304	286	76	339	345	52	17	236	186	298	353
ASPVAR	aspect variability (closer to 1 = one particular direction)	0.1	0.2	0.15	0.12	0.44	0.22	0.23	0.43	0.25	0.41	0.43	0.26	0.43
BFIHOST	Base flow index calculated from HOST classification	0.375	0.409	0.443	0.467	0.357	0.501	0.527	0.561	0.527	0.465	0.354	0.348	0.474
DPLBAR	Characterises catchment size and configuration. Mean of distances between each node on IHDTM grid and the catchment outlet (km)	9.5	10.34	13.65	24.31	1.07	8.35	10.42	4.48	5.93	7.13	7.86	9.22	1.2
DPSBAR	mean of distances between nodes and catchment outlet - characterises steepness (m/km)	158.1	149	117.7	113.4	210.9	118.6	113.5	71.2	94.6	144.5	110.2	114.7	27.2
LDP	longest drainage path (km)	20.19	23.33	29.82	45.64	2.06	15.04	18.08	8.65	12.23	12.93	14.62	15.36	2.32
PROPWET	proportion of time SMD was <= 6mm during 1961-1990	0.7	0.68	0.66	0.67	0.71	0.71	0.71	0.71	0.71	0.65	0.64	0.63	0.71
RMED-1H	mean annual maximum 1 hour rainfall (mm)	12.1	11.7	11	10.9	12.1	11.7	11.5	10.7	11	10.6	10.4	10.5	10.4
RMED-1D	mean annual maximum 1 day rainfall (mm)	54.9	51	45.1	43	54.4	50.4	49.3	40.3	42	40.2	41.2	40.2	37.9
RMED-2D	mean annual maximum 2 day rainfall (mm)	76.8	71.2	62.5	59.4	78	72.2	70.6	56.9	59.8	53.8	54.2	54.9	52.2
SAAR	standard period (1961-1990) average annual rainfall (mm)	1610	1492	1270	1188	1906	1515	1456	1018	1159	1169	1132	1116	854
SAAR4170	standard period (1941-1970) average annual rainfall (mm)	1450	1387	1318	1252	1742	1405	1367	1046	1178	1362	1374	1346	883
SPRHOST	standard percentage runoff derived from HOST	47.6	45.76	42.36	39.85	49.04	37.68	36.66	34.91	29.25	38.74	47.21	46.17	40.38
URBEXT1990	Extent of urbanised area (%)	0.0001	0.0032	0.0018	0.0023	0	0.0007	0.0007	0.0017	0	0.0007	0.0032	0.0002	0

Appendix D – Upper Eden catchment flow analyses

D1. Normalised flow duration curves (2010-2011).



D2. Annual catchment water balances

2010

Site	2010				
	P (mm)	Q (mm)	E (mm)	P-E (mm)	% diff between Q and P-E
Upland Eden	1293	897	184	1109	19.1
Eden at Kirkby Stephen	1070	862	184	886	2.7
Eden at Great Musgrave	913	714	184	729	2.1
Eden at Appleby	856	653	184	672	2.8
Gais Gill	1013	942	184	829	-13.6
Scandal Beck at Smardale	985	714	184	801	10.9
Scandal Beck at Soulby	985	695	184	801	13.2
Blind Beck	779	576	184	595	3.2
Helm Beck	785	532	184	601	11.5
Coupland Beck	858	646	184	674	4.2
Swindale Beck	1014	889	184	830	-7.1
River Belah	1016	810	184	832	2.6
Low Hall stream	719	460	184	535	14.0

2011

Site	2011				
	P (mm)	Q (mm)	E (mm)	P-E (mm)	% diff between Q and P-E
Upland Eden	2178	1668	280	1898	12.1
Eden at Kirkby Stephen	1913	1604	280	1633	1.8
Eden at Great Musgrave	1404	1222	280	1124	-8.7
Eden at Appleby	1336	1164	280	1056	-10.2
Gais Gill	1740	1649	280	1460	-12.9
Scandal Beck at Smardale	1670	1252	280	1390	9.9
Scandal Beck at Soulby	1670	1218	280	1390	12.4
Blind Beck	1429	1035	280	1149	9.9
Helm Beck	1450	983	280	1170	16.0
Coupland Beck	1431	1202	280	1151	-4.4
Swindale Beck	1788	1475	280	1508	2.2
River Belah	1548	1491	280	1268	-17.6
Low Hall stream	1131	902	280	851	-6.0

Appendix E – Water quality data

E1. Grab sample data tables: date, contaminant concentration and discharge.

See CD-ROM

E2. Determinand concentration-discharge rating coefficients.

Suspended sediment

Site	10a	b	R^2	n
Upland Eden	0.527	0.530	0.56	39
Eden at Kirkby Stephen	0.525	0.603	0.74	49
Eden at Great Musgrave	0.396	0.657	0.73	49
Eden at Appleby	0.331	0.660	0.61	49
Gais Gill	1.090	0.511	0.51	40
Scandal Beck at Smardale	0.444	0.702	0.75	48
Scandal Beck at Soulby	0.520	0.572	0.64	48
Blind Beck	1.516	1.117	0.86	49
Helm Beck	0.884	0.575	0.58	37
Coupland Beck	0.580	0.505	0.64	38
Swindale Beck	0.774	0.506	0.56	42
River Belah	0.472	0.882	0.66	46
Low Hall stream	1.935	0.852	0.69	49

Total phosphorus

Site	10a	b	R^2	n
Upland Eden	-1.635	0.238	0.60	39
Eden at Kirkby Stephen	-1.577	0.229	0.63	49
Eden at Great Musgrave	-1.622	0.290	0.70	49
Eden at Appleby	-1.686	0.327	0.69	49
Gais Gill	-1.529	0.182	0.30	40
Scandal Beck at Smardale	-1.690	0.303	0.50	48
Scandal Beck at Soulby	-1.629	0.264	0.57	48
Blind Beck	-0.881	0.665	0.81	49
Helm Beck	-1.431	0.219	0.43	37
Coupland Beck	-1.627	0.199	0.50	38
Swindale Beck	-1.364	0.223	0.52	42
River Belah	-1.666	0.303	0.33	46
Low Hall stream	-0.752	0.469	0.57	49

Soluble reactive phosphorus

Site	10a	b	R^2	n
Upland Eden	-2.0458	0.273	0.40	39
Eden at Kirkby Stephen	-1.9872	0.176	0.38	49
Eden at Great Musgrave	-1.9957	0.199	0.30	49
Eden at Appleby	-2.0969	0.303	0.36	49
Gais Gill	-1.9547	0.180	0.17	40
Scandal Beck at Smardale	-2.1024	0.217	0.22	43
Scandal Beck at Soulby	-2.0969	0.227	0.28	43
Blind Beck	-1.466	0.303	0.35	49
Helm Beck	-1.8729	0.203	0.20	37
Coupland Beck	-2.0655	0.213	0.27	33
Swindale Beck	-1.8447	0.093	0.11	42
River Belah	-2.1487	0.239	0.13	46
Low Hall stream	-1.1669	0.401	0.35	49

Nitrate

Site	10a	b	R^2	n
Upland Eden	0.168	-0.078	0.05	36
Eden at Kirkby Stephen	0.400	-0.013	0.00	43
Eden at Great Musgrave	0.567	-0.101	0.21	43
Eden at Appleby	0.677	-0.258	0.24	43
Gais Gill	0.120	-0.158	0.01	37
Scandal Beck at Smardale	0.446	-0.063	0.05	40
Scandal Beck at Soulby	0.448	-0.060	0.06	40
Blind Beck	0.810	-0.266	0.46	43
Helm Beck	0.572	0.011	0.00	34
Coupland Beck	0.289	-0.125	0.11	36
Swindale Beck	0.453	-0.063	0.08	38
River Belah	0.456	-0.145	0.13	42
Low Hall stream	0.663	-0.291	0.46	43

E3. Export coefficients from the literature

Suspended sediment

Author	Country	Land use	SS (t km^{-2} yr^{-1})	
			Average	Range
Sharpley & Smith (1990)	USA	Native grass	25.3	3.1 - 79.1
Foster & Lees (1999)	UK	Pasture		7.7 – 17.7
Wass & Leeks	UK	Upland/unimproved	17.1	
Sharpley & Smith (1990)	USA	Wheat	382.3	28.7 - 964
Foster & Lees (1999)	UK	Arable		16.5 – 24.6
Russell *et al*. (2001)	UK	Arable		77 - 122
Wass & Leeks (1999)	UK	Mixed		12.6 – 33.5
Foster & Lees (1999)	UK	Mixed	52	
Sharpley & Smith (1990)	USA	Mixed crop and grass	179.2	59.6 - 401.5
Foster and Lees (1999)	UK	Moorland		23.5 – 34.6

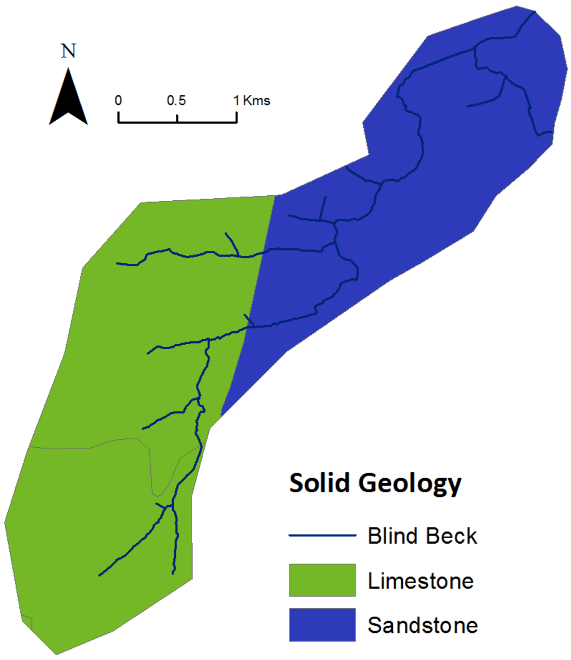
Total phosphorus

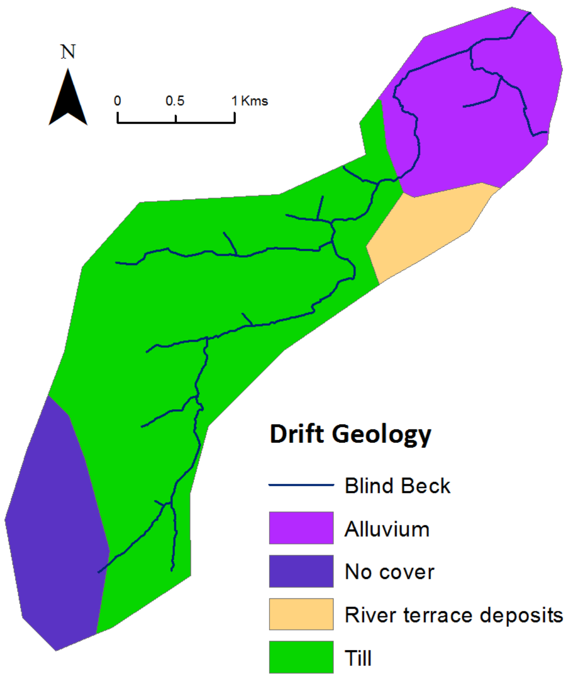
Author	Country	Land use	TP (kg km^{-2} yr^{-1})	
			Average	Range
Reckhow et al (1980)	USA	Pasture (imp grass)	150	14 - 490
Loehr et al (1989)	USA & Europe	Pasture (imp grass)		5 - 60
Marsden et al (1995)	UK	Pasture (imp grass)		40 - 100
Smith et al (1995)	Ireland	Improved grassland	100	
Johnes et al (1994)	UK	Pasture (imp grass)		10 - 80
McMuckin et al (1996)	NI	Improved grassland	80	60 - 100
Sharpley & Smith (1990)	USA	Native grass	31.4	3.1 - 95.1
Cooke (1976)	UK	Grassland	20	
Kolenbrander (1972)		Grassland		20 - 30
Haygarth and Jarvis (1996)	UK	Grassland	300	
McMuckin et al (1996)	NI	Grassland	80	60 - 100
Sharpley & Smith (1990)	USA	Wheat	237.4	84.4 - 436.5
McMuckin et al (1996)	NI	Arable	497	383 - 611
Marsden et al (1995)	UK	Arable		80 - 250
Catt et al (1998)	UK	Arable		37 - 264
Loehr et al (1989)	USA & Europe	Rural cropland		6 - 290
Clesceri et al (1986)	USA (Wisconsin)	Agriculture	26.2	
Dodd et al (1992)	USA	Agriculture	99	
Reckhow et al (1980)	USA	Mixed agriculture	113	8 - 325
Rast & Lee (1978)	USA	Rural/agriculture	50	
Sharpley & Smith (1990)	USA	Mixed crop and grass	131.5	64 - 209

Total nitrogen

Author	Country	Land use	TN (kg km^{-2} yr^{-1})	
			Average	Range
Reckhow et al (1980)	USA	Pasture (imp grass)	865	148 - 3085
Loehr et al (1989)	USA & Europe	Pasture (imp grass)		320 - 1400
Reckhow et al (1980)	USA	Row crops	1609	210 - 7960
Loehr et al (1989)	USA & Europe	Rural cropland		210 - 7960
McFarland & Hauck (2001)	USA (Texas)	Forage fields	540	
Clesceri et al (1986)	USA (Wisconsin)	Agriculture	669	
Dodd et al (1992)	USA	Agriculture	980	
Reckhow et al (1980)	USA	Mixed agriculture	1653	282 - 4150
Rast & Lee (1978)	USA	Rural/agriculture	500	

Appendix F – Blind Beck geology and drift geology maps





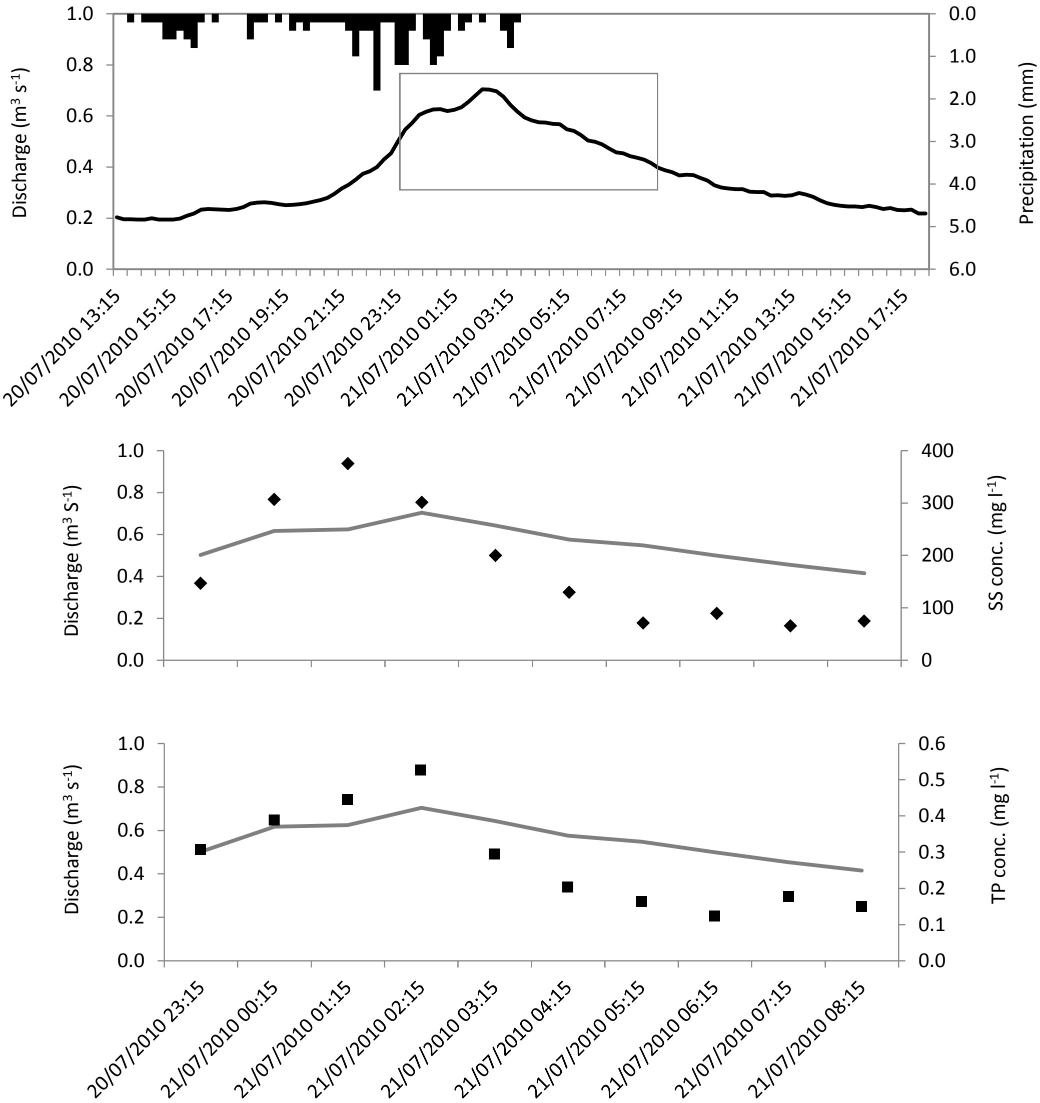
Appendix G – Blind Beck water quality

G1. Grab sample data: date, contaminant concentration and discharge.

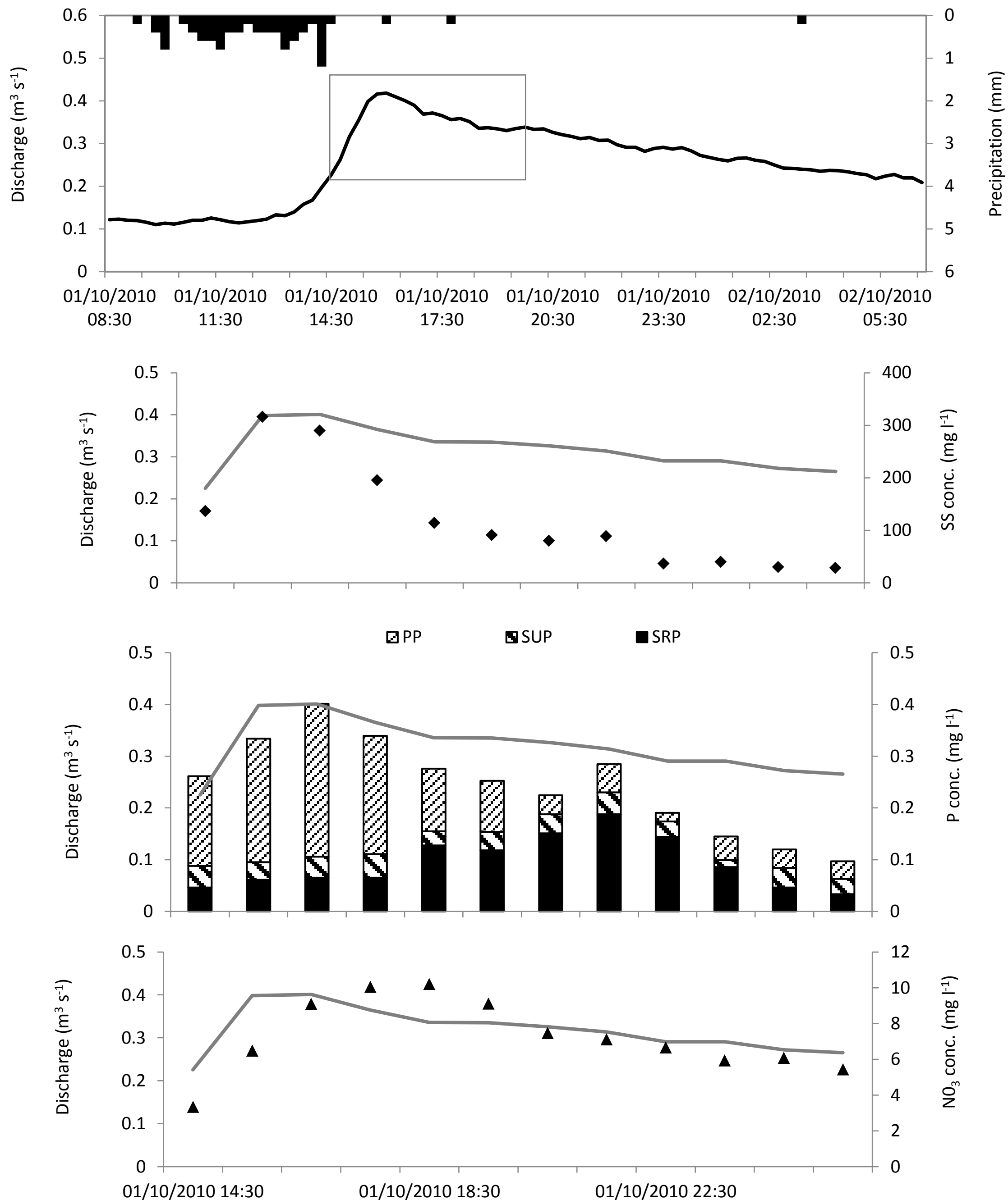
See CD-ROM.

G2. Storm event graphs.

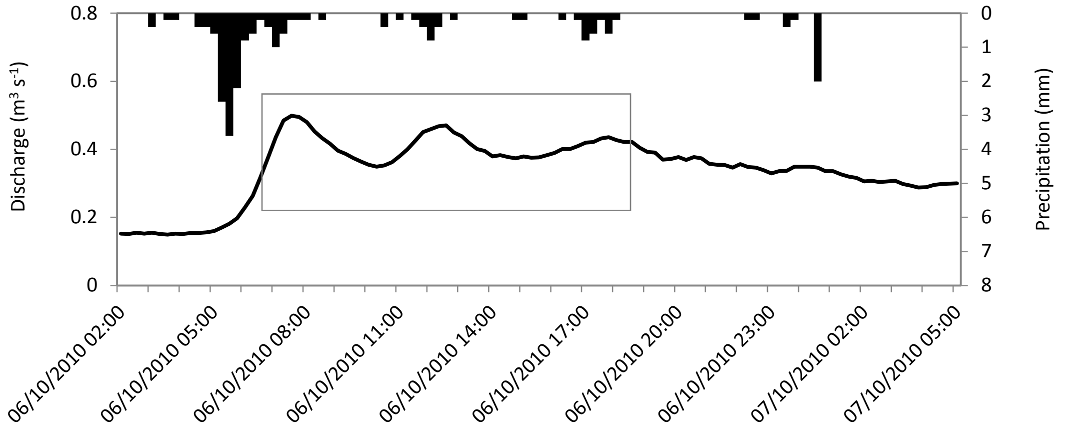
Event A – 20/07/2010

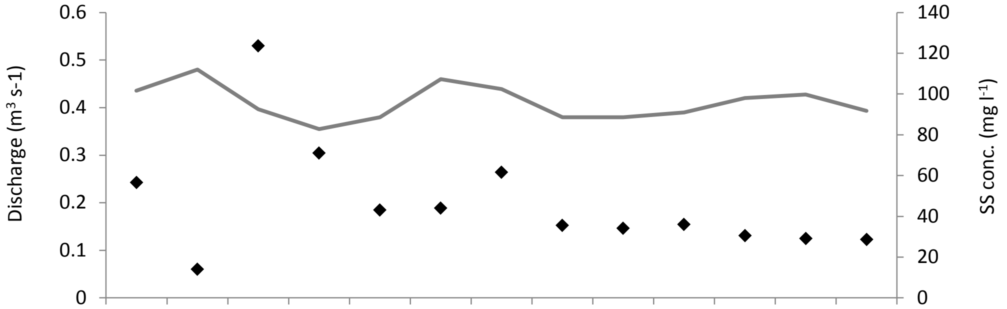


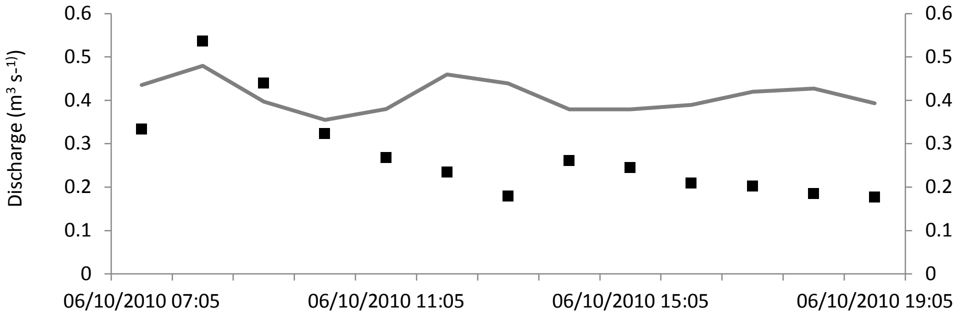
Event B – 01/10/2010



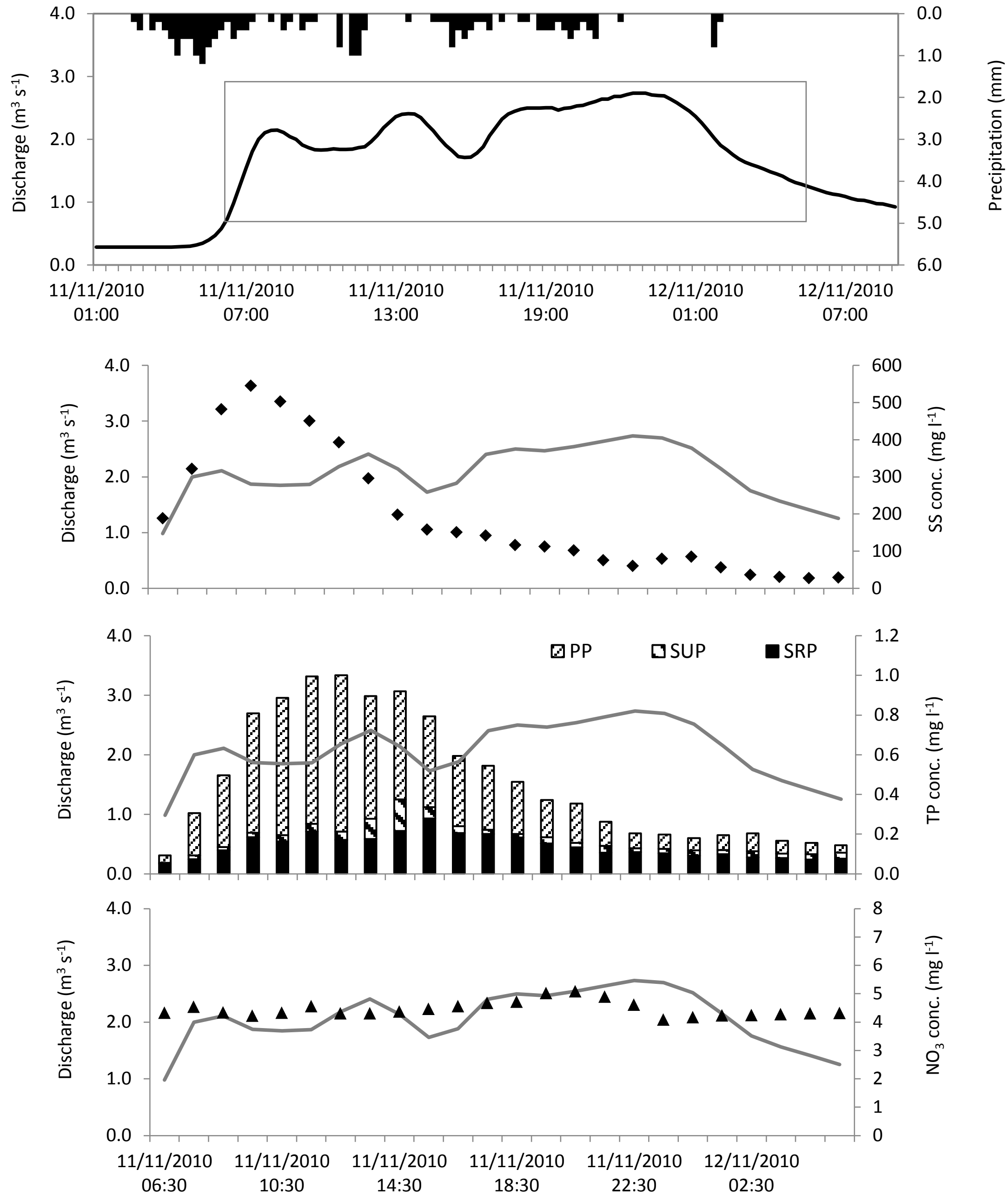
Event C – 06/10/2010



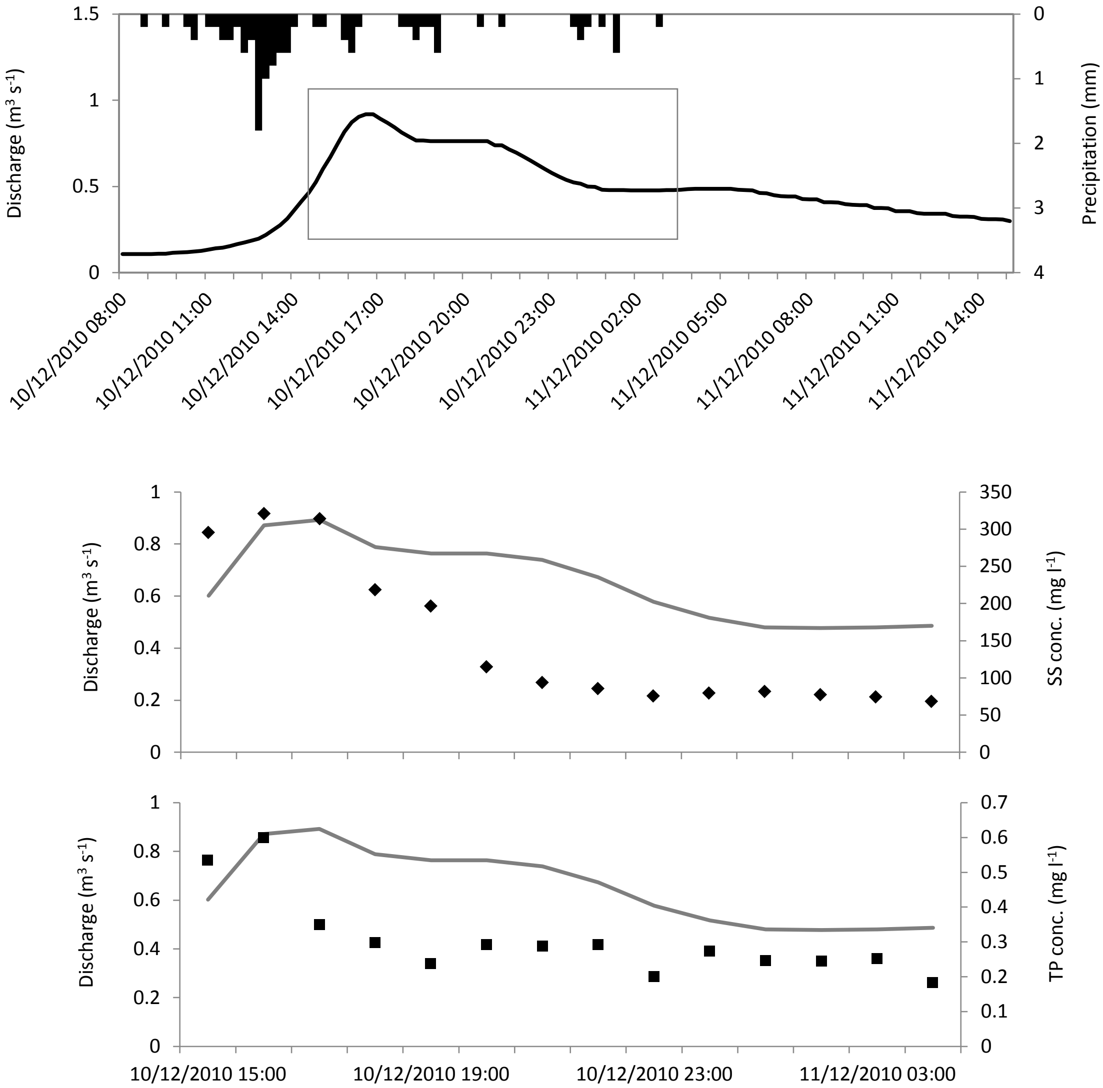




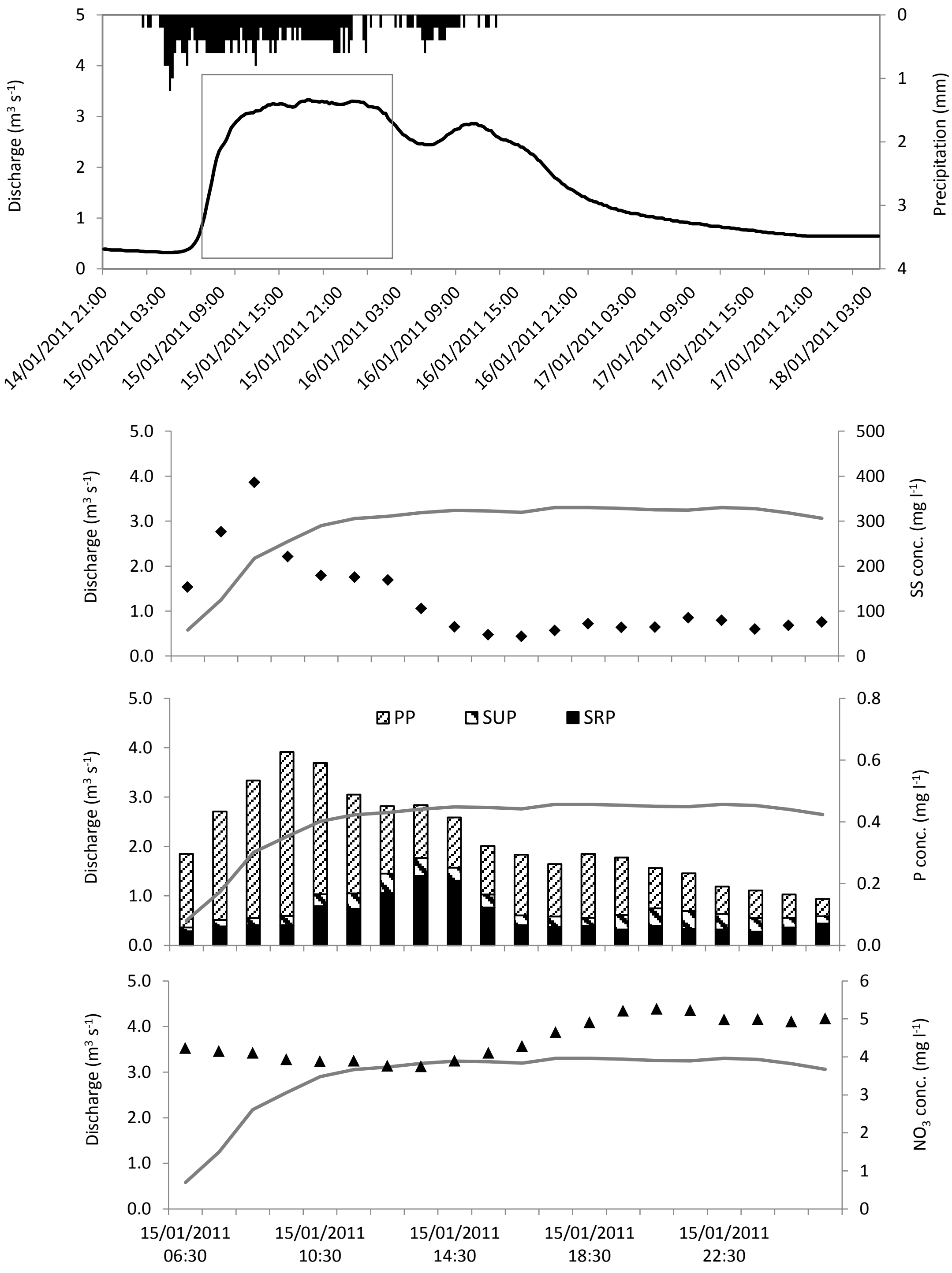
Event D – 11/11/2010



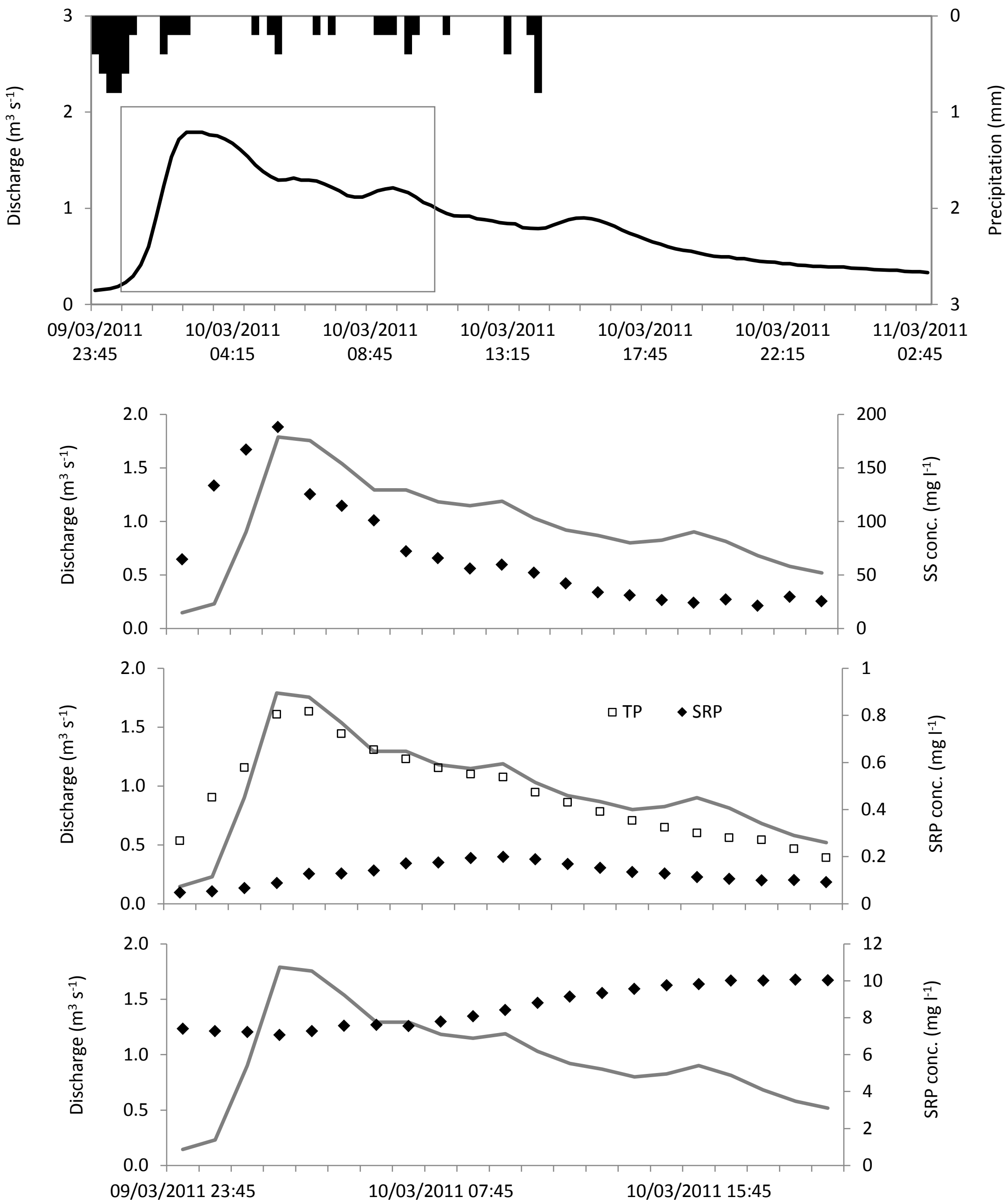
Event E – 10/12/2010



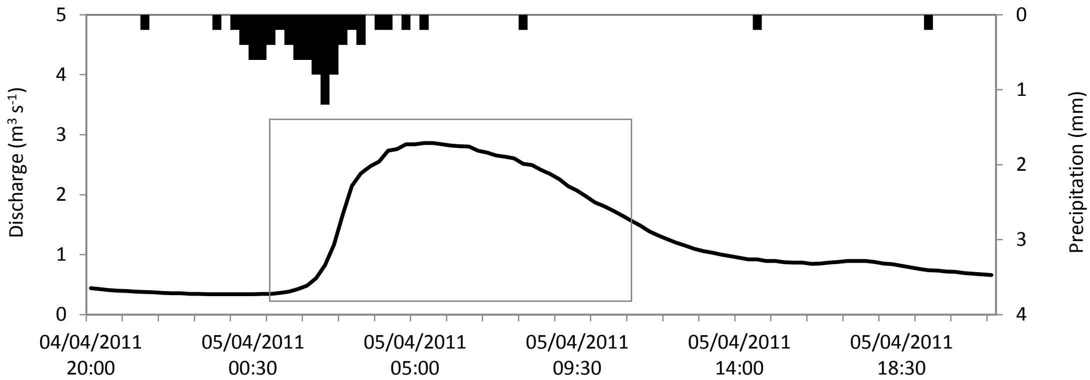
Event F – 15/01/2011

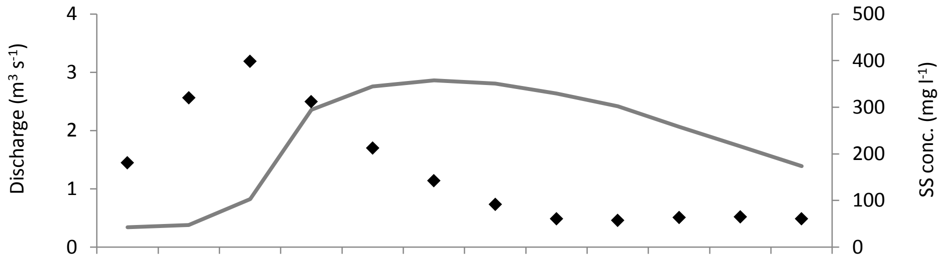


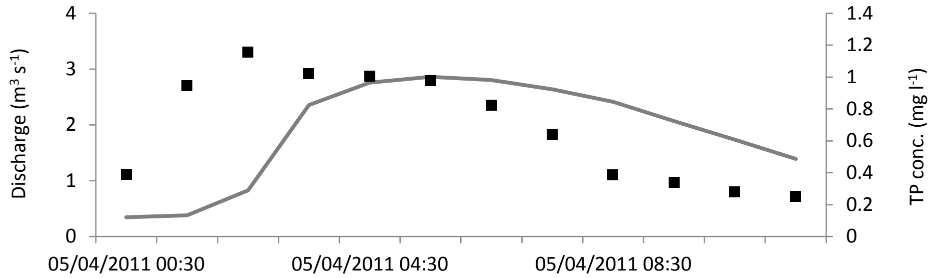
Event G – 09/03/2011



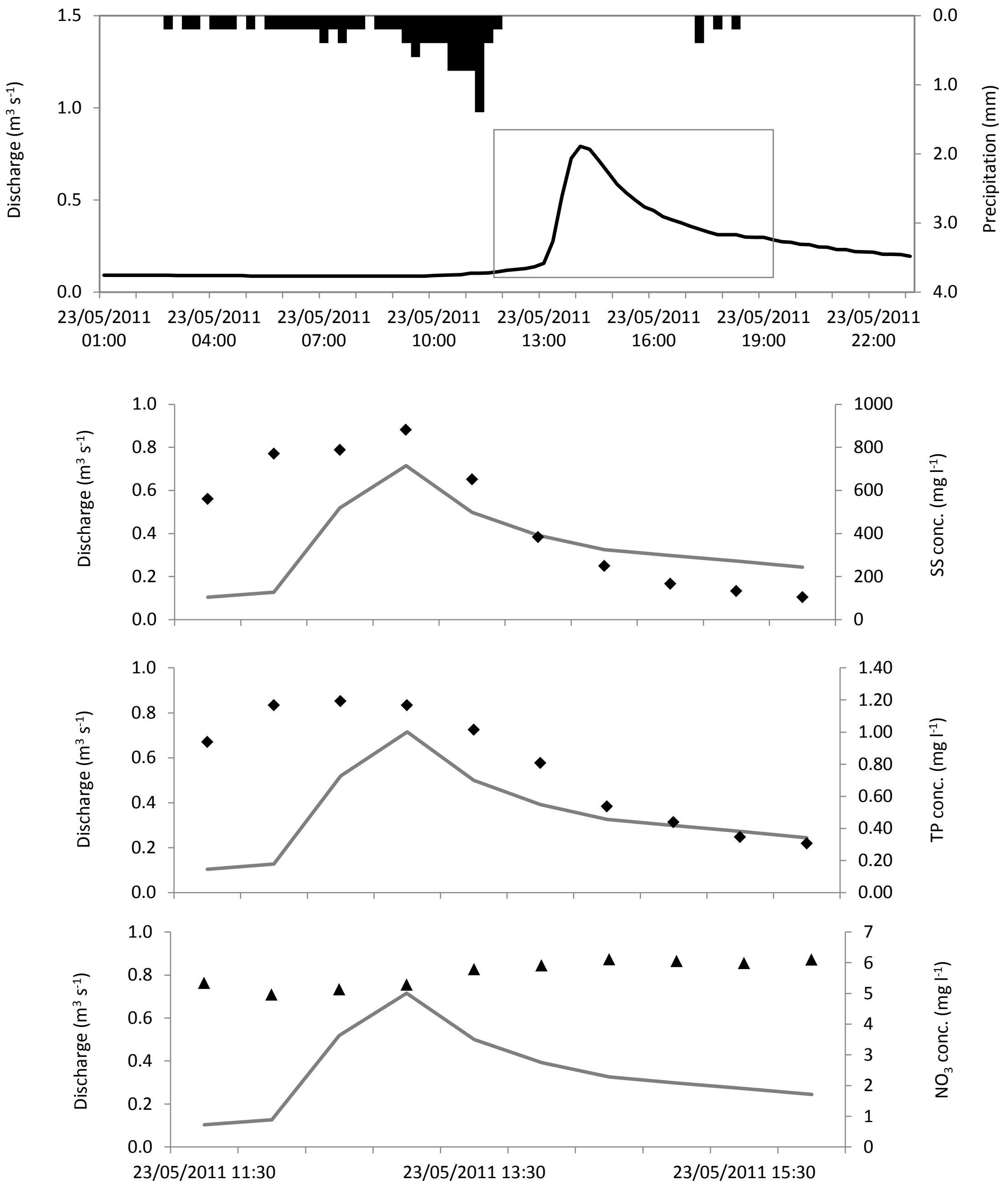
Event H – 05/04/2011



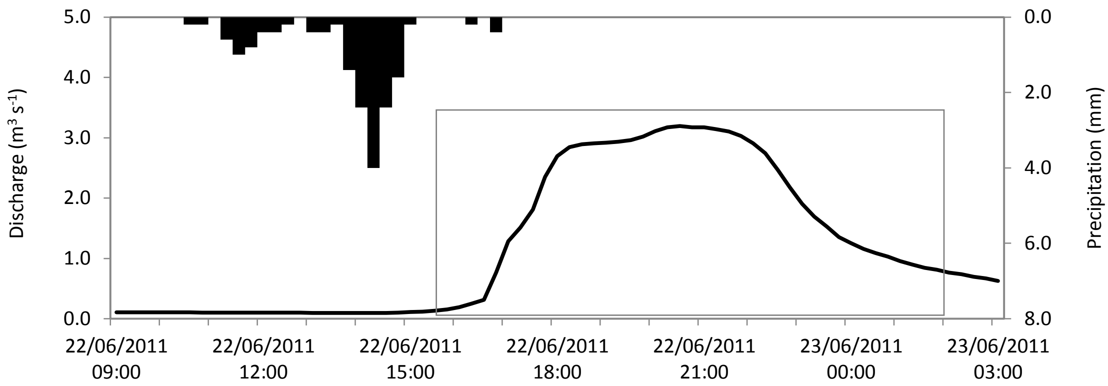


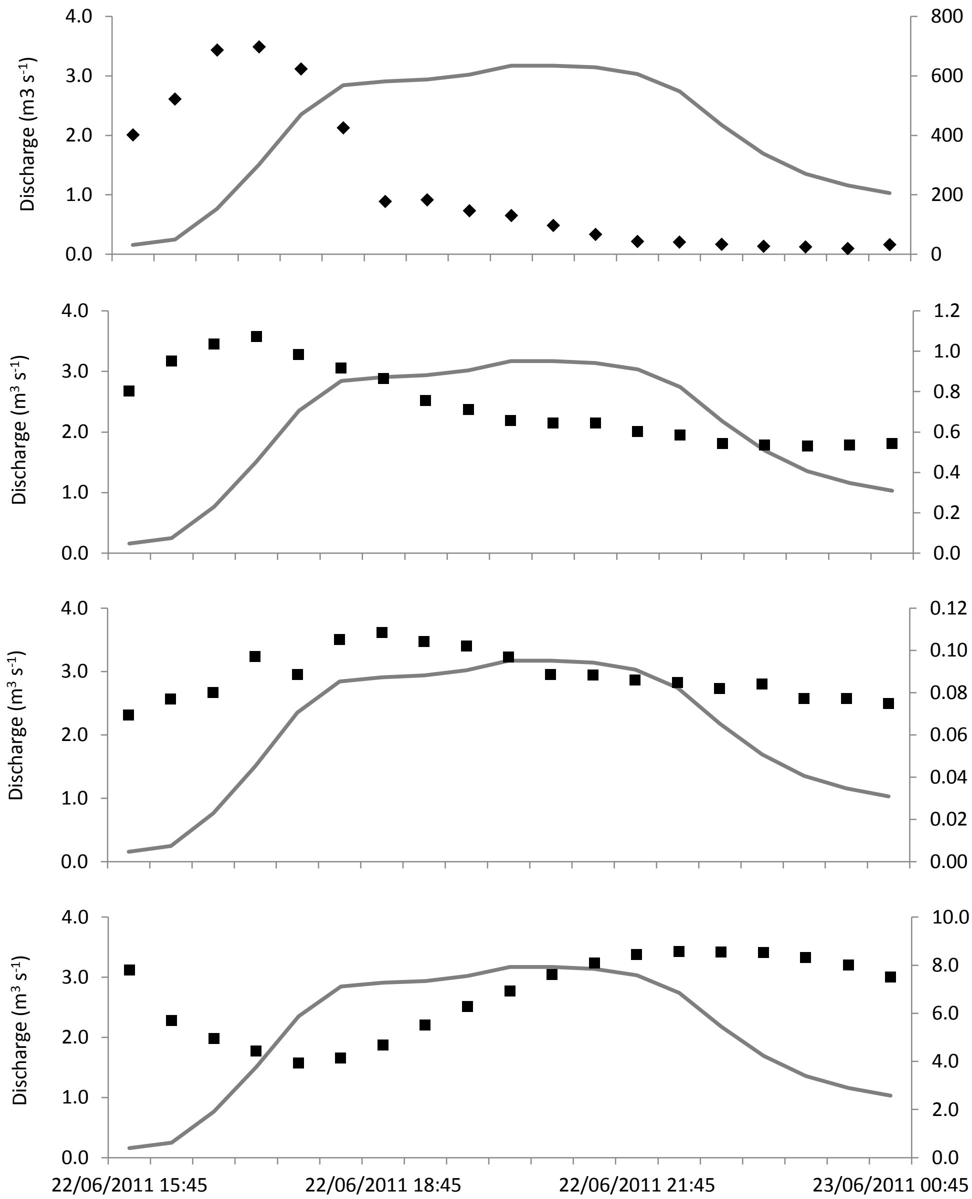


Event I – 23/05/2011

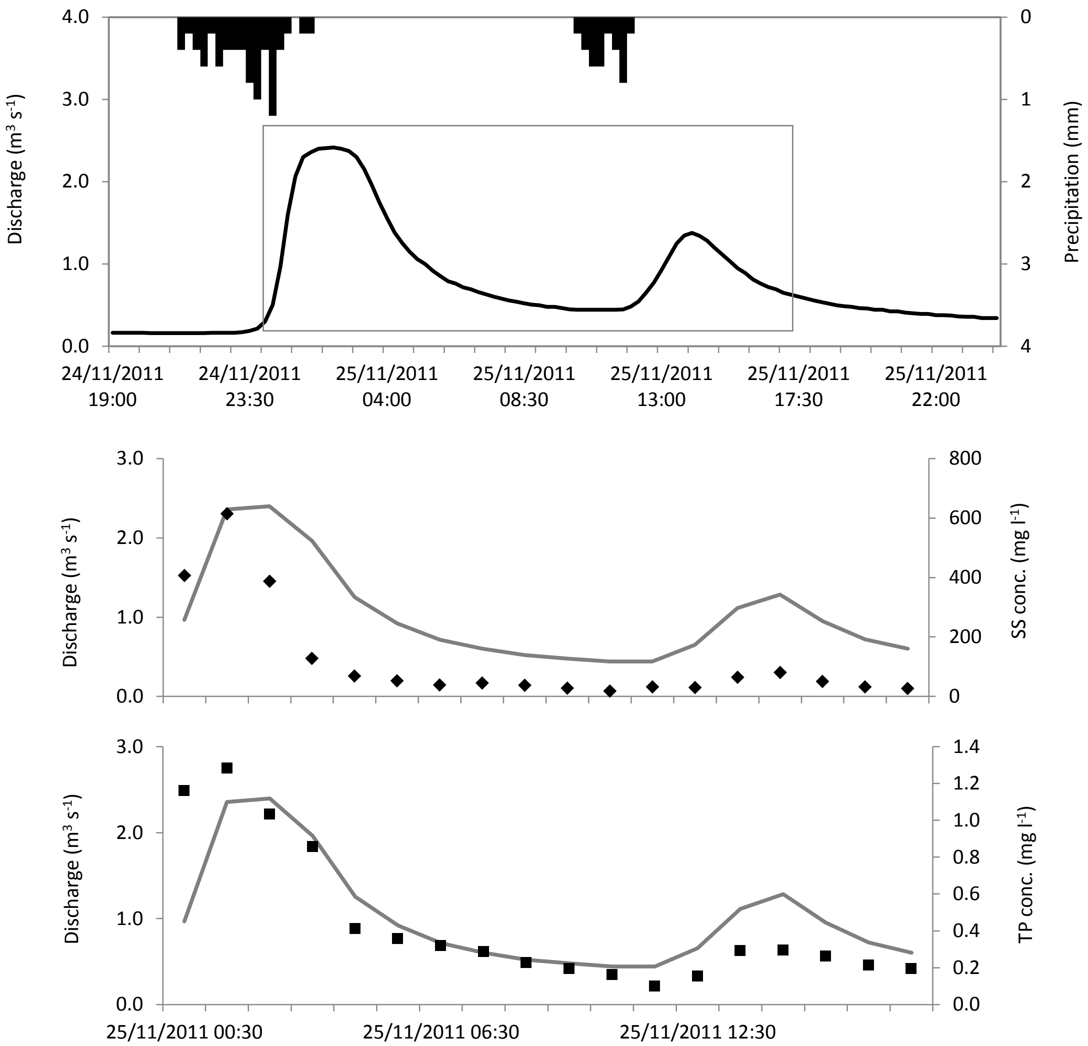


Event J – 22/06/2011





Event K – 25/11/2011

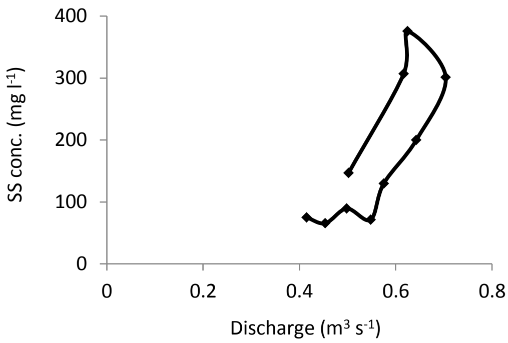


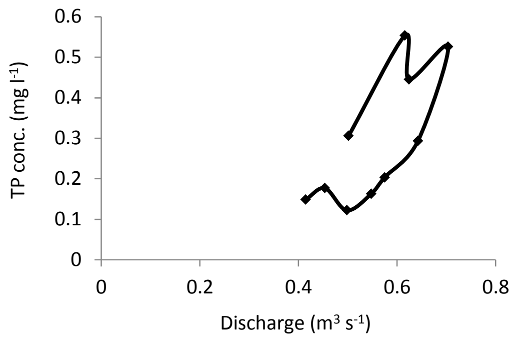
G3. Storm event raw contaminant concentration and discharge data.

See CD-ROM.

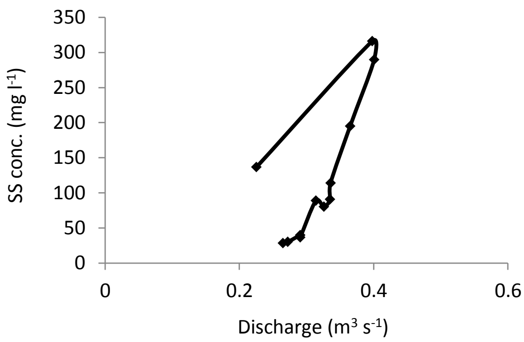
G4. Hysteresis graphs.

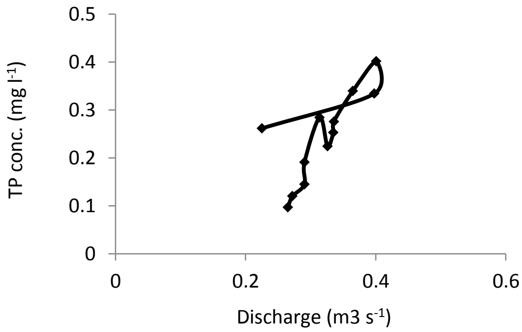
Event A – 20/07/2010

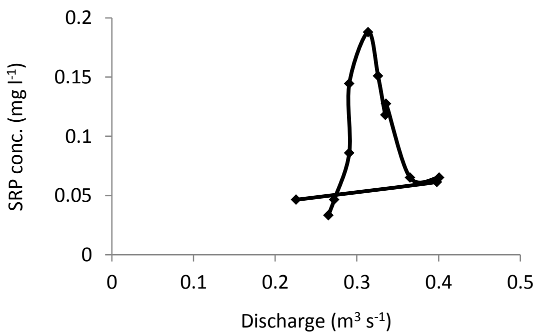


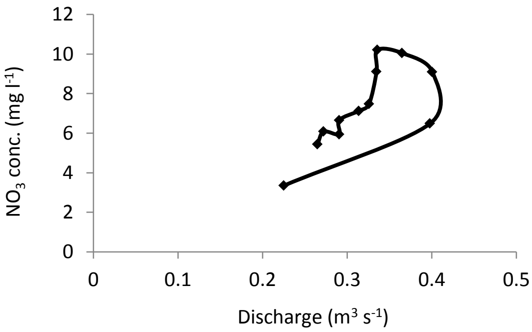


Event B – 01/10/2010

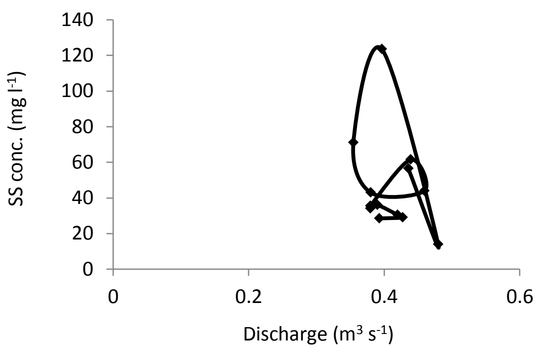


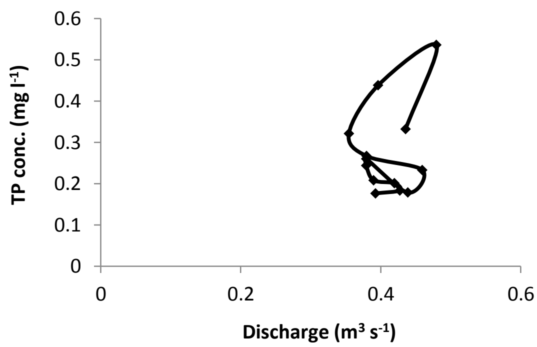




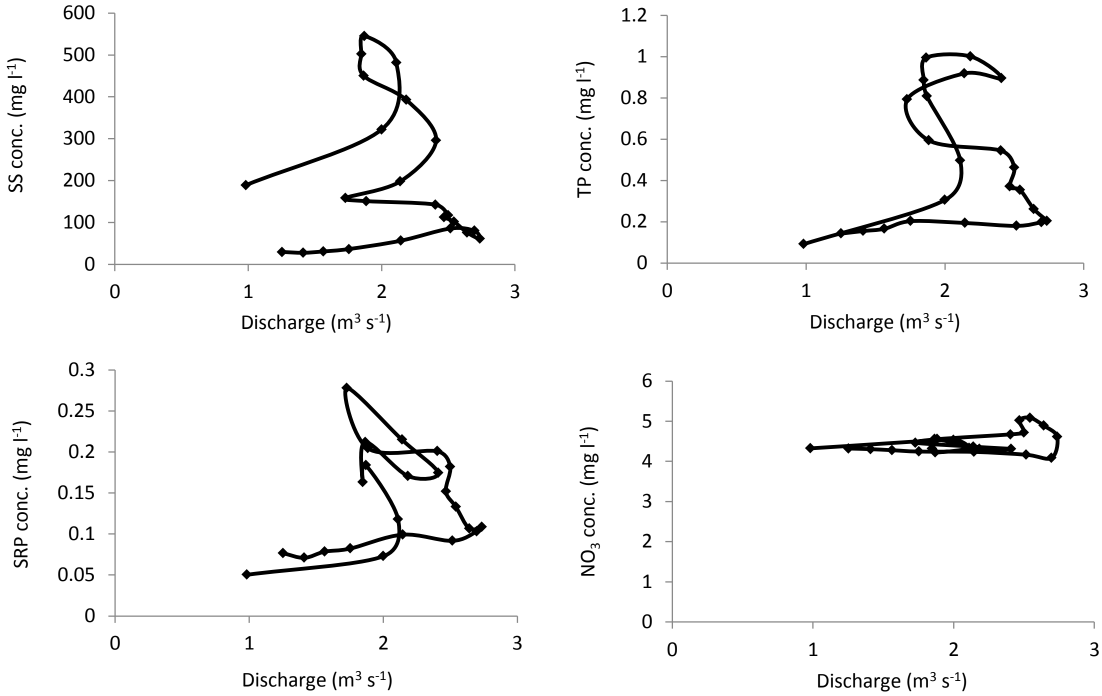


Event C – 06/10/2010

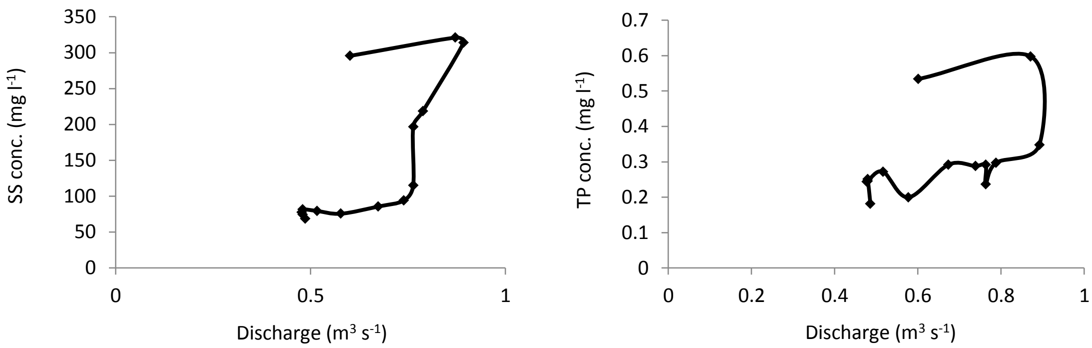




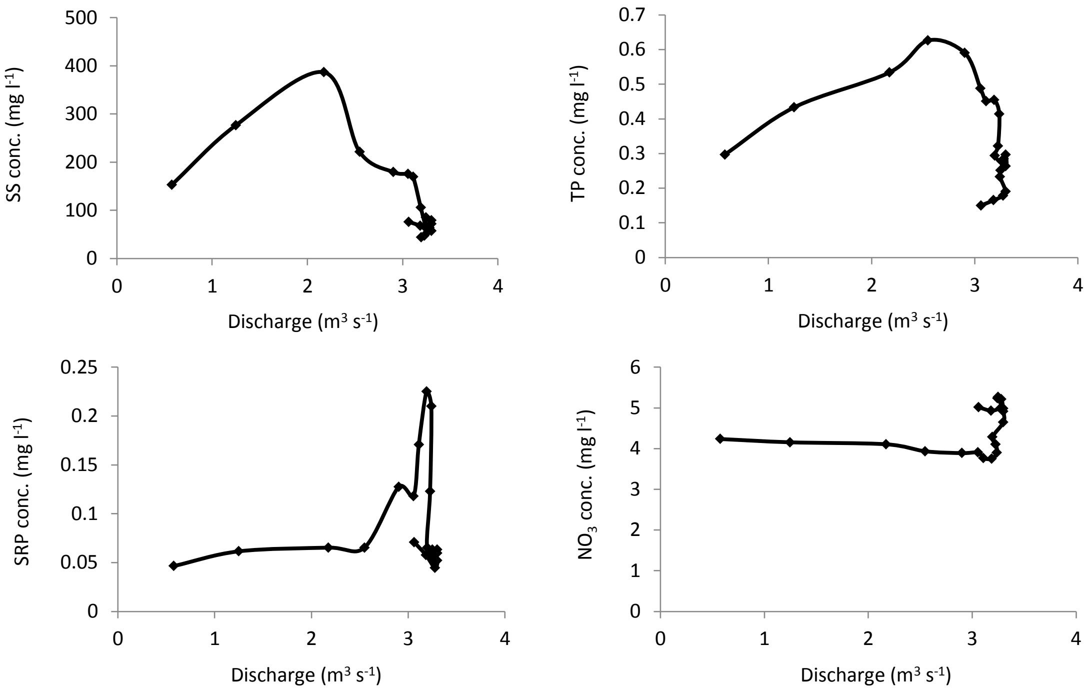
Event D – 11/11/2010



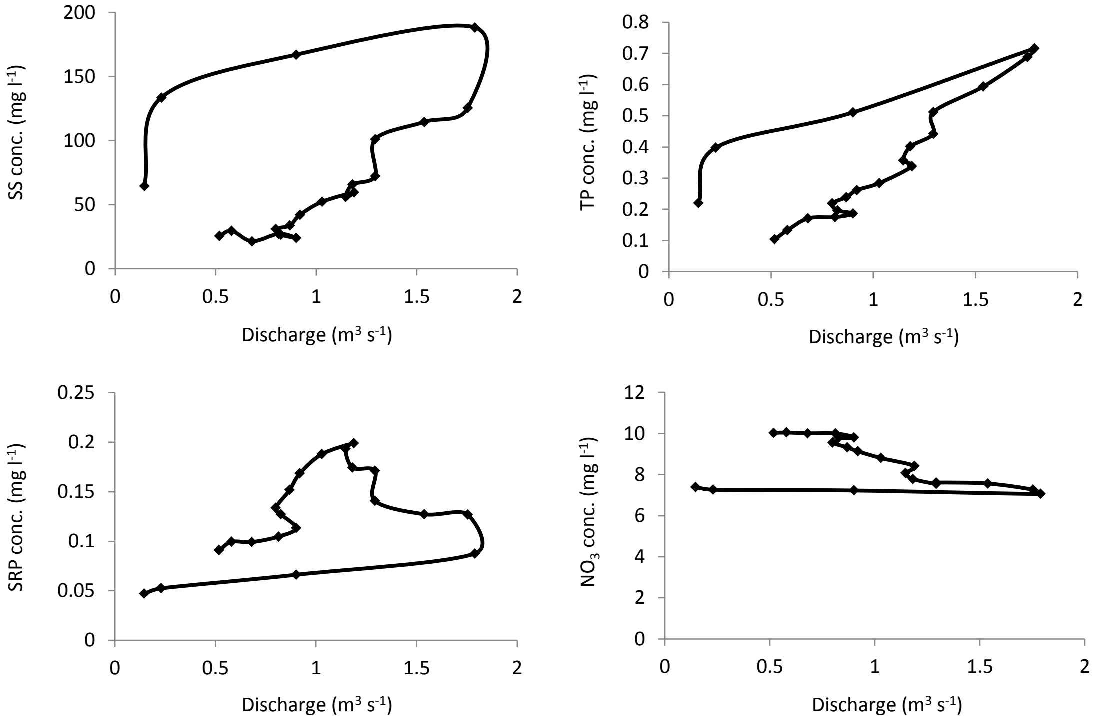
Event E – 10/12/2010



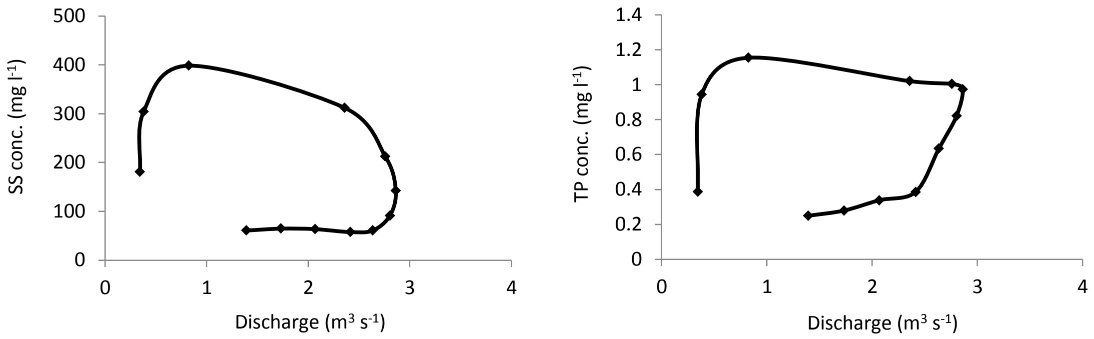
Event F – 15/01/2011



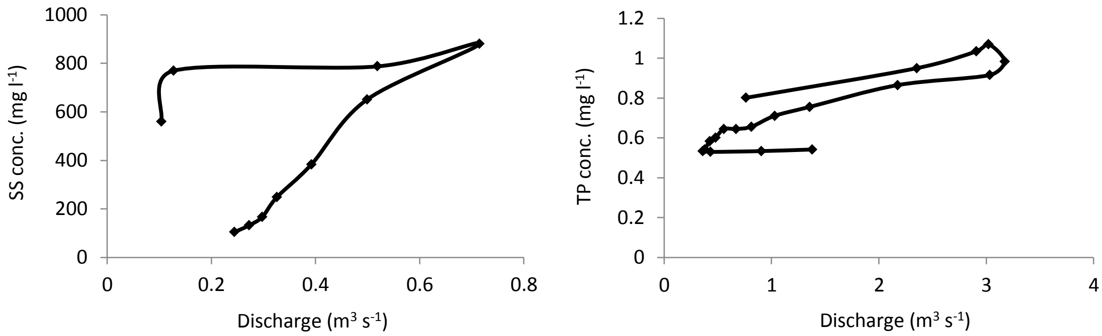
Event G – 09/03/2011



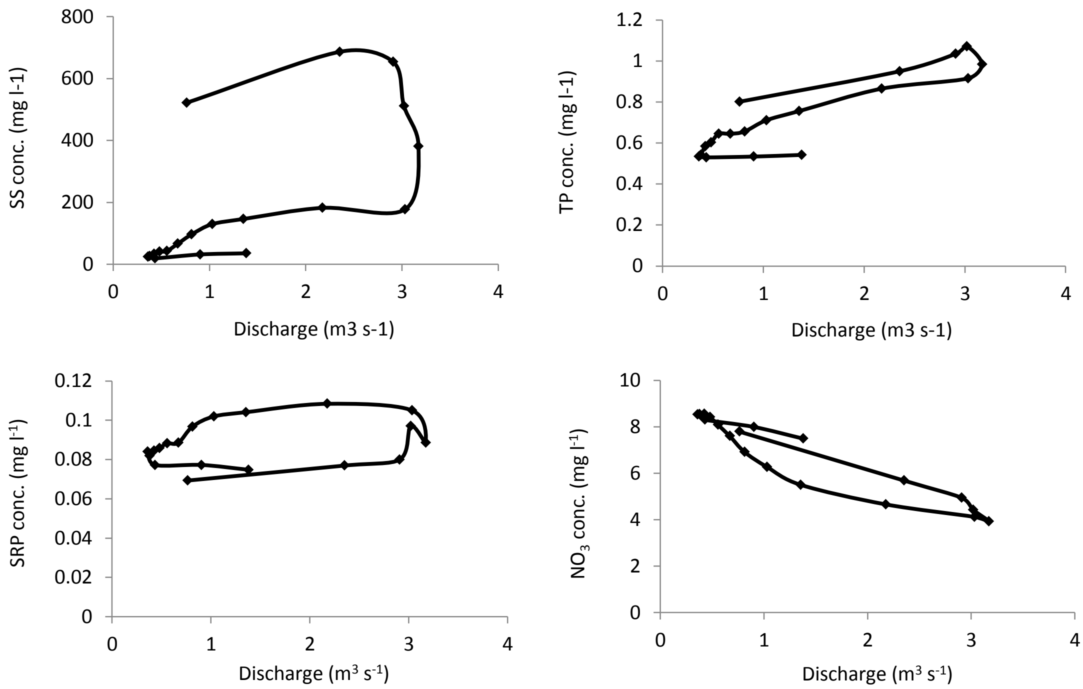
Event H – 05/04/2011



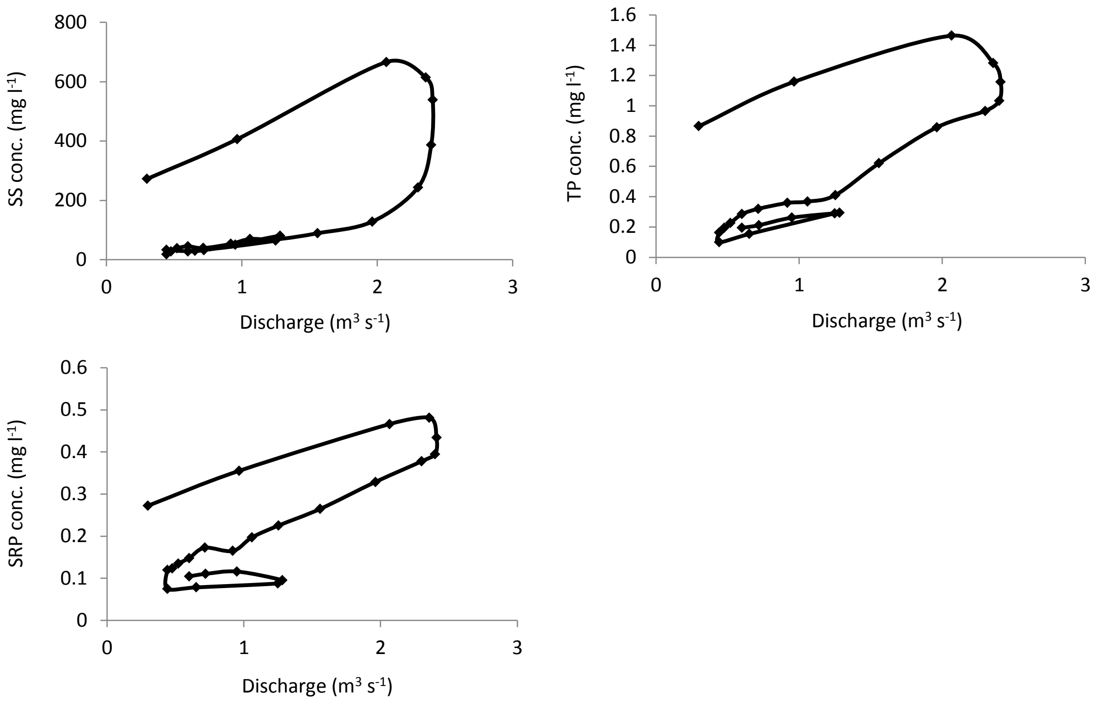
Event I – 23/05/2011



Event J – 22/06/2011



Event K – 25/11/2011



G5. Revised determinand concentration-discharge rating coefficients.

Constituent	10a	b	R^2	Number of samples
SS	1.8511	0.9358	0.44	218
TP	-0.5432	0.6757	0.45	218
SRP	-1.1612	0.4693	0.4	145
NO_3	0.7961	-0.268	0.54	139

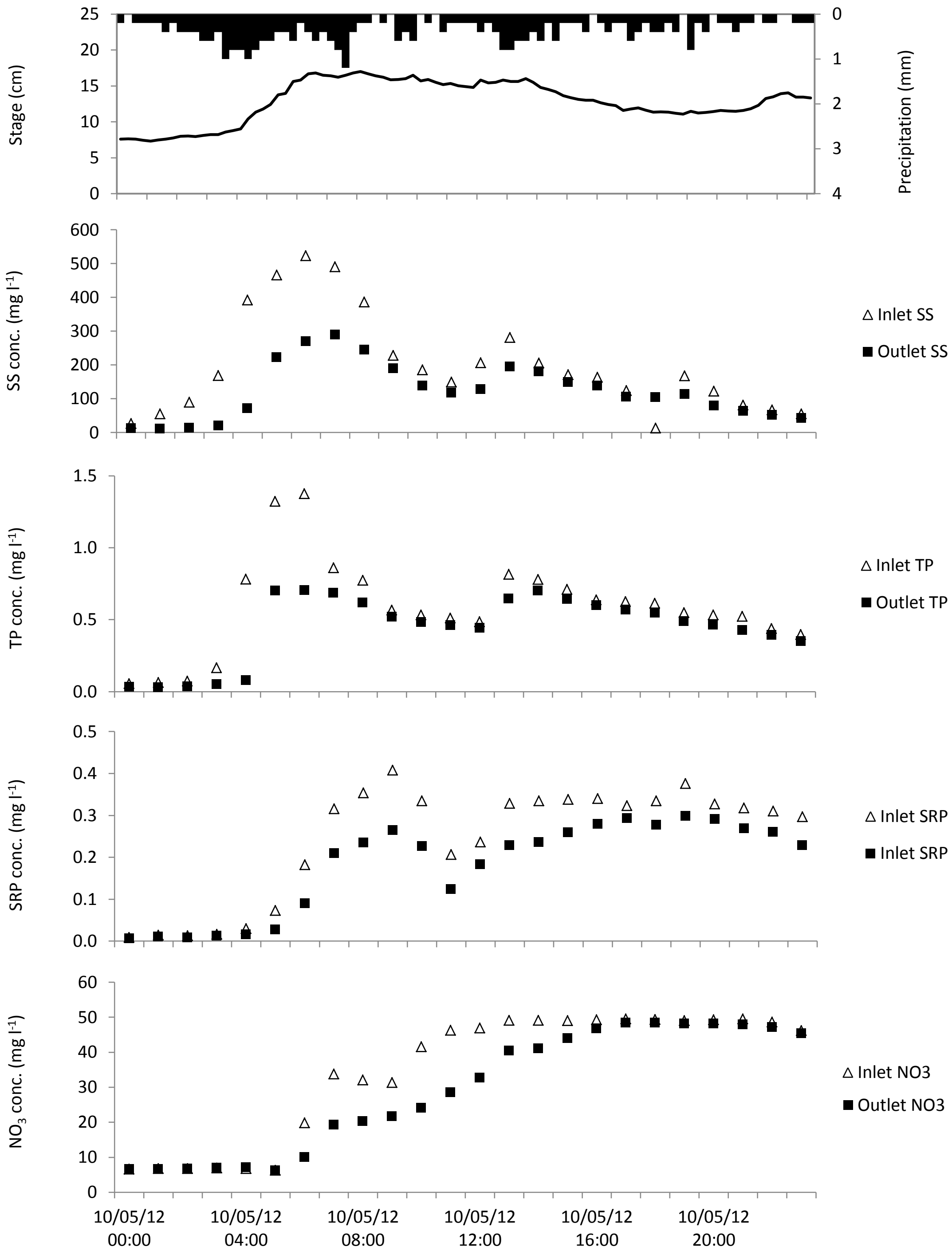
Appendix H – Belford

H1. Storm event raw contaminant concentration and discharge data.

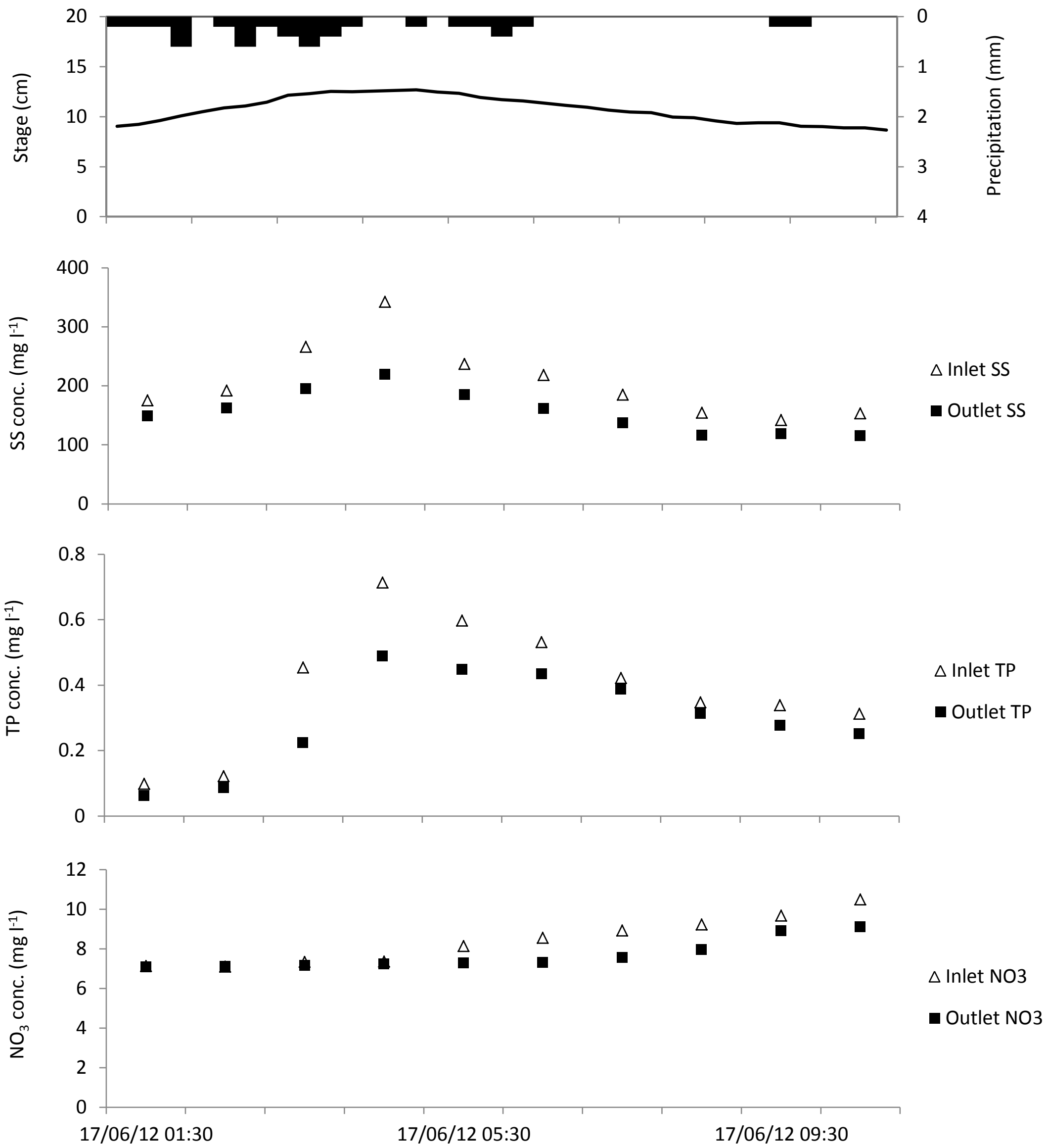
See CD-ROM.

H2. Storm event graphs.

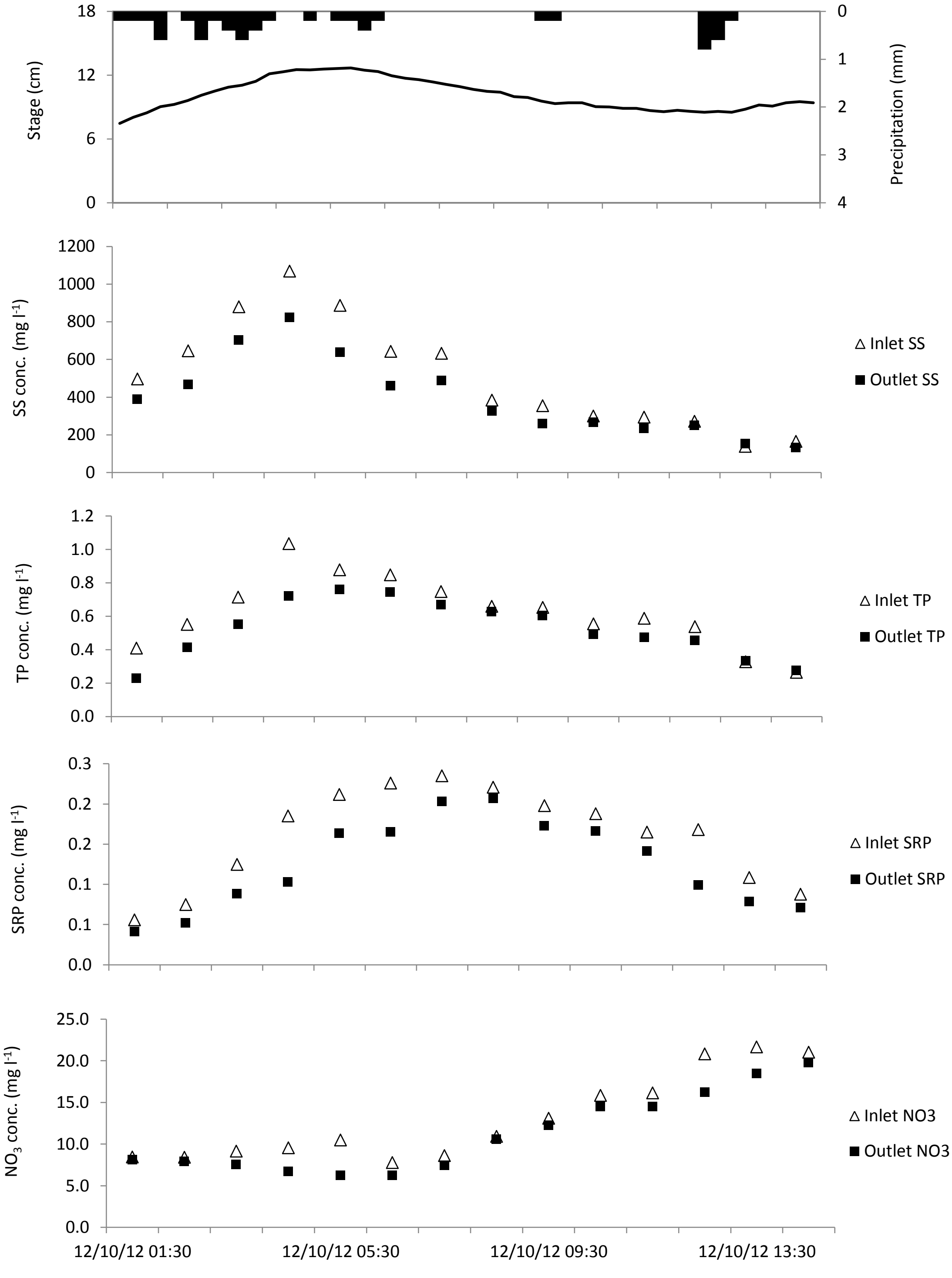
Event 1 – 10/05/2012



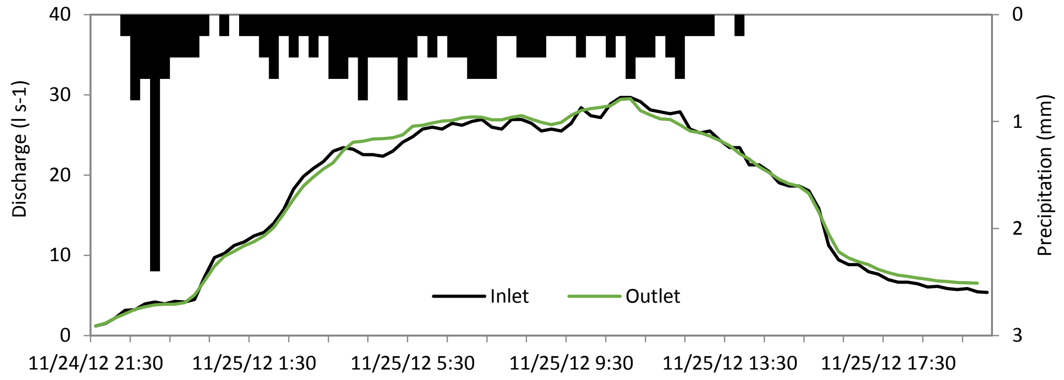
Event 2 – 17/06/2012

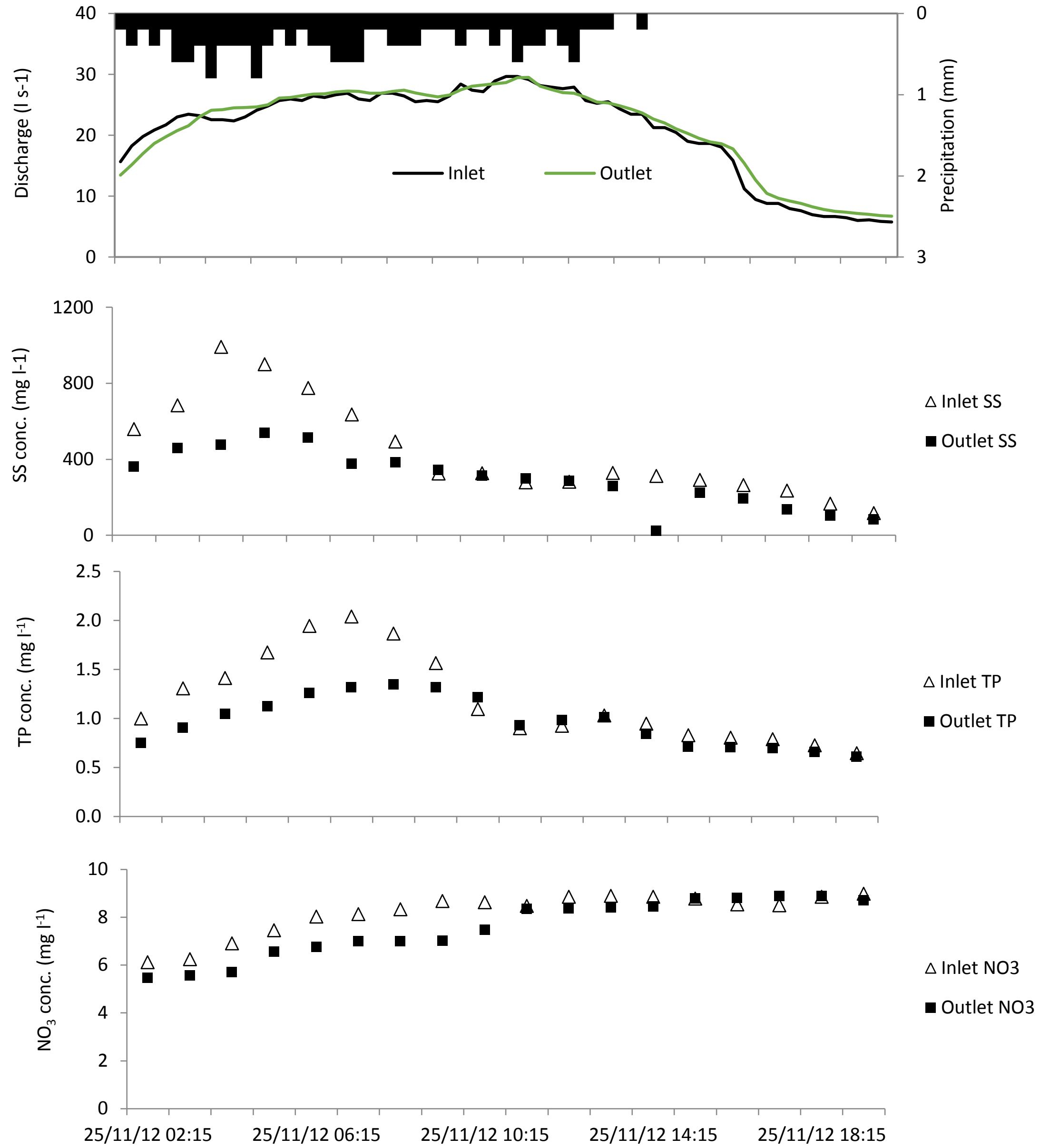


Event 3 – 12/10/2012

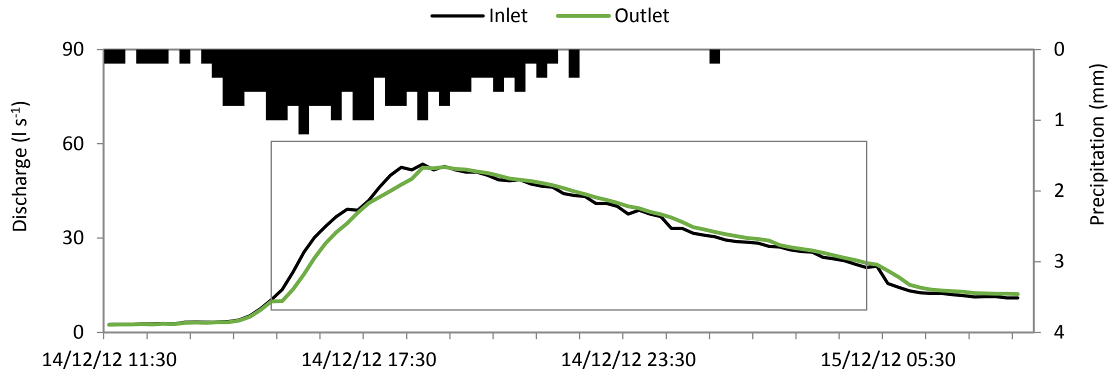


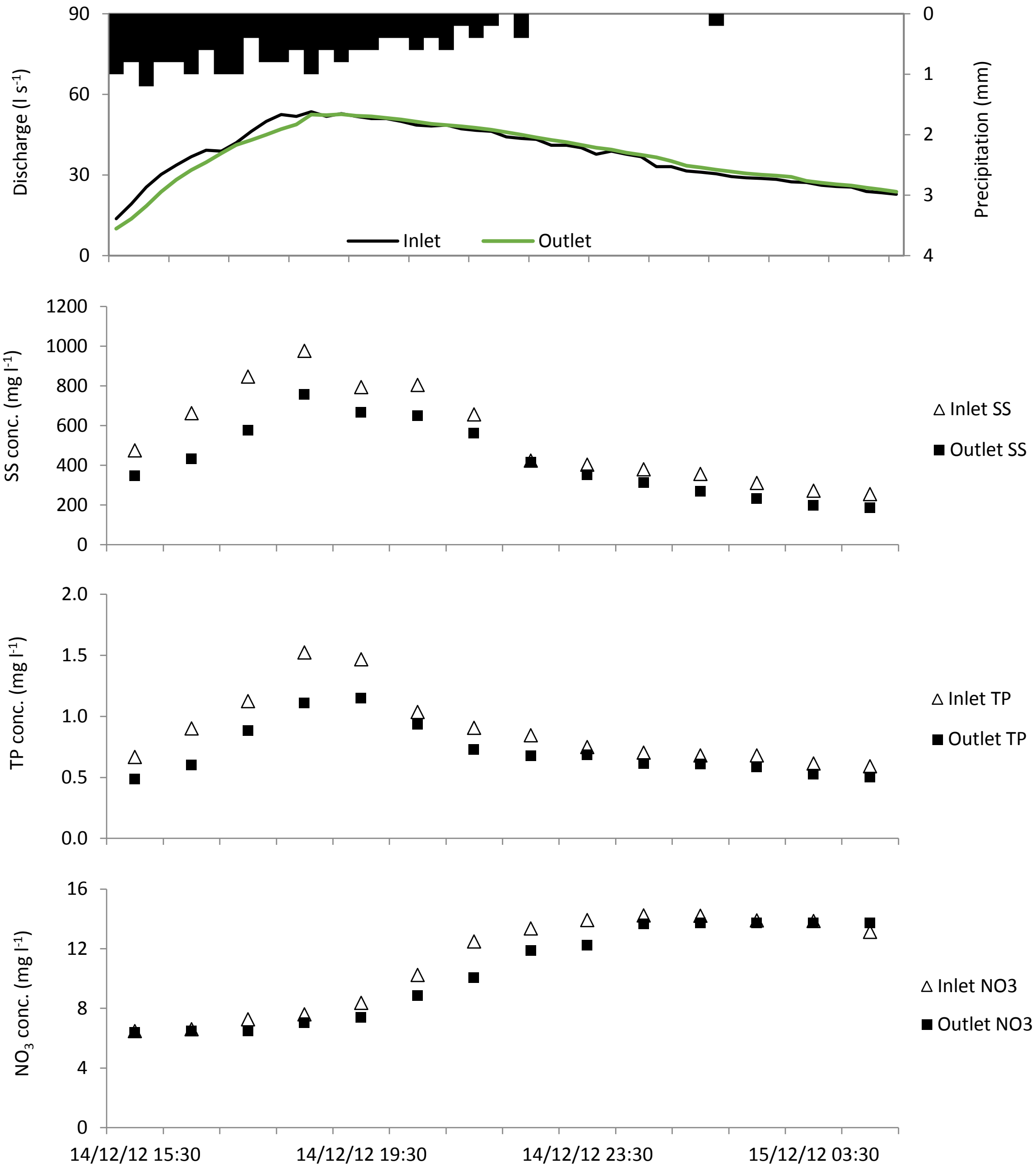
Event 4 – 25/11/2012



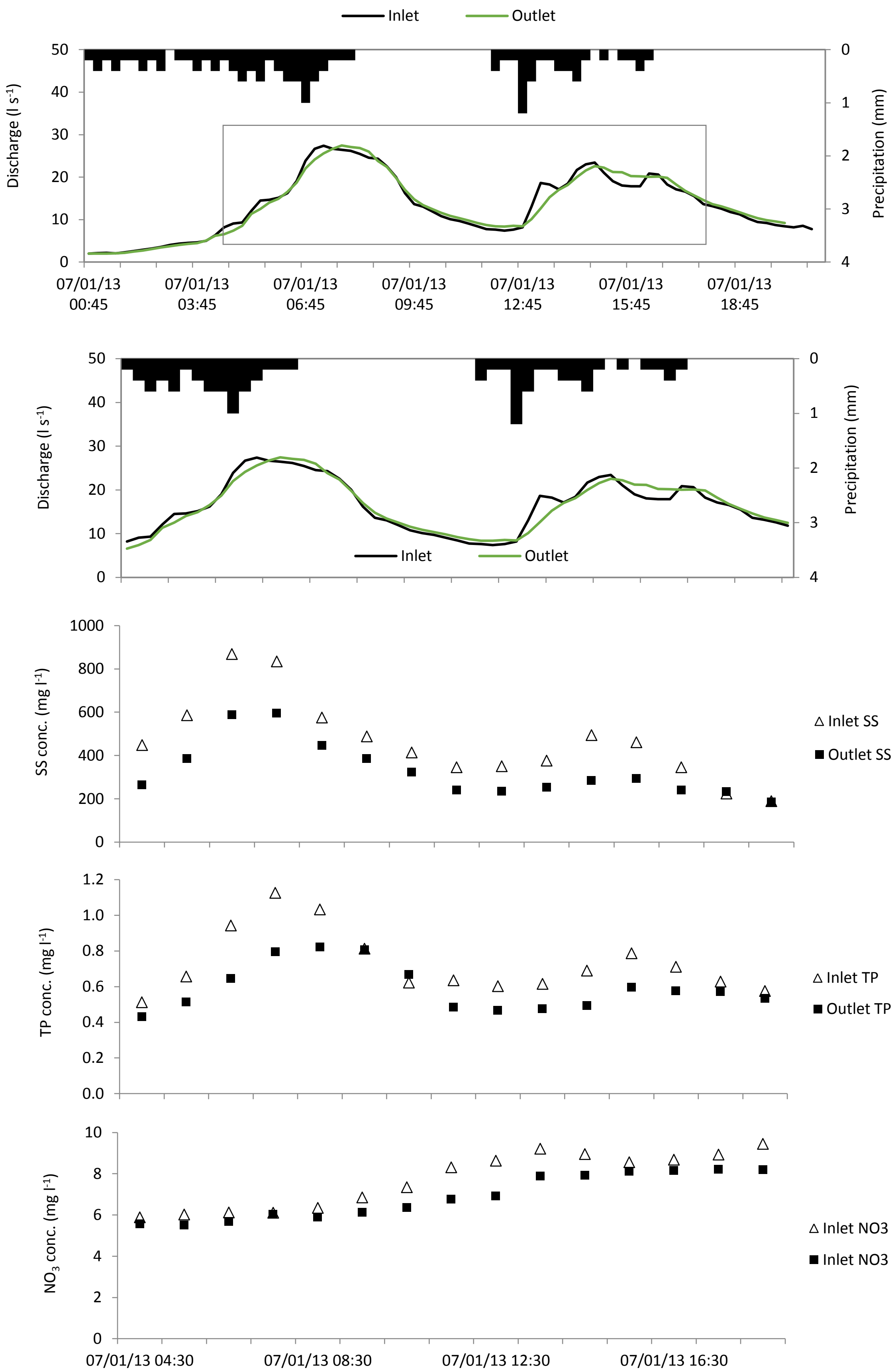


Event 5 – 14/12/2012

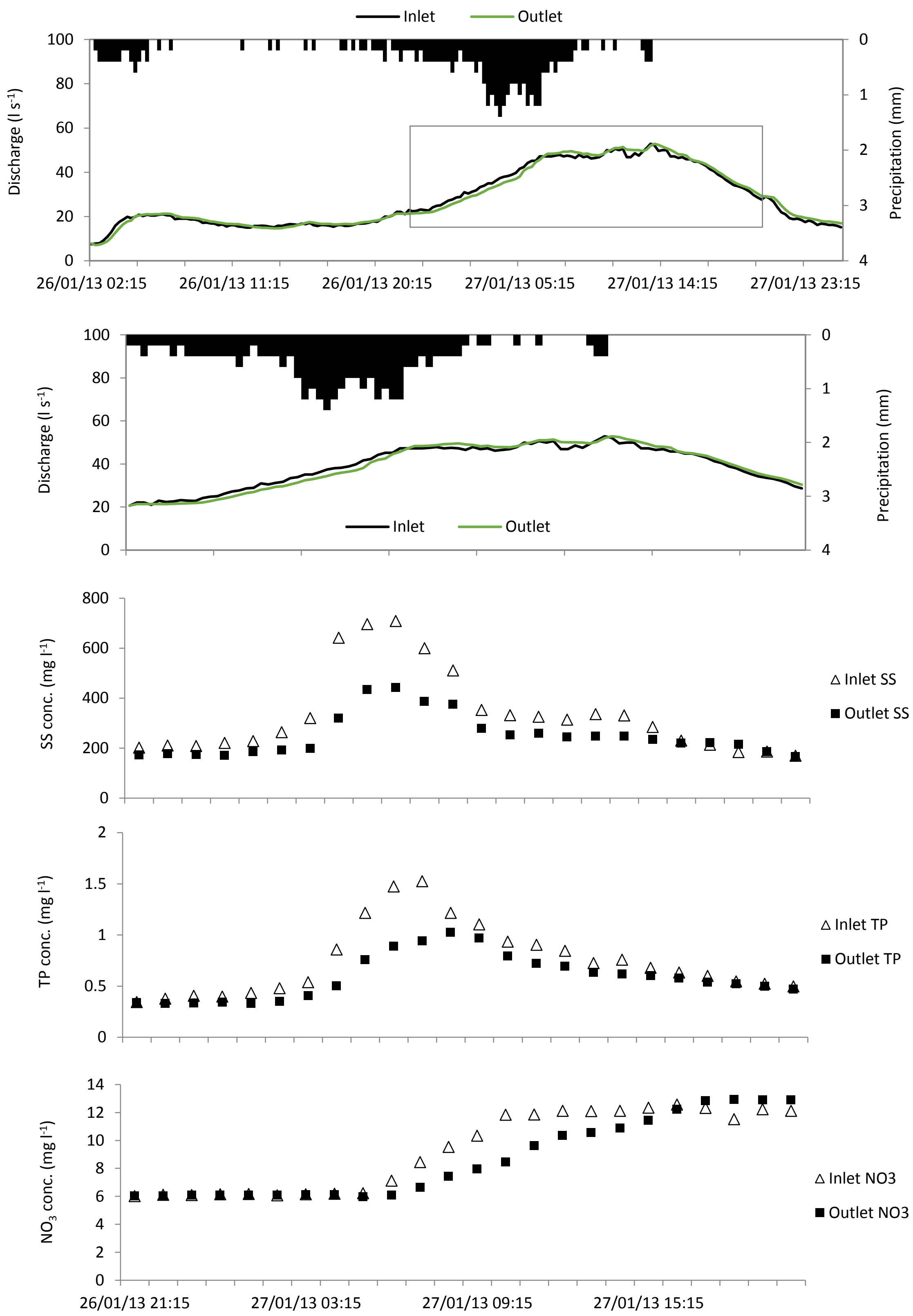




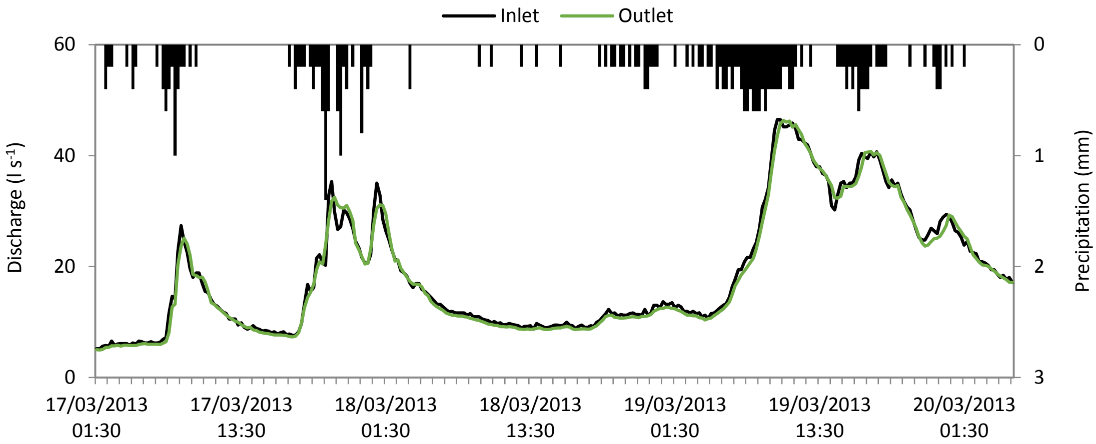
Event 6 – 07/01/2013

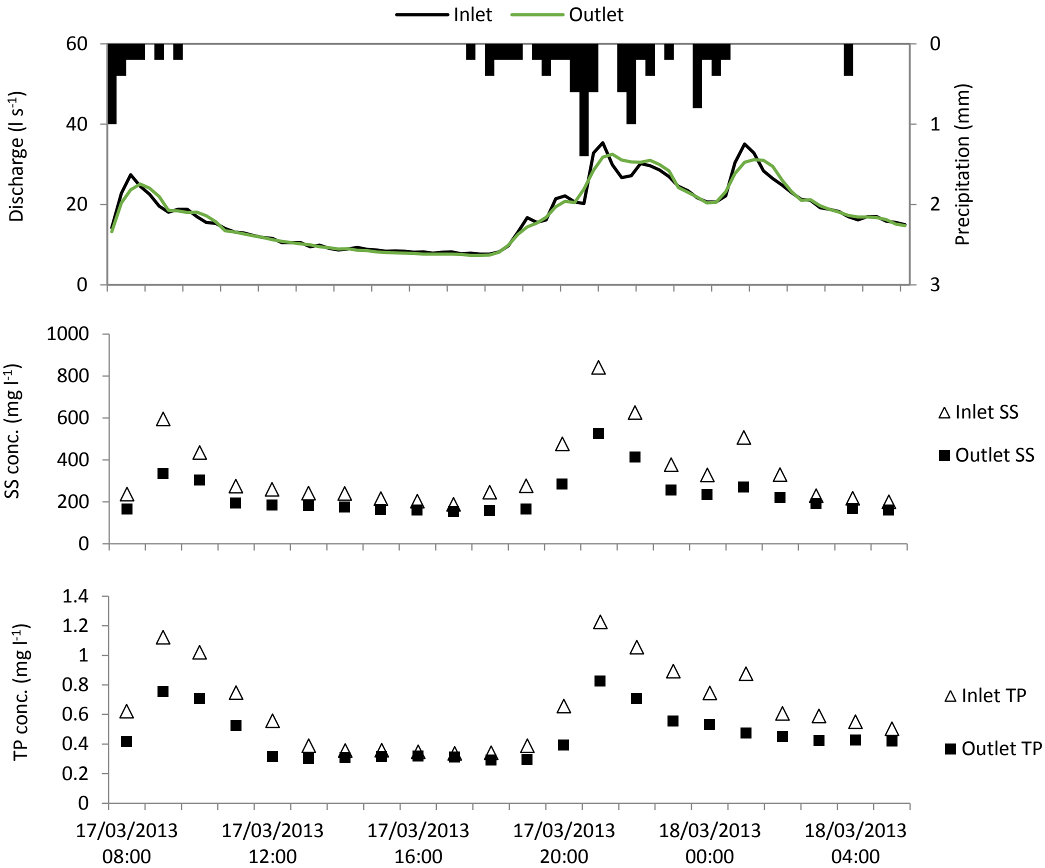


Event 7 – 26/01/2013

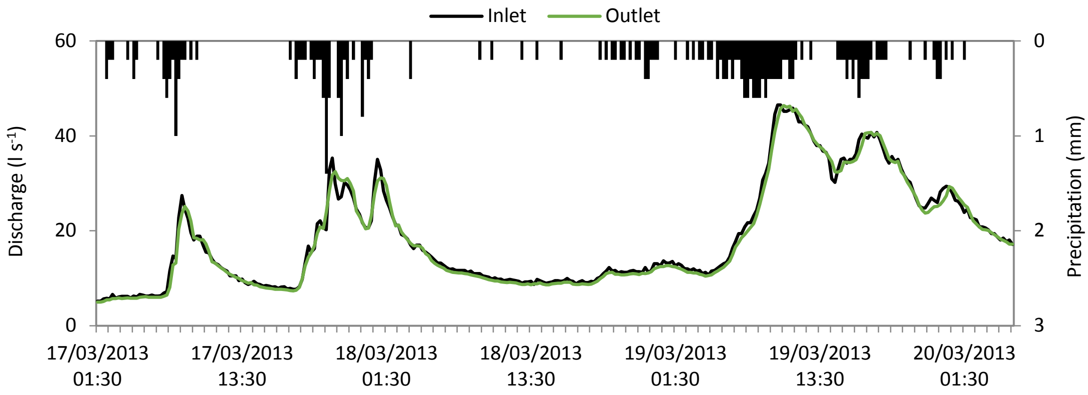


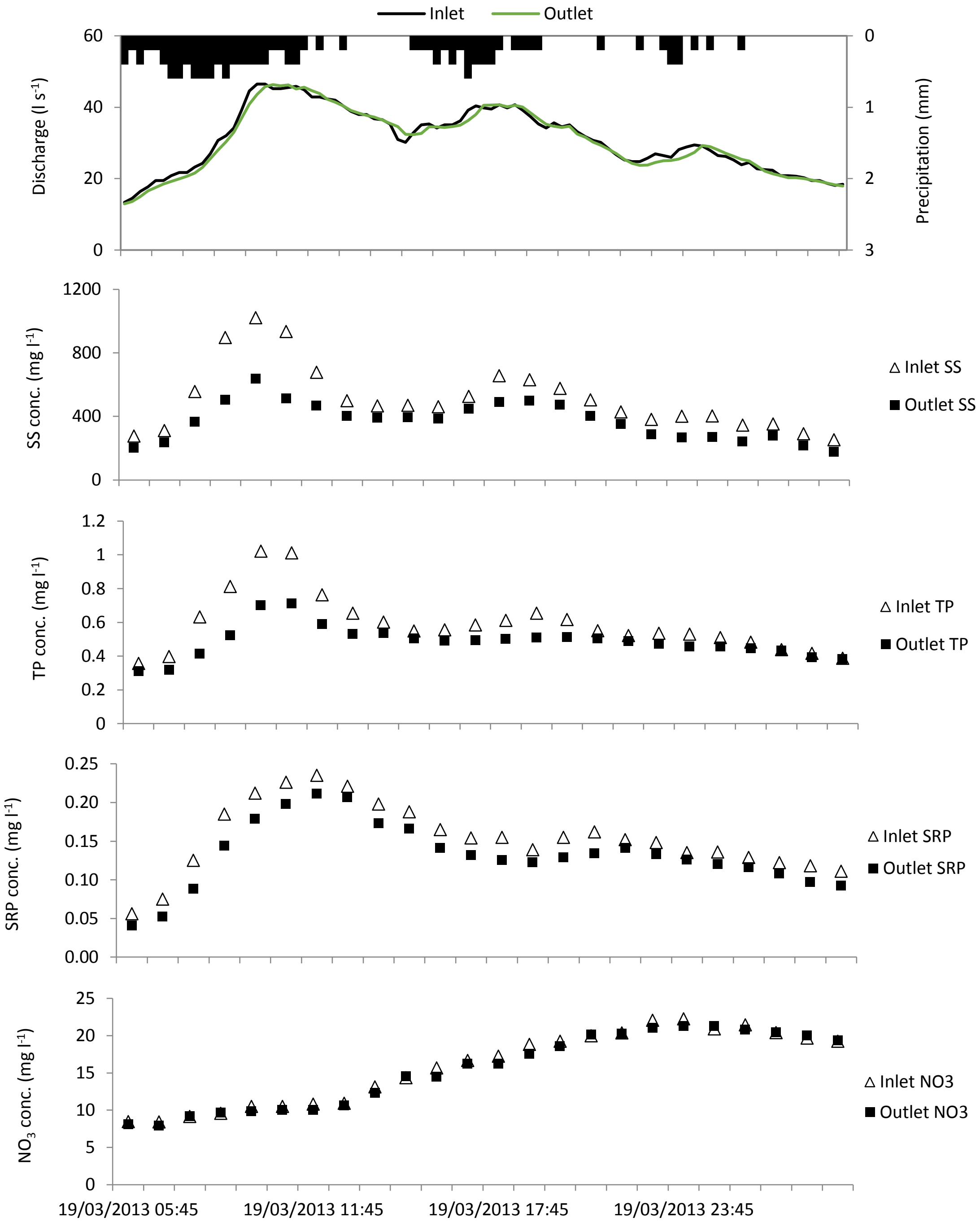
Event 8 – 17/08/2013





Event 9 – 19/03/2013





H3. Summary tables of multi-RAF performance (recorded during 9 sampled storm events 2012-2013).

Suspended sediment

Event	Date	Duration (hrs)	Suspended sediment reduction (%)			
			Concentration (instantaneous)		Load (hourly)	
			Range	Mean	Range	Total
1	10/05/2012	24	12.2 - 87.9	37.6		
2	17/06/2012	12	15.4 - 36.0	23.4		
3	12/10/2012	14	-9.4 - 28.3	18.9		
4	25/11/2012	18	-7.2 - 51.8	25.1	-13.6 - 65.4	31.8
5	14/12/2012	14	1.9 - 34.7	21.7	-4.1 - 64.2	23.8
6	07/01/2013	15	-3.6 - 42.5	28.0	12.1 - 55.5	31.6
7	26/01/2013	24	-16.3 - 50.2	20.2	-20 - 53.7	26.5
8	17/03/2013	22	22 - 47	30.5	16.7 - 49.8	37.2
9	19/03/2013	24	15.2 - 45.3	26.2	11.6 - 48.9	29.5
			Mean	25.7		30.1

Total phosphorus

Event	Date	Duration (hrs)	Total phosphorus reduction (%)			
			Concentration (instantaneous)		Load (hourly)	
			Range	Mean	Range	Total
1	10/05/2012	24	6.1 - 89.9	25.2		
2	17/06/2012	12	8.1 - 50.7	24.8		
3	12/10/2012	14	-4.0 - 44.2	15		
4	25/11/2012	18	-11.0 - 35.5	15.2	-64 - 59.6	20.8
5	14/12/2012	14	8.6 - 33.4	18.3	-2.3 - 63.4	21.3
6	07/01/2013	15	-7.1 - 39.2	19.2	-24.2 - 43.5	22
7	26/01/2013	24	2.0 - 41.7	18.3	0.2 - 45.8	21.4
8	17/03/2013	22	7.5 - 46.1	26.6	10.6 - 48.8	33.9
9	19/03/2013	24	1.8 - 35.8	13.9	2.0 - 41.5	20.2
			Mean	19.6		23.3

Soluble reactive phosphorus

Event	Date	Duration (hrs)	Soluble reactive phosphorus reduction (%)			
			Concentration (instantaneous)		Load (hourly)	
			Range	Mean	Range	Total
1	10/05/2012	24	9.3 - 62.0	27.8		
2	17/06/2012	12				
3	12/10/2012	14	6.6 - 44.4	23.5		
4	25/11/2012	18				
5	14/12/2012	14	- 9.1 - 28.1	9.2	-20.6 - 58.3	10
6	07/01/2013	15				
7	26/01/2013	24				
8	17/03/2013	22				
9	19/03/2013	24	6.6 - 30.7	15.2	3.4 - 37.5	14.7
			Mean	18.9		12.4

Nitrate

Event	Date	Duration (hrs)	Nitrate reduction (%)			
			Concentration (instantaneous)		Load (hourly)	
			Range	Mean	Range	Total
1	10/05/2012	24	- 5.7 - 48.9	13.8		
2	17/06/2012	12	- 0.2 - 15.3	8		
3	12/10/2012	14	3.0 - 40.8	14.5		
4	25/11/2012	18	-4.4 - 19.0	7.9	-81.7 - 52.1	8.7
5	14/12/2012	14	-4.4 - 19.5	6.8	-16 - 48.3	6.3
6	07/01/2013	15	1.3 - 19.7	10.1	-11.5 - 33.3	9.4
7	26/01/2013	24	-12.2 - 28.6	6.6	-16.4 - 27.3	8.1
8	17/03/2013	22				
9	19/03/2013	24	-1.8 - 7.7	3.9	-7.8 - 15.6	5.3
			Mean	9.0		7.6

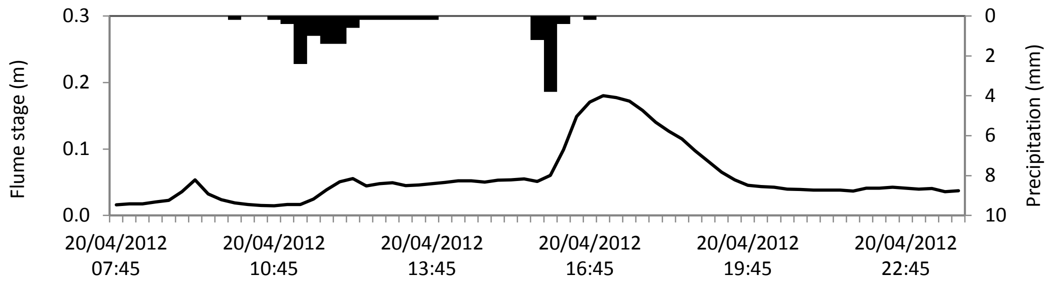
Appendix I – Netherton

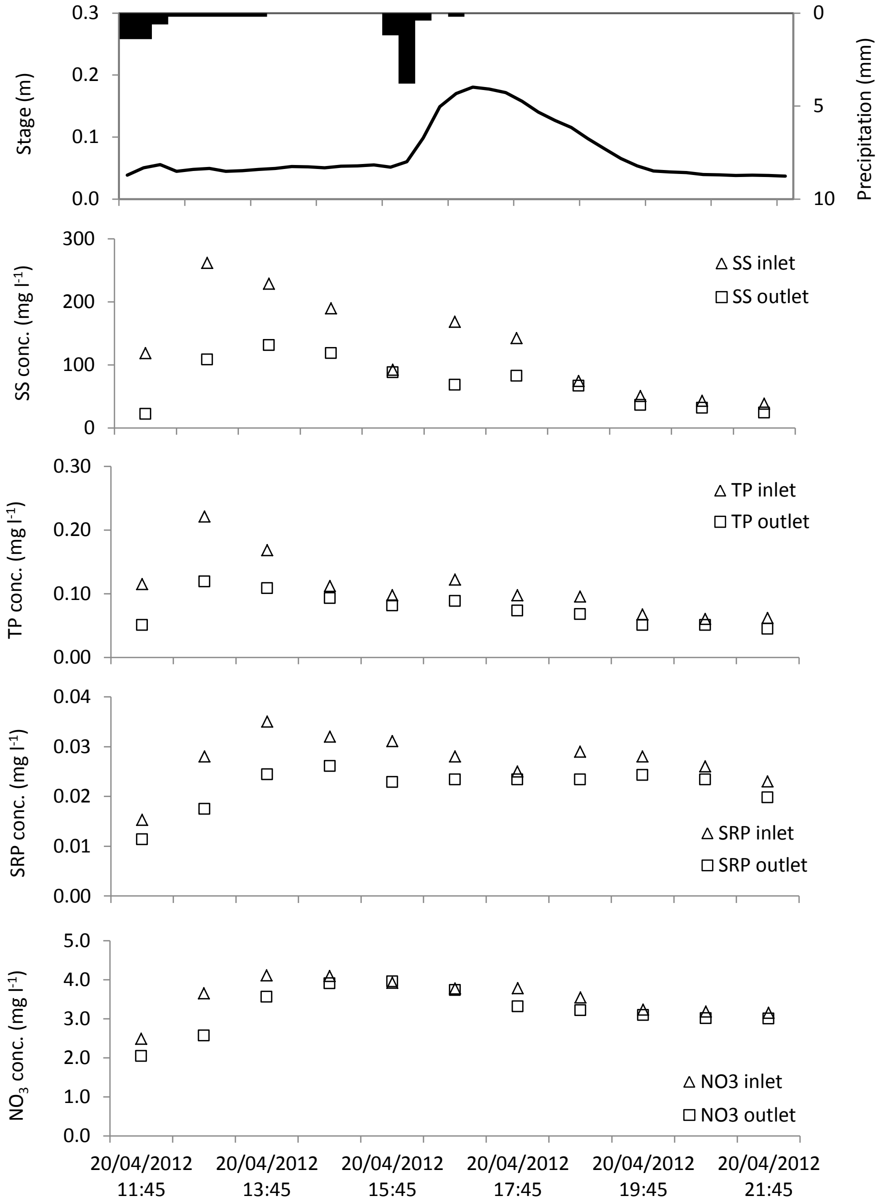
I1. Storm event raw contaminant concentration and discharge data.

See CD-ROM.

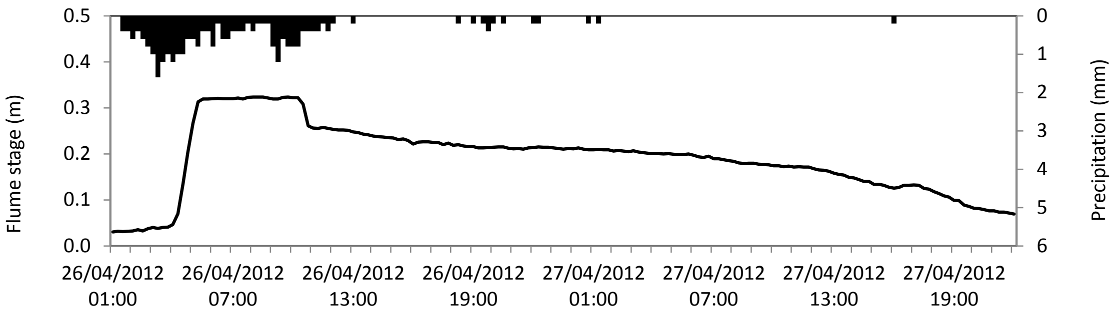
I2. Storm event graphs.

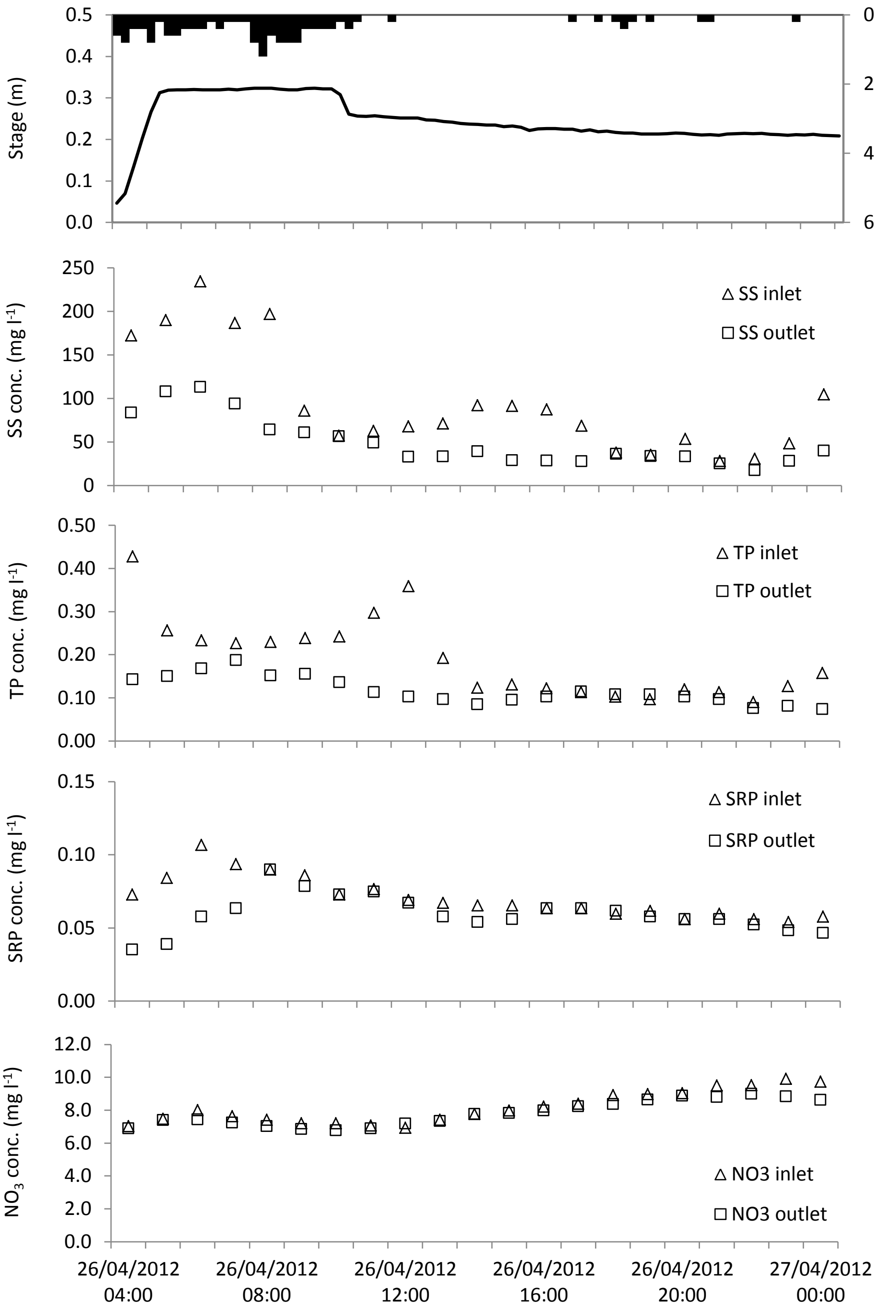
Event 1 - 20/04/2012



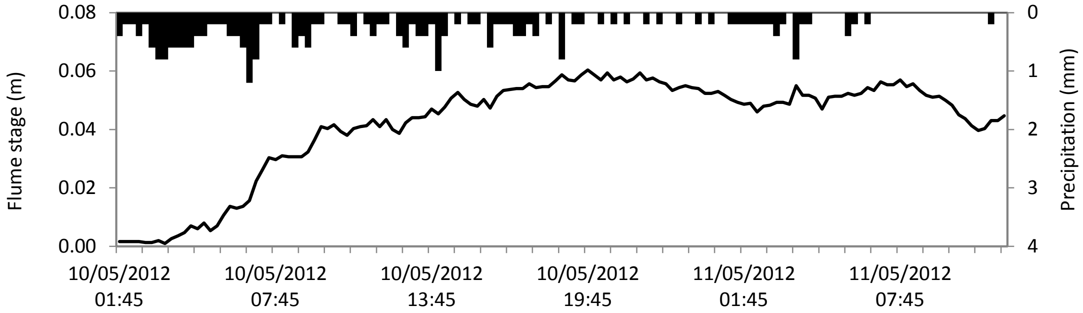


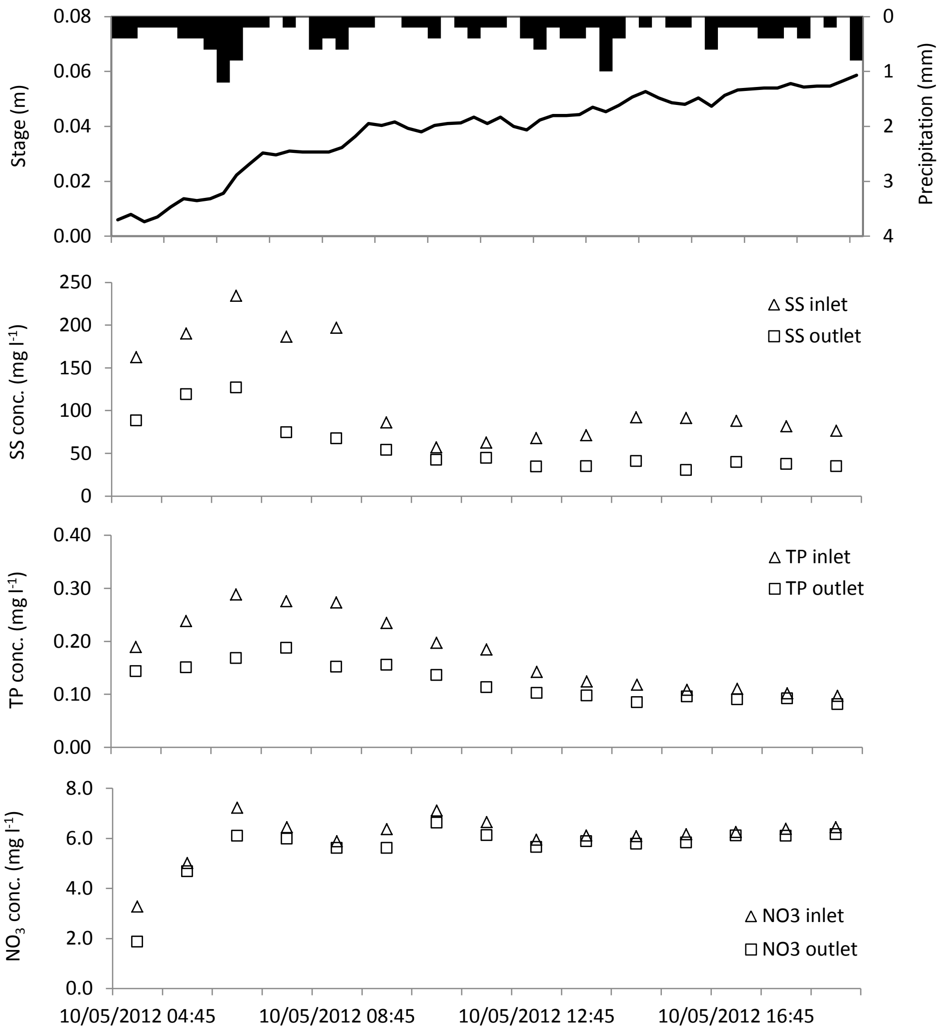
Event 2 - 26/04/2012



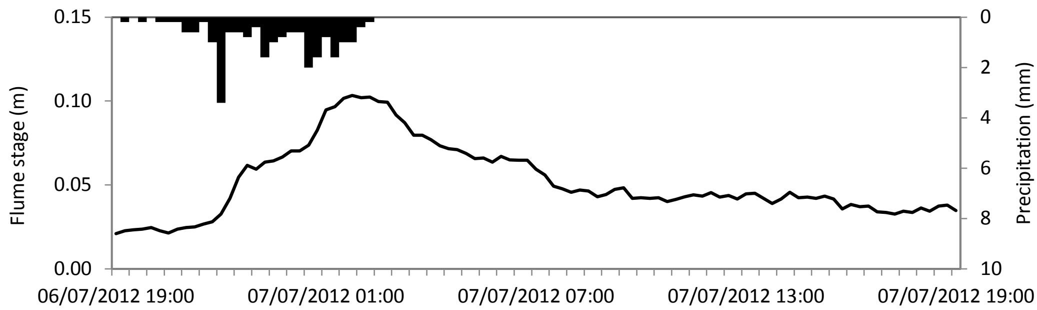


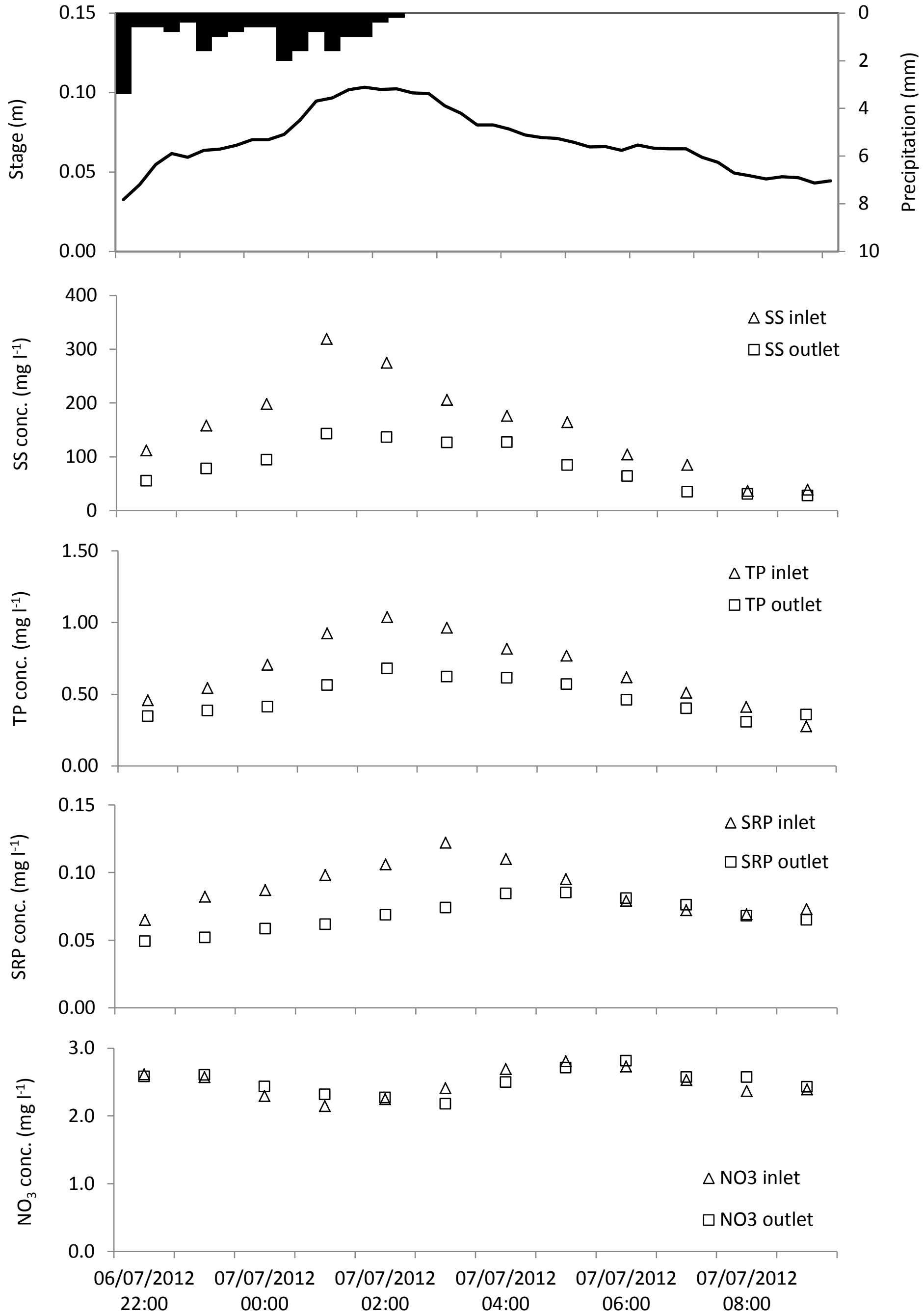
Event 3 - 10/05/2012



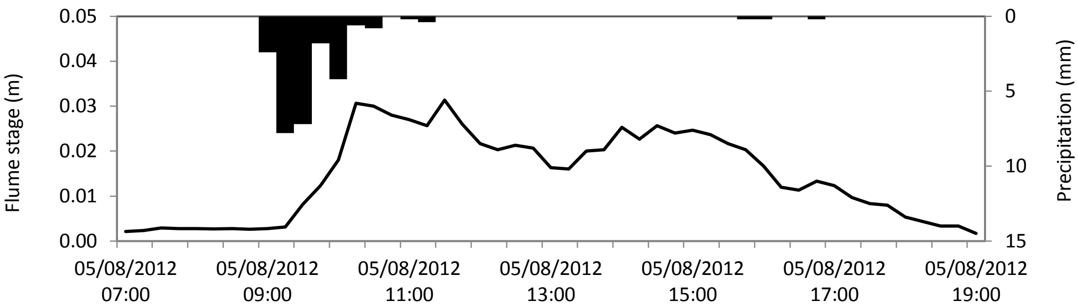


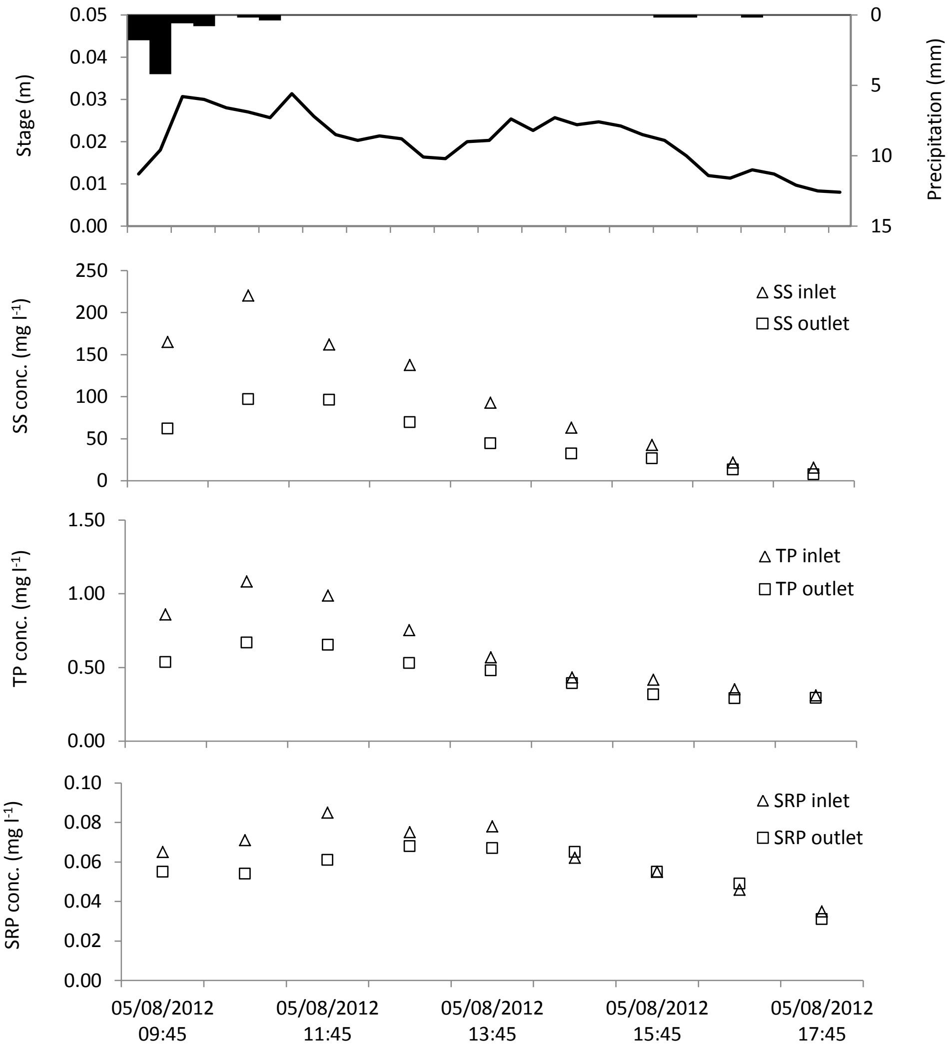
Event 4 - 06/07/2012



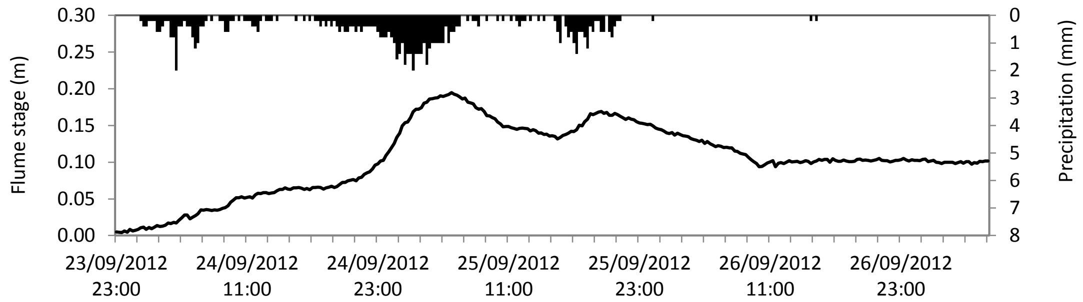


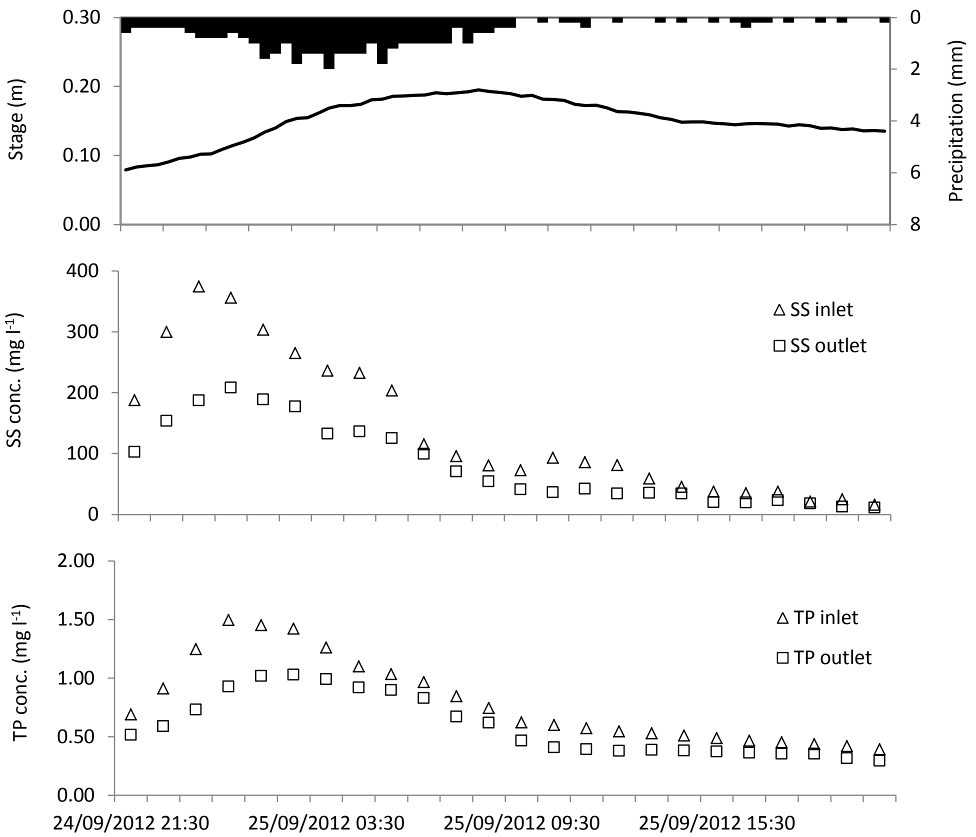
Event 5 - 05/08/2012



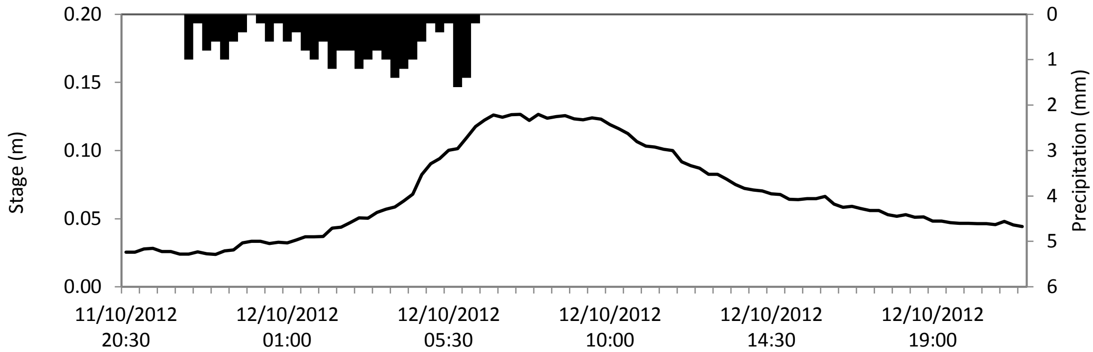


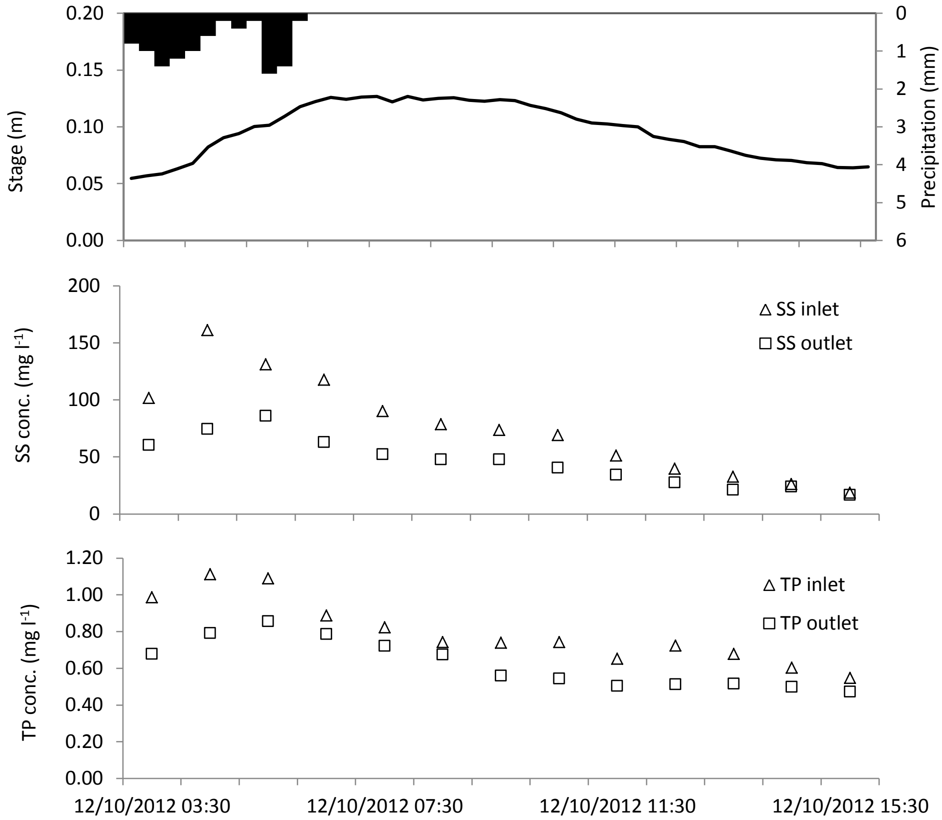
Event 6 - 24/09/2012



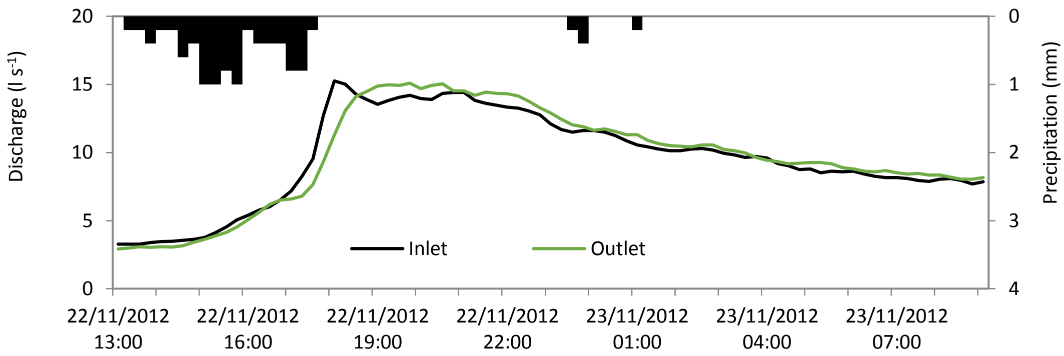


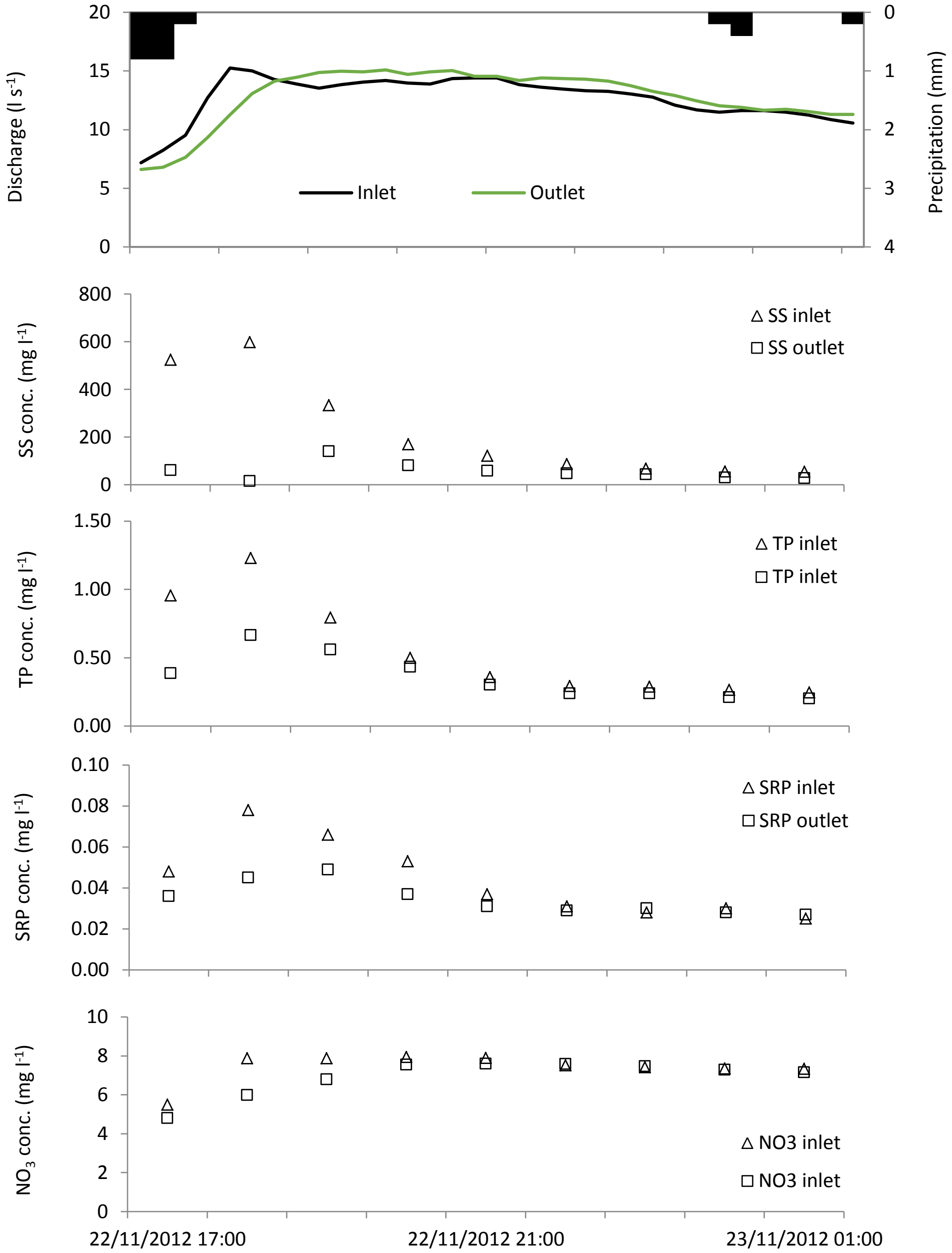
Event 7 - 12/10/2012



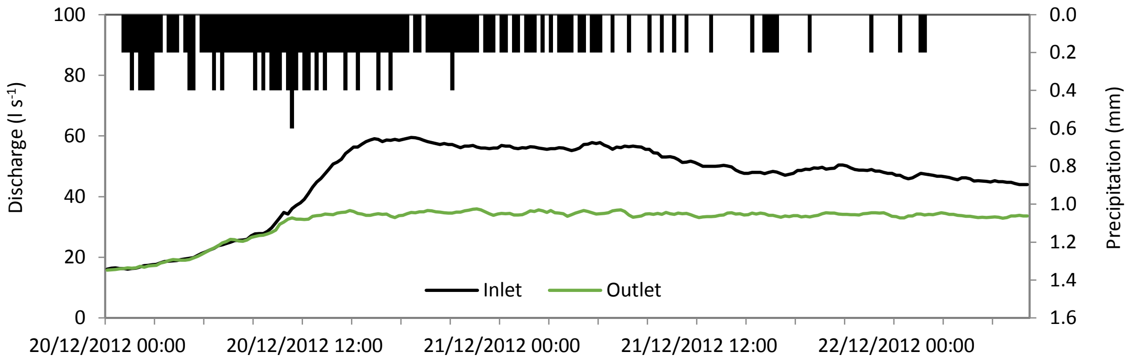


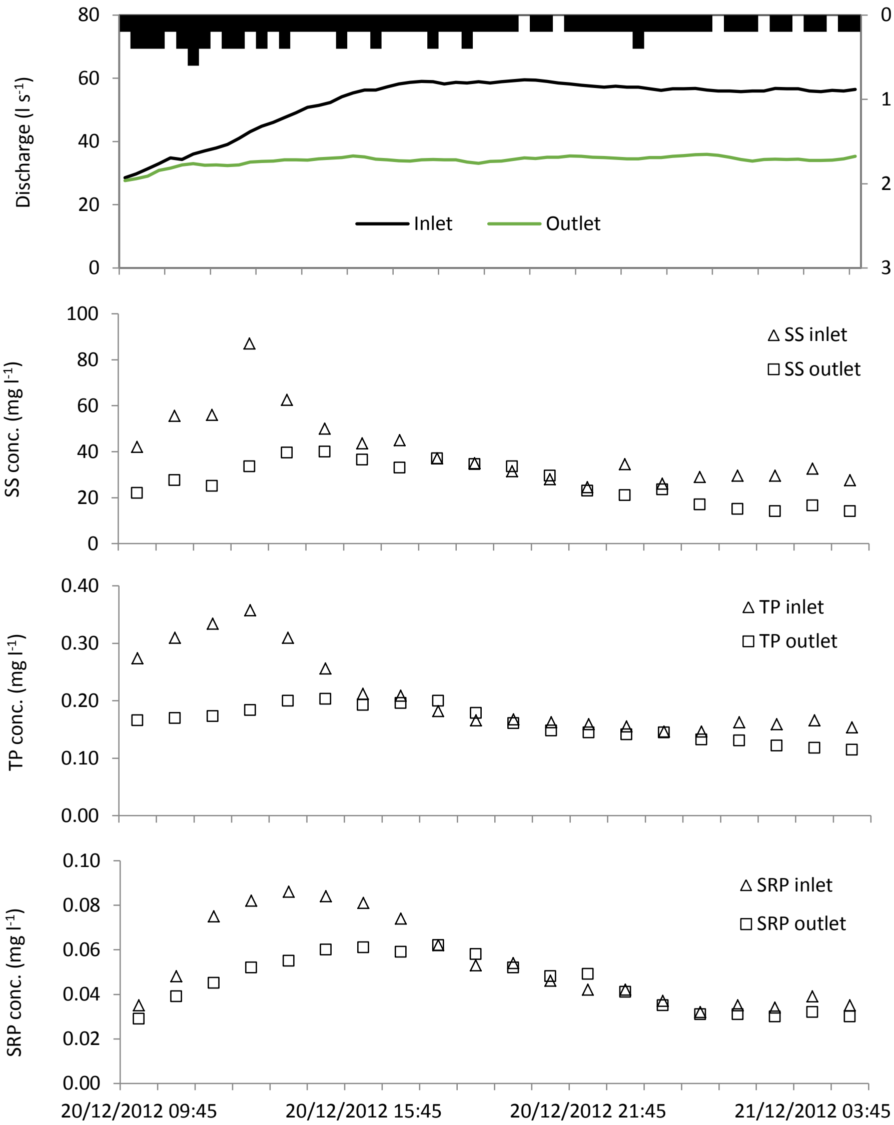
Event 8 - 22/11/2012



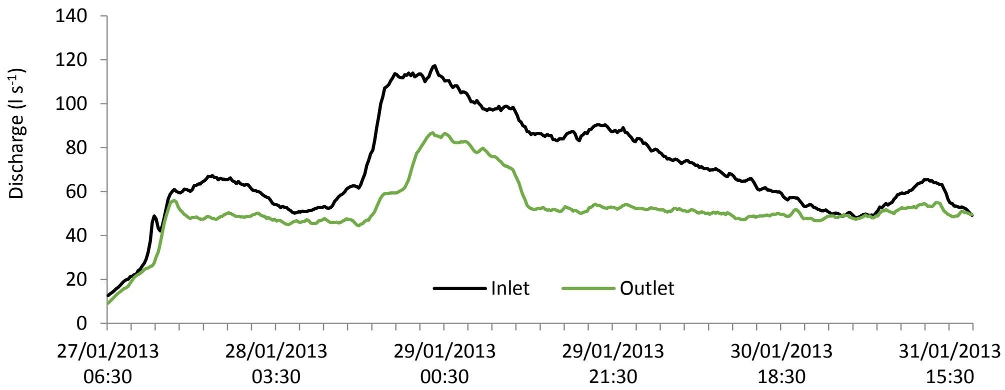


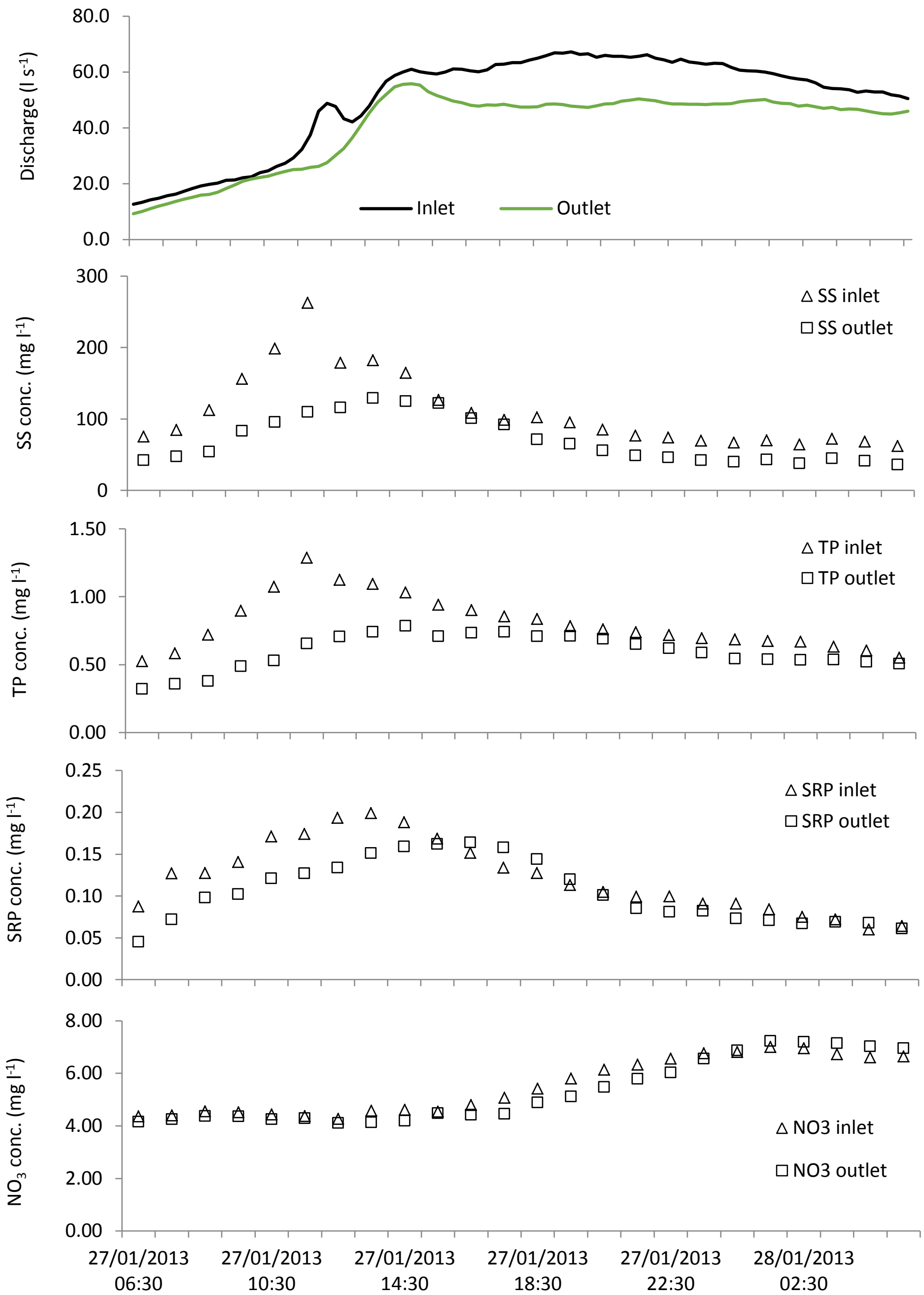
Event 9 - 20/12/2012



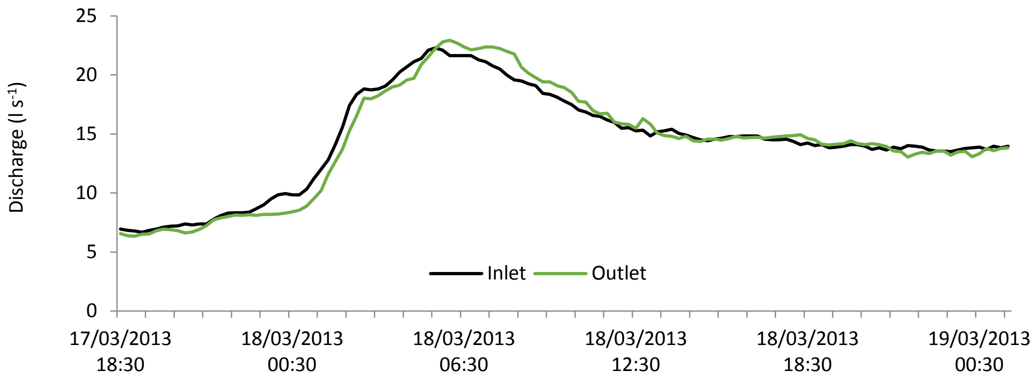


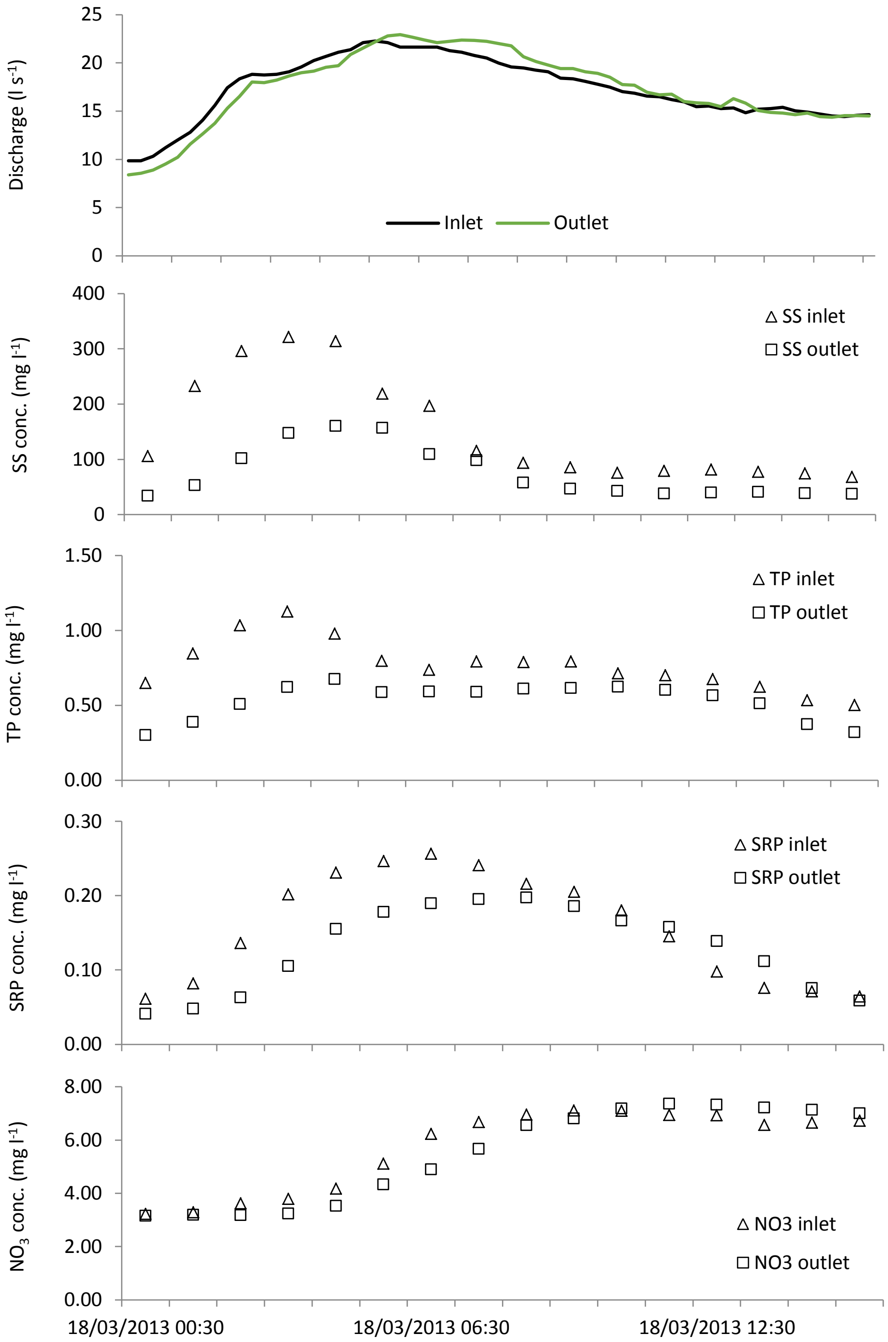
Event 10 - 27/01/2013





Event 11 - 18/03/2013





I3. Summary tables of three-cell RAF performance (recorded during 11 sampled storm events 2012-2013).

Suspended sediment

Event	Start time/date	Duration sampled (hrs)	Suspended sediment reduction (%)			
			Concentration (instantaneous)		Load (hourly)	
			Range	Mean	Range	Total
1	20/04/2012 11:45	11	4.3 - 81.4	38.8		
2	26/04/2012 09:15	21	0.8 - 68.3	41.7		
3	10/05/2012 04:45	15	25.4 - 66.7	48.7		
4	06/07/2012 22:30	12	16.4 - 59.4	43.2		
5	05/08/2012 10:15	9	37.6 - 62.4	48.7		
6	24/09/2012 11:00	24	13.6 - 60.9	39.8		
7	12/10/2012 03:00	13	8.3 - 53.8	36.6		
8	22/11/2012 16:00	9	34.3 - 88.3	55.2	72.1 - 91	83.1
9	20/12/2012 09:00	20	-6.3 - 61.5	30	31.3 – 61.5	29.6
10	27/01/2013 09:30	24	3.5 - 58.2	35.4	16 – 58.2	32.6
11	18/03/2013 00:30	16	14.5 - 77.2	48.3	7.9 - 80.7	49.1

Total phosphorus

Event	Start time/date	Duration sampled (hrs)	Total P reduction (%)			
			Concentration (instantaneous)		Load (hourly)	
			Range	Mean	Range	Total
1	20/04/2012 11:45	11	15.3 - 55.8	29.2		
2	26/04/2012 09:15	21	10.9 - 71.5	30.5		
3	10/05/2012 04:45	15	9.8 - 44.3	27.7		
4	06/07/2012 22:30	12	-29.7 - 41.5	24.9		
5	05/08/2012 10:15	9	5.7 - 38.4	23.5		
6	24/09/2012 11:00	24	13.1 - 41.4	25.3		
7	12/10/2012 03:00	13	9.2 - 32.2	21.6		
8	22/11/2012 16:00	9	13.5 - 59.6	26.6	59.6 - 74.1	67.3
9	20/12/2012 09:00	20	-9.7 - 48.6	18.7	29.8 – 48.6	18.2
10	27/01/2013 09:30	24	8 - 50.6	24.7	16.1 – 50.6	22.0
11	18/03/2013 00:30	16	12.4 - 54.0	29.8	8.5 - 60.9	29.9

Soluble reactive phosphorus

Event	Start time/date	Duration sampled (hrs)	Soluble reactive P reduction (%)			
			Concentration (instantaneous)		Load (hourly)	
			Range	Mean	Range	Total
1	20/04/2012 11:45	11	6.4 - 37.7	19.8		
2	26/04/2012 09:15	21	-3.2 - 53.7	13.8		
3	10/05/2012 04:45	15				
4	06/07/2012 22:30	12	-5.6 - 39.4	20.4		
5	05/08/2012 10:15	9	-6.5 - 28.2	10.1		
6	24/09/2012 11:00	24				
7	12/10/2012 03:00	13				
8	22/11/2012 16:00	9	-8 - 42.3	15.3	42.1 - 72.3	62.7
9	20/12/2012 09:00	20	-16.7 - 40	13.1	17.1 – 40.0	12.9
10	27/01/2013 09:30	24	-18.2 - 48.6	13.1	2.4 – 48.6	9.9
11	18/03/2013 00:30	16	-46.9 - 53.8	13.3	-43.4 - 59.5	17.7

Nitrate

Event	Start time/date	Duration sampled (hrs)	Nitrate reduction (%)			
			Concentration (instantaneous)		Load (hourly)	
			Range	Mean	Range	Total
1	20/04/2012 11:45	11	-0.9 - 29.6	9.2		
2	26/04/2012 09:15	21	-3.6 - 11.4	4		
3	10/05/2012 04:45	15	2.2 - 42.9	8.9		
4	06/07/2012 22:30	12	-8.7 - 9.7	-0.8		
5	05/08/2012 10:15	9				
6	24/09/2012 11:00	24				
7	12/10/2012 03:00	13				
8	22/11/2012 16:00	9	-0.8 - 24	6.8	32.6 - 60.9	56.7
9	20/12/2012 09:00	20				
10	27/01/2013 09:30	24	-6.4 - 12.2	3.9	4.4 – 12.2	3.7
11	18/03/2013 00:30	16	-10 - 21.6	4.7	-9.8 - 23	4.4

Appendix J – Published journal papers

Barber, N.J. and Quinn, P.F. (2012) 'Mitigating diffuse water pollution from agriculture using soft-engineered runoff attenuation features', *Area*, 44(4), pp. 454-462.

Wilkinson, M.E., Quinn, P.F., Barber, N.J. and Jonczyk, J. (2014) 'A framework for managing runoff and pollution in the rural landscape using a Catchment Systems Engineering approach', *Science of The Total Environment*, 468-469, pp. 1245-1254

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AREA
Area (2012) 44.4, 454–462 doi: 10.1111/j.1475-4762.2012.01118.x

Mitigating diffuse water pollution from agriculture using soft-engineered runoff attenuation features

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Runoff attenuation features (RAFs) are low-cost, soft-engineered catchment modifications designed to intercept polluted hydrological flow pathways. They are used to slow, store and filter runoff from agricultural land in order to reduce flood risk and improve water quality, specifically by mitigating diffuse water pollution from agriculture. This study focuses on a sub catchment (30 ha) of the Belford Burn catchment (5.7 km^2) where the capacity of two RAFs to reduce concentrations of suspended sediment (SS), phosphorus (P) and nitrate (NO_3) in runoff has been investigated. A field bund RAF, designed to intercept overland flow during storm events, has been shown to retain significant volumes of sediment; however, the underlying field drains are still exporting high concentrations of sediment and nutrients, sometimes exceeding 500 mg SS l^{-1}, 1 mg TP l^{-1} and 40 mg NO_3 l^{-1}. An on-line sediment pond is accumulating sediment during normal flow conditions, but event sampling has revealed a lack of retention of any pollutants during storm events, which has been attributed to remobilisation of previously deposited material. In order to address these problems and improve the quality of the water leaving the sub catchment, a novel multi-stage RAF has been constructed in the ditch network. A low-cost filter trap, using wood chippings, has been installed and will be the focus of on-going monitoring and investigations. The ability to help tackle flooding and pollution by managing runoff flow pathways does have great potential, despite being somewhat difficult to evaluate.

Key words: *diffuse pollution, mitigation, agriculture, catchment, sediment, nutrients*

Introduction

The EU Water Framework Directive (2000/60/EC) has made the abatement of diffuse water pollution from agriculture (DWPA) a priority. Cuttle *et al*. (2007) provides a comprehensive summary of management measures, the vast majority of which focus on 'source' and 'mobilisation' management. However, despite these efforts, the operation of preferential flow pathways, both surface (Sharpley 2002) and sub-surface (Chapman *et al*. 2003; Deasy *et al*. 2009), combined with the occurrence of heavy precipitation, will always lead to 'incidental' nutrient (phosphorus – P and nitrate – NO_3) and suspended sediment (SS) losses that may enter a watercourse unchecked. Thus, the principle aim of 'transport' management options is to intercept polluted runoff, normally downstream of a known contaminant source or upstream of a sensitive receptor, and improve the quality of that water through a combination of physical, chemical and biological processes.

Probably the best-known example of a 'transport' management option is the constructed wetland. Mitsch and Gosselink (2007, 4) described wetlands as 'the kidneys of the catchment' because they have the capacity to attenuate water flows and improve water quality. Wetlands positioned strategically within a farmscape can intercept and filter agricultural runoff (Kadlec *et al*. 2000; Braskerud 2001 2002) as well as being able to provide numerous secondary benefits, which include: flood storage, groundwater recharge, new wildlife habitat and aesthetic value (Díaz *et al*. 2012). Fisher and Acreman (2004) collated the results of 57 wetland studies from around the world and concluded that 80 per cent of wetlands reduced NO_3 loading, while 84 per cent reduced P loadings in the water flowing through them. However, a huge variation in wetland performance has been reported across the literature. Mitsch and Gosselink (2007) also reviewed the results from a number of wetland studies and reported a NO_3 retention range of 40–95 per cent, while for P a much greater variation ranging from 0 per cent (in some cases a net loss was recorded) to 99 per cent retention. The use of wetlands for the mitigation of DWPA has been relatively limited in the UK to date. However, the research carried out in countries such as Norway and Sweden provides strong evidence that wetlands and similar features such as ponds have the potential to deliver cost-effective water quality amelioration.

Kay *et al*. (2009, 72) reviewed agricultural stewardship measures in England in terms of their efficacy to reduce DWPA and concluded that there was a 'striking lack of scientific evidence' on which to base political catchment management decisions. In response to this, and with the WFD requiring urgent action, significant investment has been made. The Department of Environment, Food and Rural Affairs (Defra)-funded Mitigation Options for Phosphorus and Sediment (MOPS) project was initiated to gain evidence on the effectiveness of different DWPA mitigation options. MOPS 1 (2005–2008) focused on in-field mitigation options for winter cereals (Deasy *et al*. 2010), while MOPS 2 (2008–2013; see http://mops2.diffusepollution.info) will assess the use of edge-of-field wetlands. Based on two years of data, collected after the construction of ten unlined wetlands, sediment trapping rates of 0.01–0.07 t ha yr^{-1} at a clay soil site, 0.02–0.4 t ha yr^{-1} at a silt soil site and >0.5 t ha yr^{-1} at a sandy soil site have been reported (Ockenden *et al*. 2012). Phosphorus retention was also found to be highest at the sandy soil site, with P trapping rates ranging from 0.006 to 1 kg ha^{-1} yr^{-1} across all ten sites in the first year.

Runoff attenuation features

Runoff attenuation features are soft-engineered, low-cost, catchment modifications that include bunds, ponds, traps, leaky dams, physical filters and wetlands in order to slow, store and filter runoff from agricultural land (Quinn *et al*. 2007). They are multi-functional in that they can be developed to reduce flood risk (e.g. Nicholson *et al*. 2012; Wilkinson *et al*. 2010), improve water quality by reducing DWPA (Jonczyk *et al*. 2008), and also create new habitats and increase biodiversity if an integrated approach to land management is taken (Shaw *et al*. 2010). In 2006 the 'Proactive' research group[1] set up a project at Nafferton Farm (294 ha) in Northumberland, to demonstrate full-scale water-quality amelioration RAFs. Quinn *et al*. (2007) reported a reduction in TP concentrations of approximately 40 per cent from a combined sediment trap and phosphorus trap during a number of average-sized storms. However, directly following the installation of the P trap (ochre pellets were used to chemically bind dissolved P), high SS (>90%) and total P (TP) removal (>80%) rates were recorded (Jonczyk *et al*. 2008). This was attributed to the physical filtering performed by the ochre pellets and not by chemical processes for which it was originally intended. Removal of NO_3 was negligible, probably due to the short residence time in the feature.

The Nafferton Farm project, and its use of RAFs, demonstrated the potential to manipulate flow pathways and thus runoff regimes, at the small catchment scale; work that directly influenced the approach taken in the Belford project. The Belford project[2] was started in 2008 to alleviate flooding in the town of Belford using natural flood management techniques (Wilkinson *et al*. 2010). Since it began, the project has evolved to include the use of RAFs to address water quality issues as well as flooding. In the 2009 EA River Basin Management Plan for Northumberland, the Belford Burn ecological status was classed as 'poor'. Without any mitigation it is predicted that the watercourse will remain 'poor' (Environment Agency 2009), thus failing to meet the WFD 2015 targets. The aim of this research was to assess a number of RAFs in terms of sediment and nutrient mitigation (their efficacy as flood features is discussed elsewhere (Nicholson *et al*. 2012). This paper presents results from two RAFs as well as describing the design and construction of a new multi-stage water quality feature that will be subjected to on-going monitoring and investigation.

Materials and methods

The Belford study catchment

The Belford Burn catchment (5.7 km^2) is located upstream of the village of Belford (OS Grid Reference NU-339107), in northeast England (Figure 1a). Land use is predominantly rural, split between arable in the eastern, lower half and pastoral in the western, upper half of the catchment. The catchment bedrock is chiefly Alston formation, a mix of limestone, sandstone, siltstone and mudstone, and superficial geology is dominated by Devension till. The soil (95 per cent coverage) is Dunkeswick association, a typical stagnogley soil with fine loamy topsoil and clayey subsurface horizons (Jarvis *et al*. 1984), and described by the National Soil Resources Institute[3] as 'slowly permeable, seasonally wet, basic loams and clays with impeded drainage'. Mean annual rainfall is 695 mm (Wilkinson *et al*. 2010). The Lady's Well sub catchment (34 ha) (Figure 1b), located in the north-east of the Belford catchment, was selected to test two RAFs. Land use is a mixture of arable, principally winter wheat, and pastoral. As a result of the soil type, specifically the clayey subsurface horizons, artificial drainage has been installed in the catchment to improve the drainage. A herringbone system has been used where collector drains (6″ clay ware tile) are aligned down the main slope and the lateral drains are aligned across the slope at a slight angle to the contours (only the main collector drains are shown in Figure 1b).

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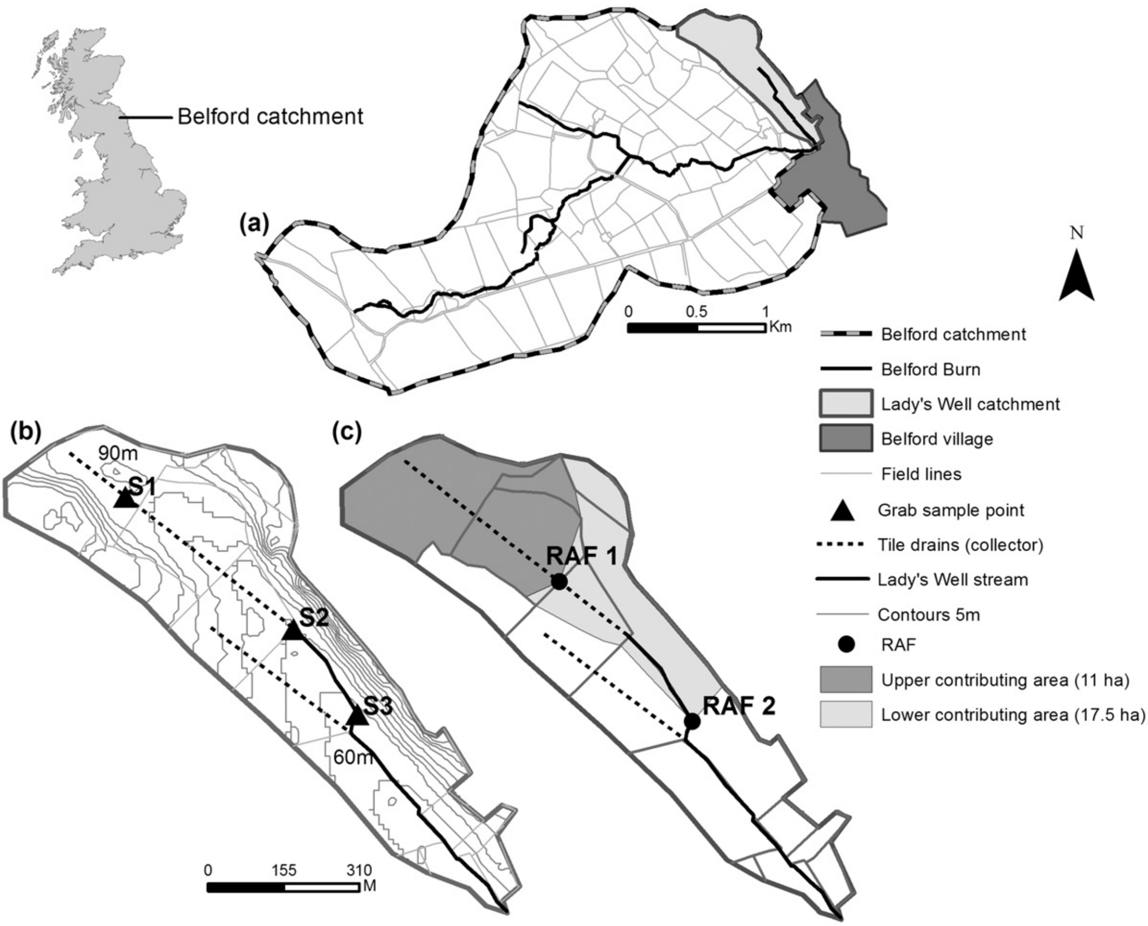

Figure 1 (a) The Belford Burn catchment; (b) the Lady's Well study catchment showing grab sample locations (S1, S2 and S3), 5-m contours, main tile drains and stream; (c) the Lady's Well catchment showing RAF locations and contributing areas

Lady's Well has an average slope of 4.3 per cent, falling 55 m over a distance of 1270 m.

The water quality in Lady's Well was characterised in 2010/2011 with 18 grab samples taken on six occasions from three locations (S1, S2 and S3; Figure 1b). S1 was from a drain inspection point at the top of the catchment; S2 was where the field drain discharges into a surface ditch (the beginning of the stream with a contributing area of 15 ha), and S3 from the stream 250 metres downstream of S2 (17.5 ha contributing area). At this time no flow measurement equipment was installed in the sub catchment so the data only exist as concentrations. No samples were collected during the summer, because there was no observed drain flow; sampling was resumed in October 2010. Together the grab samples and event recordings (described below) allowed the potential of the feature to mitigate pollution to be evaluated.

Runoff attenuation features

RAF 1 (for location see Figure 1c), constructed in November 2010, is a field bund designed to intercept and temporarily store surface runoff during storm events (Plate 1). Although this arable field is under-drained, overland flow was known to occur during heavy precipitation and act as a fast hydrological flow pathway. The bund was constructed across the main drainage thalweg with a maximum height of 1 m; this provides a storage capacity of approximately 500 m^3, 0.45 per cent of the 11 ha contributing area. The bund has a 220 mm diameter outlet pipe installed at mid-height to help prevent over-topping and possible erosion of the bund. It also helps to drain the feature in several hours; this is important in the event of a second flood peak. The RAF also doubles as a raised farm track, which prevents the farmer trafficking this previously water logged area.

Plate 1 Field bund RAF in operation during a storm

Plate 2 RAF 2 sedimentation pond during a storm event

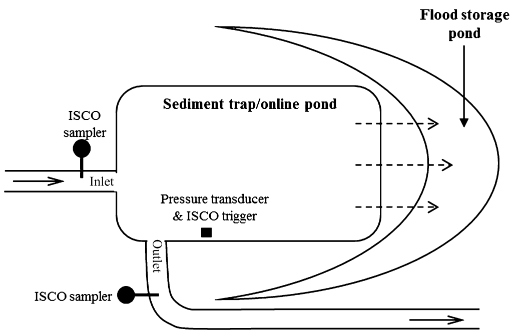

Figure 2 A schematic of RAF 2 (not to scale)

The long-term accumulation of sediment in RAF 1 is difficult to quantify because any retained sediment is intentionally ploughed back in on an annual basis. However, Palmer (2012) carried out a survey of deposited material following an event in January 2011 when the field had a low crop cover, meaning significant areas of bare soil were exposed. An estimated 0.99 tonnes of sediment had been retained in this single event and consisted of clay/silt and fine-sand sediment fractions.

RAF 2, also constructed in November 2010, has two components (Figure 2): a permanent on-line pond feature to retain sediment (Plate 2) and a higher level separate crescent-shaped pond to store flood water once a stage threshold has been exceeded and the pond spills. The sediment-trapping pond has a capacity of approximately 200 m^3, 0.11 per cent of the 17.5 ha contributing area. After construction the pond quickly began to fill with sediment and a delta could be seen developing at the inlet. A pressure transducer was installed in the pond to record stage at five-minute intervals and paired ISCO automatic samplers were deployed on the inlet and outlet of the feature in 2010. The samplers were programmed to take a sample every hour. A float switch located next to the pressure transducer initiated sampling.

Laboratory methods

All samples were analysed for suspended sediment (SS), nitrate (NO_3), total phosphorus (TP), soluble reactive phosphorus (SRP) and total soluble phosphorus (TSP) concentrations. Suspended sediment concentrations were determined using a standard method of filtration and drying at 105°C to a constant weight. Nitrate concentrations were determined using ion chromatography (Dionex 100) after filtration using a 0.45 μm cellulose acetate membrane filter. Phosphorus concentrations were determined by the colorimetric molybdate-blue method (British Standards Institute 1997); SRP after filtration using a 0.45 μm cellulose acetate membrane filter, TP after digestion with peroxodisulphate and TSP after filtration and digestion.

Results

Grab sample campaign

Table 1 contains the results from the grab sampling survey. There is a clear increase in all determinands between S1 and S2.

Sediment and TP concentrations fell slightly, on average, between S2 and S3, while SRP and NO_3 increased slightly. This suggests that water from a subsurface pathway may be entering the ditch, but this would take further investigation to verify. On every occasion, TP concentrations exceeded the EA recommended maximum concentration of 0.1 mg l^{-1} and SS concentrations as high as 400 mg l^{-1} that significantly surpass the 25 mg l^{-1} acceptable threshold prescribed under the Freshwater Fish Directive (2006/44/EC) were recorded. The data strongly suggest that the field drains transfer a significant proportion of polluted runoff; these findings are consistent with other studies (Deasy *et al.* 2009). Although the

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Table 1 Grab sample data collected from three locations in Lady's Well between January 2010 and February 2012 with sample mean and standard deviation (refer to Figure 1b for sample locations)

	Concentration (mg l^{-1})											
	SS			**TP**			**SRP**			**NO_3**		
Date	S1	S2	S3	S1	S2	S3	S1	S2	S3	S1	S2	S3
22/01/2010	12.4	297.5	284.0	0.054	0.661	0.621	0.021	0.256	0.215	6.8	10.5	9.2
04/03/2010	5.6	92.5	84.0	0.031	0.192	0.221	0.027	0.121	0.146	8.5	12.0	10.9
19/10/2010	44.0	400.8	283.6	0.062	0.249	0.329	0.054	0.110	0.132	14.7	40.8	44.6
09/11/2010	10.6	475.0	302.2	0.065	0.488	0.449	0.038	0.122	0.143	5.6	8.8	9.4
17/01/2011	10.7	163.6	75.0	0.056	0.286	0.168	0.033	0.054	0.054	13.7	15.8	16.3
07/02/2011	12.5	312.2	305.0	0.042	0.388	0.394	0.028	0.187	0.195	6.6	9.5	9.9
Mean	16.0	290.3	222.3	0.052	0.377	0.364	0.034	0.142	0.147	9.3	16.2	16.7
SD	14.0	142.9	111.0	0.013	0.174	0.164	0.012	0.070	0.056	3.9	12.3	13.9
N	6	6	6	6	6	6	6	6	6	6	6	6

majority of the grab samples were collected during rainfall events and soil conditions may have been nearing saturation, there was no overland flow in the catchment on these occasions. In a situation such as this where field drains have been identified as an important transfer route for sediment and sediment-phase nutrients, the number of mitigation options available are limited. Perhaps the most effective would be the reversion of land use from arable back to grass, but this may be unfavourable to the farmer. Another option, which would allow for the continuation of arable land use, would be to 'treat' the runoff as it leaves the drains and enters the ditch network. This idea is developed further in the Discussion section.

Evaluation of RAF 1

Palmer (2012) estimated that 0.99 tonnes of sediment were retained in RAF 1 during an event on 11 January 2011, the equivalent of 91 kg ha^{-1}. This is evidence that the feature is working to retain sediment; in this case the dominant fractions were clay/silt and fine-sand. However, two issues occurred during large events; the discharge pipe in the bund meant that a certain amount of water was allowed to flow into the next field, and the tile drain that underlies this feature was quickly surcharged, resulting in fast, highly erosive overland flows that transferred any pollutants not retained behind the bund to the lower ditch system. It would be preferable therefore to construct a second bund RAF in the next field to help address this issue. However, construction was not allowed because of planning restrictions.

Evaluation of RAF 2

Figure 3 shows inlet and outlet ISCO data from a 19-hour event on 26 February 2010. Antecedent conditions were wet following steady rainfall (22 mm over the previous seven days) before 21 mm of rain fell on the 26 February, with a peak rainfall intensity of 3 mm h^{-1}.

At the onset of sampling both SS and TP concentrations were slightly higher at the inlet than the outlet, but only for the first three hours. After this, concentrations were higher at the outlet. SS peaked at 530 mg l^{-1} (at the outlet), which coincided with the maximum pond stage after four hours. Total P concentration peaked just after the maximum pond stage at 1.22 mg l^{-1} (inlet), but the highest TP concentration of 1.24 mg l^{-1} was recorded two hours later at the outlet, at the start of the recession. Suspended sediment concentrations remained higher at the outlet until the 11th hourly sample, after which higher concentrations were recorded at the inlet. A similar pattern occurred with TP. Nitrate concentrations differed very slightly between the inlet and outlet for the entire sampling duration. The overall pattern was a slight reduction during the rising limb, followed by a steady increase; this suggests a dilution effect followed by possible leaching of NO_3 causing concentrations to increase during the recession. The NO_3 concentration of 11 mg l^{-1} may not represent the peak because higher concentrations were recorded in the grab sample campaign (Table 1). In this instance the sampling sequence was stopped because of a fall in pond stage, therefore less of the recession was recorded. Overall, during this event, there was a net loss of SS and NO_3 (2.3 and 2.5 per cent respectively), and a small 1.6 per cent net retention of TP; these percentages are based on concentrations alone as loads could not be calculated. Clearly higher downstream pollution levels contradict the initial design goal of the feature. If this effect is common in ponded features it may suggest that this type of RAF needs further development work.

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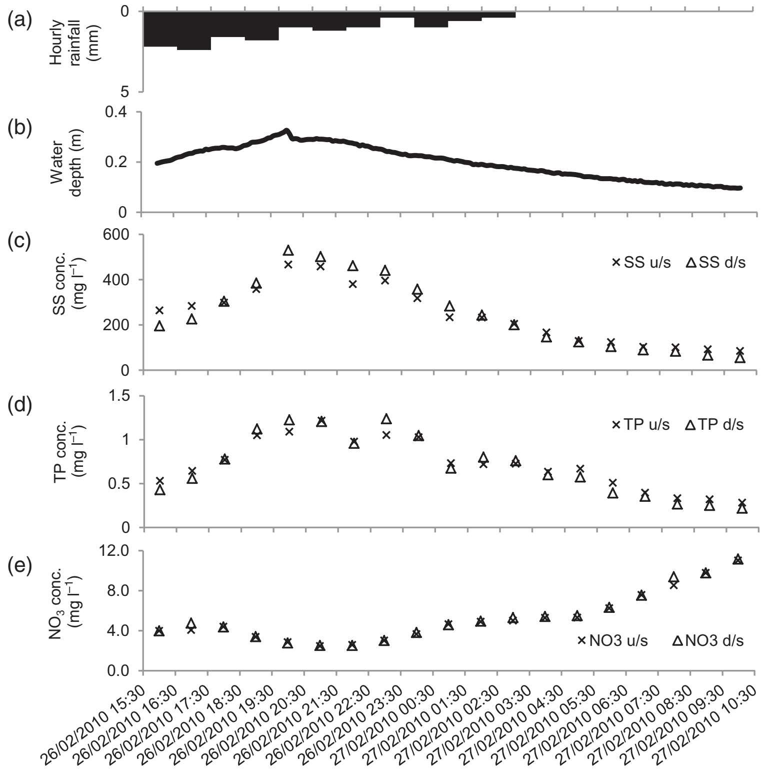

Figure 3 ISCO data recorded during an event (26–27 February 2010) at RAF 2 showing: (a) rainfall, (b) pond stage, and upstream/downstream concentrations of: (c) SS, (d) TP, (e) NO_3

Discussion and future work

Event data from RAF 2 suggest that the feature was not functioning to retain significant levels of sediment and nutrients during storm events, although more evidence, including calculation of pollutant loads, is needed to confirm this. However, it was observed that the feature was accumulating significant amounts of silt throughout 2010, particularly at the inlet. Johannesson *et al*. (2011) found a similar pattern occurring in a wetland in Sweden where sediment thickness was over four times higher at the inlet; they also found that the P content of that sediment corresponded to almost 80 per cent of the P load. In RAF 2 it is hypothesised that the bulk of the sediment comes from chronic runoff delivered by the field drains and deposited in the pond in small events. Where higher concentrations have been recorded at the RAF outlet than at the inlet, during the rising limb and at peak runoff, it would appear that previously deposited material is being remobilised. This could be a fundamental problem with using ponds as pollution traps. To overcome the problem, rapid removal of sediment from the trap provides one possibility but the frequency would make this impractical. Increasing the number and capacity of ponds is another option but available space is the biggest issue. Ponds that fully dry down (as in RAF 1) do offer easier opportunities to remove sediment or to plough sediment back into the field.

In response to these findings, it was thought that a suite of secondary pollution RAFs were required to target smaller events and long-term recessional flow. Drawing upon experience from the experiments at Nafferton Farm it was decided to use the ditch network itself to manage the pollution. Some of the design principles used in constructed wetlands were also considered to improve sediment and nutrient retention potential, including increasing storage volume to catchment area ratio in order to increase residence time (Kadlec *et al*. 2000) and the use of vegetation and obstructions to slow the velocity of the runoff (Braskerud 2001) and promote particle settlement (Uusitalo *et al*. 2003). It was decided to build a feature, or set of features, specifically to retain the fine sediment (<125 μm) and reduce remobilisation. The design would use a range of easily constructed sedimentation traps and cheap filter materials.

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Multi-stage water quality RAF – ditch management trial

A multi-stage RAF was constructed in a 150-m section of ditch, upstream of RAF 2, in February 2011. The first component was a sediment trap to remove the bulk of the coarse sediment (>125 μm), followed by a series of three brash filters, and finally a fine-sediment filter. Figure 4 shows the location and a schematic representation of the feature.

The sediment trap is of simple design to allow quick construction and simple maintenance. The feature has a capacity of 18 m^3. The ditch was both widened and deepened and a rock and earth bund constructed to dam the water. A 150-mm diameter riser pipe was installed in order to drain the feature from the surface to help minimise the remobilisation of previously deposited material. The pipe is situated at approximately 50 per cent of the total depth, therefore allowing extra storage when flow increases in the ditch. The sediment trap was partially lined with 600 × 600 mm concrete slabs to provide a solid bottom to aid sediment recovery by mechanical digger and also enable the measurement of sediment accumulation. Accumulation will be measured at the inlet and near the outlet using incremented measuring posts. The volume of retained sediment will be calculated by multiplying the surface area with the mean sediment thickness. Additional samples will be collected using steel cylinders for dry density determination. Three woven willow check dams were installed in the channel downstream of the sediment trap with brash screens placed upstream and pinned into place to prevent them from being washed away. This feature will help to slow the flow in a particularly steep section of ditch and will also have a partial damming effect on the flow, causing it to back up and promote sedimentation. Once small pockets of coarser sediment are trapped, it is hoped that these may have a flocculation effect on the fine clay material, as reported by Braskerud (2002).

The final component is an in-channel fine filter feature designed to retain fine sediment (<125 μm) and associated nutrients. To achieve a high level of filtration whilst ensuring that water can pass through easily, wood chippings were used as a cost-efficient filter media. The channel was deepened and widened to accommodate around 5 m^3 of chippings that were held in place by a timber pen lined with wire mesh and supported by wooden stakes. It is important, however, that the feature does not become a sediment source during high flow events. To prevent this, a spillway channel was dug around the feature to allow water to by-pass the filter when the stream exceeds a certain stage. This does mean that high flows will not be filtered and some estimate of the proportion of events that can be mitigated may be needed. However, there should be a suitable crossover point that will allow the ditch mitigation features to focus on smaller events and long-term recession flow and the operation of overland flow interception RAFs to target higher flow and flood events. The filter media will require periodic renewal, the time-scale of which is to be established during the experiment. Wood chips can then be spread to land following removal. Rules concerning the maintenance of RAFs are still under consideration; questions such as 'who should remove the material', 'at what rate' and 'at what cost' are important to the study.

Whilst semi-quantitative information has been helpful to build the understanding of the features and aid in their design, ultimately robust water quantity and quality data will be needed to refine pollution mitigation approaches. Rectangular flumes have been constructed in the ditch upstream and downstream of the RAF and instrumented with pressure transducers to provide a continuous flow record using a measured stage-discharge rating curve. ISCO samplers collect water samples from the flumes during storms to allow the calculation of flow-weighted concentrations and pollutant loads. A third ISCO is located after the sediment trap. The ISCOs are programmed to take samples at 30-minute intervals and will be initiated by a float switch installed in the upstream flume; the samplers will take synchronised samples.

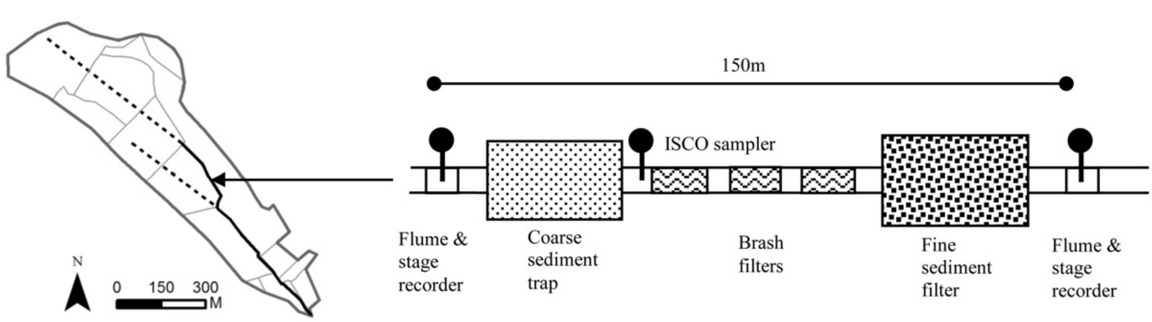

Figure 4 Schematic of the multi-stage Lady's Well RAF and its location in the catchment (not to scale)

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Conclusions and recommendations

Runoff attenuation features 1 and 2 were constructed in the Lady's Well sub-catchment principally for flood attenuation purposes, but with water quality benefits being designed in, using the prevailing evidence that features could have multiple benefits. This study began as a low-cost attempt to gain some quantitative evidence on their impact on the catchment sediment and nutrient regime, which led to some important interim results and has begged a number of fundamental questions about pollution mitigation approaches. RAF 1, the overland flow field bund, has demonstrated the potential to retain significant amounts of sediment during events. This type of feature could be recommended for construction in steep arable fields and could also be located in field corners. Despite retaining 0.99 tonnes of sediment in one event, data from the grab sample campaign strongly suggest that the field drain underlying this part of the catchment delivers significant concentrations of sediment and nutrients to the ditch during the wet season. Land use change would perhaps yield the biggest positive impact on this issue, but may not be a realistic option for the farmer. Therefore an attempt has to be made to mitigate the pollution in the surface ditch network. By tackling the pollution problem coming from the drains, a chronic issue is being addressed; however, we also need to deal with the acute, overland flow events, which can be responsible for increased concentrations of sediment and nutrients.

The implication of adding storage capacity to catchments and the requirement for filter materials is still needed. The attributes that enable RAFs and wetlands to be effective in reducing SS, NO_3 and P loadings need consideration when constructing or managing wetlands. Moreover, to provide the capacity necessary to give the residence time required to remove high levels of SS, P and NO_3 during peak flows, a catchment would require either very large features or large numbers of smaller ones. It is highly unlikely that this is a viable option for the majority of farmers. To help overcome this issue, a ditch management scheme has been designed and a physical filter is being trialled. The maintenance costs of the RAFs are still being determined and by whom and when the RAFs need to managed is not fully known. Their effective use could be an important strategy for meeting the WFD DWPA requirements and perhaps a new means of assessing 'success' is needed. The ability to tackle flooding and pollution by managing runoff flow pathways does have great potential despite being somewhat difficult to evaluate. The strength of the work so far in projects such as Nafferton, MOPS and Belford is that the features have been built and trialled at full-scale on real farms. The willingness to re-design, modify and optimise features as part of the project is vital to gaining the evidence needed for policy makers and stakeholders.

Acknowledgements

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Notes

1 See the website: http://research.ncl.ac.uk/iq (Accessed 5 September 2011)
2 See the website: http://research.ncl.ac.uk/proactive/Belford (Accessed 20 September 2011)
3 See the website: http://www.landis.org.uk (Accessed 24 August 2011)

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A framework for managing runoff and pollution in the rural landscape using a Catchment Systems Engineering approach

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HIGHLIGHTS

- A framework to achieve multiple-benefit catchment management plans is presented.
- Catchment Systems Engineering is an approach that seeks to manage flow pathways.
- Mitigation measures have been created that slow, store and filter catchment runoff.
- Several measures have been optimised for reducing diffuse pollution from agriculture.
- Results suggest that optimised features are reducing pollutant concentrations during storms.

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ABSTRACT

Intense farming plays a key role in increasing local scale runoff and erosion rates, resulting in water quality issues and flooding problems. There is potential for agricultural management to become a major part of improved strategies for controlling runoff. Here, a Catchment Systems Engineering (CSE) approach has been explored to solve the above problem. CSE is an interventionist approach to altering the catchment scale runoff regime through the manipulation of hydrological flow pathways throughout the catchment. By targeting hydrological flow pathways at source, such as overland flow, field drain and ditch function, a significant component of the runoff generation can be managed in turn reducing soil nutrient losses.

The Belford catchment (5.7 km^2) is a catchment scale study for which a CSE approach has been used to tackle a number of environmental issues. A variety of Runoff Attenuation Features (RAFs) have been implemented throughout the catchment to address diffuse pollution and flooding issues. The RAFs include bunds disconnecting flow pathways, diversion structures in ditches to spill and store high flows, large wood debris structure within the channel, and riparian zone management.

Here a framework for applying a CSE approach to the catchment is shown in a step by step guide to implementing mitigation measures in the Belford Burn catchment. The framework is based around engagement with catchment stakeholders and uses evidence arising from field science. Using the framework, the flooding issue has been addressed at the catchment scale by altering the runoff regime. Initial findings suggest that RAFs have functioned as designed to reduce/attenuate runoff locally. However, evidence suggested that some RAFs needed modification and new RAFs be created to address diffuse pollution issues during storm events. Initial findings from these modified RAFs are showing improvements in sediment trapping capacities and reductions in phosphorus, nitrate and suspended sediment losses during storm events.

1. Introduction

Intensive farming practices have the potential to increase local runoff rates, resulting in various water quality issues and local flooding problems (e.g. O'Connell et al., 2004, 2007; Parrott et al., 2009). Reducing diffuse pollution caused by agricultural activities is a major challenge in many European catchments where the sustainability of the ecosystems and water uses is compromised by intensive agriculture (Laurent and Ruelland, 2011). The European Community Water Framework Directive (WFD: 2000/60/EC) has highlighted the issues of diffuse pollution and is intended to foster the improvement of the ecology and amenity value of the UK surface waters. A central issue is excess nutrient inputs from agriculture and households to surface waters, leading to eutrophication (Hilton et al., 2006; Neal et al., 2008), which is still the most significant reason for water bodies failing to achieve good ecological

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status (GES) in 2015 and beyond (Scheuer and Naus, 2010). Agriculture in the EU contributes circa 70% of the suspended sediment (SS), 40–80% of the nitrate (NO_3) and 20–40% of the phosphorus (P) entering surface waters (OECD, 2001). Despite the 2009 deadline for the adoption of river basin management plans (RBMPs) and programmes of measures (PoMs) to meet GES by 2015, it is estimated by the European Environment Agency (2012) that only 52% of European water bodies will meet GES by 2015, with diffuse pollution from agriculture being the significant pressure for 40% of European water bodies. River basin management plans and PoMs, as key tools of WFD, take into consideration the variability in ecosystem characteristics and as such a catchment-specific approach for implementation of mitigation measures is required (Johnes et al., 2007; Doody et al., 2012). The latest review of the RBMP by the European Commission in 2012, recommended that member states should "*step up ambition in taking measures to achieve good status*" and "*in case of uncertainties in effectiveness, take no-regret measures*" (European Commission, 2012b). It is thus becoming more expedient in agricultural catchments to implement the type of management necessary to reduce sediment and nutrient losses to standing water bodies (Jordan et al., 2007).

The Floods Directive (2007/60/EC) has created a platform for the management of flood risk with aims to reduce the adverse consequences to human health, economic activity, the environment and cultural heritage associated with floods. Flooding is a significant hazard in England and Wales with approximately 1.85 million homes, 185,000 commercial properties, circa 5 million people, and half of the most productive agricultural land at some risk from flooding (Parker, 2000; Environment Agency, 2001; Penning-Rowsell et al., 2006). There is widespread concern that shifts in extreme weather events associated with climate change could exacerbate damages globally or even reverse development gains in some regions (UNDP, 2007; Wilby and Keenan, 2012). Sustainable flood risk management measures need to have a prominent role in the implementation of the directive. In the UK, sustainable flood risk management embodies a shift from a traditional, predominantly piecemeal and reactive method towards a catchment-based approach that takes account of long-term social and economic factors, which uses natural processes and natural systems to slow down and store water (Scottish Environment LINK, 2007).

Methodologies for mitigating water quantity and quality share many commonalities and these aspects should be considered together in order to maximise benefits and persuade local actions. This idea is embraced in policy within the *Blueprint to safeguard Europe's waters* document (European Commission, 2012a); however, it is rarely applied in practice. Achieving water quality targets at minimum economic cost is one of the underlying principles driving the selection of mitigation measures (Balana et al., 2012). However, there is an urgent need to tackle multiple issues in a holistic way, whilst delivering more for less and demonstrating the impact at the catchment scale. The current financial constraints throughout Europe means that multi-objective measures to meet the targets of the above directives are critical and action needs to be taken. The commonality between many western European catchments is in the intensity of the farming, the wide range of recognised environmental concerns, a highly regulated governance regime and a vulnerability to climate and demographic changes. Hence, there is potential for agricultural management to become a major part of improved strategies for controlling runoff for better water quantity and quality.

This paper presents a case study for which a framework has been developed for implementing multi-purpose measures in the Belford Burn catchment, Northumberland, UK. It provides a guide to implementing measures in the catchment, which could be easily applied to other catchments of a similar scale. The key to this uptake is to have a demonstration catchment that has soft engineering features imposed on it and to show stakeholders and regulators how the benefits were achieved and at what cost. The paper presents the framework methodology and a description of these steps. In the final two steps provisional assessment and discussion of the data are performed.

2. Study area and catchment issues

The Belford Burn catchment (5.7 km^2) lies in Northumberland in the northeast of England and drains through the village of Belford (OS Grid Reference NU-339107). The stream flows into Elswick Burn, which then drains into Budle Bay (~30 km^2). Over recent decades, increasing summer blooms of macrophytic algae (mainly *Enteromorpha*/*Ulva intestinalis*) have occurred in Budle Bay (Palmer, 2012). This is a concern as it forms part of the Lindisfarne Special Protection Area protected under the Birds Directive (79/409/EEC), and is also designated a Natura 2000 site and Ramsar wetland. The River Water Body WFD Ecological Status in 2009 classed the Belford Burn as poor; without appropriate mitigation it is predicted to remain so in 2015 (Environment Agency, 2009) and therefore fail the WFD targets. Table 1 indicates that average annual (2006–2009) reactive P (RP) concentrations consistently exceeded levels prescribed under the WFD, whilst other water quality determinands were below recommended thresholds. The main sources of water pollution were identified by the Environment Agency for England and Wales (EA) as agricultural diffuse pollution and domestic septic tanks.

The headwaters of the Belford catchment are predominately pasture and cultivated grasslands. Grasslands and arable land dominate the lowlands. The topography is relatively steep (elevation change of 150 m over 4 km river length), which is a contributing factor to the flashy response to heavy rainfall. Belford has a long history of flooding; the 2002 flood caused damage to a number of properties and businesses, which culminated in the EA commissioned flood defence pre-feasibility study (see Halcrow, 2007). The analyses concluded that traditional flood defences were not suitable for Belford because of the high-cost, lack of space for flood walls and banks, and the small number of properties at risk; therefore the town did not meet the criteria for Grant-in Aid funding. This situation is typical for many small rural villages that are at risk of flooding. Five months after the pre-feasibility study was published, the July 2007 storm occurred and caused flooding to more than 10 properties. The feeling in the village community was highlighted by the local press headline "Sick of sandbags and sympathy" (12th July 2007, Northumbrian Gazette). Owing to the high costs of traditional flood defences, there was a desire by the local EA Flood Levy Team and the Northumbria Regional Flood Defence Committee at the EA to deliver an alternative catchment-based solution to the problem (Wilkinson et al., 2010b). The original work for this study was carried out at the farm-scale (Nafferton Farm ~1 km^2) where mitigation measures were installed for water quality management purposes (Quinn et al., 2007; Jonczyk et al., 2008; Shaw et al., 2011).

3. Methodology: A runoff management framework

Applying upstream multi-purpose mitigation measures in Belford was a new approach to flood risk management for the local EA Flood Levy Team, whilst the primary goal was to reduce the risk of flooding in Belford secondary objectives which included: to work with the community to design, locate and construct measures; to gain evidence from the measures to investigate their effectiveness in reducing flood risk; and to assess the multiple environmental benefits (e.g., modifying flood measures to be better adapted to reduce diffuse pollution).

Table 1
Average (from 36 samples) yearly ammonia, dissolved oxygen, nitrate (N) and phosphate (P) levels in Belford Burn at Ross Law, 2 km downstream of the village of Belford.
(Source: Environment Agency http://maps.environment-agency.gov.uk/wiyby/ accessed December 2010.)

Average	2009	2008	2007	2006
Ammonia (mg $l-^1$)	0.125	0.116	0.101	0.094
Dissolved oxygen (%)	95.58	95.75	95.78	97.47
Nitrates (mg $l-^1$)	22.43	22.89	23.05	23.68
Phosphates (mg l^{-1})	0.13	0.12	0.1	0.16

However, the framework methodology could be used in a catchment where diffuse pollution is the primary issue and these features could be adapted to offer flood reduction benefits. Evidence is vital for influencing and informing policy and creating a local catchment plan. In order for the lessons learned in the Belford catchment to be transferable to other catchments, a runoff management framework has been developed. The framework is based around implementing mitigation measures that target crucial pathways, engaging with catchment stakeholders and using evidence from field science and effective management protocols. Stakeholder engagement is the foundation of the framework (Fig. 1). Ensuring that all stakeholders are well informed and can all actively contribute to solving the problems will lead to greater stakeholder confidence and better outcomes being reached (Collins et al., 2012). Only through engagement with stakeholders' concerns can research output lead to improvements in farming practice and realistic policy (Hewett et al., 2009). The framework aims to facilitate cross-issue communication in order to find the most holistic solution.

Fig. 1 shows the runoff management framework that was developed in the Belford case study; it is a modification of the Hewett et al. (2009) model for a multi-scale framework for the strategic management of diffuse pollution. The framework begins by identifying catchment environmental issues; these issues are subject related and are usually poorly connected in terms of communicating and achieving multiple benefits (Fig. 1; top catchment). Within catchments there are many issues that need to be resolved, but rarely is a project funded to deliver numerous benefits. The steps of the framework as shown in Fig. 1 will be described in the following sections. Finally, the loop commences again (via modification) and the long-term catchment plan evolves further; especially if the future issues are made more ambitious, for example tackling water quality and ecological issues downstream.

3.1. Step 1: The concept for catchment change — Catchment Systems Engineering approach

The objective of the *concept for catchment change* step is to come to a consensus, through engagement, on a vision for a local catchment management plan. In Belford a Catchment Systems Engineering (CSE) approach was used. CSE follows the principles of Earth Systems Engineering and management (see Allenby, 2000, 2007; Schneider, 2001; Hall and O'Connell, 2007). CSE is an interventionist approach to altering the catchment scale runoff regime and nutrient dynamics through the manipulation of hydrological flow pathways to manage water quality and quantity sustainably (Quinn et al., 2010). It seeks first to describe catchment function (or role) as the principal driver for evaluating how it should be managed in the future. The term '*systems*' in CSE relates to both the natural and human functioning of a catchment as ultimately the stakeholders must agree with the interventions proposed. The success of CSE depends upon long-term commitment, which can only be sustained by building a consensus (Hall and O'Connell, 2007).

Runoff Attenuation Features (RAFs) are a practical component of CSE. On-farm impacts can be mitigated through good land use management practices that delay or attenuate runoff (O'Connell et al., 2004; O'Donnell et al., 2011). RAFs are based on the concept of the storage, slowing, filtering and infiltration of runoff on farms, at source, by targeting surface flow pathways in fields and farm ditches (Quinn et al., 2007; Wilkinson et al., 2010a, 2010b; Barber and Quinn, 2012b; Nicholson et al., 2012; Wilby and Keenan, 2012). RAFs have the potential to create a simple multi-purpose solution, which aims to cover all the issues highlighted in Fig. 1. RAFs include bunds, drain barriers, runoff storage features (both *online* — located within the main channel; and *offline* — located adjacent to the channel), large woody debris dams, buffer strip management, and willow barriers.

3.2. Step 2: Catchment characterisation

The objective of *catchment characterisation* is to set up a monitoring platform in order to characterise the hydrological functioning of the catchment and gather evidence on the effectiveness of mitigation strategies. Moreover, evidence is required to determine the impact of the CSE approach and to underpin future management plans. Evidence is usually available in two forms: qualitative and quantitative. However, policy makers have tended to favour quantitative forms of evidence and systematic reviews of hydrological data often ignore the benefits of soft evidence (e.g. Seibert and McDonnell, 2002). It is important to characterise the catchment pre-, during and post-change. Ideally, it would be useful to have a long period of pre-change data allowing the catchment to be understood before any mitigation strategies are put in place. In many cases this is not a feasible option, especially where flood defence schemes require urgent execution. However, it is vital to

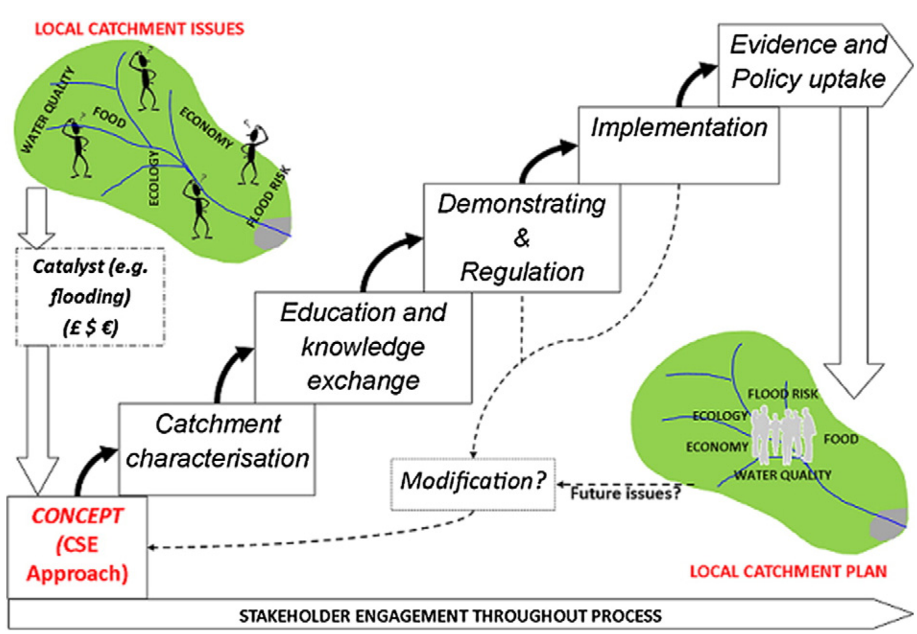

Fig. 1. The runoff management framework developed in the Belford Burn catchment.

characterise the catchment at the earliest stage possible to allow for some pre-change data to be collected; this can be achieved whilst mitigation measures are being considered. Stakeholders have a large role to play in providing expert opinion concerning the design of monitoring networks; for example, in identifying which water bodies are at risk of failing to achieve the WFD objectives and assessing the most appropriate water quality elements to monitor at suitable surveillance monitoring points (Collins et al., 2012).

A multi-scale, nested hydrometric experiment was deployed in the Belford Burn catchment in November 2007 (Fig. 2). This consisted of a rain gauge, five stream level stations and six water level recorders within RAFs. There was a desire from the EA and Newcastle University to show the multi-purpose potential of the approach. Thus, four automatic water pump samplers (ISCO 3700 and 6700) were deployed in 2009 to take samples from the stream during storm events. These were followed by a further two samplers in 2011, to monitor a ditch management RAF.

3.3. Step 3: Education and knowledge exchange

The objective of the *education and knowledge exchange* step is to share knowledge and use tools that help stakeholders understand informative concepts therefore helping to facilitate links between several issues (Fig. 1). A decision support tool was used to help accelerate the knowledge exchange process during stakeholder meetings. In academia, as well as in professional consultancy, a number of decision support systems (DSS) for river basin management have been proposed to comply with WFD, but have rarely been used by the competent authorities (de Kok et al., 2009; Klauer et al., 2012). It is important that all stakeholders are able to use and have an understanding of the DSS tool. Initially a combination of different conceptual runoff scenarios provided end users with a number of ways to visualise the effects of different land management practices. The Floods and Agriculture Risk Matrix (FARM) tool is an example of an education tool that focuses on runoff risk from farms (Wilkinson et al., 2013). The FARM tool was built around the findings of the FD2114 Defra research project (O'Connell et al., 2004) and was further refined during consultation with stakeholders in the Ripon Multi-Objective Pilot project (Posthumus et al., 2008). Primarily used at farmer meetings, with regulators in attendance, the tool reflects what stakeholders consider to be 'slow and low' and 'fast and high' runoff rates, respectively. Earlier forms of the same tool exist for pollution management including the Nutrient Export Risk Matrix (NERM) for NO_3 losses (see Quinn, 2004), and the Phosphorus Export Risk Matrix (PERM) for P losses (Hewett et al., 2004). These tools allow non-expert stakeholders to understand conceptually the underlying issues behind the problems and empower them with adequate knowledge to participate in formulating a solution.

3.4. Step 4: Demonstration and regulation

The objective at the *demonstration and regulation* step is to exhibit to catchment stakeholders and regulators how catchment intervention using RAFs will work. In Belford, a pilot/demonstration RAF site and design were agreed and constructed for this purpose (for more details refer to Wilkinson et al. (2008, 2010b)). This proved to be a long process with many regulators raising different issues about the design and location of features as well as the environmental, ecological (for example, in stream RAFs need to consider fish habitats and riparian zone features needed to consider vole habitats), and archaeological impacts of the interventions. At first, it was difficult to manage the plethora of EA advice, regulation and administration; however, the engagement with all the EA parties proved very useful and a robust solution was developed. The pilot RAF is an offline intervention (capacity ~ 1000 m^3), which consists of a 1 m high wooden bund, crossing a hollow in the landscape collecting both surface runoff and high flows spilled from the nearby stream. If the RAF is full it is allowed to overflow via a controlled spillway slot at the end of the wooden bund, reducing the risk of soil erosion from overspilling. The pilot RAF has performed well during the storms presented in Table 3. During the September 2008 event (a storm with a 24 h return period of 20 years) the pond was full at around the same time as the main peak indicating that RAF was functioning well (Wilkinson et al., 2010b). However, it is likely for a storm with a higher return period that this pond may have overtopped earlier. Information on the functioning of the pilot RAF during this event can be found in Wilkinson et al. (2010b).

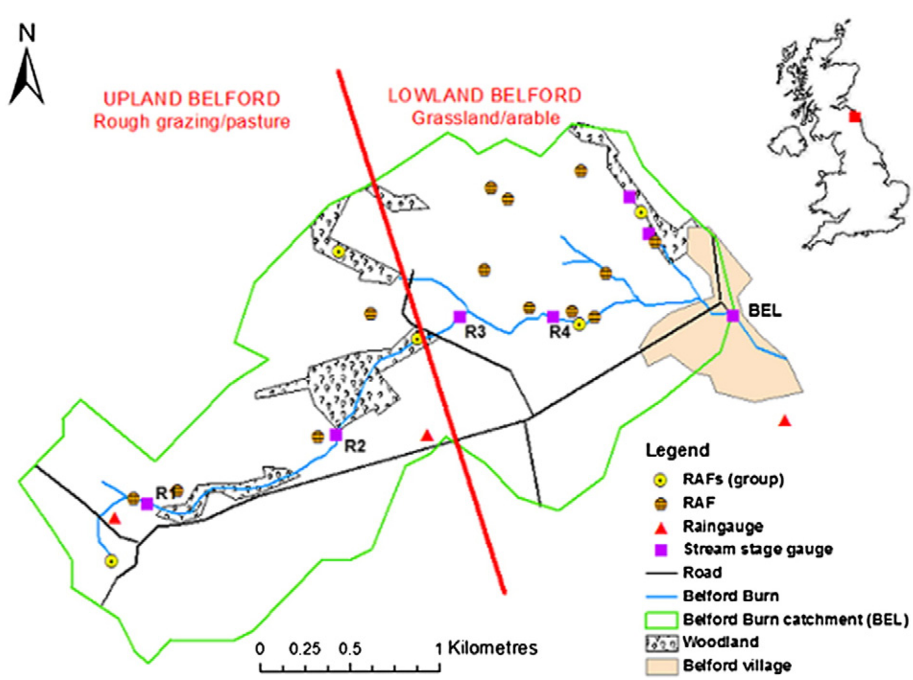

Fig. 2. The Belford Burn catchment showing the hydrometric network and Runoff Attenuation Feature (RAF) sites.

Table 2
A construction and operational summary of six instrumented RAFs in the Belford Burn catchment.

RAF number	Type	Construction and operation	Capacity (approx.) m^3
0	Offline flow storage/incepting fast overland flow pathway	Timber permeable barrier; disconnects major overland flow pathway and diverts high flows from stream	800
1	Offline flow storage	Soil bund; high flows diverted from stream	310
2	Online flow storage/wetland area	Earth dam; wetland pond with extra storage during high flows	700
3	Offline flow storage	Soil bund; high flows diverted from stream	360
4	Incepting fast overland flow pathway	Soil bund with wooden sluice; disconnects major overland flow pathway	3000
12	Online flow storage/wetland area	Soil bund around wetland pond; wetland pond with extra storage during high flows	250

3.5. Step 5: Implementation

The *implementation* step involves the construction of RAFs across the entire catchment in order to address the identified issues (Fig. 1). In a three-year period (2008–2011) over 30 RAFs were constructed in the Belford catchment (Fig. 2). A detailed summary of these features can be found in Wilkinson et al. (2010a) and Nicholson et al. (2012) (also see http://research.ncl.ac.uk/proactive/belford: accessed May 2012). Six instrumented RAFs are selected for analysis with differing construction and operation regimes. These are summarised in Table 2 and cover the broad range of features in the catchment.

Farmer participation was fundamental to the effective implementation of RAFs. Often farmers would suggest suitable sites and modify/optimise the initial design in order to gain the most advantageous environmental and agri-economic benefits. Thus, the agreed final design was based on local knowledge on current details such as land use; for example, woody debris was placed only in wooded channel sections, ponds were placed within existing riparian area buffer zones, and a range of ditch barriers were created using wood within upland ditches and willow within the lowland arable ditches. Usually, an attempt was made to solve more than one environmental problem; for example, bunds were placed across hollows in fields prone to overland flow. One bund was constructed as a farm track thus removing the problem of trafficking through a frequently saturated zone whilst also allowing the feature to be much larger. Several of the features were instrumented and assessed so that the RAF functioning could be understood. This allowed a provisional assessment of the data to take place using the methodology framework so far to understand whether the evidence was sufficient for policy uptake.

4. Results and discussion

4.1. Provisional assessment of collected data

The provisional assessment of data feeds into the evidence and policy uptake step. If the provisional evidence suggests that the RAFs could be optimised then appropriate changes could take place. Over a four year period (2008–2012) that the Belford mitigation measures have been in place, the catchment has witnessed an unusual high number of flood level storm events (Table 3).

Initially, qualitative evidence (such as photographic evidence, videos and two farmers visited some of the RAFs during the September 2008 flood) showed the stakeholders that the RAFs were clearly holding water upstream of the village despite the lack of quantitative evidence. However, the numerous storms have subsequently provided a large dataset of the RAF hydrological functioning; this is highlighted for the largest recorded flood during the project, the March 2010 event (Fig. 3; Table 3).

Fig. 3 shows the performance of six RAFs during the March 2010 event (the largest recorded during the catchment characterisation period). Fig. 3 shows that RAFs 1, 3, 4 and 12 have a peak (in water level) after the observed peak in the stream at R3 (Fig. 2). These RAFs are located nearby stream monitoring point R3. RAF 4 is one of the last features to peak and this occurs 2 h after the stream peaks (Fig. 3). This is owing to its relatively large capacity and its ability to capture a major overland flow pathway, which continues to produce runoff after the peak of the flood (as seen in Fig. 4). The pilot feature (RAF 0) peaks before the observed peak at R3, however, this site is located in the headwaters near R1 and it is likely that the peak has passed this site. Wilkinson et al. (2010b) found this feature to be functioning as intended during the September 2008 event; data indicated that the time of travel of a peak increased by 15 min over a 1 km stretch of the stream by comparing data from several storm events before and after the installation. Fig. 3 demonstrates that most RAFs are performing as specified: storing runoff at and after the peak, and then emptying within half a day of the last peak (Fig. 3). However, it is evident that RAF 2, an in-stream dam/online pond feature, reaches storage capacity some time before peak stream level, meaning that it has little-to-no effect on flow storage/attenuation during this critical part of the storm. Despite this, visual evidence suggested that RAF 2, along with other online features, was accumulating sediment. RAF 2 was surveyed in May 2010 and again 19 months later in December 2011, which revealed a reduction in storage capacity of approximately 190 m^3 (Barber et al., 2011). This could be translated into a long-term estimate of trapped sediment.

Fig. 4 shows cumulative runoff and rainfall recorded during the March 2010 event. The photograph in Fig. 4 shows that overland flow taking place before peak stage is observed in the main channel. The rainfall runoff ratio for this event was estimated at 91% (based on a runoff calculation using an extrapolation of the rating curve at R3 [Fig. 2] to estimate runoff), which could be attributed to the land drain network becoming surcharged, leading to a rapid increase in overland flow. Significant proportions of the catchment exhibit overland flow during large events, which have the potential to cause significant soil erosion and sediment (and associated nutrient) losses.

4.2. Modification and optimisation

All RAFs are under continuous review and a number of them are undergoing varying degrees of modification and optimisation (Fig. 1). A number of offline ponds have required the inlet level to be raised in order to target the peak of storms in a more timely fashion (Nicholson et al., 2012), thus ensuring a more efficient use of storage capacity. In terms of optimisation, a number of new features will be built differently based on the experience gained throughout the project; for example, despite the relative high cost of using treated timber (as an alternative to earth bunds) its versatility makes it easy to work with, its inherent strength provides resistance to the attention of livestock (particularly cattle), and it requires little in the way of space — thus having a lesser

Table 3
A summary of the top 3 extreme storm events in the Belford Burn catchment (during 2008–2011); storm return periods calculated using the Flood Estimation Handbook.

Rank	Dates	Storm duration (hrs)	Rainfall (mm)	% of yearly average rainfall	24 h rainfall return period
1st	29–30th Mar 2010	30	62.4	9	12.5 years (58.8 mm)
2nd	17th July 2009	43	102.6	15	12.5 years (58.2 mm)
3rd	5–7th Sept 2008	45	99.6	14	20 years (65.8 mm)

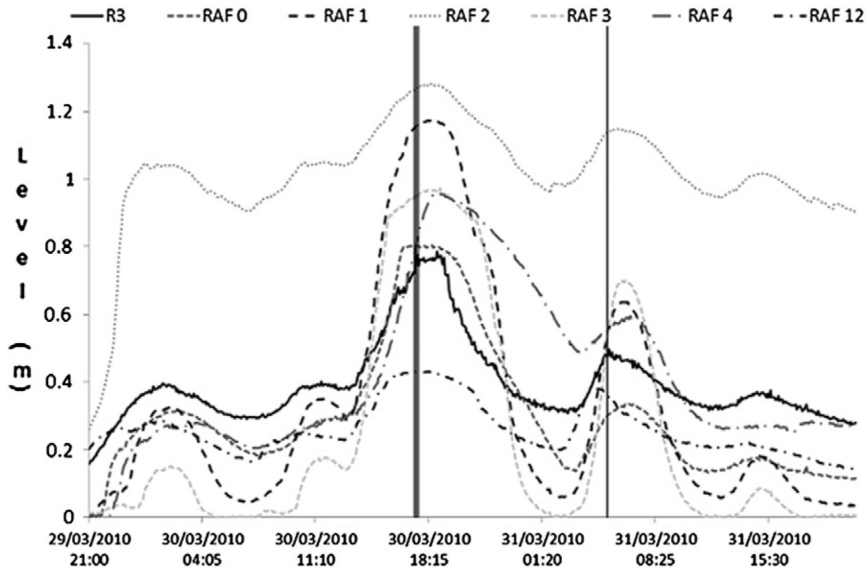

Fig. 3. The performance of six RAFs (see Table 2) during the March 2010 flood (observed at R3). RAF 0 collects runoff and stream spill, RAFs 1 and 3 collect stream spill, RAF 2 is in the stream and RAF 12 is a combination of an in-stream feature with spill overflow. Solid vertical lines indicate the peak observed in the stream at R3.

impact on agricultural activities. Soil bunds also have their merits; they can be low-cost and relatively simple to construct but suit locations with fewer livestock and where space is available to build a wider bund. Using other locally sourced materials, such as stone, from local quarries or construction sites can build much stronger RAFs that can carry the weight of vehicles. Visits from ecologists, RSPB officers and wildlife groups, inter alia have provided insight into ways the features can be further optimised to create niche habitats for valuable species. For example, ensuring that a small amount of water remains permanently in ponds can improve habitat diversity. There have been some unverified local reports of Great Crested Newts in several of the online ponds; if present this is positive in terms of biodiversity, but sediment removal from those features would be more difficult to justify and maintenance of these features will, in future, need to consider the lifecycle stage of any valuable inhabitants. Not all RAFs need to be multi-functional; for example, there may be a need to sacrifice flood storage capacity in order to enhance water quality amelioration potential.

The process of RAF modification and optimisation in order to provide multiple benefits in the Belford catchment has raised an important management issue that is summarised by the following questions:

1 Could a RAF designed for flood attenuation purposes be optimised for water quality amelioration?or
2 Would new bespoke RAFs designed specifically for diffuse water pollution management be required?

These questions will be addressed in the following section.

4.2.1. Collected evidence on the impact to water quality (based on RAF modification)

As described previously, Belford Burn (along with other streams) eventually discharges into Budle Bay, a sensitive downstream receptor. The consensus view by the EA and Natural England was that eutrophication from freshwater tributaries was causing thick mats of marine macroalgae to develop, which were a threat to benthic ecology and the large populations of wading birds (Palmer, 2012). In response, an

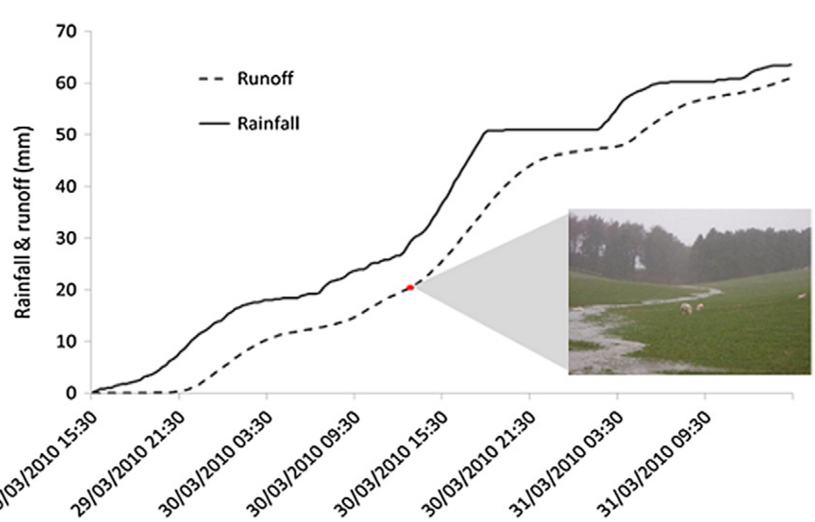

Fig. 4. Cumulative rainfall and runoff during the March 2010 flood event. The image shows overland flow occurring over the field before the main peak of the flood (red spot indicates the time that the photograph was taken in relation to the data).

investigation was begun in 2009 to establish the effectiveness of existing RAFs to reduce losses of sediment and nutrients. The investigation included a catchment-wide grab sampling campaign to characterise the sediment and nutrient regime and to identify locations that were contributing elevated levels of agricultural diffuse pollution. Four auto-samplers were also deployed at two online ponds (RAFs 2 and 12 — two samplers per feature: one directly upstream and one directly downstream). The online features were chosen for monitoring as a result of visual evidence that suggested that sedimentation was occurring. It was therefore desirable to find out whether they were retaining sediment and nutrients during storm events (please refer to Barber and Quinn (2012b) for details of the methodology used for the determination of SS, P and NO_3 concentrations).

Data from the grab sampling campaign indicated that background sediment and nutrient concentrations in the main burn, during baseflow conditions were of no ecological concern (under WFD guidelines). However, it also highlighted that certain parts of the catchment were characterised by significantly higher concentrations of SS, P and NO_3. At the outfall of one 17.5 ha sub-catchment TP concentrations exceeded the EA recommended maximum concentration of 0.1 mg l^{-1} on every sampling occasion, and SS concentrations as high as 400 mg l^{-1} were recorded that significantly surpass the 25 mg l^{-1} average annual threshold prescribed under the Freshwater Fish Directive (2006/44/EC). An average NO_3 concentration of 16.7 mg l^{-1} was recorded, with a maximum of 44.6 mg l^{-1} (Barber and Quinn, 2012b). Although below the 50 mg l^{-1} maximum concentration prescribed by the Drinking Water Directive (98/83/EC), Skinner et al. (2003) and Hickey and Martin (2009) argue that such concentrations are potentially of ecological significance. As this part of the catchment was fed principally by field drains, the data highlighted the importance of subsurface drainage as a significant conveyor of sediment and nutrients (also reported by Deasy et al., 2009), particularly during residual flow conditions when the majority of flow is being transferred by the drains.

Data collected by the auto-samplers during storm events drew attention to two important characteristics: firstly, that P and SS concentrations increased significantly above background levels; and secondly, that online ponds were not retaining pollutants during the rising limb and peak of events. Maximum TP and SS concentrations of 1.24 mg l^{-1} and 530 mg l^{-1}, respectively, were recorded during an 'average' sized event in February 2010. The paired inflow and outflow concentration data suggested that no sediment or nutrient retention occurred, particularly during the early high-flow component of the event. Based on this single event, net losses of SS, TP and NO_3 of 9%, 14.5% and 0.7%, respectively, were recorded at RAF 2 (Barber and Quinn, 2012a). Results from RAF 12 (recorded during the same event) showed that there were also net losses of SS and NO_3, 2.3 and 2.5%, respectively, but a small 1.6% net retention of TP (based on concentrations) (Barber and Quinn, 2012b). These results clearly contradict the observation that the online ponds were filling with sediment in the long term. Thus, online features appear to be functioning to reduce chronic losses of SS (and sediment-phase nutrients), but are far less effective in (acute) storm events. It is strongly suspected that remobilisation of previously deposited material is the principle problem.

Another RAF that is considered to be multi-functional is the within-field retention bund. Fig. 5 shows one example built across the main valley thalweg (the line following the lowest part of the valley) of an arable field (4.1 ha) designed to intercept and temporarily store overland flow. The field has a gradient of approximately 4° and its land cover (predominantly winter wheat during the study period) makes it highly susceptible to soil erosion. The RAF is located at the top of the 17.5 ha sub-catchment draining into RAF 12 and also doubles as a raised track. Construction was carried out by the farmer using locally-sourced materials, thus incurred relatively low cost. Whilst its ability to retain overland flow is obvious (the feature can store approximately 500 m^3 of flood water) its sediment trapping capabilities were more difficult to quantify. However, following a large runoff event in January 2011, Palmer (2012), by surveying the rills and gullys (erosion) and sediment fan left behind the retention bund (deposition), and by determining the particle size distribution and bulk density of the material, was able to calculate the mass of sediment retained by the RAF. It was estimated that 0.99 tonnes of sediment (consisting mainly of silt/clay and fine-sand) was captured but that a proportion of fine sediment was lost via the feature's outlet pipe and bypassed by sub-surface drains. Trapped sediment becomes re-incorporated back into the topsoil during annual ploughing.

In response to question 1, which asked if flood RAFs could be modified to ameliorate water quality, it has become evident that different features operate to retain pollutants under contrasting flow conditions. The flood RAFs were designed and constructed to intercept strategic pathways, either surface or subsurface and as sediment/nutrient transfer is driven chiefly by hydrology, it stands to reason that there is potential to intercept contaminants moving along the intercepted pathway. The in-field retention bund has shown the potential to reduce diffuse pollution but only functions to do so during overland flow events; although this is arguably the case when the largest pollutant loads are exported from a catchment (Haygarth et al., 2005). However, in this particular location it is apparent (according to evidence presented by Palmer (2012) and the grab sampling data described by Barber and Quinn (2012b)) that a significant proportion of the pollution is lost via the sub-surface field drains, therefore by-passing the feature. The sub-surface drains could be broken and allowed to spill into the RAF, but this would impact on the workability of the land, and could negatively impact farm operations.

Intercepting the sub-surface pathway, in the existing ditch network provides an alternative location that is more favourable to the farm. The online pond RAFs were constructed to target and 'slow and store' the subsurface pathway. They appear to retain sediment (and associated nutrients) during residual flow conditions but not during flood peaks. To improve pollutant retention, the residence time in the features could be increased by adding baffles, or introducing vegetation (as reported by Braskerud (2002)), to increase settlement time whilst not having to increase the overall RAF size. Also, in order to maximise the lifespan and water storage capacity of the online pond RAFs it would be favourable to construct upstream sediment traps to attenuate the sedimentation rate in the main ponds. Although these modifications have not been made to existing RAFs, alterations will be made to future designs based on the experience gained to further improve performance.

Concerning question 2, it was felt that a new, optimised RAF was required to meet some of the shortcomings highlighted above. A bespoke multi-stage RAF was constructed in February 2011 in a 150 m length of ditch, directly upstream of RAF 12 and approximately 500 m down the catchment from the in-field retention bund (Fig. 5). The design represents the culmination of experience gained from the Nafferton Farm and Belford projects and has the following objectives:

- mitigate polluted drain flow, *which will help to*
- reduce pollutant concentrations during residual flow conditions
- reduce remobilisation of previously settled sediment (and associated nutrients) in ponds during storm events.

The RAF consists of an upstream sediment trap, followed by a filtering system consisting of leaky willow barriers and brash screens, and a wood chip barrier/filter (Fig. 6). The feature has been instrumented with water level recorders and upstream/downstream auto-samplers to determine its performance. Initial findings in Belford suggested that it was important to create more sediment traps, especially upstream of on-line ponds to help reduce their sedimentation, thus prolonging maximum flood storage capacity. Sediment traps help to determine where in the system material is stored; therefore it is vital to ensure easy access to allow periodic emptying. Barber et al. (2011) reported that six months after construction, the sediment trap (with an area of 12 m^2) had an average sediment depth of 10 cm, giving a wet volume

Fig. 5. A field bund RAF storing water during a storm.

of circa 1.2 m^3; sediment mass and P concentration are still to be determined. The willow dams and brash screens are designed to slow the flow, reduce channel erosion, and provide a coarse level of filtration, possibly aided by flocculation (as reported by Braskerud (2002)). The wood chip filter is designed to remove fine sediment — particles less than 106 μm (fine sand, silt and clay), and associated nutrients. The use of a wood chip bioreactor is a method for removing NO_3 from drainage water by denitrification (in which NO_3 is converted to nitrous oxide and nitrogen gas). Bioreactors have been studied in Illinois and have been shown to effectively reduce NO_3 levels by 33% on average, but up to 100% during certain conditions (Woli et al., 2010). Greenan et al. (2009) also reported positive results from trials carried out in the United States, as did Saliling et al. (2007) who conducted a series of laboratory experiments using wood chips as a media for denitrification. The bioreactor is designed to 'treat' the persistent low levels of NO_3 that can have subtle but important effects on aquatic species (Earl and Whiteman, 2009).

Fig. 7 shows sediment and nutrient data taken simultaneously upstream and downstream of the multi-stage RAF during a May 2012 storm event. The average reduction in pollutant concentrations over the duration of the storm (24 h) is as follows: 40% SS, 26% TP, 25% soluble RP, and 15% NO_3. Over the course of 2012, which included several storm events of varying magnitudes and durations, inflow and outflow sampling has revealed reductions of 30–45% SS, 14–25% TP, 25–30% soluble RP and 8–38% NO_3 concentrations. Thus, although based on concentration only, these preliminary results suggest that the feature is working to reduce sediment and nutrient losses from this part of the catchment during storms. Ockenden et al. (2012) reported a 60% reduction in SS concentration during an event at a paired-pond sediment trap, which formed part of the Mitigation Options for Phosphorus and Sediment (MOPS) project. Although the MOPS feature and the multi-stage RAF had similar sized contributing areas, the MOPS sediment trap was much larger (area = 200 m^2) that may explain the higher percentage removal. Of course, many other variables can influence the retention capacity of such features but residence time is arguably one of the most important (Braskerud, 2002; Reinhardt et al., 2005).

The impact of all the RAFs on water quality is somewhat difficult to prove, as monitoring can be extremely expensive and time consuming. Also, it may take several years before any change in the sediment and nutrient regime is detected at the catchment scale (Haygarth, 2010). Therefore, management at the field- and farm-scale remains crucial to water quality outcomes and delivering on the WFD requires

Fig. 6. Wood chip filter (left) and a sediment trap (right) placed in ditch upstream of RAF 12.

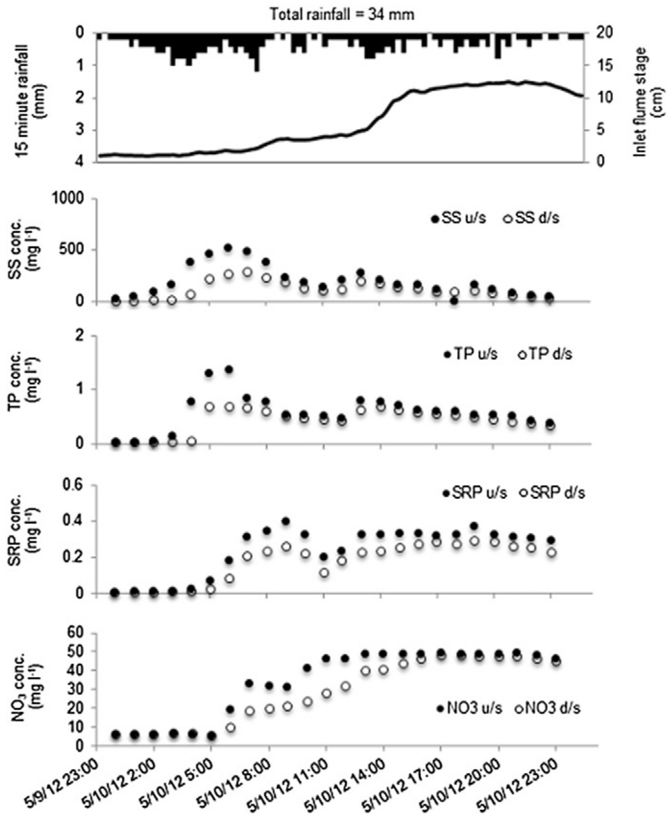

Fig. 7. Storm sample data collected up- and downstream of the multi-stage RAF (Fig. 6) during a May 2012 event.

coordination that transcends a continuum of scales (Winter et al., 2011). However, a number of underlying design criteria are being determined and some solid, local evidence is being accrued.

5. Conclusions

Many environmental issues have been identified in the Belford catchment. The motivation for this project was to reduce the flood risk in Belford whilst delivering multipurpose benefits. Natural, upstream mitigation measures have been identified by the Floods Directive and WFD to reduce flood risk and improve water quality, respectively. However, there was a desire in Belford to create a catchment plan that would have multi-purpose benefits meeting the aims for the two directives. A framework has been developed throughout the project at Belford showing how a catchment management plan can be achieved; to which the CSE approach and stakeholder engagement are the key. RAFs are at the heart of the CSE approach; they are soft-engineered structures that could potentially provide a cost-effective solution to achieving multiple benefits. However, they do not offer a single solution and should be considered alongside traditional flood defences. RAFs also require maintenance; the potential need to recover trapped sediment and whether it is of agronomic value to the farmers is part of the on-going assessment.

The framework continues to show, step-by-step, how evidence can be gathered to underpin new policy and how a catchment management plan can be achieved using the CSE approach (Fig. 1). A provisional assessment of the current data suggests that most RAFs are functioning as intended and many fast flow pathways are being intercepted. The degree to which the local catchment system has been 'engineered' is still being determined. The overall performance of these RAFs in terms of addressing pollution and ecology is also difficult to quantify and requires a further weight of evidence. As the initial findings are positive, with sediment accumulating and the creation of new ecological niches, these measures can be categorised as the 'no-regret measures' being pursued by the European Commission. In the study shown here it is important to stress the simple underlying concepts that flow pathway behaviour can be changed using soft engineering. For flood flow, and nutrient losses, the simple concept of disconnecting fast flow pathways, adding storage and attenuating flow pathways can be applied to a catchment.

Finally, local stakeholders have had a say in creating a local catchment plan. Many other stakeholders who have similar issues in their own catchments are now assessing the Belford project. The framework has been developed in Belford but it may have generic applicability to many other catchments. Belford is not unique in its issues; many other similar scaled catchments have flood risk and diffuse pollution issues. Although initial water quality impacts proved to be complicated, the dedicated sediment traps and filters are exhibiting positive impacts on sediment and nutrient losses. CSE has endeavoured to change the flood flow regime of the catchment whereby an adaptive approach is required and must continue in the future. Intervention is required at many locations but with the help of local stakeholders and regulators the potential framework for holistic environmental management has been trialled at the small catchment scale.

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Date	Discharge (m3/s)	Determinand concentration (mg/l)						
		SS	NO3	TP	PP	SRP	SUP	TSP
#########	24.9	45.0	2.466	0.066		0.018		
#########	14.6	19.2	2.237	0.045		0.02		
#########	5.71	7.0	3.215	0.040		0.014		
#########	32.1	23.0	2.517	0.055		0.018		
#########	1.06	2.0	2.144	0.026		0.016		
#########	0.962	3.5	2.158	0.031		0.011		
#########	0.408	4.0	2.668	0.021		0.01		
#########	1.31	2.2	2.148	0.027		0.01		
#########	8.59	12.9	1.970	0.032	0.021	0.007	0.004	0.011
#########	2.66	2.4	2.050	0.017	0.004	0.008	0.005	0.013
#########	0.335	2.2	2.940	0.012	0.007	0.005	0.000	0.005
#########	0.215	1.1	2.110	0.015	0.006	0.006	0.003	0.009
#########	0.161	0.8		0.022	0.005	0.012	0.005	0.017
#########	0.172	2.0	2.360	0.020	0.006	0.009	0.005	0.014
#########	0.102	0.4	2.240	0.017	0.007	0.005	0.005	0.010
#########	0.113	0.8	2.840	0.017	0.007	0.006	0.004	0.010
#########	0.55	6.4	1.550	0.025	0.005	0.010	0.010	0.020
#########	0.962	2.1	2.400	0.015	0.003	0.010	0.002	0.012
#########	0.344	1.0	1.080	0.018	0.006	0.005	0.007	0.012
#########	1.07	3.0	1.870	0.027	0.005	0.020	0.002	0.022
#########	2	14.6	1.080	0.044	0.019	0.011	0.014	0.025
#########	0.519	2.5	2.390	0.031	0.006	0.017	0.008	0.025
#########	19.2	35.8	1.550	0.039	0.013	0.019	0.007	0.026
#########	2	5.5	2.554	0.045	0.009	0.022	0.014	0.036
#########	0.621	2.1		0.042	0.016	0.019	0.007	0.026
#########	1.16	7.2	2.370	0.037	0.013	0.017	0.007	0.024
#########	2.33	6.6		0.033	0.017	0.012	0.004	0.016
#########	6.4	8.9	4.986	0.070	0.018	0.029	0.023	0.052
#########	60	44.5	1.640	0.084	0.049	0.026	0.009	0.035
#########	16.2	11.0	4.220	0.044	0.016	0.022	0.006	0.028
#########	2.72	5.0	3.540	0.024	0.010	0.010	0.005	0.015
#########	8.59	5.5	3.750	0.025	0.018	0.011	0.005	0.016
#########	1.26	5.5	5.670	0.046	0.022	0.010	0.014	0.024
#########	7.94	6.0	2.210	0.035	0.021	0.012	0.002	0.014
#########	0.413	4.8	3.650	0.022	0.011	0.005	0.006	0.011
#########	0.344	2.0	3.770	0.029	0.008	0.015	0.006	0.021
#########	2.13	4.8	3.340	0.028	0.007	0.011	0.010	0.021
#########	0.353	3.5	2.880	0.018	0.006	0.009	0.003	0.012
#########	0.666	0.7	2.440	0.020	0.005	0.011	0.004	0.015
#########	4.04	8.8	3.660	0.029	0.015	0.006	0.008	0.014
#########	0.321	2.8	2.450	0.015	0.005	0.008	0.002	0.010
#########	0.89	4.0		0.019	0.009	0.008	0.002	0.010
#########	38.8	18.5	2.030	0.058	0.033	0.020	0.015	0.025
#########	0.942	1.0	3.430	0.028	0.008	0.011	0.009	0.020
#########	49.4	68.5	1.890	0.095	0.061	0.021	0.019	0.034
#########	1.16	2.0	2.250	0.025	0.010	0.011	0.004	0.015
#########	0.556	4.0	3.380	0.022	0.013	0.005	0.004	0.009
#########	5.96	28.5	2.050	0.064	0.035	0.011	0.018	0.029

#########	1.38	3.5	0.040	0.023	0.008	0.009	0.017

Determinand instantaneous load (g/s)

SS	NO3	TP	PP	SRP	SUP	TSP
1120.5	61.40	1.643		0.448		
280.3	32.66	0.657		0.292		
40.0	18.36	0.228		0.080		
738.3	80.80	1.766		0.578		
2.1	2.27	0.028		0.017		
3.4	2.08	0.030		0.011		
1.6	1.09	0.009		0.004		
2.9	2.81	0.035		0.013		
110.8	16.92	0.275	0.180	0.060	0.034	0.094
6.4	5.45	0.045	0.011	0.021	0.013	0.035
0.7	0.98	0.004	0.002	0.002	0.000	0.002
0.2	0.45	0.003	0.001	0.001	0.001	0.002
0.1		0.004	0.001	0.002	0.001	0.003
0.3	0.41	0.003	0.001	0.002	0.001	0.002
0.0	0.23	0.002	0.001	0.001	0.001	0.001
0.1	0.32	0.002	0.001	0.001	0.000	0.001
3.5	0.85	0.014	0.003	0.006	0.006	0.011
2.0	2.31	0.014	0.003	0.010	0.002	0.012
0.3	0.37	0.006	0.002	0.002	0.002	0.004
3.2	2.00	0.029	0.005	0.021	0.002	0.024
29.2	2.16	0.088	0.038	0.022	0.028	0.050
1.3	1.24	0.016	0.003	0.009	0.004	0.013
687.4	29.76	0.749	0.250	0.365	0.134	0.499
11.0	5.11	0.090	0.018	0.044	0.028	0.072
1.3		0.026	0.010	0.012	0.004	0.016
8.4	2.75	0.043	0.015	0.020	0.008	0.028
15.4		0.077	0.040	0.028	0.009	0.037
56.9	31.91	0.448	0.115	0.186	0.147	0.333
2670.0	98.40	5.040	2.940	1.560	0.540	2.100
178.2	68.36	0.713	0.259	0.356	0.097	0.454
13.6	9.63	0.065	0.027	0.027	0.014	0.041
47.2	32.21	0.215	0.155	0.094	0.043	0.137
6.9	7.14	0.058	0.028	0.013	0.018	0.030
47.6	17.55	0.278	0.167	0.095	0.016	0.111
2.0	1.51	0.009	0.005	0.002	0.002	0.005
0.7	1.30	0.010	0.003	0.005	0.002	0.007
10.2	7.11	0.060	0.015	0.023	0.022	0.045
1.2	1.02	0.006	0.002	0.003	0.001	0.004
0.5	1.63	0.013	0.003	0.007	0.003	0.010
35.6	14.79	0.117	0.061	0.024	0.032	0.057
0.9	0.79	0.005	0.002	0.003	0.001	0.003
3.6		0.017	0.008	0.007	0.002	0.009
717.8	78.76	2.250	1.280	0.776	0.582	0.970
0.9	3.23	0.026	0.008	0.010	0.008	0.019
3383.9	93.37	4.693	3.013	1.037	0.939	1.680
2.3	2.61	0.029	0.012	0.013	0.005	0.017
2.2	1.88	0.012	0.007	0.003	0.002	0.005
169.9	12.22	0.381	0.209	0.066	0.107	0.173

4.8	0.055	0.032	0.011	0.012	0.023

Date	Discharge (m3/s)	Determinand concentration (mg/l)			
		SS	TP	SRP	NO3
19/11/2009	0.271	8.6	0.046	0.033	1.06
25/11/2009	0.244	23.6	0.055	0.022	1.34
08/12/2009	0.061	2.3	0.006	0.002	1.27
16/01/2010	0.176	14.5	0.054	0.027	
27/01/2010	0.030	7.3	0.053	0.030	0.00
17/02/2010	0.016	2.0	0.033	0.024	1.29
23/02/2010	0.013	0.0	0.077	0.042	4.56
16/03/2010	0.018	4.5	0.092	0.076	1.33
31/03/2010	0.072	15.0	0.103	0.059	1.67
06/04/2010	0.018	3.0	0.133	0.092	1.86
21/04/2010	0.012	2.0	0.157	0.125	5.41
05/05/2010	0.010	2.0	0.073	0.061	4.59
18/05/2010	0.010				
09/06/2010	0.010	0.0	0.106	0.077	3.96
21/06/2010	0.010	0.0	0.113	0.075	6.12
02/07/2010	0.010	4.4	0.150	0.112	3.51
26/07/2010	0.014	4.6	0.067	0.042	4.41
17/08/2010	0.012	2.5	0.063	0.045	5.00
02/09/2010	0.013	1.0	0.083	0.075	3.09
22/09/2010	0.015	2.0	0.055	0.046	3.61
05/10/2010	0.019	8.6	0.033	0.017	1.34
19/10/2010	0.013	1.5	0.047	0.024	1.23
02/11/2010	0.142	18.5	0.048	0.014	1.52
23/11/2010	0.019	3.0	0.044	0.019	2.85
01/12/2010	0.014	2.2	0.040	0.018	
14/12/2010	0.016	3.5	0.031	0.009	1.86
10/01/2011	0.086	9.5	0.048	0.013	
17/01/2011	0.095	5.8	0.057	0.032	3.56
04/02/2011	0.362	28.0	0.074	0.027	1.03
09/02/2011	0.132				2.71
23/02/2011	0.022	4.2	0.045	0.022	2.68
12/03/2011	0.062	4.5	0.025	0.008	3.28
16/03/2011	0.021	4.1	0.039	0.014	2.88
06/04/2011	0.045	3.5	0.029	0.010	3.52
21/04/2011	0.012	2.5	0.047	0.036	2.19
11/05/2011	0.009	0.0	0.054	0.031	3.85
25/05/2011	0.012	2.8	0.032	0.023	2.47
07/06/2011	0.010				3.41
29/06/2011	0.011	1.2	0.076	0.055	2.06
20/07/2011	0.039	8.8	0.056	0.021	3.80
03/08/2011	0.010				
23/08/2011	0.013	3.5	0.054	0.031	

06/09/2011	0.182	12.5	0.034	0.013	1.34
28/09/2011	0.018				
12/10/2011	0.334	19.5	0.066	0.015	0.00
08/11/2011	0.018	2.9	0.033	0.021	1.97
22/11/2011	0.013				
14/12/2011	0.220				
19/12/2011	0.035				

Determinand instantaneous load (mg/s)

SS	TP	SRP	NO3
2318.9	12.42	8.99	286.8
5751.7	13.41	5.29	327.2
141.2	0.38	0.15	77.5
2549.2	9.49	4.75	
217.9	1.59	0.90	0.0
31.9	0.52	0.38	20.6
0.0	0.99	0.54	58.5
79.8	1.63	1.35	23.5
1072.7	7.37	4.22	119.4
54.1	2.40	1.66	33.5
24.9	1.96	1.56	67.4
20.4	0.74	0.62	46.8
0.0	1.02	0.74	38.1
0.0	1.11	0.74	60.1
42.0	1.43	1.07	33.5
64.8	0.94	0.59	62.0
29.9	0.75	0.54	59.9
13.4	1.11	1.00	41.3
29.2	0.80	0.67	52.8
159.2	0.61	0.31	24.8
20.1	0.63	0.32	16.4
2624.0	6.81	1.99	215.6
56.1	0.82	0.36	53.3
31.1	0.56	0.25	
57.4	0.51	0.15	30.5
821.0	4.15	1.12	
548.7	5.39	3.03	336.8
10146.9	26.82	9.78	373.3
			356.4
92.8	0.99	0.49	59.2
280.1	1.56	0.50	204.1
84.7	0.81	0.29	59.5
157.6	1.31	0.45	158.4
29.6	0.56	0.43	25.9
0.0	0.46	0.26	32.8
33.1	0.38	0.27	29.2
			33.9
13.1	0.83	0.60	22.5
340.4	2.17	0.81	147.0
45.1	0.70	0.40	

2277.1	6.19	2.37	244.1
6520.9	22.07	5.02	0.0
52.7	0.60	0.38	35.8

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EVENT A	Date/time	Discharge (m3/s)	SS	TP	TSP
1	20/07/2010 23:15	0.502	146.7	0.306	
2	21/07/2010 00:15	0.616	306.7	0.388	
3	21/07/2010 01:15	0.624	375.4	0.445	
4	21/07/2010 02:15	0.704	301.3	0.526	
5	21/07/2010 03:15	0.643	200.0	0.293	
6	21/07/2010 04:15	0.575	129.7	0.203	
7	21/07/2010 05:15	0.548	71.3	0.163	
8	21/07/2010 06:15	0.498	89.3	0.122	
9	21/07/2010 07:15	0.454	65.7	0.177	
10	21/07/2010 08:15	0.415	75.0	0.148	

EVENT B	Date/time	Discharge (m3/s)	SS	TP	TSP
1	01/10/2010 14:30	0.225	136.5	0.262	0.088
2	01/10/2010 15:30	0.398	316.2	0.334	0.095
3	01/10/2010 16:30	0.401	289.6	0.401	0.106
4	01/10/2010 17:30	0.365	195.0	0.339	0.111
5	01/10/2010 18:30	0.336	113.8	0.276	0.155
6	01/10/2010 19:30	0.335	90.8	0.253	0.154
7	01/10/2010 20:30	0.326	80.0	0.224	0.188
8	01/10/2010 21:30	0.314	88.7	0.285	0.230
9	01/10/2010 22:30	0.291	36.5	0.191	0.174
10	01/10/2010 23:30	0.291	39.6	0.145	0.099
11	02/10/2010 00:30	0.272	30.0	0.120	0.084
12	02/10/2010 01:30	0.265	28.1	0.097	0.063

EVENT C	Date/time	Discharge (m3/s)	SS	TP	TSP
1	06/10/2010 07:05	0.436	56.5	0.332	
2	06/10/2010 08:05	0.480	140	0.536	
3	06/10/2010 09:05	0.397	123.5	0.439	
4	06/10/2010 10:05	0.355	71	0.322	
5	06/10/2010 11:05	0.380	43	0.267	
6	06/10/2010 12:05	0.460	44	0.233	
7	06/10/2010 13:05	0.439	61.5	0.178	
8	06/10/2010 14:05	0.380	35.5	0.260	
9	06/10/2010 15:05	0.380	34	0.244	
10	06/10/2010 16:05	0.390	36	0.209	
11	06/10/2010 17:05	0.420	30.5	0.201	
12	06/10/2010 18:05	0.427	29	0.184	
13	06/10/2010 19:05	0.393	28.5	0.177	

EVENT D	Date/time	Discharge (m3/s)	SS	TP	TSP
1	11/11/2010 06:30	0.982	189.0	0.092	0.055
2	11/11/2010 07:30	2.000	322.0	0.306	0.092
3	11/11/2010 08:30	2.111	481.5	0.497	0.135
4	11/11/2010 09:30	1.870	545.5	0.808	0.206

5	11/11/2010 10:30	1.848	502.4	0.886	0.195
6	11/11/2010 11:30	1.866	450.4	0.995	0.251
7	11/11/2010 12:30	2.184	393.0	1.001	0.212
8	11/11/2010 13:30	2.409	296.0	0.895	0.277
9	11/11/2010 14:30	2.140	198.0	0.919	0.375
10	11/11/2010 15:30	1.728	158.0	0.793	0.335
11	11/11/2010 16:30	1.884	151.0	0.594	0.239
12	11/11/2010 17:30	2.404	142.5	0.545	0.222
13	11/11/2010 18:30	2.499	117.0	0.463	0.200
14	11/11/2010 19:30	2.467	112.5	0.371	0.184
15	11/11/2010 20:30	2.542	102.0	0.354	0.155
16	11/11/2010 21:30	2.641	76.0	0.262	0.141
17	11/11/2010 22:30	2.736	61.0	0.203	0.129
18	11/11/2010 23:30	2.696	80.0	0.198	0.125
19	12/11/2010 00:30	2.515	85.5	0.180	0.118
20	12/11/2010 01:30	2.145	56.4	0.194	0.119
21	12/11/2010 02:30	1.754	36.4	0.203	0.114
22	12/11/2010 03:30	1.563	30.7	0.166	0.102
23	12/11/2010 04:30	1.411	27.5	0.155	0.098
24	12/11/2010 05:30	1.253	29.5	0.143	0.106

EVENT E	**Date/time**	**Discharge (m3/s)**	**SS**	**TP**	**TSP**
1	10/12/2010 15:00	0.601	295.6	0.534	
2	10/12/2010 16:00	0.872	321.0	0.598	
3	10/12/2010 17:00	0.893	314.0	0.348	
4	10/12/2010 18:00	0.788	218.5	0.297	
5	10/12/2010 19:00	0.763	196.7	0.237	
6	10/12/2010 20:00	0.763	115.0	0.292	
7	10/12/2010 21:00	0.739	93.9	0.288	
8	10/12/2010 22:00	0.673	85.6	0.292	
9	10/12/2010 23:00	0.578	75.6	0.200	
10	11/12/2010 00:00	0.516	79.4	0.272	
11	11/12/2010 01:00	0.480	81.7	0.246	
12	11/12/2010 02:00	0.478	77.2	0.244	
13	11/12/2010 03:00	0.480	74.4	0.251	
14	11/12/2010 04:00	0.486	68.3	0.182	

EVENT F	**Date/time**	**Discharge (m3/s)**	**SS**	**TP**	**TSP**
1	15/01/2011 06:30	0.578	153.3	0.297	0.058
2	15/01/2011 07:30	1.250	276.7	0.433	0.083
3	15/01/2011 08:30	2.174	386.7	0.534	0.088
4	15/01/2011 09:30	2.548	221.7	0.626	0.096
5	15/01/2011 10:30	2.902	179.4	0.591	0.166
6	15/01/2011 11:30	3.056	175.6	0.488	0.169
7	15/01/2011 12:30	3.110	169.4	0.451	0.233
8	15/01/2011 13:30	3.189	106.1	0.455	0.283
9	15/01/2011 14:30	3.239	65.0	0.414	0.252

10	15/01/2011 15:30	3.227	47.8	0.322	0.165
11	15/01/2011 16:30	3.196	43.9	0.293	0.097
12	15/01/2011 17:30	3.301	57.2	0.263	0.094
13	15/01/2011 18:30	3.301	72.2	0.297	0.089
14	15/01/2011 19:30	3.282	63.9	0.285	0.098
15	15/01/2011 20:30	3.251	64.4	0.251	0.120
16	15/01/2011 21:30	3.245	85.6	0.233	0.111
17	15/01/2011 22:30	3.301	79.4	0.191	0.102
18	15/01/2011 23:30	3.276	60.0	0.178	0.088
19	16/01/2011 00:30	3.183	68.3	0.165	0.089
20	16/01/2011 01:30	3.062	76.1	0.150	0.095

EVENT G	**Date/time**	**Discharge (m3/s)**	**SS**	**TP**	**TSP**
1	09/03/2011 23:45	0.146	64.6	0.220	
2	10/03/2011 00:45	0.230	133.5	0.398	
3	10/03/2011 01:45	0.902	167.0	0.511	
4	10/03/2011 02:45	1.789	188.2	0.716	
5	10/03/2011 03:45	1.754	125.4	0.689	
6	10/03/2011 04:45	1.538	114.5	0.594	
7	10/03/2011 05:45	1.294	101.0	0.512	
8	10/03/2011 06:45	1.294	72.2	0.443	
9	10/03/2011 07:45	1.182	65.7	0.402	
10	10/03/2011 08:45	1.147	56.0	0.357	
11	10/03/2011 09:45	1.189	59.5	0.338	
12	10/03/2011 10:45	1.030	52.2	0.284	
13	10/03/2011 11:45	0.920	42.0	0.262	
14	10/03/2011 12:45	0.869	33.7	0.239	
15	10/03/2011 13:45	0.800	31.0	0.219	
16	10/03/2011 14:45	0.825	26.5	0.197	
17	10/03/2011 15:45	0.902	24.0	0.186	
18	10/03/2011 16:45	0.814	27.2	0.175	
19	10/03/2011 17:45	0.681	21.4	0.171	
20	10/03/2011 18:45	0.580	29.5	0.133	
21	10/03/2011 19:45	0.519	25.5	0.104	

EVENT H	**Date/time**	**Discharge (m3/s)**	**SS**	**TP**	**TSP**
1	05/04/2011 00:30	0.343	180.9	0.388	
2	05/04/2011 01:30	0.381	320.0	0.944	
3	05/04/2011 02:30	0.825	398.5	1.155	
4	05/04/2011 03:30	2.357	312.2	1.021	
5	05/04/2011 04:30	2.758	212.4	1.005	
6	05/04/2011 05:30	2.861	142.6	0.975	
7	05/04/2011 06:30	2.804	91.3	0.822	
8	05/04/2011 07:30	2.635	60.9	0.634	
9	05/04/2011 08:30	2.414	57.6	0.386	
10	05/04/2011 09:30	2.067	63.6	0.338	
11	05/04/2011 10:30	1.732	64.6	0.279	

	12 05/04/2011 11:30	1.392	60.7	0.249	

EVENT I	Date/time	Discharge (m3/s)	SS	TP	TSP
1	23/05/2011 11:30	0.104	559.5	0.938	0.134
2	23/05/2011 12:00	0.127	769.5	1.168	0.136
3	23/05/2011 12:30	0.519	787.5	1.193	0.142
4	23/05/2011 13:00	0.715	881	1.168	0.148
5	23/05/2011 13:30	0.499	651	1.015	0.186
6	23/05/2011 14:00	0.392	383	0.807	0.122
7	23/05/2011 14:30	0.326	248	0.536	0.098
8	23/05/2011 15:00	0.298	166.5	0.439	0.094
9	23/05/2011 15:30	0.272	132	0.345	0.085
10	23/05/2011 16:00	0.244	104	0.306	0.106

EVENT J	Date/time	Discharge (m3/s)	SS	TP	TSP
1	22/06/2011 15:45	0.156	401	0.802	
2	22/06/2011 16:15	0.250	522	0.950	
3	22/06/2011 16:45	0.763	686.4	1.035	
4	22/06/2011 17:15	1.510	697	1.071	
5	22/06/2011 17:45	2.352	623	0.984	
6	22/06/2011 18:15	2.844	425	0.916	
7	22/06/2011 18:45	2.908	177.6	0.865	
8	22/06/2011 19:15	2.937	182.4	0.756	
9	22/06/2011 19:45	3.020	146.5	0.711	
10	22/06/2011 20:15	3.171	130	0.656	
11	22/06/2011 20:45	3.171	96.5	0.645	
12	22/06/2011 21:15	3.141	66.4	0.645	
13	22/06/2011 21:45	3.032	43	0.602	
14	22/06/2011 22:15	2.741	40.8	0.584	
15	22/06/2011 22:45	2.174	33.6	0.542	
16	22/06/2011 23:15	1.693	26.5	0.534	
17	22/06/2011 23:45	1.354	25	0.530	
18	23/06/2011 00:15	1.158	19.4	0.534	
19	23/06/2011 00:45	1.030	32	0.542	

EVENT K	Date/time	Discharge (m3/s)	SS	TP	TSP
1	25/11/2011 00:30	0.966	406.5	1.159	0.355
2	25/11/2011 01:30	2.357	614.5	1.283	0.481
3	25/11/2011 02:30	2.399	387	1.033	0.395
4	25/11/2011 03:30	1.963	128	0.858	0.329
5	25/11/2011 04:30	1.253	68.5	0.410	0.225
6	25/11/2011 05:30	0.920	53	0.359	0.165
7	25/11/2011 06:30	0.715	39	0.320	0.173
8	25/11/2011 07:30	0.601	45	0.285	0.148
9	25/11/2011 08:30	0.521	38	0.226	0.135
10	25/11/2011 09:30	0.478	28	0.194	0.124
11	25/11/2011 10:30	0.443	18	0.163	0.120

12	25/11/2011 11:30	0.443	32.5	0.099	0.075
13	25/11/2011 12:30	0.653	30	0.154	0.078
14	25/11/2011 13:30	1.114	64	0.290	0.088
15	25/11/2011 14:30	1.283	80.5	0.293	0.095
16	25/11/2011 15:30	0.950	50.5	0.262	0.116
17	25/11/2011 16:30	0.720	32	0.212	0.110
18	25/11/2011 17:30	0.601	27	0.194	0.105

and concentration (mg/l)

SRP	PP	SUP	NO3

SRP	PP	SUP	NO3
0.046	0.174	0.042	3.35
0.062	0.239	0.033	6.49
0.065	0.295	0.041	9.10
0.065	0.228	0.046	10.05
0.127	0.121	0.028	10.21
0.118	0.099	0.036	9.12
0.151	0.036	0.037	7.48
0.188	0.055	0.042	7.12
0.144	0.017	0.030	6.66
0.086	0.046	0.013	5.94
0.046	0.036	0.038	6.08
0.033	0.034	0.030	5.44

SRP	PP	SUP	NO3

SRP	PP	SUP	NO3
0.050	0.037	0.005	4.33
0.073	0.214	0.019	4.54
0.118	0.362	0.017	4.34
0.184	0.602	0.022	4.22

0.163	0.691	0.032	4.32
0.212	0.744	0.039	4.56
0.171	0.789	0.041	4.31
0.174	0.618	0.103	4.31
0.215	0.544	0.160	4.37
0.278	0.458	0.057	4.47
0.205	0.355	0.034	4.55
0.201	0.323	0.021	4.67
0.182	0.263	0.018	4.72
0.152	0.187	0.032	5.02
0.133	0.199	0.022	5.09
0.107	0.121	0.034	4.89
0.109	0.074	0.020	4.62
0.103	0.073	0.022	4.09
0.092	0.062	0.026	4.17
0.099	0.075	0.020	4.24
0.082	0.089	0.032	4.25
0.078	0.064	0.024	4.28
0.071	0.057	0.027	4.31
0.077	0.037	0.029	4.32

SRP	PP	SUP	NO3

SRP	PP	SUP	NO3
0.046	0.239	0.012	4.24
0.062	0.350	0.021	4.16
0.065	0.446	0.023	4.11
0.065	0.530	0.031	3.93
0.127	0.425	0.039	3.89
0.118	0.319	0.051	3.90
0.171	0.218	0.062	3.77
0.225	0.172	0.058	3.75
0.210	0.162	0.042	3.90

0.123	0.157	0.042	4.11
0.065	0.196	0.032	4.29
0.060	0.169	0.034	4.65
0.063	0.208	0.026	4.91
0.052	0.187	0.046	5.21
0.063	0.131	0.057	5.27
0.054	0.122	0.057	5.23
0.052	0.089	0.050	4.99
0.045	0.090	0.043	4.99
0.058	0.076	0.031	4.93
0.071	0.055	0.024	5.02

SRP	**PP**	**SUP**	**NO3**
0.047	0.173		7.39
0.053	0.346		7.26
0.066	0.445		7.23
0.088	0.628		7.06
0.127	0.562		7.26
0.127	0.467		7.56
0.141	0.371		7.61
0.171	0.271		7.54
0.174	0.227		7.78
0.194	0.163		8.07
0.199	0.139		8.42
0.188	0.096		8.80
0.169	0.093		9.13
0.152	0.088		9.32
0.134	0.085		9.55
0.127	0.069		9.75
0.113	0.073		9.81
0.105	0.070		10.00
0.099	0.072		10.01
0.100	0.033		10.05
0.091	0.013		10.03

SRP	**PP**	**SUP**	**NO3**

SRP	PP	SUP	NO3
0.086	0.804	0.048	5.348
0.095	1.032	0.041	4.968
0.097	1.051	0.045	5.135
0.094	1.020	0.054	5.288
0.144	0.829	0.042	5.787
0.067	0.685	0.055	5.912
0.052	0.438	0.046	6.109
0.039	0.345	0.055	6.055
0.058	0.260	0.027	5.989
0.067	0.200	0.039	6.102

SRP	PP	SUP	NO3
0.069			7.79
0.077			5.69
0.080			4.95
0.097			4.42
0.089			3.93
0.105			4.12
0.108			4.66
0.104			5.50
0.102			6.27
0.097			6.91
0.089			7.60
0.088			8.09
0.086			8.42
0.085			8.56
0.082			8.54
0.084			8.53
0.077			8.31
0.077			7.99
0.075			7.49

SRP	PP	SUP	NO3
	0.804		
	0.801		
	0.638		
	0.529		
	0.185		
	0.194		
	0.147		
	0.136		
	0.091		
	0.071		
	0.043		

0.024
0.075
0.202
0.198
0.145
0.102
0.090

				Determinand concentrat	
EVENT 1_INLET	**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
1	10/05/2012 00:00		27	0.055	0.009
2	10/05/2012 01:00		56	0.063	0.014
3	10/05/2012 02:00		90	0.074	0.013
4	10/05/2012 03:00		169	0.166	0.016
5	10/05/2012 04:00		392	0.782	0.030
6	10/05/2012 05:00		466	1.323	0.073
7	10/05/2012 06:00		524	1.376	0.182
8	10/05/2012 07:00		491	0.861	0.316
9	10/05/2012 08:00		387	0.775	0.353
10	10/05/2012 09:00		229	0.566	0.408
11	10/05/2012 10:00		186	0.532	0.334
12	10/05/2012 11:00		150	0.511	0.206
13	10/05/2012 12:00		207	0.486	0.237
14	10/05/2012 13:00		282	0.815	0.329
15	10/05/2012 14:00		206	0.780	0.334
16	10/05/2012 15:00		172	0.711	0.338
17	10/05/2012 16:00		164	0.639	0.340
18	10/05/2012 17:00		125	0.626	0.323
19	10/05/2012 18:00		130	0.614	0.334
20	10/05/2012 19:00		168	0.550	0.376
21	10/05/2012 20:00		123	0.534	0.327
22	10/05/2012 21:00		82	0.524	0.318
23	10/05/2012 22:00		67	0.439	0.310
24	10/05/2012 23:00		55	0.396	0.297
EVENT 1_OUTLET					
1	10/05/2012 00:00		13	0.033	0.007
2	10/05/2012 01:00		11	0.030	0.011
3	10/05/2012 02:00		15	0.035	0.009
4	10/05/2012 03:00		21	0.051	0.013
5	10/05/2012 04:00		72	0.079	0.016
6	10/05/2012 05:00		224	0.700	0.028
7	10/05/2012 06:00		270	0.706	0.090
8	10/05/2012 07:00		290	0.688	0.210
9	10/05/2012 08:00		245	0.621	0.235
10	10/05/2012 09:00		191	0.522	0.265
11	10/05/2012 10:00		139	0.485	0.227
12	10/05/2012 11:00		119	0.462	0.124
13	10/05/2012 12:00		129	0.444	0.184
14	10/05/2012 13:00		196	0.647	0.229
15	10/05/2012 14:00		181	0.704	0.237
16	10/05/2012 15:00		150	0.644	0.259
17	10/05/2012 16:00		140	0.600	0.280
18	10/05/2012 17:00		107	0.570	0.293
19	10/05/2012 18:00		105	0.547	0.278
20	10/05/2012 19:00		114	0.490	0.299

	21	10/05/2012 20:00		80	0.467	0.291
	22	10/05/2012 21:00		63	0.426	0.269
	23	10/05/2012 22:00		52	0.393	0.261
	24	10/05/2012 23:00		44	0.350	0.229
EVENT 2_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	17/06/2012 01:30		175	0.098	
	2	17/06/2012 02:30		192	0.121	
	3	17/06/2012 03:30		266	0.454	
	4	17/06/2012 04:30		342	0.713	
	5	17/06/2012 05:30		237	0.597	
	6	17/06/2012 06:30		218	0.531	
	7	17/06/2012 07:30		185	0.422	
	8	17/06/2012 08:30		154	0.347	
	9	17/06/2012 09:30		142	0.338	
	10	17/06/2012 10:30		153	0.312	
EVENT 2_OUTLET						
	1	17/06/2012 01:30		148	0.062	
	2	17/06/2012 02:30		162	0.085	
	3	17/06/2012 03:30		195	0.224	
	4	17/06/2012 04:30		219	0.488	
	5	17/06/2012 05:30		185	0.447	
	6	17/06/2012 06:30		161	0.433	
	7	17/06/2012 07:30		137	0.388	
	8	17/06/2012 08:30		116	0.312	
	9	17/06/2012 09:30		118	0.276	
	10	17/06/2012 10:30		115	0.251	
EVENT 3_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	12/10/2012 01:30		497	0.408	0.056
	2	12/10/2012 02:30		646	0.551	0.075
	3	12/10/2012 03:30		880	0.715	0.125
	4	12/10/2012 04:30		1068	1.035	0.185
	5	12/10/2012 05:30		887	0.877	0.212
	6	12/10/2012 06:30		642	0.847	0.226
	7	12/10/2012 07:30		632	0.748	0.235
	8	12/10/2012 08:30		384	0.660	0.221
	9	12/10/2012 09:30		355	0.653	0.198
	10	12/10/2012 10:30		300	0.554	0.188
	11	12/10/2012 11:30		293	0.588	0.165
	12	12/10/2012 12:30		271	0.538	0.168
	13	12/10/2012 13:30		139	0.327	0.109
	14	12/10/2012 14:30		165	0.263	0.088
EVENT 3_OUTLET						
	1	12/10/2012 01:30		388	0.228	0.041
	2	12/10/2012 02:30		466	0.415	0.052
	3	12/10/2012 03:30		702	0.551	0.088

	4	12/10/2012 04:30		822	0.722	0.103
	5	12/10/2012 05:30		639	0.759	0.163
	6	12/10/2012 06:30		460	0.745	0.165
	7	12/10/2012 07:30		488	0.669	0.203
	8	12/10/2012 08:30		325	0.628	0.206
	9	12/10/2012 09:30		259	0.605	0.173
	10	12/10/2012 10:30		265	0.492	0.166
	11	12/10/2012 11:30		233	0.475	0.141
	12	12/10/2012 12:30		249	0.455	0.099
	13	12/10/2012 13:30		152	0.332	0.078
	14	12/10/2012 14:30		133	0.274	0.071

EVENT 4_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	25/11/2012 02:15	7.30	559	0.998	
	2	25/11/2012 03:15	11.66	684	1.306	
	3	25/11/2012 04:15	15.68	991	1.412	
	4	25/11/2012 05:15	21.68	900	1.674	
	5	25/11/2012 06:15	22.54	776	1.943	
	6	25/11/2012 07:15	24.10	636	2.040	
	7	25/11/2012 08:15	25.72	493	1.867	
	8	25/11/2012 09:15	26.91	325	1.566	
	9	25/11/2012 10:15	26.91	326	1.097	
	10	25/11/2012 11:15	25.48	278	0.900	
	11	25/11/2012 12:15	27.15	282	0.923	
	12	25/11/2012 13:15	29.14	329	1.030	
	13	25/11/2012 14:15	27.89	312	0.946	
	14	25/11/2012 15:15	24.33	291	0.828	
	15	25/11/2012 16:15	21.26	264	0.805	
	16	25/11/2012 17:15	18.63	236	0.789	
	17	25/11/2012 18:15	9.45	166	0.725	
	18	25/11/2012 19:15	7.63	117	0.644	
EVENT 4 _OUTLET						
	1	25/11/2012 02:15	3.92	360	0.752	
	2	25/11/2012 03:15	8.63	460	0.904	
	3	25/11/2012 04:15	11.70	478	1.042	
	4	25/11/2012 05:15	17.02	540	1.123	
	5	25/11/2012 06:15	21.56	514	1.256	
	6	25/11/2012 07:15	23.50	376	1.316	
	7	25/11/2012 08:15	25.10	385	1.346	
	8	25/11/2012 09:15	25.81	342	1.320	
	9	25/11/2012 10:15	25.88	312	1.217	
	10	25/11/2012 11:15	25.96	298	0.931	
	11	25/11/2012 12:15	26.43	285	0.982	
	12	25/11/2012 13:15	27.64	258	1.012	
	13	25/11/2012 14:15	27.47	240	0.842	
	14	25/11/2012 15:15	25.47	224	0.711	
	15	25/11/2012 16:15	23.04	193	0.704	

	16	25/11/2012 17:15	19.73	134	0.699	
	17	25/11/2012 18:15	17.14	104	0.656	
	18	25/11/2012 19:15	9.08	83	0.608	

EVENT 5_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	14/12/2012 15:30	13.66	475	0.665	0.075
	2	14/12/2012 16:30	33.66	662	0.898	0.125
	3	14/12/2012 17:30	41.95	846	1.124	0.185
	4	14/12/2012 18:30	51.73	975	1.523	0.212
	5	14/12/2012 19:30	51.73	793	1.466	0.226
	6	14/12/2012 20:30	48.56	804	1.034	0.235
	7	14/12/2012 21:30	46.52	655	0.905	0.221
	8	14/12/2012 22:30	43.23	423	0.844	0.198
	9	14/12/2012 23:30	37.67	402	0.746	0.188
	10	15/12/2012 00:30	33.11	379	0.701	0.165
	11	15/12/2012 01:30	30.43	355	0.677	0.132
	12	15/12/2012 02:30	28.39	310	0.678	0.109
	13	15/12/2012 03:30	25.72	272	0.612	0.088
	14	15/12/2012 04:30	22.76	255	0.588	0.082
EVENT 5_OUTLET						
	1	14/12/2012 15:30	7.14	346	0.485	0.062
	2	14/12/2012 16:30	18.48	432	0.598	0.095
	3	14/12/2012 17:30	34.71	575	0.879	0.133
	4	14/12/2012 18:30	45.00	755	1.107	0.177
	5	14/12/2012 19:30	52.22	666	1.145	0.198
	6	14/12/2012 20:30	51.19	648	0.933	0.208
	7	14/12/2012 21:30	48.53	561	0.724	0.206
	8	14/12/2012 22:30	45.85	415	0.674	0.188
	9	14/12/2012 23:30	42.16	351	0.682	0.174
	10	15/12/2012 00:30	38.32	311	0.612	0.172
	11	15/12/2012 01:30	33.44	268	0.605	0.144
	12	15/12/2012 02:30	30.60	231	0.584	0.111
	13	15/12/2012 03:30	27.77	197	0.524	0.084
	14	15/12/2012 04:30	25.27	184	0.499	0.073

EVENT 6_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	07/01/2013 04:30	8.21	448	0.512	
	2	07/01/2013 05:30	14.48	586	0.656	
	3	07/01/2013 06:30	19.02	869	0.942	
	4	07/01/2013 07:30	26.67	834	1.125	
	5	07/01/2013 08:30	24.56	575	1.032	
	6	07/01/2013 09:30	16.21	488	0.812	
	7	07/01/2013 10:30	10.80	414	0.622	
	8	07/01/2013 11:30	8.45	345	0.635	
	9	07/01/2013 12:30	7.63	350	0.601	
	10	07/01/2013 13:30	18.25	376	0.614	
	11	07/01/2013 14:30	22.98	494	0.688	

	12	07/01/2013 15:30	18.06		461	0.786
	13	07/01/2013 16:30	20.64		346	0.710
	14	07/01/2013 17:30	15.50		224	0.627
	15	07/01/2013 18:30	11.81		189	0.574
EVENT 6_OUTLET						
	1	07/01/2013 04:30	6.20		264	0.430
	2	07/01/2013 05:30	11.37		361	0.512
	3	07/01/2013 06:30	16.55		525	0.612
	4	07/01/2013 07:30	25.57		571	0.684
	5	07/01/2013 08:30	26.85		447	0.802
	6	07/01/2013 09:30	19.81		385	0.788
	7	07/01/2013 10:30	12.53		322	0.666
	8	07/01/2013 11:30	9.83		241	0.485
	9	07/01/2013 12:30	8.38		235	0.466
	10	07/01/2013 13:30	14.22		253	0.475
	11	07/01/2013 14:30	19.98		284	0.493
	12	07/01/2013 15:30	21.23		294	0.594
	13	07/01/2013 16:30	20.04		241	0.575
	14	07/01/2013 17:30	16.78		232	0.570
	15	07/01/2013 18:30	13.09		184	0.531
EVENT 7_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	26/01/2013 21:15	20.64		201	0.342
	2	26/01/2013 22:15	22.98		210	0.376
	3	26/01/2013 23:15	22.98		208	0.402
	4	27/01/2013 00:15	25.02		220	0.396
	5	27/01/2013 01:15	28.64		228	0.431
	6	27/01/2013 02:15	31.22		263	0.477
	7	27/01/2013 03:15	35.06		320	0.536
	8	27/01/2013 04:15	37.97		641	0.856
	9	27/01/2013 05:15	41.64		696	1.214
	10	27/01/2013 06:15	45.19		708	1.471
	11	27/01/2013 07:15	47.20		599	1.522
	12	27/01/2013 08:15	47.54		510	1.215
	13	27/01/2013 09:15	46.86		352	1.102
	14	27/01/2013 10:15	46.86		331	0.933
	15	27/01/2013 11:15	50.66		325	0.902
	16	27/01/2013 12:15	46.86		313	0.843
	17	27/01/2013 13:15	51.02		336	0.722
	18	27/01/2013 14:15	49.96		330	0.755
	19	27/01/2013 15:15	46.52		284	0.677
	20	27/01/2013 16:15	44.86		230	0.632
	21	27/01/2013 17:15	41.32		213	0.598
	22	27/01/2013 18:15	36.49		184	0.543
	23	27/01/2013 19:15	33.11		186	0.521
	24	27/01/2013 20:15	28.64		169	0.496
EVENT 7_OUTLET						

1	26/01/2013 21:15	20.65	171	0.335
2	26/01/2013 22:15	21.33	177	0.328
3	26/01/2013 23:15	21.76	174	0.331
4	27/01/2013 00:15	23.40	170	0.339
5	27/01/2013 01:15	26.42	185	0.329
6	27/01/2013 02:15	29.33	191	0.349
7	27/01/2013 03:15	32.42	198	0.402
8	27/01/2013 04:15	35.32	319	0.499
9	27/01/2013 05:15	38.09	434	0.754
10	27/01/2013 06:15	44.54	441	0.890
11	27/01/2013 07:15	48.34	387	0.939
12	27/01/2013 08:15	49.32	374	1.021
13	27/01/2013 09:15	48.23	279	0.967
14	27/01/2013 10:15	47.69	252	0.789
15	27/01/2013 11:15	50.96	258	0.721
16	27/01/2013 12:15	50.10	244	0.689
17	27/01/2013 13:15	50.31	248	0.633
18	27/01/2013 14:15	51.52	247	0.615
19	27/01/2013 15:15	48.12	234	0.601
20	27/01/2013 16:15	45.24	219	0.576
21	27/01/2013 17:15	42.57	221	0.534
22	27/01/2013 18:15	37.89	214	0.519
23	27/01/2013 19:15	33.98	185	0.494
24	27/01/2013 20:15	30.35	165	0.467

EVENT 8_INLET	**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
1	17/03/2013 08:00	14.15		236	0.621
2	17/03/2013 09:00	22.54		595	1.121
3	17/03/2013 10:00	18.83		435	1.020
4	17/03/2013 11:00	13.99		275	0.746
5	17/03/2013 12:00	11.81		259	0.556
6	17/03/2013 13:00	10.52		240	0.388
7	17/03/2013 14:00	8.70		239	0.355
8	17/03/2013 15:00	8.70		215	0.358
9	17/03/2013 16:00	8.10		203	0.347
10	17/03/2013 17:00	8.21		188	0.335
11	17/03/2013 18:00	7.63		246	0.339
12	17/03/2013 19:00	16.75		276	0.388
13	17/03/2013 20:00	22.11		476	0.655
14	17/03/2013 21:00	35.34		840	1.225
15	17/03/2013 22:00	30.17		626	1.054
16	17/03/2013 23:00	24.56		376	0.891
17	18/03/2013 00:00	20.64		327	0.744
18	18/03/2013 01:00	32.83		506	0.875
19	18/03/2013 02:00	22.98		329	0.607
20	18/03/2013 03:00	18.83		228	0.588
21	18/03/2013 04:00	16.93		216	0.549

	22	18/03/2013 05:00	14.99	200	0.502	
EVENT 8_OUTLET						
	1	17/03/2013 08:00	13.22	164	0.416	
	2	17/03/2013 09:00	24.10	333	0.753	
	3	17/03/2013 10:00	18.02	304	0.706	
	4	17/03/2013 11:00	13.46	192	0.521	
	5	17/03/2013 12:00	11.77	184	0.311	
	6	17/03/2013 13:00	10.16	180	0.302	
	7	17/03/2013 14:00	8.96	173	0.308	
	8	17/03/2013 15:00	8.19	163	0.315	
	9	17/03/2013 16:00	7.80	159	0.319	
	10	17/03/2013 17:00	7.62	152	0.310	
	11	17/03/2013 18:00	7.44	157	0.289	
	12	17/03/2013 19:00	14.39	164	0.295	
	13	17/03/2013 20:00	20.84	283	0.389	
	14	17/03/2013 21:00	31.75	523	0.824	
	15	17/03/2013 22:00	30.54	412	0.705	
	16	17/03/2013 23:00	24.22	255	0.555	
	17	18/03/2013 00:00	20.57	233	0.531	
	18	18/03/2013 01:00	31.13	268	0.472	
	19	18/03/2013 02:00	23.22	219	0.449	
	20	18/03/2013 03:00	18.83	190	0.422	
	21	18/03/2013 04:00	16.86	167	0.425	
	22	18/03/2013 05:00	14.74	159	0.418	

EVENT 9_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	19/03/2013 05:45	13.34	275	0.356	0.056
	2	19/03/2013 06:45	19.42	309	0.398	0.075
	3	19/03/2013 07:45	21.68	556	0.633	0.125
	4	19/03/2013 08:45	30.69	895	0.812	0.185
	5	19/03/2013 09:45	44.53	1021	1.022	0.212
	6	19/03/2013 10:45	45.19	934	1.012	0.226
	7	19/03/2013 11:45	42.91	677	0.764	0.235
	8	19/03/2013 12:45	40.39	498	0.655	0.221
	9	19/03/2013 13:45	36.79	465	0.602	0.198
	10	19/03/2013 14:45	30.17	468	0.549	0.188
	11	19/03/2013 15:45	34.21	459	0.555	0.165
	12	19/03/2013 16:45	39.17	525	0.584	0.154
	13	19/03/2013 17:45	40.70	655	0.612	0.155
	14	19/03/2013 18:45	37.37	629	0.655	0.139
	15	19/03/2013 19:45	34.49	575	0.617	0.155
	16	19/03/2013 20:45	30.69	503	0.551	0.162
	17	19/03/2013 21:45	25.25	426	0.522	0.152
	18	19/03/2013 22:45	26.91	379	0.535	0.148
	19	19/03/2013 23:45	28.89	399	0.531	0.135
	20	20/03/2013 00:45	26.43	402	0.510	0.136
	21	20/03/2013 01:45	24.56	343	0.482	0.129

22	20/03/2013 02:45	20.84	351	0.439	0.122
23	20/03/2013 03:45	19.42	289	0.416	0.118
24	20/03/2013 04:45	18.44	251	0.388	0.111
EVENT 9_OUTLET					
1	19/03/2013 05:45	12.94	200	0.311	0.041
2	19/03/2013 06:45	17.50	234	0.318	0.052
3	19/03/2013 07:45	20.68	366	0.415	0.088
4	19/03/2013 08:45	27.98	502	0.521	0.144
5	19/03/2013 09:45	40.83	634	0.703	0.179
6	19/03/2013 10:45	46.03	511	0.712	0.198
7	19/03/2013 11:45	44.68	464	0.589	0.211
8	19/03/2013 12:45	40.38	401	0.532	0.206
9	19/03/2013 13:45	37.23	390	0.535	0.173
10	19/03/2013 14:45	31.75	393	0.504	0.166
11	19/03/2013 15:45	34.47	384	0.491	0.141
12	19/03/2013 16:45	36.31	445	0.495	0.132
13	19/03/2013 17:45	40.72	488	0.502	0.125
14	19/03/2013 18:45	38.47	497	0.509	0.122
15	19/03/2013 19:45	34.38	471	0.513	0.129
16	19/03/2013 20:45	30.26	399	0.503	0.134
17	19/03/2013 21:45	25.43	351	0.489	0.141
18	19/03/2013 22:45	24.51	284	0.472	0.133
19	19/03/2013 23:45	26.28	266	0.455	0.126
20	20/03/2013 00:45	28.03	268	0.457	0.120
21	20/03/2013 01:45	24.93	241	0.446	0.116
22	20/03/2013 02:45	20.81	276	0.431	0.108
23	20/03/2013 03:45	19.59	214	0.392	0.097
24	20/03/2013 04:45	17.96	176	0.381	0.092

ion (mg/l)	
TSP	NO3
	6.67
	6.76
	6.79
	6.97
	6.82
	6.27
	19.82
	33.75
	32.11
	31.31
	41.55
	46.26
	46.89
	49.09
	49.05
	49.03
	49.23
	49.48
	49.34
	49.03
	49.25
	49.48
	48.62
	46.19
	6.59
	6.65
	6.74
	6.99
	7.21
	6.31
	10.12
	19.29
	20.25
	21.68
	24.13
	28.48
	32.75
	40.38
	41.09
	43.99
	46.75
	48.34
	48.48
	48.22

	48.10
	47.91
	47.12
	45.32
TSP	**NO3**
	7.12
	7.09
	7.34
	7.35
	8.13
	8.55
	8.92
	9.23
	9.68
	10.49
	7.09
	7.10
	7.15
	7.24
	7.29
	7.30
	7.55
	7.96
	8.91
	9.11
TSP	**NO3**
	8.46
	8.43
	9.15
	9.55
	10.51
	7.77
	8.64
	10.93
	13.10
	15.83
	16.16
	20.84
	21.66
	21.03
	8.11
	7.89
	7.51

		6.70
		6.23
		6.25
		7.42
		10.61
		12.28
		14.52
		14.46
		16.22
		18.48
		19.80
TSP	**NO3**	
		6.11
		6.24
		6.90
		7.45
		8.02
		8.12
		8.33
		8.66
		8.61
		8.46
		8.84
		8.89
		8.85
		8.77
		8.52
		8.49
		8.85
		8.98
		5.46
		5.54
		5.69
		6.55
		6.75
		6.98
		6.99
		7.01
		7.45
		8.33
		8.36
		8.39
		8.42
		8.77
		8.79

		8.86
		8.87
		8.69
TSP	**NO3**	
		6.45
		6.58
		7.25
		7.59
		8.35
		10.22
		12.47
		13.34
		13.90
		14.22
		14.19
		13.87
		13.85
		13.12
		6.38
		6.45
		6.47
		7.02
		7.35
		8.82
		10.03
		11.85
		12.22
		13.66
		13.70
		13.70
		13.72
		13.71
TSP	**NO3**	
		5.88
		6.02
		6.12
		6.10
		6.35
		6.84
		7.35
		8.31
		8.62
		9.21
		8.95

		8.56
		8.68
		8.93
		9.45
		5.57
		5.50
		5.68
		6.02
		5.88
		6.11
		6.36
		6.75
		6.93
		7.88
		7.93
		8.11
		8.16
		8.21
		8.18
TSP	**NO3**	
		6.02
		6.10
		6.10
		6.16
		6.16
		6.07
		6.14
		6.19
		6.23
		7.12
		8.45
		9.54
		10.34
		11.83
		11.86
		12.12
		12.10
		12.12
		12.34
		12.57
		12.33
		11.51
		12.23
		12.12

6.04
6.04
6.09
6.08
6.07
6.09
6.11
6.10
5.99
6.07
6.64
7.41
7.97
8.44
9.61
10.35
10.56
10.87
11.44
12.22
12.85
12.91
12.89
12.89

TSP **NO3**

TSP	NO3
	8.46
	8.43
	9.15
	9.55
	10.51
	10.49
	10.79
	10.93
	13.10
	14.34
	15.66
	16.65
	17.25
	18.80
	19.24
	19.98
	20.35
	22.05
	22.24
	20.90
	21.45

20.41
19.65
19.24

8.11
7.89
9.17
9.66
9.86
10.00
10.01
10.61
12.28
14.52
14.46
16.22
16.20
17.55
18.54
20.12
20.23
21.02
21.23
21.24
20.78
20.41
20.01
19.35

				Determinand concentration (mg		
EVENT 1_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	20/04/2012 11:45		118.5	0.115	0.015
	2	20/04/2012 12:45		261.5	0.221	0.028
	3	20/04/2012 13:45		228.5	0.168	0.035
	4	20/04/2012 14:45		189.5	0.112	0.032
	5	20/04/2012 15:45		92	0.097	0.031
	6	20/04/2012 16:45		168.2	0.122	0.028
	7	20/04/2012 17:45		142.5	0.097	0.025
	8	20/04/2012 18:45		74.2	0.095	0.029
	9	20/04/2012 19:45		50.5	0.067	0.028
	10	20/04/2012 20:45		43	0.060	0.026
	11	20/04/2012 21:45		38.5	0.062	0.023
EVENT 1_OUTLET						
	1	20/04/2012 11:45		22	0.051	0.011
	2	20/04/2012 12:45		108.5	0.119	0.017
	3	20/04/2012 13:45		131	0.108	0.024
	4	20/04/2012 14:45		118.5	0.093	0.026
	5	20/04/2012 15:45		88	0.081	0.023
	6	20/04/2012 16:45		68.4	0.088	0.023
	7	20/04/2012 17:45		82.5	0.073	0.023
	8	20/04/2012 18:45		67	0.068	0.023
	9	20/04/2012 19:45		36.5	0.051	0.024
	10	20/04/2012 20:45		32	0.051	0.023
	11	20/04/2012 21:45		24	0.044	0.020

EVENT 2_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	26/04/2012 04:00		172.5	0.428	0.073
	2	26/04/2012 05:00		190	0.256	0.084
	3	26/04/2012 06:00		234.5	0.233	0.107
	4	26/04/2012 07:00		186.5	0.226	0.094
	5	26/04/2012 08:00		197	0.230	0.090
	6	26/04/2012 09:00		86	0.239	0.086
	7	26/04/2012 10:00		57	0.242	0.073
	8	26/04/2012 11:00		62.5	0.297	0.077
	9	26/04/2012 12:00		68	0.359	0.069
	10	26/04/2012 13:00		71	0.193	0.067
	11	26/04/2012 14:00		92	0.124	0.065
	12	26/04/2012 15:00		91.5	0.131	0.065
	13	26/04/2012 16:00		87.5	0.122	0.063
	14	26/04/2012 17:00		68.5	0.113	0.063
	15	26/04/2012 18:00		37.5	0.102	0.060
	16	26/04/2012 19:00		35	0.097	0.062
	17	26/04/2012 20:00		53.5	0.120	0.056
	18	26/04/2012 21:00		28	0.113	0.060
	19	26/04/2012 22:00		30.5	0.090	0.056
	20	26/04/2012 23:00		48.5	0.127	0.054

	21	27/04/2012 00:00		104.5	0.157	0.058
EVENT 2_OUTLET						
	1	26/04/2012 04:00		83.6	0.143	0.035
	2	26/04/2012 05:00		107.825	0.150	0.039
	3	26/04/2012 06:00		113.05	0.168	0.058
	4	26/04/2012 07:00		94.05	0.187	0.063
	5	26/04/2012 08:00		64.125	0.152	0.090
	6	26/04/2012 09:00		60.8	0.155	0.078
	7	26/04/2012 10:00		56.525	0.136	0.073
	8	26/04/2012 11:00		48.925	0.113	0.075
	9	26/04/2012 12:00		32.775	0.102	0.067
	10	26/04/2012 13:00		33.25	0.097	0.058
	11	26/04/2012 14:00		38.95	0.085	0.054
	12	26/04/2012 15:00		28.975	0.095	0.056
	13	26/04/2012 16:00		28.5	0.102	0.063
	14	26/04/2012 17:00		27.55	0.115	0.063
	15	26/04/2012 18:00		36.1	0.108	0.062
	16	26/04/2012 19:00		33.725	0.108	0.058
	17	26/04/2012 20:00		33.25	0.102	0.056
	18	26/04/2012 21:00		25.175	0.097	0.056
	19	26/04/2012 22:00		17.575	0.076	0.052
	20	26/04/2012 23:00		28.025	0.081	0.048
	21	27/04/2012 00:00		39.9	0.074	0.046

EVENT 3_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	10/05/2012 04:45		162.5	0.189	
	2	10/05/2012 05:45		190	0.238	
	3	10/05/2012 06:45		234.5	0.288	
	4	10/05/2012 07:45		186.5	0.275	
	5	10/05/2012 08:45		197	0.273	
	6	10/05/2012 09:45		86	0.234	
	7	10/05/2012 10:45		57	0.197	
	8	10/05/2012 11:45		62.5	0.184	
	9	10/05/2012 12:45		68	0.142	
	10	10/05/2012 13:45		71	0.124	
	11	10/05/2012 14:45		92	0.118	
	12	10/05/2012 15:45		91.5	0.108	
	13	10/05/2012 16:45		88.2	0.110	
	14	10/05/2012 17:45		81.5	0.102	
	15	10/05/2012 18:45		76.5	0.097	
EVENT 3_OUTLET						
	1	10/05/2012 04:45		88.5	0.143	
	2	10/05/2012 05:45		119	0.150	
	3	10/05/2012 06:45		127	0.168	
	4	10/05/2012 07:45		74.5	0.188	
	5	10/05/2012 08:45		67.5	0.152	
	6	10/05/2012 09:45		54	0.156	

	7	10/05/2012 10:45		42.5	0.136	
	8	10/05/2012 11:45		44.8	0.113	
	9	10/05/2012 12:45		34.5	0.103	
	10	10/05/2012 13:45		35	0.097	
	11	10/05/2012 14:45		41	0.085	
	12	10/05/2012 15:45		30.5	0.095	
	13	10/05/2012 16:45		39.8	0.090	
	14	10/05/2012 17:45		37.5	0.092	
	15	10/05/2012 18:45		35	0.081	

EVENT 4_INLET		Date/time	Discharge (l/s)	SS	TP	SRP
	1	06/07/2012 22:00		112	0.456	0.065
	2	06/07/2012 23:00		158	0.542	0.082
	3	07/07/2012 00:00		198	0.703	0.087
	4	07/07/2012 01:00		319	0.922	0.098
	5	07/07/2012 02:00		274.5	1.037	0.106
	6	07/07/2012 03:00		206	0.962	0.122
	7	07/07/2012 04:00		176	0.815	0.110
	8	07/07/2012 05:00		164.5	0.768	0.095
	9	07/07/2012 06:00		104	0.616	0.079
	10	07/07/2012 07:00		85	0.509	0.072
	11	07/07/2012 08:00		36.5	0.410	0.069
	12	07/07/2012 09:00		38.5	0.274	0.073
EVENT 4_OUTLET						
	1	06/07/2012 22:00		55	0.344	0.049
	2	06/07/2012 23:00		78	0.385	0.052
	3	07/07/2012 00:00		94	0.411	0.058
	4	07/07/2012 01:00		142.5	0.561	0.062
	5	07/07/2012 02:00		136	0.678	0.069
	6	07/07/2012 03:00		126.5	0.620	0.074
	7	07/07/2012 04:00		127	0.611	0.084
	8	07/07/2012 05:00		84	0.568	0.085
	9	07/07/2012 06:00		64	0.459	0.081
	10	07/07/2012 07:00		34.5	0.400	0.076
	11	07/07/2012 08:00		30.5	0.306	0.068
	12	07/07/2012 09:00		27.5	0.355	0.065

EVENT 5_INLET		Date/time	Discharge (l/s)	SS	TP	SRP
	1	05/08/2012 09:45		165	0.858	0.065
	2	05/08/2012 10:45		220	1.083	0.071
	3	05/08/2012 11:45		162	0.987	0.085
	4	05/08/2012 12:45		137.5	0.752	0.075
	5	05/08/2012 13:45		92.5	0.568	0.078
	6	05/08/2012 14:45		63	0.431	0.062
	7	05/08/2012 15:45		42.5	0.416	0.055
	8	05/08/2012 16:45		21.5	0.350	0.046
	9	05/08/2012 17:45		15.5	0.309	0.035

EVENT 5_OUTLET						
	1	05/08/2012 09:45		62	0.534	0.055
	2	05/08/2012 10:45		97	0.667	0.054
	3	05/08/2012 11:45		96	0.653	0.061
	4	05/08/2012 12:45		69.5	0.529	0.068
	5	05/08/2012 13:45		44.5	0.479	0.067
	6	05/08/2012 14:45		32	0.391	0.065
	7	05/08/2012 15:45		26.5	0.315	0.055
	8	05/08/2012 16:45		13	0.290	0.049
	9	05/08/2012 17:45		7.5	0.292	0.031
EVENT 6_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	24/09/2012 21:30		188	0.689	
	2	24/09/2012 22:30		300	0.911	
	3	24/09/2012 23:30		374.5	1.247	
	4	25/09/2012 00:30		356	1.497	
	5	25/09/2012 01:30		303.5	1.453	
	6	25/09/2012 02:30		265	1.422	
	7	25/09/2012 03:30		236	1.261	
	8	25/09/2012 04:30		232.5	1.100	
	9	25/09/2012 05:30		203.5	1.033	
	10	25/09/2012 06:30		115	0.965	
	11	25/09/2012 07:30		96	0.846	
	12	25/09/2012 08:30		80.5	0.745	
	13	25/09/2012 09:30		72.5	0.622	
	14	25/09/2012 10:30		93	0.601	
	15	25/09/2012 11:30		86	0.574	
	16	25/09/2012 12:30		81	0.545	
	17	25/09/2012 13:30		59	0.529	
	18	25/09/2012 14:30		45.5	0.511	
	19	25/09/2012 15:30		37.5	0.489	
	20	25/09/2012 16:30		34.5	0.466	
	21	25/09/2012 17:30		37.5	0.453	
	22	25/09/2012 18:30		21	0.437	
	23	25/09/2012 19:30		25	0.421	
	24	25/09/2012 20:30		16	0.392	
EVENT 6_OUTLET						
	1	24/09/2012 21:30		102.58	0.516	
	2	24/09/2012 22:30		153.64	0.589	
	3	24/09/2012 23:30		187.22	0.731	
	4	25/09/2012 00:30		208.288	0.929	
	5	25/09/2012 01:30		189.06	1.017	
	6	25/09/2012 02:30		177.56	1.028	
	7	25/09/2012 03:30		132.48	0.992	
	8	25/09/2012 04:30		136.16	0.919	
	9	25/09/2012 05:30		125.12	0.897	
	10	25/09/2012 06:30		99.36	0.831	

	11	25/09/2012 07:30		70.38	0.671	
	12	25/09/2012 08:30		54.28	0.619	
	13	25/09/2012 09:30		40.94	0.466	
	14	25/09/2012 10:30		36.34	0.410	
	15	25/09/2012 11:30		42.32	0.393	
	16	25/09/2012 12:30		34.04	0.379	
	17	25/09/2012 13:30		35.42	0.387	
	18	25/09/2012 14:30		34.04	0.383	
	19	25/09/2012 15:30		19.78	0.374	
	20	25/09/2012 16:30		19.32	0.362	
	21	25/09/2012 17:30		23	0.356	
	22	25/09/2012 18:30		17.94	0.354	
	23	25/09/2012 19:30		12.88	0.315	
	24	25/09/2012 20:30		11.04	0.294	
EVENT 7_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	12/10/2012 03:30		101.5	0.985	
	2	12/10/2012 04:30		161	1.112	
	3	12/10/2012 05:30		131	1.088	
	4	12/10/2012 06:30		117.5	0.887	
	5	12/10/2012 07:30		90	0.822	
	6	12/10/2012 08:30		78.5	0.743	
	7	12/10/2012 09:30		73.5	0.739	
	8	12/10/2012 10:30		69	0.743	
	9	12/10/2012 11:30		51	0.651	
	10	12/10/2012 12:30		39.5	0.723	
	11	12/10/2012 13:30		32.5	0.677	
	12	12/10/2012 14:30		26	0.602	
	13	12/10/2012 15:30		18.5	0.546	
EVENT 7_OUTLET						
	1	12/10/2012 03:30		60.3	0.678	
	2	12/10/2012 04:30		74.34	0.791	
	3	12/10/2012 05:30		85.95	0.855	
	4	12/10/2012 06:30		63	0.786	
	5	12/10/2012 07:30		52.2	0.721	
	6	12/10/2012 08:30		47.7	0.674	
	7	12/10/2012 09:30		47.7	0.559	
	8	12/10/2012 10:30		40.5	0.543	
	9	12/10/2012 11:30		34.2	0.503	
	10	12/10/2012 12:30		27.45	0.511	
	11	12/10/2012 13:30		21.15	0.515	
	12	12/10/2012 14:30		23.85	0.498	
	13	12/10/2012 15:30		16.65	0.472	
EVENT 8_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	22/11/2012 17:00	7.09	524.7	0.954	0.048
	2	22/11/2012 18:00	15.09	598.7	1.230	0.078

	3	22/11/2012 19:00	13.75	334.0	0.793	0.066
	4	22/11/2012 20:00	14.19	170.7	0.499	0.053
	5	22/11/2012 21:00	15.09	122.7	0.357	0.037
	6	22/11/2012 22:00	13.54	86.7	0.292	0.031
	7	22/11/2012 23:00	12.90	68.0	0.286	0.028
	8	23/11/2012 00:00	11.49	56.0	0.263	0.030
	9	23/11/2012 01:00	10.92	55.3	0.244	0.025
EVENT 8_OUTLET						
	1	22/11/2012 17:00	5.47	61.3	0.385	0.036
	2	22/11/2012 18:00	7.23	160.0	0.663	0.045
	3	22/11/2012 19:00	7.13	140.7	0.559	0.049
	4	22/11/2012 20:00	6.23	81.3	0.431	0.037
	5	22/11/2012 21:00	6.13	59.3	0.301	0.031
	6	22/11/2012 22:00	5.76	48.0	0.239	0.029
	7	22/11/2012 23:00	5.48	44.7	0.237	0.030
	8	23/11/2012 00:00	5.23	30.7	0.209	0.028
	9	23/11/2012 01:00	5.38	28.0	0.200	0.027

EVENT 9_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	20/12/2012 09:45	23.76	42	0.274	0.035
	2	20/12/2012 10:45	25.52	55.5	0.309	0.048
	3	20/12/2012 11:45	27.05	56	0.334	0.075
	4	20/12/2012 12:45	28.51	87	0.357	0.082
	5	20/12/2012 13:45	34.80	62.5	0.309	0.086
	6	20/12/2012 14:45	38.00	50	0.256	0.084
	7	20/12/2012 15:45	44.85	43.5	0.212	0.081
	8	20/12/2012 16:45	50.78	45	0.209	0.074
	9	20/12/2012 17:45	55.33	37	0.182	0.062
	10	20/12/2012 18:45	58.19	35	0.166	0.053
	11	20/12/2012 19:45	58.19	31.5	0.168	0.054
	12	20/12/2012 20:45	58.53	28	0.163	0.046
	13	20/12/2012 21:45	59.40	24.5	0.159	0.042
	14	20/12/2012 22:45	57.85	34.5	0.155	0.042
	15	20/12/2012 23:45	57.17	26	0.147	0.037
	16	21/12/2012 00:45	56.66	29	0.147	0.032
	17	21/12/2012 01:45	55.99	29.5	0.163	0.035
	18	21/12/2012 02:45	55.98	29.5	0.159	0.034
	19	21/12/2012 03:45	55.99	32.5	0.166	0.039
	20	21/12/2012 04:45	56.49	27.5	0.154	0.035
EVENT 9_OUTLET						
	1	20/12/2012 09:45	23.78	22	0.166	0.029
	2	20/12/2012 10:45	25.77	27.5	0.170	0.039
	3	20/12/2012 11:45	26.58	25	0.173	0.045
	4	20/12/2012 12:45	27.66	33.5	0.184	0.052
	5	20/12/2012 13:45	31.62	39.5	0.200	0.055
	6	20/12/2012 14:45	32.62	40	0.203	0.060
	7	20/12/2012 15:45	33.71	36.5	0.193	0.061

	8	20/12/2012 16:45	34.08	33	0.196	0.059
	9	20/12/2012 17:45	35.40	37	0.200	0.062
	10	20/12/2012 18:45	33.89	34.5	0.178	0.058
	11	20/12/2012 19:45	34.17	33.5	0.161	0.052
	12	20/12/2012 20:45	33.72	29.5	0.148	0.048
	13	20/12/2012 21:45	34.64	23	0.145	0.049
	14	20/12/2012 22:45	35.31	21	0.141	0.041
	15	20/12/2012 23:45	34.55	23.5	0.145	0.035
	16	21/12/2012 00:45	35.30	17	0.132	0.031
	17	21/12/2012 01:45	35.60	15	0.131	0.031
	18	21/12/2012 02:45	34.27	14	0.122	0.030
	19	21/12/2012 03:45	33.98	16.5	0.118	0.032
	20	21/12/2012 04:45	35.31	14	0.115	0.030
EVENT 10_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	27/01/2013 06:30	12.69	75.5	0.526	0.088
	2	27/01/2013 07:30	15.70	84.5	0.585	0.127
	3	27/01/2013 08:30	19.25	112.6	0.720	0.127
	4	27/01/2013 09:30	21.42	156	0.898	0.141
	5	27/01/2013 10:30	24.65	198.5	1.075	0.171
	6	27/01/2013 11:30	32.29	262.5	1.288	0.174
	7	27/01/2013 12:30	47.67	178.5	1.126	0.194
	8	27/01/2013 13:30	47.96	182.2	1.094	0.199
	9	27/01/2013 14:30	59.99	164.5	1.032	0.188
	10	27/01/2013 15:30	59.33	126.6	0.943	0.169
	11	27/01/2013 16:30	60.51	108.7	0.902	0.152
	12	27/01/2013 17:30	62.88	99	0.857	0.134
	13	27/01/2013 18:30	64.97	102.5	0.838	0.127
	14	27/01/2013 19:30	67.29	95.2	0.784	0.113
	15	27/01/2013 20:30	66.04	85	0.762	0.105
	16	27/01/2013 21:30	65.68	76.7	0.739	0.099
	17	27/01/2013 22:30	63.57	74	0.719	0.100
	18	27/01/2013 23:30	62.88	69.5	0.697	0.091
	19	28/01/2013 00:30	60.66	67	0.686	0.091
	20	28/01/2013 01:30	59.49	70.2	0.675	0.084
	21	28/01/2013 02:30	57.18	64.4	0.671	0.075
	22	28/01/2013 03:30	54.00	72.5	0.633	0.072
	23	28/01/2013 04:30	52.91	68.5	0.604	0.060
	24	28/01/2013 05:30	50.47	62.2	0.551	0.064
EVENT 10_OUTLET						
	1	27/01/2013 06:30	9.26	42.24	0.322	0.045
	2	27/01/2013 07:30	12.80	47.696	0.358	0.072
	3	27/01/2013 08:30	15.97	54.12	0.379	0.098
	4	27/01/2013 09:30	19.56	83.248	0.488	0.102
	5	27/01/2013 10:30	22.75	95.48	0.531	0.121
	6	27/01/2013 11:30	25.25	109.648	0.655	0.127
	7	27/01/2013 12:30	30.16	116.16	0.708	0.134

	8	27/01/2013 13:30	45.43	129.448	0.741	0.151
	9	27/01/2013 14:30	55.61	124.96	0.786	0.159
	10	27/01/2013 15:30	51.66	122.144	0.710	0.162
	11	27/01/2013 16:30	48.14	101.2	0.734	0.164
	12	27/01/2013 17:30	48.49	91.96	0.741	0.158
	13	27/01/2013 18:30	47.55	71.456	0.709	0.144
	14	27/01/2013 19:30	47.78	65.032	0.712	0.120
	15	27/01/2013 20:30	48.62	56.056	0.690	0.101
	16	27/01/2013 21:30	50.43	48.752	0.651	0.085
	17	27/01/2013 22:30	48.61	46.376	0.622	0.081
	18	27/01/2013 23:30	48.37	42.416	0.588	0.082
	19	28/01/2013 00:30	49.34	40.216	0.544	0.073
	20	28/01/2013 01:30	49.33	43.032	0.541	0.071
	21	28/01/2013 02:30	48.14	37.928	0.535	0.067
	22	28/01/2013 03:30	46.60	45.056	0.538	0.069
	23	28/01/2013 04:30	45.56	41.536	0.522	0.068
	24	28/01/2013 05:30	46.02	35.992	0.507	0.061
EVENT 11_INLET		**Date/time**	**Discharge (l/s)**	**SS**	**TP**	**SRP**
	1	18/03/2013 00:30	9.842	105.8	0.648	0.061
	2	18/03/2013 01:30	12.016	232.5	0.845	0.082
	3	18/03/2013 02:30	17.425	295.6	1.034	0.136
	4	18/03/2013 03:30	18.824	321.0	1.125	0.202
	5	18/03/2013 04:30	20.674	314.0	0.977	0.231
	6	18/03/2013 05:30	22.275	218.5	0.797	0.246
	7	18/03/2013 06:30	21.642	196.7	0.737	0.256
	8	18/03/2013 07:30	20.760	115.0	0.792	0.241
	9	18/03/2013 08:30	19.485	93.9	0.788	0.216
	10	18/03/2013 09:30	18.341	85.6	0.792	0.205
	11	18/03/2013 10:30	17.014	75.6	0.712	0.180
	12	18/03/2013 11:30	16.192	79.4	0.701	0.145
	13	18/03/2013 12:30	15.259	81.7	0.675	0.098
	14	18/03/2013 13:30	15.262	77.2	0.623	0.076
	15	18/03/2013 14:30	14.698	74.4	0.533	0.071
	16	18/03/2013 15:30	14.628	68.3	0.501	0.064
EVENT 11_OUTLET						
	1	18/03/2013 00:30	8.40	34.2	0.301	0.041
	2	18/03/2013 01:30	10.21	52.9	0.389	0.048
	3	18/03/2013 02:30	15.28	101.5	0.507	0.063
	4	18/03/2013 03:30	18.23	147.8	0.620	0.105
	5	18/03/2013 04:30	19.56	160.5	0.675	0.155
	6	18/03/2013 05:30	22.22	157.0	0.586	0.178
	7	18/03/2013 06:30	22.37	109.3	0.591	0.190
	8	18/03/2013 07:30	22.35	98.3	0.588	0.195
	9	18/03/2013 08:30	20.64	57.5	0.611	0.197
	10	18/03/2013 09:30	19.42	46.9	0.615	0.185
	11	18/03/2013 10:30	17.77	42.8	0.624	0.166

12	18/03/2013 11:30	16.76	37.8	0.602	0.158
13	18/03/2013 12:30	15.48	39.7	0.566	0.139
14	18/03/2013 13:30	14.87	40.8	0.512	0.112
15	18/03/2013 14:30	14.42	38.6	0.374	0.075
16	18/03/2013 15:30	14.48	37.2	0.320	0.059

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NO3

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NO3

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NO3

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