University of Southampton

Faculty of Environmental and Life Sciences

School of Ocean and Earth Science

Monitoring the Efficacy of Nature-based Solutions for Diffuse Pollution Mitigation in a Lowland Catchment

by

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Abstract

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Natural Flood Management (NFM) is an increasingly used Nature-based Solution (NBS) for flood risk reduction in the UK. It has been suggested that such NBS may also be able to deliver multiple benefits concerning wider ecosystem services such as improvement of stream water quality. This thesis aims to address the gap in the NFM and NBS evidence-base concerning their potential to deliver multiple benefits through the mitigation of diffuse pollution in a lowland agricultural catchment. Hydrological and water quality monitoring was carried out over multiple years in two small headwater sub-catchments of the Evenlode catchment, a largely rural upper tributary of the river Thames. A number of interventions including online ponds, offline storage areas, and instream leaky barriers were implemented within these sub-catchments as part of the Littlestock Brook NFM pilot scheme, one of the first of its kind in the Thames Basin.

Online pond interventions were able to reduce concentrations of dissolved nutrients (nitrate by 5 % and soluble reactive phosphorus by 29 %) and suspended sediment by 32 % during baseflows. During storm events, online ponds were able to attenuate sediment delivery downstream, however they also showed potential to act as sources of sediment in higher-magnitude events. Rapid sediment accumulation rates diminished storage capacity in the upstream-most pond and highlighted the need for frequent sediment removal to maintain functionality. Offline ponds with a primary purpose of temporarily storing water during events were found to provide storage of sediment and nutrients, trapping 47.9 t of sediment over 2-3 years. When combined with accumulations in the online ponds, this was the equivalent of 14.7 % of the suspended sediment yield from the catchment over the same period. Accumulation rates were influenced by hydrological connectivity, with enhanced sediment, phosphorus, and organic carbon capture in offline ponds that filled via overbank flows induced by leaky barriers.

Instream monitoring showed that attributing changes in suspended sediment loading to the implementation of NBS was challenging at a sub-catchment scale due to hydroclimatic variability and land cover change. Therefore, future research should aim to adopt long-term, multi-scale monitoring approaches to better assess the efficacy of such interventions for diffuse pollution mitigation. NFM and NBS provide a valuable suite of options for catchment management, however their efficacy is dependent on their design, connectivity, and maintenance. It is recommended that future NBS research focus on upscaling combined effects of interventions to determine the extent to which implementation is needed to improve downstream water quality.

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List of Accompanying Materials

The following datasets collected as part of this PhD have been published and are accessible via the Environmental Information Data Centre (EIDC):

Robotham, J.; Old, G.; Rameshwaran, P.; Trill, E.; Bishop, J. (2022). **High-resolution time series of turbidity, suspended sediment concentration, total phosphorus concentration, and discharge in the Littlestock Brook, England, 2017-2021**. NERC EDS Environmental Information Data Centre. (Dataset). https://doi.org/10.5285/9f80e349-0594-4ae1-bff3-b055638569f8

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I declare that this thesis and the work presented in it are my own and has been generated by me as the result of my own original research.

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1. This work was done wholly or mainly while in candidature for a research degree at this University;
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Definitions and Abbreviations

AIC .... Akaike Information Criterion. A statistical metric of model prediction error used for model selection.

ANOVA .... Analysis of Variance. Statistical models that compare differences among means.

API .... Antecedent Precipitation Index. An indicative measure of the moisture stored within a catchment prior to a storm event.

BACI .... Before-After Control-Impact. An experimental study design used to assess the impact of an intervention relative to a control.

BFI .... Baseflow Index. A long-term measure of the contribution of slow (sub-surface) flow to stream flow within a given catchment.

BMPs .... Best Management Practices.

BOD .... Biochemical Oxygen Demand.

BGS .... British Geological Survey.

C .... Concentration, measured as constituent mass per volume of water.

CAP .... Common Agricultural Policy. A European Union policy that concerns the implementation of agricultural subsidies.

CI .... Confidence Interval.

CROME .... Crop Map of England.

CW .... Constructed Wetland. An artificially created wetland used to treat polluted water.

D_{50} .... Median particle diameter.

DEFRA .... Department for Environment, Food & Rural Affairs. The UK government department responsible for environmental protection and agriculture.

DHP .... Dissolved Hydrolysable Phosphorus.

DIN .... Dissolved Inorganic Nitrogen.

DOP .... Dissolved Organic Phosphorus.

DP .... Dissolved Phosphorus.

DS.. Downstream.

DTC...................................... Demonstration Test Catchments. A UK government-funded project that looked at cost-effective measures to control diffuse pollution from agriculture.

DTM .... Digital Terrain Model. A topographic model of the bare earth elevation typically used in geospatial analyses.

ELM .... Environmental Land Management. A set of agri-environmental schemes introduced following the departure of the UK from the European Union.

ER .... Enrichment Ratio. A ratio comparing sediment constituents to their source soils.

EU.. European Union.

GAM.................................... Generalised Additive Model. A statistical modelling technique in which the impact of predictor variables is captured through linear or non-linear smoothing functions.

GHG..................................... Greenhouse gas(es). A gas that contributes to the greenhouse effect by absorbing and emitting radiant energy.

GIS....................................... Geographic Information System. A computer system for analysing and displaying georeferenced data.

GLM .... Generalised Linear Model. A statistical modelling technique where the dependent variable is linearly related to the response variable and any covariates using a specified link function.

GPS...................................... Global Positioning System.

HI_{mid}..................................... Hysteresis Index. A measure quantifying the magnitude and direction of hysteretic behaviour of a water quality variable in response to a storm event.

HOST .... Hydrology Of Soil Types. A classification system of UK soils based on their hydrological properties.

ICM...................................... Integrated Catchment Management. A paradigm of sustainable environmental management that aims to holistically manage catchment water resources, the landscape, and their interconnections.

IOM .... Inorganic Matter.

IP .... Inorganic Phosphorus.

IUCN .... International Union for Conservation of Nature.

LCM .... Land Cover Map. A UKCEH geospatial dataset that classifies the UK land surface based on satellite imagery.

LIDAR.................................... Light Detection and Ranging. A laser-based remote sensing method used for determining earth surface elevation.

LOESS.................................... Locally Estimated Scatterplot Smoothing. A statistical method used to fit a smooth curve through a set of data points.

LOI .... Loss-on-ignition. An analytical technique that involves measuring the change in mass of a sample after heating it to a high temperature.

LWD...................................... Large Woody Debris. Large pieces of wood in stream channels, either naturally occurring or artificially placed.

mASL .... Metres above sea level.

NBS....................................... Nature-based Solutions. Interventions or actions that protect, restore, and sustainably manage ecosystems to address both societal and environmental challenges.

NDVI .... Normalised Difference Vegetation Index. An index measuring the greenness (or absence of) of vegetation which is derived from remote sensing data.

NERC.................................... Natural Environment Research Council.

NFM...................................... Natural Flood Management. A suite of approaches used to reduce flood risk (and provide multiple benefits) through storing water in the landscape and slowing its movement downstream through catchments.

NGO...................................... Non-governmental organisation.

NVZ...................................... Nitrate Vulnerable Zone. Areas designated as being vulnerable to nitrate pollution from agriculture.

OC... Organic Carbon.

OLP .... Online Pond. A pond that is directly connected to a watercourse which it receives and releases flow from and to.

OM .... Organic Matter.

OP .... Organophosphorus.

POC .... Particulate Organic Carbon.

PP .... Particulate Phosphorus.

Q .... Stream discharge, measured as volume of water per unit of time.

RAF .... Run-off Attenuation Feature. Modifications to the landscape designed to prevent the rapid transfer of water into stream channels by slowing and storing overland flow.

RBI .... Richard-Baker flashiness Index. A long-term measure of the rapidness of the hydrological response of a stream to storm events.

RBMPs .... River Basin Management Plans. Region-specific plans that set out how waterbodies will be protected and improved during a specified period.

RDA .... Redundancy Analysis. A multivariate statistical technique that explains and summarises a set of response variables using a set of explanatory variables.

RSuDS .... Rural Sustainable Drainage Systems. Land management techniques that help to reduce flood risk in rural settings.

RTK .... Real-time Kinematic. A technique that improves the accuracy of GPS field measurements by correcting the baseline position in real-time.

SCIMAP .... Sensitive Catchment Integrated Modelling and Analysis Platform

SD .... Standard Deviation.

SPR .... Standard Percentage Runoff. The percentage of rainfall that contributes to surface run-off in a given area.

SRP .... Soluble Reactive Phosphorus.

SS .... Suspended Sediment or Suspended Solids.

SSA .... Specific Surface Area. A measure of the total surface area of particles per unit of mass.

SSC .... Suspended Sediment Concentration.

STW .... Sewage Treatment Works.

SuDS .... Sustainable Urban Drainage Systems. Drainage solutions that are designed to manage stormwater locally in urban environments.

TDP .... Total Dissolved Phosphorus.

TP .... Total Phosphorus.

TSS .... Total Suspended Solids.

UKCEH .... UK Centre for Ecology & Hydrology.

US .... Upstream.

VSC .... Volatile Solids Concentration.

WFD .... Water Framework Directive. A European Union directive that obligates member states to meet targets for the ecological and chemical status of their waterbodies.

WwNP .... Working With Natural Processes. A catchment management concept that aims to protect, restore, and emulate natural hydrological processes and catchment functions.

Chapter 1 Introduction

1.1 Introduction

Water quality is a key component of the functioning of river ecosystems (Keeler *et al.*, 2012). Rivers, streams, and riparian wetlands transport, transform and retain particulate matter and solutes derived from their catchments as part of natural physical, chemical, and biological processes (Fritz *et al.*, 2018). Some of these processes have been significantly accelerated or altered as a consequence of anthropogenic influences in the landscape (Akhtar *et al.*, 2021). This has negative consequences on the functioning and resilience of freshwater ecosystems (Angeler *et al.*, 2014; Pelletier *et al.*, 2020). Conserving, restoring, and emulating the natural processes of river catchments is increasingly being seen as an important management tool for multiple environmental issues including flood risk management and pollution control (Ellis, Anderson, *et al.*, 2021). To manage these issues more effectively and target sustainable and cost-effective mitigation, a greater understanding of the efficacy of such Nature-based Solutions (NBS) is needed.

The wide range of NBS for catchment management and water pollution are reviewed and summarised in Chapter 2. The need for effective management measures has become increasingly recognised in the UK due to a widespread failure to meet the regulatory 'good ecological status' under the European Union (EU) Water Framework Directive (WFD) (2000/60/EC), and observed declines in rural water quality (Whelan *et al.*, 2022). Furthermore, the need for NBS for climate change adaptation (e.g. Natural Flood Management (NFM)) is being recognised as a potential tool for offsetting the hydrometeorological impacts of climate change, alongside traditional approaches to flood risk management (Kay *et al.*, 2019). The recent uptake of NBS provides opportunity for gathering empirical evidence on the efficacy of these interventions to provide multiple environmental benefits, helping to fill this knowledge gap across UK catchments. This thesis explores the functioning and efficacy of a number of storage-based pond and wetland features to provide pollution mitigation and water quality benefits in a lowland headwater catchment (Chapter 3, Chapter 4, and Chapter 5). Section 1.2 of this chapter provides a broad background to this research topic and its importance in terms of UK policy as well as globally-relevant environmental concerns. Section 1.3 outlines the overarching gaps in the knowledge surrounding NBS and NFM; these are considered further in Chapter 2. Specific aims and objectives of the thesis are given in Section 1.4; the methodological approach of this research is described in Section 1.5; and the overall structure of the thesis is outlined in Section 1.6.

1.2 Research Background and Rationale

1.2.1 Diffuse Pollution and Agri-environmental Policy

Across the UK, diffuse (non-point source) agricultural pollution is a common reason given to explain the failure of waterbodies to meet 'good ecological status' targets set by the WFD, with agriculture and rural land management listed as a reason for failure in 30 % of waterbodies in England between 2013 and 2016 (France, 2019). Whilst the UK is no longer a member of the EU, the WFD has been retained in UK law and therefore the UK is obligated to implement WFD objectives until at least 2027 (Croner-i, 2021). It has been estimated that losses of phosphorus (P) from agriculture to surface waters account for approximately 34 % of global fertiliser use (Brownlie *et al.*, 2022). Excessive loading of P is widely acknowledged to be a key driver of the degradation of freshwater ecosystems through the processes of nutrient enrichment and eutrophication (Correll, 1998; Foy, 2005). The other major nutrient that pollutes freshwater is Nitrogen (N), largely in the form of nitrate, of which approximately 70 % of total inputs in England (and the UK as a whole) come from agriculture (Bell *et al.*, 2021; Environment Agency, 2021b). Due to its soluble nature, nitrate is able to easily pollute groundwater and poses a threat to drinking water sources, especially in drier regions or under low flow conditions where there is heavier reliance on groundwater supplies (Collins, McGonigle, *et al.*, 2009). In addition to nutrients, sediment is also a pollutant of concern for waterbodies, particularly in catchments dominated by intensive agriculture that typically deliver high loads of fine sediment (Collins *et al.*, 2011; Jones and Schilling, 2011; Naden *et al.*, 2016). In England and Wales, agricultural sources have been found to dominate fluvial sediment inputs, contributing 72-76 % (Collins, Anthony, *et al.*, 2009; Zhang *et al.*, 2014). Sediment can act as an important vector of nutrients and also other chemical contaminants such as pesticides and heavy metals (Kronvang *et al.*, 2003; Pavanelli and Selli, 2013; Astatkie *et al.*, 2021). Elevated concentrations of suspended sediment in stream water and excess sedimentation of channel beds can have ecologically harmful consequences for these habitats and their biota (Bilotta and Brazier, 2008; Kemp *et al.*, 2011; Von Bertrab *et al.*, 2013).

The pressures faced by freshwater ecosystems and their resources in the UK have received significant policy attention in recent years, with a notable focus on the pressures from agriculture and how these can be managed (Wentworth and Peck, 2022). As part of Britain's exit from the European Union (EU), the UK government are in the process of phasing out the Common Agricultural Policy (CAP) scheme in exchange for a new Environmental Land Management (ELM) scheme to be fully implemented in England and Wales by 2028. Together with the new UK Environment Act, this provides significant opportunity to improve upon the failings of previous EU environmental legislation (Klaar, Carver, *et al.*, 2020). The Department for Environment, Food &

Rural Affairs (Defra) have based the ELM scheme on the principal of "*public money for public goods*" which includes "*maintaining and improving the quality of our water*" (DEFRA, 2018). Under this legislation it is likely that payments will be available to land managers for the implementation of measures and interventions aimed at improving water quality. Therefore, it is important to provide robust evidence on how NBS can contribute to water quality improvements in agriculturally productive landscapes.

1.2.2 Climate Change

The projected changes in air temperature, rainfall and the intensity/frequency of hydrometeorological extremes are likely to exacerbate current water quality and flooding issues due to their effect on river regimes, rainfall run-off, flow velocities, water levels and residence times (Whitehead *et al.*, 2009; Arnell *et al.*, 2015; Kay, 2021; Kay *et al.*, 2021). Projections of climate change impacts in the UK include an increased frequency of extreme weather events such as intense rainfall during winter (Lavers *et al.*, 2013; Kendon *et al.*, 2018; Davies *et al.*, 2021). These events have a higher potential to generate rapid run-off which facilitates soil erosion and sediment losses into streams, particularly when fields are bare (Zhang *et al.*, 2022). These predictions suggest that larger pulses of diffuse pollution will occur during such storm events, with UK average P loads increasing by up to 30 % by 2050 (Ockenden *et al.*, 2017). It is likely that such increases in extreme rainfall would also exacerbate the impact of agricultural practices on soil structural degradation which have been linked to both soil erosion and flooding (Holman *et al.*, 2003). Soils in catchments with poor agricultural land management (e.g. overstocking) will likely be more vulnerable to extremes and have less resilience to the effects of climate change. The impacts of warmer and drier summers have been predicted to increase concentrations of soluble P in some lowland rivers in southern England as a result of lower flows and reduced dilution (Whitehead *et al.*, 2008). Consequently, climate change is serving to intensify the symptoms of eutrophication in freshwaters (e.g. reduced dissolved oxygen levels), which in turn has ecosystem-wide implications for both organisms and biogeochemical processes (Cox and Whitehead, 2009; Whitehead *et al.*, 2009; Moss *et al.*, 2011).

Climate change clearly presents a growing challenge for both catchment managers and farmers (Gupta *et al.*, 2021). Research suggests that current best practices in diffuse agricultural pollution control are insufficient to counter the exacerbated pollution from climate change, and therefore future agri-environmental policy needs to be refined to account for this (Pulley and Collins, 2021). Further empirical data on the functioning of NBS in different catchment typologies will improve our understanding of their potential ability to provide resilience to the impacts of climate change and increased risk from both floods and droughts.

1.2.3 Phosphorus Use, Sustainability, and Food Security

Farmers worldwide rely on P for achieving economically viable crop yields using nitrogen, phosphorus, and potassium (NPK) fertilisers. Predictions suggest that peak P production will occur around 2030 and so there is concern over the future scarcity of this important resource (Cordell *et al.*, 2009; 'Approaching peak phosphorus.', 2022). This issue is globally important due to the role of P in food security, and the increasing pressure on the agricultural industry to produce enough food to feed the growing human population. Due to this concern, there is an impetus to use P more sustainably in agricultural systems (Wirth *et al.*, 2021; Carvalho *et al.*, 2022). A better understanding of the linkages between P on the land and in watercourses is required for the development of sustainable practical solutions and policy to address this issue. The global importance of P-related issues and the need for sustainable P use has been recognised across the scientific community over recent years, and emphasised in a report that aims to provide direction on this topic (Brownlie *et al.*, 2022). Recommendations from this report include a call to governments globally to reduce P pollution and increase its recycling by 2050. NBS provide an opportunity to reduce P losses through the interception and storage of surface run-off and eroded nutrients, encouraging re-use on farms to reduce reliance on industrial fertilisers. There are currently limited data on the re-use of intercepted sediment-bound nutrients; further evidence is needed to better assess their potential agronomic value.

1.2.4 Nature-based Solutions for Catchment Management

NBS is an umbrella term and is defined by the International Union for Conservation of Nature (IUCN) as "*actions to protect, sustainably manage, and restore natural or modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits*" (Cohen-Shacham *et al.*, 2016). NBS encompass a broad range of environmental themes and societal challenges and are therefore gaining multi-disciplinary interest in fields such as climate science, ecology, and sustainable development (Cohen-Shacham *et al.*, 2016; Seddon *et al.*, 2020). In terms of water resources and catchment management, NBS are gaining traction due to their potential role in mitigating hydrological extremes, both floods and droughts, and due to the multiple benefits and ecosystem services that they may provide (Acreman *et al.*, 2021; Heneghan *et al.*, 2021; Lashford *et al.*, 2022). The most notable subset of water-related NBS in the UK is NFM which is focussed on slowing flow and conserving, restoring and enhancing catchment processes such as water storage that have been affected by human intervention (Dadson *et al.*, 2017). In recent years, NFM has been implemented relatively widely across the UK as a multi-purpose catchment management strategy, particularly since a £15 million investment from the government to fund 60 pilot projects (Environment Agency, 2021a). The objective of this investment was to use these pilot projects to gather evidence to help enable the mainstreaming of NFM in future flood risk management. Whilst the primary aim of NFM is to reduce flood risk, the Environment Agency have emphasised the importance of multiple benefits as part of the implementation of such schemes (Environment Agency, 2018). Future investment and funding mechanisms for NFM rely on building a robust evidence-base, one that is currently growing, in part due to numerous detailed studies stemming from the Natural Environment Research Council (NERC) NFM research programme (UK Research and Innovation, 2022). The popularity of NFM within research and practitioner communities has also enabled the development of detailed guidance on best practices and considerations for NFM implementation (e.g. CIRIA's NFM manual; Wren *et al.* (2022)). In the UK, current theory on NBS such as NFM is grounded in our broad understanding of hydrological processes that has been significantly improved through observational data from experimental studies in instrumented catchments (e.g. the Coalburn catchment experiment (Robinson *et al.*, 1998), and the Plynlimon catchment study (Robinson *et al.*, 2013). More detailed understanding of specific NBS and their potential role in catchment management has since been developed, partially through empirical studies (e.g. Lockwood *et al.* (2022)), but largely through studies using modelling approaches (e.g. Nicholson *et al.* (2020)). Future uptake and more widespread use of NBS for catchment management depends upon a better understanding of their potential to enhance hydrological processes and provide multiple benefits. Key areas in NBS research that are lacking knowledge (and the reasons for this) are highlighted in Section 1.3. Comprehensive descriptions of specific categories of NBS and NFM, their functions, and an evaluation of current evidence on their efficacy are given in Chapter 2.

1.3 Knowledge Gaps

This section introduces and summarises the key knowledge gaps that were identified through a review of literature in Chapter 2 and form the basis for the aims and objectives of this thesis (Section 1.4).

The field of hydrology is well-studied given the fundamental importance of water to life on Earth, however there are still significant knowledge gaps in our understanding. Although much of our existing knowledge on hydrological processes is relevant to understanding NBS, in many cases NBS represent enhancements of existing catchment features and modifications to processes, and therefore new science is required to better understand these. Wagener *et al.* (2021) identified a number of key gaps in terms of the general hydrology of Great Britain, several of which directly relate to NBS and their functioning; for example, gaps in our understanding of the impacts of changing land cover on surface partitioning.

One of the key gaps in relation to NBS is the lack of empirical data, which is largely limited to field-scale studies focussed on single or clustered interventions (Cooper *et al.*, 2019; Brunet *et al.*, 2021), also typically over short timescales (Yuan *et al.*, 2009), or only using event-based monitoring (Barber and Quinn, 2012). Whilst there are now an increasing number of examples of NFM schemes in the UK, most of these projects lack detailed monitoring and evaluation, meaning there is limited evidence of their success and effectiveness. Where evidence is available, it often comes in the form of grey literature, or is not always publicly-available. This has meant there are gaps in knowledge regarding the effectiveness of interventions in combination, at varying spatial scales, and in extreme events, as well how effectiveness is influenced by different designs and catchment settings. A lack in observational data is partly due to some NBS such as NFM being relatively new in their practical implementation within the UK. Furthermore, the many challenges faced in implementing and monitoring NFM act as a barrier to the generation of data on the effectiveness of these interventions. Lane (2017) describes NFM as a '*wicked problem*', a concept coined by Rittel & Webber (1973) to conceptualise complex policy issues that cannot easily be resolved. The defining characteristics of '*wicked problems*' (summarised in Table 1.1) underpin the lack of evidence on the effectiveness of NFM and uncertainty in the current evidence-base.

Table 1.1 Characteristics that define '*wicked problems*' and how they relate to NFM (information synthesised from Lane (2017).

'*Wicked Problem*' Characteristic	**NFM Context**
The problem is not fully understood until a solution has been developed	The testing of NFM interventions is mainly driven by the need to reduce flood risk where it is a known issue
The problem has no stopping rule.	NFM interventions will not completely prevent flooding, but only reduce the magnitude and frequency of some flood events
Solutions to the problem are neither right nor wrong, having different costs/benefits	NFM interventions may benefit some stakeholders (e.g. property owners downstream) but represent a cost to others (e.g. upstream agricultural land that will store flood water)
Each problem is unique due to its specific characteristics and context	Hydrological properties and responses vary spatially, meaning that NFM interventions will not produce the same effect across different catchments and landscapes (e.g. in uplands compared to lowlands)
The problem has many solutions, some of which may not even be known or tested	There is a wide range of current NFM interventions, however potential new interventions are only limited by creativity

A considerable amount of current evidence focusses on modelling the potential effects of NFM, so there is a need for empirical evidence from field studies to confirm or challenge their findings (Dadson *et al.*, 2017; Environment Agency, 2018). From a review of evidence, Kay *et al.* (2019) estimated that less than 25 % of studies on the effectiveness of NFM report empirical findings, with the majority being based on model results. The representation of interventions within models is one of the challenges faced by modellers, with assumptions and approximations being made during this process, thus necessitating model validation with observed data (Brauman *et al.*, 2022). This would enable hydrological models to be improved, and uncertainties decreased, particularly with the use of high-resolution observed data which may provide greater insight into catchment processes at a finer spatial and temporal scale. Recent technological advances have made high-resolution in-situ water quality monitoring more viable as a cost-effective monitoring tool (Silva *et al.*, 2022). Building long-term time series for water quality will provide data on the effect of NBS on hydrochemical processes at multiple temporal scales (e.g. diel, seasonal, annual) (Appling *et al.*, 2018). Furthermore, empirical data on NBS functioning is needed to better model and predict potential effects across larger spatial scales. Wagener *et al.* (2021) propose that robust hydrological understanding and improved prediction can be built by integrating a depth of knowledge from detailed model-based analyses with a breadth of knowledge from empirical regionalisation approaches. Further data on the effectiveness of NBS interventions operating in combination would benefit the hydrological modelling research community in their efforts to upscale the impact of NBS.

Understanding the variability of NBS functioning and effectiveness between different catchment contexts and typologies is a key area of uncertainty. Ockenden *et al.* (2012) compared the effectiveness of constructed wetland ponds across catchments with different soil types, finding that sediment retention was much higher for the sandy soil site compared to the silty and clay soil sites. Further research comparing NBS across catchments with a spectrum of geologies, topographies and landscape configurations will aid in determining suitable future opportunities for NBS.

More specifically, the existing evidence on NFM focusses mostly on upland catchments in the UK (e.g. Shuttleworth *et al.*, 2019; Bond *et al.*, 2021; Goudarzi *et al.*, 2021; Murphy *et al.*, 2021; Lo *et al.*, 2022), and so presents a knowledge gap on understanding NFM effectiveness in lowland catchments. These upland catchments are not only topographically very different to lowlands, but they also have largely dissimilar land-uses and covers (e.g. blanket bog peatland, acid grassland and rush pasture, moorland/heathland) which in turn influence their sediment and water quality dynamics. The expansion of evidence on NFM in arable-dominated (typically highly modified) lowland catchments which suffer from diffuse pollution would be particularly valuable given their significant proportion of land cover across the UK; arable land covered approximately 20 % of the UK's total land surface in 2021 (UKCEH, 2022b).

The focus of NFM as a NBS has unsurprisingly been largely on flood alleviation, particularly geared towards reducing the risk of economic losses resulting from flood events (Lane, 2017). Consequently, the evidence-base surrounding the wider environmental, social, and cultural benefits of NFM is lacking in its range (Dadson *et al.*, 2017). More holistic and integrated studies into the multiple benefits for all catchment stakeholders would help to address this.

Whilst there are some instances of long-term hydrometeorological monitoring for experimental catchment studies in the UK (e.g. 45 years monitoring of afforestation in the Coalburn catchment (Birkinshaw *et al.*, 2014)), such examples are rare. For NBS projects, considerably shorter monitoring periods mean that hydrometeorological extremes (e.g. large flood events and droughts) are not captured within data, and therefore their functioning cannot be tested under these infrequent but hydrologically significant occurrences. Continuous long-term high-resolution monitoring would enable interventions to be tested during a wider range of hydrological conditions and would help contribute to current investment into infrastructure for flood and drought research in the UK (Old *et al.*, 2022). Therefore there is a clear need for long-term monitoring of NBS to observe hydrological, geomorphological, and ecological changes in order to assess their effectiveness and robustness over longer time-scales.

Spatial scale is also important to consider for monitoring the effects of NBS. It is acknowledged that there is currently a paucity of empirical data on NBS at large catchment scales where these data are crucial for being able to test and validate models (Dadson *et al.*, 2017). For example, Stratford *et al.* (2017) highlight the need for a greater breadth of observational studies on the effect of tree cover on flood peaks, especially in large UK catchments.

The establishment and development of NBS (e.g. vegetation succession and siltation in ponds, growth of trees) are likely to play important roles in how they function. However, the impact of such processes on intervention effectiveness remains understudied. This knowledge is important for understanding how multiple benefits derived from NBS may change over time and can inform maintenance requirements (e.g. desilting) to optimise functioning (Heneghan *et al.*, 2021). Previous research on pollutant transport through sustainable urban drainage system (SuDS) networks over a 12-month period has demonstrated how fine sediment can be repeatedly resuspended and redeposited within SuDS features as a result of multiple rainfall events (Allen *et al.*, 2017, 2019). Sediment retention was found to vary throughout the monitored period. Further research on sediment dynamics beyond individual events is needed (particularly in rural settings) to help to build the evidence-base for such interventions and provide more insightful guidance on their management so as to avoid any unintentional dis-benefits and resource wastage.

1.4 Aims and Objectives

The aim of this research is to develop an improved understanding of the effects of NBS on water quality and sediment dynamics in lowland agricultural streams within a headwater catchment in England. This research seeks to address the relevant research gaps identified in Chapter 2 (and summarised in Section 1.3). The data generated from this research and the insight they provide will help to support the modelling of NBS effectiveness at larger spatial scales. This thesis addresses the following research objectives:

1) To fully characterise the hydrological regime of the study catchment and its suspended sediment and nutrient dynamics to assess the potential of NBS to mitigate diffuse pollution.
2) To assess the ability of different designs of NBS and NFM interventions to mitigate diffuse sediment and nutrient pollution from agriculture.
3) To contribute to the advancement of best practice and guidance for the monitoring and management of NBS and NFM interventions with respect to water quality and flood risk.

1.5 Methodological Approach

The methodological approach used to address the research aims and objectives stated in Section 1.4 is focussed on gathering empirical data from the Littlestock Brook study catchment. A hydrometric and water quality monitoring network was set up across two sub-catchments of the Littlestock Brook to enable the collection of high-resolution stream data and intervention-scale data. Full details of the monitoring network are provided in a report (Trill *et al.,* 2022a; Appendix A). Event-based water sampling campaigns were carried out to calibrate the in-situ sensor data and provide further insight into water quality dynamics. Sediment within the NBS was sampled through the use of sediment trapping devices and core sampling, with surveying of the features and their accumulations carried out alongside this. A before-after control-impact (BACI) approach was also employed to statistically evaluate the impacts of the NBS on sediment and P loading at a sub-catchment scale.

1.6 Thesis Structure

The thesis is subdivided into six chapters (as summarised in Figure 1.1), including this introductory chapter (Chapter 1) which sets out the context and rationale for the research and defines its aims and objectives. Chapter 2 critically reviews the current literature surrounding the themes of NBS, diffuse pollution and catchment management, helping to prioritise areas for further research. Chapters 3, 4, and 5 form the core body of research within the thesis and take the form of stand-alone journal papers (Chapters 3 and 4 have been published as two peer-reviewed journal papers: see Appendix B and Appendix C). Chapter 3 evaluates the effectiveness of a set of online pond features to help mitigate diffuse pollution in an agricultural lowland setting (Evenlode catchment, UK). This work provides detailed insight into the hydrological and biogeochemical functioning of this particular set of NBS features over different time-scales and examines their ability to retain sediment and nutrients. Chapter 4 continues this theme to look at the effect of a wider variety of both online and offline NBS on the accumulation and storage of sediment and nutrients in the same study catchment. This work also explored the factors that influenced the rate at which the NBS accumulated sediment, and the implications of this for their management and maintenance. Chapter 5 presents the results of multi-year sub-catchment scale monitoring of suspended sediment and nutrients, the data from which were used in a before-after control-impact analysis to test for the effect of a NFM scheme on catchment processes over time. This research explores some of the complexities of detecting and attributing change in hydrogeochemical timeseries data and the difficulties associated with catchment-scale monitoring in a highly modified agricultural landscape. Chapter 6 concludes the thesis by summarising the key findings of this research and their implications, and then proposes recommendations for further research in this area.

Appendix A contains a report that details the monitoring and subsequent data analysis of the Littlestock Brook NFM scheme, to which the research of this PhD contributed to significantly. The Appendices also include the two published journal papers from this PhD research (Appendix B and Appendix C), as well as the supporting documentation for a published dataset that was also generated from this research (Appendix E). Additional supporting information for Chapters 4 and 5 is given in Appendix D and Appendix F respectively.

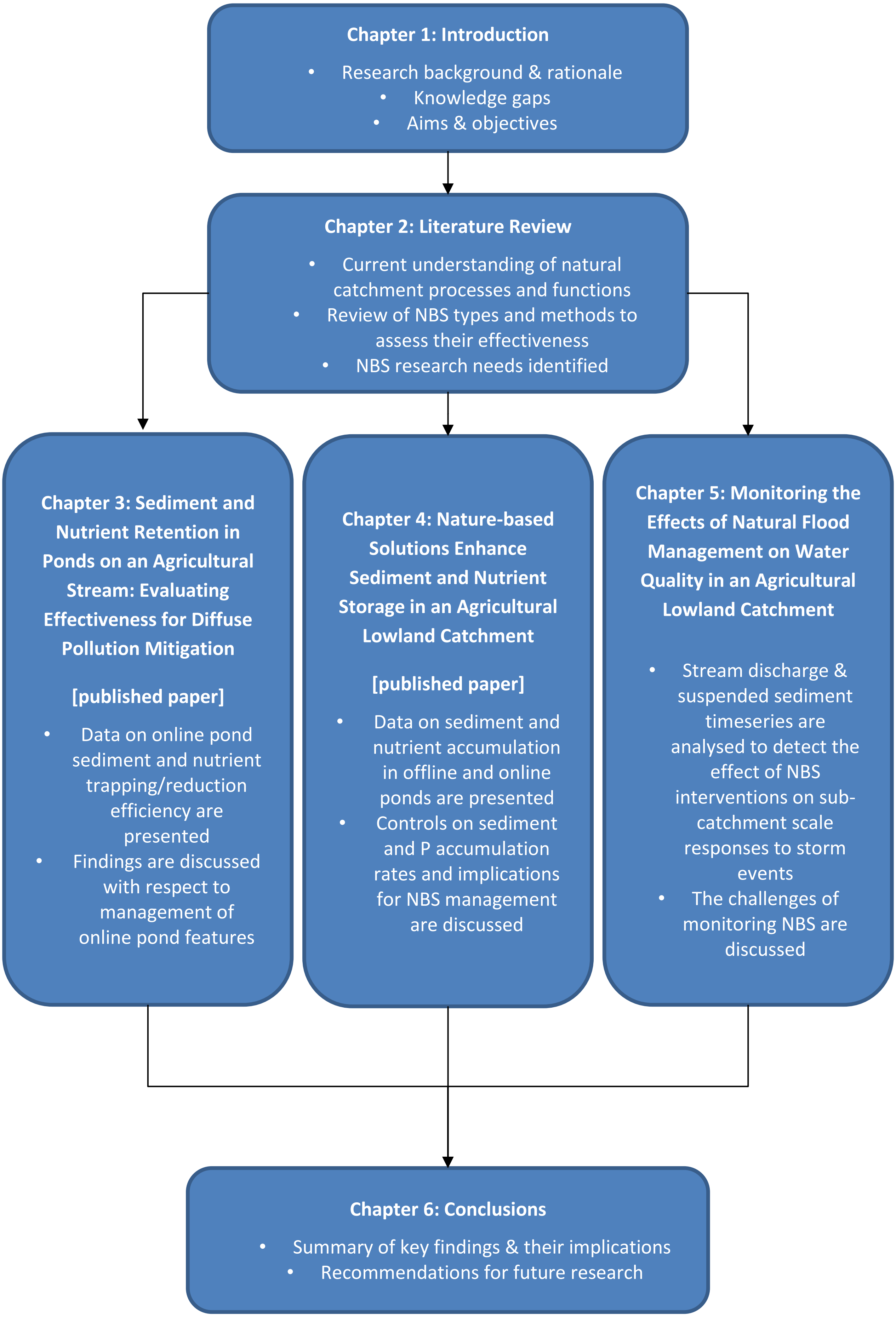


Figure 1.1 Flow diagram summary of the thesis structure.

Chapter 2 Literature Review

2.1 Introduction

Section 2.2 of this chapter provides a wider contextual understanding of the need for research into NBS, in terms of both policy and catchment management with respect to the UK. Section 2.3 provides an overview of our current understanding of natural catchment functions and processes, focussing in detail on those most relevant to water storage-based NBS (e.g. mechanisms controlling run-off in Section 2.3.2). This is followed by insight into the effects of hydrological processes on pollutant dynamics (Section 2.3.3), and the connections between catchment processes, anthropogenic pressures, and NBS (Section 2.3.4). Section 2.4 provides a detailed review of the existing knowledge base on NBS in order to highlight knowledge gaps of priority. The NBS within this section are categorised into three key groups: wetlands, ponds, and water storage features; land and soil management; and trees, woodland, and instream wood. Section 2.5 evaluates methods that have previously been used to assess the efficacy of NBS. Finally, Section 2.6 summarises the key findings of this literature review and the knowledge gaps identified.

2.2 Nature-based Solutions for Catchment Management and Water Pollution

2.2.1 Policy

A key driver of the use of NBS and wider mitigation measures is the need to adhere to environmental regulations such as those set by the European Union in order to protect water resources and aquatic ecosystems. Key regulations include the Water Framework Directive (WFD) (2000/60/EC), the Urban Waste Water Treatment Directive (91/271/EEC), and the Nitrates Directive (91/676/EEC). The WFD has a more holistic remit with an overall objective to achieve 'good ecological status' in all member state waterbodies (surface, ground, and coastal waters) compared to the older directives that addresses specific pollutants or water resource contexts (Collins, McGonigle, *et al.*, 2009). Ecological status encompasses the physical, chemical, and biological aspects of waterbodies, all of which are considered within River Basin Management Plans (RBMPs) which act as frameworks for identifying issues and improving ecological status in different regions. The WFD and the successful implementation of RBMPs and mitigation measures is therefore of importance to a wide variety of stakeholders such as water companies, non-governmental organisations (NGOs), and environmental regulatory bodies. This is often achieved through collaborative efforts and partnerships that work together with landowners and companies to gain maximum benefit for all stakeholders and the wider public (Gaddis *et al.*, 2010; Koontz and Newig, 2014).

In the UK, investment in technological improvements such as P stripping in the tertiary treatment of wastewater has helped to reduce excessive soluble reactive phosphorus (SRP) loading into waterbodies (Kinniburgh and Barnett, 2010). For example, SRP concentration reductions of up to 90 % have been seen in some rivers within the Thames Basin following sewage treatment works (STW) upgrades commissioned prior to 2003 (Neal *et al.*, 2010; Bowes *et al.*, 2012). However, these studies also found that despite the significant reductions in SRP concentrations, algal growth persisted and the risk of eutrophication remained high. Bowes *et al.* (2012) recommend that other measures such as the planting of riparian trees are needed to reduce eutrophication risk in the Thames and achieve 'good ecological status' required by the WFD. This demonstrates how even effective technological solutions to water quality issues are not necessarily enough on their own to achieve sufficient environmental change for meeting policy targets.

2.2.2 Management Approaches and Paradigms

In the 21st century there has been an increased interest in integrated catchment management (ICM) which recognises the need to tackle water-related issues at a broad scale and aims to address them through cooperation between scientists, policy-makers, and stakeholders (Falkenmark, 2004; Stewardson *et al.*, 2017). More widely, there has also been an increased popularity in the concept of nature-based solutions (NBS) which also incorporates ideas such as natural capital, ecosystem services, and green/blue infrastructure (Cohen-Shacham *et al.*, 2016; Dick *et al.*, 2020). NBS are acknowledged to form part of the answer to addressing multiple globally-relevant environmental issues, rather than a panacea (Calliari *et al.*, 2022). NBS are not intended to replace, but instead they work alongside other solutions to such issues that may typically involve grey (hard-engineered) infrastructure. Typically, NBS and natural infrastructure have been compartmentalised according to land-use, with the main distinction being between urban and rural settings. Urban water-related NBS are often referred to as sustainable urban drainage systems (SuDS), and are topical given increasing urbanisation globally (Golden and Hoghooghi, 2018). The global relevance of rural NBS is also high given the agricultural intensification that has taken place in order to meet the food demands of the growing population (Miralles-Wilhelm, 2021). A recent critical review of sustainable catchment management with respect to flooding called for a greater consideration of scale in order to shift thinking from distinct urban or rural management approaches to an integrated catchment-wide approach (Lashford *et al.*, 2022).

Within the UK, the focus on ICM and NBS can be seen in the Environment Agency's drive to grow the evidence-base on 'working with natural processes' (WwNP) and NFM, and their investment in trialling these approaches (Environment Agency, 2018). The UK government recognised the important role of WwNP in their '25 Year Environment Plan', and committed to a greater use of NFM solutions, investing £15 million for implementation up to 2021 (DEFRA, 2018). WwNP and NFM are primarily focussed on reducing flood risk, but also aim to deliver multiple environmental, social, and cultural benefits, including water quality improvement, soil erosion mitigation, capture and storage of carbon, habitat and biodiversity improvement, and enhancement of recreational opportunities (Sparks, 1995; McLean, Beevers, Pender, Haynes and M. Wilkinson, 2013; Iacob *et al.*, 2014; Dadson *et al.*, 2017). Following the departure of the UK from the EU, it is likely that future agri-environment schemes in the UK (e.g. environmental land management (ELM) schemes) will incorporate payments for the provision of services and benefits such as these (Klaar, Carver, *et al.*, 2020). With the recent investments in NBS for catchment and water resource management, this has become a growing area of research, with study into the effectiveness of different techniques in different contexts being needed to inform and advise on future implementation to maximise their potential benefits (Ruangpan *et al.*, 2020; Ellis, Anderson, *et al.*, 2021).

2.3 Natural Catchment Processes and Functions

The scientific principles which underpin NBS are grounded in a fundamental understanding of the hydrological (and biogeochemical) processes operating in natural catchments. This section of the chapter outlines these key processes and functions, and draws links between the pressures that impact them, as well as how NBS relate to them.

2.3.1 Perceptual Model of Hydrological Processes

Figure 2.1 shows the key components of the terrestrial water cycle, including storage within the atmosphere, snowpack, surface waters, vegetation, soils, and groundwater.

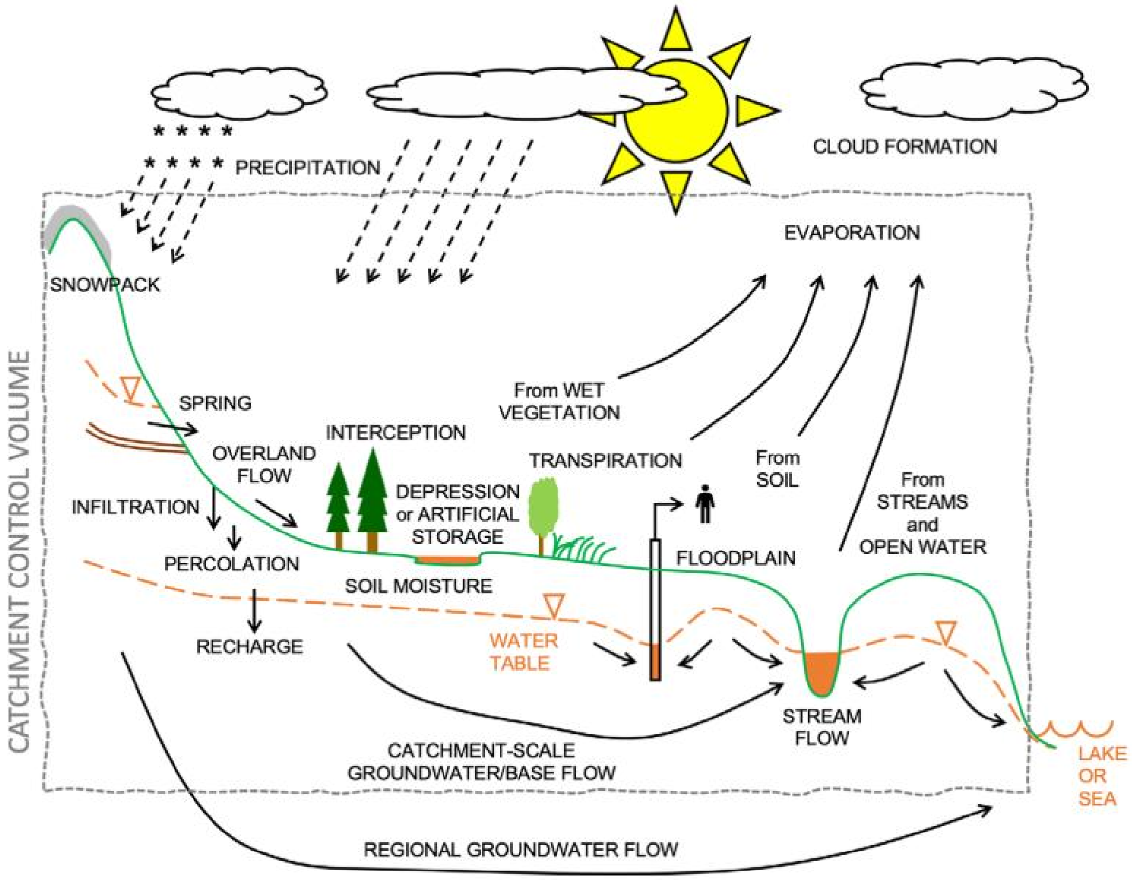


Figure 2.1 A simplified perceptual model of the hydrological processes that may occur in a typical catchment within Great Britain. Reproduced from Wagener *et al.* (2021).

Hydrological processes such as interception and infiltration slow down the movement of water from its origin as precipitation to potentially becoming streamflow. A number of factors determine the extent to which rainfall will be intercepted or infiltrated, including the cover and type of vegetation, and soil properties such as texture and porosity. Another crucial process is the generation of rainfall run-off (overland flow), particularly due to its importance in flood hydrology, a topic which has seen increased attention in both science and policy given the impacts of climate change (Pitt, 2008). Much work has been carried out to assess the relative importance of surface and sub-surface flows and land management types on flood generation (Marshall *et al.*, 2009). Research suggests that post-war agricultural intensification in the UK has augmented run-off generation at a local scale (O'Connell *et al.*, 2007) through a number of mechanisms (Figure 2.2).

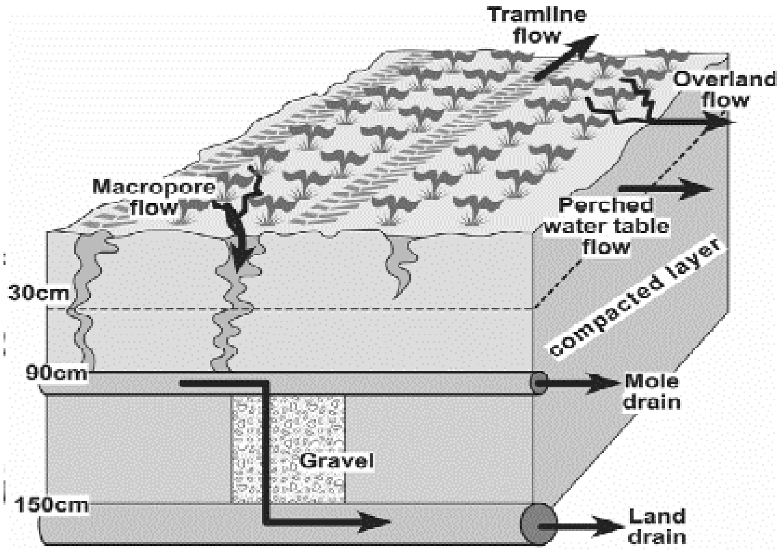


Figure 2.2: Potential local scale mechanisms of run-off generation in an intensively farmed arable field. Reproduced from O'Connell *et al.* (2007).

2.3.2 Mechanisms and Properties Controlling Run-off

Mechanisms that are known to enhance run-off include soil compaction from agricultural machinery and the subsequent concentration of run-off in tramlines which can rapidly transport flow. A field study in the Hampshire Avon catchment demonstrated how tramlines running parallel to the slope generated 46 % more run-off compared to a control plot (Withers *et al.*, 2006). Areas of compaction also act as enhanced pathways for diffuse pollutants, with the same field study reporting sediment losses up to fivefold higher, and total phosphorus (TP) losses up to fourfold higher from tramlines. Rainfall run-off is recognised as a fundamental driver of soil erosion from the land and its transfer into watercourses (Gelder *et al.*, 2018). A field-based determination of the controls on fine sediment generation from livestock fields found that sediment particles were detached from the soil by raindrop impact and mobilised through diffuse saturation excess overland flow (Pulley and Collins, 2019). The phenomenon of saturation excess overland flow is considered to be a bottom-up process in the soil profile and occurs when additional moisture inputs fill soil pore volume to the extent of saturation, initiating overland flow (Stewart *et al.*, 2019). Consequently, soil porosity is understood to play an important role in controlling run-off and associated diffuse pollution (Lipiec *et al.*, 2006; Robinson *et al.*, 2022).

The texture of soil has an influence on both its porosity and permeability and is determined by the proportion of sand, silt, and clay particles that it is made up of. Research showed that soil texture altered the threshold amount of rainfall required for run-off to occur, with a clay loam soil having the lowest threshold of only 7.8 mm compared to a loam soil with a threshold of 17.4 mm (Lee *et* *al.*, 2006). Due to the spatially heterogenous nature of soil textures, different landscapes have varying potentials to generate run-off and diffuse pollution, both between and within catchments, even prior to the consideration of anthropogenic influences on soil hydrology. For example, the dominant soil texture of catchments has been shown to influence the source and speciation balance of nutrient fluxes to headwater streams (Lloyd *et al.*, 2019). The study found that particulate phosphorus (PP) and dissolved organic phosphorus (DOP) dominated the P loads in clay/mudstone sub-catchments in the southwest of England. Observed spikes in TP were largely driven by PP and were linked to flow events and the hydrologically responsive (flashy) behaviour of the clay sub-catchments. Mellander *et al.* (2022) found that P loss risk from agricultural catchments was largely a function of their physical characteristics and flashiness.

In addition to the soil, the underlying geology also plays a crucial role in determining catchment behaviour and hydrochemistry. Lloyd *et al.* (2019) found that in chalk sub-catchments, nitrate was the dominant N species delivered to streams, owing mostly to groundwater inputs. Chronic nitrate pollution in groundwater is common in agricultural areas due to excessive use of nitrogenous fertilisers, and is often identified through observed dilution of nitrate concentrations during storm event hydrograph peaks (Lloyd *et al.*, 2016b; Mehdi *et al.*, 2021; Winter *et al.*, 2022).

2.3.3 Hydrological Processes and Pollutant Dynamics

The relationships between pollutant concentrations and stream discharge have been widely used to characterise the biogeochemical behaviour of catchments and to infer sources and pathways of pollution (Bowes *et al.*, 2015; Perks *et al.*, 2015; Rose *et al.*, 2018; Pohle *et al.*, 2021). The export of solutes and suspended sediment from catchments is often classified based on concentration-discharge relationships and the extent to which they exhibit hysteresis (Figure 2.3). Hysteresis can be defined as a non-linear relationship occurring when solute or sediment concentrations for a given discharge are different on the rising limb of a hydrograph compared to its falling limb (Lloyd *et al.*, 2016a; Malutta *et al.*, 2020). Clockwise hysteresis is indicative of a rapid response in concentration, suggesting nearby sources or a high degree of connectivity between sources, pathways, and the stream. Conversely, anticlockwise hysteresis suggests that sources are distal or there is a lower hydrological connectivity within the catchment. Concentration-discharge relationships and hysteresis patterns are also used to help inform catchment management through identifying priorities for pollution mitigation and determining the most suitable management practices and interventions (Bowes *et al.*, 2014; Sherriff *et al.*, 2016).

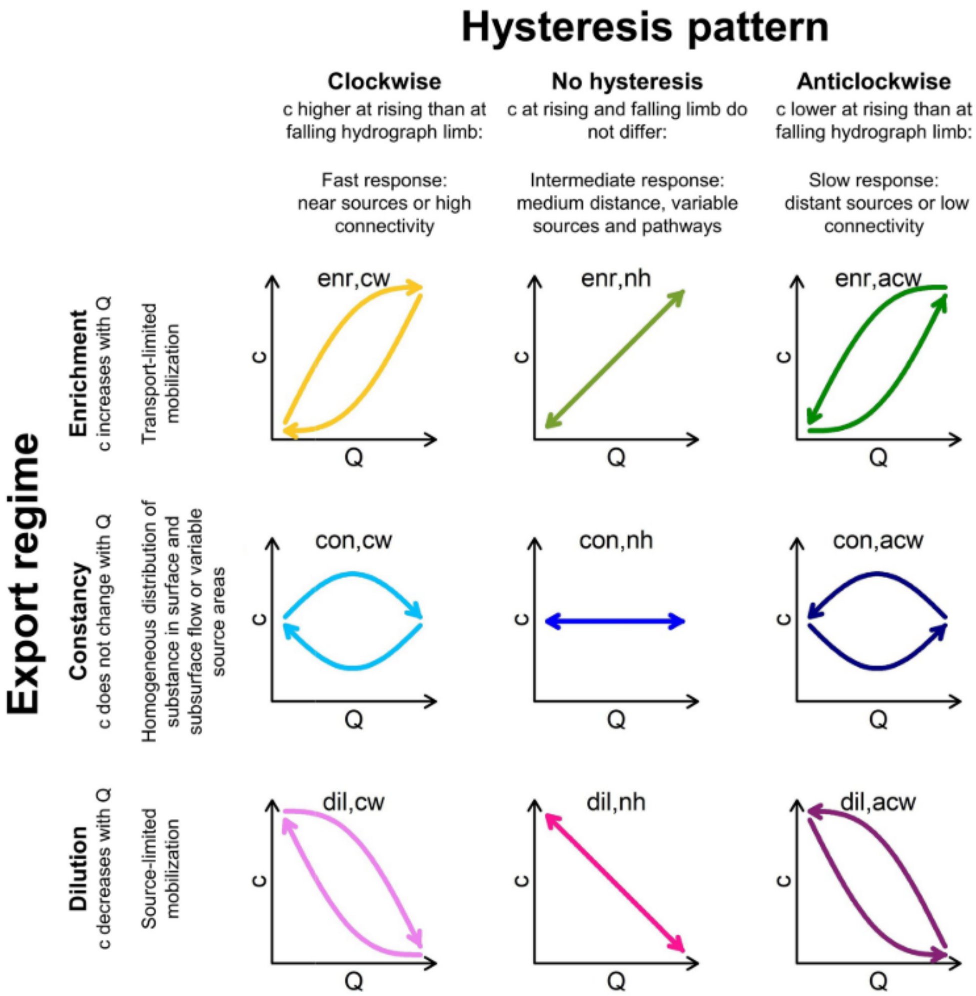
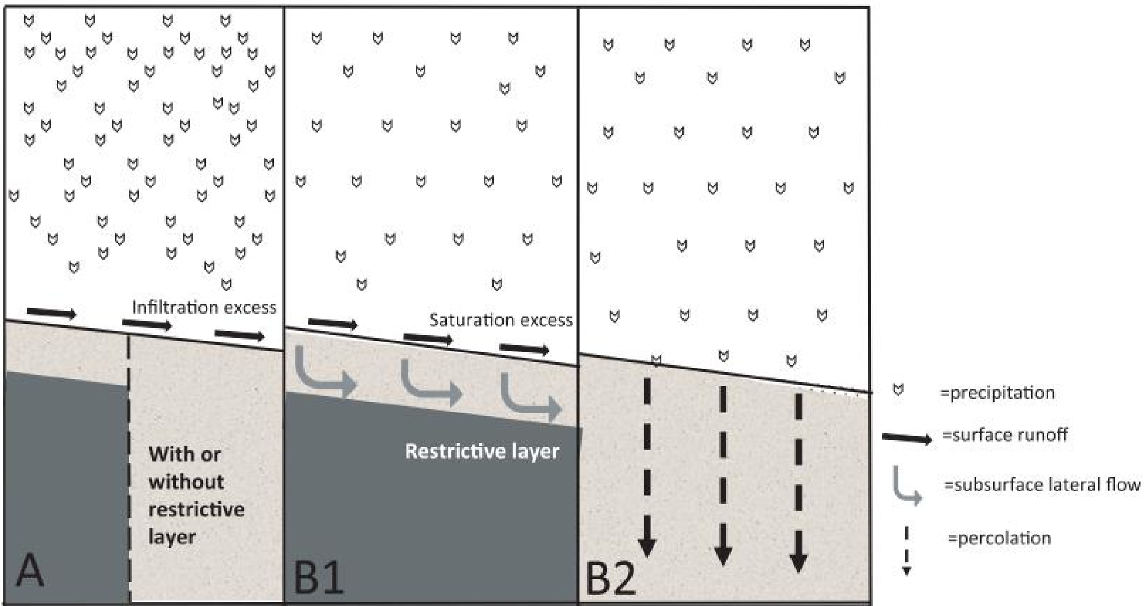


Figure 2.3: Classification of concentration (c) – discharge (Q) relationships in storm event hydrographs according to their export regime and hysteresis pattern. enr = enrichment; con = constancy; dil = dilution; cw = clockwise; nh = no hysteresis; acw = anticlockwise. Reproduced from Pohle *et al.* (2021).

Many anthropogenic influences on catchments affect hysteresis patterns, particularly due to augmentation of hydrological connectivity. For example, land drainage in arable catchments is often heavily modified through artificial drain features (see Figure 2.2), sometimes also called tile or field drains. These features have been shown to act as rapid pathways for delivering water and mobilised pollutants from critical source areas to watercourses via the subsurface (Heathwaite *et al.*, 2005; Deasy, Brazier, *et al.*, 2009). Agricultural catchments therefore typically have a lower ability to attenuate diffuse pollutants, and a limited capacity for natural processes such as denitrification which would normally occur in saturated soil given a longer residence time in the sub-surface (Collins *et al.*, 2017). Figure 2.4 presents a conceptual framework of different hydrological land types based on their dominant flow pathways and how this determines probable pollutant transport and the suitability of NBS (referred to as best management practices;

BMPs). Degradation of soil, ground and surface water is more likely to occur as a result of diffuse pollution where these pathways are intensified by anthropogenic influences.



	Precipitation intensity>infiltration capacity	Precipitation intensity<infiltration capacity	Precipitation intensity<infiltration capacity
Soil depth	Shallow or Deep	Shallow	Deep
Dominant Flow Paths	**Infiltration excess**, regardless of presence of restrictive horizon or slope	**Lateral flow** and/or **saturation excess; tile drainage**	**Percolation** and/or **tile drainage**
Probable Pollutant Transport	Pesticides, phosphorus, soil loss	Pesticides, nitrates, phosphates, soil loss	Pesticides, nitrates, phosphates
Effective BMPs	Mulch tillage, no tillage, terraces, CRP, buffers*, cover crops, NMPs, IPMs, constructed wetlands	Mulch tillage, no tillage, CRP, cover crops, tile drainage, NMPs, IPMs, constructed wetlands	CRP, buffers*, Cover crops, controlled drainage, crop rotations, forested buffers, NMPs, IPMs, constructed wetlands

Figure 2.4: A conceptual framework of pollution pathways in three different land types. Type A has a restrictive soil surface layer; type B1 has a high infiltration rate but restrictive layer in the shallow sub-surface; and type B2 has deep, well-draining soils with no restrictive layer in the sub-surface. NMP = Nutrient Management Plan; CRP = Conservation Reserve Program; IPM = Integrated Pest Management. Reproduced from Rittenburg *et al.* (2015).

2.3.4 Linking Catchment Processes, Pressures, and NBS

Section 2.3 has highlighted numerous hydrological processes and how these are impacted by pressures (particularly from agriculture). Table 2.1 provides a summary of the key hydrological processes and functions of catchments with examples of some of the anthropogenic pressures influencing them, and suggestions of which NBS may be suitable for mitigating these pressures.

Table 2.1 Natural hydrological processes/functions, the pressures within catchments that impact these processes, and potential NBS for addressing these pressures.

Process/function	Pressures	NBS
Infiltration & percolation	Agricultural intensification; compaction of soil; removal of vegetation	Woodland creation; run-off attenuation features
Run-off & overland flow	Mismanagement of agricultural land, urbanisation, reductions in surface roughness	Land and soil management (e.g. controlled traffic, cover crops)
Interception & evapotranspiration	Deforestation; overgrazing	Woodland creation
Flood storage & attenuation	Artificial straightening and embanking of river channels; floodplain disconnection; agricultural modifications of watercourses for land drainage purposes	Floodplain and river restoration (e.g. reconnection of river channels to riparian corridor and floodplains); re-meandering watercourses; run-off attenuation features; leaky barriers
Nutrient attenuation & biogeochemical processing	Wetland drainage/conversion to arable land; ditch creation; installation of field/tile drains	Arable reversion; ditch/drain blocking; peatland restoration; riparian wetland creation

2.4 Types of Nature-based Solutions

This section has been sub-divided into key categories of NBS within the realm of catchment management and water quality (with a focus on rural settings) to cover the wide-range of interventions, measures and strategies discussed within the literature. The first section (2.4.1) discusses NBS that primarily function to store water in the landscape e.g. wetlands. The following section (2.4.2) discusses how land and soil management can act as NBS. The final section (2.4.3) discusses the use of trees and instream wood as NBS. Due to the interconnected nature of NBS and WwNP, some of the categories discussed within sections are not mutually exclusive and so there is a degree of overlap in their scopes.

Figure 2.5 provides an overview of the main forms of NBS and their typical configuration within catchments. Interventions are either spatially diffuse or concentrated in their nature and target different parts of the catchment from source to downstream. NBS in the headwaters include catchment woodland, the implementation of instream structures e.g. leaky wooden dams, and non-floodplain wetlands and ponds. Further down in the catchment, measures on agriculturally productive land include management practices such as cover crops, as well as constructed interventions e.g. sediment traps. The lower reaches of river systems typically present opportunity for the restoration of river morphology (e.g. increasing sinuosity) and enhancing lateral connectivity between river channels and their floodplains.

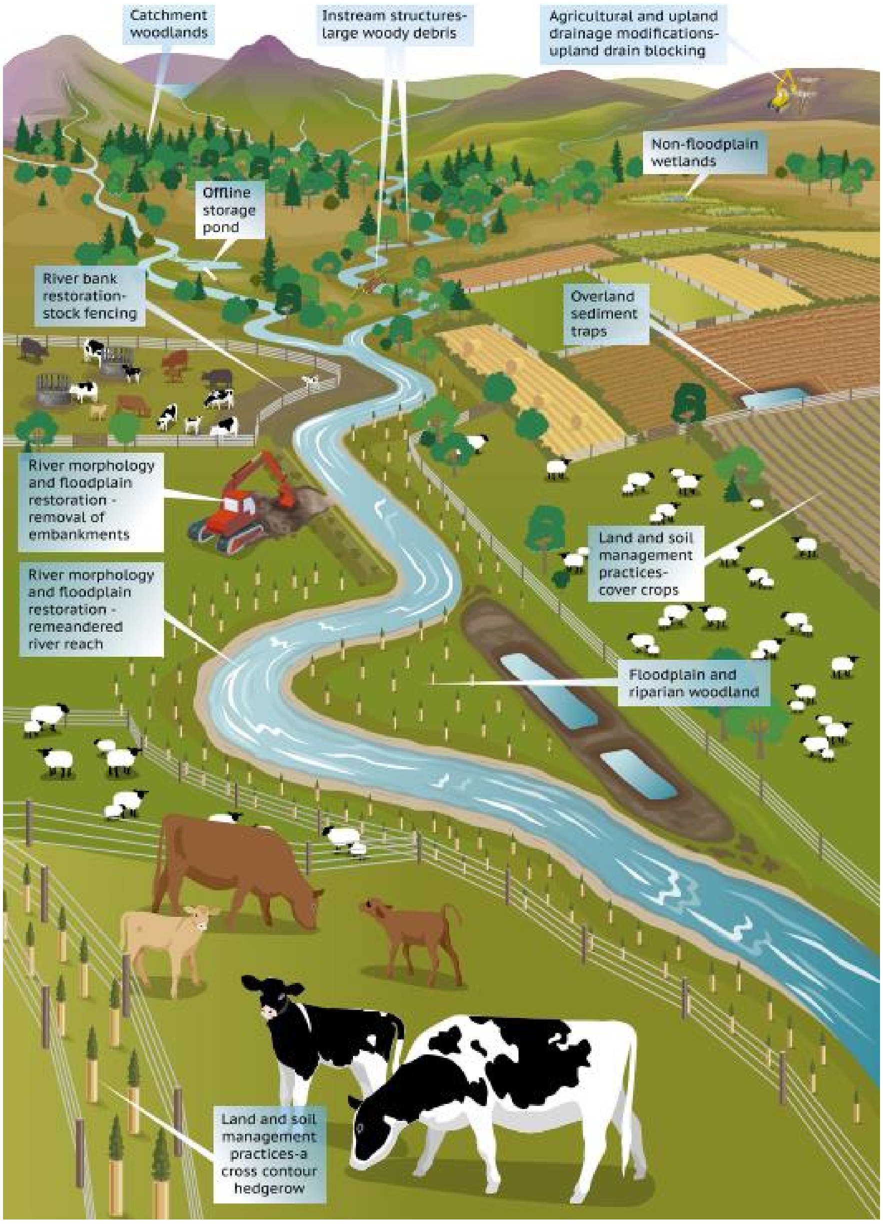


Figure 2.5: Diagrammatic representation of the different types of NBS that may be found within a typical UK catchment. Reproduced from Forbes *et al.* (2016).

The NBS shown in Figure 2.5 do not provide an exhaustive list of catchment interventions, with not all NBS being suitable or applicable to different catchment types e.g. upland peat catchments. The recent increase in the implementation of NBS (particularly NFM in the UK), has seen the development of guidance that details the broad range of measures available to catchment managers and practitioners e.g. the natural flood management manual (Wren *et al.*, 2022). The growing evidence-base and improvement in guidance is crucial in enabling further successful implementation of NBS.

2.4.1 Wetlands, Ponds, and Water Storage Features

A significant aspect of managing water in the natural environment is related to its storage within the landscape. Increasing water storage capacity in catchments (both above and below ground) aims to improve resilience and provide buffering against hydrological extremes (i.e. floods and droughts) (Global Water Partnership, 2021; Scanlon *et al.*, 2022). In natural systems, floodplains function to periodically retain floodwater and nutrients (Jakubínský *et al.*, 2021). Acreman and Holden (2013) suggest that floodplain wetlands have significant potential to reduce floods, however this is dependent on landscape-specific factors such as topography and soil characteristics. Wetland ecosystems play important roles in water storage and modulation, and also provide biogeochemical services including the transformation of nutrients through processes such as denitrification (Cheng and Basu, 2017; Jones *et al.*, 2018). The loss of such features within landscapes due to anthropogenic influences has therefore considerably changed catchment water storage (Richardson, 1994). NBS that focus on water storage aim to restore or mimic these natural functions. This can be achieved through approaches such as wetland restoration and floodplain reconnection which provide relatively large storage capacity, typically in the middle reaches of river systems. On a smaller scale, ponds, constructed wetlands and run-off attenuation features such as field corner bunds can provide spatially dispersed storge, typically in catchment headwaters.

2.4.1.1 Constructed Wetlands

Constructed (or artificial) wetlands (CW) are designed landscape features that mimic the functioning of natural wetlands and their ability to store and filter water. Fundamental components of a CW system include an inflow and outflow, a bed media such as sediment or gravel, and vegetation (typically including emergent macrophyte species). CW are used as cost-effective tools to treat and filter a variety of environmental pollutants from water and have been well-studied in the context of treating both point and diffuse sources including industrial wastewater, urban storm water, sewage treatment work effluent, and agricultural run-off (Koskiaho *et al.*, 2003; Ghermandi *et al.*, 2007; Lucas *et al.*, 2015; Ilyas and Masih, 2017; van Biervliet *et al.*, 2020). In the UK, the majority of CW are for tertiary wastewater treatment, accounting for 69 % of CW systems in the Constructed Wetlands Database as of 2008 (Lucas *et al.*, 2015). However, in recent decades there has been a growth in popularity to use CW for treating agricultural run-off and agro-industrial wastewater (Wang *et al.*, 2018).

Freshwater pollutants are diverse in their physical and chemical form and consequently have a variety of environmental and ecological consequences. In terms of agricultural systems, the research has focussed on the use of CW to remove pesticides, nutrients and sediment from field run-off (Vymazal and Březinová, 2015). Nutrient removal studies typically focus on phosphorus (P) fractions and nitrogen (N) species, due to excessive concentrations of these nutrients leading to nutrient enrichment and potentially eutrophication (Dodds and Smith, 2016). Studies that examine suspended solids typically focus on fine sediments (silt and clay sized particles) due to their potential to negatively impact instream habitats and freshwater ecosystem functioning (Owens *et al.*, 2005; Naden *et al.*, 2016; Mathers *et al.*, 2017). Fine sediment is also typically a key pollutant of interest as it acts as a vector for the transport of nutrients (e.g. P) and other contaminants (e.g. pesticides) that can be adsorbed to the surface of sediment particles (Poulenard *et al.*, 2008). The mitigation of excessive sediment loading into surface waters is becoming increasingly important due to the threat of exacerbated run-off and diffuse pollution under wetter winters in the UK as a result of climate change (Zhang *et al.*, 2022).

CW filter and remove pollutants through various physical, chemical, and biological mechanisms, transforming P into different dissolved or particulate forms, and N into different species throughout biogeochemical processes such as denitrification (Figure 2.6).

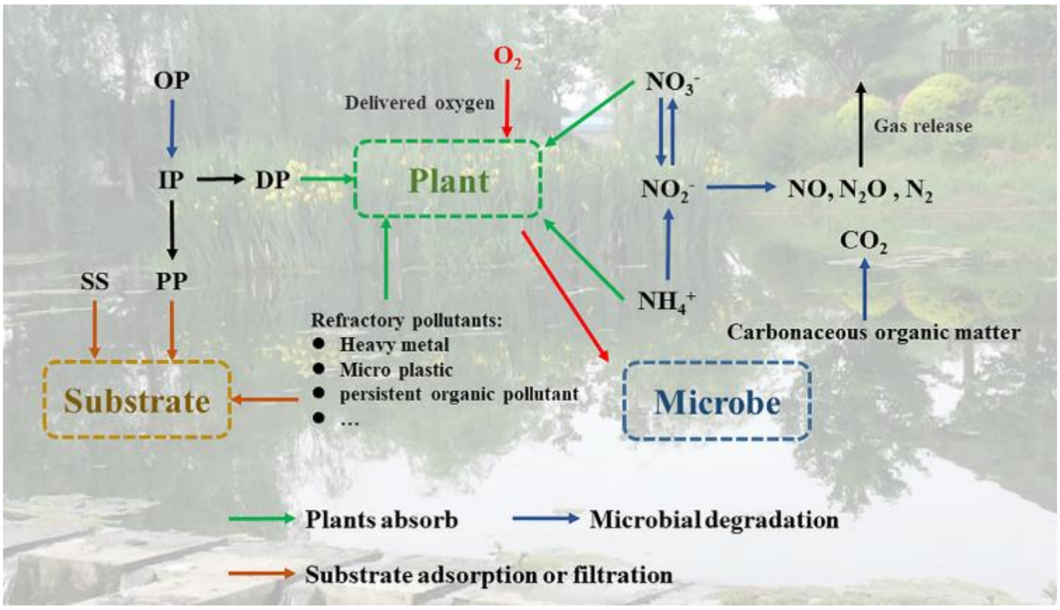


Figure 2.6: Pollutant removal mechanisms in typical CW/pond systems. SS = suspended solids; PP = particulate phosphorus; DP = dissolved phosphorus; IP = inorganic phosphorus; OP = Organophosphorus. Reproduced from Li *et al.* (2021).

Micro-organisms play a crucial role in nutrient transformation, particularly within sediment layers where microbial communities function to remove pollutants through catalysing chemical reactions, biodegradation and biosorption (Wang *et al.*, 2022). Similarly, plants play a role in trapping suspended solids and attenuating pollutants through processes such as flow velocity reduction and sedimentation effects, as well as phytomediation and nutrient uptake through roots (Vymazal, 2011b; Shelef *et al.*, 2013). Vegetation in CW has been shown to improve treatment efficiency for organic compounds and key macronutrients (N and P) (Vymazal, 2011b).

In a systematic review of on-farm CW, Newman *et al.* (2015) found that a variety of wetland types and designs were able to reduce suspended sediment (SS) and the vast majority of major nutrients. In terms of P, the reduction of SRP was highest at a mean of 97 %, with TP only at 78 %. The removal efficiency of both P fractions was found to improve with both the age and size of wetland. In terms of N species, CW were effective in reducing ammonia (NH_3) and ammonium (NH_4^+), with a mean reduction of 94 %. However, CW had no significant effect on nitrite (NO_2^-) removal, and nitrate (NO_3^-) was only reduced by CW with overland buffer strips and wetland vegetation. In terms of sediment, all CW types were found to be effective in reducing Total Suspended Solids (TSS). Newman *et al.* (2015) recommend that in order to remove at least 80 % of TSS, CW should have an area of over 2500 m^2. Whilst increasing CW area can improve its functioning as an NBS, this also reduces its practicality, particularly in an arable landscape where taking land permanently out of production can present significant economic losses to farm businesses. The trade-off between CW efficiency and agricultural productivity helps to drive the need to understand the best ways to optimise NBS to achieve environmental outcomes without impacting food production. This also highlights the importance of providing monetary compensation for the implementation of NBS through agri-environmental schemes, helping to reduce economic barriers and limitations on NBS efficiency.

Another factor which can influence the efficiency of CW is time. The pollutant removal efficiencies of a CW can change over small timescales (e.g. diel cycles (Dušek *et al.*, 2008)), but also more significantly over the longer term (e.g. annual scale), particularly during the initial years after implementation, or as a CW matures. It has been thought that N removal should increase over time as vegetation becomes established and sufficient organic carbon is available for denitrification (Kadlec and Wallace, 2009; Mustafa *et al.*, 2009). The stabilising effect of greater vegetation cover has also been observed to enhance sediment retention by mitigating resuspension (Braskerud, 2001). Kill *et al.* (2022) found that P removal was greater and more stable in CWs with a higher density of wetland vegetation. In their review, Newman *et al.* (2015) found that older CW (>18 months) had higher removal efficiencies of both SRP and TP. A newly established in-stream CW treating diffuse agricultural pollution in Estonia was found to have greatly varying removal efficiencies for different nutrients during its two year acclimatisation period (Kasak *et al.*, 2018). This CW was effective in reducing phosphate by up to 42 % during the warm season, and by up to 15 % annually, but for nitrate and total nitrogen significantly higher concentrations and loads at the CW outlet were observed. The study concluded that increases in nitrogen were likely caused by groundwater seepage, highlighting the importance of considering local hydrogeology when designing and locating CW in order to avoid adverse effects on water quality. In contrast, a four-year study of a CW in Serbia found that over the monitoring period, the concentration of suspended solids in the outflow did not change significantly over time (Josimov-Dunderski *et al.*, 2013). However, four years is relatively short when considering the overall intended lifetime of a CW. The contrasting findings highlight the need for longer term monitoring of such features in terms of both their pollutant removal efficiency and also their morphological and ecological evolution and potential feedback mechanisms that may arise as a consequence. The effect of CW management is also likely to influence functioning over time, however this is often not mentioned within the literature, even for longer-term studies where maintenance activities would typically be expected within the timeframe. Acreman and Holden (2013) suggest that removal of wetland vegetation reduces friction and potential floodplain attenuation of flood waves. Further evidence is needed to examine the impacts of wetland management on biogeochemical functions of CW and their pollutant removal efficiency.

CW are often designed to address specific catchment issues which makes the spectrum of different designs wide-ranging. For example, in the Struga Gnieźnieńska catchment (western Poland) they have been used as a preventative measure to protect lake ecosystems from eutrophication. Monitoring carried out by Kupiec *et al.* (2022) found that the novel sedimentation and biofiltration (SED-BIO) system constructed at the inflow of Jelonek Lake was able to reduce NO_3^- and PO_4^{3-} concentrations by 63 % and 19 % respectively. The SED-BIO system which consisted of multiple ponded sections with gabions, flow deflectors and vegetated zones was also able to reduce the number of potential pathogens flowing into the lake with an observed reduction in the number of faecal indicator bacteria at the system's outflow. This example illustrates an effective use of a CW to achieve multiple benefits where elements of the design were targeted to maximise different natural processes. There is a continued need to understand what designs of CW are most appropriate and effective for different catchment settings and purposes, and also how their efficacy may change over time as a result of management and geomorphological development.

2.4.1.2 Run-off Attenuation Features and Offline Storage Areas

Another type of NBS feature that aims to capture and filter water are offline storage areas and run-off attenuation features (RAFs) which are designed to operate during storm events to temporarily divert streamflow from channels and/or capture overland flows generated by run-off (Wilkinson *et al.*, 2014; Nicholson *et al.*, 2020). RAFs are typically small-scale, located within fields on grassland pasture or more specifically located in field-corners on arable land where their configuration is more restricted by the land-use. These features are targeted towards intercepting areas of high run-off generation such as farm tracks, which have also been shown to be critical source areas of sediment (Reaney *et al.*, 2019). However, most research on RAFs focusses on their use as an NFM intervention to deliver flood risk mitigation (Quinn *et al.*, 2013; Nicholson *et al.*, 2020; Lockwood *et al.*, 2022). The additional benefits for diffuse pollution mitigation have also been recognised (e.g. Barber and Quinn (2012), Wilkinson *et al.* (2014)), but not quantified as fully. This is in part due to the difficulties in obtaining robust field-scale measurements of both flow and pollutant concentrations at fine temporal resolutions (Lloyd, Freer, Johnes, Coxon, *et al.*, 2016), especially when the nature of pollution events can be highly dynamic and their sources are diffuse. Consequently, much of the current evidence is derived from studies that model the potential effects of RAFs, such as modelling by Adams *et al.* (2018) who found that adding 2000-8000 m^3 storage in the Newby Beck catchment (northwest England) provided a decrease of 5-10 % in event peak SS and TP concentrations at the outlet.

Within the UK there are several examples of where the implementation of offline storage features has been targeted towards diffuse pollution mitigation through the capture of sediment and nutrient-rich run-off from fields (Ockenden *et al.*, 2012, 2014). Detailed research funded by the Department for Environment, Food and Rural Affairs (DEFRA) on ten small (<0.1 % of catchment areas) edge-of-field features was conducted at 4 sites (Cumbria and Leicestershire) with varying soil textures. Results from three years of monitoring through the use of accumulation surveys found that all of the features were functioning to trap sediment and associated macro-nutrients, however rates between individual features varied considerably. This variation was thought to be largely due to the soil types, with features at sandy sites trapping 0.5-6 t ha^{-1} yr^{-1} compared to only 0.01-0.07 t ha^{-1} yr^{-1} at clay sites. These findings have implications for the management of such features, particularly where they have been implemented for the dual purposes of pollution mitigation and flood storage. Concerns over the effect of sediment accumulation impacting the ability of NBS (namely NFM interventions) to provide sufficient flood storage over time have been raised (Lane, 2017), and warrant further study into their maintenance requirements. Quantifying accumulation rates across a spectrum of different NBS will help to provide evidence needed to manage and maintain NBS more cost-effectively whilst also optimising their functionality.

2.4.1.3 Beaver Dams and Ponds

The reintroduction of beavers into the landscape is discussed here as a NBS in light of the role these animals play as ecosystem engineers and their ability to alter the storage and flow of water through the creation of dams (Brazier *et al.*, 2021). Due to growing evidence of their potential benefits for ecosystems, the re-introduction of the Eurasian beaver (*Castor fiber*) in England is currently being considered by the UK government (at the time of writing) following a public consultation in 2021 (DEFRA, 2021). Research has found that beaver activity can have a wide range of benefits including water quality improvement; aquatic macroinvertebrate biodiversity; sediment, carbon and nutrient storage; and flood attenuation (Law *et al.*, 2016; Puttock *et al.*, 2017, 2018, 2020; Graham *et al.*, 2022).

A considerable body of evidence is emerging from beaver reintroduction trials such as in the River Otter catchment (Devon, England) where extensive research has taken place, including monitoring of the effects on hydrological processes (Brazier *et al.*, 2020). Key findings from this five-year project include a downstream reduction in peak flows as a result of increased storage from dams, as well as improvements to water quality downstream. On one river reach, 13 beaver-created impoundments were able to reduce concentrations of suspended sediment, total oxidised nitrogen and phosphate, as well as reducing stream discharge by 34 % on average during storm events (Puttock *et al.*, 2017). In terms of water quality, previous studies carried out in other regions corroborate some of these findings. A field study in Scotland found that the mean concentrations of extractable P and NO_3^- during the growing season were 49 % and 43 % lower downstream of a series of four beaver dams (Law *et al.*, 2016). However, this study also found total suspended solids increased by 5.8 times on average downstream of the dams; this was attributed to the beavers burrowing unconsolidated bankside material. Using ecosystem engineers such as beavers is therefore arguably a riskier form of NBS given their unpredictability and potential to cause unwanted consequences. Further evidence on the long-term implications of beavers for water quality (and wider ecosystem services) would be beneficial for improving decision-making on their suitability for use in catchment management.

2.4.1.4 Potential Disbenefits

In addition to the positive aspects of NBS that focus on enhancing water storage, their potential disbenefits and trade-offs are also important to consider for sustainable catchment management (Hewett *et al.*, 2020). The main disbenefit of CW, ponds and other water storage features arises from their potential to cause unintentional pollution in other forms such as the release of greenhouse gases (GHG), namely methane (CH_4), carbon dioxide (CO_2) and nitrous oxide (N_2O). Freer (2015) found that the mean net releases of all three GHG from a CW in Cumbria were significantly higher than releases from adjacent riparian land. Additionally, evidence from experimental microcosm studies identified the disturbance of CW bed sediment by stormflows as a potential source of heightened nutrient and GHG releases. Recommendations from this research included the use of shallower, more vegetated wetlands that are better able to protect sediments from stormflows. Small temporary ponds were also found to be sources of CO_2, with emissions linked closely to changes in water level and exposed sediment (Obrador *et al.*, 2018). The drying-rewetting cycles that freshwater sediments undergo due to oscillations in water level may be exacerbated by the effects of climate change, further promoting GHG and nutrient release (Paranaíba *et al.*, 2020). A better understanding of the biogeochemical cycles operating within NBS that store water and sediment would aid their management and reduce negative impacts.

Hydrochemical and meteorological monitoring over 20 years showed that a lowland wetland-pond system was able to become a source of bioavailable nutrients during low flow periods (Jarvie *et al.*, 2020). This highlights the need for monitoring of such features over time to ensure they continue to operate as intended. This work also demonstrated the importance of accumulated nutrient legacies in wetland systems and their potential to cause water quality impairment and accelerate eutrophication risk without appropriate management. Drivers of nutrient release in the wetland-pond system included primary production and the accumulation of biomass in bed sediment. Figure 2.7 exemplifies some of the biogeochemical interactions between the different forms of P in water and sediments within freshwater systems.

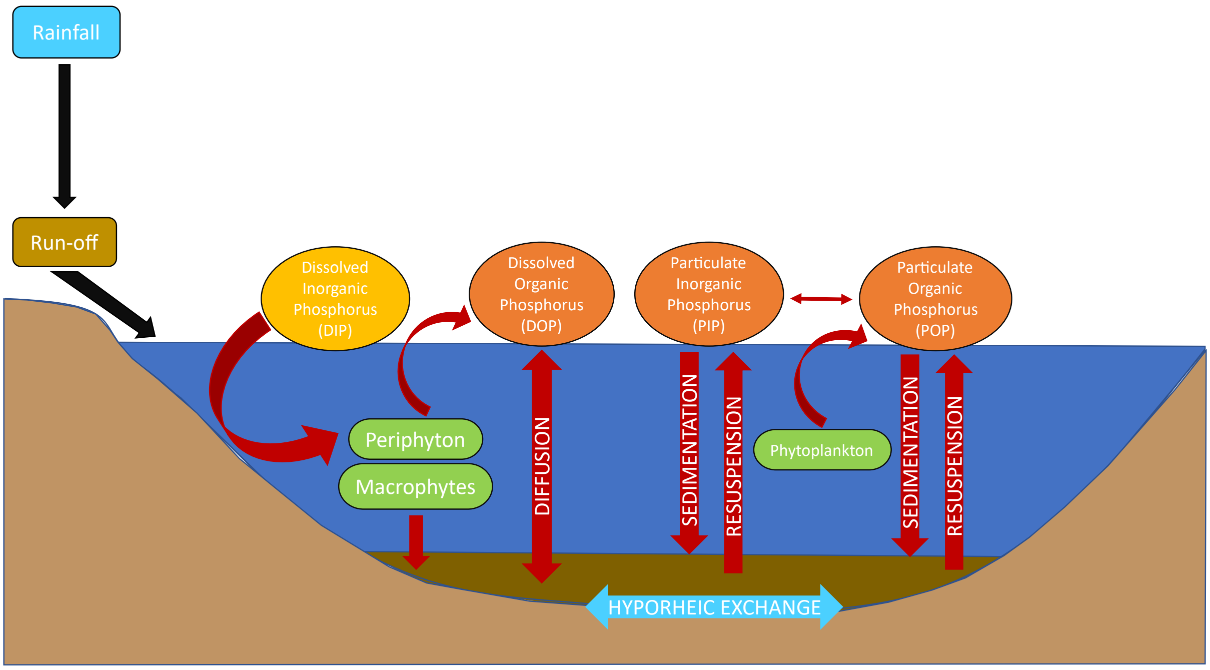


Figure 2.7: Simplified conceptual model of biogeochemical cycling of P in a freshwater system. TDP = Total Dissolved P, DHP = Dissolved Hydrolysable P, TP = Total P, PP = Particulate P.

Trade-offs can also been seen in NBS that are aimed at trapping sediment in run-off. A study of run-off attenuation ponds observed that during a large storm event, previously accumulated material within a feature was remobilised and flushed into the receiving stream (Barber and Quinn, 2012). This phenomenon presents a key risk to the efficacy of such features to mitigate pollution (Cooper *et al.*, 2019). Research into the sediment dynamics of different water storage NBS under different hydrometeorological conditions (including extremes) is needed to understand how they may behave as pollutant sources or sinks, when these states occur, and how potential risks can be reduced.

2.4.2 Land and Soil Management

The sustainable management of land and soil within catchments plays an important role in mitigating diffuse pollution due to the fundamental effect land-use has on soil erosion rates (Kosmas *et al.*, 1997). Human activity has accelerated erosion through the conversion of a significant cover of the landscape from plagioclimax woodland communities (in temperate climates) into settlements and agricultural land (Boardman, 2013).

The concern for soil erosion in the British Isles and the potential need for conservation and management of soils was first raised by Evans (1971) when evidence on this issue was still sparse. Following a period of increased scientific and societal interest, this topic was revisited by Boardman (2002), who concluded that the impacts of soil erosion and run-off are "*worthy of serious (if belated), attention*". Globally, the topic of soil erosion and conservation has become an environmental issue of increasing importance, particularly in the context of climate change and food security. The importance of soil structure and management have also been examined in terms of contributing to run-off and flood generation (Holman *et al.*, 2003). A more recent study modelling the potential impacts of land use and climate change on soil erosion by water suggests that a more energetic hydrological cycle in future (2015-2070) could increase erosion by 30-66 % globally (Borrelli *et al.*, 2020). The authors also estimated that erosion rates will be decreased through current conservation agriculture practices by approximately 5 %. This highlights the need for further action on soil conservation if we are to mitigate the impacts of climate change on soils. Collins and Zhang (2016) conclude that the current (2013-2014) low uptake of soil erosion control measures on farms in England and Wales is achieving limited benefits in reducing fine sediment delivery to rivers overall. Further evidence on the efficacy of control measures may persuade more farmers to introduce control measures, but also help to influence policy changes on payments for environmental stewardship and thereby increase uptake.

2.4.2.1 Land-use Change

Due to government policy changes and technological advancements, farming in Britain became increasingly intensive after 1945 (post World War II) with significant decreases in landscape diversity and increases in cereal crops (Robinson and Sutherland, 2002). The primary drive for maximising food production largely neglected considerations into the potential negative environmental consequences of using land for crops. Arguably the optimal solution to soil erosion on agricultural land from a pollution mitigation perspective is to greatly reduce the risk factor by reverting high risk land-use such as winter cereals back to a semi-natural state such as permanent grassland which is vegetated all year round. Arable reversion was found to be one of the most common management strategies in a survey of farms in the South Downs National Park, UK (Boardman *et al.*, 2017). Reverting arable land may require a low input of resources, however grasslands are diverse in their form and function, which has implications for the effectiveness of such land-use change measures. Research has shown that natural unimproved *Molinia caerulea* grassland had a significantly lower drainage density and greater roughness compared to improved grassland dominated by *Lolium perenne* (perennial ryegrass) (Ellis, Brazier, *et al.*, 2021). The more sinuous hydrological pathways within unimproved grasslands would also therefore help to reduce the mobilisation and transport of soil particles (Werner and Zedler, 2002). Given these findings, it would be most beneficial and effective for NBS strategies to conserve existing natural grassland where possible, rather than focussing solely on reversion to improved grassland.

Land-use change on a larger spatial scale is much rarer within the British Isles due to the high demand for land, however the concept of rewilding has gained momentum in Britain in recent years. Rewilding could potentially facilitate the mitigation of soil erosion through reducing disturbance and restoring natural processes, though current evidence suggests that rewilding has a complex set of associated environmental and societal benefits and risks (Sandom *et al.*, 2019). There is currently a lack of evidence regarding the benefits of large-scale rewilding for water quality and other potential ecosystem services in the UK.

2.4.2.2 Sustainable Soil Management

Kreiling *et al.* (2018) argue that there is currently a disconnect between management of the land and management of river networks, and that there is a need for better integration of how land and soil are managed sustainably for both agriculture and for freshwater ecosystems through targeted and effective best management practices. There are a variety of management strategies and techniques aimed at reducing run-off and soil erosion, though the evidence on their effectiveness is mixed (Rickson, 2014). Conservation agriculture and regenerative agriculture are both approaches that aim to manage land so that it benefits soil health (Mitchell *et al.*, 2019; Lal, 2020). These approaches aim to farm whilst causing minimal soil disturbance, having maximum soil cover, and increasing crop diversity (Page *et al.*, 2020). A common method for achieving minimal soil disturbance is through no-till (also called zero tillage) or minimum till agriculture which intends to optimise both crop productivity and ecosystem services (Derpsch *et al.*, 2010). A study in Brazil found that soil erosion decreased from 3.4 – 8 t ha^{-1} under conventional tillage to only 0.4 t ha^{-1} when under no-till, with the additional benefit of reduced water loss of approximately 820 t ha^{-1} (Derpsch, 2008). Keesstra *et al.* (2016) showed that soil erosion was significantly greater in both conventionally tilled fields and fields treated with herbicide when compared to plots with cover from plants, litter, or chipped wood. Where phosphorus pollution is an issue of concern, mitigation is typically focussed on reducing the diffuse delivery of the particulate form of P in run-off due to soil having a high sorption affinity for P (Dupas *et al.*, 2015). However, Badon *et al.* (2022) found that minimum till combined with cover cropping had no effect on surface run-off with respect to N or P loading and loss of suspended solids from fields with a corn and soybean rotation in the mid-southern United States. A meta-analysis comparing nitrate concentrations and loads from no-till and conventional tillage systems found that no-till resulted in overall higher nitrate concentrations in run-off, but similar loads (Daryanto *et al.*, 2017). The meta-analysis also revealed that nitrate loads lost through leaching were higher from no-till, with this largely being driven by changes in water flux. These findings highlight how soil management-related NBS can operate differently depending on the pollutant in question and the hydrological pathways involved.

Another commonly implemented soil management strategy is the planting of cover crops which aims to increase interception and evapotranspiration and thereby reduce raindrop impact and the mobilisation of soil particles in run-off (Sharma *et al.*, 2018). For example, planting cover crops such as field mustard (*Brassica rapa*) during winter can increase interception and infiltration rates and enhance surface roughness, collectively reducing the risk of soil erosion (Rittenburg *et al.*, 2015). A review study looking at temperate soils found that cover crops have the potential to reduce sediment losses by between 40 and 96 % depending on the plant biomass and the species planted (Blanco-Canqui *et al.*, 2015). Cover crops of rye and oat were found to reduce rill erosion by 54 and 89 % respectively, showing consistency across a three-year experimental field study in the midwestern United States (Kaspar *et al.*, 2001). Cover crops have also been reported to reduce dissolved nutrient losses (e.g. nitrate) in run-off, and also losses from leaching (Abdalla *et al.*, 2019). However, Blanco-Canqui (2018) concluded that cover crops are much less effective at reducing overland dissolved nutrient transport compared to leaching, with overall effectiveness being structured in the following order: nitrate leaching ≥ sediment > run-off > dissolved nutrients in run-off. This study also determined that there is still a need for further evidence on how the specific management aspects of cover crops (e.g. planting and termination dates) influences their effectiveness at conserving water quality. Furthermore, there is also a lack of evidence on the combined effects of such soil management approaches with other forms of NBS (e.g. water storage interventions), particularly beyond the field scale.

2.4.2.3 Edge-of-field Features

Conservation agriculture also considers the implementation of edge-of-field features such as riparian buffer strips which are often implemented and managed as part of agri-environment schemes (Bullock *et al.*, 2021). Particularly in arable fields, the lack of a riparian zone can degrade water quality because unlike natural systems there is no vegetative buffer to help attenuate pollutants transported in run-off (Norris, 1993). Due to their widespread use, riparian buffers are well-studied in terms of their benefits. Cole *et al.* (2020) reviewed the management of riparian buffer strips for the optimisation of ecosystem services and found that their effectiveness at mitigating diffuse pollution was positively related to their width (Knight *et al.*, 2010; Zhang *et al.*, 2010; Rasmussen *et al.*, 2011). An experimental field study found that buffer strips effectively removed sediment at a width of 7.5 m, whereas greater buffer widths of 15 m were recommended for the removal of dissolved nutrients (Schmitt *et al.*, 1999). Other studies have yielded similar results, finding that sediment trapping efficiencies tailed off after 10 m widths (Dorioz *et al.*, 2006; Liu *et al.*, 2008). The species composition of vegetation within buffer strips has also been shown to influence their efficacy, with forested strips being more efficient at removing nitrate compared to grass strips (Osborne and Kovacic, 1993). Furthermore, Stutter *et al.* (2019) highlight how the functions of buffer strips can take several years to develop due to vegetation maturation and habitat development, and therefore monitoring their effectiveness ideally requires long study periods that would capture these changes.

Simple vegetated buffer strips have been applied in agricultural settings for decades, but more recently there has been an interest in integrated buffer zones with more designed elements that aim to interrupt the flow of pollutants and also increase structural diversity in the habitat (Stutter *et al.*, 2019). One such example is the concept of three-dimensional (3D) buffer zones that operate both above and below ground to intercept pathways of pollution through design features such as deep roots, re-sculptured ground, and enhanced canopy cover (Environment Agency, 2020). 3D buffers, whilst forming part of a 'treatment train', are acknowledged to only be part of an effective solution that also involves broader soil erosion and pollution control measures. For example, Osborne and Kovacic (1993) noted that buffer strips are less likely to be effective in areas with considerable field drainage, meaning that in these areas additional measures are a necessity. Stevens and Quinton (2009) reviewed a range of mitigation measures for reducing diffuse pollution from agricultural systems (including no-till, cover crops, and riparian buffer zones), and concluded that no single option will reduce all pollutants. This emphasises the reoccurring theme of the need for multiple measures across different parts of the catchment, targeting source areas that are most at risk for the best chances of successful mitigation. Further evidence on the effectiveness of different combinations of interventions in different settings would therefore be beneficial.

2.4.3 Trees, Woodland, and Instream wood

The widely recognised benefits of trees for carbon sequestration has meant that tree planting is a commonly used NBS in the UK, particularly due to policy drivers surrounding the need to offset greenhouse gas emissions and meet net zero targets (Seddon *et al.*, 2021). Furthermore, trees and woodlands are commonly implemented in NFM schemes due to their influence on catchment hydrology and water cycling through modifying processes such as infiltration, evapotranspiration, and properties such as surface roughness (Stratford *et al.*, 2017; Page *et al.*, 2020; Baker *et al.*, 2021; Monger *et al.*, 2022).

Trees can also have impacts on stream water quality and ecology when considering the function of riparian woodland in regulating temperature, providing woody debris inputs, and retaining sediment (Studinski *et al.*, 2012). Stream water temperature and thermal regimes regulate many freshwater ecosystem processes such as metabolism and nutrient cycling which can in turn influence eutrophication risk and overall ecological status (Caissie, 2006). An analysis of historic records of stream and river temperatures in the United States shows statistically significant warming in 20 major watercourses, with the most rapid warming occurring in urbanising areas (Kaushal *et al.*, 2010). This trend of warming is expected to continue with further global climate change and urbanisation, making it an increasing area of concern and need for effective adaptive management strategies (Garner *et al.*, 2014). A systematic review of the effects of wooded riparian zones in on stream temperature in temperate climates found that they are able to reduce both spring and summer temperatures, with the greatest effect observed being a reduction of maximum temperatures (Bowler *et al.*, 2012). Halliday *et al.* (2016) found that shading from riparian tree canopies is a likely cause in suppressing the growth of benthic algae in the River Enborne, a river system heavily dominated by riparian shading. The water quality effects of riparian shading have also been modelled. For example, a study using photogrammetry-derived shading data for the River Thames found substantial differences in biochemical oxygen demand (BOD) and water temperature between shaded and non-shaded radiation inputs (Bachiller-Jareno *et al.*, 2019). Instream wood has also been found to modify riverbed temperature by increasing spatio-temporal variability and promoting greater localised geomorphic diversity, therefore having potential implications for both hyporheic zone biogeochemistry and streambed-dwelling communities (Klaar, Shelley, *et al.*, 2020).

Large woody debris (LWD), despite being a natural feature of watercourses, has often been perceived as a nuisance and potential risk for causing blockages and thereby impeding the flow of floodwater downstream (Chin *et al.*, 2008). However, a key intention of many NFM, WwNP, and river restoration initiatives is to use LWD to 'slow the flow' of water through a catchment (Grabowski *et al.*, 2019), flatten the shape of the flood hydrograph and crucially reduce its peak. A study of LWD installed in headwater streams in Germany found that LWD was able to delay flood wave progression and attenuate flood peaks and as a consequence reduce erosive processes and in-channel sediment transport rates (Wenzel *et al.*, 2014). In-stream interventions such as LWD could therefore be beneficial in protecting against hydromorphological damage to channels such as incision. The secondary effects of in-stream wood structures (installed for NFM purposes) on hydro-geomorphological processes have been reviewed to help infer the potential impacts of these features beyond their primary purpose of flood risk reduction (Lo *et al.*, 2021). Findings showed that LWD can increase the storage of particulate organic matter through multiple mechanisms including direct interception, pool formation and settling, and upstream accumulation in fine grains. Additionally, LWD has shown potential to alter biogeochemical processes and provide ecosystem services such as nutrient attenuation in lowland river systems (Krause *et al.*, 2014; Blaen *et al.*, 2018). Submerged LWD has potential to remove dissolved nutrients from streams, with studies showing phosphate and ammonium removal rates at least two times greater in reaches with LWD compared to those without (Warren *et al.*, 2007; Acuña *et al.*, 2013). However, research suggests that the influences of LWD varies between different stream settings and may not always result in the reduction of dissolved nutrients (Shelley *et al.*, 2017). For example, LWD can help to induce higher biogeochemical turnover rates in lowland streams due to increased residence times, whereas in steeper upland systems the impact is less notable (Krause *et al.*, 2014). Further monitoring is needed to better understand these effects and the potential use of LWD in different settings (particularly lowland streams) to modify biogeochemical processes under varying hydrological conditions.

2.5 Methods of Assessing NBS Effectiveness in Mitigating Diffuse Pollution

In order to determine to the effectiveness of NBS at mitigating diffuse pollution, field data are most often required; concentrations and masses or loads of the pollutants in question are commonly gathered and compared spatially or temporally, or to a control. Field data are not always necessarily available for locations or scales of interest and so numerical models are also used to represent NBS and estimate their potential impact on a catchment under different scenarios or conditions. Evidence on the effectiveness of NBS is largely derived from quantitative methods (rather than qualitative) due to policy-makers favouring quantitative evidence (Wilkinson *et al.*, 2014). Table 2.2 provides a summary of some of the methods that have previously been used to assess NBS efficacy, including their advantages, disadvantages, and examples from the literature.

Table 2.2: Methods of assessing the efficacy of NBS for mitigating diffuse pollution and determining impacts on catchment water quality and sediment dynamics.

Method	Uses & Advantages	Limitations & Disadvantages	Examples
Sediment traps (mats, saucers etc.)	Useful for long-term evaluation of NBS; passive method – does not require frequent sampling; inexpensive	Measures net accumulation – could be influenced by resuspension; can underestimate accumulation due to sediment loss during removal process	Ockenden *et al.* (2012, 2014)
Surveying of NBS (e.g. accumulation in ponds, changes in channel geomorphology)	Can be carried out infrequently (e.g. annually) to measure long-term sediment accumulation rate and change over time; can easily examine spatial differences	Requires expensive equipment (e.g. RTK GPS) for high accuracy measurements; surveying involves considerable disturbance to NBS; unsuitable for features with deep water	Ockenden *et al.* (2012, 2014)
Sediment coring (e.g. in ponds)	Useful for long-term evaluation of NBS; does not require a comparison (e.g. control or before) for determining NBS functioning	Unsuitable for measuring small accumulations within NBS	(Verstraeten and Poesen, 2001)
Algae sampling	Monitoring algal biomass instream provides indication of water quality over longer period of time	Cannot necessarily isolate the effect of specific NBS interventions; potential lag time between NBS and effect on algal biomass	Mason-McLean (2020); Paul *et al.* (2017)

Porous ceramic cups (soil pore water sampling)	Useful for understanding effect of NBS on solute (e.g. nitrate) concentrations & transport; can be used to examine vertical variation along soil profiles	Only samples at a small scale and can underestimate spatial variability; effectiveness of the method varies with soil type/texture	Curley *et al.* (2011); Gaimaro *et al.* (2022); Cooper *et al.* (2017)
Water sampling (spot/grab samples)	Useful for spatial studies & identifying pollution source areas; useful for verifying in situ sensor data	Only provides discrete 'snapshots' which can be difficult to interpret in highly dynamic systems	Barber and Quinn (2012); Cooper *et al.* (2017)
Water sampling (automatic/flow-weighted)	Can observe changes in pollutant concentrations over time (e.g. during an event)	Logistically difficult for remote field sites; samples require collecting & transporting quickly (<24 hours) for chemically unstable determinands (e.g. orthophosphate)	Barber and Quinn (2012); Uuemaa *et al.* (2018)
In situ water quality sensors & hydrometric monitoring	Allows collection of high temporal resolution data; useful for observing dynamic systems and cyclical/seasonal patterns	Expensive; frequent maintenance/calibration required for high quality data	Wilkinson *et al.* (2014); Cooper *et al.* (2019)
Catchment-scale models (e.g. SHETRAN, SWAT)	Allows testing of future scenarios not possible using empirical methods; well suited to large-scale NBS (e.g. land-use change)	Spatial resolution of models is often too coarse to represent small field-scale interventions	Adams *et al.* (2018)
Empirical & theoretical models of NBS	Allows testing of the effect of different designs on NBS efficacy	More difficult to model complex designs of NBS accurately; empirical models not as good for small features	Verstraeten and Poesen (2000)

2.6 Summary

This chapter has highlighted the importance of NBS, particularly in light of the need to meet policy targets and to adapt to climate change impacts on freshwater ecosystems, water resources and food production. The chapter also summarised key hydrological processes that underpin our current understanding of catchment hydrology and diffuse pollution. Current evidence suggests that there is a need for further interventions to mitigate diffuse pollution and the ecological degradation of watercourses in order to meet regulatory targets and maintain ecosystems sustainably. The need for mitigation is currently high due to the legacy of agricultural practices and intensification which is typical of lowland catchments in the UK.

By reviewing existing knowledge on NBS, their potential to mitigate diffuse pollution, and different methods by which their efficacy can be assessed, the following research gaps were identified:

1) Further evidence on the effectiveness of NBS interventions, particularly those aimed at delivering multiple benefits, is needed to refine best practices and inform future policy decisions regarding payments for environmental land management.
2) There is a clear lack of empirical evidence on the effectiveness of NBS (particularly those aimed at water storage) for mitigating diffuse pollution. This warrants the need for detailed monitoring of NBS.
3) NBS and NFM interventions are less frequently studied in lowland catchments compared to upland catchments in the UK.
4) The effectiveness of NBS in delivering benefits can be highly spatially and temporally variable and may depend on catchment characteristics, design, age and hydrometeorological conditions. There is therefore a need to examine these controls further in a wider variety of contexts.
5) Current evidence on the combined effects of NBS on stream water quality is limited (particularly at larger spatial scales e.g. beyond second-order streams).
6) There is a lack of evidence on the long-term functioning of NBS, the maintenance requirements of such interventions, and the influence of NBS maintenance and management on their efficacy.

The following thesis chapters aim to address these knowledge gaps to build upon our current understanding of NBS and their potential to mitigate diffuse pollution.

Chapter 3 Sediment and Nutrient Retention in Ponds on an Agricultural Stream: Evaluating Effectiveness for Diffuse Pollution Mitigation

3.1 Abstract

The creation of ponds and wetlands has the potential to alleviate stream water quality impairment in catchments affected by diffuse agricultural pollution. Understanding the hydrological and biogeochemical functioning of these features is important in determining their effectiveness at mitigating pollution. This study investigated sediment and nutrient retention in three connected (on-line) ponds on a lowland headwater stream by sampling inflowing and outflowing concentrations during base and storm flows. Sediment trapping devices were used to quantify sediment and phosphorus accumulations within ponds over approximately monthly periods. The organic matter content and particle size composition of accumulated sediment were also measured. The ponds retained dissolved nitrate, soluble reactive phosphorus, and suspended solids during baseflows. During small to moderate storm events, some ponds were able to reduce peak concentrations and loads of suspended solids and phosphorus; however, during large magnitude events, resuspension of deposited sediment resulted in net loss. Ponds filtered out larger particles most effectively. Between August 2019 and March 2020, the ponds accumulated 0.306 t ha^{-1} sediment from the 30 ha contributing area. During this period, total sediment accumulations in ponds were estimated to equal 7.6 % of the suspended flux leaving the 340 ha catchment downstream. This study demonstrates the complexity of pollutant retention dynamics in on-line ponds and highlights how their effectiveness can be influenced by the timing and magnitude of events.

3.2 Introduction

Intensively farmed landscapes can contribute significantly to the degradation of the water environment globally (Moss, 2008; Withers and Jarvie, 2008; Khatri and Tyagi, 2015). In many European countries, agricultural intensification has increased the risk of waterbodies failing to meet the EU Water Framework Directive (WFD) objective of '*good ecological status*' (2000/60/EC) (European Commission, 2000; Collins and Anthony, 2008; Holden *et al.*, 2017). For streams and rivers in England and Wales, diffuse sources of pollution from agriculture present one of the biggest threats to WFD failure, with key concerns being elevated nutrient concentrations, oxygen depletion and the smothering of instream habitats by fine sediment (Environment Agency, 2007). One of the main delivery mechanisms for diffuse pollution is soil erosion caused by surface run-off which is then exacerbated when arable fields are left bare or subject to soil compaction (Fullen, 1985; Holden *et al.*, 2017). Diffuse agricultural pollution in the UK is estimated to have an annual cost of £238 million, resulting from reduced water quality and associated treatments costs (Jacobs UK Ltd., 2008). These costs consider the wide-ranging negative impacts on biodiversity, ecosystem services, landscape value, rural public access and enjoyment, water and air quality, and natural resources. Additionally, soil erosion can lead to increased sedimentation in watercourses and a reduction in channel capacity, thereby increasing flood risk which is already an increasing concern as a result of climate change (Lane *et al.*, 2007). Given the substantial legislative, ecological and economic implications of diffuse agricultural pollution, there is a growing interest in the cost-effective delivery of measures to mitigate its negative effects. For example, the £6.5 million government-funded Demonstration Test Catchments (DTC) project (2009–2014) focused on four agriculturally representative catchments in England with the aim of generating evidence on how diffuse pollution can be controlled to improve water quality at multiple spatial scales (DEFRA, 2015). Mitigation measures are wide-ranging and require robust data to be able to evaluate and compare their effectiveness to enable cost-effective implementation for future land management (Ockenden *et al.*, 2012).

One commonly adopted mitigation measure is the creation of pond features on or adjacent to watercourses to intercept pollutants such as sediment, nutrients and pesticides during their transport instream or along overland pathways. These types of features have varying designs and are often referred to in the literature under different terms such as constructed wetlands, retention ponds, in-line/on-line ponds, settling ponds, run-off attenuation features and rural sustainable drainage systems (RSuDS) (Barber and Quinn, 2012; Ockenden *et al.*, 2012; Duan *et al.*, 2016; Vinten *et al.*, 2017; Cooper *et al.*, 2019). Differences in design mean they can be suited to specific agricultural, rural, or urban contexts, but generally they aim to achieve the same purpose of improving water quality. These types of features have been implemented and studied across countries worldwide, notably in northern and western Europe. In France and England, pond systems have been used to treat motorway run-off, removing heavy metals such as cadmium by up to 100% in some cases (Lee *et al.*, 1997; Revitt *et al.*, 2004). Since as early as the 1960s, countries including Denmark and Germany have used constructed wetlands to treat domestic wastewater to reduce biochemical oxygen demand (BOD and also remove nitrogen (N) and phosphorus (P), typically by 30–50 % (Brix *et al.*, 2007; Vymazal, 2011a). Wetland features have been used in countries such as Norway, Finland, and Estonia to treat nonpoint (diffuse) sources of agricultural pollution by removing suspended solids and various nutrient species, primarily total P (Braskerud, 2002; Koskiaho *et al.*, 2003; Kasak *et al.*, 2018). A systematic review of created wetlands (mostly in North America and Europe) found that on average they significantly reduce P and N transport from wastewater, urban and agricultural run-off, with median removal rates of 1.2 and 93 g m^{-2} $year^{-1}$, respectively (Land *et al.*, 2016). It was concluded that further research is needed on the effects of hydrological pulses on wetlands, as it was found that wetlands with nutrient loading rates driven by rainfall had significantly lower P removal efficiencies than wetlands with controlled loading rates.

In addition to diffuse pollution mitigation, such features have the potential to provide co-benefits (ecosystem services) including carbon storage, flood risk reduction, low flow and drought resilience, habitat provision and aesthetic quality (Institute of Grassland and Environmental Research, 2002; Evans *et al.*, 2007; Thiere *et al.*, 2009; Greenway, 2010; Ockenden *et al.*, 2012; Newman *et al.*, 2015; Dadson *et al.*, 2017; Environment Agency, 2018). Evidence on the provision of some of these services has been reported with varying degrees of effectiveness for different benefits and for different designs, climates, soils, and catchment characteristics (Braskerud, 2002; Lee *et al.*, 2009; Ockenden *et al.*, 2012; Kill *et al.*, 2018). However, it is often still assumed that these pond features continuously deliver co-benefits without consideration into how their effectiveness may change over time as a consequence of the magnitude of storm events they experience and how they are managed or maintained.

Evidence on the site-specific constraints and limitations of interventions is important to develop guidance that enables targeted mitigation for achieving maximal societal and ecosystem benefit. Contrasting catchments have different water quality issues of focus, making evidence of intervention effectiveness for different pollutants useful for catchment management plans. For example, in hydrologically 'flashy' catchments dominated by slowly permeable clay soils, where surface run-off generation is high, the key concern is typically the resulting high loads of fine suspended sediment and particle-bound P entering watercourses (Sandström *et al.*, 2020). Despite their small size and discharge, agricultural drainage ditches and streams in headwaters often form highly connected dense networks that can cumulatively convey large quantities of water, sediment, and nutrients rapidly downstream (Alexander *et al.*, 2007). This makes agricultural headwater streams an appropriate target area for both pollution and flood risk mitigation opportunities. The study catchment discussed in this paper has been managed to exploit these mitigation opportunities in recent years, as part of both a Natural Flood Management (NFM) scheme and a scheme to tackle diffuse agricultural pollution and address current failure to meet WFD targets. The ponds in this study were implemented primarily to help mitigate diffuse P and fine sediment pollution to manage water quality, alongside providing co-benefits for habitat creation and biodiversity.

Past studies examining the trapping efficiencies of constructed wetlands have suggested that greater retention of sediment can occur with increases in the run-off received per wetland surface area (hydraulic load), and that even small-scale features can be suitable for retaining fine clay particles (Braskerud *et al.*, 2000; Braskerud, 2003). More recent efforts studying multiple events have raised questions surrounding a lack of consistency in trapping efficiency, with instances of net loss of material being reported (Barber and Quinn, 2012; Cooper *et al.*, 2019). More evidence on the controls on trapping efficiencies for different sediment and nutrient fractions under a broad range of hydrological conditions will help to clarify this issue. Previous guidance on the design of constructed wetlands recommended an optimal width-to-length ratio of 1:4; however, in practice this can be difficult to achieve on farms with limited space (Ellis *et al.*, 2003). Further evidence on the trapping efficiencies of different styles of pond and wetland (including those with suboptimal designs) can provide valuable information for developing sustainable designs for their future use in a variety of landscape contexts, including NFM schemes.

In the UK, ponds and wetlands are currently not widely used as interventions for mitigating the impacts of diffuse agricultural pollution, even more so in the case of on-line features that connect to existing streams. This paper aims to provide evidence on the effectiveness of small on-line ponds with ~1:1 aspect ratios to function as diffuse pollution interventions in the context of a lowland arable catchment in the UK. We quantify key water quality benefits derived from the pond system under both baseflows and stormflows, with a particular focus on its effectiveness to trap sediment and phosphorus over multiple hydrological events. It was hypothesised that the ponds would be most effective at trapping sediment, reducing both suspended sediment and phosphorus concentrations and loads downstream of each pond. This study also quantifies the accumulation of sediment within ponds since their construction in order to evaluate their sustainability and maintenance requirements in the long-term.

3.3 Materials and Methods

3.3.1 Study Site

The study site is located in the predominantly arable 16.3 km^2 Littlestock Brook sub-catchment that lies within the Evenlode catchment (430 km^2) in the upper reaches of the Thames basin in southern England, United Kingdom (Figure 3.1). Lithology within the Evenlode catchment is dominated by the Great Oolite Group, consisting of mudstone and fine-grained limestone (Figure 3.1b). The Littlestock Brook sub-catchment on the western side of the Evenlode is mostly underlain by the Lias Group, consisting of clays, mudstones, and limestones.

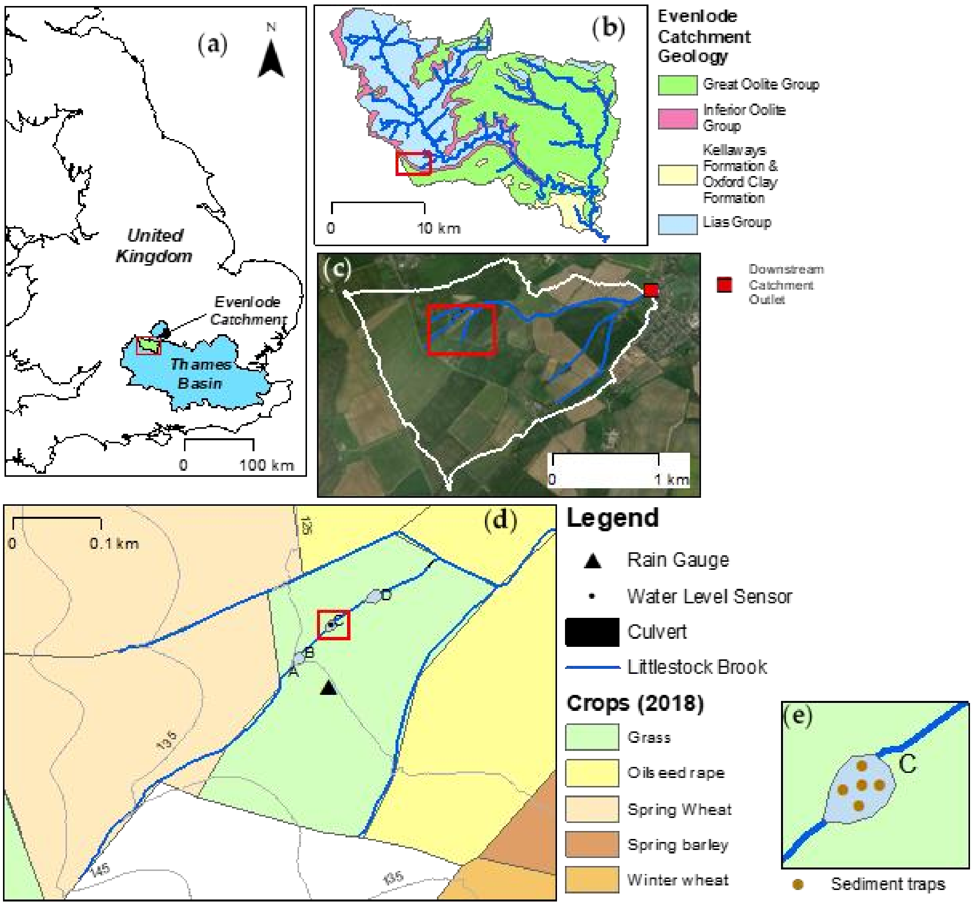


Figure 3.1 Locations of (**a**) the Evenlode catchment (green) within the Thames basin (blue); (**b**) the Downstream Catchment (outlined in red) within the Evenlode catchment and its geology; (**c**) the field site of the on-line ponds (outlined in red) within the Downstream Catchment; (**d**) the on-line ponds and monitoring sites (labelled A-D from upstream to downstream) and (**e**) sediment traps within the Central Pond. The white field contains improved grassland used as a horse paddock. Contours (grey) display elevation (metres AODN). Crop data are from UKCEH Land Cover® *plus*: Crops (2018).

The Littlestock Brook sub-catchment is being intensively monitored as part of a NFM pilot scheme that is being delivered over five years (2016–2021) to reduce flood risk in the village of Milton-under-Wychwood. A full description of the NFM project, its delivery, and interventions, is given by Old *et al.* (2019).

The ponds are situated in a first-order headwater tributary of the Littlestock Brook close to its source which rises from a limestone geology overlain by a shallow lime-rich soil that transitions down the valley into seasonally wet, slowly permeable clay soils. The average saturated hydraulic conductivity of topsoil in the catchment is approximately 50 cm day^{-1} (Panagos *et al.*, 2012). The area experiences a temperate maritime climate, with an average annual minimum temperature of 5.7 °C and maximum of 13.1 °C and receives an average annual rainfall of 765 mm (Met Office, 2021). The ponds drain an area of 0.3 km^2 and only occupy <0.2 % of this catchment. Over 75 % of this catchment area is underlain by a highly productive fissured aquifer. Both the rest of this area and the downstream catchment (3.4 km^2 area outlined in white; Figure 3.1c) are underlain by rocks with essentially no groundwater. This downstream catchment forms part of the wider high temporal resolution hydrological monitoring network for the NFM scheme, gauging flows and fluxes of waters/suspended matter leaving the NFM-impacted catchment outlet (detailed in Section 3.3.2). The ponds are situated in a steeper part of the catchment where run-off risk was identified as being high and overland flows had previously been observed.

The field containing the ponds was partly taken out of agricultural production for construction of the ponds and channel in February 2018 and was then surrounded by an area of mixed deciduous tree species planted in early 2019. The three ponds were dug out and the excavated soil used to form earth banks covering the outflows, which are comprised of layers of locally-sourced limestone that slowly allow flow through into the following stream reach. The ponds are teardrop-shaped and vary in size with the Upstream Pond having an estimated surface area and total capacity of 145 m^2 and 70 m^3, the Central Pond 126 m^2 and 90 m^3, and the Downstream Pond 156 m^2 and 95 m^3 respectively (Figure 3.2) (estimates derived from pond cross-section surveys described in Section 3.3.3). The Upstream/Downstream ponds have width-to-length ratios of ~1:1, and ~1:1.5 for the Central Pond. The earth banks and ponds were not seeded and left to colonise naturally.

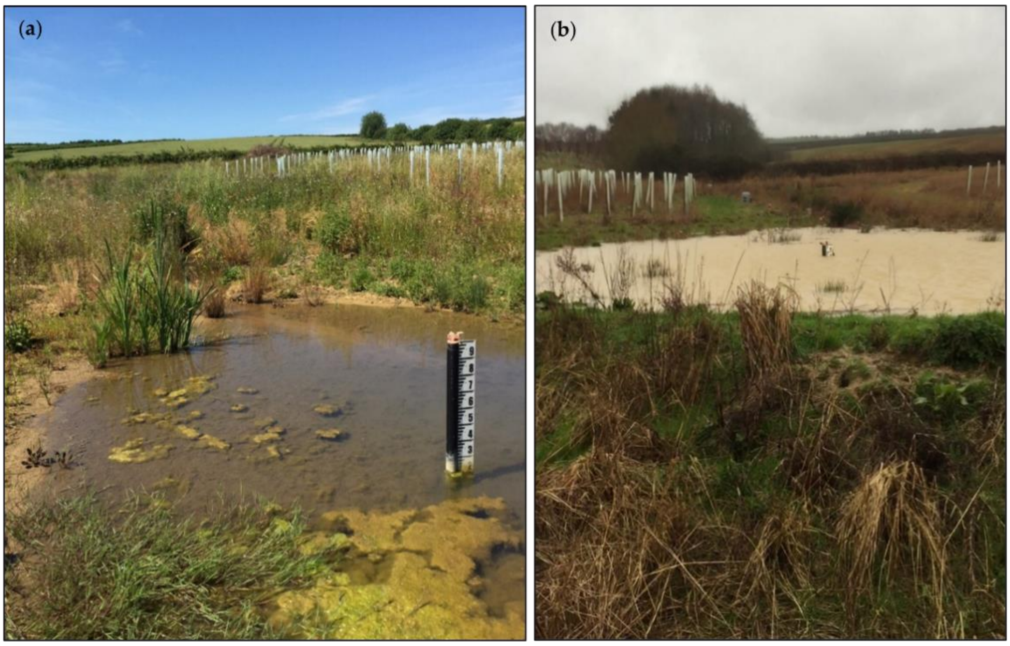


Figure 3.2 The Central Pond during (**a**) low flow conditions in July 2019; and (**b**) at capacity during a storm event on 15 February 2020. Both photos are taken facing upstream.

The newly dug stream channel is small and shallow with an average width of ~0.5 m and depth of ~0.15 m, and a gradient of 2.5 %. This contrasts to adjacent streams running along the field margins, which both have deeply incised channels. The channel planform is relatively straight, however colonisation and growth of graminoid vegetation within the channel has started to form multiple channels in places. In June 2018, a water level sensor (Rugged TROLL 100, In-Situ; Redditch, UK) was deployed in the Central Pond to measure changes in water depth and temperature at 5-min intervals. Atmospheric pressure from a nearby (<1 km) sensor was also logged and used to compensate for changes in barometric pressure. Sensor water depth was then calibrated against the observed depth on the stage board (pictured in Figure 3.2a) measured during site visits. The sensors were set to log at 5-min intervals in order to capture rapid changes in pond water level due to the 'flashy' nature of the catchment during rainfall events. In February 2019, a tipping bucket rain gauge (Casella; Sycamore, IL, USA) was installed in an adjacent clearing to measure rainfall at 2-min intervals. A storage rain gauge was also installed at the same location to aid quality control of the tipping bucket gauge. During site visits, stored rainwater was emptied into a graduated cylinder and the volume checked against the tipping bucket rainfall total for the same period to ensure measurements were within a 5 % tolerance range.

3.3.2 Water Quality Sampling and Analysis

To monitor water quality under near baseflow conditions, water samples from the on-line pond system's inlet and outlet were collected during field visits every 2–4 weeks. One unfiltered 60 mL sample was taken for total phosphorus (TP), and two 60 mL samples were immediately filtered through a 0.45 µm cellulose nitrate membrane (Whatman™ WCN grade; Maidstone, UK) for analysis of total dissolved phosphorus (TDP), soluble reactive phosphorus (SRP), and dissolved major ions (NO_2^-, NO_3^-, NH_4^+, F^-, Cl^- and SO_4^{2-}). Particulate phosphorus (PP) was taken to be the difference between TP and TDP. Approximately 500 mL was sampled using a US DH-48 sampler for determination of suspended sediment concentration (SSC) and volatile solids concentration (VSC) (as a proxy for organic matter). Water chemistry samples were refrigerated at 4 °C upon return from the field until they were analysed following Wallingford Nutrient Chemistry Laboratories procedures described in detail by Bowes *et al.* (2018). SSC was determined gravimetrically by filtering known volumes of water samples through pre-ashed, dried and weighed Whatman™ GF/C™ filter papers, which were then oven dried at 105 °C for at least 2 h. Filter papers were then reweighed after cooling in a desiccator for 30 min. VSC was then determined through loss-on-ignition (LOI) by igniting filter papers in a muffle furnace (AAF 1100, Carbolite Gero; Hope, Derbyshire, UK) at 500 °C for 30 min before being cooled and reweighed (Standing Committee of Analysts, 1984).

For monitoring storm events, automatic samplers (Sigma SD900, Hach; Loveland, CO, USA) were deployed at four locations along the stream to sample water flowing into and out of each pond (Figure 3.1c). Triggering of samplers was determined based on the rainfall forecast in order to capture samples approximately representative of the event. Grab samples of run-off were taken from contributing overland flow pathways. Samples were refrigerated upon return to the laboratory, and 60 mL subsamples were taken as soon as possible for chemical determinands of interest. To ensure representative subsampling, samples were thoroughly mixed before immediately taking an aliquot using a syringe. The remaining sample was used to determine SSC and VSC. Discharge was estimated at the ponds' outflows in higher flows using an Electromagnetic Current Meter (Valeport; Totnes, UK) and the velocity-area method (Herschy, 1993), and also under low flows using a conductivity sensor (EXO1, YSI; Yellow Springs, OH , USA) and the salt dilution method (Hongve, 1987). During storm events, run-off frequently overwhelmed the small stream channel and rendered it unsuitable for accurate flow measurement or development of a reliable stage-discharge relationship. Instead, water flowing through the ponds was estimated as a catchment area-weighted proportion of the discharge measured at a more stable gauging site (Downstream Catchment Outlet; Figure 3.1c). In order to represent timings of storm hydrographs more realistically, the estimated discharge was shifted back in time by applying a linear regression

(R^2 = 0.51) between peak discharge and the time difference between peak stage in the Central Pond and at the Downstream Catchment Outlet. It was assumed that at a given time, discharge was equal at both pond inflows and outflows.

The fluxes of total suspended sediment, silt and clay, and TP were also calculated at the Downstream Catchment Outlet site using discharge and SSC/TP data at 5-min intervals. Discharge was estimated using a stage-discharge rating curve with flow measurements taken using the methods described above, with measured discharges ranging from 6 to 587 L s^{-1} (n = 15). Turbidity was monitored using an in-stream sensor (DTS-12, FTS; Victoria, Canada) and then calibrated against SSC and TP samples (R^2 = 0.99, n = 95; R^2 = 0.79, n = 372) taken under a range of flows (sampled using the methods described above) to give estimated timeseries of SSC and TP. Turbidity data covered >99% of the monitoring period. Suspect datapoints were removed and the gaps filled by linear interpolation for periods of <12 h if no storm events took place during the missing period. Fluxes were calculated by integrating SSC/TP instantaneous load timeseries for the monitoring period. Suspended sediment particle size distributions were also sampled (n = 9) during two high flow/SSC events (measured using laser diffraction as described in Section 3.3.3). These event particle size distributions were assumed to be representative of the stream's suspended load as storm events contribute the majority of the total sediment flux. The proportions of particles <63 µm in diameter in the samples were averaged and combined to estimate the flux of silt and clay leaving the catchment.

3.3.3 Pond Sediment Sampling and Analysis

Sediment traps were deployed in each pond to quantify sediment, organic matter, and P accumulation, and determine particle size distribution. Traps were assembled from circular plastic saucers (19 cm in diameter, 4 cm in height) with weights attached to allow them to sink and rest on the pond bed. Traps were positioned in ponds as evenly as possible, with one central trap and four outer traps (e.g., Figure 3.1). Traps were deployed for periods of up to 50 days before being retrieved, emptied, and immediately redeployed. Collected sediment (including pond water pooled on the surface) from each trap was emptied into separate plastic bottles for transport back to the laboratory. Bottles were then emptied into larger plastic boxes and refrigerated for at least 48 h to allow suspended solids to settle out. The supernatant was then siphoned off into bottles and filtered following the same method described for SSC to account for the mass of any fine particles still in suspension. Macroinvertebrates found in trap samples were removed and identified to family level where possible. Sediment in the boxes was stirred thoroughly, and for each, three sub-samples of ~5 g were transferred into centrifuge tubes for particle size analysis. Grain size distributions and characteristics were determined using laser diffraction particle size analysis (Mastersizer 2000, Malvern Panalytical; Malvern, UK). Prior to analysis, samples were treated with a 5 % sodium hexametaphosphate solution to disperse particles and agitated for 5 min in an ultrasonic bath. To determine sediment mass, the remaining sediment was distributed into pre-weighed aluminium trays (~100 g sediment per tray) and oven-dried at 105 °C for at least 48 h before being cooled and reweighed. To determine volatile solids (organic matter) by LOI, one tray per trap was then ignited at 500 °C for 2 h before being cooled and reweighed. One tray per batch was reheated and reweighed to check that the sample mass remained stable. P content was determined by grinding the ignited sample into a fine powder, of which triplicate subsamples of 3 ± 0.1 mg were taken, mixed with 60 mL ultrapure water, and then analysed using the same TP methodology used for water samples. Length and width transects of pond sediment depths were surveyed in January and July 2020 following a standard method (Puttock *et al.*, 2018), and spatially interpolated in a GIS (ArcMap, Esri; Redlands, CA, USA) using the natural neighbour interpolation method to estimate stored sediment volumes. Measuring along transects aimed to minimise sediment disturbance and damage to habitat but meant that measurements were not evenly distributed across the pond area. Natural neighbour interpolation was, therefore, chosen over other methods because of its ability to perform well with an uneven sampling density and irregular distribution of data points (Ledoux and Gold, 2005).

3.3.4 Data and Statistical Analyses

Statistical procedures were carried out in RStudio v1.1.453 (RStudio Team, Boston, MA, USA, 2016) using the programming language R (R Core Team, 2018). Inlet and outlet water quality determinand concentration data were assessed for normality with the Shapiro-Wilk test, after which any non-normal variables were normalised using cube-root transformations. Paired samples T-tests were carried out on inlet-outlet samples for determinands to compare their means over the sampling period (Lee *et al.*, 2019). Similarly, sediment particle size distribution variables were tested for normality and equal variances to ensure robustness before performing one-way ANOVAs and post hoc Tukey's tests on the data (Rice and Church, 1998). Baseflow removal efficiencies of determinands from ponds were calculated using Equation 3.1:

$$Removal\ Efficiency\ (\%) = \frac{Inflow\ Concentration\ -\ Outflow\ Concentration}{Inflow\ Concentration} \times 100 \qquad (3.1)$$

Similarly, removal efficiencies for storm events were determined using total loads at monitoring locations calculated using the estimated event discharge. Antecedent Precipitation Index (API) was calculated for storm events with daily rainfall records since data collection began using Equation 3.2:

$$API_d = k \times API_{d-1} + P_d \tag{3.2}$$

where API_d is the API for day, d; k is a decay factor and P_d is rainfall for day d. A fixed value of 0.95 was used for k following the method described by (Hill *et al.*, 2015). Simple linear regressions were carried out to test the effect and strength of water temperature on determinand removal efficiencies where previous research had suggested the relationships exist (Kim *et al.*, 2011; Mallin *et al.*, 2012).

3.4 Results

3.4.1 Near Baseflow Water Quality

Outside of rainfall events, 19 sets of samples were taken between March 2019 and March 2020. Significant differences between inlet and outlet concentrations were found for dissolved nitrate, SRP, SSC and VSC, which all showed a decrease in mean concentration at the outlet (paired samples t-test, $p < 0.01$, $n = 19$) (Figure 3.3).

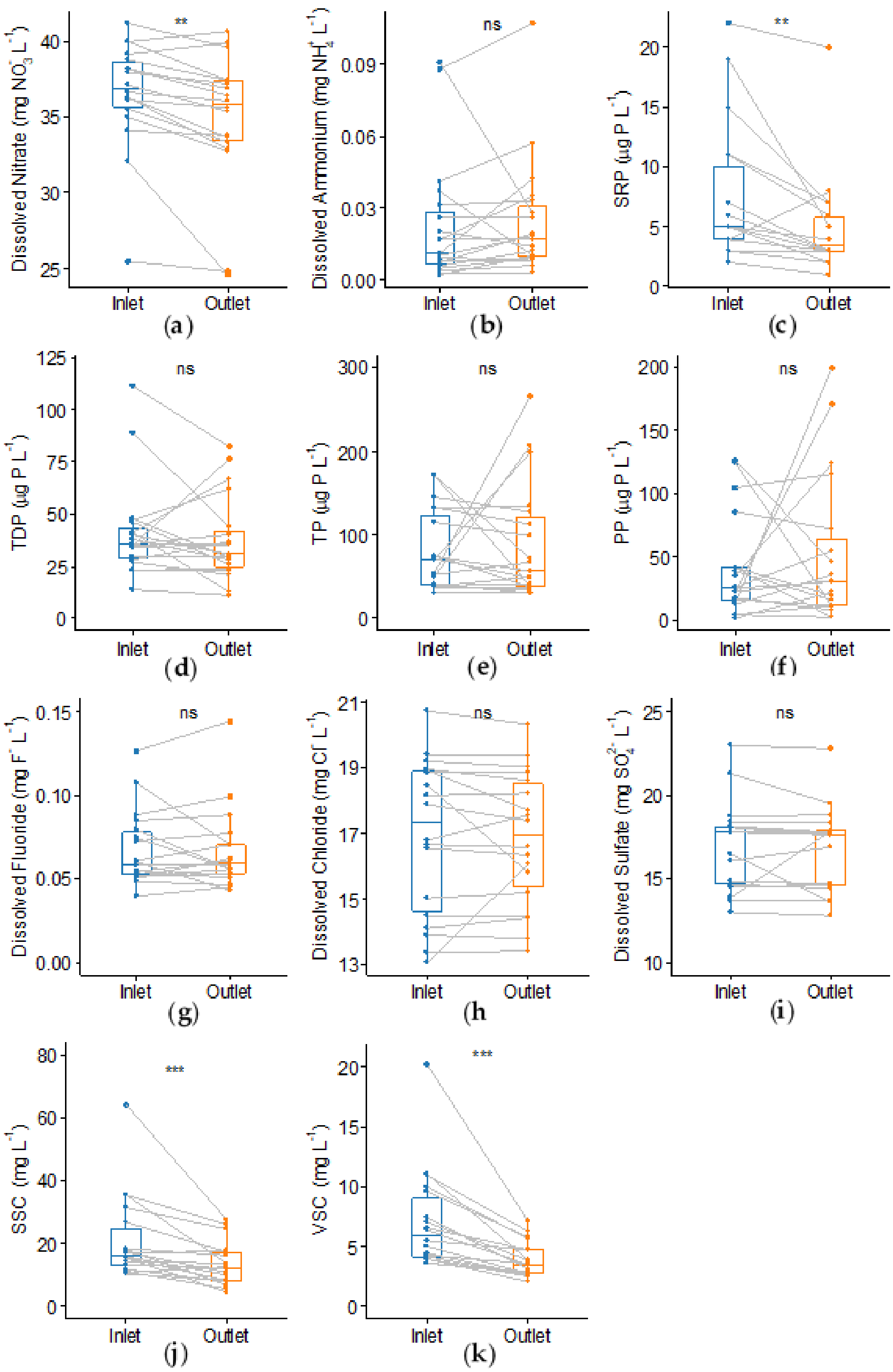


Figure 3.3 Boxplots showing paired on-line pond inlet and outlet concentrations for **(a)** dissolved nitrate; **(b)** dissolved ammonium; **(c)** SRP; **(d)** TDP; **(e)** TP; **(f)** PP; **(g)** dissolved fluoride; **(h)** dissolved chloride; **(i)** dissolved sulfate; **(j)** SSC and **(k)** VSC. Median values are represented by bold lines. Significance levels for results of paired samples T-tests are indicated with: *** ($p < 0.001$), ** ($p < 0.01$), ns ($p > 0.05$).

The other determinands generally showed minimal variance between the inlet and outlet; however, in some cases TP/PP concentrations increased by over 100 % at the outlet. Nitrite (NO_2^-)

was excluded from the statistical tests due to a majority (67 %) of both inlet and outlet samples measuring 0 mg NO_2^- L^{-1}.

Removal efficiencies exhibited considerable variability between determinands during baseflows, ranging from extreme negative values (net export from the pond system) for PP, to more consistently positive values (net retention) for SSC and VSC (Table 3.1). Overall, the majority of mean removal efficiencies for the sampling period were positive, with the exceptions being PP, TP, and NH_4^+.

Table 3.1 Mean (±SD) inflow and outflow concentrations (mg L^{-1}), and mean (±SD), minimum, and maximum removal efficiency (%) of the on-line pond system for water quality determinands sampled during (near) baseflow conditions.

Determinand	Mean Inflow Concentration (mg L^{-1})	Mean Outflow Concentration (mg L^{-1})	Mean Removal Efficiency (%)	Minimum Removal Efficiency (%)	Maximum Removal Efficiency (%)
SRP	0.008 ± 0.006	0.005 ± 0.004	29 ± 37	−100	74
TDP	0.041 ± 0.023	0.038 ± 0.02	3 ± 43	−117	68
PP	0.04 ± 0.04	0.052 ± 0.059	−237 ± 579	−2100	95
TP	0.081 ± 0.048	0.089 ± 0.069	−34 ± 125	−314	77
NH_4^+	0.023 ± 0.026	0.024 ± 0.025	−61 ± 118	−400	73
NO_3^-	36.56 ± 3.585	34.903 ± 4.4	5 ± 6	−2	23
F^-	0.068 ± 0.023	0.067 ± 0.024	0 ± 18	−23	35
Cl^-	16.913 ± 2.382	16.835 ± 2.045	0 ± 7	−23	14
SO_4^{2-}	17.006 ± 2.652	16.904 ± 2.488	0 ± 9	−29	18
SSC	21.2 ± 4.153	13.464 ± 6.943	32 ± 24	−17	70
VSC	7.09 ± 4.153	3.901 ± 1.469	40 ± 15	15	66

3.4.2 Storm Event Water Quality

Four storm events were captured between March 2019 and February 2020 (Table 3.2); however, it was not always possible to trigger all four automatic samplers for every storm. The event captured in February was during Storm Dennis and had the highest rainfall; total monthly rainfall in February was 170 % above average for the area. Estimated peak discharge was highest during the November event, with a return period of 5.5 years (Bishop *et al.*, 2021). API was highest prior to the October 14th event following a rapid wetting of the catchment at the end of September.

Table 3.2 Mean (±SD) SSC (mg L^{-1}) for each pond monitoring site during four storm events, estimated discharge (L s^{-1}) prior to the event and at its peak, and the sampling duration (hours). Rainfall (mm) is the total event precipitation and Antecedent Precipitation Index (API) (mm) is given for the day prior to each event.

Storm Event	Mean SSC (mg L^{-1})				Sampling Duration (h)	Estimated Discharge (L s^{-1})		Rainfall (mm)	API (mm)
	Upstream Pond Inlet	Upstream Pond Outlet	Central Pond Outlet	Downstream Pond Outlet		Pre-event	Peak		
12/13 March 2019	45 ± 47	30 ± 33	29 ± 27	35 ± 30	23	8.9	18.7	8.8	51.9
14 October 2019	258 ± 365	161 ± 152	143 ± 94	126 ± 55	5.75	8.4	58.6	23.1	104.1
14 November 2019	92 ± 67	27 ± 11	24 ± 7	-	5.75	9.2	74	31.8	97.6
15/16 February 2020	-	87 ± 63	98 ± 79	-	23	12.3	55.7	32.2	64.8

The March 2019 event was the smallest in magnitude, with the least rainfall and lowest API, but still resulted in a peak SSC of >200 mg L^{-1} at the inlet to the Upstream Pond, with the peak then being reduced by ~50 % downstream at the outlet of the Downstream Pond (Figure 3.4d). Streamflow responded rapidly to rainfall with a lag time of less than two hours (Figure 3.4a-b). The response of suspended sediment was partially staggered, with lag times increasing downstream at each monitoring point except for water leaving the Downstream Pond, which peaked simultaneously with water leaving the Central Pond. SSC at the Downstream Pond outlet had a less steep gradient on the falling limb compared to the other monitoring locations. Volatile solids made up <20 % of the total solids during peaks, but as high as 78 % on the receding limb (Figure 3.4c).

The response of TP and PP closely reflected that of SSC and VSC; however, TDP did not exhibit a rising limb and remained relatively constant at the inlet and outlet of the Upstream Pond (Figure 3.4e-g). TDP showed a somewhat different pattern at the outlet of the Central Pond with the concentration abruptly dropping below 10 µg P L^{-1} after 19:00 p.m. At the Downstream Pond outlet, TDP remained under 20 µg P L^{-1}, which was lower than both the inlet and outlet of the Upstream Pond which almost always stayed above 20 µg P L^{-1}. On the rising and receding limbs of the event, PP accounted for the majority (57–91 %) of transported P, after which TDP at the inlet and outlet of the Upstream Pond exceeded the particulate fraction. Automatic sampler SRP data are not presented as samples could not be analysed within 48 h of sampling and showed a 60–100

% decrease when compared to grab samples analysed within 48 h. Grab samples showed that SRP made up to 42 % of the TDP at 12:00 p.m.

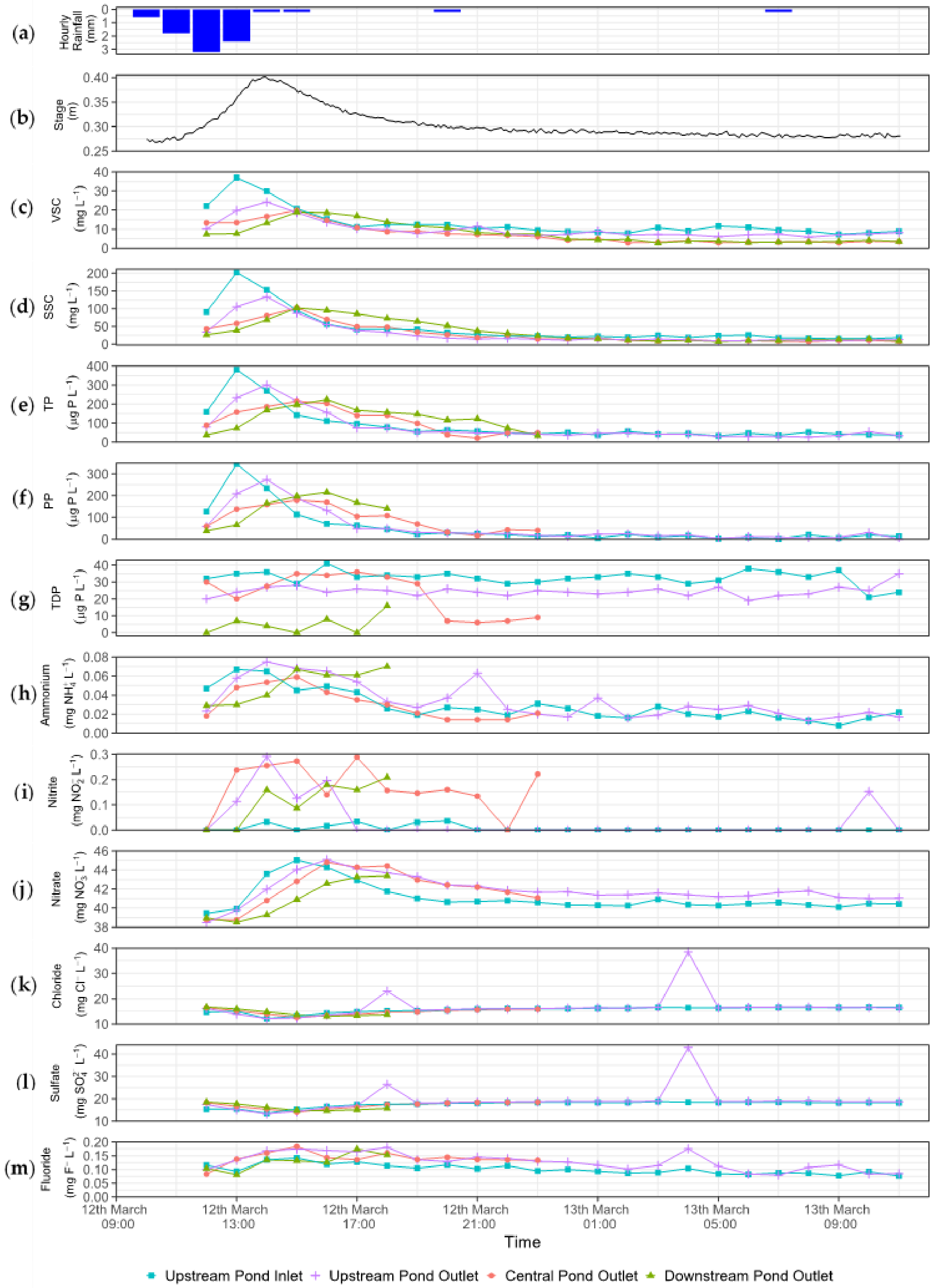


Figure 3.4 Timeseries during a storm event on 12/13 March 2019 showing: (**a**) hourly rainfall (mm); (**b**) stage (m) in the Central Pond; and concentrations of water quality determinands: (**c**) VSC and (**d**) SSC (mg L^{-1}); (**e**) TP, (**f**) PP, and (**g**) TDP (µg P L^{-1}); (**h**) ammonium (mg NH_4^+ L^{-1}); (**i**) Nitrite (mg NO_2^- L^{-1}); (**j**) Nitrate (mg NO_3^- L^{-1}); (**k**) chloride (mg Cl^- L^{-1}); (**l**) Sulfate (mg SO_4^{2-} L^{-1}) and (**m**) Fluoride (mg F^- L^{-1}) at each pond inlet/outlet sampling site.

Both dissolved ammonium and nitrate concentrations increased during the March event, with nitrate having more defined peaks and ammonium having a more variable response (Figure 3.4h and Figure 3.4j). Dissolved nitrite displayed rising limbs at the pond outlets but remained comparatively low at the Upstream Pond inlet (Figure 3.4i). Throughout the event, the majority of dissolved N transported was made up by nitrate. Concentrations of nitrate after the peak remained consistently higher leaving the Upstream Pond than those at the inlet. Dissolved fluoride showed a rising limb during the storm event after which the concentration decreased gradually and returned to a similar level as at the start of the event (Figure 3.4m). Dissolved chloride and sulfate concentrations exhibited almost identical patterns, with both solutes showing a small dilution between 14:00 p.m. and 15:00 p.m. coinciding with the peak water level (Figure 3.4k-l). There was minimal variation in chloride and sulfate concentration between sampling sites with the exception of two sudden peaks at the outlet of the Upstream Pond.

During the sampled storm events, total suspended sediment loads entering the pond system varied from 55 to 220 kg, and between 0.08 and 0.44 kg for TP, reflecting both the event magnitude and duration (Table 3.3). Load removal efficiencies varied greatly between ponds and events; however, the Upstream Pond was consistently the most efficient in all events sampled for both suspended sediment and TP. Generally, load removal efficiencies were higher for suspended sediment than TP, with negative removal efficiencies occurring more frequently for TP. During the March event, the Downstream Pond showed the lowest (negative) removal efficiency for suspended sediment out of all sampled events as a result of elevated concentrations at its outlet during the falling limb. The highest removal efficiency was observed in the November event for the Upstream Pond which indicated a net retention of 50 kg suspended sediment during <6 h. Sediment load removal efficiency of the Upstream Pond in the October event was 40 % lower, but still retained 66 kg also across a <6-h period. Overall sediment retention in the March event was comparatively much smaller at only 9 kg over a longer 23-h period. In the context of the wider catchment area, the net sediment load retained by the ponds in the March event was equivalent to 0.85 % of the flux leaving the Downstream Catchment during the same time period. This proportion was almost 2.5 times greater during the October event, with 2.1 % of the Downstream Catchment flux retained by the ponds.

Table 3.3 Load (kg) of suspended sediment, TP at each monitoring location and removal efficiency (%) of each pond for the sampled storm events. Ups. = Upstream Pond; Cent. = Central Pond; Down. = Downstream Pond; Catch. = gauged catchment. (NB Loads for TP in the March 2019 event are calculated for the first half of the event period).

Storm Event	Load (kg)										Load Removal Efficiency of Pond (%)					
	Suspended Sediment					TP					Suspended Sediment			TP		
	Ups. Inlet	Ups. Outlet	Cent. Outlet	Down. Outlet	Catch. Outlet	Ups. Inlet	Ups. Outlet	Cent. Outlet	Down. Outlet	Catch. Outlet	Ups.	Cent.	Down.	Ups.	Cent.	Down.
March 2019	54.73	38.9	37.81	46.21	1005	0.08	0.078	0.078	0.087	1.81	28.91	2.81	−22.21	2.53	−0.31	−11.8
October 2019	219.96	154.05	141.80	126.94	4438	0.437	0.347	0.391	0.357	8.81	29.96	7.95	10.48	20.53	−12.51	8.76
November 2019	70.55	20.53	17.81	-	1382	-	-	-	-	2.98	70.9	13.24	-	-	-	-
February 2020	-	213.5	250.79	-	10566	-	0.588	0.679	-	21	-	−17.47	-	-	−15.51	-

3.4.3 Pond Sediment Quality

From the manual surveying of sediment depths approximately two years after their construction, it was estimated that 13.89 m^3 of matter had accumulated in the Upstream Pond and 7.36 m^3 in the Central Pond. This meant that the Upstream Pond had filled ~20 % of its total capacity, and the Central ~8 %. At the time of surveying in January, depths in the Downstream Pond were unable to be measured due to the water level being too high. The Downstream Pond was able to be surveyed in July at the earliest (due to the Covid-19 pandemic), and had accumulated 9.89 m^3 of matter, equating to ~10 % of its total capacity.

Sediment traps were first deployed in March 2019, after which traps were deployed continuously from August 2019 with sediment collection taking place on six occasions until March 2020 to capture run-off during the wet season. Throughout this seven-month period, rates of accumulation were variable, but the Upstream Pond had the highest overall accumulation, and the Downstream Pond had the lowest (Table 3.4). Sediment accumulation rates varied considerably between the trap placements within ponds as shown by the large standard deviations. Despite only a short deployment period, the accumulations were surprisingly high during August, with ponds accumulating disproportionately more sediment (0.048 t ha^{-1}) than the yield leaving the Downstream Catchment (0.001 t ha^{-1}). Over the whole period, the ponds accumulated 6.1 % of the downstream catchment silt + clay flux, and 7.6 % of all suspended sediment. P accumulation in ponds generally showed the same pattern as sediment, and on average made up ~0.1 % of the total accumulated mass (Table 3.5). Total accumulated P in ponds only made up 3.2 % of the Downstream Catchment P flux. LOI showed that deposited sediments were largely made up of inorganic matter (IOM), accounting for >75 % of the accumulated sediment mass throughout the sampling period. The organic matter (OM) content ranged from 10–23 % and consistently decreased downstream along the pond sequence in each deployment period. OM content was highest between August and October. OM content of pond sediment was significantly enriched compared to the soil in the arable fields of the contributing area, which had an OM content of 5–7 %, typical of arable fields in this area.

Table 3.4 Accumulated sediment (±SD) (t) in each pond, all three ponds, and only the silt + clay (<63 µm) for sediment trap monitoring periods. Accumulated sediment yield (t ha^{-1}) for all ponds from the contributing area (30 ha), the flux of sediment and silt + clay (t) and the exported yield (t ha^{-1}) from the downstream catchment area (340 ha) are given for the same periods.

Monitoring Period	Days	Rainfall (mm)	Accumulated Sediment (t)					All Ponds Sediment Yield (t ha^{-1})	Catchment Sediment Flux (t)	Catchment silt+clay Flux (t)	Catchment Sediment Yield (t ha^{-1})
			Upstream Pond	Central Pond	Downstream Pond	All Ponds	All Ponds (silt+clay)				
8 August 2019 – 30 August 2019	22	62	0.56 ± 0.27	0.54 ± 0.35	0.33 ± 0.35	1.43 ± 0.56	1.01	0.048	0.34	0.3	0.001
30 August 2019 – 3 October 2019	34	128	0.63 ± 0.55	0.17 ± 0.05	0.25 ± 0.04	1.06 ± 0.55	0.71	0.035	7.4	6.47	0.022
3 October 2019 – 30 October 2019	27	132	0.69 ± 0.27	0.32 ± 0.11	-	1.01 ± 0.29	0.65	0.034	19.06	16.66	0.056
30 October 2019 – 4 December 2019	35	140	0.63 ± 0.37	0.39 ± 0.2	-	1.02 ± 0.42	0.67	0.034	21.93	19.16	0.065
4 December 2019 – 22 January 2020	49	167	0.67 ± 0.27	0.82 ± 0.28	0.67 ± 0.23	2.16 ± 0.45	1.57	0.072	32.79	28.65	0.096
22 January 2020 – 12 March 2020	50	177	0.98 ± 0.35	1.05 ± 0.33	0.49 ± 0.31	2.52 ± 0.57	1.77	0.084	38.63	33.76	0.114
Total	217	871	4.15 ± 0.89	3.29 ± 0.6	1.74 ± 0.52	9.18 ± 1.19	6.38	0.306	120.18	104.99	0.353

Table 3.5 Accumulated phosphorus (±SD) (kg) in each pond and all three ponds for sediment trap monitoring periods. Accumulated P yield (kg ha^{-1}) for all ponds from the contributing area (30 ha), the flux of P (kg) and the exported P yield (kg ha^{-1}) from the downstream catchment area (340 ha) are given for the same periods.

Monitoring Period	**Days**	**Rainfall (mm)**	**Accumulated P (kg)**				**All Ponds P Yield (kg ha^{-1})**	**Catchment P Flux (kg)**	**Catchment P Yield (kg ha^{-1})**
			Upstream Pond	**Central Pond**	**Downstream Pond**	**All Ponds**			
8 August 2019 – 30 August 2019	22	62	0.58 ± 0.27	0.51 ± 0.34	0.29 ± 0.28	1.38 ± 0.52	0.046	1.06	0.003
30 August 2019 – 3 October 2019	34	128	0.69 ± 0.55	0.22 ± 0.08	0.27 ± 0.04	1.18 ± 0.56	0.039	16.42	0.048
3 October 2019 – 30 October 2019	27	132	0.65 ± 0.18	0.36 ± 0.12	-	1.01 ± 0.22	0.034	43.87	0.129
30 October 2019 – 4 December 2019	35	140	0.56 ± 0.29	0.4 ± 0.19	-	0.96 ± 0.35	0.032	54.7	0.161
4 December 2019 – 22 January 2020	49	167	0.6 ± 0.22	0.81 ± 0.27	0.69 ± 0.25	2.1 ± 0.43	0.07	77.64	0.228
22 January 2020 – 12 March 2020	50	177	0.91 ± 0.22	0.94 ± 0.48	0.42 ± 0.27	2.27 ± 0.59	0.076	87.91	0.259
Total	217	871	3.99 ± 0.77	3.24 ± 0.69	1.68 ± 0.46	8.91 ± 1.13	0.297	281.6	0.828

Total P content of sediment from traps varied from 695 to 1634 mg kg^{-1} (Figure 3.5). The highest median P content in each pond occurred during September and then showed a downward trend in the following months. During most deployment periods, P content decreased along the pond sequence.

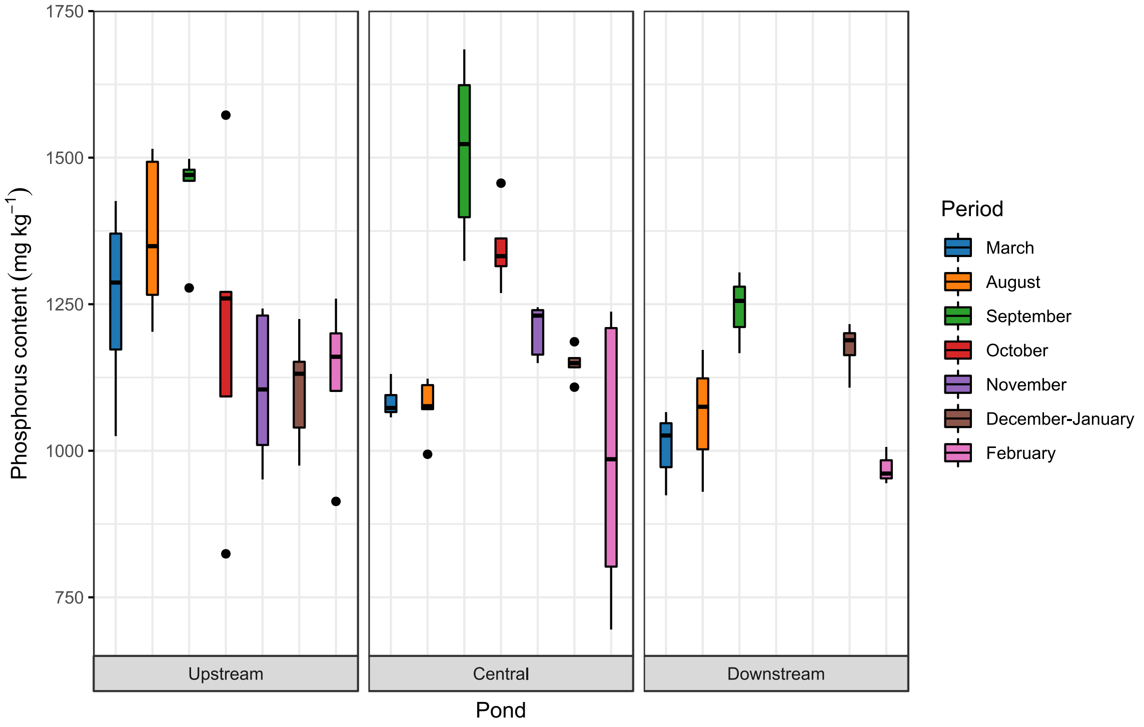


Figure 3.5 Boxplots showing the range of phosphorus content (mg kg^{-1}) of deposited sediment in each pond for each trap deployment period. Median values are represented by bold lines.

The P content of pond sediment was found to be positively correlated with OM content ($p < 0.05$) (Figure 3.6). This relationship was strongest in the Central Pond, with OM explaining 54 % of variation in P content.

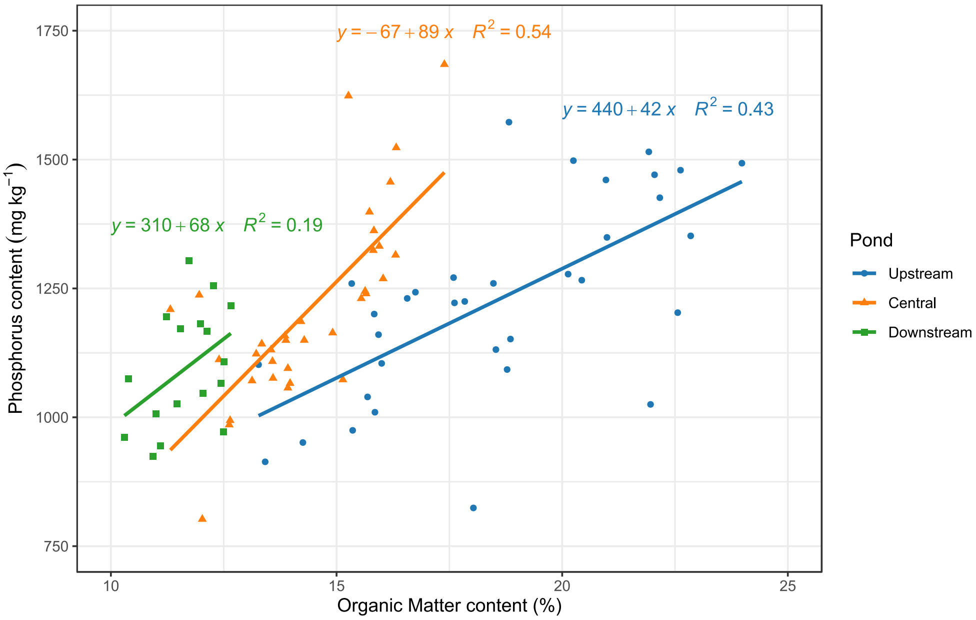


Figure 3.6 Linear regressions of organic matter content (%) and P content (mg kg^{-1}) of deposited sediment in each pond (Upstream Pond n = 34; Central Pond n = 35; Downstream Pond n = 20).

Deposited sediment in all three ponds was mainly comprised of the silt fraction, followed by sand, and then clay, which only accounted for up to 6 % of particles (Figure 3.7). Both clay and silt content showed an increasing trend downstream along the pond sequence, whilst sand showed a decrease. All pairwise comparisons show significant differences between group means with the exception of clay content between the Central and Downstream Ponds. Soil in the contributing area is known to have a clay content of 10–25 %, silt content of 50–60 %, and sand content of 15–30 %, broadly mirroring the composition of the deposited pond sediments.

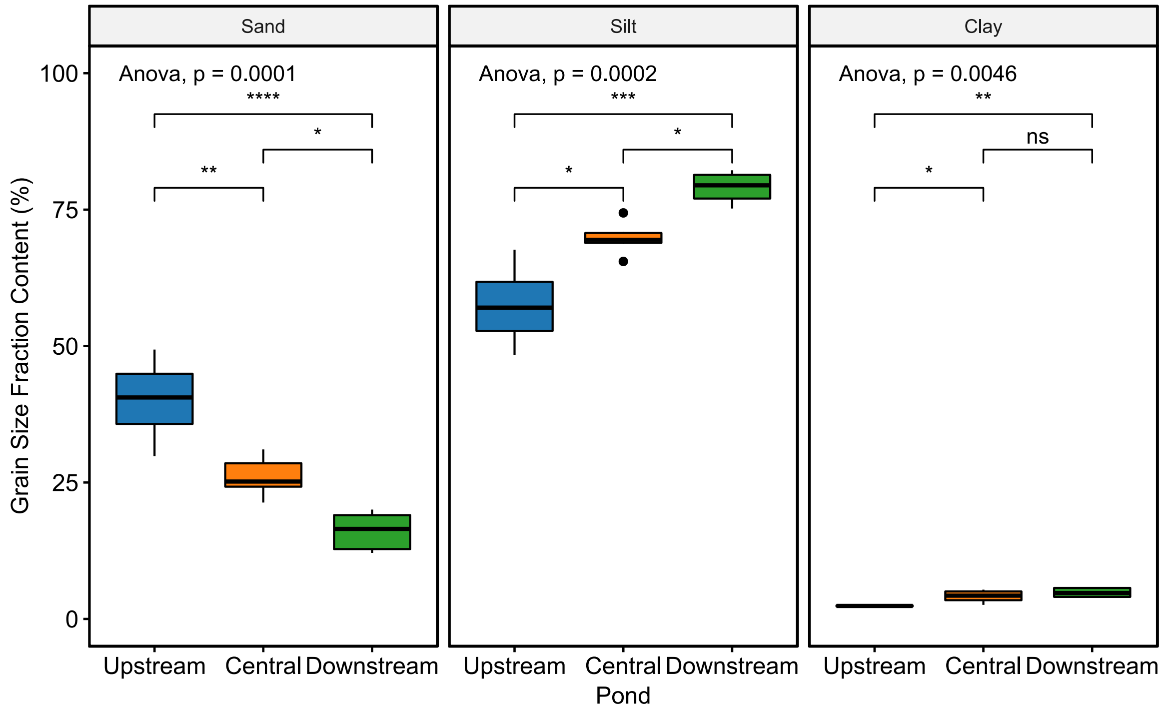


Figure 3.7 Boxplots showing the range of sediment grain size content (%) of sand, silt, and clay in each pond from trap deployment during March 2019 (Upstream Pond *n* = 4; Central Pond *n* = 5; Downstream Pond *n* = 5). Median values are represented by bold lines and significance levels for Tukey's tests by **** (p < 0.0001), *** (p < 0.001), ** (p < 0.01), * (p < 0.05), ns (p > 0.05).

Median particle diameter (D_{50}) of deposited sediment in traps was shown to decrease downstream along the pond sequence, ranging from a maximum of 61.85 µm in the Upstream Pond to a minimum of 15.8 µm in the Downstream Pond (Figure 3.8). Inversely, specific surface area (SSA) increased along the pond sequence, ranging from 0.42 $m^2 g^{-1}$ in the Upstream Pond to 0.89 $m^2 g^{-1}$ in the Downstream Pond. Pairwise comparisons showed significant differences ($p < 0.05$) in both D_{50} and SSA between the Upstream and Central Pond, and the Upstream and Downstream Pond.

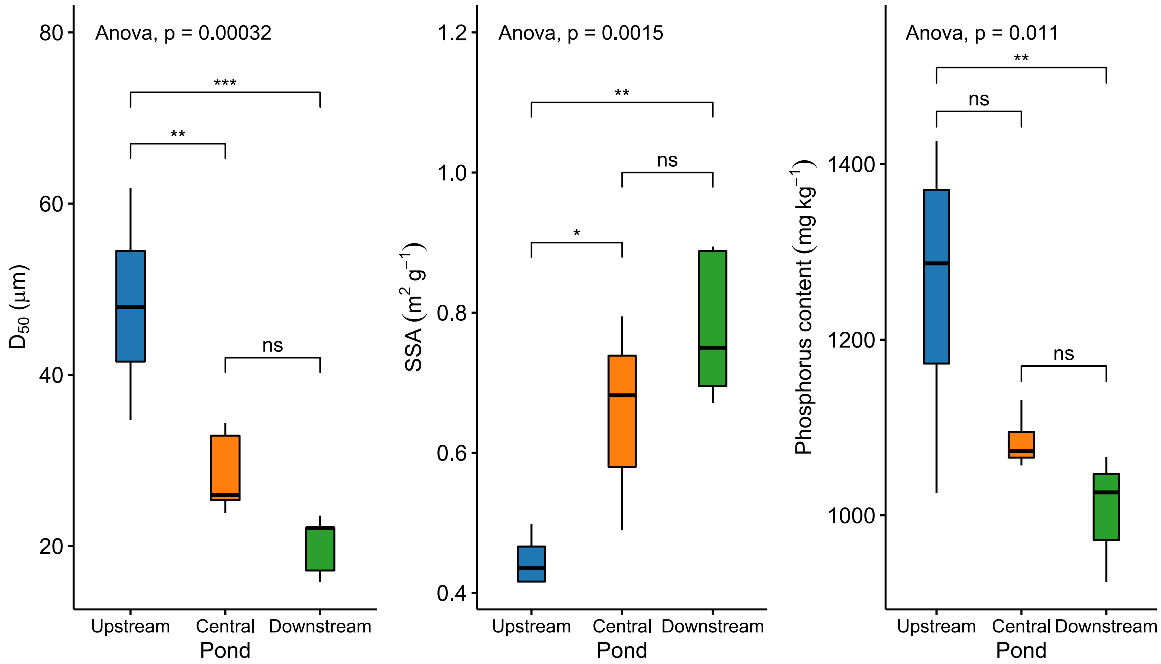


Figure 3.8 Boxplots showing the range of D_{50} (µm), SSA (m^2 g^{-1}), and phosphorus content (mg kg^{-1}), of sediment in each pond from trap deployment during March 2019 (Upstream Pond *n* = 4; Central Pond *n* = 5; Downstream Pond *n* = 5). Median values are represented by bold lines and significance levels for Tukey's tests by *** ($p < 0.001$), ** ($p < 0.01$), * ($p < 0.05$), ns ($p > 0.05$).

The P content of accumulated sediment within ponds showed much less variation compared to the P content of suspended sediment sampled during the March 2019 storm event (Figure 3.9). P content of suspended samples varied from 0 to ~2500 mg kg^{-1}. The suspended samples showed a general increasing trend in P content from upstream to downstream along the pond sequence; however, this enrichment effect appeared to level off between the Central Pond Outlet and Downstream Pond Outlet.

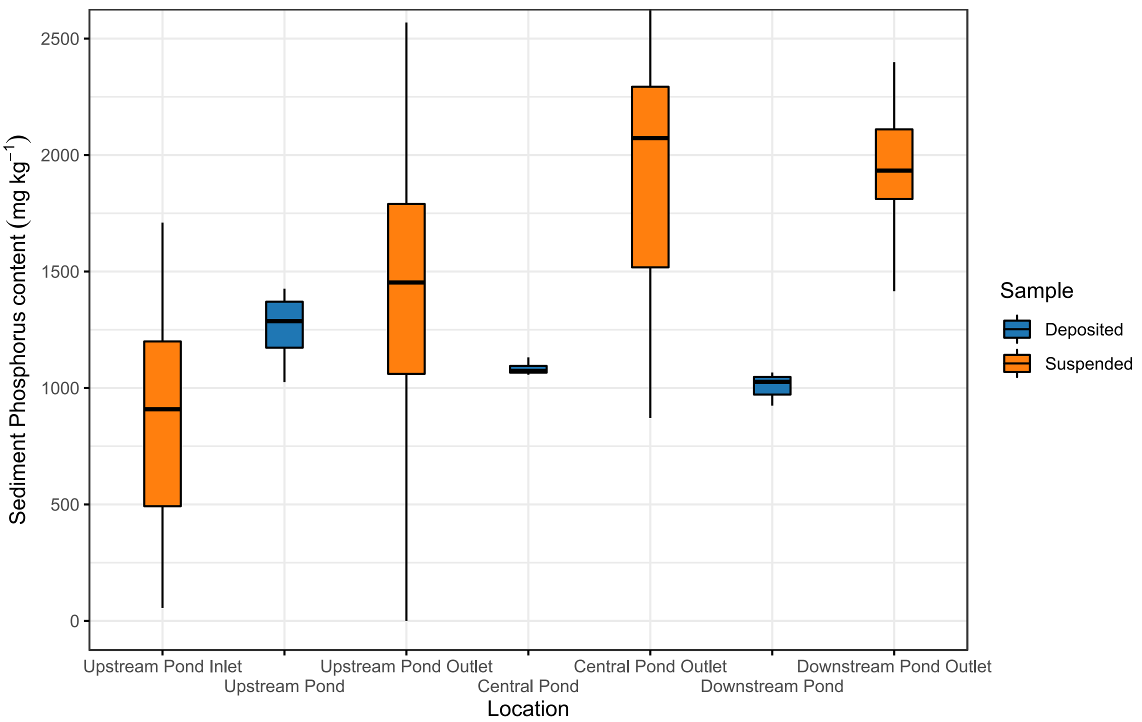


Figure 3.9 Boxplots showing the range of sediment phosphorus content (mg kg^{-1}) at different locations along the on-line pond system. Deposited samples were from sediment traps deployed during March 2019, and suspended samples from water sampled during the storm event in the same month. Median values are represented by bold lines.

3.5 Discussion

3.5.1 Near Baseflow Water Quality

The on-line ponds were shown to be effective at removing dissolved nitrate and SRP, which are both bioavailable forms of N and P. The removal efficiency of nitrate showed seasonality, peaking during the summer at 23 % but was below 10 % the rest of the time, and only exhibited negative removal efficiencies on two occasions (in December and January). This is considerably lower than the average nitrate removal efficiencies of between 72 % and 83 % reported by previous studies (Braskerud, 2002; Kim *et al.*, 2011; Mallin *et al.*, 2012). SRP removal efficiency of the on-line pond system had a wider range of up to 74 %, though it is important to note that this represents a reduction of only ~15 µg P L^{-1} due to low concentrations of SRP in the baseflow. In contrast to nitrate, SRP removal efficiency did not show any apparent seasonality, though the lowest removal efficiency of -100 % was also observed in winter. Evidence from a created wetland in Ohio, also draining an agricultural catchment, found no significant difference in removal between seasons; however, the influence of season on SRP removal has been shown to be important in certain wetlands, which exhibit removal increases during the warm seasons (Fink and Mitsch, 2004; Kasak *et al.*, 2018). Other studies have found average SRP (or orthophosphate) removal efficiencies of between 12 % and 87 % (Wedding, 2000; Braskerud, 2002; Mitsch *et al.*, 2014). The removal efficiency of bioavailable nutrient fractions is likely limited within the on-line ponds by several factors, a key one being the abundance and density of macrophytes and algae. During the first growing season (spring/summer 2018), limited establishment of vegetation was observed in the ponds. This could partly be due to extreme temperatures (2018 being the hottest summer on record in England at the time of writing), but also potentially a result of not enough time for natural colonisation to occur. In contrast, by the end of summer 2019 the ponds had been partly colonised, particularly the Central Pond with several stands of *Typha latifolia* and *Juncus* spp. In both years, all three ponds showed substantial algal growth, often forming thick mats of filamentous green algae that covered up to a quarter of pond surfaces. As data collection only began in 2019, the effect of the presence of vegetation on nutrient removal could not be analysed. However, it is thought that, over time, the ponds are likely to increase their nutrient removal capacity given further macrophyte succession. This is supported by a review of constructed wetlands that found that SRP and TP removal was higher in older (>18 months) wetlands (Newman *et al.*, 2015). SRP removal is also influenced by the underlying pond bed sediment and its P sorption/desorption capacity, typically quantified as the Equilibrium Phosphorus Concentration (EPC_0) (Jarvie *et al.*, 2005). However, in constructed wetlands with considerable algal growth, EPC_0 has been shown to play a less important role in P removal compared to algal uptake (Yoo *et al.*, 2006). Our study did not undertake EPC_0 measurements for consideration of SRP removal; however, bed sediment P enrichment in ponds was measured using the sediment traps (Section 3.5.3). SRP retention in the ponds may also be aided by the persistently high nitrate concentrations, which can buffer the reductive dissolution of Fe and thereby limit any redox-mediated SRP release from sediment (Dupas *et al.*, 2017, 2018; Jarvie *et al.*, 2020).

An important biological nutrient removal mechanism for N in wetlands is bacterial metabolism, most commonly through nitrification and denitrification pathways (Vymazal *et al.*, 1998). In our study, dissolved ammonium concentrations showed no significant difference between the inlet and outlet, suggesting that nitrification is an unlikely cause of N removal in these ponds. However, denitrification is more likely to be occurring in the ponds to reduce nitrate to nitric oxide, nitrous oxide, and nitrogen gas (Cheng *et al.*, 2016). Previous studies showed that denitrification rates are increased under anoxic or low dissolved oxygen conditions, warmer temperatures, and an optimum pH of between 6 and 8 (Vymazal, 2007). A positive correlation between mean daily water temperature in the Central Pond and nitrate removal efficiency (linear regression, R^2 = 0.32,

p = 0.06) supports these findings. Temperature was only able to explain almost a third of the variation in removal efficiency, but this is justifiably low due to the other influential factors mentioned above not being considered. Nitrate removal in a constructed wetland in North Carolina showed a similar temperature dependence, with removal efficiencies of >90 % during the growing season (Mallin *et al.*, 2012). Although our monitoring showed reduced levels of nitrate removal during the winter, the net losses from the ponds during this period were minimal (<2 % concentration increase in the outflow) and therefore, not significantly affecting water quality. Further monitoring of the ponds (particularly dissolved oxygen measurements) may provide further data to help explain nutrient removal efficiencies and processes.

PP was not significantly reduced at the outlet, but both SSC and VSC were and had removal efficiencies of up to 70 % and 66 % respectively. It was hypothesised that the majority of the inflowing P load would be sediment-bound and settle out in the ponds, but results show that PP in the outflow remained just as high. A potential explanation could be that under lower flows, there was an increased export of planktonic algae from their proliferation in the ponds, as well as clay particles that remained suspended during low flow velocities. A study on stormwater control structures found evidence of PP release during low flows which were attributed to SRP release from anaerobic sediment, which was then adsorbed onto clay particles or assimilated by algae (Duan *et al.*, 2016). The observed reduction in SSC and VSC was expected due to the rapid reduction in flow velocity within ponds, which likely resulted in the deposition of larger particles with higher settling velocities nearest the inflow.

3.5.2 Storm Event Water Quality

The ponds were most effective at reducing suspended solids (SSC and VSC) downstream, and to a lesser extent P. The ponds may have a lower removal efficiency for TP/PP compared to SSC/VSC due to a high proportion of the P being bound to clay particles, which are more likely to remain in suspension compared to the coarser-grained particles. Studies show that the particulate fraction of P is often adsorbed onto the surface of particles such as metal oxides (e.g., iron oxides), or on clay particles (van der Grift *et al.*, 2018). Although water samples were unable to be analysed for particle size, the downstream increase in P content of suspended sediment in storm samples suggests that heavier particles settled out in the Upstream and Central Ponds, thereby reducing SSC, but having a smaller effect on PP concentrations, which are more heavily influenced by finer particles. Generally, SSC was able to explain most of the variation in suspended PP concentration, particularly under storm conditions (Figure 3.10).

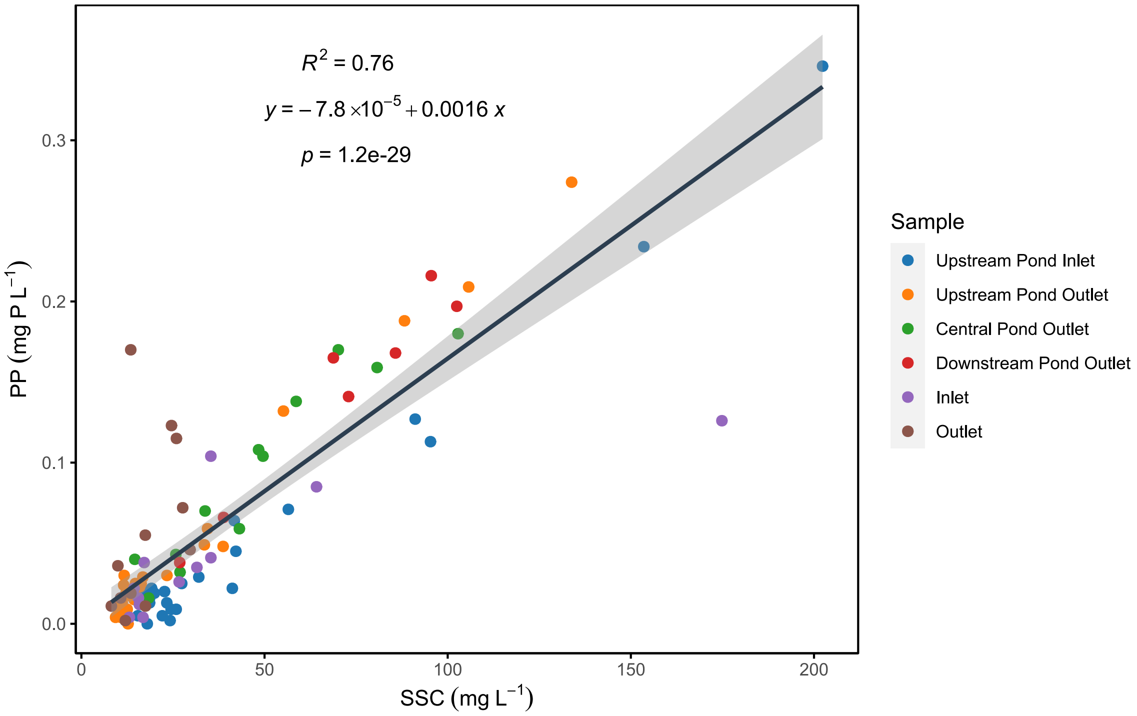


Figure 3.10 Linear regression of SSC (mg L^{-1}) and PP (mg P L^{-1}) from Pond Inlet/Outlet sampling sites during a storm event on the 12th and 13th March 2019, and near baseflow from the Inlet and Outlet (n = 96). The grey band represents 95 % confidence intervals.

It is thought that under near baseflow conditions, PP at the outlet tended to be high despite the low SSC because the finest particles, with much lower masses remaining in suspension even during very low flow velocities. The ~1:1 width-to-length ratios of the ponds are likely to have limited their potential to remove PP and clay sized particles, with previous work suggesting that width-to-length ratios play an important role in controlling hydraulic efficiency and pollutant removal efficiency (Persson and Wittgren, 2003). Longer ponds, with width-to-length ratios of greater than 1:4, have increased flow pathways and may, therefore, have improved ability to settle out and retain finer sediment particles and, consequently, PP (Persson, 2000; River and Richardson, 2018).

Particulate forms of P are also associated with organic P compounds (e.g., organophosphates used as pesticides), altogether making this fraction chemically and physically complex with highly variable stabilities, bondings and exchangeabilities (Poulenard *et al.*, 2008). A study modelling agricultural best management practices found that ponds were more effective at removing organophosphates, such as chlorpyrifos, that are more readily attached to sediment particles (Zhang and Zhang, 2011). Sediment-bound P can be released into the water column through resuspension and desorption, which is more likely to occur during high magnitude storm events.

This phenomenon is also likely to explain the increase in suspended sediment and TP loads downstream of the Central Pond during the large February 2020 event. Furthermore, during multiple events, both the Upstream and Central Pond were seen to be partially overtopping their banks when outflows were not able to drain fast enough to accommodate the inflowing discharge.

Barber and Quinn (2012) found that during a storm event an on-line run-off attenuation feature was not able to reduce SSC, TP, or nitrate by significant levels at its outlet. In terms of sediment and P they attributed these findings to the resuspension of previously deposited material, highlighting this as a key drawback of interventions of this style. Our study also found that nitrate was not retained during the storm event in March 2019, but instead appears to have been flushed out of the Upstream Pond at a higher concentration.

Suspended material entered the pond system at notably higher concentrations during the October and February events, where it was observed that a substantial overland flow pathway located just upstream of the Upstream Pond was active. During the October event, run-off from this pathway (shortly after the storm's peak) contributed an SSC of 219 mg L^{-1} and a TP concentration of 0.75 mg L^{-1}, whilst SSC at the inflow was approximately four times higher, with a TP concentration of 1.1 mg L^{-1}. It is likely that a combination of antecedent conditions and intense rainfall brought about the activation of this critical source area to significantly increase stream sediment delivery from the hillslope.

3.5.3 Pond Sediment Quality

Sediment particle size is a key parameter in determining the transport and fate of pollutants in streams (Walling and Woodward, 2000). The chief concerns for water quality are the smallest sediment particles (clay and fine silt) that are capable of transporting large quantities of bound P when entrained and remain in suspension for longest (River and Richardson, 2018). High concentrations of fine sediment typically result in turbid water and have been shown to have adverse ecological consequences (both in suspension and when deposited) for primary productivity, aquatic food webs, benthic macroinvertebrate communities and salmonid spawning habitats (Ryan, 1991; Henley *et al.*, 2000; Harrison *et al.*, 2007; Sear *et al.*, 2017). Results from the sediment traps show that it was mostly the silt fraction (2–63 µm) being deposited in all three ponds; however, there was a shift towards a smaller median particle size along the sequence. This is largely a consequence of the deposition and filtering out of sand particles within upstream ponds which significantly decreases their proportion within sediment downstream. Visual observations showed that there was considerable build-up of coarse matter at pond inlets. Similar results were found by Cooper *et al.* (2019), who showed a decrease in the mean particle size of deposited sediment along the length of a constructed wetland. Comparable results were found by Ockenden *et al.* (2014) who also observed that in paired field wetland ponds, the median particle size was typically larger in the first pond, which reflects the results of our study. They also found that sediment nutrient concentrations were generally higher in the second pond of the pair. In our study, the observed increase in SSA along the pond sequence would suggest that the Downstream Pond may have a higher capacity to trap P, given the importance of particulate surface area for adsorption of sediment-associated contaminants (Horowitz and Elrick, 1987; Walling *et al.*, 2000). However, our data do not support this idea and, in fact, show the opposite relationship, suggesting that particle size characteristics are not the dominant influence on sediment-bound P within this system. A potential explanation for this may be that organic matter within ponds has a greater contribution to sediment P enrichment. Sediment P content appeared to show a seasonal pattern that peaked during September. The increase in P content with organic matter suggests that a significant proportion of P within the pond sediment was derived from plant material, with the strongest correlation seen in the Central Pond. A similar relationship was observed in riverbed sediment from the River Blackwater, where organic matter and iron content explained 59 % of variation in P (House and Denison, 2002). During the autumn, there was a noticeable increase in leaf litter found within sediment traps, particularly in the Upstream Pond which is immediately downstream of a ~350 m length of stream with dense riparian tree cover. Decomposition of the macrophytes within ponds is likely to have been a key source of autochthonous organic matter and P, particularly in the Central Pond where macrophyte cover was greatest. It is likely that the ponds may have different rates of internal P cycling processes such as the release of dissolved P from sediment back into the water column as a result of organic matter breakdown and mineralization (Sinke *et al.*, 1990). This dissolved P may also accumulate within interstitial pore water in the pond sediment, allowing it to be assimilated by rooted macrophytes, e.g., *Typha*, or potentially released into the water column if significant disturbance and remobilisation occurs as a result of a storm event (Reddy *et al.*, 1999; Frost *et al.*, 2019).

The ratios of PP to suspended sediment at both the inlet and outlet of the Upstream Pond during the March storm event were over double those of deposited sediment within the traps. However, the ratio at the inflow during near baseflow conditions was 20 % lower. These relationships between sediment and PP content suggest that there may be release of P from sediment within the pond during events, similar to the findings of Barber and Quinn (2012), whereas the sediment is more likely to become enriched in P under average flow conditions.

3.5.4 Pond Capacity

The ponds showed significant accumulation during their first two years of being in operation, with an estimated annual reduction in capacity of 10 % for the Upstream Pond, and 5 % for both the Central and Downstream Pond. The storm event sampling data suggest that loads of up to 66 kg sediment can be retained during stormflows in the absence of flushing; however, up to 37 kg could be lost when flushing occurs. Further establishment of pond vegetation is likely to reduce the risk of flushing, with previous evidence demonstrating a decrease in sediment resuspension with increasing macrophyte cover (Braskerud, 2001). Continued monitoring over successive years is needed to investigate this effect.

Without further intervention to maintain storage capacity, it is also possible that the ponds may undergo periods of net accumulation followed by net export if event magnitude is sufficiently large. In light of this, ponds may be even more prone to flushing in the future with climate change predicted to intensify extreme precipitation events and flood risk (Tabari, 2020). In order to overcome and limit the issue of remobilisation and flushing of accumulated matter downstream during high magnitude events, regular maintenance will be required. Ponds can be dredged most efficiently during periods of low flow in summer when water levels are minimal. The first pond within a sequence will need dredging more frequently (at least every two years) than the following ponds. There should also be consideration of the impacts of dredging on pond ecology where maintenance activities may damage habitat and remove vegetation. Keeping sections of vegetated sediment intact will aid recolonisation and reduce the risk of resuspension following maintenance. In practice, maintenance frequency will likely be a trade-off between its cost and the effectiveness of the ponds as mitigation measures.

3.5.5 Ecology

During the ~2.5 years since creation, the stream reach, ponds, and marginal areas were colonised from bare soil into wetland habitat, with a plant species richness of 31 as of August 2020. A range of benthic macroinvertebrate taxa were also recorded throughout the monitoring period from a total of 22 different families. The presence of filamentous green algae was observed within all ponds throughout the monitoring period, likely due to a lack of shading and persistently high nitrate concentrations.

In terms of the potential ecological impact of P being exported from the ponds, it can be said the risk for eutrophication downstream is low due to the flushing phenomenon being observed during the winter period when flows are typically high enough for sufficient dilution of P. Intense convective storms during summer are likely to pose a greater risk for eutrophication, though their occurrence is less frequent. In terms of fine sediment flushing, there are potential risks of contributing to benthic smothering, since during winter many benthic spawning biota have eggs in the riverbed. Further research into the ecological impacts of pond features on downstream communities would be beneficial for a more holistic evaluation of overall costs and benefits, particularly if monitored over a longer timescale.

3.5.6 Implications for Catchment Management

Trapping effectiveness was highly variable across the monitoring period for different water quality determinands and hydrological conditions. The surprisingly high accumulation of sediment in the ponds compared to the downstream catchment flux during August may be a result of several convective storms occurring during this period. These events may have been considerably localised, thereby mobilising sediment upstream of the ponds, but only having minimal impact on sediment transport in the rest of the catchment. In the context of the wider 340 ha catchment, the total accumulations in the ponds made up significant proportions within the overall budget (7.6 % of suspended sediment; 6.1 % of silt and clay; 3.2 % of P) given that the ponds only drain 8.8 % of the Downstream Catchment. It is important to note that the proportion of the flux trapped by ponds is likely to represent an upper estimate, because not all of the sediment would have necessarily been transported to the Downstream Catchment Outlet, particularly the larger particles. The estimated flux of clay and silt is, therefore, a more realistic representation of the suspended load exported from the catchment. Our findings highlight how the ponds show most potential for reducing downstream sediment loads, but are less efficient for mitigating diffuse agricultural P pollution. Despite only covering a small area (<0.02 %) of the wider catchment, the ponds trapped a disproportionately large percentage of the fine sediment flux leaving the catchment. This highlights the importance of locating ponds where they will intercept high yielding run-off pathways within the catchment, and also makes them a particularly beneficial mitigation intervention where space is limited, and it is not economically viable for farms to lose large areas of agriculturally valuable land. Currently in the UK (and under the WFD) there are no regulatory limits on fluvial suspended sediment concentrations or yields. Without robust and specific sediment targets, the estimated pond sediment accumulations are difficult to assess in terms of ecological and regulatory significance; nevertheless, such interventions show useful potential as management tools in the delivery of on-farm pollution mitigation.

This paper provides further evidence on how the trapping efficiency of in-stream pond features is often dependent on the magnitude and frequency of storm events they experience, with high discharge and sediment inputs leading to a rapid reduction in storage capacity and causing ponds to overflow. These issues could potentially be alleviated by altering pond designs to allow greater storage capacity or incorporating additional features (e.g., vegetated swales, woody debris dams) to capture and filter overflow. It is important to note that the young age of the ponds may also play a role in their limited ability to remove pollutants such as fine sediments. The expectation from catchment management efforts is often that observable benefits in pollutant reductions will be delivered shortly following implementation. The evidence presented here only shows a 'snapshot' of the ponds' functioning and trapping efficiency in the short term, and it is very likely to change with continued geomorphological evolution of the stream channel and further colonisation and succession of vegetation. Continued monitoring would be beneficial for evaluating the ponds' performance over a time period that allows for maintenance and revegetation to take place. The capacity reduction of the ponds observed during this two-year period necessitates regular maintenance and poses the potential opportunity for disposal of deposited pond sediment back into the landscape. The sediment has value for farmers that can capitalise on its nutrient content by redistributing it on arable fields as a soil conditioner, though critical source areas should be avoided to minimise the risk of mobilisation following application. The accumulated pond sediment properties show good suitability for agricultural application, having high organic matter and silt content (silty loam texture) and thus good water holding capacity. Previous research demonstrates that dredged fluvial sediments can increase crop productivity if added to soil with poor agricultural characteristics, for example where soil organic matter has been depleted (Darmody and Ruiz Diaz, 2017).

Even with the implementation of on-line pond features in agricultural headwaters, the delivery of other mitigation interventions and sustainable management practices are still required to enable the best chance of achieving ecologically significant improvements to water and habitat quality in downstream catchments (Melland *et al.*, 2014). The value of the wider co-benefits from pond features is also important to consider in their overall evaluation and contribution to achieving catchment management objectives related to habitat and the aesthetic quality of the landscape. Monitoring of water quality and ecology can help assess benefits and risks post-implementation, thereby informing decisions on adaptive management for improving interventions.

3.6 Conclusions

This paper demonstrates how the effectiveness of on-line ponds for the mitigation of diffuse agricultural pollution on clay soils with a 2.5 % slope can be highly variable due to the different retention capacities of sediment and nutrient fractions under different hydrological conditions. During baseflows, ponds reduced dissolved nitrate and SRP concentrations by averages of 29 % and 5 %, respectively. Despite their small size (<0.05 ha) and contributing area (30 ha), the on-line ponds were capable of accumulating significant pollutant masses over a seven-month period, equating to 7.6 %, 6.1 % and 3.2 % of the wider catchment (340 ha) suspended sediment, silt and clay, and P fluxes, respectively. However, data suggest that net losses of sediment and P can occur during higher magnitude storm events, with this risk likely to increase as pond storage capacity reduces. The ponds are most advantageous for capturing silt and sand-sized material during smaller to medium events typically experienced during winter. This design of on-line pond with a ~1:1 width-to-length ratio is less effective at mitigating TP loading. We recommend that pond maintenance should be considered on a biennial basis, and removed sediment be reapplied to arable land as an organic-rich soil conditioner. In addition to the implications for water quality, these interventions provide benefits for habitat diversity and potential for flood attenuation in NFM schemes. On-line ponds are likely to be most effective when they are well-managed and used in combination with other mitigation measures, particularly helping to improve functioning during more extreme storm events. Further research into the longer-term evolution of the on-line pond system would help evaluate changes in its functioning over time with continued development of its geomorphology and vegetation.

3.7 Author Contributions

Conceptualization, J.R., G.O., P.R. and D.S.; methodology, J.R., G.O., P.R., D.S. and D.G.-T.; field work, J.R. and J.B.; data analysis, J.R.; investigation, J.R.; resources, G.O. and P.R.; writing—original draft preparation, J.R.; writing—review & editing, G.O., D.S., P.R., D.G.-T., J.B., J.O. and D.M.; visualization, J.R.; supervision, G.O., D.S., P.R. and D.G.-T.; funding acquisition, G.O. and D.S. All authors have read and agreed to the published version of the manuscript (Appendix B).

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Chapter 4 Nature-based Solutions Enhance Sediment and Nutrient Storage in an Agricultural Lowland Catchment

4.1 Abstract

In this paper, nature-based solutions (NBS) include: (1) Natural Flood Management (NFM) interventions with a primary function of flood risk reduction but with additional multiple benefits for water quality improvements through the mitigation of diffuse pollution; and (2) ponds with a primary function of water quality improvement. This study assesses the ability of these NBS to trap pollutants in run-off within two small (3.4 km^2) agricultural catchments (Upper Thames, UK). The masses of sediment, phosphorus and organic carbon trapped by 14 features (since construction 2-3 years previously) were quantified through sediment surveying and sampling. Streamflow and suspended sediment monitoring downstream of features enabled catchment yields to be calculated. The features trapped a total of 83 tonnes sediment, 122 kg phosphorus, and 4.3 tonnes organic carbon. Although the footprint of the features was <1 % of the catchment area, they drained 44 % of the total land area and captured the equivalent of 15 % of the total suspended sediment yield, 10 % of the total phosphorus yield, and 8 % of the particulate organic carbon yield as monitored at the catchment outlet over the monitoring period. Results reveal that accumulation rates were influenced by hydrological connectivity, with greater accumulation in features constructed directly on streams (online ponds), and those offline features that filled from overbank flows. The low to moderate accumulation rates observed in offline features suggests that their floodwater storage potential is only likely to significantly reduce in the medium-term, necessitating maintenance after ~10-years. Compared with topsoil in each contributing area, trapped sediment was enriched in phosphorus and carbon in the majority of features, having on average 50 % higher phosphorus and 17 % higher organic carbon concentrations than surrounding arable soils, highlighting its potential value for redistribution on farmland. Monitoring results demonstrate the potential of NBS, including NFM, to mitigate diffuse pollution in lowland catchments.

4.2 Introduction

Soils are crucial to sustaining agricultural production and food security globally (FAO, 2015; Pozza and Field, 2020). However, soils are threatened by the acceleration of erosion from water due to anthropogenic pressures including land-use and climate change (O'Neal *et al.*, 2005; Ockenden *et al.*, 2016; Borrelli *et al.*, 2020). Intensification of the water cycle as a result of climate change is predicted to bring more intense rainfall and associated flooding which will in turn exacerbate the issue of soil loss (Burt *et al.*, 2016; IPCC, 2021). In temperate regions, shifts in the timing of heavy rainstorms from summer to autumn may also increase soil loss, particularly in bare arable fields where soil is susceptible to erosion following harvest (Routschek *et al.*, 2014). Historically agricultural landscapes and their watercourses have typically been heavily modified to enable efficient drainage and maximise crop production (Evans *et al.*, 2007; Pierce *et al.*, 2012). Technological advances such as the mechanisation of farming and changing trends in the growing of certain crops have impacted soils in many ways and intensified their erosion over time. Increased hydrological connectivity of the land to streams facilitates the transfer of water, mobilised soil particles and solutes into watercourses via surface run-off or subsurface drains. This has negative onsite impacts in terms of soil health and nutrient losses, but also costly and undesirable offsite consequences on downstream flood risk, water quality and biodiversity (Pimentel, 2006; Evans, 2010; Boardman, 2013, 2021; Mondon *et al.*, 2021).

Soil conservation practices such as no-till farming can be implemented on arable fields to help mitigate soil erosion and associated impacts, with experimental evidence suggesting that reduced tillage can significantly reduce sediment delivery on both conventional and organic farms (Seitz *et al.*, 2019). Conservation agriculture has been found to enhance soil organic carbon and in turn improve soil structure, infiltration and water storage which reduce soil loss (Page *et al.*, 2020). However, further evidence shows how conservation practices can be less economical and less effective for mitigating certain nutrient losses. Bertol *et al.* (2017) found that nutrient and organic carbon concentrations in run-off from no-till were higher than from conventional tillage, with the cost of erosion losses from no-till being 29 % higher in terms of phosphate fertiliser. These differences demonstrate potential trade-offs and highlight how the effectiveness of soil conservation practices may vary considerably across different landscapes due to factors such as soil or crop type (Deasy, Quinton, *et al.*, 2009; Choden and Ghaley, 2021). Although changing agricultural practices may be part of the solution, mitigating soil loss and diffuse pollution may require additional interventions.

In recent years there has been an increased interest in nature-based solutions (NBS) and natural infrastructure to mitigate environmental problems such as climate change, biodiversity loss, pollution, and hydrometeorological hazards in a more integrated way (Seddon *et al.*, 2020; Suttles *et al.*, 2021). One such approach being adopted in the UK and across Europe is Natural Flood Management (NFM), which aims to work with hydrological processes to slow and store water in the landscape to deliver multiple environmental and societal benefits (Lane, 2017). NFM is part of the wider concept of working with natural processes (WwNP) which the Environment Agency describe as aiming to "*protect, restore and emulate the natural functions of catchments, floodplains, rivers and the coast*" (Environment Agency, 2018; Fryirs and Brierley, 2021). NFM encompasses a broad variety of interventions, including the creation of woodland, addition of instream leaky woody dams/barriers, and construction of offline storage features. These offline features are used to temporarily hold back water in the landscape, reducing flood risk through attenuating run-off or by receiving overflow from stream channels, thereby also capturing diffuse pollutants, creating wetland habitat and storing carbon (Evrard *et al.*, 2008; Barber and Quinn, 2012; Ockenden *et al.*, 2014; Williams *et al.*, 2020). Offline features typically fill from diffuse overland flow, but can also be designed to store overbank flows. On the other hand, online features can be defined as ponds receiving flow directly from a stream, and are typically used as a NBS for water quality improvement.

Current evidence on the effectiveness of NBS to deliver multiple benefits is limited, but the rollout of several small NFM schemes has created new opportunities for gathering empirical data (Dadson *et al.*, 2017; Wingfield *et al.*, 2019). Findings from the Belford catchment (northeast England) suggest that offline features are able to retain significant volumes of sediment, but online features showed a lack of retention during storm events (Barber and Quinn, 2012; Wilkinson *et al.*, 2014). Modelled evidence suggests that peak suspended sediment and total phosphorus concentrations could be reduced by 5 to 10 % from adding 2000 to 8000 m^3 of storage in the pasture-dominated Newby Beck catchment (Adams *et al.*, 2018). Despite the policy relevance and growing interest in NBS such as NFM, the knowledge base (particularly on offline features) is lacking evidence for lowland catchments that cover large parts of the south and east of England (Lockwood *et al.*, 2022). Questions have also been raised over the sustainability of water storage in such features where rapid sediment deposition could diminish storage capacity over time (Lane, 2017). Evidence on their efficacy and the delivery of benefits is needed to support agri-environmental policies such as the UK government's Environmental Land Management (ELM) scheme which could provide farmers with financial incentives for adopting NFM and other NBS, thereby increasing uptake more widely (Holstead *et al.*, 2017; Bark *et al.*, 2021). This study therefore aims to quantify the accumulation of sediment, phosphorus, and organic carbon in offline NFM features and online pond features within a small predominantly arable lowland catchment. Specifically, two key research questions are addressed:

1. How has the implementation of NBS altered the catchment storage and yields of sediment, total phosphorus, and particulate organic carbon?
2. What factors influence accumulation rates within offline and online features?

The sustainability of these features over the long-term is considered and the suitability of the accumulated sediment for redistribution on arable land is assessed to help inform management guidance for NFM schemes.

4.3 Methodology and Methods

4.3.1 Study Site

Accumulations of sediment, phosphorus and carbon were measured across a variety of offline and online features implemented as part of the Littlestock Brook NFM scheme, upstream of the village of Milton-under-Wychwood, Oxfordshire (Figure 4.1; see Appendix D (S1) for photographs of storage features). The studied features were constructed between February 2018 and February 2019 and vary in their design and hydrology (Table 4.1). Further details on the Littlestock Brook NFM trial are given by Old *et al.* (2019) and Robotham *et al.* (2021).

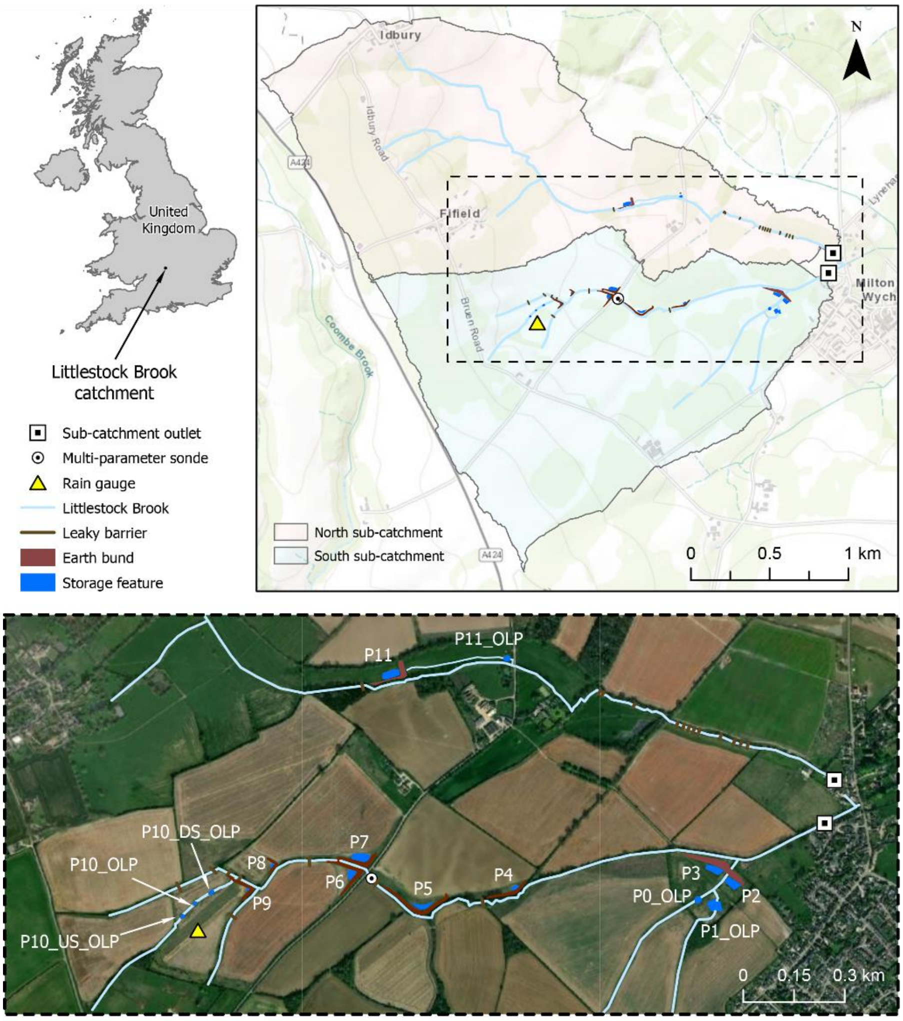

Figure 4.1 Location of the Littlestock Brook, the NFM features, online ponds and monitoring instrumentation within the north and south sub-catchments (both 3.4 km^2). Features and ponds are labelled according to the naming conventions detailed in Table 1.

Table 4.1 Characteristics of offline and online features in the Littlestock Brook NFM scheme. Features appended with 'OLP' denote online pond features, and those without denote offline features. The 'US' and 'DS' prefixes are used to denote the most upstream and downstream features in the series of ponds, respectively.

Storage Feature	Construction Date/Location	Max. Volume (m^3)	Contributing Area (ha)	Description
P0_OLP	February 2019 (South sub-catchment)	35	41.0	Permanently wet, draw-down in summer.
P1_OLP		440	30.5	Seasonally wet, connected to stream during winter.
P2		53	1.1	Seasonally wet, fills in large run-off events.
P3		514	0.4	Permanently wet, fills in large run-off events, fed by field drains.
P4	February 2018 (South sub-catchment)	857	3.9	Normally dry, fills in large run-off events, partly connected to leaky barrier spillway.
P5		3504	4.0	Permanently wet, fills in large run-off events, connected to leaky barrier spillway.
P6		2647	6.5	Normally dry, fills in large run-off events, drains easily due to connection to downslope field drain.
P7		2719	9.5	Permanently wet, fills in large run-off events, fed by upslope field drain.
P8		569	1.0	Normally dry, fills in large run-off events.
P9		860	20.1	Normally dry, fills in large run-off events.
P10_US_OLP		70	30.0	Permanently wet, draw-down in summer.
P10_OLP		90		Permanently wet, draw-down in summer, fills from outflow of P10_US_OLP.
P10_DS_OLP		95		Permanently wet, draw-down in summer, fills from outflow of P10_OLP.
P11	February 2019 (North sub-catchment)	2533	13.3	Normally dry, fills in large run-off events, connected to leaky barrier spillway.
P11_OLP		83	0.6	Permanently wet, draw-down in summer, filled by P11 outflow in large events.

The Littlestock Brook catchment is located within the predominantly rural River Evenlode catchment, a tributary of the River Thames (southern England, United Kingdom). The area upstream of Milton-under-Wychwood is drained by two sub-catchments (referred to as North and South). The North sub-catchment consists mainly of permanent improved grassland used for grazing cattle and sheep, whilst the South sub-catchment is largely arable. The area has a minimum and maximum elevation of 103 m and 202 m respectively, and an average slope of 6.3 %. The western part of the catchment is underlain by a limestone geology with shallow lime-rich soils. Further down the catchment, the soils are largely seasonally wet, slowly permeable clay and loamy soils with some impeded drainage. The area receives an average annual rainfall of 765 mm and experiences an average annual minimum and maximum temperature of 5.7 °C and 13.1 °C respectively (Met Office, 2021).

Online features are defined as areas that are connected to a watercourse, either directly (e.g. constructed on a pre-existing stream), or indirectly (e.g. via a newly-excavated channel that allows flow into and out of the area). Some indirectly connected online features are filled by seasonal intermittent or ephemeral flow. For example, P1_OLP is filled by a stream channel that flows regularly during October to March, but outside of this window, the channel is only activated temporarily in response to significant rainfall. Offline features are areas that typically fill from overland flow during rainfall events (Figure 4.2a). However, many features are also co-located with instream wood features (leaky barriers) and spillways/swales that allow the features to fill from diverted streamflow in higher magnitude events (Figure 4.2b). The contributing areas of features were estimated in a GIS using Environment Agency National LIDAR Programme DTM (digital terrain model) 2020 at a 1 metre resolution (Environment Agency, 2022). To take account of overbank flow diverted by leaky barriers during higher magnitude storms, 'event contributing areas' were also calculated to estimate the drainage areas upstream of spillways where this phenomenon was observed. To do this, ArcMap hydrology tools were used to delineate the area draining to each spillway associated with a feature, and then the overland contributing areas of any upstream features that fell within this delineated area were subtracted to avoid double counting.

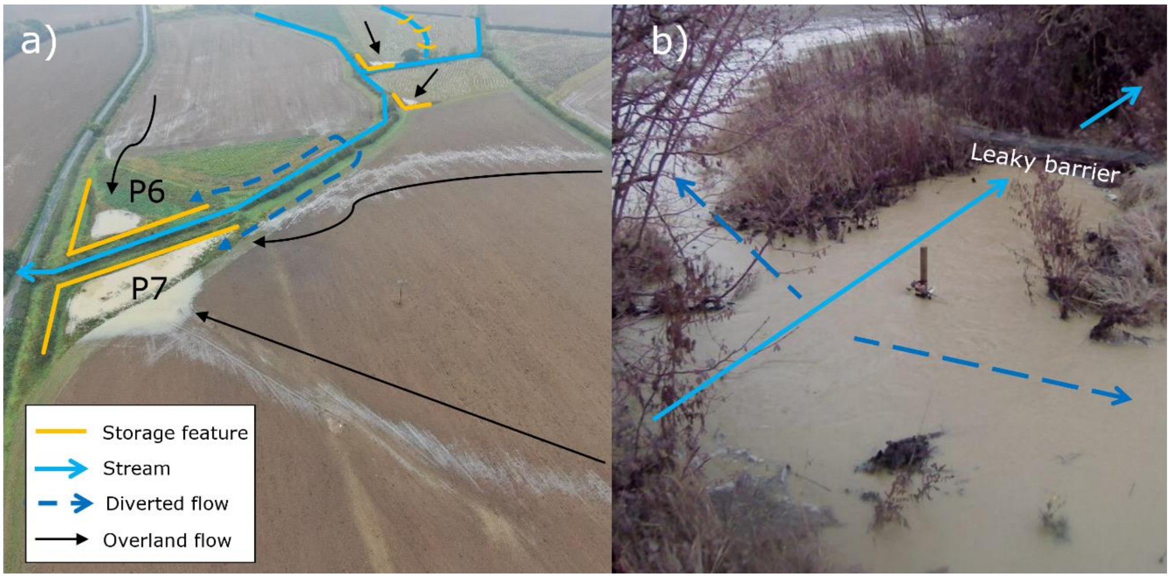

Figure 4.2 **(a)** Offline features (P6 and P7) filling from overland flow pathways during a storm event on 4th October 2020; **(b)** A leaky barrier diverting streamflow into P6 and P7 via spillways during a storm event on 23rd December 2020.

4.3.2 Sediment and Soil Sampling

Sediment accumulations were estimated from measurements of sediment depth and bulk density. Sediment cores were sampled within each feature to determine the average bulk density of accumulated sediment. A coring device suitable for sampling soft, submerged sediment was made from 1 m long copper pipe (2.6 cm in diameter), cut at a 45° angle on one end to aid insertion into the sediment. Six cores were taken from each storage feature (half in shallower sections closer to feature margins, and half in deeper central sections). Sediment depth (down to the solid base of the storage feature) was also measured at each coring location to determine the original core length prior to any potential compaction that occurred during coring. Cores were stored in plastic sample bags and refrigerated at 4°C until being transferred into aluminium trays and oven-dried at 105°C for at least 36 hours before being weighed. Dry bulk density was calculated following guidance of Wood (2006). Loss-on-ignition (LOI) was quantified as a proxy measure for organic matter (OM) content. The samples were heated for 2 hours at 500°C before being cooled in a desiccator and re-weighed (Standing Committee of Analysts, 1984). OM was converted into organic carbon (OC) content using a 0.58 conversion factor chosen based on the literature (Bhatti and Bauer, 2002; De Vos *et al.*, 2005; Rollett *et al.*, 2020). The total phosphorus (TP) concentration of the sediment was determined spectrophotometrically. The ashed sample was crushed into a fine powder and combined into a bulk sample for each feature from which triplicate sub-samples of 3±0.1 mg were then taken for determining average TP content. Sub-samples were mixed with 20 ml ultrapure water and analysed following the modified molybdenum blue methodology of Eisenreich *et al.* (1975).

Alongside the cores, additional sediment was sampled for use in determining absolute particle size distribution and characteristics using laser diffraction particle size analysis (Mastersizer 2000, Malvern Panalytical; Malvern, UK). Prior to analysis, 0.5 to 0.6 g sub-samples of sediment were treated with a 5 % sodium hexametaphosphate solution and agitated for 5 minutes in an ultrasonic bath to disperse particles and prevent agglomeration.

Topsoil within each of the feature's contributing areas (listed in Table 4.1) was sampled for the determination of OC, TP, and absolute particle size distribution. Within each contributing area, five soil samples were taken using a trowel to dig out the top 5 cm of soil, following a W-shape pattern across the dominant land-use to obtain spatially representative samples (Peters and Laboski, 2013). A total of 60 soil samples were taken across three land-use types (arable, grassland, and arable reversion). In the lab, 0.5 to 0.6 g sub-samples were taken from each soil sample for particle size analysis. The remaining soil was air-dried in aluminium trays at 30°C for at least 72 hours before being crushed and sieved to <0.4 mm, also removing roots and stones from the sample. The soil was then oven-dried overnight at 105°C before being weighed. OM content was determined using LOI at 500°C and also converted to OC. Soil TP content of samples was then analysed and averages determined for each contributing area. Variability of soil properties between samples was visualised prior to averaging (see Appendix D (S2) for boxplots of key properties). Enrichment Ratios (ER) were calculated by dividing mean constituent concentrations in sediment samples by mean concentrations in soil samples (Sharpley, 1980). Uncertainties for ER values were calculated as 95 % confidence intervals using Fieller's theorem (Fieller, 1940).

4.3.3 Storage and Sediment Volumes

The depths of accumulated sediment within each feature were surveyed along transects spanning the length and width of the feature, with measurements being taken at 1 to 2 m intervals. Depths were measured to the nearest cm from the solid base of the feature to the surface of the soft sediment layer using a metre rule. Transects were positioned so that they approximately captured the deepest section of the feature and a handheld GPS (eTrex, Garmin; Olathe, KS, USA) with a horizontal accuracy of 3 m was used to locate the start and end points of each transect. Where possible, one of the measurements was taken at a known reference point (stage board) in each feature to allow transects to be linked to this datum. Maintenance work to remove sediment from the series of P10 online ponds following their surveying in January and June 2020 meant that any future surveying would not represent the accumulation since construction. As a result, sediment depths measured for these features represent a shorter period of accumulation compared to the other features which were measured following a longer period post-construction and with no maintenance. Sediment depths were spatially interpolated using the natural neighbour interpolation method (ArcMap 10.5, Esri; Redlands, CA, USA) to estimate stored volumes. The bulk density measurements were then used to convert sediment volumes into masses, and concentration data were used to calculate total stored nutrient masses. Uncertainties are reported as standard deviation unless otherwise stated. A combination of LIDAR and real-time kinematic global positioning system (RTK GPS) (GS14, Leica Geosystems; St. Gallen, Switzerland) surveys of the features post-construction were similarly used to estimate their total storage volumes. Stage boards and water level sensors (Rugged TROLL 100, In-Situ; Redditch, UK) were installed in 11 features (the exceptions being P0_OLP, P10_DS_OLP, P10_US_OLP, and P11_OLP) to measure water depth at 5-minute intervals. Depth-volume relationships were derived in a GIS and used to produce time-series of stored water volumes in the different features. The length-to-width ratios of features were measured by dividing length by width. To keep the metric as consistent as possible between the different types of features, lengths were defined as the distance from the inlet to the outlet.

Instream stage boards and water level sensors located by leaky barrier spillways were used to determine when certain features were filling from the stream. The overflow elevation of spillways were surveyed with RTK GPS, with overflow into P6 during storm events also being verified with hourly time-lapse camera imagery.

4.3.4 Catchment Yields and Water Quality

Yields (±95 % confidence intervals (CI)) of total and fine suspended sediment (SS), particulate organic carbon (POC), and TP were calculated at the two sub-catchment outlets using discharge and concentration data at 5-minute intervals. Stream discharges were estimated using a stage-discharge rating curve developed from flow measurements made using an Electromagnetic Current Meter (Valeport; Totnes, UK) and the velocity-area method (Herschy, 1993). Some low flow measurements were made using conductivity sensors (EXO1, YSI; Yellow Springs, OH, USA) and the salt dilution method (Hongve, 1987). Measured discharges ranged from 6 to 587 L s^{-1} (n=15) for the south sub-catchment, and from 3 to 946 L s^{-1} (n=15) for the north sub-catchment. Instream turbidity sensors (DTS-12, FTS; Victoria, Canada) co-located at gauging sites were calibrated against suspended sediment concentration (SSC) and TP samples taken under a range of flows using a US DH-48 sampler and automatic samplers (Sigma SD900, Hach; Loveland, CO, USA). Time-series were quality controlled to remove suspect datapoints, with gaps of <12 hours filled by linear interpolation if no storm events were known to have taken place during the period.

Overall, turbidity/concentration data coverage was >99 % for the monitoring period. Particulate OC concentration was estimated using linear regressions of SSC against particulate OM concentration of water samples at each sub-catchment outlet (south sub-catchment R^2=0.97, *n*=184; north sub-catchment R^2=0.96, *n*=127) and the OC conversion applied. Time-series of instantaneous loads were calculated as products of discharge and concentration, and were then integrated to estimate yields over the monitored periods (Equation 4.1).

$$Flux = \int_{t_1}^{t_2} Q(t)C(t)dt \tag{4.1}$$

where Q = stream discharge; C = concentration of SS/TP/OC, and t = time.

Fine sediment (<63 µm) yields in each sub-catchment were estimated based on particle size distributions sampled during two high flow events (*n*=9 per sub-catchment). These particle size distributions were assumed to be generally representative of the stream's suspended load as large storm events typically deliver most of the total sediment yield (Chappell *et al.*, 2004). The proportions of fine particles in the samples were averaged and combined to estimate the yields of silt and clay from each sub-catchment. The stored masses of sediment, TP, and OC within NBS features were divided by the yields leaving the sub-catchment for each monitoring period to calculate stored masses as proportions of the total sub-catchment yield. Calculations and statistical analyses were carried out in R (R Core Team, 2018).

Instream water quality parameters (including turbidity and ammonium) were also measured at hourly intervals using a multi-parameter sonde (EXO2, YSI; Yellow Springs, OH, USA) deployed as part of Thames Water's 'Smarter Water Catchments' initiative (Thames Water, 2020). The sonde was located between P5 and P6/P7 (Figure 4.1) and operated using a pumped flow cell system which minimised sensor fouling. Rainfall was recorded at 2-minute intervals using a tipping bucket rain gauge (Casella; Sycamore, IL, USA), and quality controlled using a storage rain gauge by ensuring the measurements were within a 5 % tolerance.

4.4 Results

4.4.1 Sediment and Nutrient Storage

The total sediment, TP and OC captured by the NBS features varied by two orders of magnitude, ranging from 0.2 to 20.1 t sediment during the 2 to 3 years since construction (Table 4.2). Bulk density of the accumulated sediment had a mean of 0.69 ± 0.23 g cm^{-3} for online features and

0.93 ± 0.22 g cm^{-3} for offline features. The total accumulated mass of sediment in the eight offline features was 47.8 t, and 39 t in the six online ponds. Cumulatively, the 13 features within the south sub-catchment stored 83 tonnes sediment with a total volume of 108.8 m^3. The features were most effective in trapping sediment, with 14.7 % of the total sediment yield and 14.1 % of the fine (clay and silt) sediment yield stored compared to only 9.5 % and 7.5 % of the TP and POC yields respectively.

Table 4.2 Yields (±95 % CI), masses of accumulated sediment (t), Total Phosphorus (kg), and Organic Carbon (t) in NBS features and their equivalent proportion of the total suspended sediment, fine suspended sediment, TP, and particulate OC yields leaving the 3.4 km^2 south sub-catchment.

NBS Feature	Time period	Rainfall (mm)	Total sediment yield (t)	TP yield (kg)	POC yield (t)	Stored sediment (t)	Stored TP (kg)	Stored POC (t)	Total sediment yield stored (%)	Fine sediment yield stored (%)	TP yield stored (%)	POC yield stored (%)	Sub-catchment area drained (%)
P0_OLP	Feb 19–Mar 21	2126	498±24	1095±50	50±1	5.8	8.0	0.3	1.2	1	0.7	0.6	12.1
P1_OLP						6.2	14.4	0.4	1.3	1.4	1.3	0.8	9.0
P2						0.4	0.5	0.03	0.07	0.06	0.04	0.06	0.3
P3						20.1	28.9	1.1	4.0	3.7	2.6	2.2	0.1
P4	Feb 18–Mar 21	2810	565±30	1278±65	57±2	0.3	0.4	0.01	0.05	0.05	0.03	0.02	1.1
P5						7.0	11.0	0.3	1.2	1.3	0.9	0.5	1.2
P6						8.7	14.2	0.4	1.6	1.8	1.1	0.7	1.9
P7						10.6	16.7	0.4	1.9	2.1	1.3	0.7	2.8
P8						0.2	0.2	0.01	0.03	0.04	0.02	0.02	0.3
P9						0.6	0.6	0.02	0.1	0.1	0.05	0.03	5.9
P10_OLP	Feb 18–Jan 20	1634	160±10	417±26	16±1	4.6	5.5	0.2	2.9	2.5	1.3	1.3	8.8
P10_US_OLP						10.7	13.1	0.8	6.7	4.6	3.2	5.0	
P10_DS_OLP	Feb 18–Jun 20	1944	207±12	533±30	21±1	7.8	8.5	0.3	3.8	3.6	1.6	1.4	
P11_OLP	Feb 19–Mar 21	2126	605±102	1614±250	52±6	3.8	5.9	0.2	0.6	0.7	0.4	0.4	0.2
Total†		2810	565±30	1278±65	57±2	83.0	121.8	4.3	14.7	14.1	9.5	7.5	43.5

† P11_OLP is located within the north sub-catchment and is therefore excluded from the totals.

4.4.2 Factors Influencing Accumulation Rates

Due to differences in contributing area size and the influence of spillways, we expected the hydrological regimes of the features to be notably varied. Volume-duration curves exhibited a range of patterns (Figure 4.3). These curves show the variation in the frequency that the volume of water stored by each feature is exceeded in terms of a percentage of the features estimated maximum storage volume. P3 showed the greatest retention of water with 60 % of its capacity exceeded 50 % of the time, equating to a median storage volume of 338 m^3. P1_OLP and P2 both displayed a similar curve shape, however P1_OLP sustained water storage year-round whereas P2 stayed essentially dry during summer. P8 filled infrequently and only ever filled to 12 % (68 m^3) of its potential storage capacity during this period. P6 also had a flashy filling regime but stored significantly more water, reaching 26 % capacity (688 m^3), one order of magnitude greater than P8. In comparison P5 had a less steeply sloping curve, sustaining water storage for a greater duration and at its peak filling to 1475 m^3, 42 % of its potential capacity.

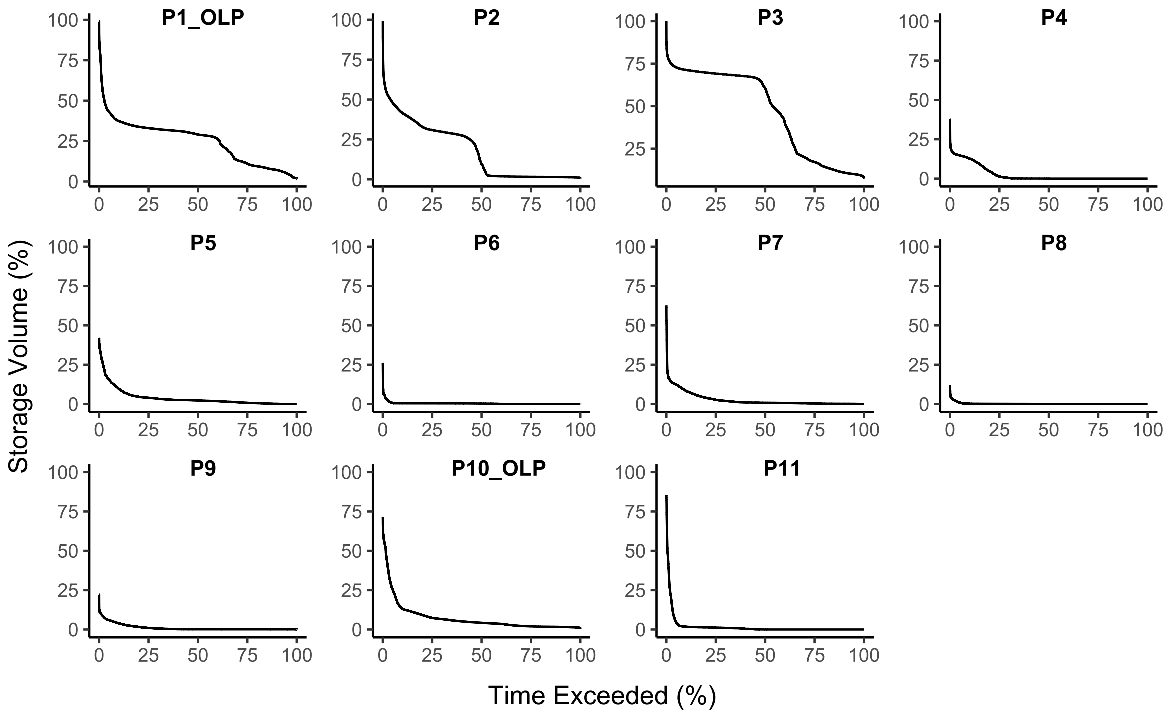

Figure 4.3 Volume-duration curves for different NBS features, showing storage volume (%) as a percentage of the maximum capacity of each feature during the 2019-2020 hydrological year. Volume was not monitored in the online features: P10_US_OLP, P10_DS_OLP, and P0_OLP.

On average, the sediment accumulation rate was 3.3 times higher in online features (20.8 ± 9.8 kg m^{-2} y^{-1}) than in offline features (6.3 ± 5.2 kg m^{-2} y^{-1}) when taking into account the ponded area of each feature. The length-to-width ratio of features explained some of the variation in accumulation rates, with positive relationships observed for both sediment (R^2=0.42, p<0.05) and TP accumulation (R^2=0.54, p<0.01). Length-to-width ratios were generally low and ranged from ~0.25 to 2.0, with P1_OLP having the highest ratio. Contributing area was also found to positively influence sediment accumulation rate (R^2=0.49, p<0.05). Differences in accumulation rate were better explained by event contributing area which broadly clusters the offline features into those activated by leaky barriers and those that were not (Figure 4.4). Features such as P9 were never observed to fill from overbank flows whereas P6 was frequently observed to do so during event peaks in winter storms (Figure 4.2b; Figure 4.5).

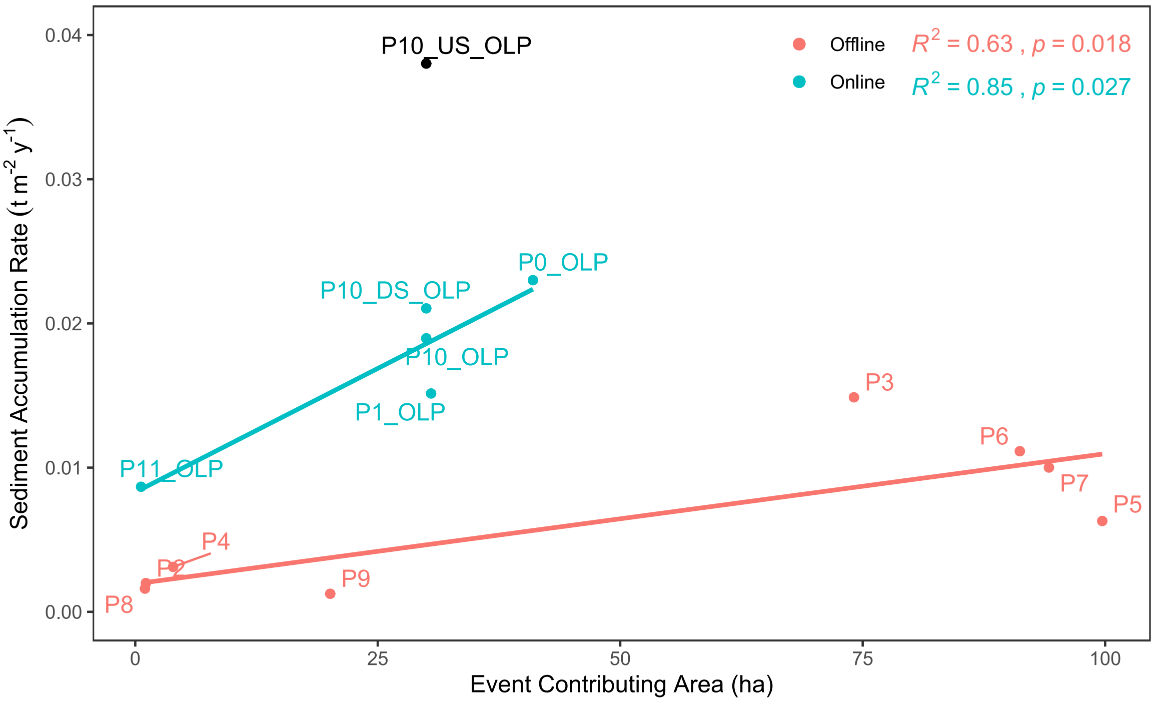

Figure 4.4 Linear regressions between event contributing area (ha) and sediment accumulation rate (t m^{-2} y^{-1}) for offline and online NBS features. P10_US_OLP is excluded from the regression.

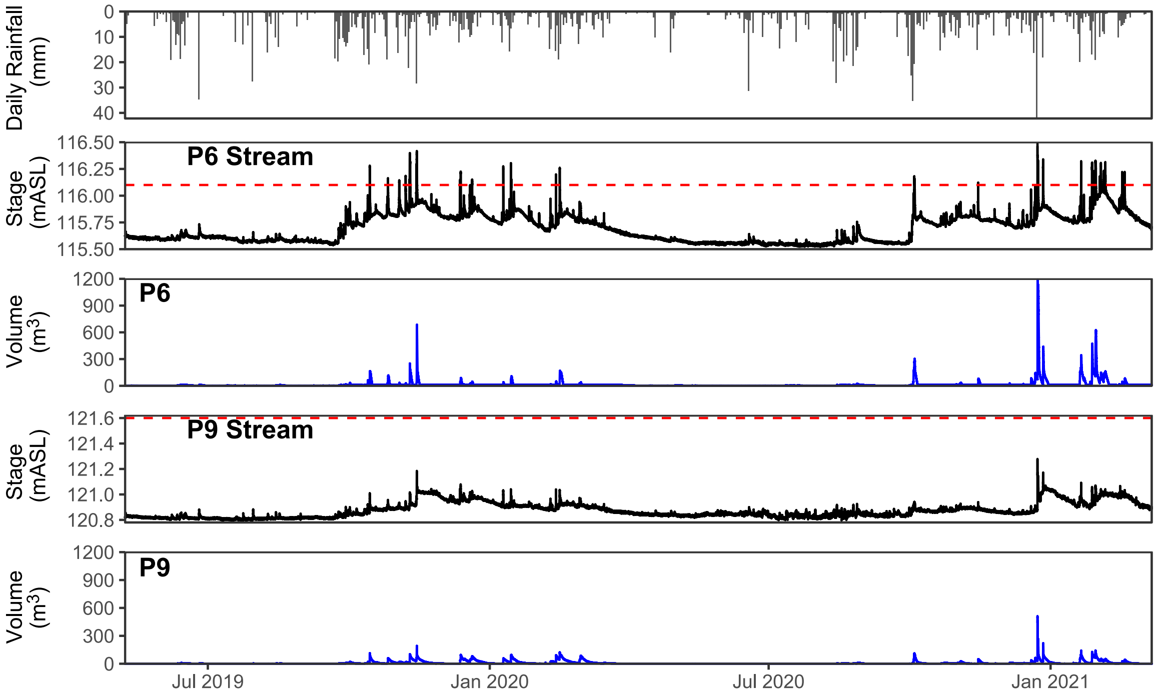

Figure 4.5 Time-series of daily rainfall (mm), and stream stage (mASL) at leaky barriers and water volume (m^3) in features P6 and P9. Dashed red lines indicate the threshold at which spillways are activated. mASL = metres above sea level.

The water storage dynamics of features along with stream stage data at their spillways provided insight into how features that filled from overbank flow via spillways compare to those that did not. We hypothesised that this additional hydrological connectivity would augment sediment delivery and thereby accumulation within features that received overbank flow. P6 and P9 exemplify this contrast (Figure 4.5). Overbank flows by the leaky barrier and spillway connected to P6 occurred in over 20 storm events between October 2019 and March 2021 and helped to fill the feature. In contrast, the threshold for overbank flow was never reached at the P9 spillway, even at the peak of the highest magnitude event in December 2020 the water level was still 0.3 m below the threshold. During this event, peak storage in P6 reached over double the volume in P9. The timing of overbank flow is well aligned with stream SSC, allowing the highest sediment load to be diverted into P6 during event peaks (Figure 4.6).

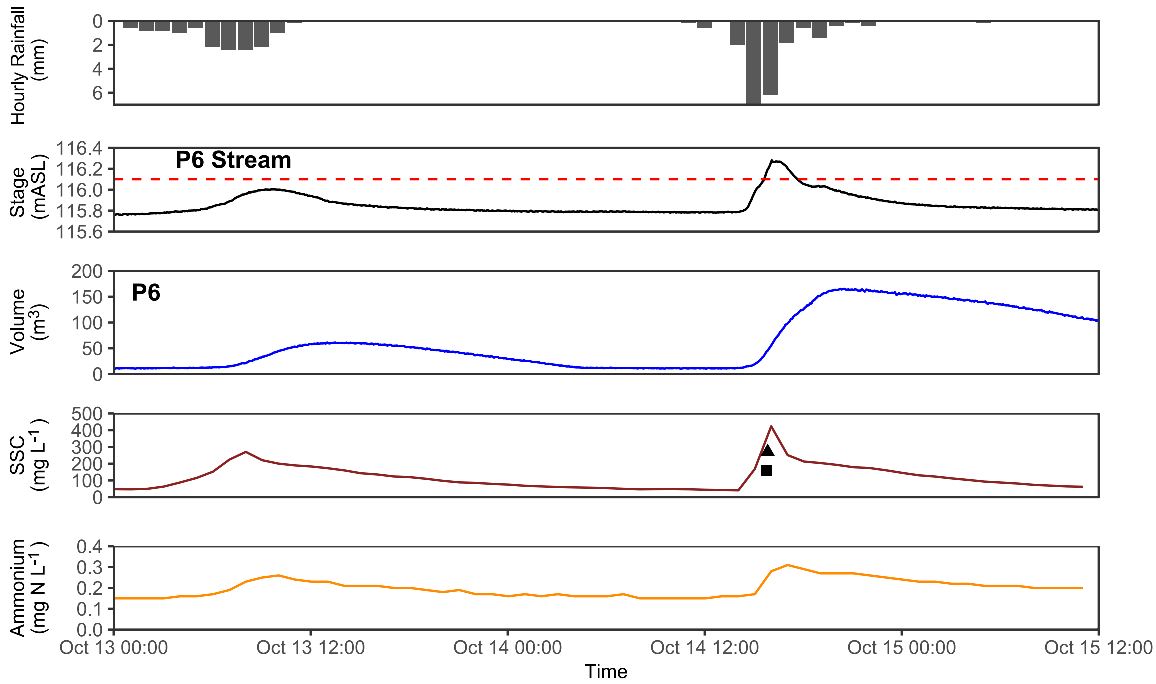

Figure 4.6 Time-series of hourly rainfall (mm), stream stage (mASL) at the P6 leaky barrier, water volume (m^3) in P6, stream SSC (mg L^{-1}) and ammonium concentration (mg N L^{-1}) during an event in October 2019. Square = surface run-off SSC at 15:45; triangle = surface run-off SSC at 15:50.

During the October 2019 event the filling rate of P6 was at its highest during the period in which the spillway was active, suggesting that the contribution of diverted streamflow likely exceeded that of surface run-off. Diverted streamflow was also likely to be a greater source of sediment, with run-off grab samples from adjacent fields having lower SSC than instream at the time of sampling. The relationship between stream stage and SSC shows clockwise hysteresis occurred during the smaller October 13th event, followed by a figure-of-eight pattern in the larger event (Figure 4.7). The highest SSC coincided with peak stage, however the peak in ammonium concentration occurred following peak stage, showing an anti-clockwise hysteresis loop. During the event on the 14th, it was estimated that a sediment load of 52 kg entered P6.

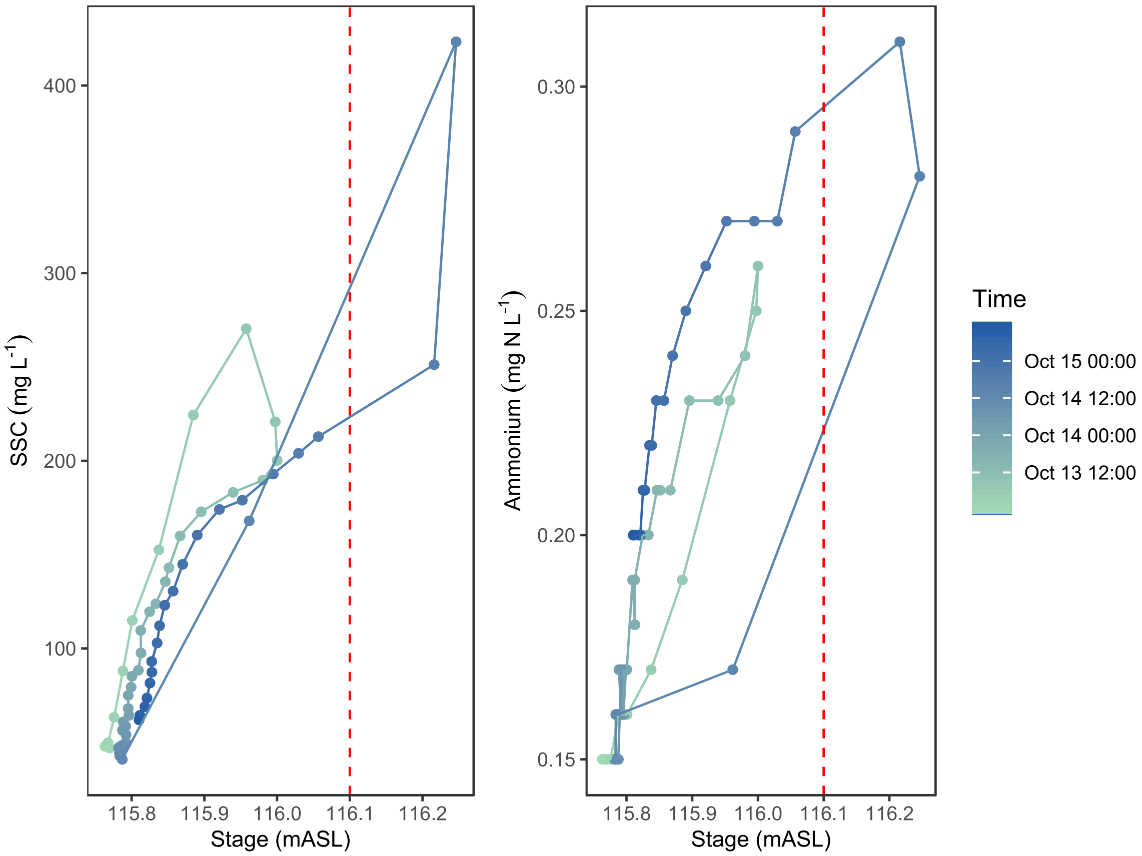

Figure 4.7 Stage-concentration relationships for SSC (mg L^{-1}) and ammonium (mg N L^{-1}) at the P6 leaky barrier during a storm event. Dashed red lines indicate the threshold of spillway activation.

4.4.3 Sediment Enrichment

Sediment deposited within features was found to be significantly enriched in TP (paired samples *t*-test, $p<0.01$, $n=14$), with an average concentration 1.5 times greater than the surface soil in contributing areas. The highest TP enrichment ratio of 2.66 (1.60-3.81) was observed for P1_OLP (see Appendix D (S3) for full table of ER values). On average the sediment was composed of 86 % silt and clay particles. Clay ER was typically higher for offline features, with a mean of 1.24 ± 0.32 compared to 0.76 ± 0.68 for online features. The opposite trend was seen for sand, with ER as low as 0.05 for offline features (P6) and up to 4.84 for online features (P10_US_OLP). Mean OC ER across all features was >1, however there was no apparent difference between the offline and online features.

A negative non-linear relationship was observed between median particle diameter (D_{50}) and OC content of soil and sediment (Appendix D (S4)). A significant correlation was seen for samples from arable (or recently reverted arable) fields (non-linear least-squares regression, $p<0.01$, $R^2=0.72$). Clay content was positively correlated with OC, though samples from permanent grassland with considerably higher organic content did not fit this relationship (linear regression, $p<0.01$, $R^2=0.13$). Sediment OC content was found to be negatively correlated with bulk density (linear regression, $p<0.01$, $R^2=0.33$; Appendix D (S5)). Particle size and clay content were both unable to explain differences in TP concentration ($p>0.05$).

4.4.4 Reductions in Storage Capacity

In the 2 to 3 years since construction, the majority of features did not lose significant volumes of their maximum storage capacity as a result of sediment loading (Appendix D (S6)). Average annual losses in storage capacity during the monitoring period ranged from 0.01 % in P4 and P8 up to 12.9 % in P0_OLP. In order to maintain their ability to fill and drain effectively during and after events, offline features require their outlets to remain sufficiently above the level of accumulated sediment, thereby helping to prevent siltation within drains. When considering the remaining storage capacity up to the drain height of features, the accumulated sediment volumes had a much greater impact. Potential further storage for sediment is most reduced in the online features, with P10_US_OLP, P10_OLP, and P0_OLP all predicted to fill beyond their outlet drain heights within 10 years (based on current accumulation rates). Whilst having a high accumulation rate, at a 10-year timescale P1_OLP is predicted to still retain >50 % of its storage capacity up to its outlet. P1_OLP with its deeper design had a mean water depth of 0.71 m during autumn and winter in contrast to only 0.3 m in P10_OLP with its shallow design and comparatively low outlet elevation. Interestingly, loss of storage capacity in P11 was negligible due to the sediment accumulation rate being too small to quantify even after over 2 years since construction. However, P11_OLP (connected to the outflow of P11) lost almost 5 % of its total storage within the same period.

4.4.5 Synthesis of Overall Functionality of NBS

On average, online features accumulated sediment more rapidly compared to offline features. Sediment trapped in online ponds was typically characterised by low bulk density and high TP concentration relative to the soil. In comparison, offline features selectively trapped a higher proportion of fine (clay) particles. Several factors including event contributing area and length-to-width ratio were found to partially explain differences in accumulation rates and the observed variation in trapped sediment properties. Instream leaky barriers enabled delivery of sediment-rich flows to offline features during storm peaks. Whilst this enhanced sediment accumulation, the observed rates were low enough to not compromise the flood storage capacity of offline features. However, online ponds with smaller capacities tended to accumulate sediment rapidly; this was enough to diminish their overall volume significantly within 3 years. The reasons underpinning observed differences are discussed in the following section along with their potential implications.

4.5 Discussion

4.5.1 Sediment and Nutrient Storage

It is evident that the construction of NBS (NFM features and online ponds) have cumulatively resulted in significant storage of sediment and nutrients in this small agricultural catchment within a timescale of 3 years. Despite having a total footprint covering <1 % of the catchment area, the features were able to accumulate 83 tonnes, the equivalent of almost a quarter of the total sediment yield leaving the sub-catchment. Within the monitoring period, both 2019 and 2020 experienced considerably wet conditions (25 % and 32 % above the rainfall average respectively). In this context, the observed sediment and nutrient storage is promising and suggests that the features are delivering multiple benefits despite hydrological extremes. Furthermore, the level of observed storage is significant when considering that sediment capture is a benefit secondary to the main flood risk mitigation function of the offline features. The results of this study demonstrate that numerous small-scale landscape modifications that alter and intercept floodwater pathways are capable of delivering beneficial outcomes for diffuse pollution alongside flood attenuation. Such benefits may also function to mitigate the need for channel maintenance of higher-order watercourses which has previously been used in conventional flood risk management approaches that aim to maintain conveyance downstream (CIWEM, 2014).

Research by Cooper *et al.* (2019) showed that a constructed wetland of a similar scale to the offline and online NBS features can trap over 7 tonnes sediment during its first year of operation, having an accumulation rate comparable to P7 when accounting for contributing area. Alternative NFM approaches such as beaver re-introduction have reported similar effect sizes in terms of their sediment and nutrient storage benefits (Puttock *et al.*, 2018). The creation of 13 beaver-engineered ponds in a small (0.2 km^2) enclosed headwater catchment resulted in the storage of ~100 tonnes sediment during a 3 to 5 year period. It is advantageous that similar magnitudes of mitigation can be achieved through contrasting NBS (i.e. those targeting flood risk reduction or water quality improvement) , thereby providing a more diverse toolkit for catchment managers to best suit interventions to different land-uses and settings. Table 4.3 demonstrates how sediment deposition rates within the NBS features are comparable to those observed in other catchments within similar pond features as well as naturally occurring deposition on floodplains. The measured accumulation rates were most comparable to those reported by Ockenden *et al.* (2014) for edge-of-field wetlands intercepting agricultural run-off on silt loam and clay soils (broadly similar to the soil texture within the Littlestock Brook catchment). However, our observed accumulation rates were relatively low compared to those on sandy soils in Cumbria. Putting the results in this context highlights the importance of soil properties and the erodibility of the surrounding landscape in determining how rapidly features will accumulate sediment. These comparisons indicate that NFM storage features are likely to have wide applicability across different catchment types within the temperate maritime climate of western Europe.

Table 4.3 Comparison of sediment storage rates observed in different features within catchments in England.

Type	**Catchment characteristics**	**Location**	**Catchment area (ha)**	**Mean storage ($t\ m^{-2}\ y^{-1}$)**	**Min/max storage ($t\ m^{-2}\ y^{-1}$)**	**Reference**
Offline NBS features	Lowland, arable/grass, silt loam/clay	Oxfordshire, England	0.4 – 20.1	0.006	0.001 – 0.01	This study
Online NBS features			0.6 – 41.0	0.02	0.009 – 0.04	
Constructed wetlands	Lowland, arable, clay	Leicestershire, England	4.0 – 10.0	0.005	0.0006 – 0.02	Ockenden *et al.* (2014)
	Upland, arable/grass, silt loam	Cumbria (Crake Trees Manor), England	10.0 – 50.0	0.04	0.002 – 0.1	
	Lowland, arable/grass, sand	Cumbria (Whinton Hill), England	1.5 – 30.0	0.4	-0.004 – 2.0	
Constructed wetlands	Lowland, arable (roadside), clay loam/sandy clay loam	Norfolk, England	23.8	0.06	0.06	Cooper *et al.* (2019)
Floodplain	Lowland, grass, various loams	Devon, England	27600.0	0.0005	NA	Lambert and Walling (1987)
Beaver dam ponds	Lowland, grass, clay loam/silt loam	Devon, England	20.0	0.02	0.009 – 0.05	Puttock *et al.* (2018)

4.5.2 Factors Influencing Accumulation Rates

Accumulation rates were highly variable between features, with these differences primarily being attributed to the size of contributing areas and the extent to which features were hydrologically connected. Many of the online features had considerably higher accumulation rates due to continually receiving streamflow and capturing suspended sediment from numerous sources. For example, resuspended channel bed sediment mobilised during small magnitude events will have been flushed into online ponds, whereas offline features would have received minimal sediment input under these circumstances due to insufficient surface run-off from contributing fields. Brainard and Fairchild (2012) also report significantly higher accumulation rates in ponds with inflows compared to those without. These results support the observations of Barber and Quinn (2012) who found that an online run-off attenuation feature accumulated a significant volume of silt throughout its first winter in operation. Additionally, during small events online ponds drain large areas with multiple fields whereas during similar events offline features typically only drain single fields. Despite a subset of the offline features receiving sediment-rich redirected streamflow during higher magnitude events, the relatively high elevation of spillways and raised design of the leaky barriers meant that this only occurred during short periods of high discharge. Whilst our study did not directly measure accumulations at an event scale, it is likely that the high magnitude storms contributed a significant proportion of the sediment in offline features. Palmer (2012) estimated that an offline run-off attenuation feature (with a contributing area comparable to P7) retained approximately 1 tonne of sediment during a single event. Our estimate of 52 kg sediment entering P6 during an event is considerably less, but is likely explained by factors including the differences in catchment characteristics (e.g. slope) and also the timing of the spillway activation during the event. This event was typical of the catchment response, with clockwise SSC-discharge hysteresis being the dominant pattern observed during the monitoring period (Robotham, 2022, *in preparation*). Clockwise hysteresis means that SSC peaks prior to discharge, potentially resulting in misalignments of spillway activations and the periods of highest suspended sediment loads, thereby delivering less to features. In slowly permeable catchments, clockwise hysteresis has been associated with in-channel sediment sources where there is enhanced deposition during baseflows and a readily available sediment supply (Lloyd *et al.*, 2016b; Sherriff *et al.*, 2016). This suggests that leaky barriers may play a role in modifying sediment dynamics during events as a result of their ability to increase in-channel storage. Reducing the porosity of barriers is likely to produce a greater backwater effect, forcing more water out of the channel and thereby increase rates of accumulation in offline features (Muhawenimana *et al.*, 2021). It is also likely that lowering the threshold required for overbank flow into the spillways would coarsen the grain size distribution of accumulated sediment, given typical vertical profiles of suspended sediment transport where coarse particle load increases with proximity to the streambed (Lupker *et al.*, 2011; Lamb *et al.*, 2020). The generally high threshold required for overbank flow into the features in this study explains, at least in part, the relatively low accumulation rates within offline features.

River restoration techniques such as reconnecting streams to their floodplains can enhance sediment deposition through increasing the frequency of overbank flows. Millington (2007) shows how floodplain deposition is closely related to overbank suspended sediment load at a restored site with instream wood jams in the New Forest, Hampshire. Following restoration, sediment deposition of 26.3 kg m^{-2} was observed upstream of wood jams over a flood season, which compares to the annual mean accumulation rate of 6.3 kg m^{-2} in the offline NBS features. This suggests that restoration techniques may be better suited to emulate natural depositional processes than NFM approaches. However in an arable context, floodplain restoration is generally considered incompatible with the land-use, and so spatially targeted NBS (online and offline features) offer a good compromise for landowners and catchment managers.

Several features stood out as outliers in terms of their accumulation rate. P10_US_OLP had an exceptionally high rate compared to other online features because it is the most upstream feature in a series of three connected features. Robotham *et al.* (2021) showed that P10_US_OLP was generally the most effective at trapping sediment. Therefore, where NBS interventions are placed in series, features that are located farthest upstream are likely to require more frequent maintenance. P3 is also somewhat of an outlier in terms of its contributing area, with its higher accumulation potentially owing to the contribution of sediment from subsurface drainage via broken field drains. During monitored events, it was observed that P3 typically reached peak water storage later than other features, inferring that filling was not driven by rapid rainfall run-off, but via the subsurface. Studies have shown that field drains can act as pathways for sediment, particularly in fine-grained soils (Stone and Krishnappan, 2002; Coelho *et al.*, 2010). NBS in highly modified agricultural landscapes hold potential for mitigating diffuse pollution from subsurface pathways if located appropriately.

The length-to-width ratio of features was also shown to play a role in sediment accumulation, with performance of features improving with higher ratios, as also found by Persson and Wittgren (2003). Similarly, a moderate positive correlation between this ratio and both particle and P retention was found by Johannesson *et al.* (2015) in constructed wetlands in southern Sweden. Increasing length-to-width ratios gives the influent a greater residence time and opportunity to settle out fine matter (e.g. clay particles) (Fifield, 2011). Most of the offline features were constructed in field corners thereby only taking small proportions of the fields out of agricultural production, however this also restricted their length-to-width ratios (0.2-0.8) compared to online features (1.1-2.0) with the result that they trapped less sediment.

Further discussion of factors influencing accumulation rates is given in the Supporting Information (Appendix D (S7)).

4.5.3 Sediment Enrichment

In the context of this study, enrichment ratios are influenced by soil erosion processes and by the trapping efficiency of features. As expected, P enrichment was observed in the majority of features, with the three highest ER occurring in online ponds. Features with greater residence times or increased hydraulic roughness from vegetation are able to more effectively settle out finer particles with larger surface areas and typically higher P content (Vargas-Luna *et al.*, 2015; River and Richardson, 2018). Evidence of this can be seen in 75 % of the offline features which were more enriched in clay. The permanently flowing online features showed the opposite, with a considerably higher sand ER as a result of the transition from high to low velocity upon entering the ponded area causing rapid deposition of large particles. Coarser sediment is therefore typically found closer to the inflow of features (Ockenden *et al.*, 2014; Cooper *et al.*, 2019; Robotham *et al.*, 2021).

Particle size was found to significantly influence OC concentrations, with higher organic content being associated with greater proportions of finer sediment particles. This relationship has been observed in similar field wetlands and ponds with the larger surface area of finer particles allowing greater potential for binding of organic matter (Ockenden *et al.*, 2014; Cooper *et al.*, 2019). The generally higher OC and lower bulk density of sediment in online features may be in part explained by a more extensive cover of wetland vegetation leading to greater carbon inputs into the waterlogged hypoxic or anoxic sediment (Were *et al.*, 2020). A study of sediment in small natural ponds in Northumberland found a similar pattern, with the highest OC (up to 15 %) in permanent, vegetated ponds, and the lowest in temporary ponds with little vegetation (Gilbert *et al.*, 2014). Over the 2 to 3 years since construction, the permanently wet features have been colonised by emergent wetland vegetation (e.g. *Typha* sp.) which is likely to enhance their overall trapping efficiency. Braskerud (2001) found that vegetation aids sediment retention by mitigating resuspension of trapped material, reducing it to negligible levels after 5 years. In future, sediment in NBS features that continue to develop wetland vegetation may become more enriched in finer particles due to greater stabilisation and a positive feedback effect (Corenblit *et al.*, 2009). Increased trapping may pose management implications if the effect of vegetation enhances the rate of accumulation to a point where flood storage capacity is significantly compromised.

However, in terms of biodiversity this feedback and natural succession may be more beneficial, whereas undergoing regular disturbance to remove sediment and maintain storage capacity will result in a plagioclimax community.

The sediment trapped within features was generally more enriched in OC compared to the arable soils in the catchment which on average contained an OC content of 0.8 % less. A similar range of OC ER have been observed in simulated rainfall erosion experiments which showed evidence of selective transportation of OC via finer particles (Schiettecatte *et al.*, 2008; Nie *et al.*, 2015). This sediment has potential viability for spreading back onto surrounding fields to boost soil organic matter, which is an important property for sustaining healthy soil biology and can improve crop yield (Whitmore *et al.*, 2017). The accumulated sediment, primarily composed of silt and clay, also shows potential for nutrient reclamation with an average TP concentration of 1424 mg P kg^{-1}, which is 438 mg P kg^{-1} higher than the arable soils. The recovery and recycling of P is becoming increasingly important for the future sustainability of food production and could help reduce fertiliser costs for farmers and nutrient losses to waterbodies (Tonini *et al.*, 2019).

4.5.4 Considerations for NBS Management and Design

One of the concerns that has been raised in literature discussing NFM approaches is the issue of flood storage capacity being consumed as a result of sedimentation (Lane, 2017). The evidence from this study suggests that in the short-term, sedimentation does not pose a major threat to the ability of the offline features to function as NFM interventions. The total sediment accumulations equated to a reduction of 110.8 m^3 storage capacity across all the monitored features (<1 % of total storage lost over 3 years). This leaves almost 15000 m^3 available for potential flood storage in both sub-catchments, which drain a combined area of 6.8 km^2. Due to their smaller volume and more rapid accumulation, online ponds require more frequent maintenance to remove stored sediment. However, this is to be expected from the online ponds as they were constructed primarily to address diffuse P pollution. The results suggest that such features should undergo desilting on a biennial basis to reduce the potential risk of blocking outflow drains and the remobilisation and flushing of sediment downstream (Wilkinson *et al.*, 2010). Without appropriate management, there is a risk that online features may act as a source of sediment and pollutants (Barber and Quinn, 2012). However, evidence shows that overall such features are still net sediment sinks despite their potential to act as temporary sources during large events (Robotham *et al.*, 2021). Whilst more frequent desilting makes online features more expensive to maintain, they have potential for high natural capital value through their provision of semi-permanent wetland habitat. In contrast, offline features are only likely to require maintenance in the medium-term after ~10-years of operation, with sediment removal being more easily achieved during summer when features are dry or at their shallowest. The management of such offline NFM features should aim to strike a balance between their primary purpose of flood mitigation and their additional benefits for biodiversity and water quality. If appropriately maintained, both online and offline NBS have the potential to become long-term anthropogenic landforms of sustainable agricultural landscapes.

The design and configuration of online and offline features can play an important role in their ability to effectively intercept and store water and eroded matter. However, current best practice guidance for NFM does not typically consider optimising intervention design for increasing the removal efficiency of diffuse pollutants (Forbes *et al.*, 2016; Highways England, 2021; Wren *et al.*, 2022). Our findings show that despite having length-to-width ratios below the recommended 5:1 ratio for optimal trapping efficiency, the features still accumulated significant masses of pollutants. Therefore it can be said that diffuse pollution mitigation is still possible with interventions optimised for flood storage opposed to pollutant removal efficiency. If a feature is found to act as a pollutant source, the issue could be remediated retroactively by introducing greater hydraulic complexity e.g. adding berms or vegetation zones perpendicular to the direction of flow to enhance trapping (Persson and Wittgren, 2003).

Another consideration for the implementation and management of NBS features is the potential risk of harmful algal blooms forming in the nutrient-enriched stagnant water. A cyanobacterial bloom was observed in P5 during June 2021, but posed minimal threat due to its location away from livestock and routes of public access. Future climate change may increase the occurrence and intensity of blooms such as this due to the effects of warming water temperatures on algal abundance (Richardson *et al.*, 2019). Consequently there is a rationale for allowing marginal trees to develop, providing shade to mitigate against extreme heat and the potential for such disbenefits to occur (Kail *et al.*, 2021). Trees may also help to mitigate the potential disbenefit of enhanced greenhouse gas emissions from temporary ponds which have been observed as a result of sediment drying-rewetting cycles (Obrador *et al.*, 2018; Paranaíba *et al.*, 2020). Our results indicate that features are significant sinks for POC, however the extent to which this carbon remains in situ is not yet fully understood. The NFM evidence-base would benefit from further empirical research into the impact of such features on biogeochemical cycles to better understand their environmental trade-offs and potential implications for pollution swapping.

4.5.5 Opportunities for Further Study

This study used a pragmatic approach to estimate the ability of NBS features to store sediment and nutrients and puts this into the context of yields estimated from high-resolution monitoring at the catchment outlet. The surveying method used to obtain sediment accumulations was based on transects and therefore provided an estimate of sediment depths. Full bathymetric surveying covering the entire footprint of features would reduce the uncertainties of these estimates, particularly in permanently ponded features. This would overcome the need for sampling and spatial interpolation and its associated issues (Li and Heap, 2011). Additionally, a greater density of core samples and analysis of sub-samples along vertical sediment profiles would also improve estimates and allow greater insight into changes in composition and accumulation rates over time. Monitoring within a continually changing farmed landscape has inherent challenges, particularly whilst the catchment was subject to disturbance from the phased construction of interventions and changes in cropping over the three monitored years. Future studies would benefit from surveying NBS features over a longer time period, starting immediately after construction and then taking repeat measurements over multiple years following a period of acclimatisation. This would better capture changes in NBS features' responses to hydrometeorological extremes, land-use and management change, and ecological development, as well as the effect of any maintenance activity. This study characterises NBS functionality within the specific context of a lowland arable catchment, but there is still a need to develop further understanding of such features in a wider range of landscape contexts.

4.6 Conclusions

Online and offline NBS are net stores of sediment that are capable of accumulating significant masses of sediment and nutrients, helping to mitigate fluvial soil loss and diffuse pollution from agricultural land, whilst also creating new wetland habitat. The features within this study occupied a total surface area <1 % of the catchment, yet trapped the equivalent of 15 % of the estimated catchment sediment yield over 3 years without compromising high value arable land and farm productivity. This enhanced sediment storage also accounted for the equivalent of up to 14 %, 10 %, and 8 % of the fine suspended sediment, TP, and POC catchment yields respectively. The majority of the monitoring period experienced above average rainfall, with 2019 and 2020 receiving 25 % and 32 % above the annual average respectively. This enabled the functioning of features to be tested under notably wet conditions that posed a higher soil erosion and diffuse pollution risk. The magnitude of sediment and nutrient mitigation observed is therefore promising in light of this context.

The design of NBS is important in optimising their potential for both flood storage and water quality improvement. Results suggest that rates of sediment and nutrient accumulation are largely explained by differences in the hydrological connectivity and drainage areas of features. Online pond features showed higher accumulation rates, but the activation of leaky barriers and spillways augmented accumulation in offline features. Based on these differences, maintenance requirements are more frequent for online features, whereas accumulation in offline features only necessitates sediment removal in the medium-term to prevent reducing effective flood storage capacity. The enriched sediment stored within features shows potential nutrient reclamation benefits for farmers through redistributing on fields as a soil conditioner.

These findings provide valuable insight into the delivery of diffuse pollution mitigation by NBS in a small lowland catchment of the Upper Thames, albeit a snapshot over a relatively short period of their intended lifetime. An extended monitoring record covering a range of interannual hydrological conditions and extremes is needed to better understand the long-term impact of NBS, their multiple benefits, trade-offs, and roles within farm businesses. Interventions such as offline storage and leaky barriers are able to deliver benefits for sediment and nutrient storage beyond their primary aim of managing flood risk. NBS show good potential for use in integrated catchment management and should be incorporated into future environmental land management schemes in order to deliver their benefits more widely.

4.7 Author Contributions

Conceptualization: John Robotham, Gareth Old, David Sear, Ponnambalam Rameshwaran, David Gasca-Tucker. Funding acquisition: Gareth Old, David Sear, Ponnambalam Rameshwaran. Methodology: John Robotham, Gareth Old, David Sear, Ponnambalam Rameshwaran, David Gasca-Tucker. Investigation: John Robotham, James Bishop, Emily Trill. Resources: David Gasca-Tucker, Joanne Old, David McKnight, James Bishop. Supervision: Gareth Old, David Sear, Ponnambalam Rameshwaran, David Gasca-Tucker. Writing – initial draft: John Robotham. Writing – reviewing and editing: John Robotham, Gareth Old, David Sear, Ponnambalam Rameshwaran, Emily Trill, James Bishop, David Gasca-Tucker, Joanne Old, David McKnight. All authors have read and agreed to the published version of the manuscript (Appendix C).

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Chapter 5 Monitoring the Effects of Natural Flood Management on Water Quality in an Agricultural Lowland Catchment

5.1 Abstract

Natural Flood Management (NFM) schemes comprise a series of interventions implemented to store water and slow its movement through a catchment to help attenuate flooding downstream. It is thought that such interventions may also provide multiple benefits (ecosystem services) including water quality improvement through the mitigation of diffuse pollution. This study used multiyear continuous high-resolution stream monitoring data to analyse the impact of NFM interventions on suspended sediment (SS) and total phosphorus (TP) concentrations, loads, and their dynamics in two small (3.4 km^2) adjacent rural sub-catchments using a Before-After Control-Impact (BACI) approach. In the Before-NFM period, the annual loads of SS and TP from the Impact sub-catchment were higher than from the Control, whereas the reverse was true in the After-NFM period. Monitoring data also revealed notable differences in the hydrological responses of these sub-catchments as well as the rainfall totals for the Before and After periods. To minimise the effects of these potentially confounding differences in the BACI analysis, various *estimated run-off* indices were derived based on the hydrology of soil types (HOST), impermeable bedrock geology, slowly permeable soils, and average slope for inclusion in Generalised Linear Models (GLMs). GLMs using data from 148 storm events between 2017 and 2021 (occurring at varying stages of NFM scheme implementation) found mean rainfall intensity and total rainfall/estimated run-off to be significant predictors of peak suspended sediment concentration (SSC) and total suspended sediment load, respectively. When modelling SSC, a significant interaction effect was observed between the final phase (4) of NFM implementation and the Control-Impact sub-catchment. Further analyses suggested that this effect was in fact a result of intense rainfall combined with contrasting sub-catchment land cover notably elevating SSC in the Impact sub-catchment within this period. Results indicate that any effect of NFM interventions and their added storage within the sub-catchment was masked by additional sources and inputs of sediment caused by hydrometeorological variability and changing land cover. Hysteresis behaviour indicated that the likely sources of suspended sediment differed between sub-catchments and storm events and were controlled by a combination of factors including land-use,

lack of crop cover, and antecedent conditions. Difficulties in detecting NFM effects were exacerbated by a dry and relatively short pre-intervention period followed by a wet post-intervention period. This study highlights the importance of establishing robust pre-intervention data for catchment monitoring to improve hydrological BACI analyses. We recommend the implementation of multi-scale monitoring for evaluating the effectiveness of diffuse pollution mitigation measures in order to best detect and understand their impact against the noise of other environmental change within catchments.

5.2 Introduction

Natural Flood Management (NFM) is a catchment-based approach that aims to slow, filter, and store the flow of water through a catchment using a variety of low-cost modifications to the landscape that enhance these hydrological processes, thereby reducing flood risk downstream (Lane, 2017). It is also widely thought that through modifying the flows of water through a landscape, NFM can provide a number of environmental, ecological, and cultural co-benefits (Wingfield *et al.*, 2019). These natural capital benefits include aspects such as water quality improvement, carbon sequestration, habitat provision, and aesthetic enhancement (McLean, Beevers, Pender, Haynes and M. I. Wilkinson, 2013; Iacob *et al.*, 2014). With NFM typically being carried out in rural settings, often on farmland, many existing NFM schemes use interventions that are also targeted at mitigating the effects of diffuse agricultural pollution. Common agricultural pollutants of streams include fine sediment, nutrients, and pesticides, all of which NFM has the potential to mitigate through attenuating run-off and preventing their delivery to watercourses. Run-off attenuation and storage features can be used to trap particulate phosphorus (PP), particularly in catchments with heavy (clay-rich) soils, where this is typically the dominant P fraction due to the increased probability of surface run-off generation and soil erosion in these areas (Lloyd *et al.*, 2019).

Drivers such as climate change are putting increasing pressure on freshwater systems, with a key mechanism being the increasing occurrence of extreme hydrometeorological conditions (i.e. floods and droughts) (Jeffries *et al.*, 2013; Puczko and Jekatierynczuk-Rudczyk, 2020). Evidence suggests that wetter winters as a result of climate change in the UK will lead to increased losses of nitrate and sediment from grassland and even more so from arable land (Zhang *et al.*, 2022). Nutrient enrichment increases the risk of eutrophication which has significant ecological consequences as well as high costs in monetary terms (Pretty *et al.*, 2003). In the UK, agriculture has been the largest source of nitrogen (N) to freshwater over the past two centuries whereas sewage treatment works are the dominant P source (Bell *et al.*, 2021). However, Worrall *et al.* (2016) note that currently ~77 % of sewage sludge is applied to agricultural land, thereby providing further potential for nutrients entry into watercourses through run-off. The need for mitigation efforts on agricultural land is therefore high.

Initiatives such as *Smarter Water Catchments*, trialled by Thames Water within the Evenlode catchment as part of their asset management plan period for 2015-2020 provide potential new funding streams for many of the catchment-based interventions that NFM encompasses (Thames Water, 2020). The departure of the UK from the European Union (EU) also provides potential funding mechanisms for these interventions through the Environmental Land Management (ELM) schemes which by 2025 will replace the current schemes available through the EU's Common Agricultural Policy (CAP). Water quality and flood mitigation are increasingly being seen as public goods with the landowners and farmers who actively deliver these societal benefits being compensated accordingly, as opposed to a system where subsidies are determined by landholding size (Cusworth and Dodsworth, 2021).

The Demonstration Test Catchments (DTC) programme trialled and investigated a variety of management measures and nature-based solutions (NBS) aimed at mitigating diffuse agricultural pollution in six operational catchments across England (DEFRA, 2020). These study catchments cover the majority of broad catchment typologies in England and Wales; however the least well-represented typology is the '*Midlands and South Coast'* catchment group. This means the evidence gathered by the DTC programme on the effectiveness of mitigation measures is less applicable to these lowland catchments in central and southern England of which there are 66, many of which are located within the Thames River Basin and still suffer from diffuse pollution issues. Whelan *et al.* (2022) conclude that since 1940 nutrient pollution has increased in rural river catchments in Britain, largely as a consequence of intensive agriculture. To better understand the potential benefits of NBS and their effect on issues such as fine sediment loading in these types of catchments, further evidence on their efficacy is required.

Evidence on the effectiveness of NBS for reducing sediment loading tends to be limited to field-scale studies focussing on single or clustered interventions (Cooper *et al.*, 2019; Brunet *et al.*, 2021), also typically over short timescales (Yuan *et al.*, 2009), or only using event-based monitoring (Barber and Quinn, 2012). The ability to monitor multiple determinands across multiple interventions, sub-catchments, and across interannual timescales is constrained largely by financial resources and limited capacity to start monitoring programmes far enough in advance of intervention implementation to gather sufficient data for robust evaluation of their effectiveness. Continued advancements in sensor technology have helped enable field data to be collected at higher-resolution with reduced costs. Digital turbidity sensors allow us to measure stream turbidity in situ continuously across a monitoring period and use this as a surrogate for suspended sediment concentration to provide insight into catchment soil erosion dynamics (Jordan and Cassidy, 2022).

To the best of our knowledge, one of the most-intensively monitored examples of NFM in the UK is the Littlestock Brook NFM scheme, situated in the Evenlode catchment in the upper part of the Thames River Basin (Old *et al.*, 2019; Trill, Robotham, Bishop, *et al.*, 2022). This saw the implementation of interventions such as storage ponds, instream leaky woody barriers, tree planting, and conservation agriculture (e.g. no-till). The scheme was developed to not only help alleviate flooding in the villages downstream, but also to address some of the water quality issues faced by the catchment. The most recent (2019) assessment of the Littlestock Brook's Water Framework Directive (WFD) status resulted in an overall classification of '*poor*' due to the phosphate, and macrophytes and phytobenthos categories. The underlying causes of this poor ecological status have been attributed to both sewage discharge and agricultural land management. Investments to improve wastewater treatment performance and capacity in rural communities are often deemed disproportionately expensive relative to their benefits. Therefore, in small rural catchments such as this, low-cost mitigation measures are required in order to attempt to reverse declines in water quality and ecological status. The NFM scheme presented an opportunity for integrated catchment management addressing multiple issues and testing the effectiveness of a suite of interventions for both flood risk and diffuse pollution mitigation.

Earlier work in the Littlestock Brook catchment demonstrates the ability of a variety of water storage features to intercept key pathways of diffuse pollution and thereby store significant masses of both sediment and nutrients (Robotham *et al.*, 2021; Robotham *et al.*, 2023). This study aims to investigate the combined effect of the NFM scheme on the receiving watercourse and determine if a Before-After Control-Impact (BACI) approach is able to statistically detect changes to the catchment's suspended sediment (SS) dynamics and hydrological response.

The success of mitigation strategies can prove a challenge to determine due to the difficulties in detecting their effects on catchment responses against background environmental variation (Lloyd *et al.*, 2014). This is further compounded by nonstationarity, with an increasing frequency of extreme weather events and greater variance in their magnitude making 'typical' representative baseline conditions hard to establish (Slater *et al.*, 2021). This is particularly the case where baseline monitoring is constrained to a short timeframe due to the logistical and practical aspects of setting up a monitoring programme and implementing catchment initiatives. A robust monitoring design to test the impact of measures on the environment requires considerable amounts of data, with one of the most optimal study designs being the Before-After Control-Impact (BACI) approach (Smokorowski and Randall, 2017; Thiault *et al.*, 2017). Such study designs have been employed to assess catchment restoration efforts, but are not without their challenges (Spray *et al.*, 2022). Catchment projects typically have considerably limited time and resources to be able to thoroughly monitor highly temporally and spatially variable systems, making it important to maximise the value for effort from data collection. Another challenge arises from the fact that the implementation of interventions may not always occur within one discrete period, but instead projects may adopt a phased approach over multiple years due to funding or planning constraints. To the best of our knowledge this paper is the first catchment monitoring study to evaluate the effects of NFM interventions using a multiple-phase BACI approach.

Within this paper we identify and characterise the main sources and pathways of sediment and nutrients within two small (3.4 km^2) neighbouring sub-catchments with broadly similar land-uses. We aim to address the following questions:

1. What are the key drivers of suspended sediment (and associated nutrient) delivery and transport within these sub-catchments?
2. To what extent is it possible to detect the effects of NFM on suspended sediment concentrations and loads at a sub-catchment scale using a statistical BACI approach and multiyear monitoring data?
3. What is the potential of NFM for mitigating elevated dissolved nutrient concentrations associated with chronic and legacy agricultural pollution?

5.3 Materials and Methods

5.3.1 Study Site

The Littlestock Brook NFM study site is located in the Upper Thames (Southern England), in the largely rural lowland headwaters of the Evenlode catchment. The catchment has a mean altitude of 142 mASL and standard annual average rainfall of 691 mm for the period 1961-1990 (UKCEH, 2022a). The majority of the Littlestock Brook NFM scheme is concentrated on the area of land upstream of the village of Milton-under-Wychwood, the community at risk from flooding. It is here that two main tributaries of the Littlestock Brook meet. These two tributaries both drain areas of approximately 3.4 km^2, and within this study the sub-catchment to the north will also be referred to as the *Control*, and the sub-catchment in the south as the *Impact* (Figure 5.1).

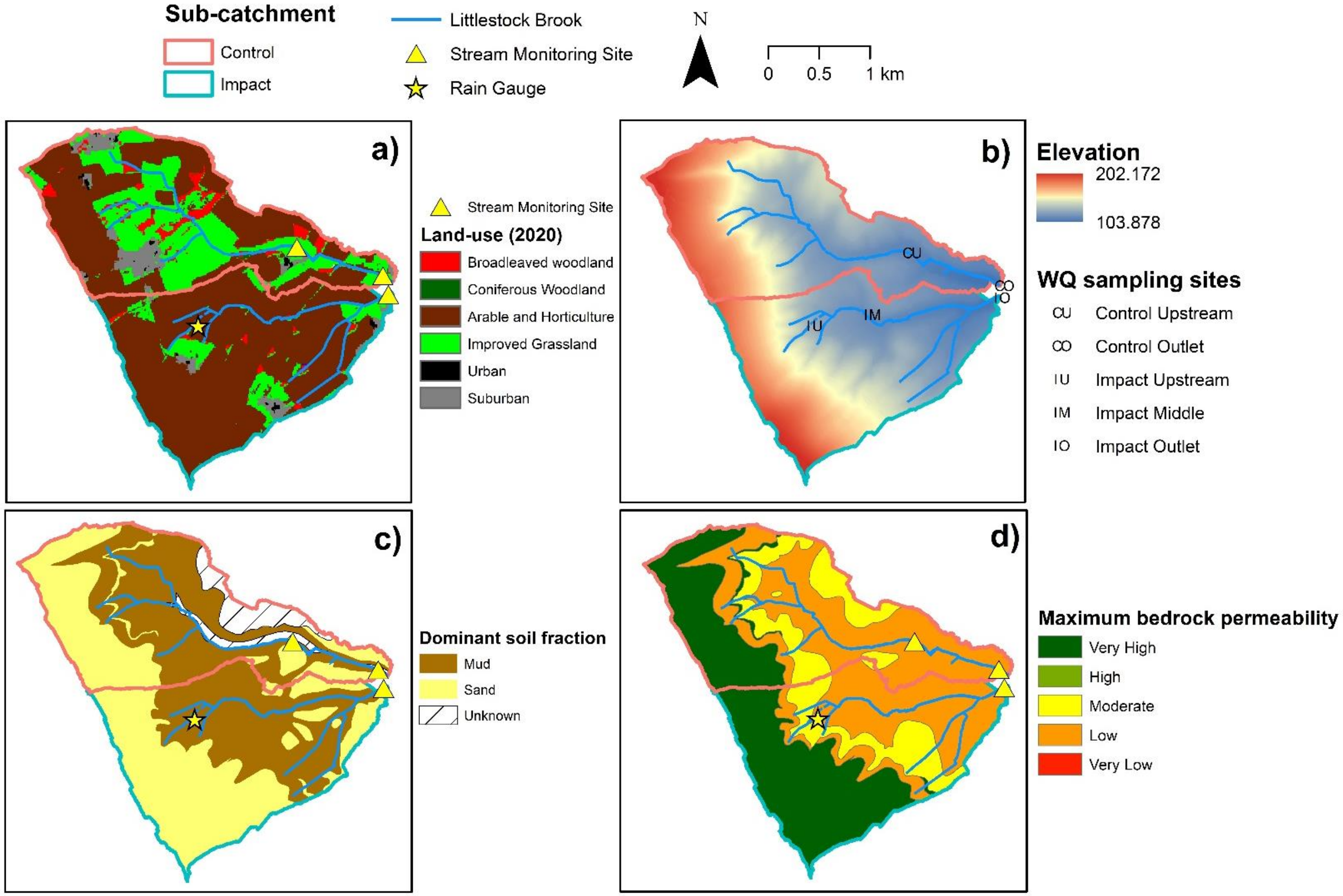

Figure 5.1 The Control and Impact study sub-catchments showing **a)** land-use; **b)** elevation (m) and water quality sampling sites; **c)** dominant soil fraction; and **d)** maximum bedrock permeability. Land-use data was obtained from the UKCEH LCM 2020; elevation data from the Environment Agency National LIDAR Programme; soil fraction data from the BGS Soil Parent Material Model Dataset; and bedrock permeability data from the BGS Permeability Dataset.

Table 5.1 provides several hydrological metrics and catchment properties that summarise the differences in flow regime between the study sub-catchments. Baseflow index (BFI) and Richard-Baker flashiness index (RBI) were computed for each sub-catchment from daily mean flows recorded at the outlet stream monitoring sites between 2017 and 2021. Standard Percentage Runoff based on the Hydrology of Soil Types (SPR_{HOST}) were obtained using the Flood Estimation Handbook methodology (Bayliss, 1999).

Table 5.1 Hydrological, geological and soil properties of each sub-catchment. Flow statistics were computed from daily mean flows.

Catchment property		Sub-catchment	
		Control	**Impact**
Flow (L s^{-1})	Q_{mean}	34.9	64.7
	Q_{50}	8.2	56.5
	Q_{10}	97.5	118.6
	Q_1	279.7	232.9
BFI		0.33	0.75
BFI_{HOST}		0.43	0.52
SPR_{HOST} (%)		39.17	34.25
RBI		0.41	0.16
Stream drainage density (m km^{-2})		2150.1	2145.8
Average slope (%)		6.9	5.8
Bedrock geology (%)	Limestone	30.0	45.0
	Mudstone	49.6	38.8
	Siltstone & mudstone (interbedded)	20.4	16.2
Soil type (%)	Well-drained, shallow, lime-rich	36.8	50.5
	Slightly acid, loamy and clayey, slightly impeded drainage	6.6	14.3
	Slowly permeable, seasonally wet, acid or slightly acid & base rich, loamy and clayey, impeded drainage	56.5	35.2

The upper part of each sub-catchment is characterised by shallow lime-rich soil underlain by Cotswold limestone that transitions into a slowly permeable, seasonally wet soil with a loamy and clayey texture. The lower part of the Control sub-catchment is dominated by heavy soil with poor drainage that also covers the Impact sub-catchment to a lesser extent. Small areas in the middle reaches of the sub-catchments are characterised by an intermediate soil with slightly impeded drainage that is also similar in texture.

The Littlestock Brook NFM scheme consists of a variety of catchment-based interventions, namely 14 bunded offline storage areas and on-line ponds, and multiple series of instream leaky woody dams. Arable reversion was carried out across 14.4 ha of woodland planting with a mix of native trees (e.g. *Quercus*, *Alnus* and *Salix* sp.). The scheme was implemented in five phases from 2018 to 2021 (Table 5.2). Additionally, several field-edge sediment/nutrient traps were constructed across the sub-catchments towards the end of the monitored period during winter 2020/2021.

Table 5.2 Timeline of the phased installation of NFM interventions and the potential cumulative storage volumes (m^3) added to the Impact and Control sub-catchments. NB: Phase 1 interventions were not part of the official NFM scheme delivery.

Phase	**Implementation**	**Impact sub-catchment**		**Control sub-catchment**	
		Interventions	**Cumulative storage (m^3)**	**Interventions**	**Cumulative storage (m^3)**
0	January 2017	None (monitoring commenced)	0	None (monitoring commenced)	0
1	March 2017	None	0	Woody check dams (for bedload transport control)	0
2	February 2018	Leaky woody dams; field corner bunds and offline storage areas; woodland planting; on-line ponds	11500	Woodland planting, offline storage area	140
3	February 2019	Field corner bunds and offline storage areas; on-line ponds	14700	Field corner bund and offline storage area; leaky woody dam and swale; on-line pond	2020
4	Sept/Oct 2020	None	14700	Field corner bunds and offline storage areas	8420
5	Winter 2020/21	Sediment/nutrient traps	14700	Sediment/nutrient traps and ponds	8420

5.3.2 Study Design and Monitoring

The monitoring of sub-catchments was set up to enable the BACI study design to be used to test the effects of the NFM interventions. Three stream monitoring sites were established in December 2016 and January 2017, just over a year prior to the start of construction on the NFM scheme. During Phase 2-3, the Control sub-catchment was subject to the installation of a comparably small set of NFM interventions (and then two larger storage areas in Phase 4), thereby changing the study design into a partial BACI setup due to the loss of a '*true control*'.

The stream monitoring sites were each equipped with a water level sensor (Level TROLL, In-Situ; Redditch, UK) and a turbidity sensor (DTS-12, FTS; Victoria, Canada), both of which measured and logged data at a 5-minute resolution. For each site, turbidity readings were calibrated against suspended sediment concentration (SSC) samples taken using US DH-48 sampler and automatic samplers (Sigma SD900, Hach; Loveland, CO, USA). Flow velocity measurements at each monitoring site were taken under a range of flow conditions using an electromagnetic current meter (Valeport) and discharge (Q) was calculated using the velocity-area method. During low flow conditions where the current meter was unsuitable, Q was measured using electrical conductivity sensors (YSI/EXO1) following the salt dilution method. Stage-discharge rating curves were constructed for each monitoring site to develop timeseries of stream discharge at 5-minute intervals. Further details on flow measurement and SSC sampling methodology and quality control procedures are described by Robotham *et al.*, 2022 (Appendix E). Fluxes were calculated using Equation 5.1:

$$Flux = \int_{t_1}^{t_2} Q(t)C(t)dt \qquad (5.1)$$

where Q = stream discharge; C = concentration of SS/TP, and t = time.

Water chemistry samples were taken regularly at stream monitoring sites (Figure 5.1) between March 2019 and 2020 (ending prematurely due to restrictions imposed as a result of the Covid-19 pandemic) and analysed for major ions and P fractions (see Robotham *et al.* (2021) for further details of sampling, and Bowes *et al.* (2018) for analytical methodology). Concentration (C) data were used to examine Q-C relationships to characterise nutrient inputs within the sub-catchments. Land-use and cropping data for each year in the study period were obtained from the Rural Payments Agency's crop map of England (CROME) dataset (Rural Payments Agency, 2021) in order to gauge the extent of land-use change over time in each sub-catchment.

5.3.3 Data Processing

For the BACI analysis, a dataset of storm events was compiled by extracting variables from the Q and SSC timeseries at the sub-catchment outlet monitoring sites along with combined rainfall data from our tipping bucket rain gauge and the hourly Met Office weather station at Little Rissington, ~3 km from Milton-under-Wychwood (Met Office, 2006). Estimated run-off was initially calculated using SPR_{HOST} by multiplying rainfall totals by the SPR_{HOST} value for each sub-catchment. Similarly, estimated run-off indices based on impermeable bedrock geology, slowly permeable soils, and slope were calculated. Estimated run-off indices using all possible combinations of the geology/soil/slope coefficients were also derived. For the purposes of this study, events were defined as occurring when a rise in Q of greater than three times the initial value was observed alongside a continuous period of rainfall. Events were split where marked double peaks in the hydrograph and hyetograph were observed. Antecedent precipitation index (API) was calculated as a proxy for soil moisture using Equation 5.2:

$$API_d = k \times API_{d-1} + P_d \tag{5.2}$$

Where API_d is the API for day, d; k is a decay factor and P_d is rainfall for day d. The decay factor was set to 0.95 following the method of Hill *et al.* (2015).

A hysteresis index (HI_{mid}) was used to quantify the direction and magnitude of hysteresis observed in Q-C relationships during storm events (Lawler *et al.*, 2006). For clockwise hysteresis, HI_{mid} was calculated using Equation 5.3:

$$HI_{mid} = \left(\frac{C_{RL}}{C_{FL}}\right) - 1 \tag{5.3}$$

And for anti-clockwise hysteresis, HI_{mid} was calculated using Equation 5.4:

$$HI_{mid} = \left(\frac{-1}{\left(\frac{C_{RL}}{C_{FL}}\right)}\right) + 1 \tag{5.4}$$

Where: C_{RL} and C_{FL} are the concentration values at Q_{mid} on the rising and falling limbs respectively. Q_{mid} was calculated using Equation 5.5:

$$Q_{mid} = 0.5(Q_{max} - Q_{min}) + Q_{min} \tag{5.5}$$

The slopes of rising discharge limbs were calculated for each event using Equation 5.6:

$$Q\ Slope = \frac{dQ}{dt} \tag{5.6}$$

Where: $dQ = Peak\ Q - Initial\ Q$, and $dt = duration\ (hours) from\ initial\ Q\ to\ Peak\ Q$. Slopes of rising SSC limbs were also calculated following the same approach.

To examine the effect of land cover on sediment, the percentage of unvegetated or sparsely vegetated land within sub-catchments was obtained from Sentinel-2 satellite imagery at a 10 m resolution downloaded using the '*sen2r*' package (Ranghetti *et al.*, 2020). The percentage was derived by calculating the area of land with a normalised difference vegetation index (NDVI) of <0.4 for each storm event using the most recent image with a cloud cover of <15 %.

The stream monitoring timeseries data were also used to analyse variability in 'pulses' of Q and SSC over time (before and after NFM) at the Impact sub-catchment outlet. Following the method described by Archer and Newson (2002), pulses were defined as occurrences of a rise above a specified threshold, and pulse durations as the time between rising above and subsequently falling below the threshold. The number of pulses and their durations were extracted from SSC and Q timeseries which were split into 12-month periods of data. The five 12-month periods ran from the start date of data collection at the monitoring site (17/01/2017), and ended in 2022. However, the final period (2021) for SSC only runs for ~9.5 months until 02/11/2021. This was carried out using the median (M) values as a threshold, and repeated using various multiples of the medians as thresholds. The thresholds ranged from half of the median value (M0.5) to 100 times the median value (M100). The median Q and SSC were calculated using data from across the entire monitoring period.

5.3.4 Statistical Analysis

Kruskal-Wallis tests were carried out to compare differences in the average hydrological and sediment responses to storm events between the different phases of implementation of the NFM scheme in the Control and Impact sub-catchments. This non-parametric test was chosen due to the highly skewed distribution of the response data. The '*kruskalmc*' function in the '*pgirmess*' package in R was then used to carry out post-hoc pairwise comparisons to determine which phases were significantly different from one another.

Comparing average responses to storm events over time provides an indication of whether NFM may be having an impact on these variables, however it does not account for the differences in rainfall between events and the different phases monitored. Therefore, generalised linear models (GLMs) were used to examine the effect of rainfall, antecedent conditions, soil cover, and NFM interventions (both as before-after and as NFM phases) on response variables (peak SSC, total SS load, peak Q, total Q) in storm events, using the sub-catchment (Control/Impact) as a covariate. To account for observed differences in sub-catchment hydrological behaviour, GLMs were also run using each of the various estimated run-off variables in turn as a predictor in place of total rainfall. These BACI GLMs aimed to detect statistically significant interaction effects between the before-after/NFM phases and the sub-catchment whilst controlling for other potential influences. Similar approaches were previously used by Puttock *et al.* (2020) and Graham *et al.* (2022) to successfully demonstrate the effect of beaver dams on flood flow attenuation. The GLMs were run with a gamma error distribution and identity link function, chosen due to suitability for right-skewed data. The model formulations followed the example in Equation 5.7:

$$\begin{aligned} Peak\ SSC = Total\ Rainfall + API + Exposed\ Soil\ Cover \\ + NFM\ Phase \times Sub\text{-}catchment \end{aligned} \tag{5.7}$$

Parsimonious GLMs were achieved by removing non-significant continuous predictors in turn until the remaining continuous predictors were significant. A stepwise process (‘*stepAIC*’ function in the ‘*MASS*’ package) was then used to select the model with lowest AIC (Akaike Information Criterion) for each GLM.

Redundancy analysis (RDA) was used to better understand the relative importance of explanatory variables (presence of NFM, rainfall, antecedent conditions, soil cover etc.) on hydrological and sediment responses to storm events. The RDA aimed to explain the variation in responses in each sub-catchment into key gradients of influence. RDA is a constrained ordination technique which summarises multivariate data through a combination of principal components analysis and the regression of multiple response variables with multiple explanatory variables. This analysis was carried out using the R ‘*vegan*’ package (Oksanen *et al.*, 2020). Before running the RDA, the variables were transformed using a Box-Cox transformation and then centred and scaled for reducing the influence of outliers. To create a parsimonious model, a forward stepwise selection procedure was used to remove variables with high collinearity without reducing the explanatory power of the RDA. The resulting model was assessed using a Monte Carlo permutation test with 1000 permutations to determine the significance of the RDA axes.

Additionally, generalised additive models (GAM) were used for examining the non-linear effects of peak water storage measured within NFM offline and online pond features on storm event response variables. Information on how water storage volume data used within the GAM were collected is detailed by Trill *et al.* (2022a, 2022b) (see Appendix A, section A.3.7).

5.4 Results

5.4.1 Sub-catchment Characterisation

Stream monitoring data prior to the implementation of NFM revealed notable differences in the hydrological regime between the two sub-catchments (Figure 5.2).

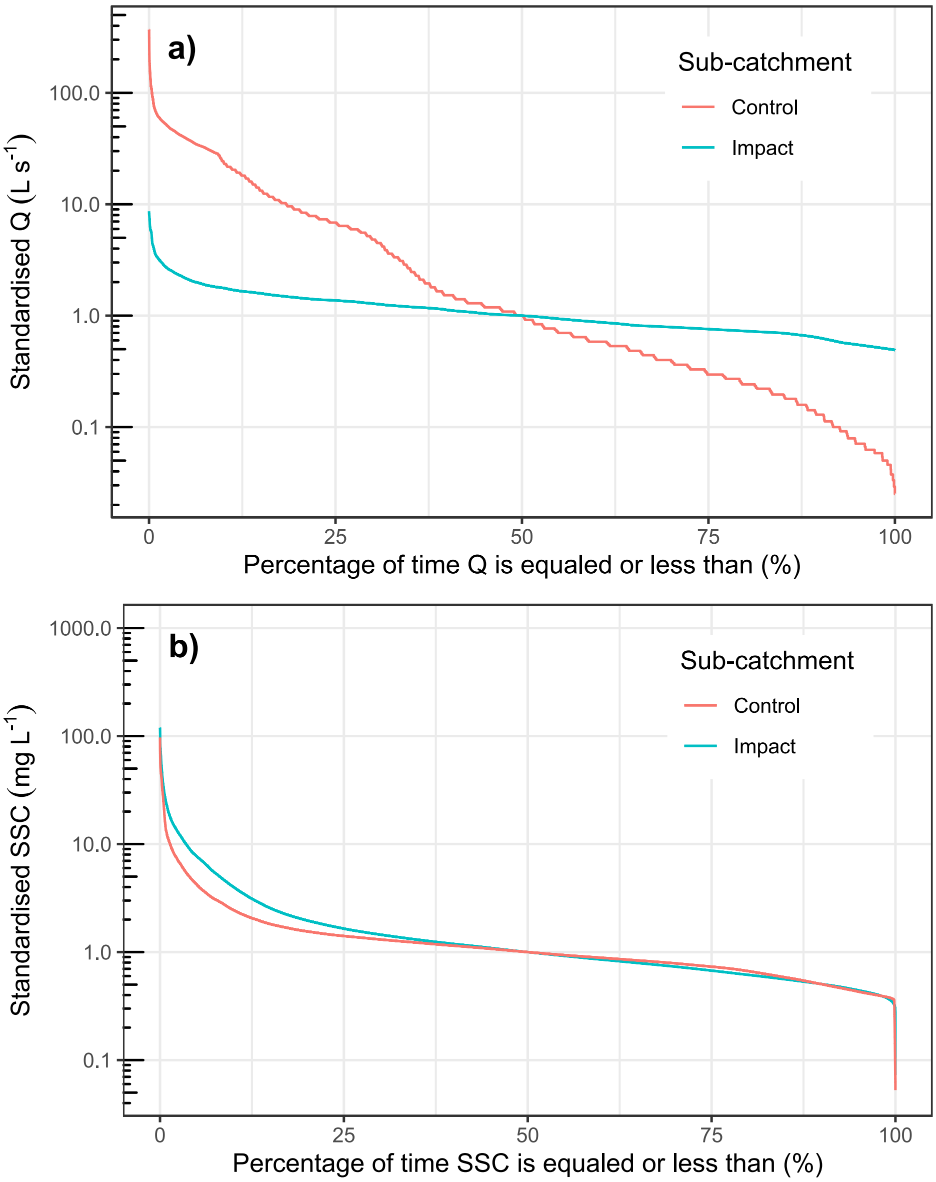

Figure 5.2 Standardised duration curves for **a)** Q (L s^{-1}) and **b)** SSC (mg L^{-1}) in the Control and Impact sub-catchments using stream monitoring data prior to the implementation of NFM (Jan 2017 – Feb 2018). Note the log scale of the Y-axis and data standardisation.

The Control sub-catchment exhibited a flashier behaviour as shown by the considerably steeper duration curve, with the highest flows exceeding those in the Impact sub-catchment. The Impact sub-catchment had a damped regime showing relatively little variation in comparison. The SSC regime was found to be more similar between sub-catchments, with the main difference being slightly elevated concentrations within the Impact sub-catchment during the highest ~35 % (Figure 5.2).

Annual SS and TP fluxes show considerable variation between the monitored years, particularly at the Control site (Table 5.3). Fluxes in both sub-catchments were lowest in 2017 and highest in 2020. On average across the monitoring period sediment fluxes were highest at the Control site, however in 2017 the flux at the Impact site was marginally greater by 6.3 tonnes. Sediment yields varied from 16-99 t km^{-2} y^{-1} which are considered within the typical range for a lowland setting, with the UK average reported as being in the region of 50 t km^{-2} y^{-1} (Walling *et al.*, 2007). Total discharges were consistently higher from the Impact sub-catchment, most notably in 2017. However in the wetter years of 2019 and 2020 (see Figure 5.4 and Figure 5.7 for rainfall), total discharges were more comparable between the sub-catchments.

Table 5.3 Annual suspended sediment and total phosphorus fluxes (t) and total discharges (million m^3) with lower/upper uncertainty bounds and yields for the Impact and Control sub-catchments. In years with missing data, the flux/total discharge calculations excluded these periods for both sub-catchments.

Sub-catchment	**Year**	**SS/TP flux (t) Total discharge (million m^3)**	**Lower 95 % bound (t)**	**Upper 95 % bound (t)**	**SS/TP yield (t km^{-2} y^{-1}) Water yield (million m^3 km^{-2} y^{-1})**	**Missing data (days)**
Suspended Sediment						
Impact	2017	61.94	57.34	66.66	18.22	19.5
	2018	76.61	69.81	83.69	22.53	15.5
	2019	94.75	89.79	99.81	27.87	0
	2020	272.76	259.24	286.52	80.22	0
Control	2017	55.64	49.20	62.01	16.36	63
	2018	95.14	76.15	115.91	27.98	0
	2019	182.03	154.01	216.35	53.54	0
	2020	335.13	281.55	396.4	98.57	0
Total Phosphorus						
Impact	2017	0.18	0.16	0.19	0.05	19.5
	2018	0.21	0.19	0.23	0.06	15.5
	2019	0.24	0.23	0.25	0.07	0

Sub-catchment	Year	SS/TP flux (t) Total discharge (million m³)	Lower 95 % bound (t)	Upper 95 % bound (t)	SS/TP yield (t km^{-2} y^{-1}) Water yield (million m³ km^{-2} y^{-1})	Missing data (days)
	2020	0.59	0.57	0.62	0.17	0
Control	2017	0.16	0.14	0.18	0.05	63
	2018	0.28	0.24	0.33	0.08	0
	2019	0.55	0.48	0.64	0.16	0
	2020	0.85	0.73	1.00	0.25	0
Total Discharge						
Impact	2017	1.92	1.81	2.03	0.56	19.5
	2018	1.94	1.83	2.05	0.57	0
	2019	1.78	1.70	1.85	0.52	0
	2020	2.16	2.10	2.22	0.64	0
Control	2017	0.40	0.37	0.43	0.12	0
	2018	0.73	0.67	0.79	0.21	0
	2019	1.45	1.34	1.66	0.43	0
	2020	1.50	1.37	1.73	0.44	0

The slope of the positive linear relationship between SSC and TP was consistent between the two sub-catchments, however the intercept was significantly higher (by 0.105 mg P L^{-1}) in the control sub-catchment (*p*<0.001; Figure 5.3).

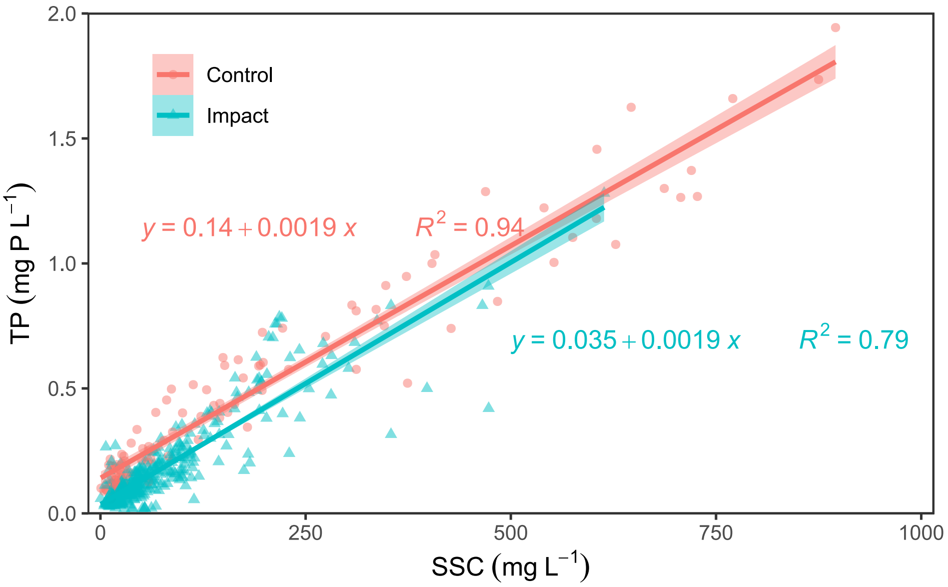

Figure 5.3 Linear regressions of SSC (mg L^{-1}) and TP (mg P L^{-1}) from samples taken at monitoring sites within the Control (red) and Impact (blue) sub-catchments.

5.4.2 Monthly-scale BACI Analysis

The mean SS flux difference in the *before* period was positive (0.6 t), indicating that on average the monthly fluxes leaving the Impact sub-catchment were greater than those leaving the Control (Figure 5.4). Conversely, in the after period, the mean flux difference was negative (-3.9 t), however there was considerably greater variation in monthly flux differences within this period compared to pre NFM. Notably large confidence intervals in months such as December 2020 reflect the increased uncertainty in Q estimates during high flows.

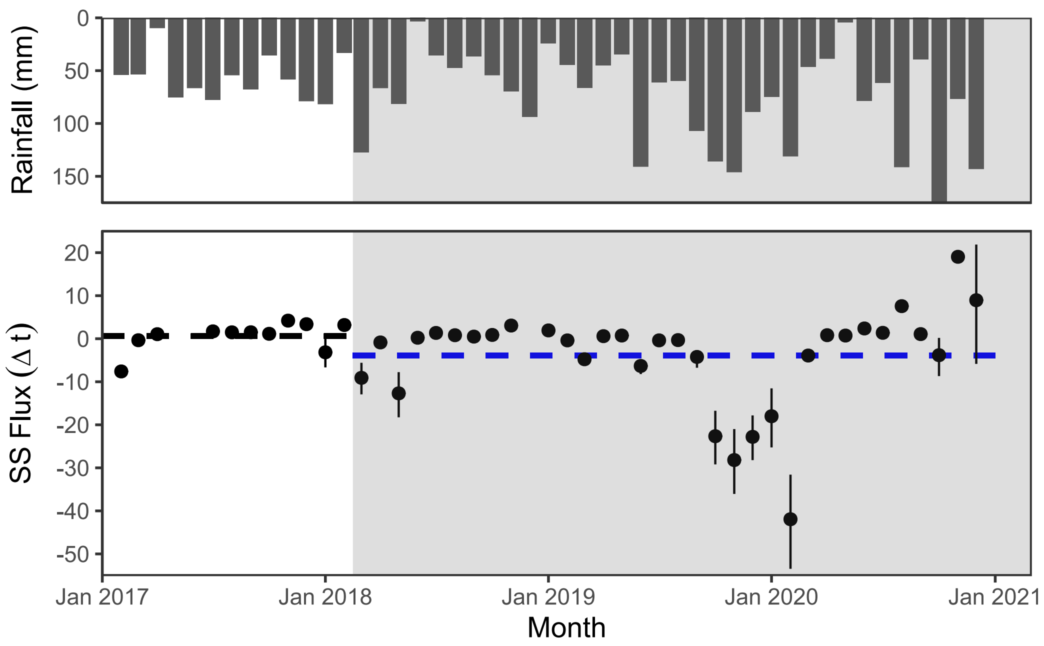

Figure 5.4 Monthly rainfall (mm) and suspended sediment flux difference (Δt) between the Impact and Control sub-catchments (±95 % confidence intervals) throughout the study period (grey shading denotes the after period). Mean SS flux differences before and after NFM implementation are shown as black and blue lines respectively.

5.4.3 Event-scale BACI Analysis

A greater number of storm events occurred during the *after* period due to it spanning multiple years compared to the shorter *before* period, but also a consequence of wetter conditions as highlighted by the higher API and rainfall intensities (Table 5.4). Mean event peak Q and SSC were higher in the Control sub-catchment both *before* and *after*. Both mean SS load and total Q were markedly higher in the *after* period. Mean SSC slope was higher in the Control sub-catchment *before*, but only marginally different (<1 mg L^{-1} h^{-1}) *after*, whereas mean Q slope remained higher in the Control during both periods. On average, HI_{mid} were higher in the Control both *before* and *after*, with positive values indicating that event hysteresis loops were mostly clockwise.

Table 5.4 Summary statistics (mean, minimum and maximum) of selected hydrological and catchment variables calculated from storm events occurring *before* (*n*=14) and *after* (*n*=134) NFM interventions in the Control and Impact sub-catchments.

Variable	Summary statistic	Intervention period and sub-catchment			
		Before		After	
		Control	Impact	Control	Impact
Total Rainfall (mm)	Mean	12.1		11.8	
	Min	3.4		1.8	
	Max	31.2		46.2	
Peak Instantaneous Rainfall Intensity (mm h^{-1})	Mean	78.3		100.5	
	Min	18		18	
	Max	144		408	
Mean Instantaneous Rainfall Intensity (mm h^{-1})	Mean	13.2		17.1	
	Min	1.5		4.2	
	Max	37.5		70.8	
API	Mean	44.54		65.21	
	Min	24.53		15.56	
	Max	60.21		112.54	
Exposed Soil Cover (%)	Mean	20.9	25.2	26.6	41.2
	Min	16.6	18.7	4.2	0.8
	Max	29.7	35.8	41.9	65.7
Peak SSC (mg L^{-1})	Mean	769.6	493.4	618.1	546.6
	Min	171.3	159.7	49.6	30.0
	Max	1764.3	1268.4	3296.4	2955.6
Peak Q (L s^{-1})	Mean	247.9	244.4	301.3	226.6
	Min	71.9	100.6	5.9	10.1
	Max	887.9	531.6	2518.8	846.7
SS Load (kg)	Mean	4314	3399	3554	2458
	Min	854	444	7	17
	Max	14033	10953	34636	27462
Total Q (m^3)	Mean	13152	18604	11308	10609
	Min	2094	5604	194	305

	Max	27447	33499	69093	58647
Variable	**Summary statistic**	**Intervention period and sub-catchment**			
		Before		**After**	
		Control	**Impact**	**Control**	**Impact**
SSC Slope (mg L^{-1} h^{-1})	Mean	76.69	50.94	85.35	86.06
	Min	11.73	15.57	1.63	1.89
	Max	209.26	131.49	941.09	509.06
Q Slope (L s^{-1} h^{-1})	Mean	19.93	14.56	25.83	19.38
	Min	3.45	3.04	0.70	0.22
	Max	91.29	49.08	230.67	179.24
HI_{mid}	Mean	1.27	0.28	0.63	0.09
	Min	0.02	-3.37	-6.57	-21.57
	Max	2.72	3.51	21.71	10.68

Examining storm event response variable data in higher granularity by comparing between the different phases of NFM scheme implementation (as described in Table 5.2) highlighted several changes in the hydrological response of the Impact sub-catchment (Appendix F (*A1*)). Peak Q was significantly higher in Phase 4 compared to Phase 3 (p<0.05). Total event Q was significantly lower in Phase 3 compared to both Phase 0 and Phase 2 (p<0.001). Peak SSC was significantly higher in Phase 4 compared to both Phase 2 and Phase 3 (p<0.001). SS Load was significantly lower in Phase 3 compared to both Phase 0 and Phase 4 (p<0.001). No significant differences between the phases were detected for any of the storm event response variables in the Control sub-catchment.

The GLMs were unable to find a significant interaction between the *before-after period* and the Control-Impact sub-catchment for all response variables. When using the NFM phases as a term in the GLM, there was a significant interaction between NFM Phase 4 and the sub-catchment (p<0.01), but only when modelling Peak SSC. Scatterplots were used to visualise relationships between mean instantaneous rainfall intensity and Peak SSC, and total event rainfall and Total SS load (Figure 5.5). The scatterplots also compare how these relationships differed between the NFM phases. The responses of SSC and SS load to mean rainfall intensity and total event rainfall were highly variable both within and between each phase. Phase 4 had distinctly elevated Peak SSC in the Impact sub-catchment, even at low rainfall intensities (Figure 5.5b). Total rainfall was found to be a significant predictor in the models for SS Load, Peak Q, and Total Q. Rainfall intensity was a significant predictor of Peak SSC in the final model. API was a significant predictor of Peak SSC, Peak Q and Total Q, whereas Exposed Soil Cover was a significant predictor of SS

Load. The GLM that included Peak Q as an additional variable showed that it was also a significant predictor of Peak SSC. The GLM for Total event Q showed significant pairwise differences in the predictor-response relationships observed between NFM Phase 3 and Phase 0, and also between NFM Phase 4 and Phase 0, irrespective of the sub-catchment in which events occurred (Appendix F (*A2*)). See Appendix F (*A3*) for the parsimonious model results for each GLM.

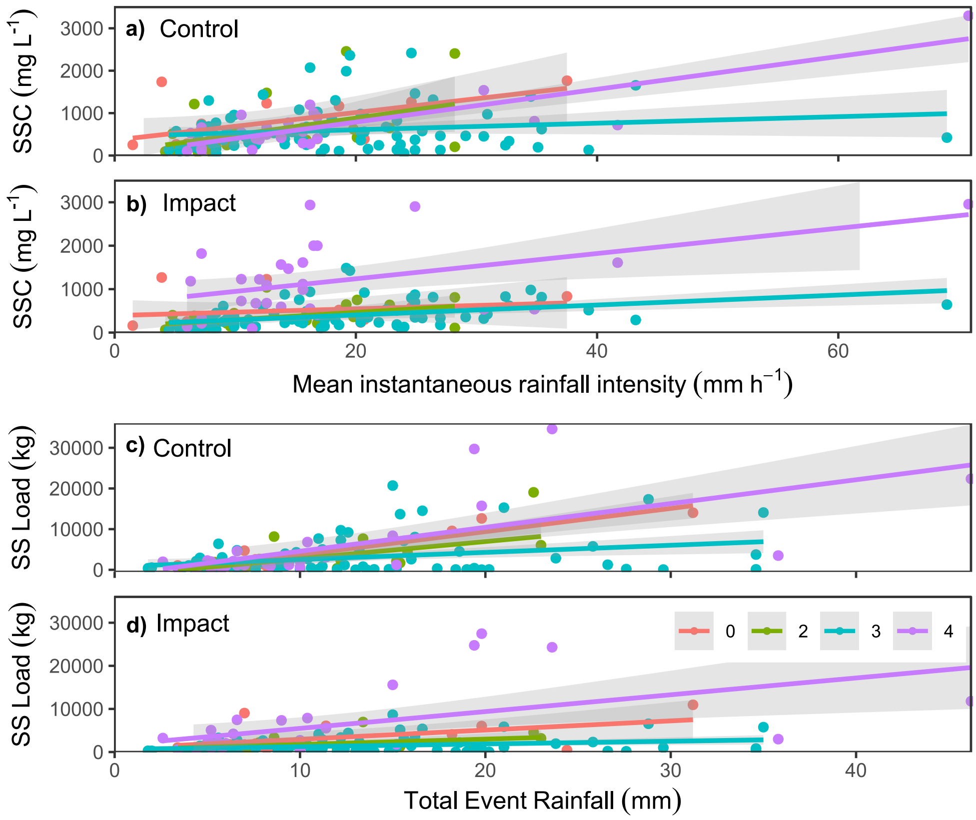

Figure 5.5 Storm event peak SSC (mg L^{-1}) and SS load (kg) in the Control and Impact sub-catchments as functions of **a-b)** mean instantaneous rainfall intensity (mm h^{-1}) and **c-d)** total event rainfall (mm), and the NFM Phase in which each event occurred (as modelled in GLMs). Shaded bands represent 95 % confidence intervals. NB: Phase 0 events include those that occurred prior to Phase 2 (i.e., pre-NFM).

GLMs were also run with different modifications to the total rainfall variable in order to control for the measured differences in sub-catchment hydrology (as observed in Section 5.4.1). The estimated event run-off indices that were derived for each sub-catchment from the SPR_{HOST}, the impermeable geology, the slowly permeable soil, and the average slope all did not alter the interaction between the NFM Phase and sub-catchment for all response variables.

Storm event response variables in each sub-catchment were plotted against each other, showing the changes in relationships between Control and Impact responses within each NFM phase (Figure 5.6).

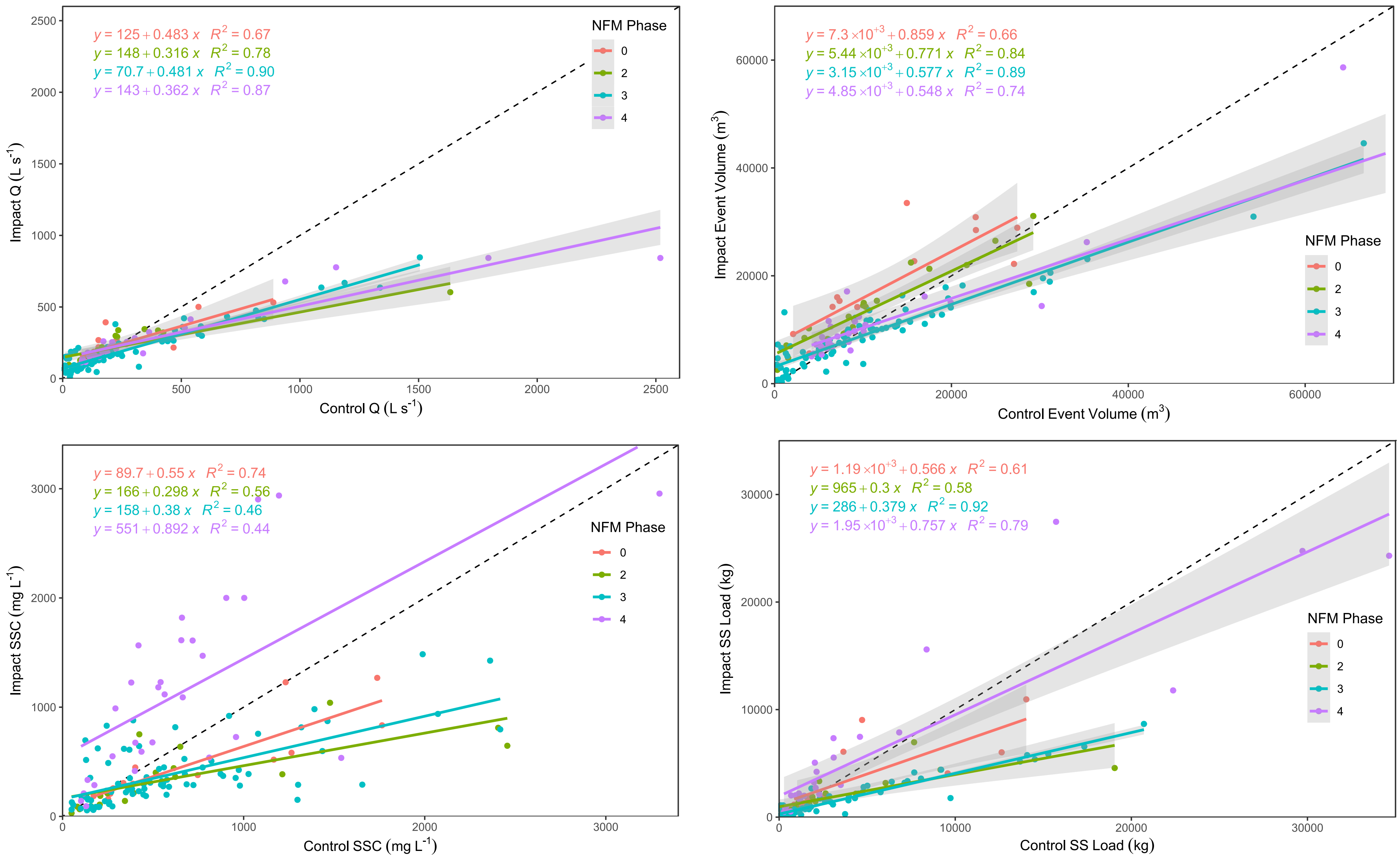

Figure 5.6 Linear relationships between Control and Impact sub-catchment responses for storm event peak Q (L s^{-1}), event volume (m^3), peak SSC (mg L^{-1}), and SS load (kg) in each NFM Phase. Shaded bands represent 95 % confidence intervals (not shown for SSC due to large intervals for Phase 4). The dashed line represents the 1:1 relationship. NB: Phase 0 events include those that occurred prior to Phase 2 (i.e., pre-NFM).

In lower magnitude events peak Q was frequently higher in the Impact sub-catchment in all NFM phases, however above ~300 L s^{-1} peaks were consistently higher in the Control sub-catchment. Differences in the slope of this relationship between the NFM phases were small and showed no significant difference pre and post implementation of NFM. When looking at the total volume of stream water leaving each sub-catchment, the relationship is more distinct between the phases which show a progressively decreasing slope with each phase of NFM implementation. Peak SSC exhibits the most divergent response in Phase 4, where peak concentrations in the Impact sub-catchment vastly exceeded those in the Control. The prior phases all exhibited the opposite trend for peak SSC. When looking at the total SS load, Phase 4 also stands apart, highlighting the influence of elevated SSC in this period which resulted in higher loads in the Impact sub-catchment for smaller magnitude events. However, during larger magnitude events the response is mediated by discharge with significantly higher Q from the Control sub-catchment resulting in a higher total SS load. HI_{mid} showed weak correlation between the sub-catchments across all NFM phases ($R^2 \leq 0.28$).

The peak water storage in the Impact sub-catchment NFM features ranged up to almost 10,000 m^3 during the monitored events. GAMs which modelled peak Q and total event Q showed a considerable divergence in the response of the sub-catchments during higher magnitude storm events (Appendix F (*A4*)). The Impact sub-catchment appeared to show a levelling off, unlike the Control which maintained high Peak Q and Total Q in such events. Peak SSC exhibited a highly variable response which the GAM did not explain well in either sub-catchment. SS load response showed less variation but did not show an attenuation effect in higher magnitude events in the same way that Peak Q and Total event Q did. It should be noted that the highest peak NFM storage volumes occurred during two consecutive storm events (on 23rd December 2020), but for the purposes of the total event Q and SS load GAMs they were analysed as one double-peaked event.

5.4.4 Q and SSC Variability Analysis

The number of Q pulses was found to be highest between M0.5 and M2 across all 12-month periods; the number of pulses steadily decreased at higher multiples up to M15, beyond which there were no pulses that exceeded the specified thresholds (Appendix F (*A5*)). In terms of SSC, the number of pulses followed a similar distribution, however unlike Q, pulses were observed at considerably higher multiples of the median SSC, occurring at up to M100 in 2020 and 2021 (Appendix F (*A6*)). For both Q and SSC, the lowest number of pulses were observed during the 2017 (pre-NFM) period for almost all thresholds. In contrast, the 2019 and 2020 periods had the highest number of Q pulses; however, for SSC the 2021 period had the highest number of pulses at both low (M2 to M4) and high (M50 to M100) thresholds. The 2020 period had the highest total duration of Q and SSC pulses above thresholds of M2 and higher. The 2020 period also had the consistently highest mean duration of SSC pulses for thresholds greater than M. There was considerably less variation in the distribution of the mean duration of Q pulses between the 12-month periods.

5.4.5 Redundancy Analysis

RDA found that the storm event response variables in the Control sub-catchment were significantly constrained by explanatory variables along two axes (RDA1: 76.6 %, p<0.001; RDA2: 15.2 %, p<0.05) (Appendix F (*A7*)). Response variables in the Impact sub-catchment were significantly constrained along three axes (RDA1: 53.2 %, p<0.001; RDA2: 33.4 %, p<0.001; RDA3: 8.4 %, p<0.05). The adjusted R^2 of the final models for the Control and Impact sub-catchments were 0.37 and 0.46 respectively after removing the event total rainfall variable in the forward stepwise selection. Total Q was strongly seasonal, with higher fluxes of water leaving the sub-catchments in winter events (more so in the Control sub-catchment). Conversely, total Q was negatively correlated with the number of days since the occurrence of the previous event. Total SS load was most closely associated with peak Q. The rate of rise to peak SSC (*SSCslope*) was most correlated with peak rainfall intensity in the Impact sub-catchment but with mean rainfall intensity in the Control sub-catchment. The NFM Phase variable was included in the final RDAs but had less explanatory power than most other variables in both sub-catchments. Each of the NFM Phase vectors were considerably distinct from one another. Phase 4 was mostly closely associated with the rate of rise to peak Q (*Qslope*), with this association being strongest in the Impact sub-catchment.

5.4.6 Hydrometeorological Conditions, Catchment Cover, and Land-use

The hydrometeorological conditions varied considerably between the initial part of the study period and the latter part, with greater daily rainfall totals falling from the start of the 2020 hydrological year (Figure 5.7). There is a notable upward trend in API caused by the near-continuous daily rainfall throughout the winter of 2019-2020. Despite a relatively dry spring in 2020, the API rebounded after multiple intense rainfall events throughout the rest of the year during within which the highest daily total of the entire monitoring period was observed. HI_{mid} showed no clear seasonal or temporal trend, however the sub-catchment response diverged somewhat during the end of 2020 with an increase in the proportion of negative HI_{mid} values in the Impact sub-catchment.

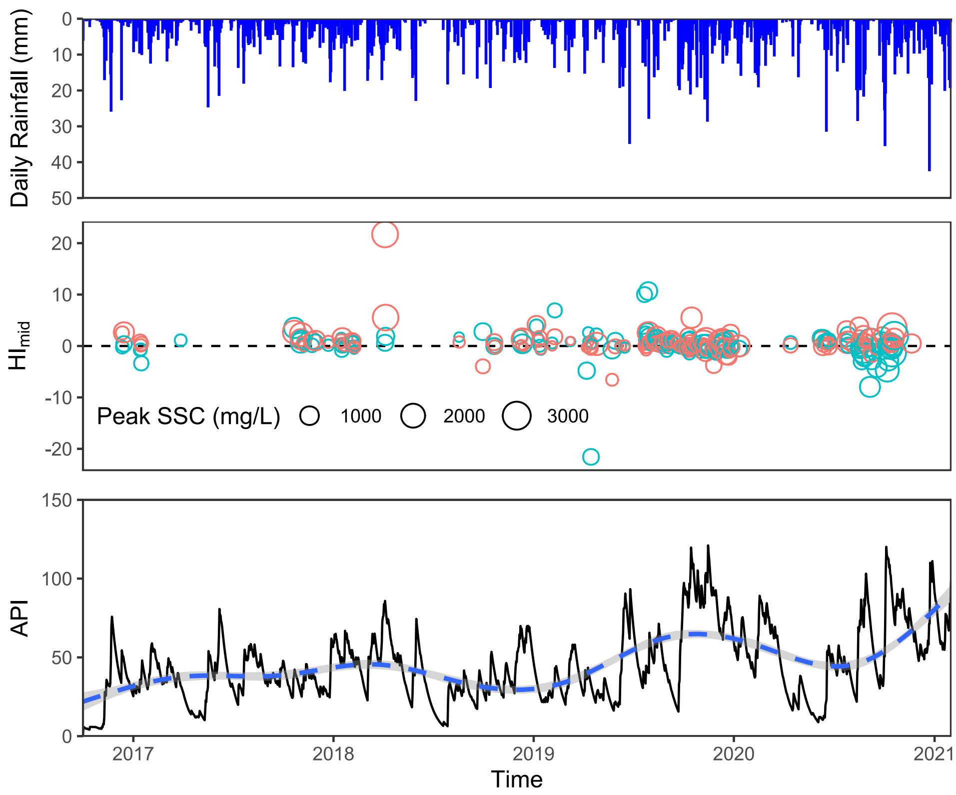

Figure 5.7 Daily rainfall (mm), HI_{mid} and antecedent precipitation index (API) from Oct 2016 to Feb 2021 in the study area. The size of HI_{mid} circles indicates the event peak SSC in the Control (red) and Impact (blue) sub-catchments. The blue dashed line shows the temporal trend in API produced from a GAM (shaded bands represent 95 % confidence intervals).

Total annual rainfall in both 2019 and 2020 was considerably above the annual average (Figure 5.8). The timing of rainfall throughout each year broadly fluctuated in a similar pattern until mid-August in 2020 and towards the end of September in 2019. Both years proceeded to diverge from 2017 and 2018 with continued wetter than average conditions, with 2020 having a total cumulative rainfall 32 % above the average for 1981-2010 and 25 % above the 1991-2020 average.

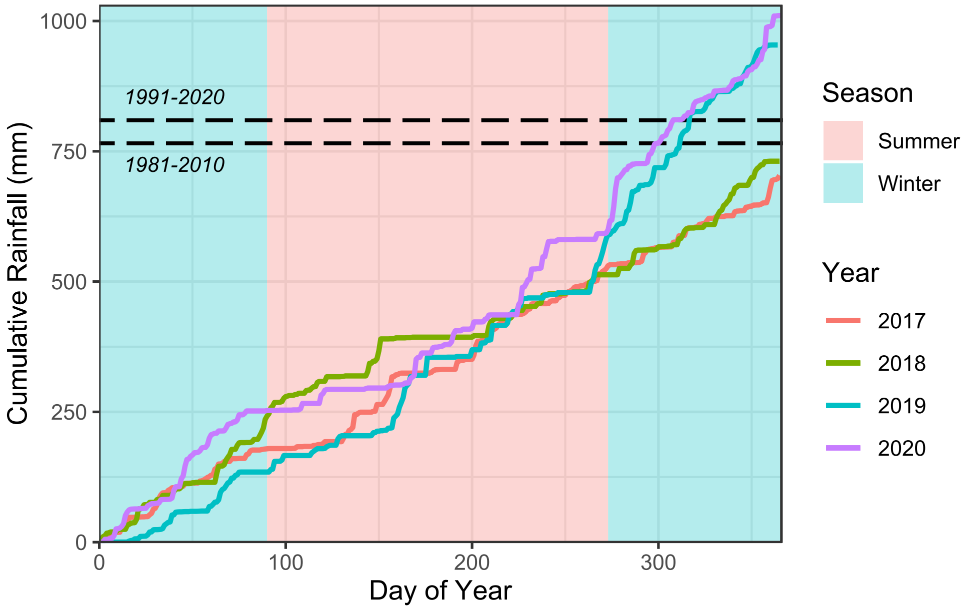

Figure 5.8 Cumulative rainfall (mm) in the study area for each calendar year of the study period, with hydrological seasons shaded in blue (winter) and red (summer). The dashed lines represent the average annual rainfall for the study area during 1981-2010 and 1991-2020 (Met Office, 2021).

The dominant land-use in the Impact sub-catchment was consistently cereal crops (>50 % cover) whereas in the Control sub-catchment grassland became dominant in 2019 but then was replaced by an increase in leguminous crops in 2020 (Figure 5.9). Tree and scrub cover was consistently higher in the Control sub-catchment, but only constituted under 10 % cover in the Impact sub-catchment. The dominant crops in both sub-catchments were typically spring barley, winter wheat and winter oilseed.

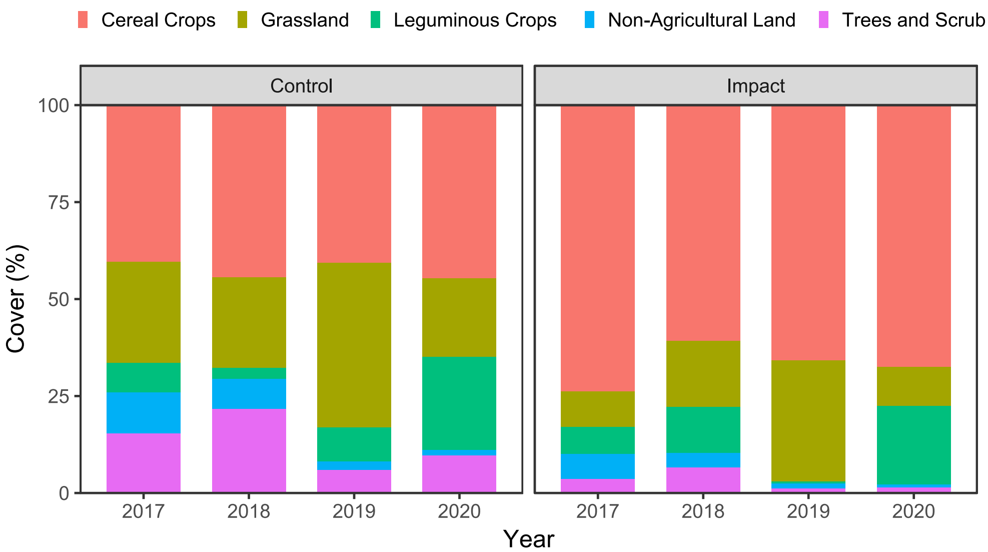

Figure 5.9 Changes in Control and Impact sub-catchment land-use cover (%) over the monitoring period.

The exposed soil metric provides further insight into the extent to which the sub-catchments were susceptible to erosion during each monitored storm event (Figure 5.10c). Both sub-catchments showed increasing trends in exposed soil over time (p<0.01), but the Impact sub-catchment saw a greater rate of rise due to notably bare fields throughout winter 2019/2020 and autumn 2020 where it reached a high of >65 % at the start of the hydrological year in October. The mean rainfall intensity of storm events was positively correlated with their peak SSC in both sub-catchments, however exposed soil did not appear to show a clear trend in moderating this relationship (Figure 5.10a). The Peak Q of events was also positively correlated with Peak SSC, and took the form of a non-linear relationship in the Control sub-catchment, showing a levelling off at the highest discharges (Figure 5.10b). The Impact sub-catchment did not show the same indication of sediment exhaustion; however this may be a result of the Peak Q range only being less than half of that of the Control sub-catchment. Again, the exposed soil showed no apparent trend in moderating this relationship.

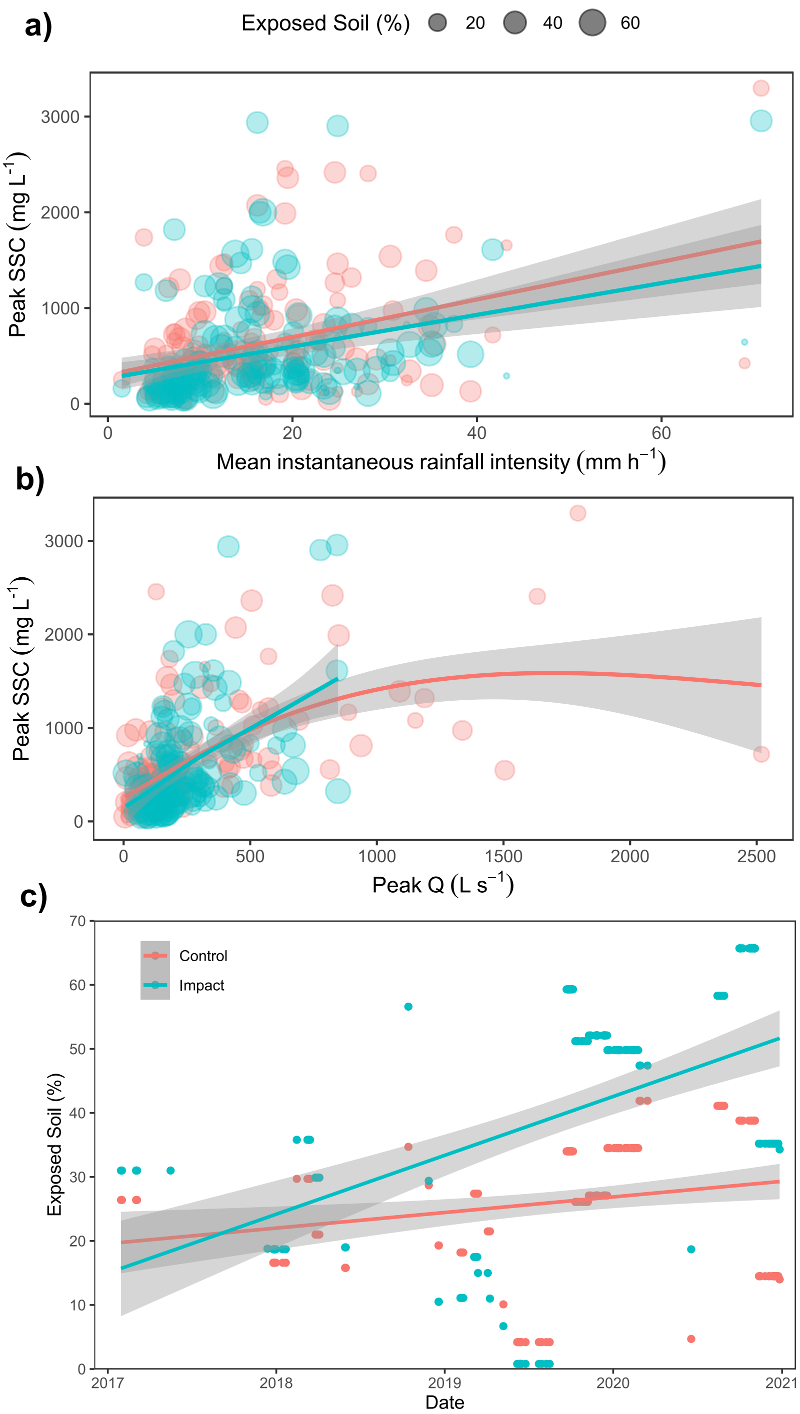

Figure 5.10 Correlations between event **a)** mean instantaneous rainfall intensity (mm h^{-1}) and peak SSC (mg L^{-1}) , and **b)** peak Q (L s^{-1}) and peak SSC. The size of points represents the exposed soil cover (%) at the time of each storm event. **c)** Exposed soil cover at the time of each event within each sub-catchment as a function of time. Shaded bands represent 95 % confidence intervals.

5.4.7 Dissolved Nutrients

Regular water sampling of dissolved nutrients showed that throughout the year concentrations of soluble reactive P (SRP), total dissolved P (TDP), ammonium and nitrite remained relatively stable with the exception of spikes during March/April and July (Figure 5.11). Nitrate concentrations showed a seasonal trend which gently peaked towards the end of the hydrological year and reached a trough during summer when daily mean discharge was also at its lowest. SRP concentrations were consistently higher in the control sub-catchment whereas nitrate was broadly higher in the impact sub-catchment. Within the control sub-catchment, nitrate became more concentrated downstream on average, whereas the impact sub-catchment showed seasonal nitrate attenuation downstream between July and October during this low flow period.

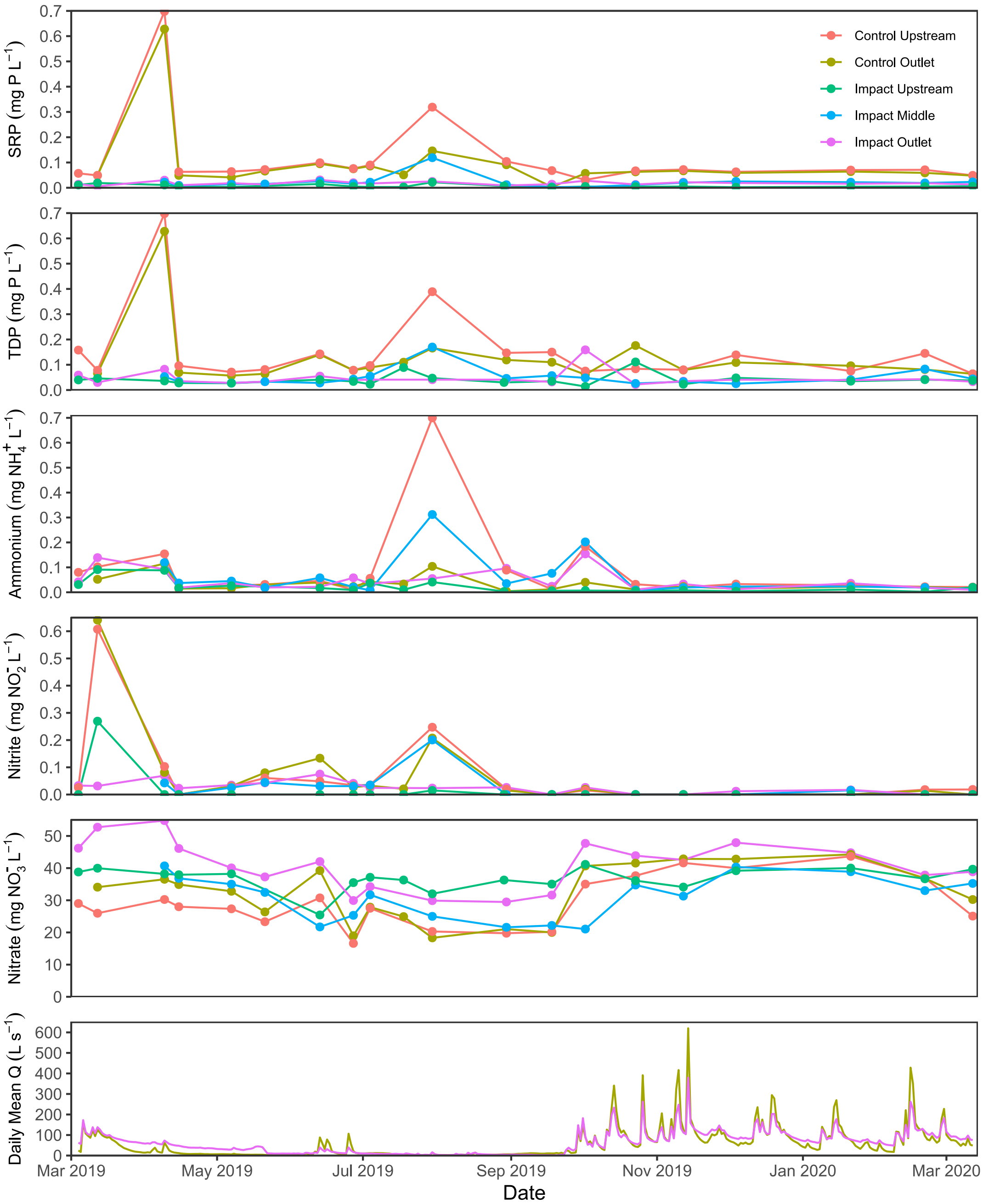

Figure 5.11 Timeseries of nutrient concentrations (mg L^{-1}) sampled regularly between March 2019 and 2020 at different sites within the study sub-catchments (see Figure 5.1 for locations), and daily mean discharge (L s^{-1}) at the sub-catchment outlets. SRP = soluble reactive P; TDP = total dissolved P.

The relationship between discharge and stream nitrate concentration followed a similar pattern in each sub-catchment, though peak concentrations in the Impact sub-catchment were notably higher, reaching almost double those in the Control sub-catchment (Figure 5.12). The Q-C relationship showed a concentration of nitrate with increasing flow during summer, whereas in winter dilution occurred during higher flows of above ~50 L s^{-1}.

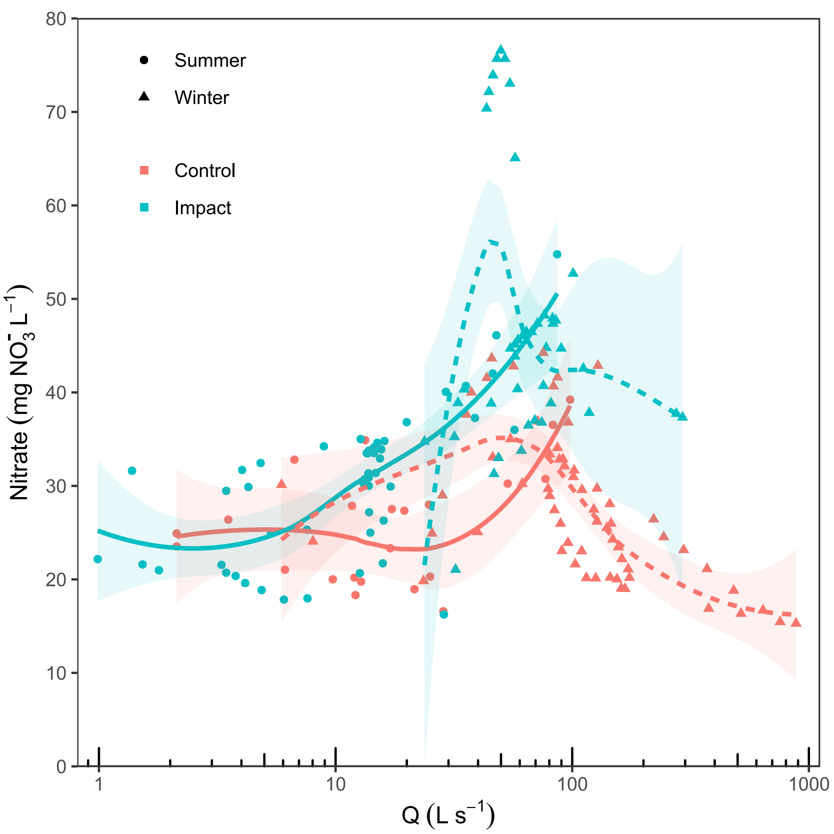

Figure 5.12 Seasonal discharge (L s^{-1}) – nitrate concentration (mg NO_3^- L^{-1}) relationships in the Control and Impact sub-catchments. Note that Q is plotted on a logarithmic scale. Locally estimated scatterplot smoothing (LOESS) are shown in solid lines (summer) and dashed lines (winter) with shaded bands representing 95 % confidence intervals.

5.5 Discussion

5.5.1 Suspended Sediment and P Transport – Sources and Pathways

Results suggest that suspended loads in both sub-catchments are largely transport-limited rather than supply-limited. However, the considerable degree of variability observed in HI_{mid} between the sub-catchments during storm events demonstrates how sediment transported to each outlet is likely to have been derived from different source types or locations (Lloyd *et al.*, 2016b; Malutta *et al.*, 2020; Sherriff *et al.*, 2016). The typically higher HI_{mid} of the Control sub-catchment showed greater clockwise hysteresis, indicative of readily available proximal sources such as streambed sediment. HI_{mid} was generally lower in the Impact sub-catchment but notably so during the wet Phase 4 period, during which hysteresis was predominantly anti-clockwise, indicating that the SSC response lagged behind Q in these events. This suggests an activation of distal sediment sources, likely as a result of the greater hydrological connectivity in the landscape brought about by saturated soil and activation of overland flow pathways. The differences in sub-catchment hysteresis response may in part be explained by the influence of soil hydrology and geology, with a greater proportion of shallow well-drained soil and highly permeable limestone underlying the Impact sub-catchment, thereby enhancing throughflow. These properties are also thought to explain the contrasting shape of the sub-catchment flow duration curves. Sherriff *et al.* (2016) found that a poorly-drained headwater catchment rarely exhibited anti-clockwise hysteresis, in contrast to a moderately-drained one in which anti-clockwise hysteresis was dominant during winter. The predominance of clockwise hysteresis in the more poorly-drained Control sub-catchment likely resulted from its flashy hydrology and lower baseflow. These characteristics enable highly erosive flows and encourage channel sediment accumulation due to deposition on rapidly receding limbs, in turn creating an easily mobilised sediment supply in future events (Mellander *et al.*, 2015). Additionally, the higher cover of permanent grassland reduced the erodibility of fields within the sub-catchment and therefore limited the probability of delivery from distal sources (Sherriff *et al.*, 2016). The relationship between discharge and SSC indicated that sediment exhaustion may have occurred at the highest flows in the Control sub-catchment.

The strong relationships observed between SSC and TP highlights particulate P (adsorbed to sediment) as the dominant form of P exported from both sub-catchments. However, the elevated intercept of this relationship in the Control sub-catchment indicates a greater contribution from dissolved P during baseflows compared to the Impact sub-catchment. This is supported by the data from regular baseflow sampling showing consistently higher SRP at the Control sub-catchment outlet. Walk-over stream surveys and spatial sampling from different tributaries suggest that point source contributions from the small settlements upstream in the sub-catchment are the main cause of the chronically elevated dissolved P concentrations.

5.5.2 Detecting the Effects of NFM

The BACI approach was unable to detect a statistically significant reduction in any of the storm event response variables, both when considering the implementation of NFM interventions in a standard before-after design and when using a more pragmatic approach with incremental phases of additional NFM storage. The GLMs showed elevated peak SSC in the Impact sub-catchment during Phase 4 that were not observed in the Control sub-catchment, but this effect is thought to be an effect of significant differences in sub-catchment soil cover in this period (discussed further in Section 5.5.3). However, the GLMs did still provide insight into the hydrological changes over time and between the sub-catchments. In the GLM that examined the total Q of storm events as a response variable, significant differences were observed between Phase 0 and 3, and Phase 0 and 4, irrespective of the sub-catchment. The fact that this reduction in total Q over time was seen in both sub-catchments would suggest that this change in hydrological response is due to a factor other than the NFM. However, the difference in response is most notable in the Impact sub-catchment where a cluster of events in Phase 0 exhibit notably high total Q despite relatively low rainfall totals (see Figure 5.6 and Appendix F (*A2*)). This may indicate that prior to NFM interventions being installed, the sub-catchment was conveying a greater proportion of stormwater to the outlet during events. However, a lack of data points within this pre-NFM phase (0) makes it difficult to fully test this hypothesis, particularly given the high natural variability in hydrological responses both between events and year to year.

The variability analysis (Section 5.4.4) also demonstrated the contrast in Q and SSC between the monitored periods, with fewer pulses of both water and sediment in the relatively dry pre-NFM period. This approach aimed to evaluate the degree of hydrological change following the implementation of the NFM scheme, however the lack of long-term data (particularly pre-NFM data) meant that it was not possible to decouple the effect of rainfall variability between periods from the effect of the interventions. This method has previously been used by Archer (2007) to show the effects of afforestation on catchment flood flow responses over a 30-year period. The findings highlighted the value of using variability analysis to detect more subtle catchment-scale hydrological change which were unable to be identified by analysing peak flows (O'Connell *et al.*, 2004). This type of analysis is more likely to yield conclusive results on the effects of NBS where monitoring data include a longer pre-intervention period.

Evidence from our monitoring demonstrates that the NFM storage features were operating to intercept and store run-off as intended, with up to almost 10,000 m^3 water held at peak storage within the Impact sub-catchment during one of the highest magnitude events monitored. Assuming that in the absence of the NFM features this water would have rapidly become streamflow, for this event the total sub-catchment discharge would have been 23 % greater. This does not account for antecedent storage in the features, but nonetheless this estimate suggests that a significant volume of the flood wave was attenuated relative to the overall water flux leaving the sub-catchment. Analysis by Trill *et al.* (2022a) demonstrated that these NFM interventions were able to reduce downstream flood peaks by 14-55 % in the Impact sub-catchment. It would therefore not be unreasonable to assume that this degree of run-off capture would also contribute to a reduction in sediment loading, especially in light of earlier findings that directly measured accumulation within features (Robotham *et al.*, 2021; Robotham *et al.*, 2023). This raises the question of why this trapping effect is not reflected in the statistical BACI models that use the sub-catchment outlet data.

One reason for the absence of a significant signal from the NFM is due to the potential lag time in the stream water quality response to the implementation of interventions. Meals *et al.* (2010) undertook a review of such lag times in responses to BMPs for managing diffuse pollution; key processes influencing lag time were found to be hydrology, vegetation growth, transport rates and pathways, residence times, pollutant sorption properties, and ecosystem linkages. The study highlights how lag time magnitudes can be highly site and pollutant specific, with pollutants such as P and sediment potentially taking years to decades to respond to management practices. A study assessing farm-scale impacts of cover crops on nutrient losses found that despite recording significant reductions in soil pore water NO_3-N during cover cropping which covered 20 % of the catchment, there were no reductions in instream concentrations in the corresponding period (Cooper *et al.*, 2017). These findings were hypothesised to be a consequence of pre-existing legacy stores from decades of fertiliser application (Outram *et al.*, 2016). Similarly, the mobilisation of in-channel legacy stores of sediment in the Littlestock Brook may be masking the effect of NFM interventions on downstream concentrations and loads.

Another possible reason is the effect of dynamic sediment sources in the sub-catchments. The complex relationship between the volume of NFM storage utilised in the Impact sub-catchment and the peak SSC at its outlet suggests that despite targeted interventions there still remained significant sediment sources and pathways within the sub-catchment. Despite the sub-catchments having almost identical drainage densities, sediment and discharge dynamics were highly variable as a result of numerous catchment characteristics which likely played a role in the NFM detection effect ability. In-channel sediment from processes such as bank erosion and collapse during high flows, and also bioturbation by invasive signal crayfish (*Pacifastacus leniusculus*), the presence of which is known in both sub-catchments are likely sources. Sanders *et al.* (2021) found that rate of bank collapse is strongly associated with an increase in the density of *P. leniusculus* burrows and can contribute significant masses of sediment to streams. These geomorphic processes are compounded by the largely steep and incised channels seen in the sub-catchment, particularly where channels have previously been modified for agricultural drainage. The SCIMAP (sensitive catchment integrated modelling and analysis platform) tool suggests that the channels in the Impact sub-catchment are at a higher risk of in-channel sediment accumulation compared to the Control sub-catchment (Reaney *et al.*, 2019). Additionally, roads were not targeted by the NFM scheme for run-off interception, and our observations from wet weather surveying indicate they can act as significant pathways for sediment delivery into streams. Our findings do suggest that both hydrometeorological and land cover differences between the NFM Phases are likely to have played an important role in determining sediment sources and stream responses and also the ability of the BACI analyses to isolate the signal of NFM alone. The following section discusses some of these temporal changes and their implications.

5.5.3 Variability in Catchment Conditions

It is important to recognise that at a sub-catchment scale, a BACI study design is inherently imperfect due to the inability to maintain static conditions within the Control. To account for this within the analyses, such changes in catchment condition were quantified where possible with primary data (e.g. precipitation) or open sources of secondary data (e.g. land cover).

Changes in land cover throughout the study period were largely due to changes in cropping rather than as a direct result of the NFM scheme. Whilst one cereal field within the impact catchment was taken out of production from 2018 to accommodate the creation of on-line pond features and woodland, the majority of NFM interventions did not alter the landscape to such an extent. For example, most tree planting took place in patches along riparian zones and field margins and did not cover enough land to constitute a land-use change in any one given area, especially given that trees were planted as saplings with no or minimal leaf cover. It is acknowledged that the CROME data used to examine changes in cropping have an inherent degree of error, particularly with respect to specific crop type classification due to the random forest machine learning method used in its derivation (Rural Payments Agency, 2021). Despite its disadvantages, the data is useful in characterising the dominant arable crop types within the sub-catchments and therefore valuable in assessing the risk of its contribution to soil erosion and how this changed during the study period.

The susceptibility of arable land to soil erosion from rainfall varies as a result of crop types due to their growing season, growth rate, and planting pattern (Boardman, 2013). Winter oilseed poses a low erosion risk due to its quick establishment and plants can grow close together typically providing sufficient ground cover around the start of the hydrological year. On the other hand, winter cereals such as wheat are more likely to result in erosion because they are only able to establish sufficient cover (30 %) by around January, which leaves soils particularly vulnerable during late autumn and early winter when rainfall is typically highest. The autumn/winter of 2019 was notably wet, with November receiving almost 150 mm of rain, which is the highest monthly total during the study period and nearly double the monthly average for the area. The SS flux difference between sub-catchments was at its second most extreme during November 2019, with the Control sub-catchment having the fourth highest monthly flux of the study period, whereas the Impact sub-catchment only had the 18^{th} highest. During 2019, the Control sub-catchment had a winter wheat cover of 14 %, compared to 34 % in the Impact sub-catchment. This suggests that the large difference in fluxes is less likely to be a result of the differences in crop type cover between the two, especially given the extensive cover of permanent grassland (42 %) in the Control.

After arable land, pasture was the next biggest agricultural land-use. Grassland cover was highest in 2019 for both sub-catchments, with this category including fallow land (either bare or partially grass-covered ground) which accounted for <3 % cover. The contribution of grassland to soil erosion is significantly influenced by its management, most crucially whether or not it is grazed, and how it is grazed. Erosion risk is increased through the processes of soil compaction and poaching which are facilitated by high stocking densities and the outdoor wintering of animals on grassland (Bilotta *et al.*, 2007; Evans *et al.*, 2017). Within the Littlestock Brook catchment both sheep and cattle were grazed at relatively low densities on a rotational basis throughout the monitoring period. Soil compaction was observed at one site in the Control sub-catchment during the winter of 2019/2020, though this was limited to a small portion (~0.5 ha) of the field. It is thought that the generally good livestock management will have limited sediment transfers via infiltration excess overland flow from grassland.

Whilst crop type and land-use play an important role in diffuse pollution risk, the timing of farming activities and crop phenology can also crucially determine how vulnerable the landscape is to erosion. The percentage of exposed soil and the NDVI data from which the metric was derived gave an indication of how this risk varied both temporally and spatially (Figure 5.13). Autumn 2020 showed a notable contrast in NDVI between the sub-catchments, particularly in fields adjacent to the Littlestock Brook which nearly all had high vegetation cover in the Control sub-catchment. The opposite was true in the Impact sub-catchment, which had long reaches of stream that were only separated from bare arable fields by generally narrow riparian buffer strips and hedgerows (visible in Figure 5.13).

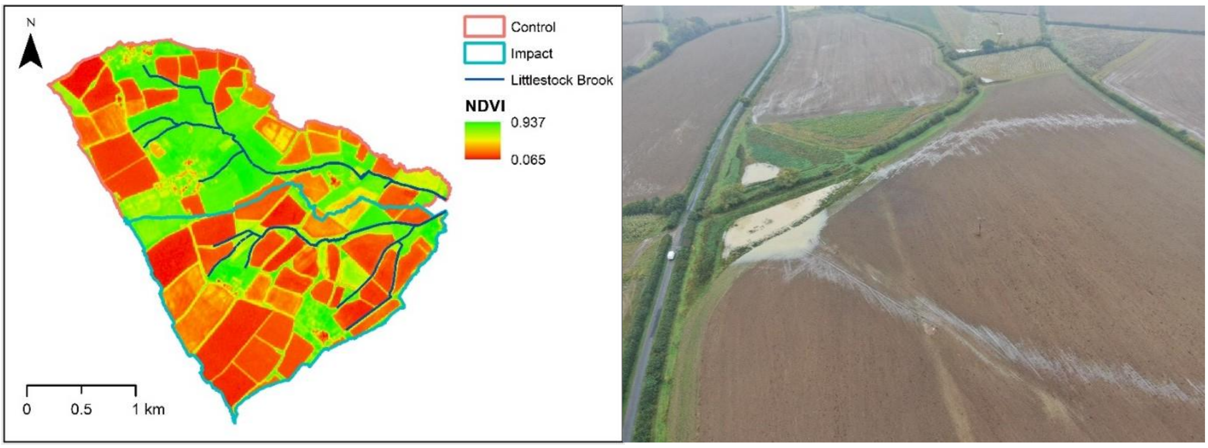

Figure 5.13 NDVI in the study sub-catchments derived from Sentinel-2 imagery taken on 21st September 2020 (left). Drone imagery showing overland flow pathways in the Impact sub-catchment during a storm event on 4th October 2020 (right).

It was also during this time that intense rainfall led to elevated API which suggests that antecedent soil moisture would have been relatively high, especially given the lack of crop cover and thereby reduced evapotranspiration (Meyer *et al.*, 2019). Under these conditions, it is likely that saturation excess overland flow may have occurred. In turn, the risk of soil erosion would have been increased and sediment transported downslope via sheet flow or preferential flow in rills, both of which were observed during storm events (Figure 5.13). This coupling of a lack of soil cover with above average rainfall and wet antecedent conditions provides a likely explanation for the notably high SSC and SS loads observed downstream during Phase 4.

GLMs highlighted that the relationships between these variables are complex and exhibit high levels of dispersion. Higher rainfall intensities correlated with higher peaks of SSC; however this did not appear to be significantly moderated by exposed soil. In terms of the total sediment load exported during events, exposed soil was an important contributing factor alongside total rainfall. Bare fields provided an easily mobilised source of sediment across the catchment, with reduced buffering from vegetation facilitating its delivery to stream channels. Relatively low rainfall totals were shown to be capable of exporting high sediment loads at times when soil was more prone to erosion. Although intense rainfall events were shown to result in highly-concentrated pulses of sediment with high SSC peaks, results indicate that associated total loads can be relatively low. This is thought to be due to the shorter duration of intense rainfall events that were characterised by flashy storm hydrographs where discharge rapidly returned to baseflow levels and the total volume of rainfall and run-off was relatively low. High rainfall intensities are more likely to cause infiltration excess overland flow and thereby rapidly activate critical source areas of sediment. Additionally, intense rainfall and consequent rapid increases in streamflow can easily mobilise channel bed sediment leading to concentrated SS pulses. Evidence to support these mechanisms of sediment delivery was seen in the redundancy analysis which showed that both the rate of rise in SSC and Peak SSC were most closely associated with rainfall intensity variables (maximum and mean). Previous studies have also found significant relationships between rainfall intensity and SSC (Lana-Renault *et al.*, 2007; Liu *et al.*, 2020). The role of antecedent conditions, vegetation and discharge were highlighted as key interacting factors in changing the response of sediment concentrations.

5.5.4 Dissolved Nutrient Dynamics

Nitrate was shown to be the dominant N species in the Littlestock Brook, contributing to high levels of dissolved inorganic nitrogen (DIN) in both sub-catchments. The dilution pattern of nitrate observed during periods of high stream discharge in winter indicate that its transport is associated with baseflow and slow flow pathways (Johnson and Stets, 2020; Knapp *et al.*, 2020). The shallow soils and limestone underlying upper parts of the catchment are thought to have enhanced transport through sub-surface pathways and thereby augmented baseflows, more so in the Impact sub-catchment which contains a higher proportion of permeable bedrock and soil. In heavily-modified agricultural landscapes, field drains can aid delivery of high concentrations of dissolved nutrients to streams as a result of precipitation flushing mobile solutes from nutrient-rich arable soils (Grose *et al.*, 2022). The maximum observed concentrations in the Impact sub-catchment reached almost 80 mg NO_3^- L^{-1} during an event in early March 2019 where it is likely that field drains were acting as preferential pathways for the export of nitrate-rich soil water (Cooper *et al.*, 2018). Our monitoring suggests that nitrate transfers may partly be controlled by antecedent conditions, with the highest observed concentration occurring after a relatively dry February with low API. Knapp *et al.* (2020) proposed that nitrate mobilisation observed in Q-C relationships under drier conditions may be due to evapoconcentration of solutes within the soil. Such high concentrations in surface waters are well above the limit of 50 mg NO_3^- L^{-1} set by the Nitrates Directive (91/676/EEC). The catchment falls within a nitrate vulnerable zone (NVZ) which attempts to limit the risk of N pollution from agriculture, however the issue persists due to significant stores of nitrate within the vadose zone (Ascott *et al.*, 2017).

Spatial sampling of stream water quality helped to identify nutrient source areas and determine the extent of instream biogeochemical processing and its potential links to NFM. During summer, downstream attenuation of nitrate was observed within the Impact sub-catchment, however this effect appeared to be negligible in the Control sub-catchment. Conversely, during winter, nitrate concentrations within each sub-catchment were almost always highest at their outlets. Part of the seasonal nitrate attenuation can be attributed to on-line (connected) pond features. NFM interventions within the study area were not explicitly designed and implemented with the aim of reducing diffuse N pollution and therefore nitrate did not form an extensive part of the overall monitoring campaign. However, intervention-scale monitoring of three small on-line ponds in the Impact sub-catchment found that these permanent ponds were capable of reducing nitrate concentrations from this first-order stream by an annual average of 5 % (Robotham *et al.*, 2021). The impact of the N removal effect of the ponds is harder to assess at the sub-catchment scale due to the influence of an unmonitored tributary which also contains two separate on-line pond features. However, even the nitrate attenuation from the on-line ponds and any natural instream attenuation are not sufficient enough to mitigate the effects of the significant agricultural N legacy in the catchment's soil, groundwater and streams. Basu *et al.* (2022) acknowledge the challenges posed by legacy stores of N which often result in time lags between the implementation of catchment conservation measures and water quality improvements. They propose that measures to control agricultural sources (e.g. nutrient management) should be used in conjunction with downstream measures such as ponds and wetlands that promote nitrate removal through denitrification and plant uptake. Designing pond and wetland features in such a way that promotes continual wetting will also help to create the anoxic conditions required for full conversion of nitrate to nitrogen gas (Negi *et al.*, 2022). Understanding the drivers of nitrate losses to watercourses will enable more strategic targeting of mitigation measures for different catchments (McAleer *et al.*, 2022).

The several permanently ponded offline NFM storage features that exist within the Impact sub-catchment are likely to be intercepting and denitrifying concentrated subsurface flow from the soil and field drains, thereby reducing some of the nitrate entering the stream via hyporheic exchange. It has also been suggested that instream woody features such as leaky barriers may enhance instream nitrate processing due to slower flows enhancing residence times and increasing potential for hyporheic exchange (Howard *et al.*, 2022).

Earlier research found that on-line ponds were more effective in removing particulate fluxes compared to dissolved constituents (particularly during storm events), thereby reducing the overall transport of P to a greater extent than N (Robotham *et al.*, 2021). Due to the biological significance of the N/P ratio, interventions that only mitigate P loading may have unintended consequences for downstream freshwater ecosystems if N remains present in relatively high quantities (Peñuelas and Sardans, 2022). Further research into the potential for different NFM interventions to attenuate dissolved nutrients (namely nitrate and phosphate) would benefit our understanding of the role of NBS in mitigating against eutrophication.

5.5.5 Implications for Catchment Monitoring and Management

Despite earlier intervention-scale monitoring demonstrating significant trapping of sediment in the offline and online NFM features (Robotham *et al.*, 2021; Robotham *et al.*, 2023), the BACI analyses were inconclusive in detecting a statistically significant reduction in sediment loading attributable to the implementation of interventions. This result becomes more plausible when considering some of the limitations of monitoring schemes and BACI configurations. Earlier research that undertook monitoring of catchment restoration in the Scottish Borders highlighted several difficulties associated with empirical monitoring studies and the BACI designs that are typically used for assessing success of interventions (Spray *et al.*, 2022). These difficulties included: the reliance on natural events that are hard to predict in both their timing and scale; the need to capture a diversity of event magnitudes in the pre-intervention period; and underlying variability in catchment conditions e.g. rainfall intensity, antecedent soil moisture. The high degree of commonality between these challenges and those experienced in our study suggest that multi-scale monitoring approaches are required to better understand the effects of interventions such as NFM or river restoration on catchment functioning, particularly in highly dynamic headwaters or large heterogeneous catchments. Examples of previous multi-scale monitoring efforts include the Pontbren catchment (Jackson, Wheater, *et al.*, 2008), and the Demonstration Test Catchments (DTC) (Defra, 2020). Detecting signals of change within data from small streams is made harder by uncertainty in sediment and nutrient flux estimations that can arise due to factors such as non-stationarity in stage-discharge relationships (Lloyd *et al.*, 2016c). Establishing robust and detailed monitoring programmes in such environments can prove challenging for numerous reasons, particularly given their typically hydrologically flashy nature and therefore the need for automated sampling systems (Harmel *et al.*, 2018). There is a clear need for further monitoring of NFM projects and the establishment of longer baselines (Short *et al.*, 2018). Baseline data that covers a wider range of hydro-climatic conditions allows for more robust BACI comparisons and helps to increase the chances of detecting the effects of mitigation efforts in catchment systems where signal-to-noise ratio is low. With NFM becoming more mainstream, there may be future opportunity to develop monitoring approaches that involve citizen/community science to more effectively crowd-source hydrometric or water quality data and improve data coverage, particularly in small ungauged catchments (Starkey *et al.*, 2017; Njue *et al.*, 2019). Detailed multiyear hydrogeochemical monitoring typically occurs on watercourses of a high stream order rather than on headwater tributaries draining small catchments. Headwater streams are considered to be vulnerable waters which are hydrologically dynamic and biogeochemically reactive systems that are essential to catchment resilience (Lane *et al.*, 2022).

There is a need to monitor hydrological processes at a low-order and intervention-scale to enable better interpretation of the potential large-scale impacts, particularly in a changing climate.

The observed differences in sub-catchment responses highlighted in our study demonstrate the need for better understanding of the hydrological functioning of headwater catchments in order to explain controls on sediment and nutrient transfers. Future NFM schemes aiming to deliver multiple benefits should consider selection of interventions for catchments based on their dominant hydrological pathways in order to effectively target pollutants that pose the greatest risk to water quality (Mellander *et al.*, 2015). Integrated management of flood risk and diffuse pollution is needed so as not to risk mitigating one at the unintentional expense of the other. Mitigation efforts are constrained by a paucity of data on diffuse pollution which also limits the usefulness of catchment modelling for management purposes (Harmel *et al.*, 2018). Catchment monitoring is required not only to help determine the success of mitigation measures, but also to assist in the identification of new sources of pollution and implement management decisions quickly in response.

Despite the monitored sub-catchments being equal in area and adjacent to one another, results show that they have differing hydrological regimes and responses to storm events. This means that the Control sub-catchment is somewhat of an imperfect comparator, thereby causing further difficulties in detecting intervention effects within the Impact sub-catchment. The uniqueness of catchments is an inherent theme within hydrological sciences and presents challenges across both empirical paired-catchment studies and when modelling processes (Beven, 2000). In the case of our study sub-catchments these challenges arose from the influence of sub-surface flows and the assumption that sub-catchments were hydro-geologically isolated. Research has found that groundwater boundaries and effective catchment areas are often underrepresented, therefore overlooking potentially significant inter-catchment groundwater flows of water and solutes (Azimi *et al.*, 2022; Schwamback *et al.*, 2022). Adjusting event rainfall totals based on the differences in sub-catchment geology and soil types within BACI GLMs sought to compensate for the contrasting hydrological regimes, but this approach did not account for any potential sub-surface transfers from neighbouring catchments. The underlying lithological characteristics of catchments also have complex interrelationships with other catchment parameters due to their effect on soil type and depth, vegetation type, land-use and cover (Bloomfield *et al.*, 2009). Further study to examine groundwater influences in small headwater catchments with underlying and adjacent carbonate geology may help to inform the targeting of NFM interventions to maximise their effectiveness at reducing flooding and attenuating diffuse nutrient pollution.

We recommend that evaluation of the efficacy of NFM (or similar NBS) interventions for diffuse pollution mitigation should adopt a weight-of-evidence approach using multi-scale monitoring i.e. at intervention and catchment scales. High-resolution stream monitoring data provide useful insight into water quality dynamics and potential pollutant sources; however this study demonstrates the difficulties in using this evidence alone to assess intervention effects in responsive headwater catchments. Our results highlight the sensitivity of suspended sediment transport to changes in catchment conditions (e.g. rainfall, land cover) in a highly modified agricultural landscape.

5.6 Conclusions

This study set out to test if it was possible to detect the effect of the successive implementation of NFM interventions on storm event suspended sediment dynamics in a small headwater catchment. A BACI approach using GLMs was unable to detect statistically significant differences in hydrological and hydrogeochemical responses as a result of the presence of NFM interventions. This approach sought to control for variability in rainfall, antecedent conditions, and bare soil cover between events, as well as differences in geology and soil between the Control and Impact sites. Despite the high-resolution data spanning multiple years, a lack of storm events (particularly those of higher magnitude) in the *before* period limited the statistical power of the BACI analyses. Analysis based on the frequency and duration of Q and SSC exceedance above multiple thresholds also highlighted the contrast in the hydrological response of the Impact sub-catchment between the pre-NFM period and subsequent years. Less than average rainfall in the monitoring period prior to interventions compared to above average rainfall experienced post-intervention resulted in contrasting catchment conditions that play a large role in controlling sediment delivery. The timing of rainfall and catchment wetting in combination with the predominantly arable land-use (in the Impact sub-catchment) was shown to have a strong effect on sediment and P delivery that masked the effect of the NFM interventions. This highlights the need for multiyear baseline monitoring data to capture a greater degree of variability in hydro-climatic and catchment conditions that allows for a more robust before-after comparison. Post-intervention monitoring may need to span significantly longer timescales to take into account the effect of lag times in water quality responses to NFM. Catchment scale data are useful in putting intervention scale data into context, but they cannot solely be relied upon for determining the success of mitigation where highly dynamic diffuse pollutants are concerned; therefore a weight-of-evidence approach should be adopted in such evaluations. Further empirical research is needed to better disentangle the effects of NFM from other influences and determine its effectiveness in providing hydrological and biogeochemical buffering against the impacts of climate change and intensive food production.

5.7 Acknowledgements

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Chapter 6 Conclusions

6.1 Introduction

This research has used intervention and catchment-scale monitoring to determine the effects of multiple NBS on sediment dynamics and water quality, adopting a weight-of-evidence approach. The study provides insight into potential solutions for the mitigation of diffuse agricultural pollution in lowland headwater catchments.

Section 6.2 provides a summary of the key findings from this research and their implications for catchment management and the use of NBS. Section 6.4 identifies key strengths and limitations of this research and discusses recommendations for future research directions in this field. Concluding remarks are presented in Section 6.5.

6.2 Summary of Findings

6.2.1 Hydrological Regime, Suspended Sediment and Nutrient Dynamics of the Littlestock Brook

High temporal resolution (5-minute) monitoring of stream water levels and discharge over multiple years showed contrasting hydrological regimes between the two 3.4 km^2 sub-catchments of the Littlestock Brook (Section 5.4.1). The Control sub-catchment exhibited flashy behaviour, with rapid rising limbs and high peaks during storm events compared with the Impact sub-catchment. The maximum recorded discharge in the Control sub-catchment reached 2519 L s^{-1} compared to only 847 L s^{-1} in the Impact. Outside of storm events, baseflows were considerably higher in the Impact sub-catchment which was found to have a BFI of 0.75 compared to only 0.33 in the Control sub-catchment. These observed differences are consistent with soil types and underlying geology, with the Control sub-catchment having a higher proportion of slowly permeable clay-rich soil in contrast to the greater extent of shallow well-drained soil in the Impact sub-catchment. The hydrological regimes are therefore likely to have been driven by different processes, with higher potential for rainfall run-off in the Control sub-catchment, and greater sub-surface flow in the Impact sub-catchment.

The influence of the sub-catchment hydrology was reflected in the suspended sediment response to storm events, with a higher mean concentration of 770 mg L^{-1} in the Control sub-catchment compared to 493 mg L^{-1} in the Impact sub-catchment (prior to the implementation of NFM). The concentrations of TP also followed this pattern due to the dominance of sediment-bound particulate P transported in flow events. Rainfall intensity was found to be a key driver of peak SSC, whereas total rainfall was more important in determining the event sediment load. High SSC and TP during times of high flow meant that storm events contributed significantly to overall sediment and P yields from both sub-catchments. In contrast, nitrate concentrations were typically higher in the Impact sub-catchment and showed dilution at higher discharges during winter storm events. This Q-C relationship highlighted how nitrate sources were associated with baseflows and its transfer was likely enhanced through sub-surface pathways in the shallow permeable soils and via groundwater in the underlying limestone.

Implications:

The rapid response of streamflow and pollutant concentrations to rainfall in both sub-catchments illustrates how there is potential for NBS to mitigate run-off and associated overland pollutant transfers in impermeable lowland catchments. The high nitrate concentrations observed during baseflows also demonstrate potential for NBS that target slow flows to remediate chronic N pollution in agricultural catchments. The observed differences between sub-catchment hydrological responses despite their matching size exemplifies the importance of considering the uniqueness of place when implementing NBS and related monitoring programmes. Such differences create challenges in interpreting the results of paired-catchment studies (as highlighted in Section 6.2.4).

6.2.2 Sediment and Nutrient Retention in Online Ponds

Detailed monitoring of online pond features (Chapter 3) explored their functioning under different flow conditions. Online ponds were observed to function as both sources and sinks of sediment and nutrients depending on hydrological conditions. Baseflow water sampling from the inflow and outflow of the set of three ponds showed statistically significant removal of SRP, nitrate, SSC, and VSC. Between March 2019 and March 2020, the mean reduction in SRP concentration was 29 %, nitrate 5 %, SSC 32 %, and VSC 40 %. The greatest reductions in dissolved nutrient concentrations occurred during summer when flows were lowest, residence times longest, and removal was likely being driven by plant uptake during the growing season when cover of *Typha latifolia* was highest. Removal efficiency for both SSC and VSC was high during baseflows, however the observed concentrations were significantly lower in comparison to those observed during storm events. During storm events, the three ponds were able to sequentially reduce peak concentrations and loads of suspended sediment and P; however, during a particularly intense event, resuspension of deposited sediment resulted in net loss from the system.

Siltation devices were also used to examine the accumulation rates of sediment and P over longer periods (approximately monthly) within each of the ponds. The overall trend showed a decrease in accumulation rate with increasing distance downstream along the pond sequence. Between August 2019 and March 2020, the ponds accumulated 0.306 t ha^{-1} sediment from the 0.3 km^2 contributing area. During this period, total sediment accumulation across all three ponds was estimated to equal 7.6 % of the total suspended yield leaving the 3.4 km^2 Impact sub-catchment (as monitored at the outlet). Accumulated sediment properties were found to vary both temporally (between sampling periods) and spatially (between ponds). Sediment P content peaked in September and then decreased during the winter months. This was thought to be due to the timing of plant-derived inputs and processing of organic matter (e.g. macrophyte decomposition) as the P content of sediment was found to be positively associated with organic matter content. On average, sediment P concentrations showed a decreasing trend downstream along the pond sequence which was also linked to differences in organic matter. In terms of physical properties, the ponds were able to filter out larger particles most effectively due to these particles having higher settling velocities. The sand and silt content of the deposited sediment varied significantly between the three ponds, however the clay content showed smaller differences between ponds. Both clay and silt content showed an increasing trend downstream along the pond sequence, whilst sand showed a decrease. This was as a result of suspended sand particles being preferentially deposited when flowing into the first pond in the sequence where both flow velocity and turbulence decreased rapidly.

Implications:

Findings from storm event monitoring demonstrate the how online pond features are capable of capturing pollutants from the upstream agricultural catchment, however they are also at risk of releasing particulate pollutants under high flow conditions in rapid response to rainfall. In terms of stream ecology, this may not present so much of an issue given that pollutant flushing is more likely to occur during winter when high flows have greater capacity to carry and dilute pollutants. In smaller flow events that are more typical of summer, online ponds are able to better protect downstream habitats during such ecologically sensitive times of year. However, in light of an increased likelihood of extreme rainfall events in summer under future climate change, this risk should still be taken into account when designing, implementing, and managing NBS. Baseflow monitoring demonstrated how online ponds may mitigate dissolved nutrient pollution, particularly during summer when stream ecosystems may be at greater risk from high dissolved concentrations in low flows.

To reduce the risk of becoming net sources of pollution, it is recommended that online ponds undergo maintenance to remove accumulated sediment on a biennial basis. For online ponds that exist in series, maintenance requirements for each pond should be adjusted accordingly, with less frequent sediment removal being necessary for those located further downstream. Accumulated pond sediment derived from a largely arable contributing area showed good suitability for agricultural application as a soil conditioner, owing to high organic matter and silt content, and thus good water holding capacity. Policy that incentivises such NBS needs to explicitly consider their management and longevity rather than purely focussing on their implementation. Encouraging the re-use of accumulated sediment will help to integrate NBS and their maintenance into farm business, thereby ensuring their long-term sustainability. This will also help shift thinking towards sediment and nutrients as valuable resources and part of a circular economy rather than as pollutants that are an accepted consequence of farming. Policy should consider monitoring of features to allow for reporting of their effectiveness. Whilst it would be unfeasible to carry out widespread detailed monitoring, simple metrics could be used to record performance over time and thereby help evaluate success. For example, a record of the accumulated sediment depth, the maintenance regime (e.g. frequency and volume of sediment removed), and dimensions of the feature (e.g. surface area) could be maintained. Reporting of the agricultural value (i.e. nutrient content) of accumulated sediment could be integrated into farm businesses through agronomic assessments which already include soil sampling.

Optimising the locations of connected pond features is key to their effectiveness. Online ponds present a good land management option in locations where their footprint is not detrimental to agricultural activities, and could be used in combination with other NBS such as arable reversion and woodland creation. Given that arable land is the second highest land cover in Great Britain, accounting for 23 % of land area as of 2020 (UKCEH, 2022b), it is likely that online ponds could have wide applicability, particularly in similar lowland catchments that cover most of southern and eastern England. They could also be applied in other agricultural contexts such as improved grassland to treat run-off from densely stocked pasture where soils are likely to be degraded and vulnerable to erosion (Bilotta *et al.*, 2007; Marshall *et al.*, 2009). However, it is important to note that small online features are only suitable for headwaters and low stream orders where they are able to cope with relatively small inflowing volumes. Where streams are located further down in a catchment and drain larger areas, it would be more appropriate to implement a NBS such as ‘Stage Zero’, an approach which aims to reconnect watercourses with their floodplains and allow natural processes to restore fluvial systems to conditions resembling pre-anthropogenic disturbance (Flitcroft *et al.*, 2022). Implementation of online ponds in catchments where surface water is highly connected to groundwater could risk transferring pollutants to groundwater (Dzakpasu *et al.*, 2012) or reduce the efficacy of nutrient removal (Kill *et al.*, 2018; Gordon *et al.*, 2021); therefore local hydrogeology should be taken into consideration.

In practice, such ponds may also be implemented for purposes other than diffuse pollution mitigation. For example, they may be used for habitat creation in order to meet wider policy objectives (e.g. Biodiversity Net Gain). Implementation and management of these NBS should be holistic and aim to maximise the delivery of multiple benefits, thereby contributing to multiple national and international policy goals (e.g. the United Nations Sustainable Development Goals for clean water and climate action). Ponds can be maintained for diffuse pollution mitigation and flood storage by regularly removing trapped fine sediment, however this maintenance should be carried out in a way that is beneficial for biodiversity. It is recommended that when undertaking desilting, a portion of the pond is left undisturbed to allow re-colonisation following maintenance. A recent report on integrated sediment management in the context of the WFD demonstrates how an integrated approach to managing sediment is key to meeting multiple environmental policy objectives such as those under the Floods Directive and the Habitats Directive (Ausili *et al.*, 2022). The authors recommend applying NBS and the principles of adaptive management to address sediment issues. Future policy on NBS (e.g. ELM schemes) therefore needs to be agile and respond to evidence on their effectiveness to ensure their long-term sustainability as catchment management measures.

6.2.3 Sediment and Nutrient Retention in Offline Ponds

The majority of pond features implemented in the study catchment were offline, having a primary function of flood risk reduction; chapter 4 investigated their ability to provide multiple benefits for water quality improvement through the mitigation of diffuse pollution. Offline ponds were observed to have highly variable rates of sediment and nutrient accumulation depending on their design and configuration. The mass of sediment accumulated in the offline features (ranging from 28 to 355 m^2 in area) varied greatly (between 0.2 and 20.1 tonnes) during the 2-3 years since their construction. In total, the offline features accumulated 47.9 tonnes of sediment, and when combined with the accumulation in the online ponds, this made up the equivalent of 14.7 % of the total suspended sediment yield from the catchment over the same period. Offline features accumulated 72.5 kg of P, and 2.3 tonnes of POC, which combined with the online ponds was equivalent to 9.5 % and 7.5 % of the TP and POC yields respectively.

Hydrological connectivity of the offline features was found to be a crucial factor in controlling the rate at which they accumulated sediment and nutrients. Greater accumulation rates were observed within features that were designed to fill via overbank flows during storm events as a result of instream leaky barriers. Conversely, features that were hydrologically isolated from the stream channel and drained smaller areas of land accumulated comparatively little material. The shape of the features was also found to influence their accumulation rate, with length-to-width ratios explaining 42 % of the variation in sediment accumulation rate, and 54 % of the variation for P. Greater length-to-width ratios resulted in higher accumulation rates, likely due to increasing residence times within features and thereby enhancing settling of particles. For the offline features, the accumulation rates measured in the 2-3 years since their construction suggests that their potential for floodwater storage is only likely to be significantly reduced in the medium term, after ~10 years of operation. Therefore maintenance to remove sediment from offline features for this purpose would only be required beyond 10 years. However, it is acknowledged that the succession of vegetation within these features may enhance accumulation rates over time, and maintenance requirements may become more frequent in future.

Enrichment ratios were calculated to enable comparisons between the topsoil and trapped sediment within offline features, finding that eroded material was enriched in both P and OC in the majority of features. On average, offline features had a lower P enrichment ratio compared to online features, however OC enrichment was found to be similar between the feature types. In terms of sediment and soil grain size composition, on average offline features were most enriched in clay particles, in contrast to online features which were highly enriched in sand particles. These findings highlight some of the potential differences in the functioning of storage-focussed NBS, with offline features more likely to effectively capture the fine sediment fraction.

Implications:

These findings demonstrate the potential for significant masses of sediment and nutrients to be trapped by offline NBS despite them having a relatively small footprint of <1 % of the total catchment area. It is therefore recommended that field corner ponds such as these should be integrated into arable landscapes where the risk of soil erosion and run-off are high. The variation in sediment accumulation between the features highlights the need to suitably locate interventions to intercept critical source areas in the landscape. For example, interventions placed on steep arable fields adjacent to watercourses would provide greater value than those located on gently sloping permanent grassland far from the stream network. Offline interventions are also likely to be effective at attenuating run-off and diffuse pollution in upland catchment contexts. Given the typically higher potential for run-off generation in upland settings with steep slopes and higher rainfall totals, it is necessary to consider the total storage capacity required to cope with these conditions. Interventions should be designed to suit different catchment contexts and hydrological responses, allowing them to function without causing unintentional risks (e.g. exceeding storage capacity and rapidly releasing large volumes of water/sediment).

Given the importance of hydrological connectivity in controlling accumulation rates, storage-based NBS should also be designed in conjunction with other features (e.g. instream leaky barriers) to maximise pollutant trapping potential alongside the benefit of increased flood attenuation. Based on the measured properties of the accumulated sediment within features, it is likely that this material, once removed for maintenance purposes, holds value for redistribution on arable land. Given their ability to capture soil nutrients, storage-based NBS can therefore help to contribute to improving the sustainability of farm businesses (as was discussed in Section 6.2.2 with respect to online NBS).

Despite their effectiveness at trapping sediment, it is recommended that offline ponds should be implemented as part of a wider suite of measures rather than in isolation to address diffuse pollution more holistically. Land managers should adopt the source-pathway-receptor model to consider how pollution can be mitigated through combining interventions that target different stages and processes. For example, pollution can be addressed at source using innovative land and soil management practices (e.g. herbal leys, controlled field traffic) that aim to increase soil organic matter content and improve soil structure. Such practices have been shown to significantly increase soil porosity (Trill, Blake, *et al.*, 2022), and are therefore likely to reduce the risk of run-off generation and erosion. Offline features should be used in addition to source measures as a second line of defence to intercept pathways of diffuse pollution. These edge-of-field interventions are likely to play an important role during times when fields are particularly vulnerable to erosion (e.g. post-harvest). This strategic approach to diffuse pollution management prioritises keeping soil and nutrients in situ, but allows any mobilised sediment to be trapped by offline features and then returned to fields following maintenance. Policies such as the ELM schemes should therefore incentivise and encourage the use of interventions in combination.

6.2.4 Monitoring Sub-catchment-scale Effects of NBS

The high temporal resolution monitoring of streamflow, SSC, and TP at the two sub-catchment outlets of the Littlestock Brook over multiple years enabled the potential effects of the NFM scheme to be studied at a larger spatial scale. Chapter 5 assessed sub-catchment-scale impacts of the NFM over time using a BACI approach and highlighted the challenges of using this monitoring design in a complex agricultural landscape.

In the year prior to the implementation of interventions, yields of SS and TP from the Impact sub-catchment were found to be greater than yields from the Control sub-catchment. In the years following the implementation of NFM, yields were higher from both sub-catchments, however they were consistently higher from the Control. When looking at data extracted from storm events, the mean event peak discharge and SSC were found to be higher in the Control sub-catchment both before and after the implementation of NFM interventions. This was also reflected in the event SS loads, with the post-NFM mean load being almost one tonne higher from the Control sub-catchment. Analysis of the frequency and duration of Q and SSC pulses above selected thresholds in the Impact sub-catchment found that there were notably fewer pulses in the pre-NFM period compared with subsequent years, reflecting differences in hydrometeorological conditions.

Given the observed differences between the sub-catchments (as discussed in Section 6.2.1), and the additional influences of variability in rainfall, these before-after NFM comparisons did not allow definitive conclusions to be drawn regarding the effect of NFM. Modelling storm event responses of stream discharge and sediment concentrations/loads as functions of event rainfall, Antecedent Precipitation Index, and exposed soil cover in a GLM framework aimed to overcome potentially confounding factors and identify differences in these relationships in the presence and absence of NFM and over time. The GLM showed that the relationship between the predictor variables and event peak SSC was significantly different between the Control and Impact sub-catchments during the final phase of NFM implementation. The relationship during this phase showed considerably elevated SSC peaks of up to 2000 mg L^{-1} in the Impact sub-catchment during low rainfall events of less than 10 mm which contrasted to the preceding phases where SSC remained below 1000 mg L^{-1}. Analysis suggested that any effect of NFM interventions and their added storage within the sub-catchment was masked by additional sources and inputs of sediment caused by hydrometeorological variability (e.g. rainfall intensity and timing) and changing land cover. Disturbance caused by the construction and addition of further NFM interventions throughout the post-NFM monitoring phases is also likely to have contributed to these difficulties in detecting the impact of NFM on sediment losses at the sub-catchment scale. This hypothesis is supported by previous research that observed increased sediment supply to a lowland river during the period following restoration works (Sear *et al.*, 1998). Hysteresis behaviour indicated that the likely sources of suspended sediment differed between sub-catchments and storm events, and were controlled by a combination of factors including land-use, lack of crop cover, and antecedent conditions.

Implications:

These findings demonstrate the highly dynamic nature of suspended sediment transport in small headwater catchments where responses to rainfall can vary significantly as a result of both natural variability and anthropogenic influences such as agriculture. Evidence of elevated sediment loading during periods with extensive areas of exposed soil emphasises the importance of additional mitigation measures that directly target source areas (e.g. cover crops) to reduce losses from soil erosion and also from soil disturbance caused by the construction of interventions themselves.

The inconclusive results of the BACI analysis highlight the need to monitor the hydrological impacts of NBS over longer timescales and at multiple spatial scales to better determine the efficacy of such interventions. More holistic monitoring that includes measurements at different scales in the landscape would help to better understand how NBS are functioning within the wider catchment system. Applying a source-pathway-receptor approach to monitoring would address this and allow the effects of NBS to be more easily detected, either at an intervention-scale, in-channel, or further downstream at the catchment outlet. Recommendations for monitoring NBS are discussed further in Section 6.3.

Despite the paucity of evidence to support the benefits of NBS for water quality at a sub-catchment scale (Chapter 5), policy should still promote the use of NBS based on the evidence of local-scale benefits (Chapter 3 and Chapter 4).

6.3 Recommendations for Future Research

The instream monitoring carried out during this research demonstrated the highly variable nature of streamflow and the transport of suspended particulates and solutes between hydrological events and years. The difficulties in detecting the effects of NBS using a BACI approach were exacerbated by this variability due to a dry and relatively short pre-intervention period followed by above average rainfall in the post-intervention period. Chapter 5 emphasises the importance of establishing robust pre-intervention monitoring data to improve the ability of BACI analyses to yield definitive results. Given the gradual phased implementation of the Littlestock Brook NFM scheme, the monitoring was not able to cover a period of stability following the disturbance from the construction of interventions and then any potential acclimatisation. Future research should seek to monitor over longer timescales to capture variability in the period pre-intervention, in the potential acclimatisation period immediately following interventions, and then over the longer term after acclimatisation. Long-term monitoring programmes should be favoured over shorter-term projects, and large-scale studies should be complimented with intervention-scale evaluation; future funding mechanisms for NBS monitoring should reflect these priorities. However, it is recognised that the ideal monitoring design may not always be feasible given potential time, planning, and funding constraints. In light of this, it is recommended that multi-scale monitoring should be adopted wherever possible, thereby allowing a weight-of-evidence approach to be taken to when answering research questions on the efficacy of catchment-based interventions.

There is still a need to more comprehensively understand how the intervention-level effects of trapping pollutants relate to the catchment scale, particularly considering the movement and storage of sediment through the landscape and stream network from source to sink. Detailed monitoring of sediment yields as well as storage within different parts of the catchment (e.g. riparian zones, channel bed) would help to provide greater insight into the movement, residence time and fate of diffuse pollutants. For example, the application of techniques such as sediment source fingerprinting (Pulley and Collins, 2018) and rare earth element labelling/tracing (Govenor *et al.*, 2021) in catchments where NBS are established would be useful in assessing the impact of interventions on pollutant transfers and potential changes in sediment provenance and dominant pathways over time.

Chapter 4 highlights how instream leaky barriers can function to enhance overbank flows during storm events, and Chapter 5 considers their potential for enhancing the attenuation of dissolved nutrients through increasing residence times. However, there is still a considerable gap in the knowledge surrounding the full effects of leaky barriers on water quality and how these may compare to natural woody debris. The leaky barriers in the Littlestock Brook catchment were designed to allow clearance below for baseflow and fish passage, however many other types of design exist and their impacts may differ as a consequence. The implementation of leaky barriers is widespread, with over 3000 across England being registered as NFM assets in the Catchment Based Approach (CaBA) database of the 79 Defra-funded NFM projects, making them the most commonly used NFM intervention (The Rivers Trust, 2019). This means that there is considerable opportunity for further research into their effects on instream biogeochemical processes (e.g. denitrification) across contrasting catchment types and different designs of leaky barrier. This research may be most pertinent in catchments where water resources are particularly vulnerable to nitrate pollution, for example in chalk catchments or intensive arable catchments (Jackson, Browne, *et al.*, 2008). Previous research suggests that in-channel NFM measures provide minimal flood reduction benefit in groundwater-dominated catchments (Barnsley, 2021), however future studies should not overlook their value as NBS to deliver wider ecosystem services.

This thesis focussed on providing detailed empirical evidence on the efficacy of NBS at a local scale, looking at two 3.4 km^2 second and third stream order sub-catchments. The issue of scale remains a notable gap in the research surrounding NFM. Dadson *et al.* (2017) note that simple extrapolation of local-scale benefits does not necessarily equate to an additive effect at a larger spatial scale, instead suggesting that local benefits may be diminished downstream by the channel network. Future work should build upon the findings of this thesis and draw from the growing wider evidence-base to examine potential impacts of more widespread implementation of NBS. Currently planned large-scale efforts to enhance the environment such as Defra's Landscape Recovery scheme provide opportunities to help address this research gap through monitoring multiple benefits. One of the scheme's initial focus areas is the restoration of waterbodies, rivers, and floodplains, and the aim to improve water quality, biodiversity, and resilience to climate change (Defra, 2022). Empirical research into the multiple benefits resulting from such large-scale and long-term initiatives would help to build scientific consensus and inform future environmental policy on NBS. In addition to monitoring, future research should aim to upscale the effects of NBS in order to test their efficacy under different scenarios based on potential land-use and climatic changes. Spatial modelling should help to determine where NBS measures will be most suitable for mitigating pollution, and to what extent they need to be adopted in order to deliver cost-effective water quality improvements that can meet legislative targets.

6.4 Concluding Remarks

This study has contributed towards an improved understanding of the potential of NBS to mitigate diffuse pollution in lowland agricultural catchments. This is important given that such catchments represent a significant proportion of land cover in the UK. The research suggests that NBS aimed at increasing catchment water storage also provide multiple benefits through their ability to intercept diffuse pollution and trap sediment and nutrients.

Detectable effects of NBS on sediment and nutrient capture at the intervention-scale were observed. These effects exhibit high temporal variability and are dependent on a number of factors, both intrinsically in terms of the designs of interventions themselves, but also due to changes in hydrometeorological conditions. The monitoring of NBS at an intervention-scale can provide valuable insight into their functioning and efficacy, which also may help inform suitable management strategies. The appropriate maintenance of interventions is important for their ability to deliver benefits for both flood storage and sediment and nutrient retention.

Instream monitoring at the catchment-scale provides insight into potential pollutant sources and hydrological pathways. However, at this scale it is challenging to isolate the effect of interventions on sediment loading, as demonstrated using a BACI study design in a complex landscape subject to land cover changes on a seasonal and annual basis. Monitoring NBS on longer timescales is therefore important for capturing greater hydrological and agricultural variability and helping in the detection of trends in sediment and nutrient loads resulting from the presence of catchment interventions. This is also needed to avoid the effects of disturbance in the initial period following NBS implementation. The variability of sediment and P losses in the Littlestock Brook catchment demonstrates the sensitivity of intensively managed arable headwaters to changes in hydrometeorological conditions. Targeting of NBS in these areas would help to build greater resilience to the impacts of future hydrological extremes, particularly given the context of the changing climate.

The evidence-base on NBS would benefit from further research to assess the efficacy of a wider variety of interventions across different settings to test the transferability of these findings to other catchment typologies. In addition, study into the impacts of NBS at larger spatial scales and under longer timescales to capture more extreme events is needed to fully understand the implications for catchment hydrology, biogeochemistry, and ecology.

Appendix A Littlestock Brook Natural Flood Management Pilot: Hydrological and Water Quality Monitoring and Analysis Report

The following report was produced by UKCEH as part of the monitoring for the Littlestock Brook NFM pilot scheme. The data collection and analyses undertaken for this PhD thesis contributed considerably to all sections of the report. Report writing was jointly carried out by E. Trill and J. Robotham.

UK Centre for Ecology & Hydrology

Hydrological and water quality monitoring and analysis report

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A.1 Executive Summary

The Natural Flood Management trial:

The Littlestock Brook Natural Flood Management (NFM) trial was a 5-year project that ran from 2016 to 2021. Key objectives were to reduce flood risk to Milton-under-Wychwood and enhance the river environment. Through partnership working the Environment Agency (EA) collaborated with Wild Oxfordshire, the Evenlode Catchment Partnership (ECP), Bruern Estate and the local community to deliver NFM measures. Through two Doctoral Training Partnership PhD studentships UKCEH has undertaken a detailed monitoring campaign to assess the effectiveness of the measures on reducing flood flows and improving water quality.

Monitoring report:

This report describes the monitoring network, observational methods, equipment adopted, the data processing and analysis undertaken.

Implementation of the Natural Flood Management trial:

The trial has been implemented over five phases between March 2017 and February 2021. In March 2017, 12 woody dams were installed in the heavily incised northern tributary channel immediately upstream of Milton-under-Wychwood. The next three phases of delivery (2018-2020) implemented interventions in the upper catchment, including soil management measures on steep clay slopes and along overland flow pathways; creating nutrient retention ponds and sediment traps in fields; constructing 15 riparian field corner bunds to store over-land run-off; and installing a further 15 in-channel, bank-full woody dams. In addition, 100 m of watercourse was de-culverted and 230 m of new watercourse was created. A Forestry Commission Woodland Grant scheme delivered 14.4 ha of new riparian woodland, which aims to improve interception of rainfall and run-off and sequester carbon over time. Phase 5 of the trial was delivered in 2020/21 and included additional retention pond creation, further riparian tree planting and 900 m of field edge nutrient trapping swales.

Monitoring established:

A detailed multi-scale monitoring network was established to measure precipitation inputs, and water quantity and quality. Observations were made at the intervention scale as well as in streams leaving the catchment. The monitoring of two sub-catchments of equal area allowed for a partial before-after control-impact (BACI) experimental set up.

In December 2016 two instream water quantity and quality monitoring stations were established in the tributary draining to the Heath. In January 2017 a similar station was established on the other tributary. Water levels were recorded and later converted to river flows, using a rating curve determined from manual flow gaugings. Water quality measurements included suspended sediment and nutrient concentrations. Continuous suspended sediment concentrations were estimated from monitoring turbidity and calibrating it to suspended sediment, using data from water samples. Instantaneous nutrient and suspended sediment concentrations were determined from samples collected either manually or using automatic water samplers.

Two rain gauges were installed during 2019 to observe precipitation inputs.

Thirteen water level sensors were installed in bunds and ponds to enable estimates of water storage when combined with topographic survey data.

Multi-parameter sondes were installed in online ponds and at specific locations in streams to observe their water quality.

Data and their availability:

Total data coverage for the monitoring period is over 90% across all sensors.

All data (raw, and where available quality controlled and processed) will be made available on the NERC Environmental Information Data Centre.

Flow, turbidity, suspended sediment concentration and total phosphorous data for the three main stream monitoring sites are already available for the period 2017-2021 on the EIDC (https://doi.org/10.5285/9f80e349-0594-4ae1-bff3-b055638569f8).

Data analysis:

Specific analysis reported include the following:

Daily and annual rainfall

Annual river flows

Annual fluxes of suspended sediment and nutrients

Sediment and nutrient accumulation in bunds

Sediment and nutrient attenuation in online ponds

Chapter 6

Water storage in bunds and estimated reductions in catchment outlet peak flows during selected events

Monitoring evaluation:

Data were evaluated using a number of approaches at multiple spatial scales in order to determine the effect of the NFM interventions. Isolating the effect of the NFM from natural variability was challenging purely using an experimental BACI approach, particularly as the catchment interventions were incrementally added throughout the monitored period. Intervention-scale monitoring (e.g. continuous measurements of water depth/volume within flood storage features) provided us with evidence of the effectiveness of interventions. The intervention-scale monitoring data were used to enable estimation of the effect (e.g. reduction in flood peak) downstream at the flood receptor.

Contents

A.2 Introduction

A.2.1 Littlestock Brook

The Littlestock Brook is a 16.3 km^2 sub-catchment of the River Evenlode catchment (430 km^2); located in the upper reaches of the Thames basin in West Oxfordshire, United Kingdom (Figure 6.1). The Evenlode catchment lithology is dominated by the Great Oolite Group, consisting of mudstone and fine-grained limestone. The Littlestock Brook sub-catchment on the west of the Evenlode is mostly underlain by the Lias Group; consisting of clays, mudstones and limestones (Robotham et al., 2021).

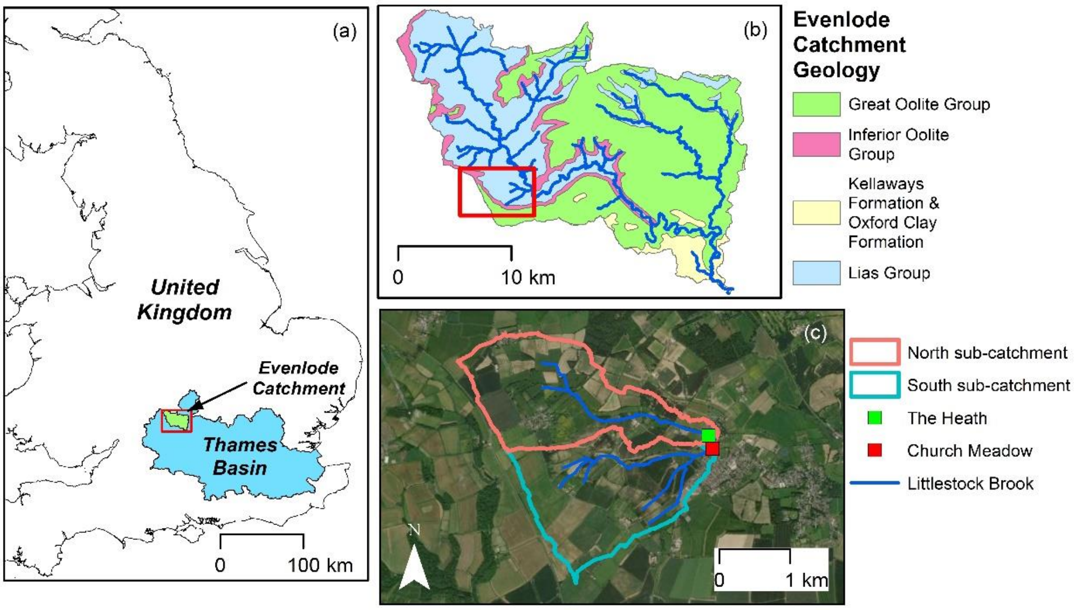


Figure 6.1: Locations of (a) the Evenlode catchment (green) within the Thames basin (blue); (b) the downstream catchment (outlined in red) within the Evenlode catchment and its geology; (c) the north and south sub-catchments with sub-catchment outlets marked. Adapted from Robotham et al. (2021).

The NFM trial in the Littlestock brook sub-catchment has been implemented and intensely monitored upstream of the Milton-under-Wychwood flood receptor between 2016 and 2022. Milton-under-Wychwood is at the confluence site of two tributaries draining the upstream study area that is comprised of two predominantly rural sub-catchments, each 3.4 km^2 and referred to as North and South in this report. The North sub-catchment consists mainly of arable land and permanent improved grassland used for grazing cattle and sheep, while the South sub-catchment is largely arable (Table 6.1).

Table 6.1: Land use of study site sub-catchments (% coverage), extracted from UKCEH Land Cover Map 2015.

North		**South**	
Broadleaved woodland	1.55	Broadleaved woodland	0
Arable and horticulture	60.59	Arable and horticulture	76.25
Improved grassland	34.66	Improved grassland	16.61
Calcareous grassland	0.66	Calcareous grassland	5.51
Suburban	2.54	Suburban	1.62

The elevation of the study area ranges from 103 m to 202 m, with an average slope of 6.4 % (Robotham et al., 2021). The area receives an average annual rainfall of 809.6 mm and experiences an average annual minimum and maximum temperature of 5.9 °C and 13.4 °C respectively (Met Office, 2022). Both sub-catchments have a low baseflow component of total stream flow, but the contribution of groundwater to river flow is greater in the south catchment. This is indicated by a higher base flow index of 0.75 relative to 0.33 in the North sub-catchment (Table 6.2). The Richard-Baker flashiness index indicates how quickly a stream increases and decreases during storm events, using changes in daily flows relative to average annual flows. The North sub-catchment has a higher flashiness index of 0.40 relative to the south sub-catchment index of 0.16, indicating that the short-term response to run-off events is faster in the North sub-catchment.

Table 6.2. Catchment and flow properties over the period January 2017 – April 2022 (Adapted from Robotham et al. 2022 (manuscript in preparation))

Catchment property		Sub-catchment	
		North	South
Flow (L s^{-1})	Q_{mean}	32.5	60.7
	Q_{50}	8.6	53.0
	Q_{10}	93.3	111.9
	Q_{1}	248.0	214.4
BFI (base flow index)		0.33	0.75
RBI (Richard Baker flashiness index)		0.41	0.16
Bedrock geology (%)	Limestone	30.0	45.0
	Mudstone	49.6	38.8
	Siltstone & mudstone (interbedded)	20.4	16.2
Average slope (%)		6.9	5.8

A.2.2 Purpose of the study

The Evenlode catchment has few significant settlements, with the rest of the catchment's population largely dispersed into many small towns and villages. Several of these settlements are in the upper and middle Evenlode and are prone to flooding, including the Wychwoods (Milton, Shipton and Ascott) (Old et al., 2019). Four properties flooded in 1990 and 1998, and 318 properties suffered fluvial flooding in July 2007. After the 2007 flooding, property level flood mitigation measures and modifications to a flood storage area (FSA) were installed to reduce the flood risk from the Littlestock Brook in Milton-under-Wychwood.

As there are a relatively small number of properties vulnerable to flooding in the Littlestock Brook sub-catchment, an engineered flood mitigation scheme could not be justified on a cost-benefit basis. As the community remained vulnerable to flooding after 2007, the Environment Agency (EA) collaborated with Wild Oxfordshire, the Evenlode Catchment Partnership (ECP), Bruern Estate and the local community to trial Natural Flood Management (NFM) measures. The trial was implemented by establishing an integrated catchment partnership approach with a working group including these key organisations, local communities and landowners to develop cost-effective, sustainable NFM solutions.

Phase 1 of NFM measure implementation began in March 2017, with installation of 12 woody dams in the heavily incised northern tributary channel immediately upstream of Milton-under-Wychwood, to reduce the transport of coarse bed material restricting flow conveyance (Table 6.3). The next three phases of delivery (2018-2020) implemented interventions in the upper catchment, including soil management measures on steep clay slopes and along overland flow pathways; creating nutrient retention ponds and sediment traps in fields; constructing 15 riparian field corner bunds to store over-land run-off; and installing a further 15 in-channel, bank-full woody dams. In addition, 100 m of watercourse was de-culverted and 230 m of new watercourse was created. A Forestry Commission Woodland Grant scheme delivered 14.4 ha of new riparian woodland, which aims to improve interception of rainfall and run-off and sequester carbon over time.

Figure 6.2: (Left) Woody dam upstream of (P9, Figure 6.3) in the South sub-catchment; (Middle) On-line pond and wider tree planting (OLP10) in the South sub-catchment; (Right) Corner bund and flood storage area (P5) in the South sub-catchment, surrounded by tree planting.

Phase 5 of the trial was delivered in 2020/21 and included additional retention pond creation, further riparian tree planting and 900m of field edge nutrient trapping swales.

The primary NFM measure was the construction of field corner flood storage bunds. The leaky woody dams divert flood flows into the scrapes and field corner flood storage areas (FSA), which then intercept overland run-off pathways and temporarily store high flows from the brook. These NFM measures provide an approximate total of 30,000 m^3 of temporary storage across the whole NFM trial area. The FSAs included within the scope of this study, in the two study sub-catchments upstream of Milton-under-Wychwood, are shown in Figure 6.1.

Table 6.3: Timeline of the phased installation of NFM interventions and the potential cumulative storage volumes (m^3) they added to the South and North sub-catchments. NB: Phase 1 interventions were not part of the official NFM scheme delivery. Adapted from Robotham et al., 2022. NB. An additional tributary containing further FSAs and additional storage volume is not included within this report.

Phase	Implementation	South sub-catchment		North sub-catchment	
		Interventions	Cumulative storage (m^3)	Interventions	Cumulative storage (m^3)
1	March 2017	None	0	Woody check dams (for bedload transport control)	0
2	February 2018	Leaky woody dams; field corner bunds and offline storage areas (P4, P5, P6, P7, P8, P9, OLP10); woodland planting; on-line ponds	11500	Woodland planting, offline storage area	140
3	February 2019	Field corner bunds and offline storage areas (OLP1, P2, P3); on-line ponds	14700	Field corner bund and offline storage area (P11, OLP11); leaky woody dam and swale; on-line pond	2020
4	Sept/Oct 2020	None	14700	Field corner bunds and offline storage areas (P12, P13)	8420
5	Winter 2020/21	Sediment/nutrient traps	14700	Sediment/nutrient traps and ponds	8420

The Littlestock Brook trial has been modelled by HR Wallingford using a full 2-dimensional InfoWorks ICM hydrodynamic model of the river channels and floodplains. HR Wallingford modelled the baseline before NFM measures were implemented, the effects of NFM measures on flooding up to

Phase 3 of NFM measure implementation, potential additional NFM measures and bund failure studies of the FSAs.

The purpose of this study was to collect data to understand the functioning of the measures enabling calibration and validation of the modelling.

A.3 The Littlestock Brook Monitoring Network

A.3.1 History of Littlestock Brook Monitoring

The Evenlode has been monitored at Cassington Mill by the EA since 1970, with daily and peak flow data available from the UK National River Flow Archive. The same site has been monitored for water quality by UKCEH since 2009 (Bowes et al., 2018).

Prior to the NFM trial implementation three monitoring stations were installed by UKCEH upstream of Milton-under-Wychwood, in the winter of 2016/2017. These stations provide a continuous time-series of water level and turbidity at the locations marked in Figure 6.3 as 'Upstream The Heath', 'The Heath' and 'Church Meadow' (UTH, TH and CM respectively).

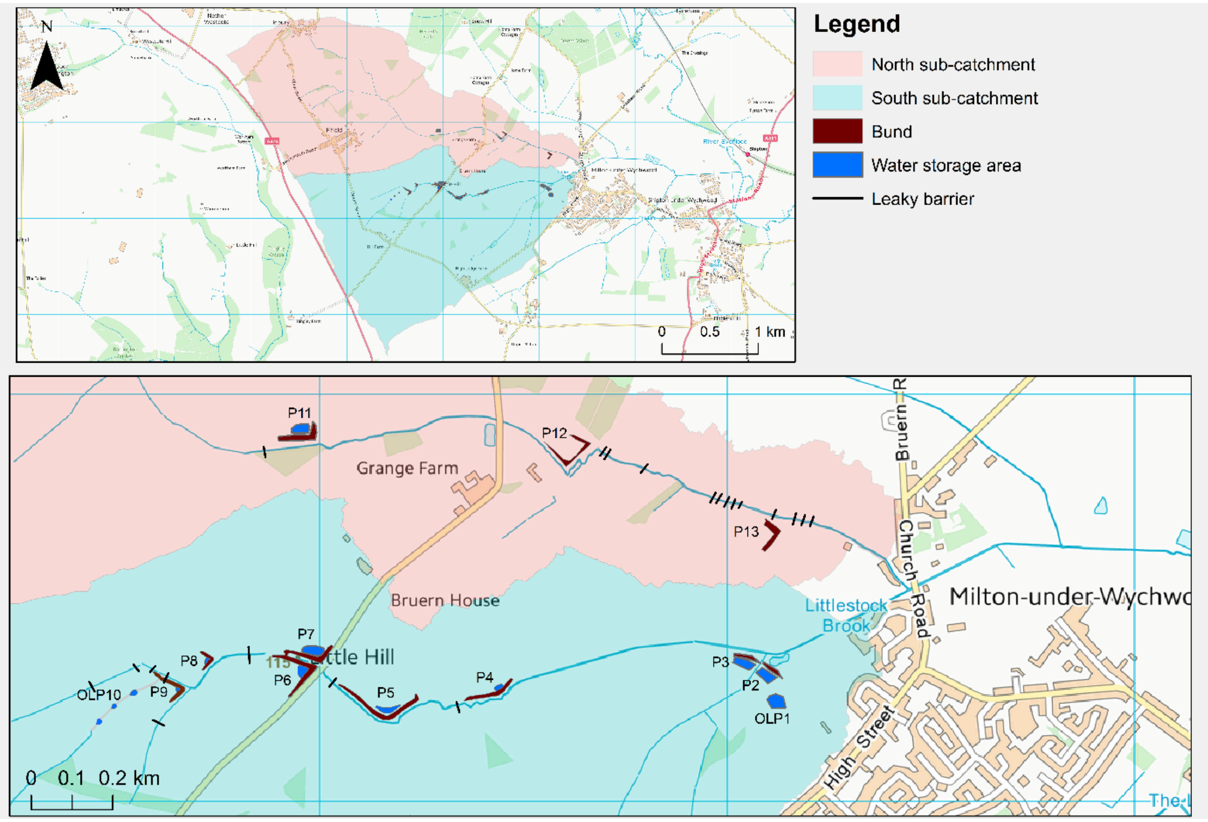
In 2018 the ECP organised a 'hydro-hack' event along with members of Oxford University, Atkins Consultancy, South East Rivers Trust to install water level sensors at several FSA and in-stream Phase 1 intervention sites.

In autumn 2018 two UKCEH PhD students started NFM research projects and further monitoring was implemented. This included 2 tipping bucket rain gauges; a storage rain gauge; 3 flow gauging sites; 6 automatic water samplers; 2 multi-parameter water quality sondes. The position of instrumentation is shown relative to NFM features in Figure 6.3. Further sediment and nutrient retention monitoring was implemented for FSAs constructed in the 2019 phase of NFM installation. Regular water quality spot sampling also started in March 2019, alongside wet-weather sampling campaigns during storm events.

After March 2021 the monitoring network was reduced to water level and rain gauge sensors. These data are available up to March 2022, when the monitoring project was completed.

The comprehensive hydrometric and water quality monitoring network focused on the south catchment, where nine Flood Storage Areas (FSAs) were monitored (Table 6.3; Figure 6.1). Three FSAs in the northern catchment were also monitored.

Discharge was estimated just upstream of the Milton-under-Wychwood confluence of both tributaries, using a stage-discharge rating curve and repeat observations of water level (corrected to datum). Rainfall was monitored using two tipping bucket rain gauges, near The Heath monitoring site at the outlet of the North sub-catchment and upstream in the Tears of Bruern in south catchment. A single storage gauge was co-located at the Tears of Bruern site and nearby Met Office Little Rissington rainfall data were available at grid reference 51.86, -1.692 (~3km from the Tears of Bruern gauges). Stream water level was monitored at eight locations across the catchment, in addition to the sites where discharge was monitored.



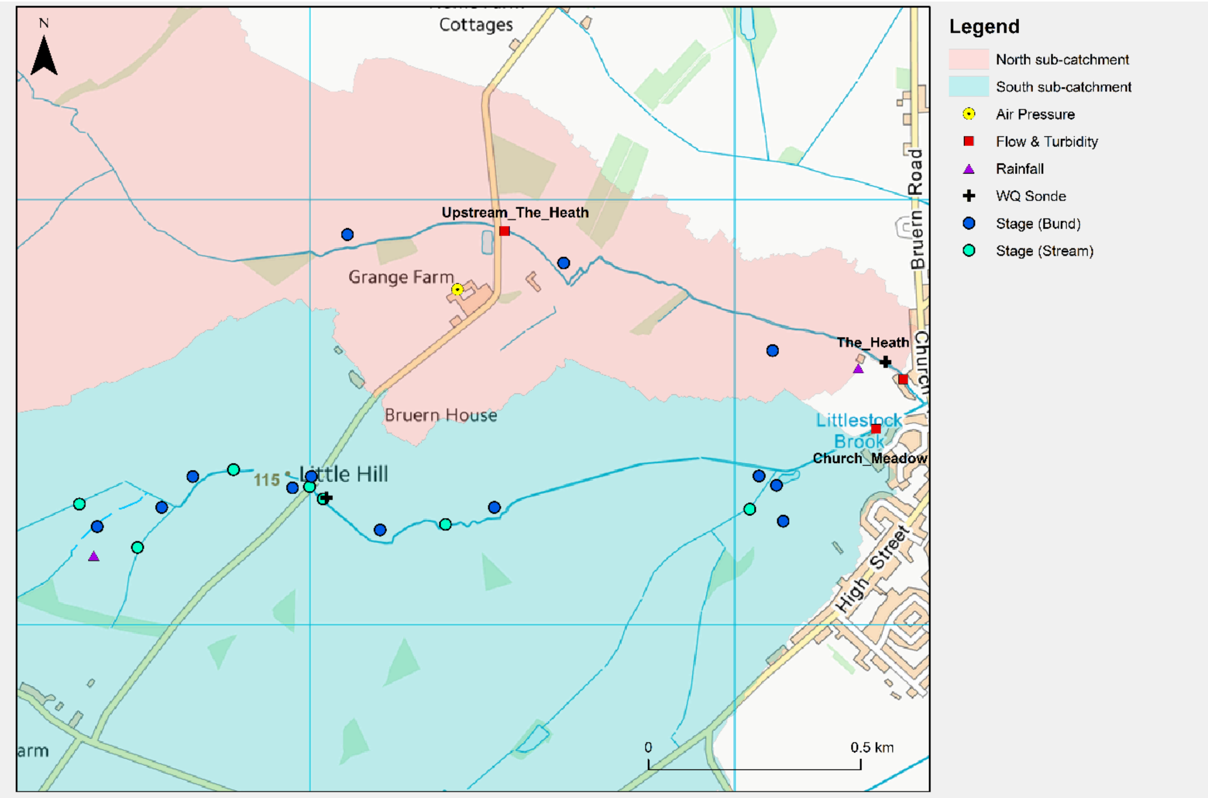


Figure 6.3: **(Top)** Overview of NFM features in the North and South sub-catchments; (middle) NFM features of the Littlestock Brook NFM Trial; **(bottom)** Hydrometric monitoring of the Littlestock brook. NB: OLP1 is also referred to as P1 and P1_OLP in different report sections, to keep consistent with linked publications. OLP10 includes the upstream (US) and downstream (DS) ponds referred to in water quality analysis sections and is also referred to as P10 and P10_OLP. P0_OLP is not shown on the map as it was not fully surveyed, did not provide additional storage potential, and did not have a level sensor installed. P0_OLP is located ~100m upstream of P2 and P3.

Water quality monitoring was set up as part of the wider hydrometric monitoring network to investigate the wider benefits of the NFM scheme. Turbidity sensors were co-located at the stream discharge monitoring sites in 2016, providing detailed time-series data on sediment dynamics downstream of NFM interventions. Automatic water samplers and spot samples have been used to capture water quality dynamics during storm events at these sites, through the determination of suspended sediment and nutrient concentrations. Water quality sondes have also monitored a suite of parameters (temperature, ammonium, dissolved oxygen, pH, and conductivity) at the Heath and downstream of P6-P9 and OLP10 (Figure 6.3). Sediment and nutrient retention were monitored within the NFM ponds and FSAs using a combination of siltation traps, automatic water samplers, sediment coring and manual surveying.

Table 6.4: Hydrometric and water quality monitoring in the Littlestock Brook.

Monitored parameter	Number of sites	Monitoring interval	Instrument
Rainfall	**2**	**2 minute**	**Casella Tipping Bucket Rain Gauge**
Flow gaugings	**2**	**Spot gaugings**	**Valeport Electromagnetic Current Meter**
In-stream water level	**8**	**5 minute**	**Level Troll 500/100 Data Logger**
Flood storage area water level (enabling volume estimation)	**11**	**5 minute**	**Rugged Level Troll 100 Data Logger**
Turbidity (enabling sediment concentration estimation)	**3**	**5 minute**	**FTS Digital Turbidity Sensor-12**
Water quality (continuous using multi parameter sondes)	**3**	**1 hour**	**EXO2 YSI sonde electrical conductivity**
Water quality (regular manual spot sampling)	**6**	**Spot samples**	**US DH-48 isokinetic manual sampler**
Storm water quality (using automatic water samplers)	**7**	**Spot samples**	**Sigma SD900 portable sampler**
Sediment accumulation in online ponds (multiple siltation traps deployed at each site)	**3**	**Spot samples**	**Purpose-built siltation traps (made by UKCEH)**
Sediment accumulation in flood storage areas (sediment coring)	**14**	**Spot samples**	**Purpose-built sediment corer (made by UKCEH)**
Event response of flood storage area and leaky barrier (using time-lapse cameras)	**2**	**1 hour**	**MCE-RPS-C 4G/3G Complete Camera Pillar System**

A.3.2 Rainfall

In February 2019, a tipping bucket rain gauge (Casella; Sycamore, IL, USA) was installed in the Tears of Bruern to measure rainfall at 2-minute intervals up to March 2022. It was installed in a clearing adjacent to the upstream on-line ponds in the south sub-catchment (Figure 6.3), free of obstructions and cleared of surface vegetation (Figure 6.4). The gauge was levelled and secured. Prior to installation the rain gauge was calibrated in the lab by levelling the tipping bucket mechanism, measuring the diameter of the collecting funnel to the nearest 0.1 cm and then calculating the volume of water ($V1t$) required for the bucket to tip using:

$$V = pir^2 * resolution$$

Once $V1t$ was known water was slowly dripped into each bucket alternately using a bulb pipette, noting the volume of water required for each tip. After five repetitions, the gauge screws were adjusted and then a further five repetitions completed. This process was repeated until the volume of water required per tip matches $V1t$.

After this a burette was used to drip water (volume sufficient to produce at least 30 tips) through the rain gauge with the collector funnel attached. The expected number of tips was then compared to the recorded number of tips. If these were within 5% of each other, the rain gauge was accepted for use using the standard calibration. If the difference exceeded 5%, a new calibration would be set based on the relationship between expected and actual number of tips.

The gauge was calibrated *in-situ* at least every 6 months by repeating the last step above. Any artificial tips were removed in data processing, these were cross-checked with alternative rainfall sources to verify accuracy.

A storage rain gauge was co-located to aid quality control of the tipping bucket gauge (Figure 6.4). During site visits, stored rainwater was emptied into a graduated cylinder and the volume checked against the tipping bucket rainfall total for the same period to ensure measurements were within a 5% tolerance range.

Additional rainfall data are available from the Little Rissington Met Office station (grid reference 51.86, -1.692 (~3km from the 'Tears of Bruern' gauges)). There is also a privately run weather station in the nearby village of Shipton-under-Wychwood. These data were in good agreement with the Littlestock Brook rain gauge and allowed rainfall data gaps to be filled using these alternative sources (Robotham et al., 2021).

Figure 6.4: Tears of Bruern Casella tipping bucket rain gauge (left, white) and storage rain gauge (right, brass).

A.3.3 Stream levels

Water level was measured and logged at 5-minute intervals at stream sites marked in Figure 6.3 using a Level TROLL 100 Data Logger (pressure sensor) submerged in a plastic stilling well to minimise data noise from water turbulence (Figure 6.5). A barometric pressure correction was applied using equation:

- $WaterLevel = 1000 \times \frac{100 \times (TotalPressure - AtmosphericPressure)}{10^3 \times 9.81}$

where water level is in millimetres and pressure in millibar. Quality controlled atmospheric pressure data were collected using a Rugged Level TROLL 100 sensor in a barn at Grange Farm (location marked on map as “air pressure”). Missing or suspect atmospheric pressure data were infilled or corrected using either a back-up barometric Level TROLL 100 located at the Heath or from the Met Office Little Rissington data.

Stream sites had stage boards mounted on wooden posts and surveyed to an accuracy of 1 cm using Real-time Kinematic Global Navigation Satellite System (RTK-GNSS) equipment, to enable conversion of water levels into meters above sea level (mASL). Stage board readings were taken during regular site visits and flow gaugings, and served as fixed points throughout the monitoring period. Raw water sensor water level data were corrected to stream stage using a linear regression developed between sensor values and observed stage board readings for each monitoring site. After this correction data were quality controlled for outliers and spikes.

Figure 6.5: Example site set up at The Heath (TH). Showing the stage board surveyed at the right of the photo, the black stilling well mounted to the wooden board and housing the TROLL pressure sensor, and the DTS-12 turbidity sensor to the left of the stilling well.

At the 3 stream monitoring sites (TH, UTH, CM), water level was measured using Vented Level TROLL 500s. These sensors did not need to be corrected for atmospheric pressure. Otherwise the level data from these sensors was processed and quality controlled in the same way as the Rugged sensors, with site specific sensor-stage board linear regression corrections and suspect data removal.

All TROLL sensors were supplied with a factory calibration and checked for clock drift at each site visit.

Additional 15-minute water level data at TH are telemetered by the EA and are publicly available at https://www.gaugemap.co.uk/#!Map/Summary/17680/13431.

A.3.4 Stream velocity

The mean channel velocity of 0.46 m s^{-1} was estimated using a salt dilution time-of-travel experiment, using an EXO1 YSI sonde electrical conductivity (EC) sensor to measure instream specific conductivity at a 1-second resolution (Hongve, 1987). Small injections (<50 g) of table salt (Sodium Chloride) were made during a storm event at the road bridge downstream of P6 and P7 in the South sub-catchment. Two EC sensors were located at the injection site and at the downstream catchment outlet monitoring site Church Meadow. The salt time-of-travel between ECs was used to estimate the mean channel velocity. It was assumed that the velocity was constant along the watercourse of approximately 1460 m.

A.3.5 Stream flow

Streamflow is available at 5-minute intervals at the 3 in-stream monitoring sites (TH, UTH, CM). Discharge was estimated under low flows using a conductivity sensor (EXO1, YSI; Yellow Springs, OH, USA) and the salt dilution method as detailed in Section A.3.4. Discharge was primarily estimated in higher flows using an Electromagnetic Current Meter ((ECM) Valeport; Totnes, UK) and the velocity-area method. For this, cross-sectional area was calculated by measuring water depths across the channel at regular intervals using a metre rule. At each point, flow velocity was then measured with an ECM, enabling the instantaneous discharge to be calculated using the below equation (Herschy,1993);

$$q_{5+6} = \frac{\overline{v}_5+\overline{v}_6}{2}\frac{d_5+d_6}{2}(b| \quad |6-b_5)$$

where q_{5+6} = discharge through segment 5-6;

$\overline{v}_5, \overline{v}_6$ = mean velocities in verticals 5 and 6

d_5, d_6= depth of flow at verticals 5 and 6;

b_5, b_6= distance from initial point on the bank to verticals 5 and 6.

The total discharge was calculated as the sum of the discharge in all segments and assumes that the velocity at each bank is zero.

Rating curves were developed for each stream monitoring site, enabling discharge estimates to be calculated from the power law relationship between observed stage and discharge. Rating curves were computed using the '*nls*' package in R, with lower and upper 95% confidence intervals calculated following the method used by Dalgaard (2004). The equations for each relationship are given in Table 6.5, with plots of each rating curve up to the maximum recorded stage included in Appendix 1 – Rating curves, with 95 % confidence intervals shown as red dashed lines.

To accurately estimate low flows at The Heath site, ratings were constructed for both low and high flows. Plotting separate ratings for low and high flows significantly improves estimates of baseflow discharge. Discharges calculated using the low flow relationship are much closer to the gauged values observed during baseflow gaugings. The low flow rating used only the low flow gauging measurements, which did not fit the full rating relationship. A curve with only high flow gaugings was used to estimate high flows. Rating curves were plotted using discharge (Q) measurements made at the Heath Site and non-linear least squares regressions were fitted.

Due to limited gauging measurements for UTH, the rating curve is only suitable for estimating discharge up to ~330 L s^{-1} and should not be used beyond this threshold (Robotham et al., 2022).

Table 6.5: Rating curve equations and confidence intervals used for discharge estimation at each monitoring site under different flow conditions. Flow units are in L s^{-1}, and stage units are in m.

Site Name	Flow Condition	Rating	Rating Curve Equation	n	Observed flow range
The Heath (TH)	Low	Estimate	$710377415.957 \times (Stage + 0.01)^{8.277}$	5	3.08 – 43.44
		Lower 95% CI	$796296879.462 \times (Stage + 0.01)^{8.350}$		
		Upper 95% CI	$636718094.631 \times (Stage + 0.01)^{8.207}$		
The Heath (TH)	High	Estimate	$6849.014 \times (Stage + 0.01)^{2.362}$	11	78.99 – 946.23
		Lower 95% CI	$4397.425 \times (Stage + 0.01)^{2.158}$		
		Upper 95% CI	$9599.230 \times (Stage + 0.01)^{2.509}$		
Upstream The Heath (UTH)	Low & high (<330 L s^{-1})	Estimate	$3914.873 \times (Stage)^{3.567}$	5	2.87 – 329.59
		Lower 95% CI	$4415.422 \times (Stage)^{3.790}$		
		Upper 95% CI	$3539.962 \times (Stage)^{3.373}$		
Church Meadow (CM)	Low & high	Estimate	$1417.271 \times (Stage - 0.013)^{1.167}$	15	4.47 – 668.35
		Lower 95% CI	$1341.966 \times (Stage - 0.013)^{1.169}$		
		Upper 95% CI	$1492.510 \times (Stage - 0.013)^{1.165}$		

A.3.6 Stream water quality

(1) Continuous turbidity

Turbidity data were measured at 5-minute intervals using a DTS-12 (Digital Turbidity Sensor, Forest Technology Systems Ltd.) located in-stream and logged to a CR1000 datalogger at the three monitoring sites (TH, UTH, CM; Figure 6.5) between winter 2016/2017 to March 2021. In June 2017 an EXO2 optical turbidity sensor was also installed at TH. The EXO2 sonde was set to take hourly samples with a pumped system. The pumped system allowed measurements to be taken during low flows, when the DTS-12 sensor was above the water level. The pumped sample is taken into the system through a strainer in order to prevent large particles from the streambed or suspended organic debris (e.g. leaf litter) being sampled. This improved the reliability of the sensor, particularly during high flow events where the DTS-12 sensor can become obscured by debris (Robotham et al. 2021).

Turbidity values from the DTS-12 sensor were measured in NTU (Nephelometric Turbidity Unit) at a resolution of 0.01 NTU and accuracy of ±2% of reading + 0.2 NTU (0-399 NTU) and ±4% of reading (400-1600 NTU). Each turbidity measurement consists of 100 instantaneous samples from which summary statistics are computed. The median turbidity value is used as opposed to the sample mean, to minimise the risk of erroneous extreme samples biasing the value. Turbidity values from the EXO2 sensor were measured in NTU at a resolution of 0.01 NTU and accuracy of ±2% (Robotham et al. 2022).

Turbidity sensors were replaced approximately twice a year so that the sensors could be returned to the lab for calibration. Raw turbidity measurements were calibrated using linear equations specific to each DTS-12 sensor for that specific deployment period. The equations were determined by calibrating the DTS-12 sensors against polymer bead solutions covering a range of concentrations. These solutions were evaluated during each calibration exercise against a certified known 1000NTU standard using an Analite turbidity probe. This ensured solutions were of a known turbidity and any changes in probe output were due to changes in the probes and not the solutions. Calibrations took place before and after each turbidity sensor deployment, allowing any sensor drift to be monitored. As no significant drift was observed, the mean of the pre/post calibration values was used for that deployment period. The pre/post calibration values determined the minimum and maximum turbidity values used as estimated uncertainty bounds, to account for error attributed to minor sensor drift within the expected range of the instrument for the deployment period.

Turbidity data were quality controlled using a set of simple rules to remove erroneous measurements, caused by things such as sensor errors or stream debris getting caught on the optical face of the sensor. The rules are as follows:

- Raw values had to be > 0 NTU. Negative and 0 values were removed.
- Raw values had to be < 1600 NTU. Values above the detection range of the sensor were removed.
- Raw values recorded during prolonged periods of sensor failure were removed, where validated *in-situ*.
- Erroneous spikes in the time-series were removed. Spikes were identified using a formula stating that the turbidity value at a given timestep should be less than 3 times the mean average of the turbidity values for the timesteps immediately before and after.
- Erroneous drops in the time-series were removed. Drops were identified using a formula stating that the turbidity value at a given timestep should be greater than the mean average of the turbidity values for the timesteps immediately before and after divided by 3.
- Gaps in the time-series were linearly interpolated where the gap was less than 12 hours (outside of storm events). This was done using the function '*fillMissing*' from the '*baytrends*' package in R. During storm events, only gaps created by individual data 'spikes' were filled due to rapid changes in turbidity in response to rainfall.
- Gaps larger than 12 hours were left in the time-series.

There are periods where the stream water level was very low and exposed the turbidity sensor to the air, giving false turbidity readings close to zero. These values were identified and removed from the dataset where possible, but some may remain. Due to this issue there are large gaps in the turbidity data at TH early in the time-series. The installation of the EXO2 turbidity sensor at this site helped reduce this data loss.

(2) Instantaneous Sediment and Nutrient Concentrations

To monitor stream water quality, samples were collected using a US DH-48 isokinetic manual sampler on a rod at regular (~monthly) visits to the 3 stream sites between 2016 and 2018. After 2018, automatic samplers (Sigma SD900, Hach; Loveland, CO, USA) were deployed at each monitoring site. These were programmed to trigger at a high water level indicative of a storm event, through pressure sensor data read by the CR1000 logger. Samples were processed for suspended sediment concentration (SSC) and volatile solids concentration (VSC) (as a proxy for organic matter) in the same way as described for the on-line pond samples in Section A.3.8.

(3) Continuous Suspended Sediment Concentration and Total Phosphorous

Calculation of the SSC time-series used simple linear regressions using turbidity to predict concentrations from spot water samples (Section (2)) taken at the same time as the turbidity measurement. Turbidity from the in-stream sensor was calibrated against SSC and Total Phosphorus

(TP) samples taken under a range of flows to give estimated time-series of SSC and TP. The regressions and the upper and lower 95% confidence intervals for each monitoring site are listed in Table 6.6. SSC was used as the predictor in regressions to calculate TP (Table 6.7).

Table 6.6: Regressions, confidence intervals, and summary statistics for the conversion of turbidity to SSC. All regressions were statistically significant at the $p<0.001$ level.

Site Name	Regression line	Lower 95% CI	Upper 95% CI	n	R^2
The Heath	SSC=1.5358×$Turbidity$	0.9354×SSC	1.0646×SSC	70	0.93
Upstream The Heath	EXO SSC=2.00248×EXO $Turbidity$	0.9482× EXO SSC	1.0518×SSC	100	0.94
Church Meadow	SSC=0.84206×$Turbidity$+4.03079	0.969×SSC	1.031×SSC	94	0.99
The Heath	SSC=1.00701×$Turbidity$	0.9806×SSC	1.019×SSC	95	0.99

Table 6.7: Regressions and summary statistics for the estimation of TP from SSC. All regressions were significant at the $p<0.001$ level.

Site Name	Regression line	n	R^2
The Heath	TP=0.0019×SSC+0.14	111	0.94
Upstream The Heath	TP=0.0018×SSC+0.15	47	0.94
Church Meadow	TP=0.0019×SSC+0.035	359	0.79

Fluxes of total suspended sediment, silt and clay, and TP were also calculated at the Downstream Catchment Outlet site, using discharge, SSC and TP data at 5-minute intervals. Fluxes were calculated by integrating the SSC/TP instantaneous load time-series for the monitoring period. Suspended sediment particle size distributions were sampled during two high flow and SSC events and measured using laser diffraction particle size analysis (Mastersizer 2000, Malvern Panalytical; Malvern, UK). Prior to analysis, 0.5 to 0.6 g sub-samples of sediment were treated with a 5% sodium hexametaphosphate solution and agitated for 5 minutes in an ultrasonic bath to disperse particles and prevent agglomeration. The event particle size distributions were assumed to be representative of the stream's suspended load as storm events contribute the majority of the total sediment flux. The proportions of particles <63 µm in diameter in the samples were averaged and combined to estimate the flux of silt and clay leaving the catchment.

(4) Multi-parameter water quality sondes

Water temperature, electrical conductivity, pH, ammonium, turbidity, and dissolved oxygen were monitored at hourly intervals at 3 sites using YSI EXO2 multi-parameter sondes at hourly intervals. One sonde was deployed as part of Thames Water's 'Smarter Water Catchments' initiative between P5 and P6/P7 (Figure 6.3). A further sonde was deployed by the EA at TH. These sondes operated using a pumped flow cell system which minimised sensor fouling. A further EA sonde was deployed downstream of the scope of this monitoring report and did not use a pumped system, from which the data have not been used.

A.3.7 FSA water levels and volumes

Water levels were monitored in 12 bunds and 1 online pond (Figure 6.3). Rugged Level TROLL 100s were installed in stilling wells adjacent to stage boards in the deepest part of each FSA (Figure 6.6), surveyed using RTK-GNSS equipment. FSA Rugged level TROLL data were corrected to mASL, for atmospheric pressure and quality controlled using the same methods as described for the in-stream Rugged level TROLLs in Section A.3.3.

Figure 6.6: Example bund monitoring site (P3), showing stilling well containing Rugged TROLL 100 pressure sensor, mounted to wooden stake and RTK-GNSS surveyed stage board in deepest part of bund.

FSA storage volume was estimated for each feature using a Digital Elevation Model (DEM), produced using 1 m horizontal resolution LiDAR data. Where no LiDAR data were available as-built manual survey data interpolated using the Natural Neighbours method were used. The vertical resolution of the LiDAR and the manual survey data are both down to micrometre resolution, however the errors are larger than this. We estimate the relative height error (random error) to be no more than ±5 cm. EA specifications require the absolute height error to be less than ±15 cm. This is the root mean square error, which quantifies the error or difference between the Ground Truth Survey and LIDAR data.

The DEMs were imported into ArcGIS to identify the maximum static water level, defined by the lowest elevation point on top of the bund. In ArcGIS the maximum FSA volume was estimated using a raster between this maximum static water level and the elevation at which the stream channel and FSA are not connected. A depth-area-volume toolset was applied to the raster to produce a depth-stored volume lookup table for each FSA.

The continuous corrected and quality-controlled water level time-series in each FSA was then used to produce a time-series of stored volume for each FSA, by matching the water level time-series to the depth-stored volume lookup table produced from ArcGIS. Note that this was not possible for the newer FSAs installed in the north sub-catchment, as no LiDAR or survey data were available.

A.3.8 Pond water quality and nutrient attenuation

The quality of the ponds was observed using water samples taken from the inflow and outflow (for suspended sediments, nutrients, and major anions) during storm events and baseflows. Changes in nutrient concentration as water flows through the three online ponds were observed to understand to what extent they were attenuated both during storm events and baseflow conditions. Pond outflow discharge was measured on two occasions using salt dilution gauging for lower flows, and also an electromagnetic current meter and the area-velocity method when the stream was deep enough to do so. Multi-parameter sondes were installed in one of the on-line ponds and at a downstream location to continuously measure temperature, dissolved oxygen (DO), electrical conductivity (EC), and chlorophyll.

The following methods form part of a journal paper (Robotham *et al.*, 2021) which details the sampling of the water quality and nutrient attenuation effect of the on-line ponds.

To monitor water quality outside of storm events (i.e. under baseflow conditions), water samples from the on-line pond system's inlet and outlet were collected during field visits every 2–4 weeks. One unfiltered 60 mL sample was taken for total phosphorus (TP), and two 60 mL samples were immediately filtered through a 0.45 µm cellulose nitrate membrane (Whatman™ WCN grade; Maidstone, UK) for analysis of total dissolved phosphorus (TDP), soluble reactive phosphorus (SRP), and dissolved major ions (NO_2^-, NO_3^-, NH_4^+, F^-, Cl^- and SO_4^{2-}). Particulate phosphorus (PP) was taken to be the difference between TP and TDP.

Approximately 500 mL was sampled using the US DH-48 sampler for determination of SSC and VSC. Water chemistry samples were refrigerated at 4°C in the UKCEH labs upon return from the field until they were analysed following Wallingford Nutrient Chemistry Laboratories procedures described in detail by Bowes et al. (2018). SSC was determined gravimetrically by filtering known volumes of water samples through pre-ashed, dried and weighed Whatman™ GF/C™ filter papers, which were then oven dried at 105°C for at least two hours. Filter papers were then reweighed after cooling in a desiccator for 30 minutes. VSC was then determined through loss-on-ignition (LOI) by igniting filter papers in a muffle furnace (AAF 1100, Carbolite Gero; Hope, Derbyshire, UK) at 500°C for 30 minutes before being cooled and reweighed.

For monitoring storm events, automatic samplers (Sigma SD900, Hach; Loveland, CO, USA) were deployed and triggered between March 2019 and February 2020 at four locations along the stream to sample water flowing into and out of each pond (Figure 6.7). Triggering of samplers was determined based on the rainfall forecast in order to capture samples approximately representative of the event. Grab samples of run-off were taken from contributing overland flow pathways. Samples were refrigerated upon return to the laboratory, and 60 mL subsamples were taken as soon as possible for chemical determinands of interest. To ensure representative subsampling, samples were thoroughly mixed before immediately taking an aliquot using a syringe. The remaining sample was used to determine SSC and VSC using consistent methods.

Discharge was estimated at the ponds' outflows in higher flows using an Electromagnetic Current Meter (Valeport; Totnes, UK) and the velocity-area method, and also under low flows using a conductivity sensor (EXO1, YSI; Yellow Springs, OH, USA) and the salt dilution method. During storm events, run-off frequently overwhelmed the small stream channel and rendered it unsuitable for accurate flow measurement or development of a reliable stage-discharge relationship. Instead, water flowing through the ponds was estimated as a catchment area-weighted proportion of the discharge measured at a more stable gauging site (CM). In order to represent timings of storm hydrographs more realistically, the estimated discharge was shifted back in time by applying a linear regression (R^2 = 0.51) between peak discharge and the time difference between peak stage in the Central Pond and at the CM site. It was assumed that at a given time, discharge was equal at both pond inflows and outflows.

Figure 6.7: Automatic water sampler at the inlet of an on-line pond.

A multi-parameter sonde was installed in the central on-line pond next to the water level sensor and stage board. A second sonde was deployed at an instream location in the reach adjacent to the P8 storage feature. Both sondes logged temperature, dissolved oxygen (DO), electrical conductivity (EC), and chlorophyll at 15-minute intervals. Sondes were equipped with wipers which cleaned the sensors in-between measurements. Individual sensors were calibrated according to EA National Water Quality Instrumentation Service protocols. The draining and significant reduction in water level of the on-line pond in drier conditions meant that sensors were exposed to the air for periods of time. Lowering the sonde deeper into the water to avoid this also posed risks of sensor burial and fouling due to the accumulation of sediment in the pond. Consequently, water and sediment sampling yielded a more reliable source of data in this context.

A.3.9 Pond sediment and associated nutrient accumulation

Siltation traps were deployed in each of the on-line ponds to quantify sediment, organic matter, and P accumulation, and determine the particle size distribution of the trapped material. Ponds were also surveyed to estimate the total volume and mass of stored sediment accumulated since their construction.

The following methods form part of a journal paper (Robotham *et al.*, 2021) which details the sampling of the sediment, organic matter, and P trapping effect of the on-line ponds.

The net accumulation of sediment and nutrients was determined in the on-line ponds using siltation traps (Figure 6.8).

Figure 6.8: Siltation trap filled with on-line pond sediment after being deployed for circa one month.

Traps were assembled from circular plastic saucers (19 cm in diameter, 4 cm in height) with weights attached to allow them to sink and rest on the pond bed. Traps were positioned in ponds as evenly as possible, with one central trap and four outer traps. The traps were deployed for periods of up to 50 days before being retrieved, emptied, and immediately redeployed. Collected sediment (including pond water pooled on the surface) from each trap was emptied into individual plastic bottles for transport back to the UKCEH laboratory. Bottles were then emptied into larger plastic boxes and refrigerated for at least 48 hours to allow suspended solids to settle out. The supernatant was then siphoned off into bottles and filtered to account for the mass of any fine particles still in suspension. Sediment in the boxes was stirred thoroughly, and for each, three sub-samples of approximately 5 grams were transferred into centrifuge tubes for particle size analysis (method as described in Section A.3.6(3). To determine sediment mass, the remaining sediment was distributed into pre-weighed aluminium trays (~100 g sediment per tray) and oven-dried at 105 °C for at least 48 hours before being cooled and reweighed. To determine volatile solids (organic matter proxy) by LOI, one tray per trap was then ignited at 500 °C for 2 hours before being cooled and reweighed. One tray per batch was reheated and reweighed to ensure that the sample mass remained stable and the LOI was complete. P content was determined by grinding the ignited sample into a fine powder, of which triplicate subsamples of 3 ± 0.1 mg were taken, mixed with 60 mL ultrapure water, and then analysed with the same TP methodology used for water samples (Section A.3.8). Length and width transects of pond sediment depths were surveyed in January and July 2020 following a standard method, and spatially interpolated in a GIS (ArcMap, Esri; Redlands, CA, USA) using the natural neighbour interpolation method to estimate the total stored sediment volumes in each pond.

Siltation trap monitoring of the on-line ponds took place between August 2019 and March 2020 with six deployment periods. This monitoring was unable to continue beyond March 2020 due to the COVID-19 pandemic.

A.3.10 Bund sediment accumulation

The accumulation of sediment (and phosphorus and organic carbon) was monitored using two approaches. Sediment deposition pins were placed at regular intervals within FSAs in order to regularly measure the depth of accumulated sediment across the area. The second approach involved the collection of sediment cores to quantify bulk density, in combination with the surveying of accumulated sediment depths across bunded areas to estimate stored sediment volumes, and from this derive estimates of the stored masses.

The following methods form part of a journal paper (Robotham *et al.*, under review) which details the determination of the accumulation of sediment, organic matter, and P within offline storage features.

The deposition pins consisted of plastic-coated metal rods (approximately 1.2 m in height) that were driven into the solid ground base within the bunded area of each storage feature (Figure 6.9). Pins were arranged in a cross-shaped formation spanning the width and length of the bunded area, making sure to include the deepest section (typically co-located with stage board level sensor). Pins were placed at between 1 and 5 m intervals depending on the size of the bunded area. At periods of ~2 months, the height of the pins above the sediment surface was measured to determine the accumulation (or erosion). Unfortunately this approach did not yield satisfactory results for several reasons. In some features (e.g. P11) the pins were disturbed by livestock when they were dry, and in one case (P3) pins were vandalised. The rapid draining of many FSAs also meant the soil underwent frequent cycles of wetting and drying which destabilised the pins, causing them to lean in places. The degree of error associated with these issues was greater than the extent of the sediment accumulation within these ~2-month periods, rendering them unsuitable for measuring the effectiveness of the NFM features at trapping sediment in this context.

Figure 6.9: Erosion pins following installation in P6.

The approach that used a combination of sediment core sampling and sediment depth surveying was able to provide a more consistently successful approach across all of the NFM storage features.

Sediment cores were sampled within each FSA to determine the average bulk density of accumulated sediment. A coring device suitable for sampling soft, submerged sediment was made from 1 m long copper pipe (2.6 cm in diameter), cut at a 45° angle on one end to aid insertion into the sediment. Six cores were taken from each storage feature (half in shallower sections closer to feature margins, and half in deeper central sections). Sediment depth (down to the solid base of the storage feature) was also measured at each coring location to determine the original core length prior to any potential compaction that occurred during coring. Cores were stored in plastic sample bags and refrigerated at 4°C before being transferred into aluminium trays and oven-dried at 105°C for at least 36 hours before being weighed. Dry bulk density was calculated following guidance of Wood (2006). LOI was quantified as a proxy measure for organic matter (OM) content. The samples were heated for 2 hours at 500°C before being cooled in a desiccator and re-weighed. OM was converted into organic carbon (OC) content using a 0.58 conversion factor chosen based on the literature (Bhatti and Bauer, 2002; De Vos et al., 2005; Rollett et al., 2020). The TP concentration of the sediment was determined spectrophotometrically. The ashed sample was crushed into a fine powder and combined into a bulk sample for each storage feature from which triplicate sub-samples of 3±0.1 mg were then taken for determining average TP content. Sub-samples were mixed with 20 ml ultrapure water and analysed following the modified molybdenum blue methodology of Eisenreich et al. (1975).

Alongside the cores, additional sediment was sampled for determining the absolute particle size distribution using the methods described in Section A.3.6(3).

Depths of accumulated sediment within each storage feature were surveyed along two transects spanning the length and width of the feature, with measurements being taken at 1 to 2 m intervals. Depths were measured from the solid base of the feature to the surface of the soft sediment layer using a metre rule. Transects were positioned so that they approximately captured the deepest section of the storage feature and a handheld GPS (eTrex, Garmin; Olathe, KS, USA) was used to locate the start and end points of each transect. Maintenance work to remove sediment from the series of P10 ponds following their surveying in January and June 2020 meant that any future surveying would not represent the accumulation since construction. As a result, sediment depths measured for these features represent a shorter period of accumulation compared to the other features which were measured following a longer period post-construction and with no maintenance. Sediment depths were spatially interpolated using the natural neighbour interpolation method (ArcMap 10.5, Esri; Redlands, CA, USA) to estimate stored sediment volumes. The bulk density measurements were then used to convert sediment volumes into masses, and concentration data were used to calculate total stored nutrient masses. A combination of LiDAR and RTK GNSS (GS14, Leica Geosystems; St. Gallen, Switzerland) surveys of the features post-construction were similarly used to estimate their total storage volumes.

A.4 Data Coverage

Total data coverage for the monitoring period is over 90 % across all sensors. Monthly data coverage is shown in Figure 6.10 for the sensors and combined parameters used for analyses within this report.

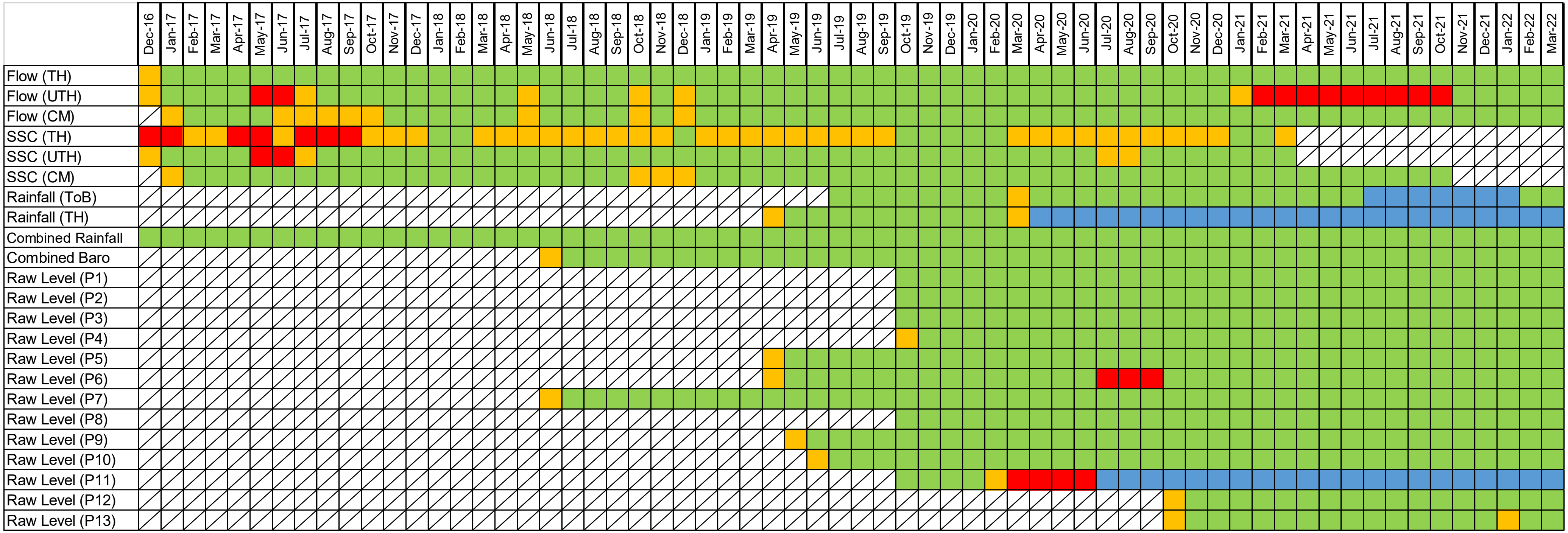


Figure 6.10: Monthly data coverage for parameters and sensors over Littlestock Brook NFM trial. Green = >90 % data coverage, amber = <90 % data coverage, red = no data, blue = suspect data. NB: P1 in this table is also referred to as OLP1 and P1_OLP in different report sections, to keep consistent with linked publications. In this table, P10 refers to the second pond within the series of three online ponds (as shown in Figure 6.3). This was the only instrumented pond in the series. This pond is also referred to as OLP10, P10_OLP and Central Pond in different report sections, to keep consistent with linked publications.

A.4.1 Flow

Figure 6.10 shows flow 100 % data coverage at TH for the monitoring period, with the exception of December 2019 when the first sensor was installed at this site. Data coverage at UTH has some months of <90 % data coverage and no data periods due to technical issues with the sensor at that time. From January 2021 there is an extended period of no data due to the sensor coming to the end of its lifespan; it was then replaced at the start of November later that year. Some data from the old sensor may be recoverable by the manufacturer a later date. Flow data at CM had several periods of <90 % coverage due to issues with the sensor. This was eventually replaced in October 2017 to give more consistent data coverage for the rest of the monitoring period.

A.4.2 Suspended Sediment Concentration

Overall the SSC data show good coverage for the majority of the monitoring period, however due to the optical nature of the turbidity sensors, data gaps and issues typically occur more frequently compared to other data collection methods. Figure 6.10 shows that SSC data coverage at TH is notably patchy for the first year of monitoring until December 2017, when the additional multi-parameter water quality sonde (measuring turbidity) was installed. The data coverage issues in this early period relate to the low water levels at this hydrologically flashy site resulting in the sensor being above the water level most of the time (except during rainfall events). Data coverage is more complete at the UTH site with the exception of two summer months in 2017 missing due to a technical issue with the datalogger. The CM site has the most complete series of data, with only 4 months of <90 % data coverage due to issues with the turbidity probe at this site.

A.4.3 Rainfall

Figure 6.10 shows that overall the combined rainfall coverage for the Littlestock Brook area is 100 % due to the multiple sources of data used to form this data series. The ToB and TH rain gauges were installed in 2019 and have good coverage until March 2020 when some of the data were lost as a result of data being automatically overwritten during the Covid-19 lockdown period. The period of suspect data for the ToB site is as a result of the tipping bucket becoming blocked. The period of suspect data at TH site is as a result of the gauge becoming unstable and potentially out of calibration. During these periods, Little Rissington rainfall data were used to ensure reliability as detailed in Section A.3.2.

A.4.4 Atmospheric Pressure

The combined barometric pressure data coverage in Figure 6.10 shows 100 % data coverage for the monitoring period. There is <90 % data in the installation month as the sensor was installed mid-month. This combined time-series is the pressure sensor in the barn Grange Farm with missing or suspect atmospheric pressure data infilled or corrected using either the back-up barometric sensor located at the Heath or from the Met Office Little Rissington data, as detailed in Section A.3.3. This is the time-series that was used to correct stream and FSA water level data.

A.4.5 FSA water levels

FSA P1-13 data coverage in Figure 6.10 is for raw sensor data, as FSAs frequently dry out resulting in prolonged no data during dry periods. Across all sensors over 98 % of total data coverage was achieved. Less than 90 % data coverage is observed for most FSAs in sensor installation months as the whole month was not monitored. There is missing P6 data for the period July to October 2020 due to a sensor error. The P11 sensor was repeatedly knocked over during strong flood events and by livestock, resulting in missing or suspect data from February 2020 onwards. This is as the replacement sensors were not surveyed before being knocked over again. The P13 sensor was knocked over by livestock in January 2022, this was recovered and replaced during the same month resulting in less than 90% data coverage.

A.5 Analysis

A.5.1 Rainfall

Daily rainfall totals for the area are presented in Figure 6.11 with the antecedent precipitation index (API) giving an indication of catchment wetness over the period. The trend-line shows how the winters in 2020 and 2021 were notably wet, with the highest daily total exceeding 40 mm on 23rd December 2020. Having higher API values prior to intense rainfall events increases the risk of rapid run-off responses. Under these conditions NFM may be particularly valuable in delaying and attenuating this response. The drought conditions that occurred during summer 2018 can be clearly seen reflected in the API which drew down close to 0.

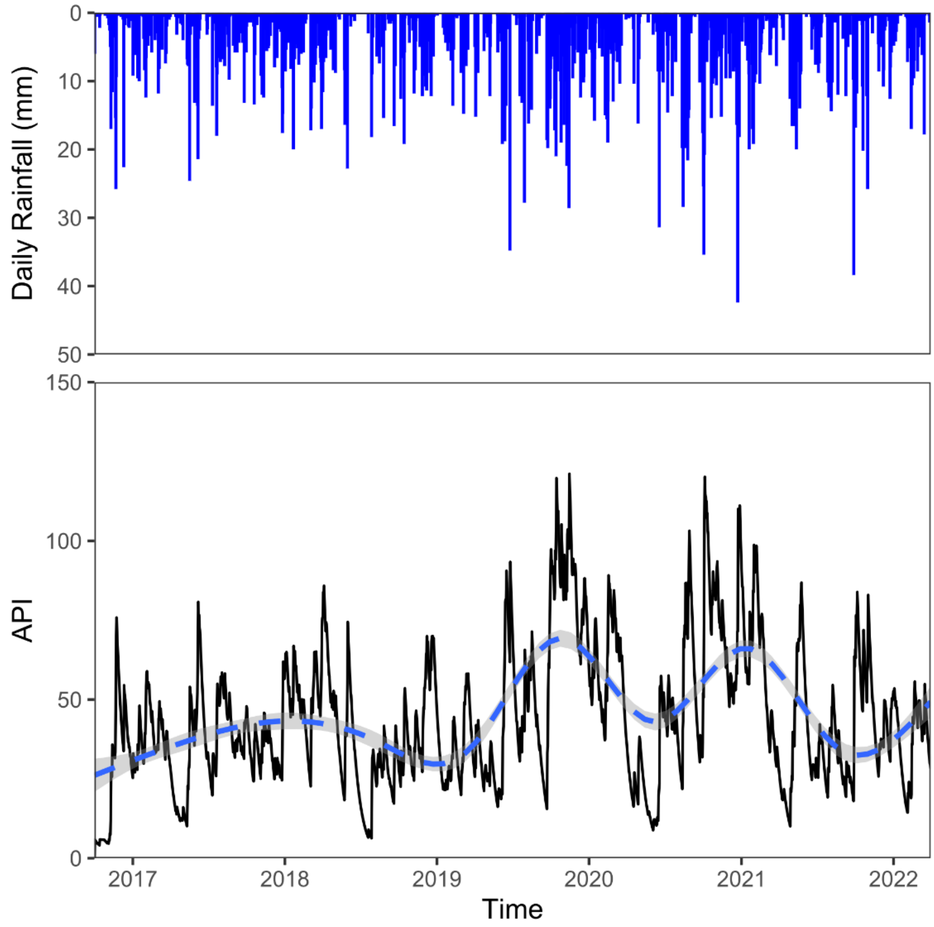


Figure 6.11: Daily rainfall totals (mm) and Antecedent Precipitation Index (API) from October 2016 to April 2022.

Table 6.8 gives the total annual rainfall in each water year. The water year for 2017 (prior to implementation of NFM) was notably dry when compared to the 30-year (1991-2020) average of 809.6 mm (Met Office, 2022). This contrasts to the particularly wet years of 2020 and 2021 following NFM implementation.

Table 6.8: Total annual rainfall (mm) for each water year in the monitoring period. 2022 total includes up to April 1st.

Water Year	Total Rainfall (mm)
2017	674.2
2018	686.2
2019	800.6
2020	988.2
2021	965.0
2022†	359.2

†Years with incomplete data (see Figure 6.10, Section A.4 'Data Coverage').

A.5.2 Annual stream discharge

Stream discharges (flows) are presented at 5-minute resolution for each water year (1st October – 30th September) at each of the three streamflow monitoring sites (Figure 6.12; Figure 6.13; Figure 6.14). It is important to note that for the UTH site, only discharges up to 330 L s^{-1} are displayed due to the uncertainty in the stage-discharge rating above this threshold at this site.

Church Meadow:

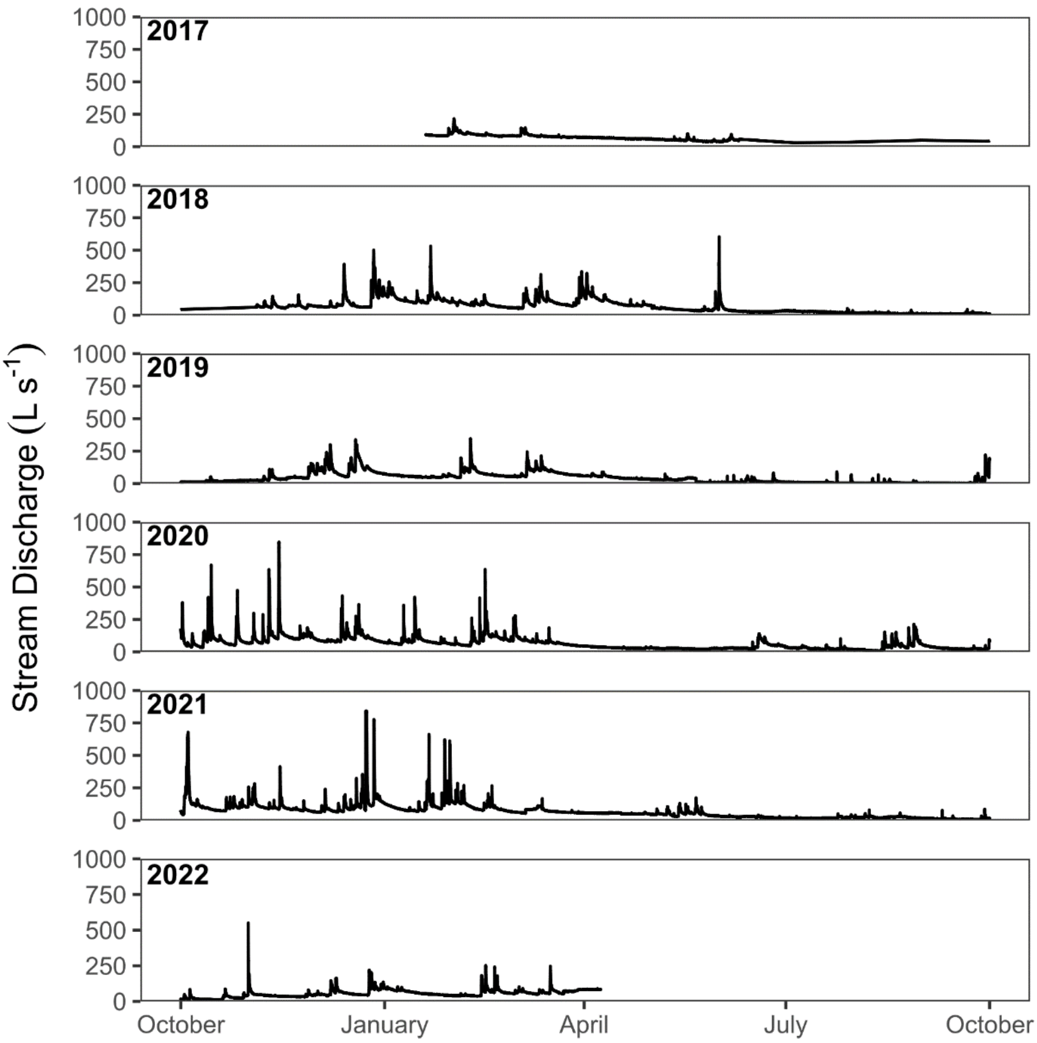


Figure 6.12: Stream discharge (L s-1) at Church Meadow (CM) in each water year of the monitoring period. The data series starts in January 2017 and ends in April 2022. NB. Y-axis scale is lower than North Catchment (TH and UTH) plots (Figures 13 and 14) to show full detail.

The Heath:

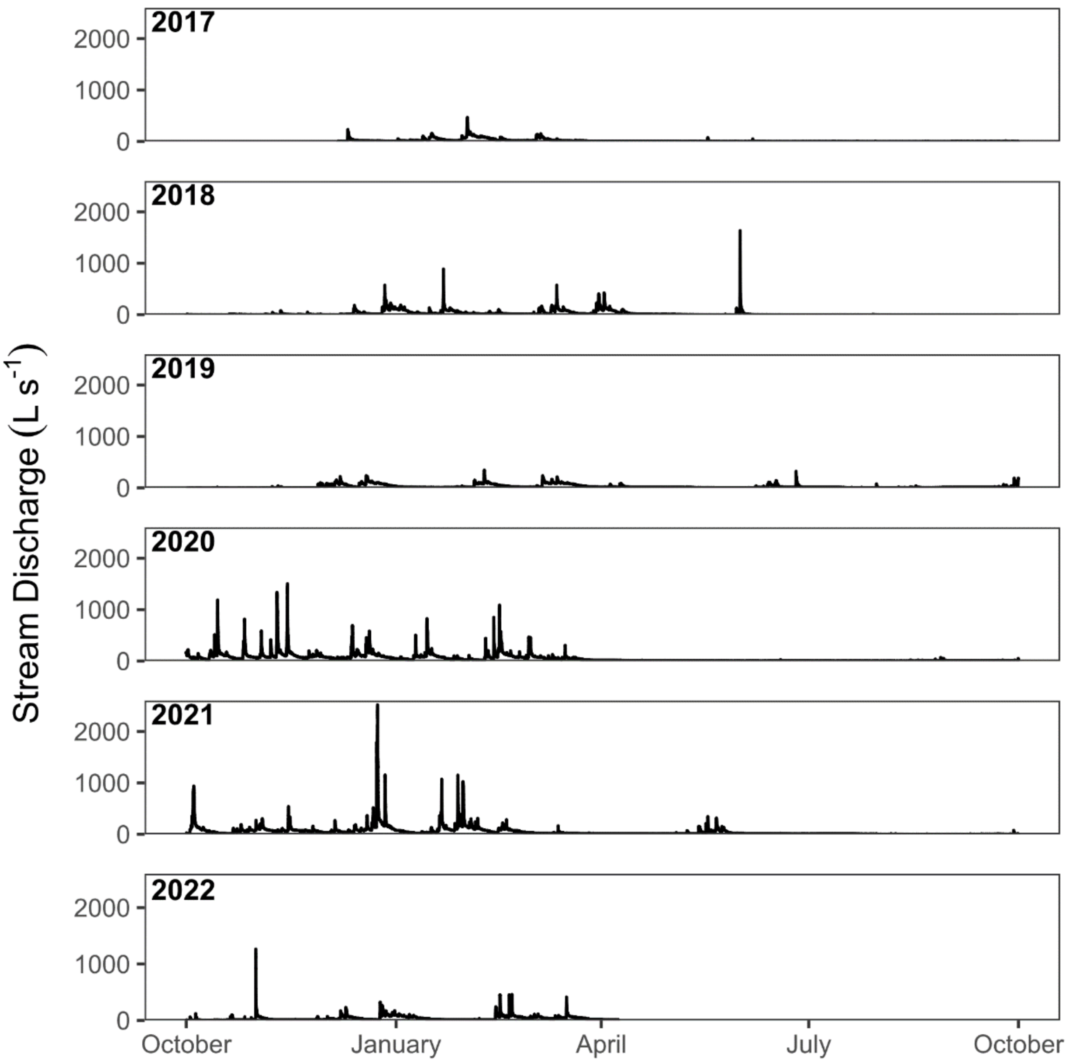


Figure 6.13: Stream discharge (L s^{-1}) at The Heath (TH) in each water year of the monitoring period. The data series starts in December 2016 and ends in April 2022. NB. Y-axis scale goes beyond that of CM in the South sub-catchment (Figure 12).

Upstream The Heath:

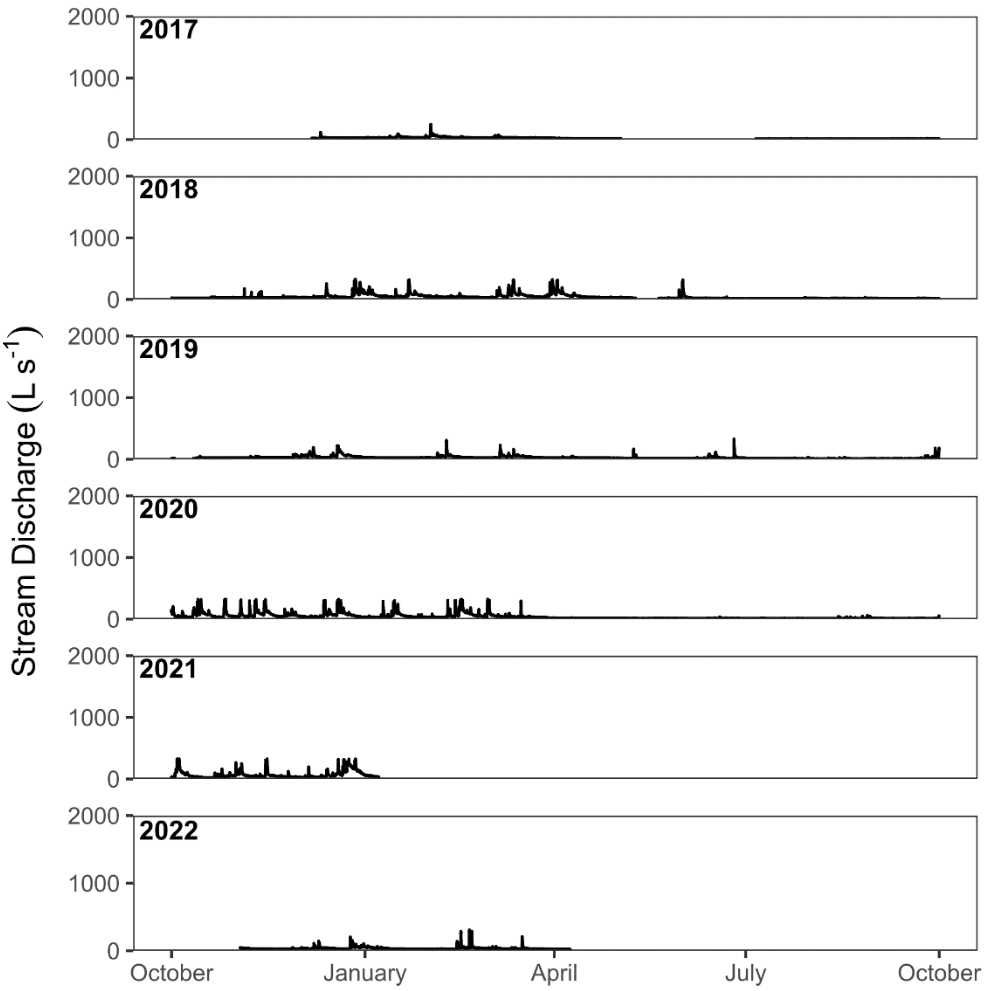


Figure 6.14: Stream discharge (L s^{-1}) at Upstream The Heath (UTH) in each water year of the monitoring period. The data series starts in December 2016 and ends in April 2022. NB. Y-axis scale goes beyond that of CM in the South sub-catchment (Figure 12).

Table 6.9 shows the total annual discharge (water flux) leaving each of the monitored sub-catchments for each water year. Despite considerably higher storm event peaks in the North sub-catchment, the total discharge was consistently higher from the South sub-catchment, reflecting its higher baseflow component influenced by a potentially larger sub-surface (groundwater) contributing area.

Table 6.9: Total annual discharge (million m^3) from the North and South sub-catchments during each water year.

Total Discharge (million m^3)		
Water Year	Sub-catchment	
	North	South
2017†	0.365	1.310
2018	0.646	2.075
2019	0.712	1.363
2020	1.626	2.116
2021	1.532	2.197
2022†	0.600	0.922

†Years with incomplete data (see Section A.4 'Data Coverage').

A.5.3 Annual fluxes of suspended sediment and nutrients

SSC is presented at a 5-minute resolution for each water year (1st October – 30th September) at each of the three SSC monitoring sites (Figure 6.15; Figure 6.16; Figure 6.17).

At CM, SSC was relatively low (<1000 mg L^{-1}) during the dry year of 2017. The 2021 water year was notably rich in events of high SSC reflecting the wet conditions that occurred early on in the autumn when arable fields were still largely bare.

Church Meadow:

NB. Data after January 2021 during storm events is truncated due to the sensor used from this point onwards having an upper measurement limit of ~1600 mg L^{-1}. Beyond this point event peaks are missed in instances where turbidity exceeded this.

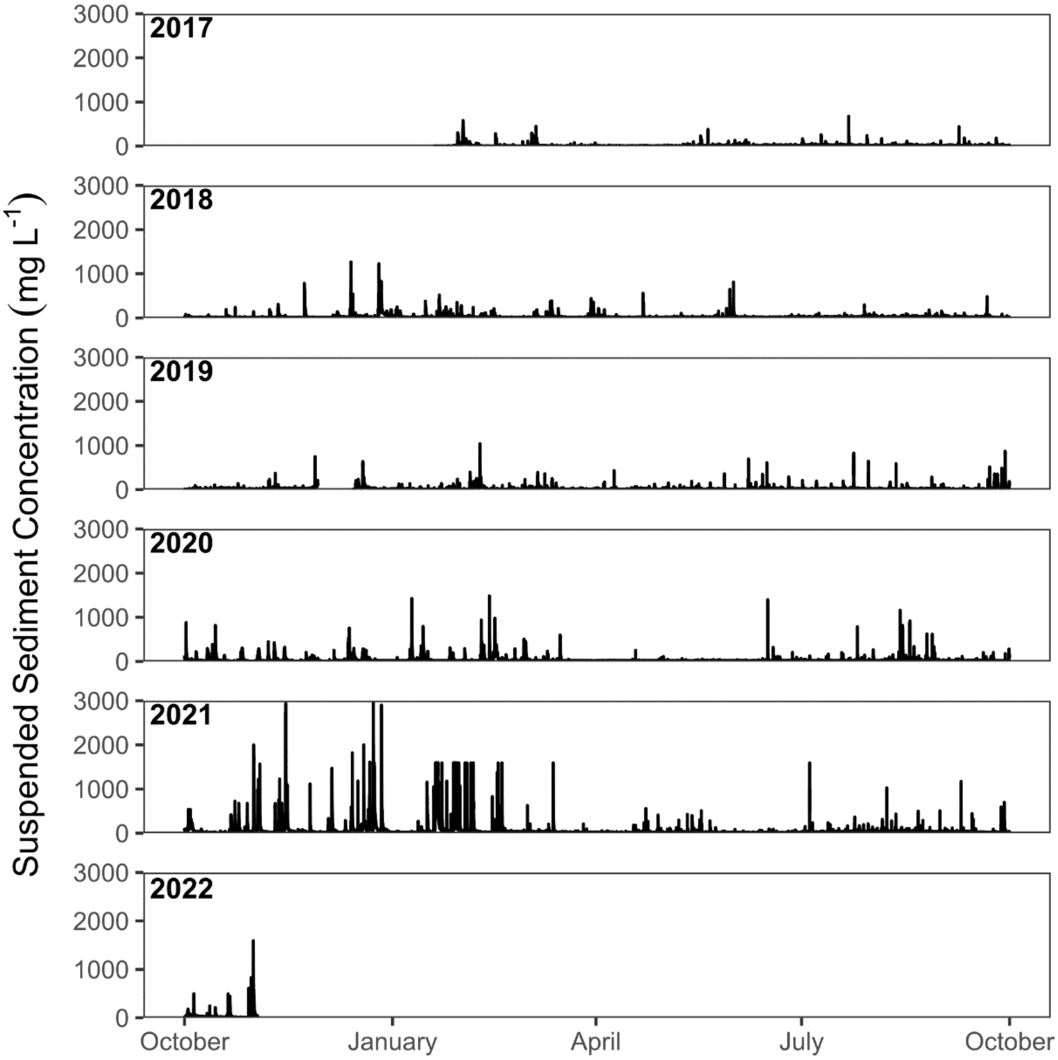


Figure 6.15: Stream suspended sediment concentration (mg L^{-1}) at Church Meadow (CM) in each water year of the monitoring period. The data series starts in January 2017 and ends in October 2022.

SSC at TH had a particularly flashy response mirroring the hydrological regime of this sub-catchment. Measured SSC was highest at this site and frequently exceeded 1000 mg L^{-1}, even reaching relatively high concentrations during the dry year of 2017.

The Heath:

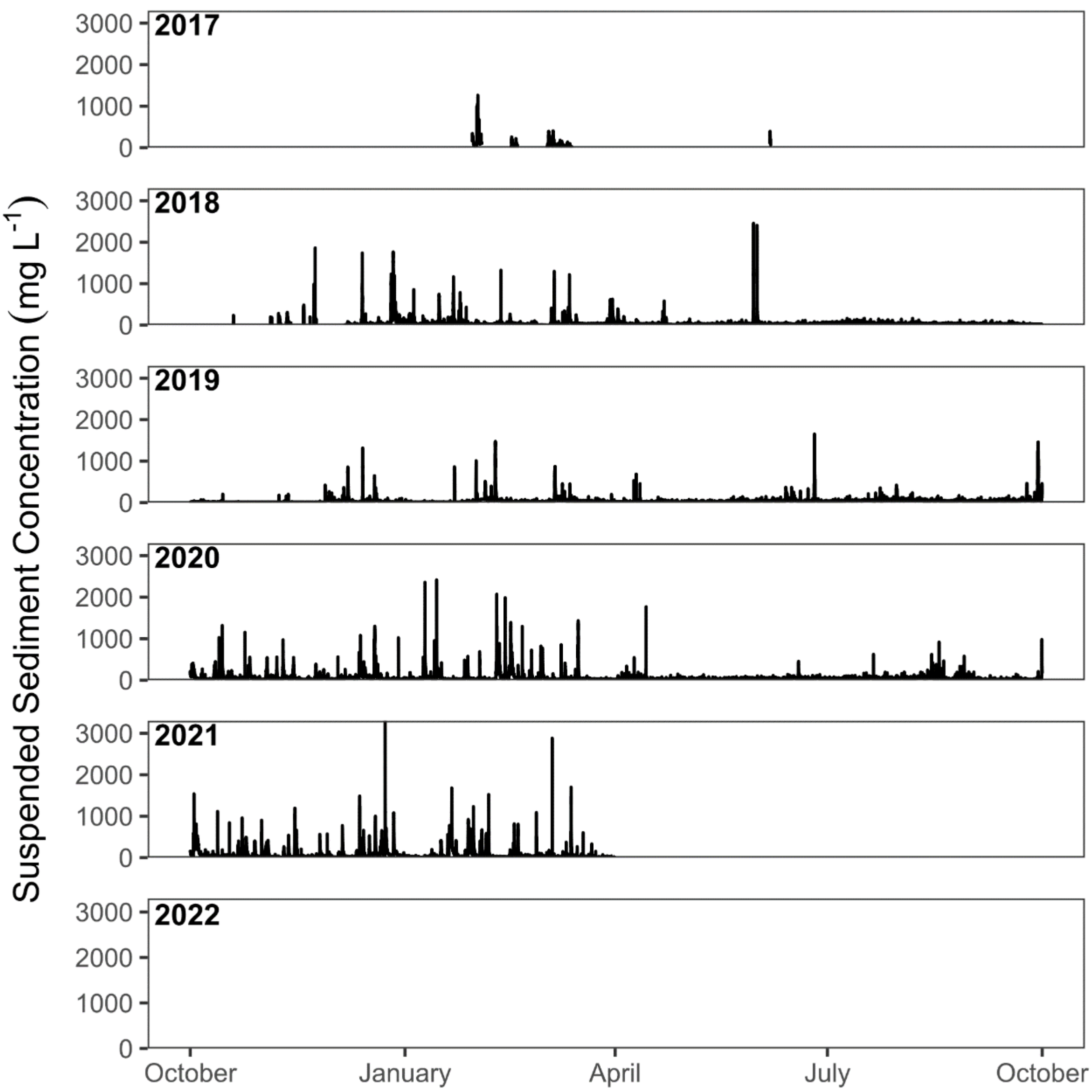


Figure 6.16: Stream suspended sediment concentration (mg L^{-1}) at The Heath (TH) in each water year of the monitoring period. The data series starts in February 2017 and ends in April 2021.

SSC at UTH is similar to TH in its response but has overall lower concentrations. This suggests that there are significant sources of sediment entering the stream between UTH and TH. Potential critical source areas are likely to include the incised and largely unvegetated channel banks in the reach just upstream of TH and the adjacent arable field.

Upstream The Heath:

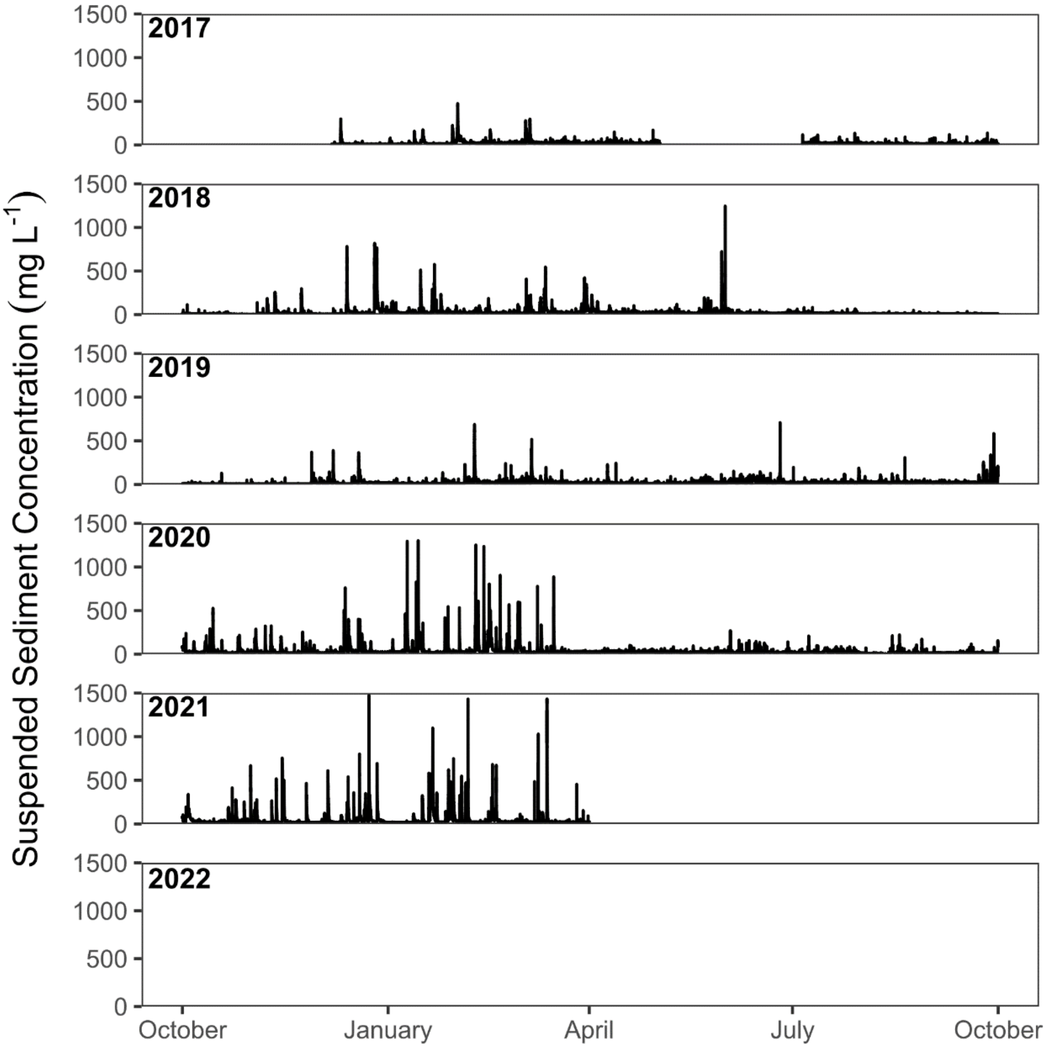


Figure 6.17: Stream suspended sediment concentration (mg L^{-1}) at Upstream The Heath (UTH) in each water year of the monitoring period. The data series starts in December 2016 and ends in April 2021.

Table 6.10, Table 6.11 and

Table 6.12 show the fluxes of suspended sediment, total phosphorus, and particulate organic carbon leaving the monitored sub-catchments in each water year.

Table 6.10: Suspended sediment flux (tonnes) and lower/upper uncertainty bounds in round brackets from the North/South sub-catchments during each water year. Values in square brackets show fluxes for periods that do not span the full water year, to allow sub-catchment comparison.

Suspended Sediment Flux (t)		
Water Year	**Sub-catchment**	
	North	**South**
2017†	29.99 (27.05 – 32.93)	24.42 (22.58 – 26.30)
2018	112.45 (96.03 – 128.82)	101.40 (93.88 – 109.11)
2019†	60.34 (54.50 – 69.57)	48.69 (44.13 – 53.48)
2020	259.09 (232.23 – 288.25)	135.29 (129.30 – 141.37)
2021†	[280.31 (251.44 – 311.34)]	358.26 (341.12 – 375.79) [344.85 (328.40 – 361.72)]

†Years with incomplete data (see Section A.4.2 'Data Coverage').

Table 6.11: Total phosphorus flux (kg) from the North/South sub-catchments during each water year. Values in brackets show fluxes for periods that do not span the entire water year to allow sub-catchment comparison.

Total Phosphorus Flux (kg)		
Water Year	**Sub-catchment**	
	North	**South**
2017†	107.02	92.23
2018	304.16	265.30
2019†	214.31	135.20
2020	719.96	331.10
2021†	(725.02)	757.58 (714.72)

†Years with incomplete data (see Section A.4.2 'Data Coverage').

Table 6.12: Particulate organic carbon flux (tonnes) from the North/South sub-catchments during water years. Values in brackets show fluxes for periods that do not span the entire water year to allow sub-catchment comparison.

Particulate Organic Carbon Flux (t)		
Water Year	Sub-catchment	
	North	South
2017†	2.57	2.46
2018	9.63	10.20
2019†	5.17	4.90
2020	22.18	13.61
2021†	(24.00)	36.05 (34.70)

†Years with incomplete data (see Section A.4.2 'Data Coverage').

Suspended fluxes of sediment, total phosphorus, and organic carbon show broadly similar patterns across the observed water years. Fluxes varied greatly between the years, largely reflecting the changing hydrometeorological conditions. Suspended sediment and POC fluxes were consistently higher from the North sub-catchment up to the 2020 water year. However in 2021 the fluxes were higher from the South sub-catchment, with the exception of TP. This is a result of the chronically elevated dissolved P component in the North sub-catchment.

A.5.4 Sediment and nutrient accumulation in FSAs

The ability of the FSAs to trap sediment and associated nutrients was assessed through multiple monitoring methods with varying degrees of success. The key findings presented within this section are taken from a research article published in the *Earth Surface Processes and Landforms* journal (Robotham et al., 2023).

<u>Sediment and nutrient storage:</u>

The accumulated masses estimated from the sediment depth surveying and core sampling are given in Table 6.13. The total sediment, TP and OC captured by the FSA and pond features varied by two orders of magnitude, ranging from 0.2 to 20.1 tonnes of sediment during the 2 to 3 years since construction. Bulk density of the accumulated sediment had a mean of 0.69 ± 0.23 g cm^{-3} for online features and 0.93 ± 0.22 g cm^{-3} for offline features. Cumulatively, the 13 features within the south sub-catchment stored 83 tonnes of sediment with a total volume of 108.8 m^3. The FSAs were most effective in trapping sediment, with 14.7 % of the total sediment flux and 14.1 % of the fine (clay and silt) sediment flux stored compared to only 9.5 % and 7.5 % of the TP and POC fluxes respectively.

Table 6.13: Fluxes (±95 % CI), masses of accumulated sediment (t), Total Phosphorus (kg), and Organic Carbon (t) in storage features and their equivalent proportion of the total suspended sediment, fine suspended sediment, TP, and particulate OC fluxes leaving the 3.4 km^2 South sub-catchment.

†Total excludes P11_OLP as this feature is in the North sub-catchment.

Storage Feature	Time period	Rainfall (mm)	Total sediment flux (t)	TP flux (kg)	POC flux (t)	Stored sediment (t)	Stored TP (kg)	Stored POC (t)	Total sediment flux stored (%)	Fine sediment flux stored (%)	TP flux stored (%)	POC flux stored (%)	Sub-catchment area drained (%)
P0_OLP	Feb 19– Mar 21	2126	498±24	1095±50	50±1	5.8	8.0	0.3	1.2	1	0.7	0.6	12.1
P1_OLP						6.2	14.4	0.4	1.3	1.4	1.3	0.8	9.0
P2						0.4	0.5	0.03	0.07	0.06	0.04	0.06	0.3
P3						20.1	28.9	1.1	4.0	3.7	2.6	2.2	0.1
P4	Feb 18– Mar 21	2810	565±30	1278±65	57±2	0.3	0.4	0.01	0.05	0.05	0.03	0.02	1.1
P5						7.0	11.0	0.3	1.2	1.3	0.9	0.5	1.2
P6						8.7	14.2	0.4	1.6	1.8	1.1	0.7	1.9
P7						10.6	16.7	0.4	1.9	2.1	1.3	0.7	2.8
P8						0.2	0.2	0.01	0.03	0.04	0.02	0.02	0.3
P9						0.6	0.6	0.02	0.1	0.1	0.05	0.03	5.9
P10_OLP	Feb 18– Jan 20	1634	160±10	417±26	16±1	4.6	5.5	0.2	2.9	2.5	1.3	1.3	8.8
P10_US_OLP						10.7	13.1	0.8	6.7	4.6	3.2	5.0	
P10_DS_OLP	Feb 18– Jun 20	1944	207±12	533±30	21±1	7.8	8.5	0.3	3.8	3.6	1.6	1.4	
P11_OLP	Feb 19– Mar 21	2126	605±102	1614±250	52±6	3.8	5.9	0.2	0.6	0.7	0.4	0.4	0.2
Total†		2810	565±30	1278±65	57±2	83.0	121.8	4.3	14.7	14.1	9.5	7.5	43.5

NB. P1_OLP is also referred to as OLP1 and P1 in different report sections, to keep consistent with linked publications. The three P10_OLPs are also referred to as Upstream, Central and Downstream ponds in Section 4.5 and as one entity as OLP10, P10 and elsewhere in the report, to keep consistent with linked publications.

Factors influencing accumulation rates:

Hydrology

The hydrology and filling of the different FSAs and ponds is notably varied, with some features being permanently ponded and thereby always having an antecedent storage component. On the other hand, some features only fill during rainfall events and then drain down and dry shortly after. P3 showed the greatest retention of water with 60 % of its capacity exceeded 50 % of the time, equating to a median storage volume of 338 m^3. P8 filled infrequently and only ever filled to 12 % (68 m^3) of its potential storage capacity during this period. P6 also had a hydrologically flashy filling regime but stored significantly more water, reaching 26 % capacity (688 m^3), one order of magnitude greater than P8. In comparison P5 exhibited less flashy behaviour, sustaining water storage for a greater duration and at its peak filling to 1475 m^3, 42 % of its potential capacity.

Connectivity & design

On average, the sediment accumulation rate was 3.3 times higher in on-line features (20.8 ± 9.8 kg m^{-2} y^{-1}) than in offline features (6.3 ± 5.2 kg m^{-2} y^{-1}) when taking into account the ponded area of each feature. The width-to-length ratio of features explained some of the variation in accumulation rates, with positive relationships observed for both sediment (R^2=0.42, p<0.05) and TP accumulation (R^2=0.54, p<0.01). Width-to-length ratios were generally low and ranged from ~0.25 to 2.0, with P1_OLP having the highest ratio. Contributing area was also found to positively influence sediment accumulation rate (R^2=0.49, p<0.05). Differences in accumulation rate were better explained by event contributing area which broadly clusters the offline features into those activated by leaky barriers and those that were not (Figure 6.18).

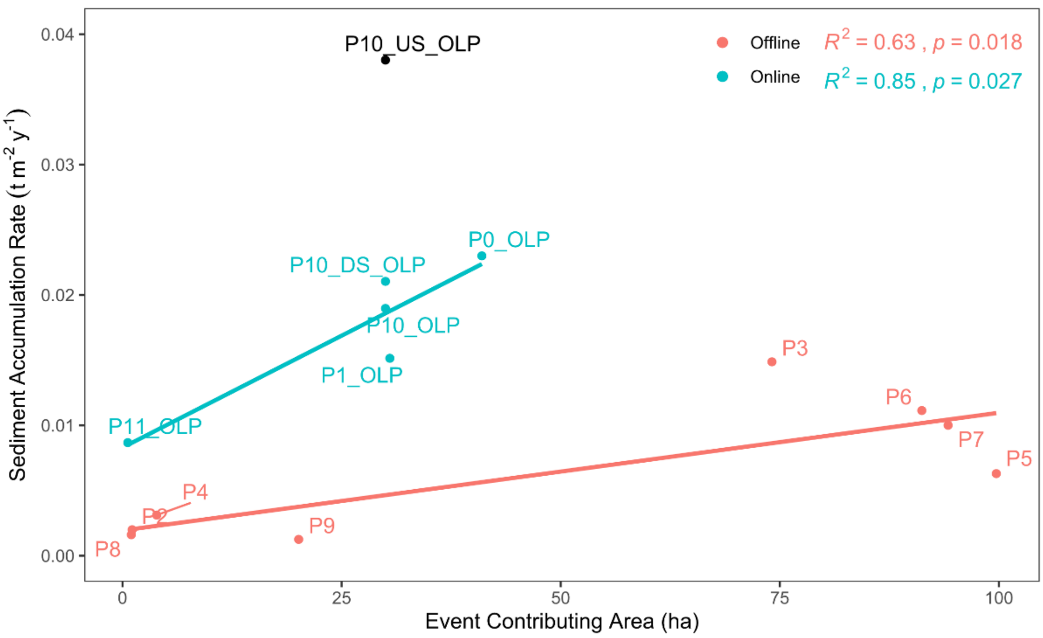


Figure 6.18: Linear regressions between event contributing area (ha) and sediment accumulation rate ($t\ m^{-2}\ y^{-1}$) for offline and online features. P10_US_OLP is excluded from the regression.

Features such as P9 were never observed to fill from overbank flows whereas P6 was frequently observed to do so during event peaks in winter storms (Figure 6.19). Overbank flows by the leaky barrier and spillway connected to P6 occurred in over 20 storm events between October 2019 and March 2021. The threshold for overbank flow was never reached at the P9 spillway; even at the peak of the highest magnitude event in December 2020 the water level was still 0.3 m below the threshold. During this event, peak storage in P6 reached over double the volume in P9. The timing of overbank flow was generally well aligned with stream SSC, allowing the highest sediment load to be diverted into P6 during event peaks.

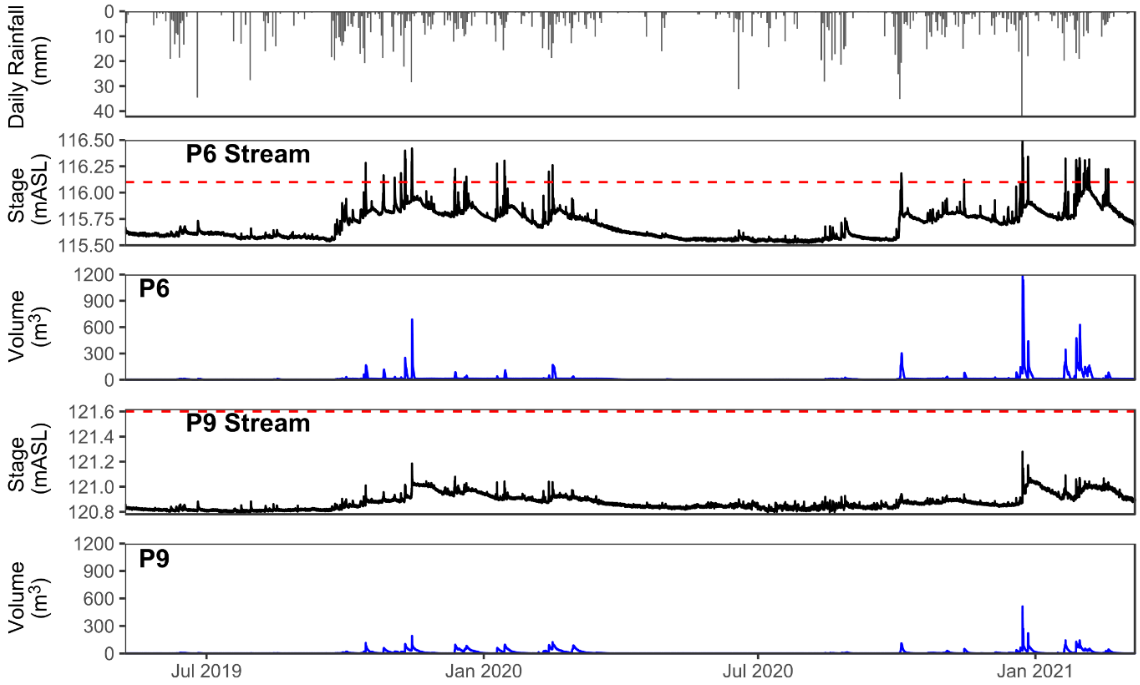


Figure 6.19: Time-series of daily rainfall (mm), and stream stage (mASL) at leaky barriers and water volume (m^3) in NFM storage features P6 and P9. Dashed red lines indicate the threshold at which spillways are activated. mASL = metres above sea level.

<u>Sediment enrichment:</u>

Sediments deposited within FSA and pond features were found to be significantly enriched in TP (paired samples *t*-test, $p<0.01$, n=14), with an average concentration 1.5 times greater than the surface soil in contributing areas. The highest TP enrichment ratio of 2.66 was observed for P1_OLP. On average the sediment was composed of 86% silt and clay particles. Enrichment of clay was typically higher in the offline features. The opposite trend was observed for sand content. In terms of OC enrichment, there were no apparent differences between the offline and on-line features.

<u>Reductions in FSA storage capacity:</u>

In the 2 to 3 years since construction, the majority of FSAs did not lose significant volumes of their maximum storage capacity as a result of sediment loading (Table 6.14). Average annual losses in storage capacity during the monitoring period ranged from 0.01% in P4 and P8 up to 12.9% in P0_OLP. In order to maintain their ability to fill and drain effectively during and after events, storage features require their outlets to remain sufficiently above the level of accumulated sediment, thereby helping to prevent siltation within drains. When considering the remaining storage capacity up to the drain height of features, the accumulated sediment volumes had a much greater impact.

Potential storage for water and sediment is most reduced in the online features, with P10_US_OLP, P10_OLP, and P0_OLP all predicted to fill beyond their outlet drain heights within 10 years (based on current accumulation rates). Whilst having a high accumulation rate, at a 10-year timescale P1_OLP is predicted to still retain >50% of its storage capacity up to its outlet. P1_OLP with its deeper design had a mean water depth of 0.71 m during autumn and winter in contrast to only 0.3 m in P10_OLP with its shallow design and comparatively low outlet elevation. Interestingly, loss of storage capacity in P11 was negligible due to the sediment accumulation rate being too small to quantify even after over 2 years since construction. However, P11_OLP (connected to the outflow of P11) lost almost 5% of its total storage within the same period.

Table 6.14: Percentage reductions in the maximum storage capacity and the storage capacity up to drain heights of FSAs since their construction. 10-year reductions in storage capacity are estimates based on the measured rates of accumulation during the monitoring period. NB. It was not possible to calculate storage capacities up to drain heights for all features.

Storage Feature	**Storage capacity to drain (m^3)**	**Reduction in max. storage capacity (%)**	**Reduction in storage capacity to drain (%)**	**Annual reduction in max. storage capacity (%)**	**10-year reduction in max. storage capacity (%)**	**10-year reduction in storage capacity to drain (%)**
P0_OLP	-	27.34	-	12.9	100	-
P1_OLP	144.01	2.95	9.00	1.39	11.4	42.4
P2	-	0.57	-	0.27	2.7	-
P3	-	4.23	-	2.00	20.0	-
P4	113.74	0.03	0.26	0.01	0.1	0.8
P5	121.93	0.19	5.40	0.06	0.6	17.3
P6	130.46	0.38	7.69	0.12	1.2	24.7
P7	310.91	0.49	4.30	0.16	1.6	13.8
P8	17.37	0.04	1.30	0.01	0.1	4.2
P9	89.55	0.07	0.72	0.02	0.2	2.3
P10_OLP	20.24	8.18	36.36	4.22	42.2	>100
P10_US_OLP	25.52	19.84	54.43	10.23	>100	>100
P10_DS_OLP	48.46	10.41	20.41	4.37	43.7	85.8
P11	-	0.00	-	0.00	-	-
P11_OLP	-	4.75	-	2.24	22.4	-

<u>Sediment deposition pins:</u>

Sediment deposition pins proved to be unsuitable for assessing the accumulation of fine sediment within the FSA and pond features for the multiple reasons listed in Section A.3.10. For future monitoring programmes we recommend that sediment deposition is best quantified through surveying of sediment depths alongside core sampling to avoid issues surrounding large degrees of uncertainty that are associated with the deposition pin method. However, if deposition pins are used, they are more likely to yield meaningful results when:

- Placed within permanently wet features
- Placed within locations with soil types less prone to swelling/shrinking
- Placed away from livestock and publicly accessible areas (vandalism/removal)

A.5.5 Sediment and nutrient attenuation in online ponds

Detailed monitoring of the on-line ponds (P10; Tears of Bruern) allowed their effectiveness for sediment and nutrient retention to be analysed. These analyses are detailed in full in a peer-reviewed scientific journal article (Robotham *et al.*, 2021), of which the abstract is given below and the key findings are presented within this report section.

Abstract:

The creation of ponds and wetlands has the potential to alleviate stream water quality impairment in catchments affected by diffuse agricultural pollution. Understanding the hydrological and biogeochemical functioning of these features is important in determining their effectiveness at mitigating pollution. This study investigated sediment and nutrient retention in three connected (on-line) ponds on a lowland headwater stream by sampling inflowing and outflowing concentrations during base and storm flows. Sediment trapping devices were used to quantify sediment and phosphorus accumulations within ponds over approximately monthly periods. The organic matter content and particle size composition of accumulated sediment were also measured. The ponds retained dissolved nitrate, soluble reactive phosphorus and suspended solids during baseflows. During small to moderate storm events, some ponds were able to reduce peak concentrations and loads of suspended solids and phosphorus; however, during large magnitude events, resuspension of deposited sediment resulted in net loss. Ponds filtered out larger particles most effectively. Between August 2019 and March 2020, the ponds accumulated 0.306 t ha^{-1} sediment from the 30 ha contributing area. During this period, total sediment accumulations in ponds were estimated to equal 7.6% of the suspended flux leaving the 340 ha catchment downstream. This study demonstrates the complexity of pollutant retention dynamics in on-line ponds and highlights how their effectiveness can be influenced by the timing and magnitude of events.

Highlights:

- On-line ponds significantly reduced concentrations of biologically-available nutrients on average (nitrate by 5 % and soluble reactive phosphate by 29 %).
- Overall, on-line ponds acted as a net sink of sediment despite some instances of sediment resuspension/flushing during larger storms.
- On-line ponds require frequent maintenance (approximately every 2 years) for efficient functioning.

Baseflows:

Under baseflow conditions (outside of storm events), 19 sets of water chemistry samples were taken between March 2019 and March 2020. Significant differences between inlet and outlet concentrations were found for dissolved nitrate (NO_3^-), SRP, SSC, and VSC which all showed a decrease in mean concentration at the outlet (paired samples t-test, $p < 0.01$, $n = 19$; Figure 6.20). Table 6.15 gives the average concentrations at the inflow and outflow of the on-line pond system along with the average, minimum and maximum removal efficiencies of each determinand.

Table 6.15: Mean (±SD) inflow and outflow concentrations (mg L^{-1}), and mean (±SD), minimum, and maximum Removal Efficiency (%) of the on-line pond system for water quality determinands sampled during baseflow conditions. Determinands that showed statistically significantly attenuation are shown in bold font.

Determinand	Mean Inflow Concentration (mg L^{-1})	Mean Outflow Concentration (mg L^{-1})	Mean Removal Efficiency (%)	Minimum Removal Efficiency (%)	Maximum Removal Efficiency (%)
SRP	0.008 ± 0.006	0.005 ± 0.004	29 ± 37	-100	74
TDP	0.041 ± 0.023	0.038 ± 0.02	3 ± 43	-117	68
PP	0.04 ± 0.04	0.052 ± 0.059	-237 ± 579	-2100	95
TP	0.081 ± 0.048	0.089 ± 0.069	-34 ± 125	-314	77
NH_4^+	0.023 ± 0.026	0.024 ± 0.025	-61 ± 118	-400	73
NO_3^-	36.56 ± 3.585	34.903 ± 4.4	5 ± 6	-2	23
F^-	0.068 ± 0.023	0.067 ± 0.024	0 ± 18	-23	35
Cl^-	16.913 ± 2.382	16.835 ± 2.045	0 ± 7	-23	14
SO_4^{2-}	17.006 ± 2.652	16.904 ± 2.488	0 ± 9	-29	18
SSC	21.2 ± 4.153	13.464 ± 6.943	32 ± 24	-17	70
VSC	7.09 ± 4.153	3.901 ± 1.469	40 ± 15	15	66

NB. Nitrite (NO_2^-) was excluded from the statistical tests due to a majority (67 %) of both inlet and outlet samples measuring 0 mg NO_2^- L^{-1}.

Under baseflow conditions removal efficiencies exhibited considerable variability between the water quality determinands. Removal ranged from extreme negative values (indicating net export from the pond system) for PP, to more consistently positive values (indicating net retention) for SSC and VSC.

Overall, the majority of average removal efficiencies for the sampling period were positive, with the exceptions being PP, TP, and NH_4^+.

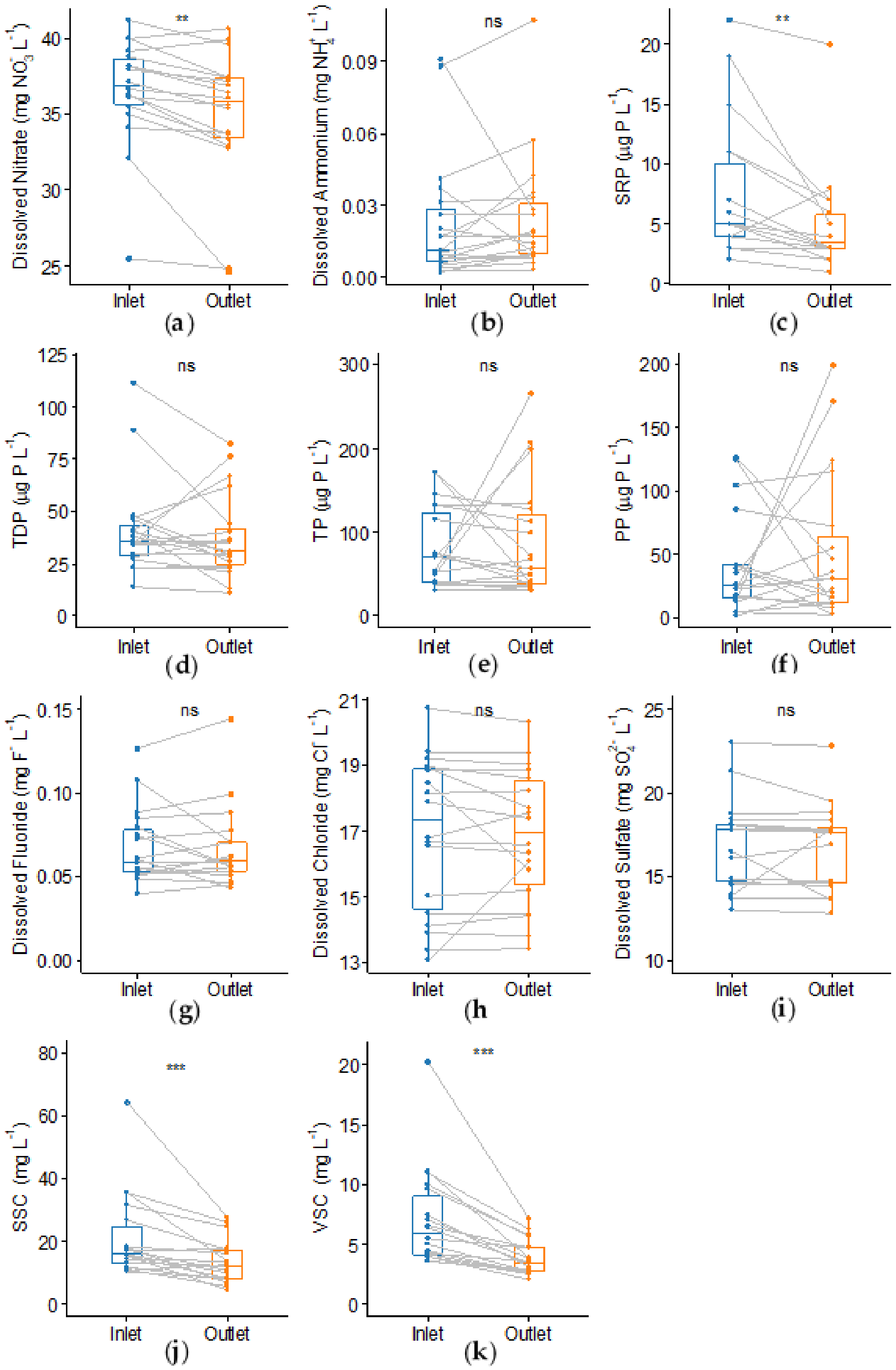


Figure 6.20: Boxplots of paired on-line pond inlet and outlet concentrations for water quality determinands. Median values are represented by horizontal lines. Significance levels for results of paired samples t-tests are indicated with: *** ($p < 0.001$), ** ($p < 0.01$), ns ($p > 0.05$).

Storm events:

Four storm events were captured between March 2019 and February 2020 (Table 6.16); however it was not always possible to trigger all four automatic samplers for every storm. The event captured in February was during Storm Dennis and had the highest rainfall (total monthly rainfall in February was 170 % above average for the area). Estimated peak discharge was highest during the November event, with an estimated return period of 5.5 years. The API gives an indication of the likely soil moisture conditions, and was found to be highest prior to the October 14th event following a rapid wetting of the catchment at the end of September.

Table 6.16: Mean (±SD) SSC (mg L^{-1}) for each pond monitoring site during four storm events, estimated discharge (L s^{-1}) prior to the event and at its peak, and the sampling duration (hours). Rainfall (mm) is the total event precipitation and Antecedent Precipitation Index (API) (mm) is given for the day prior to each event.

Storm Event	**Mean SSC (mg L^{-1})**				**Sampling Duration (h)**	**Estimated Discharge (L s^{-1})**		**Rainfall (mm)**	**API (mm)**
	Upstream Pond Inlet	**Upstream Pond Outlet**	**Central Pond Outlet**	**Downstream Pond Outlet**		**Pre-event**	**Peak**		
12th/13th March 2019	45 ± 47	30 ± 33	29 ± 27	35 ± 30	23	8.9	18.7	8.8	51.9
14th October 2019	258 ± 365	161 ± 152	143 ± 94	126 ± 55	5.75	8.4	58.6	23.1	104.1
14th November 2019	92 ± 67	27 ± 11	24 ± 7	-	5.75	9.2	74	31.8	97.6
15th/16th February 2020	-	87 ± 63	98 ± 79	-	23	12.3	55.7	32.2	64.8

The March 2019 event was the smallest in magnitude, with the least rainfall and lowest API, but still resulted in a peak SSC of > 200 mg L^{-1} at the inlet to the Upstream Pond, with the peak then being reduced by ~50 % downstream at the outlet of the Downstream Pond (Figure 6.21d). Streamflow responded rapidly to rainfall with a lag time of less than two hours (Figure 6.21a/4b). The response of suspended sediment was partially staggered, with lag times increasing downstream at each monitoring point except for water leaving the Downstream Pond which peaked simultaneously with water leaving the Central Pond. SSC at the Downstream Pond outlet had a less steep gradient on the falling limb compared to the other monitoring locations.

The response of TP and PP closely reflected that of SSC, however TDP did not exhibit a rising limb and remained relatively constant at the inlet and outlet of the Upstream Pond (Figure 6.21e - 14g). TDP shows a somewhat different pattern at the outlet of the Central Pond with the concentration abruptly dropping below 10 µg P L^{-1} after 19:00pm. At the Downstream Pond outlet, TDP remained under 20 µg P L^{-1}, which was lower than both the inlet and outlet of the Upstream Pond which almost always stayed above 20 µg P L^{-1}. On the rising/receding limbs of the event, PP accounted for the majority (57-91 %) of transported P, after which TDP at the inlet and outlet of the Upstream Pond exceeded the particulate fraction.

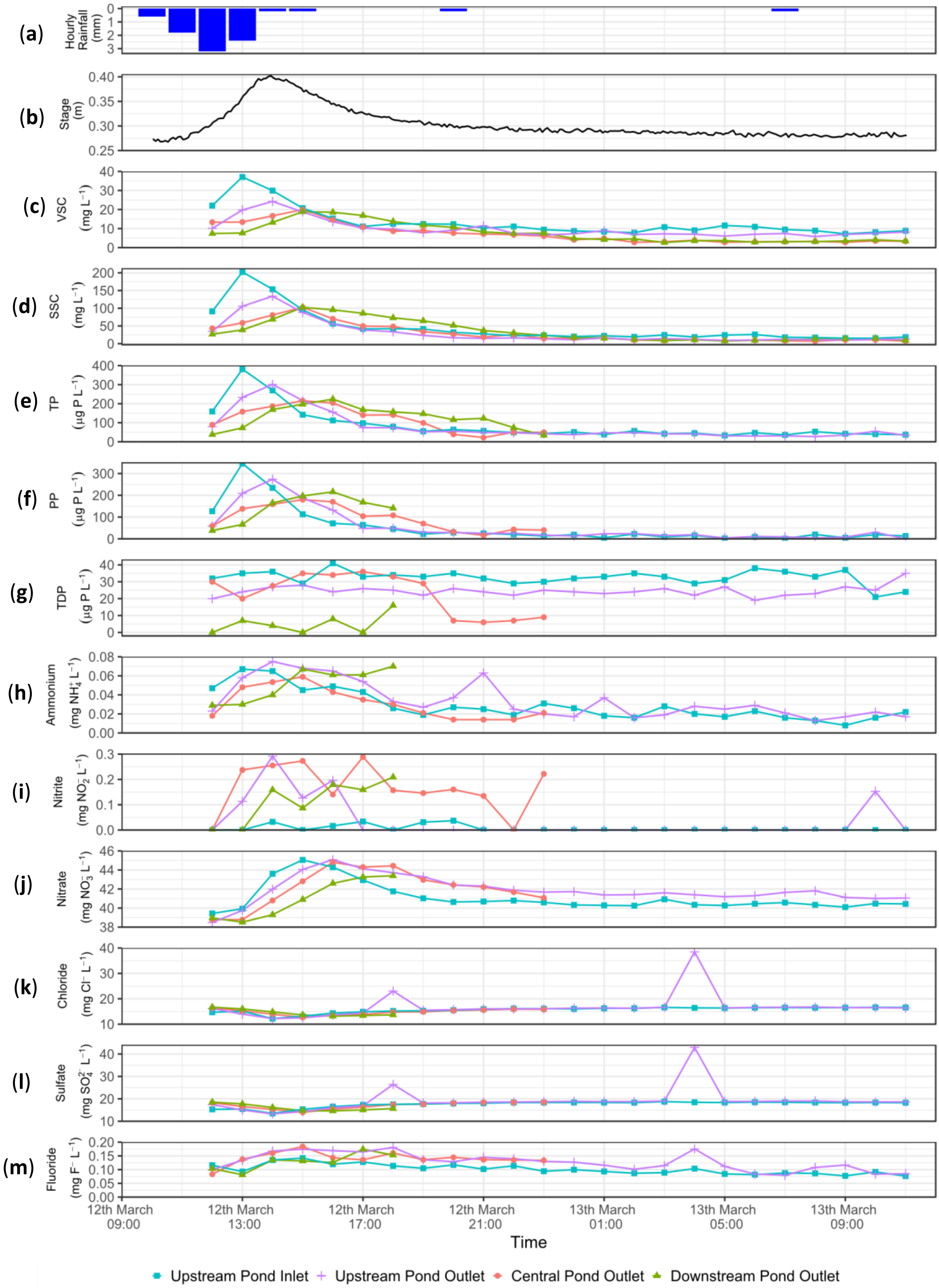


Figure 6.21: Time-series during a storm event on 12th/13th March 2019 showing: (a) Hourly Rainfall (mm); (b) Stage (m) in the Central Pond; and concentrations of water quality determinands: (c) VSC and (d) SSC (mg L-1); (e) TP, (f) PP, and (g) TDP (µg P L^{-1}); (**h**) ammonium (mg NH_4^+ L^{-1}); (**i**) Nitrite (mg NO_2^- L^{-1}); (**j**) Nitrate (mg NO_3^- L^{-1}); (**k**) chloride (mg Cl^- L^{-1}); (**l**) Sulfate (mg SO_4^{2-} L^{-1}); and (**m**) Fluoride (mg F^- L^{-1}) at each pond inlet/outlet sampling site.

<u>On-line pond sediment quality:</u>

From manual surveying of sediment depths approximately two years after their construction, it was estimated that 13.89 m^3 of matter had accumulated in the Upstream Pond, and 7.36 m^3 in the Central Pond. This meant that the Upstream Pond had filled ~20 % of its total capacity, and the Central ~8 %. At the time of surveying in January, depths in the Downstream Pond were unable to be measured due to the water level being too high. The Downstream Pond was able to be surveyed in July at the earliest (due to the Covid-19 pandemic), and had accumulated 9.89 m^3 of matter, equating to ~10 % of its total capacity.

Sediment traps were deployed continuously from August 2019, with sediment collection taking place on six occasions until March 2020 to capture run-off during the wet season. Throughout this 7-month period, rates of accumulation were variable, but the Upstream Pond had the highest overall accumulation, and the Downstream Pond had the lowest (Table 6.17). Sediment accumulation rates varied considerably between the trap placements within ponds as shown by the large standard deviations. Over the whole period, the ponds accumulated 6.1 % of the downstream catchment silt + clay flux, and 7.6 % of all suspended sediment. P accumulation in ponds generally showed the same pattern as sediment, and on average made up ~ 0.1 % of the total accumulated mass (Table 6.18). Total accumulated P in ponds only made up 3.2 % of the Downstream Catchment P flux. LOI showed that deposited sediments were largely made up of inorganic matter (IOM), accounting for > 75 % of the accumulated sediment mass throughout the sampling period. The OM content ranged from 10 – 23 % and consistently decreased downstream along the pond sequence in each deployment period. OM content was highest between August and October. OM content of pond sediment was significantly enriched compared to the soil in the arable fields of the contributing area which had an OM content of 5 – 7 %, typical of the arable fields in this sub-catchment.

Table 6.17: Accumulated sediment (±SD) (t) in each pond, all the ponds, and only the silt + clay (< 63 µm) for trap monitoring periods. Accumulated sediment yield (t ha^{-1}) for all ponds from the contributing area (30 ha), the flux of sediment and silt + clay (t) and the exported yield (t ha^{-1}) from the Downstream Catchment area (340 ha) are given for the same periods.

Monitoring Period	**Days**	**Rainfall (mm)**	**Accumulated Sediment (t)**					**All Ponds Sediment Yield (t ha^{-1})**	**Catchment Sediment Flux (t)**	**Catchment silt+clay Flux (t)**	**Catchment Sediment Yield (t ha^{-1})**
			Upstream Pond	**Central Pond**	**Downstream Pond**	**All Ponds**	**All Ponds (silt+clay)**				
08/08/2019 – 30/08/2019	22	62	0.56 ± 0.27	0.54 ± 0.35	0.33 ± 0.35	1.43 ± 0.56	1.01	0.048	0.34	0.3	0.001
30/08/2019 – 03/10/2019	34	128	0.63 ± 0.55	0.17 ± 0.05	0.25 ± 0.04	1.06 ± 0.55	0.71	0.035	7.4	6.47	0.022
03/10/2019 – 30/10/2019	27	132	0.69 ± 0.27	0.32 ± 0.11	-	1.01 ± 0.29	0.65	0.034	19.06	16.66	0.056
30/10/2019 – 04/12/2019	35	140	0.63 ± 0.37	0.39 ± 0.2	-	1.02 ± 0.42	0.67	0.034	21.93	19.16	0.065
04/12/2019 – 22/01/2020	49	167	0.67 ± 0.27	0.82 ± 0.28	0.67 ± 0.23	2.16 ± 0.45	1.57	0.072	32.79	28.65	0.096
22/01/2020 – 12/03/2020	50	177	0.98 ± 0.35	1.05 ± 0.33	0.49 ± 0.31	2.52 ± 0.57	1.77	0.084	38.63	33.76	0.114
Total	217	871	4.15 ± 0.89	3.29 ± 0.6	1.74 ± 0.52	9.18 ± 1.19	6.38	0.306	120.18	104.99	0.353

Table 6.18: Accumulated phosphorus (±SD) (kg) in each pond and all three ponds for sediment trap monitoring periods. Accumulated P yield (kg ha^{-1}) for all ponds from the contributing area (30 ha), the flux of P (kg) and the exported P yield (kg ha^{-1}) from the Downstream Catchment area (340 ha) are given for the same periods.

Monitoring Period	Days	Rainfall (mm)	Accumulated P (kg)				All Ponds P Yield (kg ha^{-1})	Catchment P Flux (kg)	Catchment P Yield (kg ha^{-1})
			Upstream Pond	Central Pond	Downstream Pond	All Ponds			
08/08/2019 – 30/08/2019	22	62	0.58 ± 0.27	0.51 ± 0.34	0.29 ± 0.28	1.38 ± 0.52	0.046	1.06	0.003
30/08/2019 – 03/10/2019	34	128	0.69 ± 0.55	0.22 ± 0.08	0.27 ± 0.04	1.18 ± 0.56	0.039	16.42	0.048
03/10/2019 – 30/10/2019	27	132	0.65 ± 0.18	0.36 ± 0.12	-	1.01 ± 0.22	0.034	43.87	0.129
30/10/2019 – 04/12/2019	35	140	0.56 ± 0.29	0.4 ± 0.19	-	0.96 ± 0.35	0.032	54.7	0.161
04/12/2019 – 22/01/2020	49	167	0.6 ± 0.22	0.81 ± 0.27	0.69 ± 0.25	2.1 ± 0.43	0.07	77.64	0.228
22/01/2020 – 12/03/2020	50	177	0.91 ± 0.22	0.94 ± 0.48	0.42 ± 0.27	2.27 ± 0.59	0.076	87.91	0.259
Total	**217**	**871**	**3.99 ± 0.77**	**3.24 ± 0.69**	**1.68 ± 0.46**	**8.91 ± 1.13**	**0.297**	**281.6**	**0.828**

A.5.6 Water storage in FSAs during storm events and estimated reductions in catchment outlet flows

FSA water level data have been analysed in order to assess the effectiveness of the south sub-catchment (3.4 km^2) FSAs, which have an estimated combined storage capacity of 15,717 m^3. The volume of each FSA was calculated using the methods detailed in Section A.3.7. This analysis has not been done in the north sub-catchment due to absence of LiDAR or survey data for the newer FSA interventions and suspect data in one of the FSAs for much of the monitoring period (Section A.4.5).

Three of the largest storm events observed in each water year from 2019/2020 to 2021/2022 have been analysed, with return periods of up to 5.5 years. These events were identified from the CM site south sub-catchment outlet discharge time-series, estimated using the methods detailed in Section A.3.5 (Figure 6.22).

The start of each event was identified by the start of locally recorded rainfall, and the antecedent storage volume was taken at this time, as a percentage of the total storage volume available.

Each FSA volume time-series was shifted to account for the travel time to the CM site sub-catchment outlet discharge location. Travel time was calculated using the estimated mean channel velocity (Section A.3.4) and the distance from FSA outlet to the CM site.

To assess the sensitivity to travel time, the analysis was repeated with ±25 % and ±10 % travel time shifts in each FSA volume time-series. For each travel time scenario the sum of all FSA volumes was averaged to a one-hour volumes to give a better estimate of the total sustained FSA volume, due to short-term variability in the 5-minute time-series. Hourly flood volume was calculated from the CM discharge time-series and hourly periods were centred on the flood peak to ensure representation of the flood maximum.

The percentage of total FSA stored flood volume was calculated as a proportion of the FSA volume increase added to the 1-hour flood volume, giving an estimate of the reduction in downstream discharge. Results showed reductions in flood peaks across all events, ranging from 14.2% to 55.2 % during the most intense rainfall event (

Table 6.19). As the proportion of water stored is highest for the largest and most intense events this indicates a threshold effect, that higher stream water levels result in more overbank flow into the FSAs to be stored. This suggests that the NFM potential for flood water storage is greater in larger events, where overbank flow was observed at spillways and leaky barriers than during smaller events where FSAs fill from runoff. However, the smallest peaks analysed (15/02/2022 and 16/03/2022) were still reduced by over 20 %, with over 1,300 m^3 combined water storage. The hydrographs in Figure 6.23, Figure 6.24 and Figure 6.25 show successful attenuation flood peaks for all events. This is demonstrated by reduced discharge due to flood water storage during the rising limb, event peak and at the start of the falling limb, after which FSA drainage increased the discharge on the falling limb.

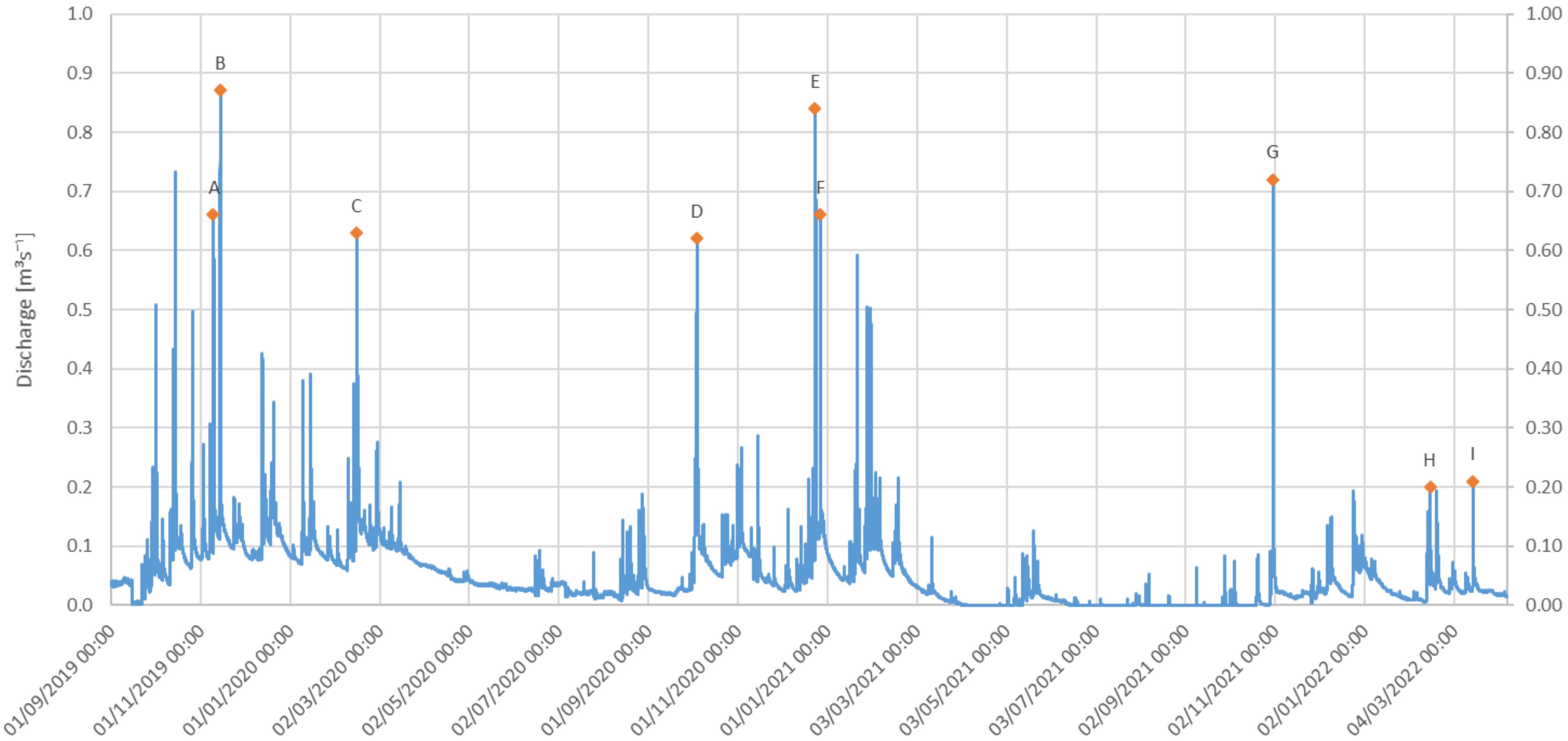


Figure 6.22: Time-series of discharge at the sub-catchment outlet monitoring site CM, with analysed storm events marked by orange symbols and assigned letter. Event dates - A: 09/11/2019, B: 14/11/2019, C: 15/02/2020, D: 04/10/2020, E: 23/12/2020, F: 27/12/202. Note that the October 2019 storm event was not analysed as the time-series data were not available for one of the larger FSAs during this period, so total flood storage volume could not be reliably estimated.

Table 6.19: Storm event summary statistics. South sub-catchment an estimated combined storage capacity of 15,717 m^3. **Instant peak discharge** = Instantaneous maximum discharge (in meters cubed per second) observed during each storm event. **Max 12 hr rainfall** = maximum amount of rainfall (mm) observed in a consecutive 12-hour period during each event. **Antecedent stored volume** = total FSA volume at the start of the event rainfall. **Max stored volume** = the maximum volume of water stored in the FSAs (i.e. the maximum total FSA volume minus the antecedent volume). **Min remaining total storage** = the amount of remaining FSA capacity at the highest FSA volume during the event. **Peak total volume** = average total FSA volume (antecedent volume plus stored volume) over the hour period centred on the flood peak. **Hourly reduction in flood peak** = stored volume as a percentage of the total flood volume plus the change in stored volume, over the hour peak of the storm event. All volumes are shown as a percentage of total storage capacity or in m^3. Volumes are estimated using the mean stream travel time (estimated using the distance of each FSA to the sub-catchment outlet monitoring site and the estimated mean channel velocity from salt dilution time-of-travel experiment detailed in Section A.3.4). The volume ranges in brackets show results from ±25% and ±10% mean travel time sensitivity analysis.

Event	Event date	Instant Peak Discharge $[m^3s^{-1}]$	Max 12 hr Rainfall [mm]	Antecedent stored volume [%]	Max stored volume [%]	Max stored volume $[m^3]$	Min remaining total storage [%]	Min remaining total storage $[m^3]$	Peak total volume [%]	Peak total volume $[m^3]$	Hourly reduction in flood peak [%]
A	09/11/19	0.66	23.6	5.4 (5.4-5.4)	13.9 (13.9-14.0)	2193 (2193-2200)	80.7 (80.6-80.7)	12680 (12673-12682)	16.6 (16.0-17.1)	2602 (2519-2693)	14.2 (13.4-14.4)
B	14/11/19	0.87	29.4	7.4 (7.4-7.5)	25.4 (25.3-25.4)	3992 (3976-3992)	67.2 (67.2-67.2)	10559 (10555-10564)	26.3 (25.1-27.8)	4132 (3943-4363)	26.8 (26.3-27.0)
C	15/02/20	0.63	14.8	8.3 (8.2-8.3)	20.3 (20.2-20.3)	3190 (3174-3190)	79.7 (79.7-79.8)	12530 (12527-12538)	17.5 (17.0-18.0)	2746 (2674-2829)	18.5 (17.8-19.2)
D	04/10/20	0.62	27.8	9.7 (9.6-9.8)	16.7 (16.6-16.9)	2625 (2609-2656)	73.6 (75.6-75.6)	11564 (11564-11566)	18.0 (17.7-18.4)	2828 (2778-2885)	14.6 (14.0-14.8)
E	23/12/20 (Peak 1)	0.83	35.8	10.7 (10.7-10.7)	49.1 (49.0-49.4)	7717 (7701-7764)	40.2 (40.0-40.3)	6326 (6284-6339)	35.5 (29.8-40.6)	5577 (4690- 6382)	55.2 (49.9-57.6)
F	23/12/20 (Peak 2)	0.84							53.6 (52.4-55.3)	8430 (8230-8697)	19.1 (14.5-23.8)
G	27/12/20	0.66	19.2	13.0 (13.0-13.0)	19.8 (19.7-19.9)	3112 (3096-3128)	67.2 (67.3-67.2)	10563 (10557-10581)	26.5 (25.3-27.9)	4170 (3970-4392)	32.4 (32.1-33.1)
H	31/10/21	0.72	25.4	2.8 (2.8-2.8)	17.0 (17.0-17.0)	2672 (2672-2672)	80.2 (80.1-80.2)	12598 (12596-12606)	18.0 (17.0-18.9)	2830 (2679-2964)	31.8 (28.6-33.2)
I	15/02/22	0.20	10.2	4.6 (4.6-4.7)	4.1 (4.0-4.1)	644 (629-644)	91.3 (91.3-91.3)	14354 (14353-14355)	6.3 (6.1-6.7)	994 (952-1047)	20.1 (15.9-18.4)
J	16/03/22	0.21	17.6	3.4 (3.4-3.4)	5.3 (5.3-5.3)	833 (833-833)	91.3 (91.3-9.13)	14352 (14351-14353)	5.9 (5.6-6.3)	931 (887-987)	21.7 (19.8-23.6)

The maximum flood reduction of 55.2 % was observed during the 23 December 2020 event which had two peaks, the longest duration and most intense rainfall. The daily rainfall and API were both highest for this event (Section A.5.1), showing that the FSA interventions were effective during notably wet preceding conditions. As significant storage capacity remained after the first peak the FSAs were also able to attenuate a second larger peak by 19.1 %, during which the FSAs held over 8,000 m^3 of water. While both peaks were successfully attenuated, drainage of the FSAs after the first peak and resultant increase in discharge at the sub-catchment outlet will have reduced the effect for the second peak. This event also showed the largest sensitivity to travel time, with an estimated 9.3 % increase in flood peak reduction between +25 % travel time and -25 % travel time. Suggesting that slower travel times and drainage would attenuate the peak further. This relationship was observed across all events.

At least 40 % of total storage capacity remained available throughout all events, suggesting that larger events than those analysed here could be successfully attenuated. Though this will be dependent on how much flood storage capacity is lost to antecedent conditions. The maximum antecedent storage observed was 13 % during notably wet 2020 winter event on 27 December 2020 which closely succeeded the longest and most intense event observed, yet over 67 % of storage capacity remained throughout the event. There is a wide range of responses to storm events dependent on antecedent conditions and rate and duration of rainfall. Further work is being carried out to investigate these relationships through a PhD studentship.

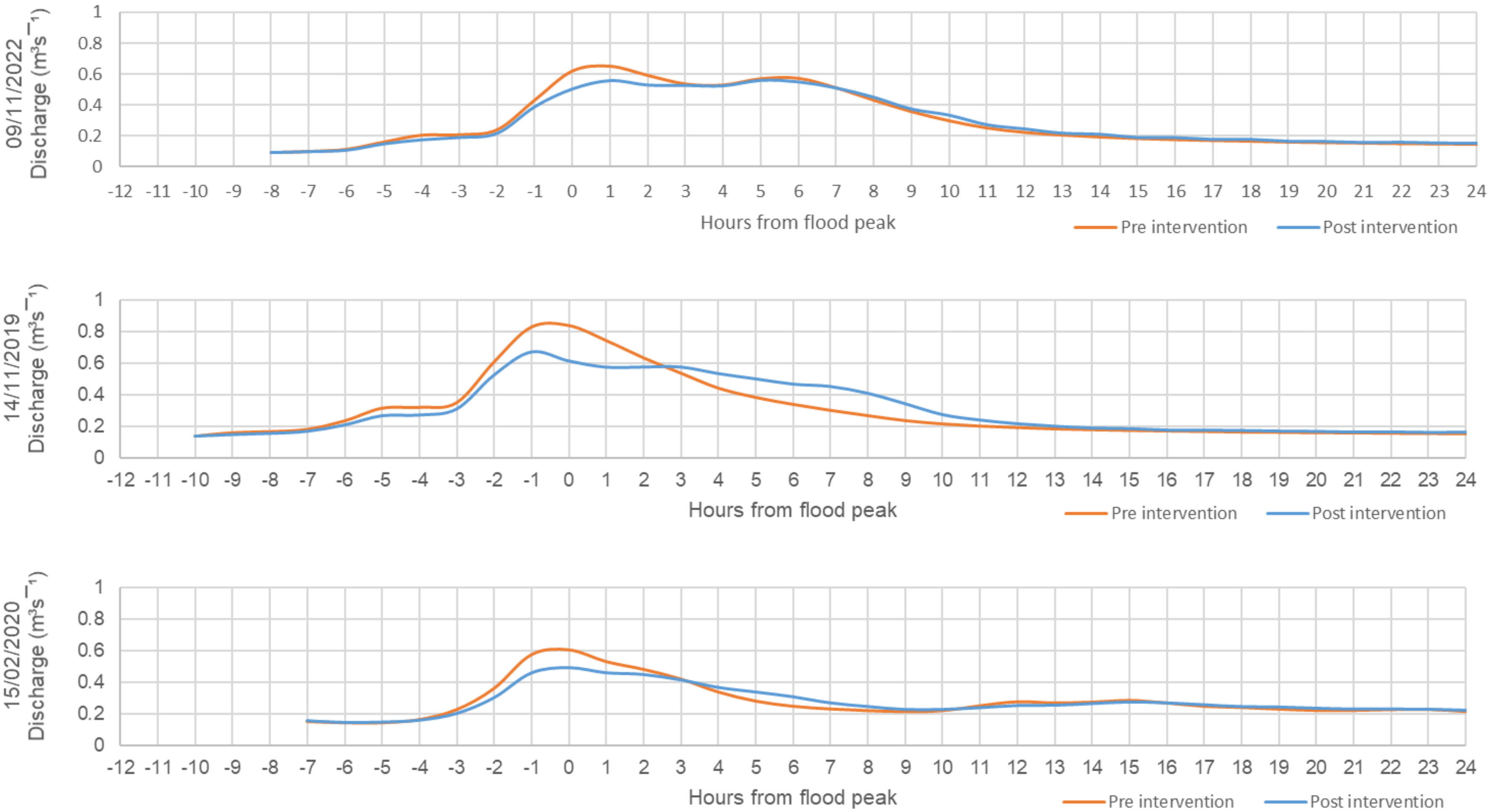


Figure 6.23: WY 2019/2020 event hydrographs for discharge pre- (orange) and post-FSA interventions (blue). The post intervention discharge is the hourly averaged value estimated at the sub-catchment outlet using the stage-discharge rating curve and repeat observations of water level. The pre intervention discharge is estimated from the post intervention discharge by subtracting the post intervention discharge multiplied by the stored FSA volume as a percentage of the combined stored and flood volume at the sub-catchment outlet.

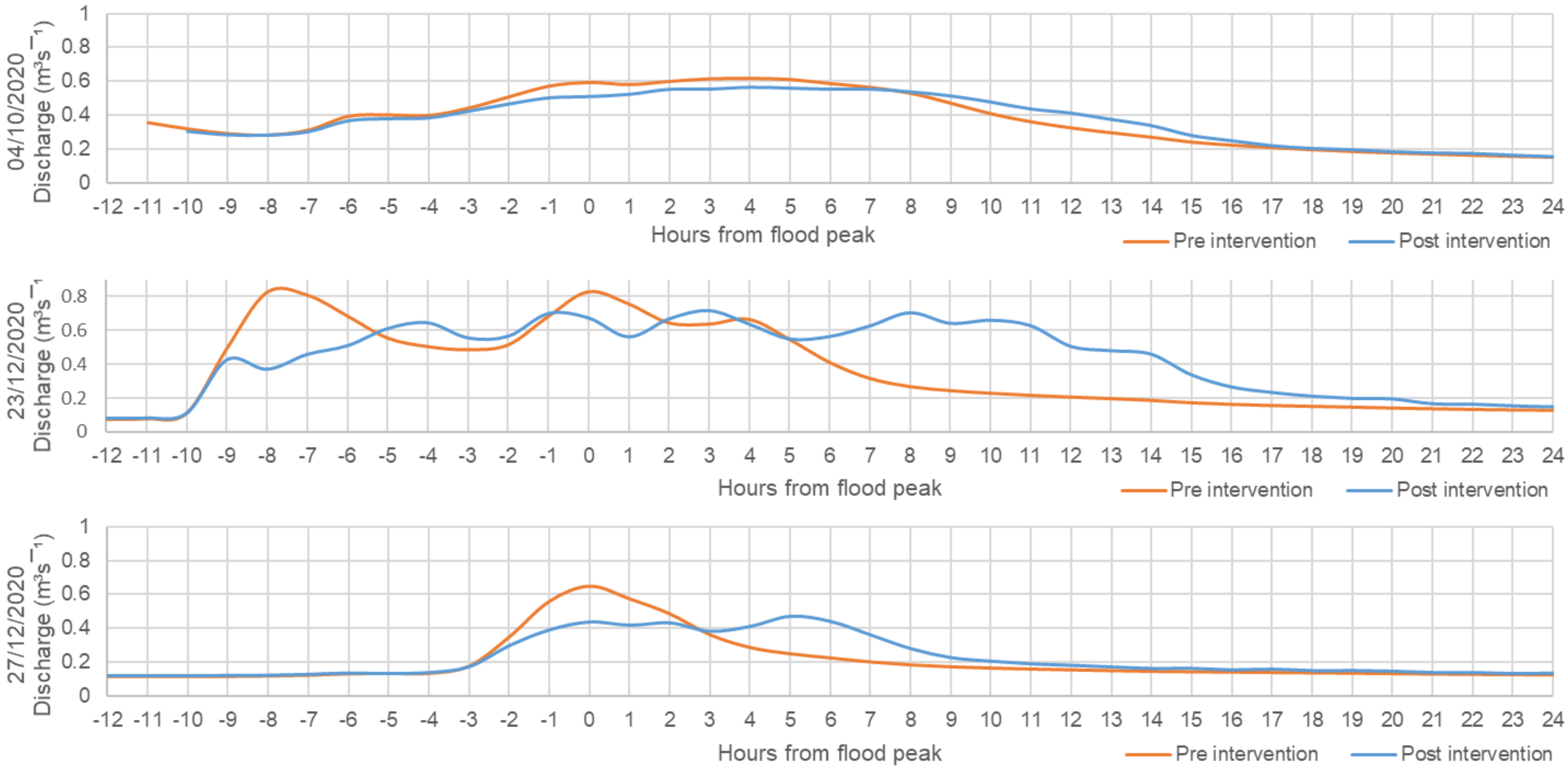


Figure 6.24: WY 2020/2021 event hydrographs for discharge pre- (orange) and post-FSA interventions (blue). The post intervention discharge is the hourly averaged value estimated at the sub-catchment outlet using the stage-discharge rating curve and repeat observations of water level. The pre intervention discharge is estimated from the post intervention discharge by subtracting the post intervention discharge multiplied by the stored FSA volume as a percentage of the combined stored and flood volume at the sub-catchment outlet.

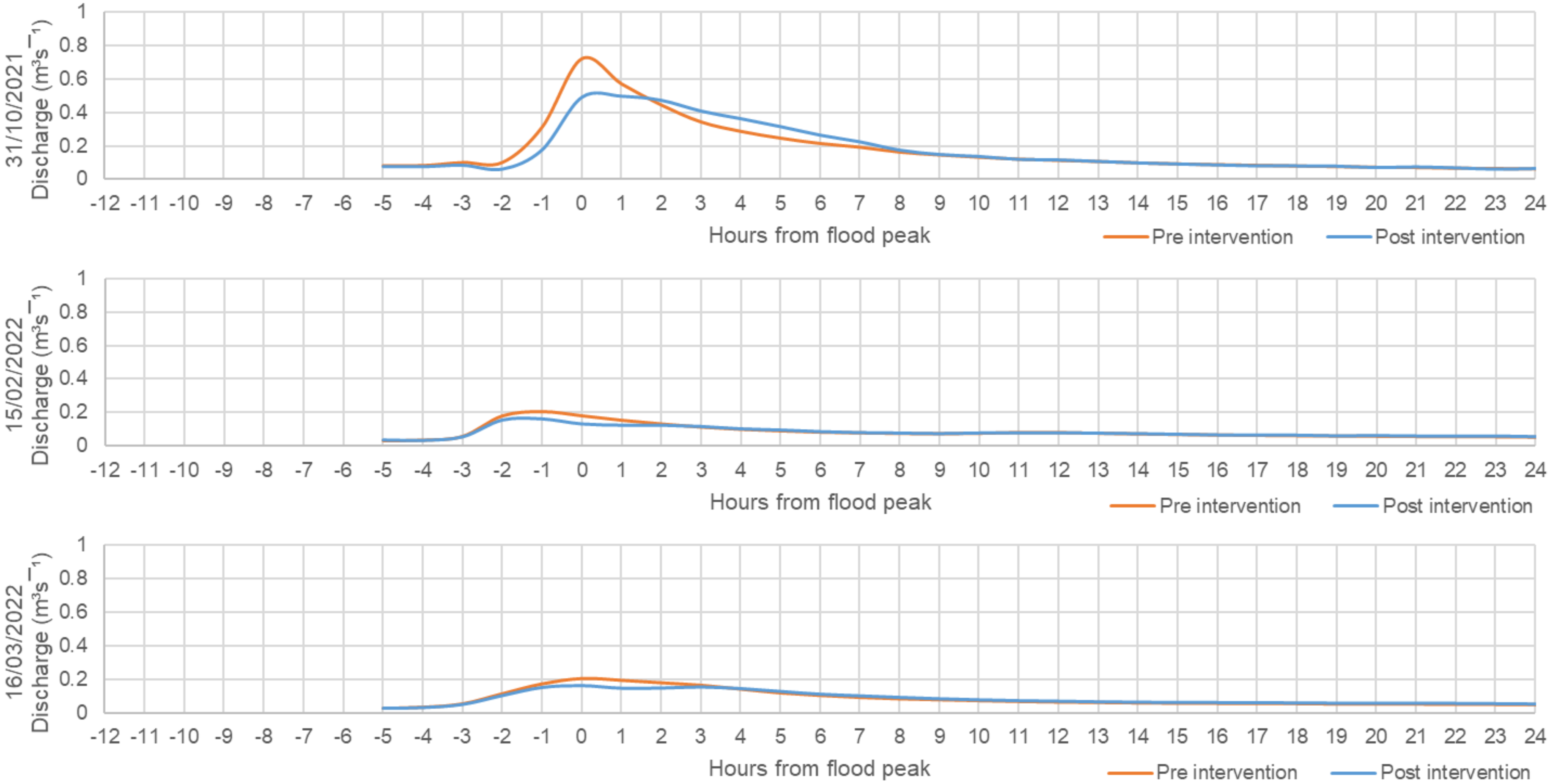


Figure 6.25: WY 2021/2022 event hydrographs for discharge pre- (orange) and post-FSA interventions (blue). The post intervention discharge is the hourly averaged value estimated at the sub-catchment outlet using the stage-discharge rating curve and repeat observations of water level. The pre intervention discharge is estimated from the post intervention discharge by subtracting the post intervention discharge multiplied by the stored FSA volume as a percentage of the combined stored and flood volume at the sub-catchment outlet.

A.6 Monitoring Evaluation

Data were evaluated using a number of approaches at multiple spatial scales in order to determine the effect of the NFM interventions.

Isolating the effect of NFM interventions from natural variability was challenging using an experimental 'Before-After Control-Impact' (BACI) approach, particularly as the catchment interventions were incrementally added throughout the monitored period (Robotham, 2022). This approach requires robust pre-intervention data to eliminate the noise of environmental and climatic change within catchments, relative to the effects of NFM interventions. This trial was characterised by a relatively short and dry pre-intervention period, with few high magnitude storm events. This was followed by a wet post-intervention period, making it difficult to compare pre- and post-NFM data to detect the effects. For a better assessment of NFM effectiveness we recommend a prolonged period of multi-scale baseline monitoring that captures a range of environmental conditions pre-intervention installation, to allow for more robust before-after evaluation.

Combining in-stream and individual intervention-scale monitoring provided us with evidence of the FSA NFM intervention effectiveness during storm events. Continuous water level monitoring in FSAs allowed us to calculate continuous flood storage volume across the whole south sub-catchment, enabling us to estimate the reduction in flood peaks at the downstream flood receptor rather than qualify. Further work is required to evaluate the FSA effectiveness and responses during storm events, with diverse antecedent conditions and rates and durations of rainfall. Analyses will also be carried out in the north sub-catchment when topographic data are available for all FSAs.

A.7 Concluding Remarks

This was a successful monitoring program with over 90 % data coverage of rainfall, stream flow, FSA level, suspended sediment and associated nutrients across 5 years and two sub-catchments of the Littlestock Brook NFM scheme. These data enabled the calibration and validation of hydrodynamic modelling of the NFM measures, as well as detailed analysis of the catchment area hydrological processes and water quality. This enabled assessment of the effectiveness of the NFM interventions, within the limits of the range of conditions observed during the monitoring period.

The analysis of the intervention-scale monitoring data showed successful attenuation of all storm event discharge peaks (14.2-55.2 % reductions) and that over 40 % of the total storage volume remained available throughout all events. The greatest peak reductions were observed in the larger and more intense rainfall events, where higher water levels lead to greater overbank flow to the FSAs. This was the case for the largest reduction (55.2 %) in flood peak for the intense rainfall 23 December 2020 event. This event was preceded by notably wet conditions and the FSA storage successfully attenuated a second storm peak by 19.1 %. Travel time sensitivity analysis showed that slower travel time from FSAs to the sub-catchment outlet would attenuate flood peaks further across the events analysed. The effects varied greatly due to event variability of antecedent conditions and rainfall intensity and duration.

The FSAs were able to provide multiple benefits through significant sediment trapping, particularly during the larger storm events where features were connected to the stream via spillways. The equivalent of 15 % of sub-catchment sediment yield was trapped by features over the 2-3 years since construction. This stored sediment also accounted for 10 % of the TP and 8 % of the POC yields. The measured sediment accumulation rates varied greatly between features, and they do not appear to compromise the primary water storage function of the FSAs; they are only likely to need maintenance every 10 years. The accumulated sediment is generally fine and enriched in nutrients thereby holding potential value for re-use in agriculture.

Detailed monitoring of the on-line pond features highlighted their benefits for water quality during baseflow conditions, significantly reducing dissolved nutrient (N and P) concentrations by 5 and 29 % respectively. Overall these small features acted as a net sink of sediment, despite some instances of sediment resuspension/flushing during the monitoring period. They developed into vegetated wetland habitat, however they also accumulated sediment rapidly and therefore require maintenance approximately every 2 years.

Streamflow monitoring in each of the sub-catchments highlighted significant differences in hydrological regimes, with the south sub-catchment having higher baseflows and lower peak discharges during storm events. The north sub-catchment had a more hydrologically flashy response with higher peaks, suggesting that the NFM features located within this area will be particularly important for intercepting rapid run-off from this land.

All data collected throughout this monitoring will now be archived and made available on the NERC Environmental Information Data Centre for further research. Further analysis of these data is also being carried out through an ongoing PhD studentship, which will analyse the effectiveness of the scheme in more detail over the range of events and conditions observed.

A.8 Acknowledgements

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A.9 References

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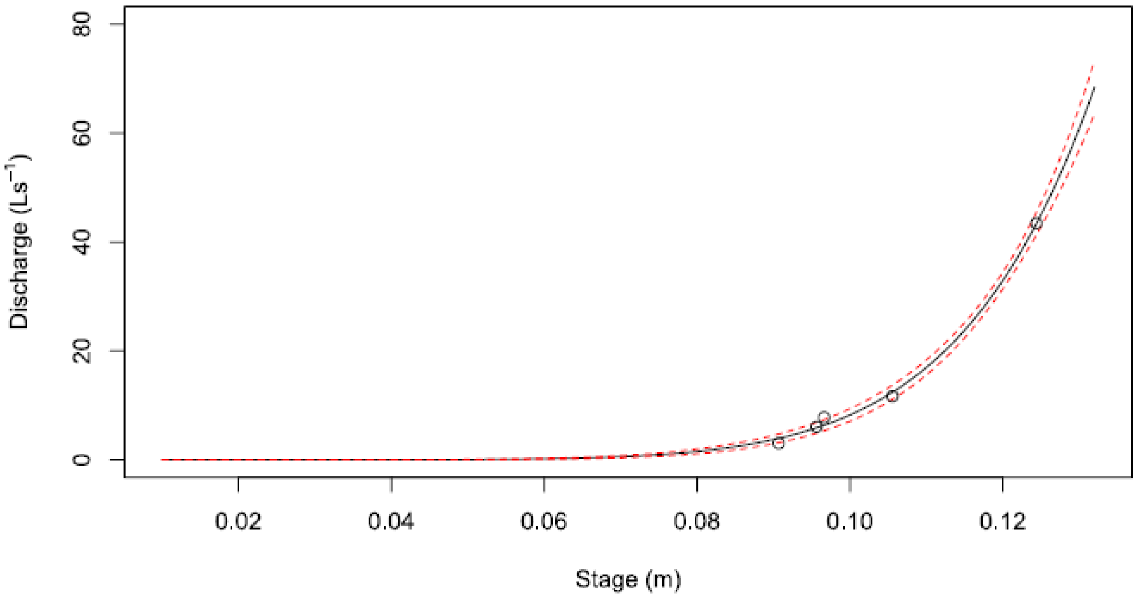
A.10 Appendices

A.10.1 Appendix 1 – Rating curves

Low Flow Rating (used where Stage < 0.132 m):

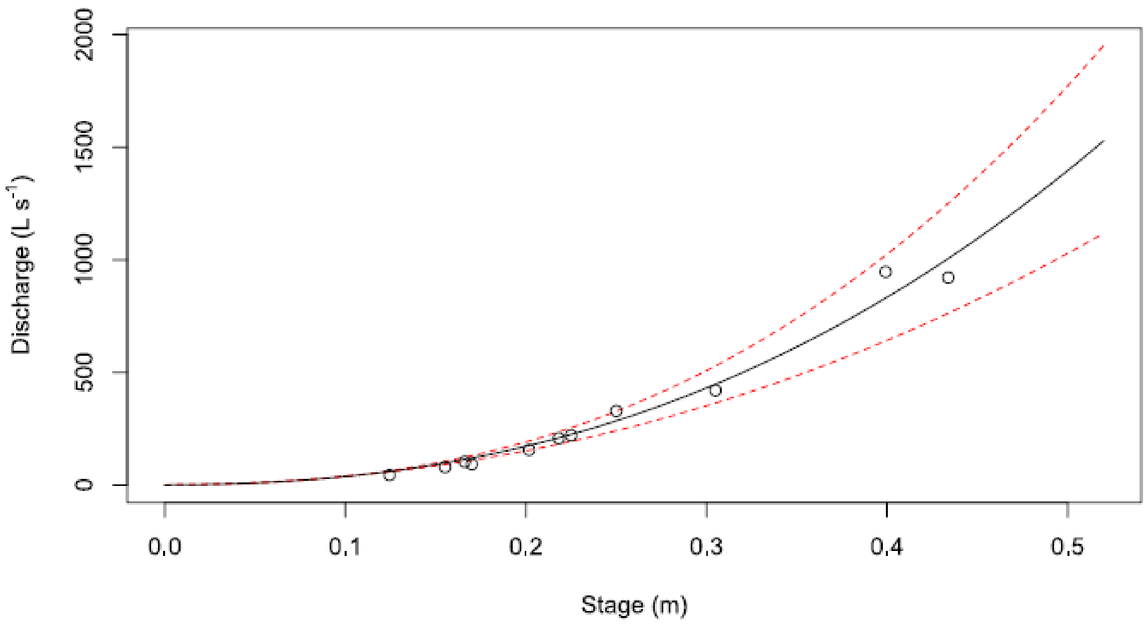
$$Q\ [L\ s^{-1}] = 710377415.9571 \times (Stage\ [m] + 0.01)^{\ 8.277}$$



The Heath:

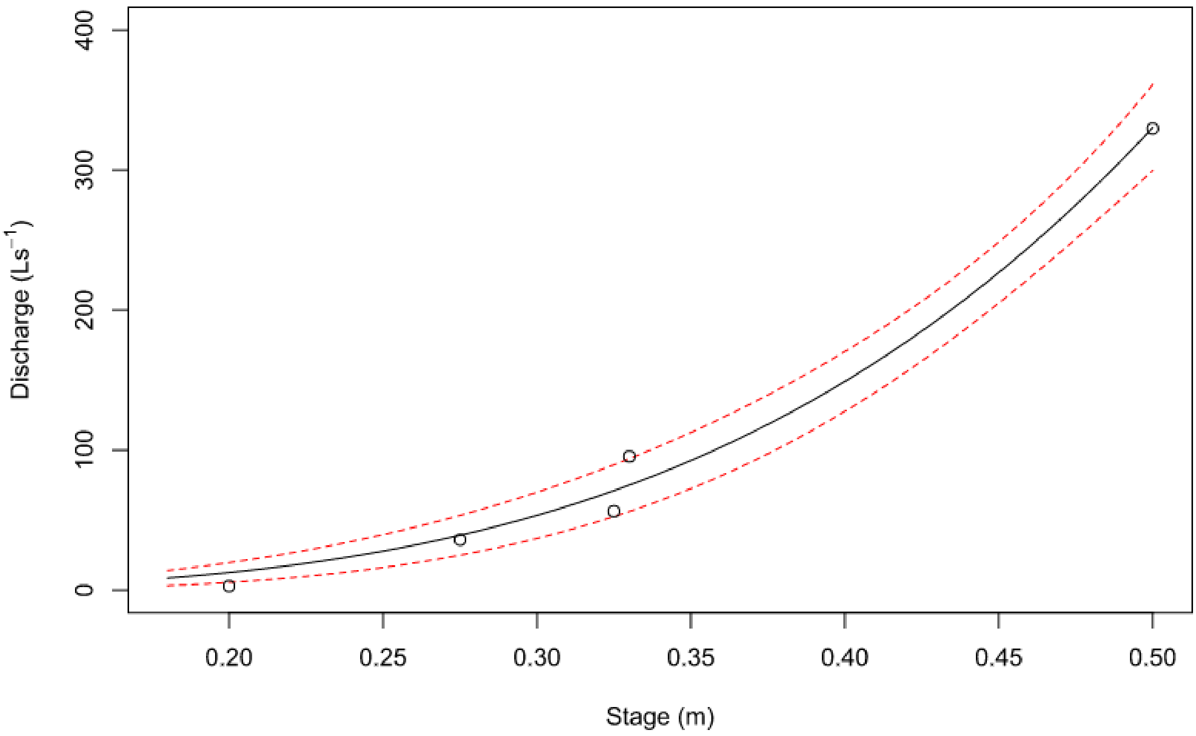
High Flow Rating (used where Stage > 0.132 m):

$$Q\ [L\ s^{-1}] = 6849.014 \times (Stage\ [m] + 0.01)^{\ 2.3622}$$



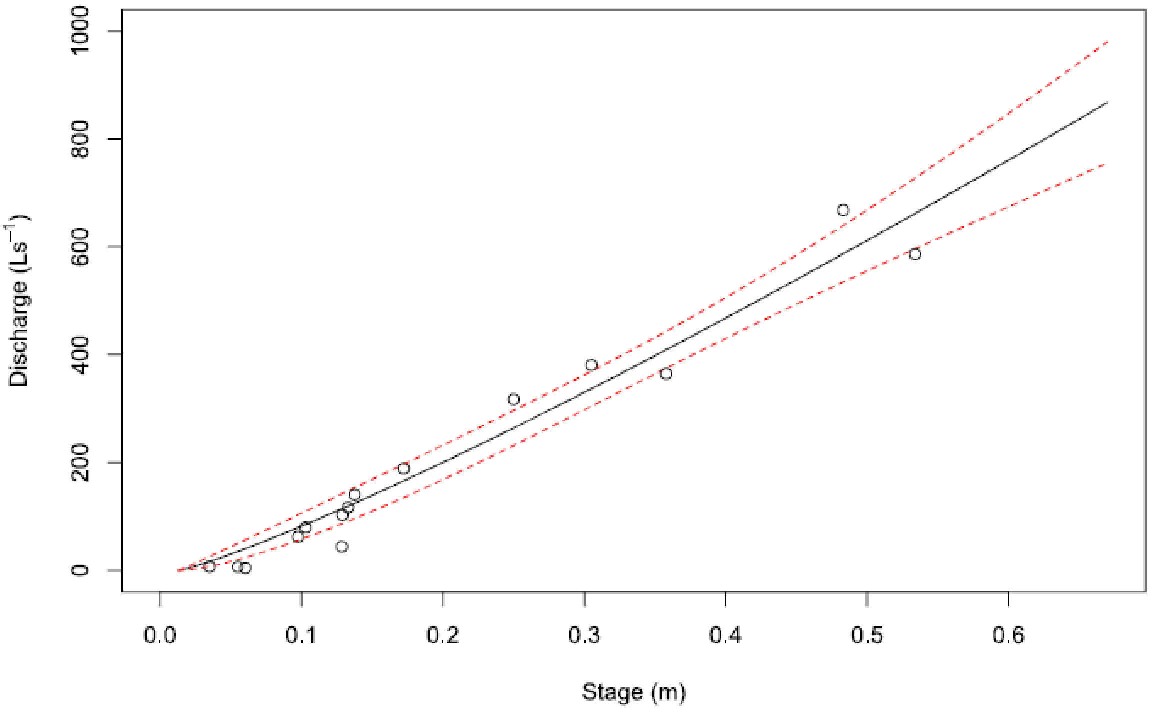
Appendix A

Upstream The Heath:



Church Meadow:

$$Q\ [L\ s^{-1}] = 1417.2706 \times (Stage\ [m] - 0.013)^{1.1669}$$



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Appendix B Sediment and Nutrient Retention in Ponds on an Agricultural Stream: Evaluating Effectiveness for Diffuse Pollution Mitigation

The following peer-reviewed journal article was published as part of this PhD thesis (Chapter 3). The first page is displayed below as an image and is linked to the full version of the published manuscript. The open access article is also available at https://doi.org/10.3390/w13121640.

Appendix B

water

MDPI

Article

Sediment and Nutrient Retention in Ponds on an Agricultural Stream: Evaluating Effectiveness for Diffuse Pollution Mitigation

John Robotham [1,2,*], Gareth Old [1], Ponnambalam Rameshwaran [1], David Sear [2], David Gasca-Tucker [3], James Bishop [1,4], Joanne Old [5] and David McKnight [5]

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Abstract: The creation of ponds and wetlands has the potential to alleviate stream water quality impairment in catchments affected by diffuse agricultural pollution. Understanding the hydrological and biogeochemical functioning of these features is important in determining their effectiveness at mitigating pollution. This study investigated sediment and nutrient retention in three connected (on-line) ponds on a lowland headwater stream by sampling inflowing and outflowing concentrations during base and storm flows. Sediment trapping devices were used to quantify sediment and phosphorus accumulations within ponds over approximately monthly periods. The organic matter content and particle size composition of accumulated sediment were also measured. The ponds retained dissolved nitrate, soluble reactive phosphorus and suspended solids during baseflows. During small to moderate storm events, some ponds were able to reduce peak concentrations and loads of suspended solids and phosphorus; however, during large magnitude events, resuspension of deposited sediment resulted in net loss. Ponds filtered out larger particles most effectively. Between August 2019 and March 2020, the ponds accumulated 0.306 t ha^{-1} sediment from the 30 ha contributing area. During this period, total sediment accumulations in ponds were estimated to equal 7.6% of the suspended flux leaving the 340 ha catchment downstream. This study demonstrates the complexity of pollutant retention dynamics in on-line ponds and highlights how their effectiveness can be influenced by the timing and magnitude of events.

Keywords: water quality; natural flood management; suspended sediment; phosphorus; nitrate; organic matter; on-line ponds; constructed wetlands; particle size

check for updates

Citation: Robotham, J.; Old, G.; Rameshwaran, P.; Sear, D.; Gasca-Tucker, D.; Bishop, J.; Old, J.; McKnight, D. Sediment and Nutrient Retention in Ponds on an Agricultural Stream: Evaluating Effectiveness for Diffuse Pollution Mitigation. *Water* 2021, *13*, 1640. https://doi.org/10.3390/w13121640

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1. Introduction

Intensively farmed landscapes can contribute significantly to the degradation of the water environment globally [1–3]. In many European countries, agricultural intensification has increased the risk of waterbodies failing to meet the EU Water Framework Directive (WFD) objective of 'good ecological status' (2000/60/EC) [4–6]. For streams and rivers in England and Wales, diffuse sources of pollution from agriculture present one of the biggest threats to WFD failure, with key concerns being elevated nutrient concentrations, oxygen depletion and the smothering of instream habitats by fine sediment [7]. One of the main delivery mechanisms for diffuse pollution is soil erosion caused by surface run-off which is then exacerbated when arable fields are left bare or subject to soil compaction [6,8]. Diffuse

Appendix C Nature-based solutions enhance sediment and nutrient storage in an agricultural lowland catchment

The following peer-reviewed journal article was published as part of this PhD thesis (Chapter 4). The first page is displayed below as an image and is linked to the full version of the published manuscript. The open access article is also available at: https://doi.org/10.1002/esp.5483.

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RESEARCH ARTICLE

ESPL WILEY

Nature-based solutions enhance sediment and nutrient storage in an agricultural lowland catchment

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Abstract

In this paper, nature-based solutions (NBS) include: (1) natural flood management (NFM) interventions with a primary function of flood risk reduction but with additional multiple benefits for water quality improvements through the mitigation of diffuse pollution; and (2) ponds with a primary function of water quality improvement. This study assesses the ability of these NBS to trap pollutants in run-off within two small (3.4 km^2) agricultural catchments (Upper Thames, UK). The masses of sediment, phosphorus, and organic carbon trapped by 14 features (since construction 2–3 years previously) were quantified through sediment surveying and sampling. Streamflow and suspended sediment monitoring downstream of features enabled catchment yields to be calculated. The features trapped a total of 83 t sediment, 122 kg phosphorus, and 4.3 t organic carbon. Although the footprint of the features was <1% of the catchment area, they drained 44% of the total land area and captured the equivalent of 15% of the total suspended sediment yield, 10% of the total phosphorus yield, and 8% of the particulate organic carbon yield as monitored at the catchment outlet over the monitoring period. Results reveal that accumulation rates were influenced by hydrological connectivity, with greater accumulation in features constructed directly on streams (online ponds), and those offline features that filled from overbank flows. The low to moderate accumulation rates observed in offline features suggests that their floodwater storage potential is only likely to significantly reduce in the medium term, necessitating maintenance after ~10 years. Compared with topsoil in each contributing area, trapped sediment was enriched in phosphorus and carbon in the majority of features, having on average 50% higher phosphorus and 17% higher organic carbon concentrations than surrounding arable soils, highlighting its potential value for redistribution on farmland. Monitoring results demonstrate the potential of NBS, including NFM, to mitigate diffuse pollution in lowland catchments.

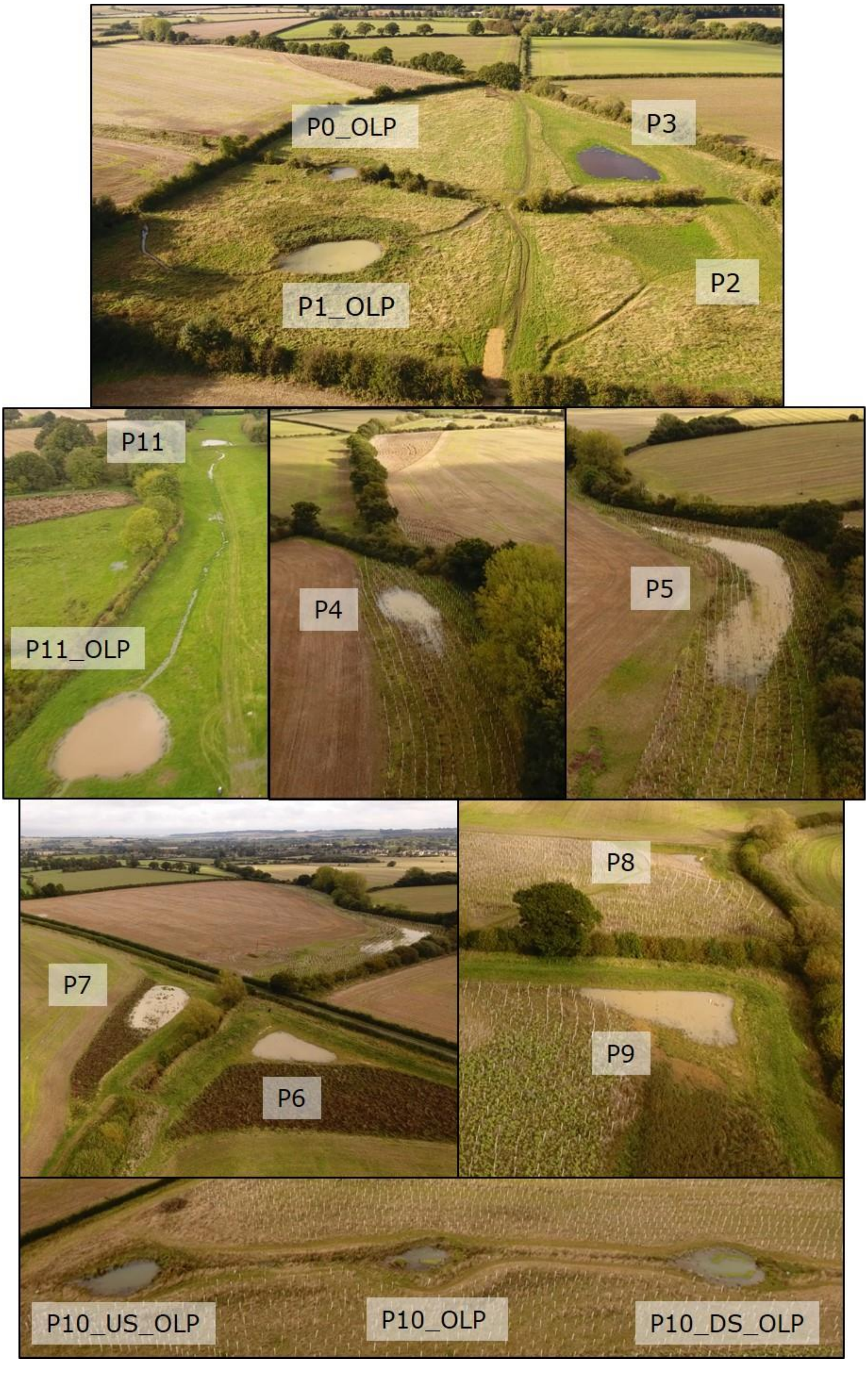
KEYWORDS

catchment management, diffuse agricultural pollution, fine sediment, multiple benefits, natural flood management, organic carbon, phosphorus, soil erosion, water quality, working with natural processes

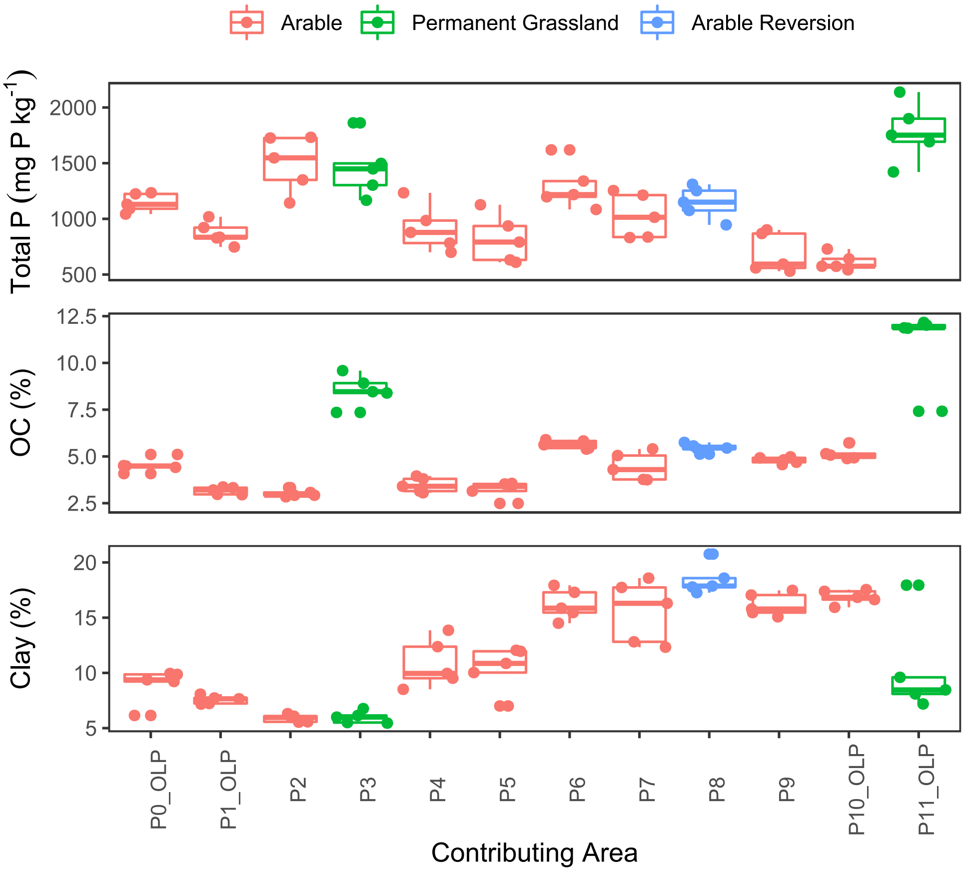
Appendix D Supporting Information for Chapter 4

This appendix contains the supporting information for Chapter 4 of this PhD thesis and the published journal article (see Appendix C).

SUPPORTING ONLINE ONLY INFORMATION



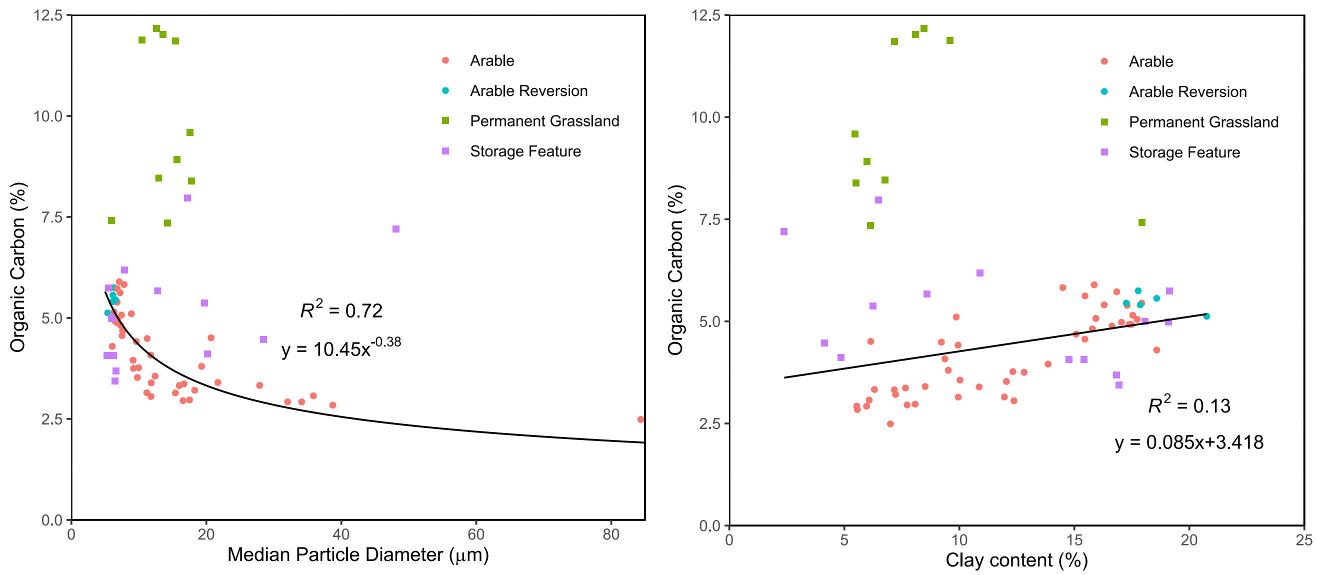
S1. Aerial photographs of the NBS features after a storm event on 14th October 2019. Images courtesy of the Environment Agency.



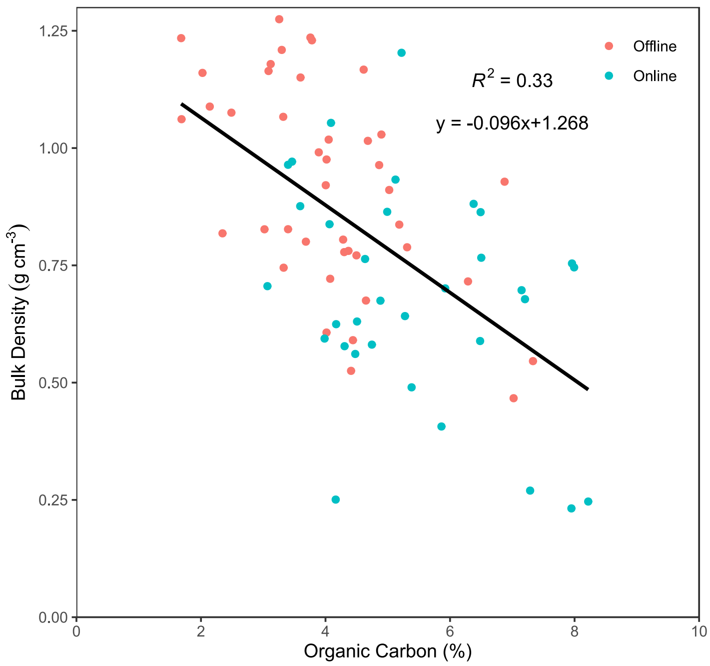
S2. Boxplots with data points showing sample variability for key soil properties: Total P content (mg P kg^{-1}), organic carbon content (%), and clay content (%) in each features contributing area. Colours indicate the dominant land-use in each contributing area that soil was sampled from.

S3. Enrichment Ratios (ER) and 95% confidence intervals for clay, silt, sand, OC, and TP content derived from samples of sediment in NBS features and soil in respective contributing areas.

NBS Feature	Enrichment Ratio (95% CI)				
	Clay	Silt	Sand	OC	TP
P0_OLP	0.70 (0.59-0.87)	1.03 (0.92-1.16)	1.04 (0.77-1.58)	1.19 (0.79-1.60)	1.19 (1.08-1.33)
P1_OLP	1.44 (1.36-1.52)	1.52 (1.49-1.55)	0.11 (0.10-0.11)	1.95 (1.54-2.37)	2.66 (1.60-3.81)
P2	1.10 (0.92-1.29)	1.34 (1.30-1.38)	0.59 (0.54-0.63)	2.64 (2.40-2.91)	0.91 (0.76-1.12)
P3	1.44 (1.31-1.60)	0.98 (0.94-1.03)	0.94 (0.79-1.16)	0.66 (0.45-0.89)	0.98 (0.74-1.31)
P4	1.76 (1.44-2.27)	1.22 (1.11-1.35)	0.29 (0.21-0.42)	1.44 (1.28-1.64)	1.58 (1.26-2.12)
P5	1.62 (1.33-2.07)	1.20 (0.98-1.54)	0.29 (0.18-0.71)	1.14 (0.99-1.35)	1.92 (1.48-2.70)
P6	0.91 (0.83-1.01)	1.15 (1.13-1.18)	0.05 (0.05-0.06)	0.72 (0.62-0.82)	1.26 (0.99-1.60)
P7	0.99 (0.83-1.24)	1.11 (1.09-1.13)	0.30 (0.23-0.43)	0.91 (0.76-1.13)	1.53 (1.24-1.97)
P8	1.04 (0.90-1.19)	1.08 (1.05-1.10)	0.19 (0.13-0.26)	1.05 (0.96-1.15)	1.07 (0.95-1.24)
P9	1.05 (0.96-1.14)	0.98 (0.96-1.00)	1.06 (0.92-1.20)	0.72 (0.52-0.92)	1.41 (1.10-1.92)
P10_OLP	0.25 (0.15-0.34)	0.93 (0.87-1.00)	3.15 (2.30-4.24)	0.87 (0.76-0.99)	1.93 (1.70-2.23)
P10_US_OLP	0.14 (0.12-0.16)	0.77 (0.66-0.88)	4.84 (3.63-6.43)	1.40 (1.25-1.56)	2.01 (1.61-2.47)
P10_DS_OLP	0.29 (0.23-0.35)	1.06 (1.00-1.11)	1.94 (1.33-2.72)	0.80 (0.69-0.92)	1.77 (1.56-2.05)
P11_OLP	1.76 (1.20-3.31)	1.05 (1.03-1.07)	0.21 (0.15-0.33)	0.45 (0.37-0.57)	0.87 (0.74-1.05)



S4. (Left) Non-linear relationship between median particle diameter (μm) and organic carbon (%) in soil and sediment; the plotted function includes arable and arable reversion datapoints only. (Right) Linear regression between clay content (%) and organic carbon (%) in soil and sediment; the plotted function excludes permanent grassland datapoints.



S5. Linear regression between organic carbon (%) and bulk density (g cm^{-3}) of sediment in offline and online NBS features.

S6. Percentage reductions in the maximum storage capacity and the storage capacity up to drain heights of NBS features since their construction. 10-year reductions in storage capacity are estimates based on the measured rates of accumulation during the monitoring period. N.B. It was not possible to calculate storage capacities up to drain heights for all features.

NBS Feature	Storage capacity to drain (m^3)	Reduction in max. storage capacity (%)	Reduction in storage capacity to drain (%)	Annual reduction in max. storage capacity (%)	10-year reduction in max. storage capacity (%)	10-year reduction in storage capacity to drain (%)
P0_OLP	-	27.34	-	12.9	100	-
P1_OLP	144.01	2.95	9.00	1.39	11.4	42.4
P2	-	0.57	-	0.27	2.7	-
P3	-	4.23	-	2.00	20.0	-
P4	113.74	0.03	0.26	0.01	0.1	0.8
P5	121.93	0.19	5.40	0.06	0.6	17.3
P6	130.46	0.38	7.69	0.12	1.2	24.7
P7	310.91	0.49	4.30	0.16	1.6	13.8
P8	17.37	0.04	1.30	0.01	0.1	4.2
P9	89.55	0.07	0.72	0.02	0.2	2.3
P10_OLP	20.24	8.18	36.36	4.22	42.2	>100
P10_US_OLP	25.52	19.84	54.43	10.23	>100	>100
P10_DS_OLP	48.46	10.41	20.41	4.37	43.7	85.8
P11	-	0.00	-	0.00	-	-
P11_OLP	-	4.75	-	2.24	22.4	-

S7. Additional comments and discussion relating to the factors influencing sediment and nutrient accumulations within NBS features.

The accumulations measured within this study reflect the short-term impact of the features within a period in which the landscape was undergoing frequent change as a result of the NFM scheme. During this time, accumulations were likely influenced by the initial soil disturbance from the construction of the earth bunds prior to their stabilisation from the establishment of vegetation. The earth bunds were seeded with a native wildflower mix following construction in February, allowing substantial cover and stabilisation to take place before the storms of the next winter. It is also important to consider the effects of other NFM *interventions implemented in parallel with the storage features (e.g. leaky barriers and riparian tree planting) which will have modified sediment and nutrient delivery in the catchment. Whilst not quantified in this study, a build-up of channel debris and sediment was observed immediately upstream of several of the leaky barriers (S8). This type of storage is a commonly observed effect of instream wood structures, enhancing both deposition and nutrient processing (Lo et al., 2021). It is possible that during high flows some of this stored channel sediment will have been resuspended and transported into offline features via spillways. The NFM scheme included over 0.1 km^2 of riparian woodland creation on arable land, helping to reduce soil erosion and buffer streams from run-off (Hickey and Doran, 2004; Stutter et al., 2019), potentially trapping sediment before it even reached the features.*

S8. Animation of overbank flow induced by the leaky barrier located upstream of P6 and P7 during a storm event on 23rd December 2020). [NB. Animation only works in .docx file available in supporting information at: https://doi.org/10.1002/esp.5483]

Appendix E Supporting Documentation for Published EIDC Dataset: "High-resolution timeseries of turbidity, suspended sediment concentration, total phosphorus concentration, and discharge in the Littlestock Brook, England (2017-2021)"

This appendix contains the supporting documentation for a dataset collected as part of this PhD thesis and was subsequently published in the Environmental Information Data Centre (EIDC): https://doi.org/10.5285/9f80e349-0594-4ae1-bff3-b055638569f8.

High-resolution timeseries of turbidity, suspended sediment concentration, total phosphorus concentration, and discharge in the Littlestock Brook, England (2017-2021)

Overview:

This dataset contains measurements of turbidity, suspended sediment concentration, total phosphorus concentration, and discharge at a 5-minute resolution over a period of several years (2017-2021) for three stream sites in the Littlestock Brook catchment (southern England). The turbidity data were collected using instream sensors. Samples of suspended sediment and total phosphorus concentrations taken under a range of flow conditions were used in a regression to predict concentrations from turbidity measurements for the entire timeseries. The discharge timeseries was derived from a stage-discharge rating curve, with flow being measured using the velocity-area method (and salt dilution method), and stage being measured by an instream water level sensor. The dataset was collected as part of a monitoring programme for the Littlestock Brook Natural Flood Management (NFM) scheme, a 5-year project funded by the Environment Agency using low-cost nature-based solutions to mitigate flood risk and diffuse pollution. UKCEH oversaw the collection of these data which were developed and interpreted as part of two NERC-funded PhD projects (see Robotham *et al.* (2021) for how these data have been used). The dataset contains some periods of missing data as a result of sensor failure etc., however these periods vary between the different stream sites. Data were quality controlled to exclude erroneous data and short periods of missing data were linearly interpolated where appropriate.

Data collection methods:

Each file within the dataset contains data collected from a stream monitoring site on the Littlestock Brook, a headwater tributary of the River Evenlode located within the upper reaches of the Thames basin. Locations of each site are given in Table 1.

Table 1: Latitude and longitude of Littlestock Brook stream monitoring sites.

Site Name	Latitude	Longitude
LSB_The_Heath	51.865283	-1.6180989
LSB_Upstream_The_Heath	51.868396	-1.6316682
LSB_Church_Meadow	51.864193	-1.6187105

Turbidity data were measured at 5-minute intervals using a DTS-12 (digital turbidity sensor) located instream and logged to a CR1000 datalogger at each monitoring sites. An EXO2 optical turbidity sensor (installed 6/12/2017) at LSB_The_Heath was also used to develop a relationship between turbidity and sampled SSC. The EXO sonde was set up with a pumped system, taking hourly water samples from the stream. The pumped system allowed measurements to be taken during low flows where the DTS-12 sensor remained above the water level at this site. The pumped sample is taken into the system through a strainer in order to prevent large particles from the streambed or suspended organic debris (e.g. leaf litter) being sampled. This improved the reliability of the sensor, particularly during high flow events where the DTS-12 sensor sometimes become obscured. Water depth was measured and logged at each site using a Level TROLL 500 Data Logger (pressure sensor) submerged into a plastic stilling well to minimise noisy data from water turbulence. Stream sites also had stage boards mounted on wooden posts which served as fixed points throughout the monitoring period and readings were taken during regular site visits and flow gaugings.

Flow gaugings were carried out across a range of flows, primarily using the velocity-area method. For this, cross-sectional area was calculated by measuring water depths across the channel at regular intervals using a metre rule. At each point, flow velocity was then measured with an Electromagnetic Current Meter (ECM), enabling the instantaneous discharge to be computed. During low flow conditions where the ECM was less suitable, the salt dilution gauging method was employed instead. This followed the method detailed in Hongve (1987) and used an EXO1 sonde to measure instream specific conductivity at a 1-second resolution. Small quantities (<50g) of table salt (NaCl) were used for these gaugings. Rating curves were constructed for each stream monitoring site, enabling discharge estimates to be computed from the power law relationship between observed stage and discharge. Rating curves were computed using the '*nls*' package in R along with lower and upper 95% confidence intervals calculated following the method used by Dalgaard (2004). Table 2 provides detail on the various rating curves used to derive the discharge timeseries in this dataset (see appendix for plots). Due to limited gauging measurements for LSB_Upstream_The_Heath, the rating curve is only suitable for estimating discharge up to ~330 L s^{-1} and should not be used beyond this threshold. The LSB_Upstream_The_Heath data therefore only includes discharges up to this threshold; values above this range are blank.

Table 2: Rating curve equations and confidence intervals used for discharge estimation at each monitoring site under different flow conditions. Flow units are L s^{-1}, and stage units are m.

Site Name	Flow Condition	Rating	Rating Curve Equation	n	Observed flow range
LSB_The_Heath	Low	Estimate	$710377415.9571\times(Stage+0.01)^{8.277}$	5	3.08 – 43.44
		Lower 95% CI	$796296879.4624\times(Stage+0.01)^{8.3497}$		
		Upper 95% CI	$636718094.63113\times(Stage+0.01)^{8.20714}$		
LSB_The_Heath	High	Estimate	$6849.014\times(Stage+0.01)^{2.3622}$	11	78.99 – 946.23
		Lower 95% CI	$4397.42522\times(Stage+0.01)^{2.15786}$		
		Upper 95% CI	$9599.22968\times(Stage+0.01)^{2.50921}$		
LSB_Upstream _The_Heath	Low & high (<330 L s^{-1})	Estimate	$3914.8733\times(Stage)^{3.567}$	5	2.87 – 329.59
		Lower 95% CI	$4415.422324\times(Stage)^{3.789925}$		
		Upper 95% CI	$3539.962256\times(Stage)^{3.373286}$		
LSB_Church _Meadow	Low & high	Estimate	$1417.2706\times(Stage-0.013)^{1.1669}$	15	4.47 – 668.35
		Lower 95% CI	$1341.966446\times(Stage-0.013)^{1.168546}$		
		Upper 95% CI	$1492.50970\times(Stage-0.013)^{1.16536}$		

Nature and units of recorded values:

1. Turbidity values from the DTS-12 sensor were measured in NTU (nephelometric turbidity unit) at a resolution of 0.01 NTU and accuracy of ±2% of reading + 0.2 NTU (0-399 NTU) and ±4% of reading (400-1600 NTU). Every turbidity measurement consists of 100 instantaneous samples from which summary statistics are computed. Our data use the median turbidity value opposed to the sample mean so as to minimise the risk of erroneous extreme samples biasing the value. NB. This is not to be confused with the mean turbidity derived from the sensor calibration described in paragraph 2 of the 'Calibration steps and values' section of this documentation.
2. Turbidity values from the EXO2 sensor were measured in NTU at a resolution of 0.01 NTU and accuracy of ±2%.
3. Suspended Sediment Concentration (SSC) was measured in mg L^{-1} at a resolution of 0.1 mg L^{-1}.
4. Total Phosphorus (TP) concentration was measured in mg L^{-1} at a resolution of 0.001 mg L^{-1}.
5. Discharge was measured in L s^{-1}.

Analytical methods:

1. SSC was measured following the protocol for determination of suspended matter (membrane filtration method) detailed in (Standing Committee of Analysts, 1984). Whatman™ GF/C™ filters with a particle retention of 1.2 µm were used for filtering the samples. A drying oven temperature of 105°C was used for all samples. Filter papers were weighed using a calibrated Sartorius balance that was regularly tested with balance check weights prior to use in SSC analysis.
2. TP concentration was measured following the modified molybdenum blue method of Eisenreich, Bannerman and Armstrong (1975).

Fieldwork and laboratory instrumentation:

1. DTS-12 Digital Turbidity Sensor (Forest Technology Systems Ltd.)
2. CR1000 Datalogger (Campbell Scientific, Inc.)
3. Level TROLL 500 Data Logger (In-Situ Inc.)
4. US DH-48 Depth-integrating suspended-sediment sampler
5. Sigma SD 900 Portable sampler (Hach Company)
6. Varian Cary 50 Scan UV Visible Spectrophotometer
7. Electromagnetic Current Meter (Valeport Ltd.)
8. EXO1 Multiparameter Sonde and Conductivity/Temperature sensor (YSI/Xylem Inc.)
9. EXO2 Multiparameter Sonde and Turbidity sensor (YSI/Xylem Inc.)
10. McVan Analite® probe NEP 390 series (Observator Instruments Pty Ltd.)

Calibration steps and values:

1. TP concentration analysis was carried out alongside calibration standards produced by the Wallingford Nutrient Chemistry Laboratories. Samples were analysed alongside Aquacheck quality control standards (LGC Standards) which acted as "blind" samples to confirm results of the analysis were sufficiently and consistently close to the actual concentration value. Further details are provided in Bowes *et al.* (2018).
2. The calibration of turbidity sensors was carried out approximately twice per year. For calibration, turbidity sensors at the monitoring sites were replaced with new sensors and the old sensors returned to the lab. The raw turbidity measurements were calibrated using linear equations specific to each DTS-12 sensor for that specific deployment period. The equations were determined by calibrating the DTS-12 sensors against standards of known NTU (polymer bead turbidity standards) in the lab, alongside independent measurements with an Analite turbidity probe. Calibrations took place prior to probe deployment and after a deployment period, allowing any sensor drift to be monitored. Where no significant drift was observed, the mean of the pre/post calibration values is used for that deployment period. This calibration process helped to ensure consistency in the turbidity data produced by the different probes at different sites at different times. The pre/post calibration values also determined the minimum and maximum turbidity values which are used as estimated uncertainty bounds to account for error attributed to minor sensor drift within the expected range of the instrument for a given period of deployment.
3. Raw water level depths measured by the sensors were corrected to stream stage using a linear regression between sensor values and observed stage board readings for each monitoring site (Table 3).

Table 3: Regressions and summary statistics for the conversion of sensor readings of water depth into stream stage at each monitoring site for given periods.

Site Name	Period	Regression line	n	R^2
LSB_The_Heath	2017 onwards	$\text{Stage}=992.01\times\text{Sensor}+15.285$	35	0.94
LSB_Upstream_The_Heath	06/12/2016 – 03/11/2017	$\text{Stage}=0.7034\times\text{Sensor}+182.89$	11	0.92
LSB_Upstream_The_Heath	03/11/2017 onwards	$\text{Stage}=1.0965\times\text{Sensor}+102.09$	11	0.97
LSB_Church_Meadow	17/01/2017 – 28/09/2017	$Stage=0.6346\times\text{Sensor}-36.25$	6	0.97
LSB_Church_Meadow	28/09/2017 onwards	$Stage=1.0374\times\text{Sensor}-50.138$	19	0.99

4. For the computation of SSC values in the timeseries, simple linear regressions were run using turbidity to predict concentrations from samples taken at the same time as the turbidity measurement. This was done for each monitoring site for which the equations of the regressions and the upper and lower 95% confidence intervals are

listed in Table 4. For the TP timeseries, SSC was used as the predictor in regressions (Table 5).

Table 4: Regressions, confidence intervals, and summary statistics for the conversion of turbidity to SSC. All regressions were statistically significant at the p<0.001 level.

Site Name	Regression line	Lower 95% CI	Upper 95% CI	n	R^2
LSB_The_Heath	$SSC = 1.5358 \times Turbidity$	$0.9354 \times SSC$	$1.0646 \times SSC$	70	0.93
	EXO $SSC = 2.00248 \times$ EXO $Turbidity$	$0.9482 \times$ EXO SSC	$1.0518 \times SSC$	100	0.94
LSB_Upstream_The_Heath	$SSC = 0.84206 \times Turbidity + 4.03079$	$0.969 \times SSC$	$1.031 \times SSC$	94	0.99
LSB_Church_Meadow	$SSC = 1.00701 \times Turbidity$	$0.9806 \times SSC$	$1.019 \times SSC$	95	0.99

Table 5: Regressions and summary statistics for the estimation of TP from SSC. All regressions were significant at the p<0.001 level.

Site Name	Regression line	n	R^2
LSB_The_Heath	$\mathrm{TP} = 0.0019 \times \mathrm{SSC} + 0.14$	111	0.94
LSB_Upstream_The_Heath	$\mathrm{TP} = 0.0018 \times \mathrm{SSC} + 0.15$	47	0.94
LSB_Church_Meadow	$\mathrm{TP} = 0.0019 \times \mathrm{SSC} + 0.035$	359	0.79

5. The Electromagnetic Current Meter was serviced and calibrated by the manufacturer once during the monitoring period.

Quality control:

1. The turbidity data were quality controlled according to a set of simple rules to help remove erroneous measurements caused by things such as sensor errors or stream debris getting caught on the optical face of the sensor. The rules are as follows:
   a. Raw values must be > 0 NTU
      i. Negative and 0 values are identified and removed.
   b. Raw values must be < 1600 NTU.
      i. Values above the detection range of the sensor are identified and removed.
   c. Raw values recorded during prolonged periods of sensor failure must be removed where this has been validated in situ.
   d. Erroneous spikes/drops in the timeseries must be removed
      i. Spikes are identified using a formula stating that the turbidity value at a given timestep should be less than 3 times the mean average of the turbidity values for the timesteps immediately before and after.
      ii. Spikes are flagged with a 1 where the following condition is true:
      $IF\ x_{t2} > 3 \times \bar{x}_{t1,t3}$

e. Drops are identified using a formula stating that the turbidity value at a given timestep should be greater than the mean average of the turbidity values for the timesteps immediately before and after divided by 3.
    i. Drops are flagged with a 1 where the following condition is true:

$$IF\ x_{t2} < \frac{\bar{x}_{t1,t3}}{3}$$

f. Gaps in the timeseries are linearly interpolated where the gap is less than 12 hours using the function '*fillMissing*' from the '*baytrends*' package in R.
g. Interpolated datapoints are marked with the QC code "I".
h. Gaps are left in the timeseries where the gap is greater than 12 hours and are marked as "M" in the 'QC_Code' column.

2. Discharge data were quality controlled to remove erroneous data (periods of rapid fluctuation or extreme values). Short gaps in the data (30 minutes or less) were linearly interpolated, and those longer were left as blank in the data files.

Details of data structure:

This dataset comprises three csv files entitled "LSB_The_Heath", "LSB_Upstream_The_Heath", and "LSB_Church_Meadow". The LSB_Upstream_The_Heath and LSB_Church_Meadow files have 14 columns labelled 'Timestamp'; 'QC_Code'; 'Turbidity_Mean'; 'Turbidity_Min'; 'Turbidity_Max'; 'SSC'; 'SSC_Lower_CI'; 'SSC_Upper_CI'; 'Q'; 'Q_Lower_CI'; 'Q_Upper_CI'; 'TP'; 'TP_Lower_CI'; 'TP_Upper_CI'. The LSB_The_Heath file contains the following additional columns: 'EXO_SSC'; 'EXO_SSC_Lower_CI'; 'EXO_SSC_Upper_CI'; and 'SSC_Combined'. Table 6 contains metadata on these variables.

Table 6: Dataset variables and their units or format where applicable.

Column name	Description	Units/format
Timestamp	Date and time of observation	UTC time zone (YYYY-MM-DD hh:mm:ss)
QC_Code	Letter indicating the QC status of the turbidity observation	'G' = Good; 'I' = Interpolated; 'M' = Missing; 'E' = EXO data
Turbidity_Mean	The mean turbidity value derived from the minimum and maximum (start/end) calibration for the period during which the sensor was deployed	NTU
Turbidity_Min	The minimum turbidity value determined by either the start or end calibration for the period during which the sensor was deployed	NTU
Turbidity_Max	The maximum turbidity value determined by either the start or end calibration for the period during which the sensor was deployed	NTU

SSC	Suspended sediment concentration as determined by the regression between turbidity readings and SSC samples	mg L^{-1}
SSC_Lower_CI	Lower 95% confidence interval for suspended sediment concentration as determined by the regression between turbidity readings and SSC samples	mg L^{-1}
SSC_Upper_CI	Upper 95% confidence interval for suspended sediment concentration as determined by the regression between turbidity readings and SSC samples	mg L^{-1}
Q	Discharge as determined by the stage-discharge rating curve	L s^{-1}
Q_Lower_CI	Lower 95% confidence interval for discharge as determined by the stage-discharge rating curve	L s^{-1}
Q_Upper_CI	Upper 95% confidence interval for discharge as determined by the stage-discharge rating curve	L s^{-1}
TP	Total phosphorus concentration as determined by the regression between SSC and TP samples	mg L^{-1}
TP_Lower_CI	Lower 95% confidence interval for total phosphorus concentration as determined by the regression between SSC and TP samples	mg L^{-1}
TP_Upper_CI	Upper 95% confidence interval for total phosphorus concentration as determined by the regression between SSC and TP samples	mg L^{-1}
EXO_SSC	Suspended sediment concentration as determined by the regression between EXO sensor turbidity readings and SSC samples. These data are linearly interpolated between hourly readings to get the 5-minute resolution timeseries	mg L^{-1}
EXO_SSC_Lower_CI	Lower 95% confidence interval for suspended sediment concentration as determined by the regression between EXO sensor turbidity readings and SSC samples	mg L^{-1}
EXO_SSC_Upper_CI	Upper 95% confidence interval for suspended sediment concentration as determined by the regression between	mg L^{-1}

	EXO sensor turbidity readings and SSC samples	
SSC_Combined	Observations of SSC (using the DTS-12 sensor) where data are available, with the gaps filled by the EXO_SSC data	mg L^{-1}

Miscellaneous:

There are some periods where the stream water level was very low and therefore may have exposed the turbidity sensor to the air, thereby giving false readings close to zero. These have been identified and removed from the dataset as best possible, but some may remain, particularly in LSB_The_Heath and LSB_Upstream_The_Heath. Due to this issue there are large gaps in the turbidity data at the LSB_The_Heath site towards the start of the timeseries. The installation of the EXO2 turbidity sensor at this site helped to reduce this data loss.

Due to a pressure sensor malfunction at the LSB_Church_Meadow site from 05/07/2017 13:20:00 UTC onwards, the discharge data until 28/09/2017 (when a new sensor was installed) were linearly interpolated between stage board readings taken during site visits. Data from the other LSB sites confirms that no significant storm events took place during the interpolated period, however these data should be used with caution if the purpose pertains to low or baseflows.

When using these data, it is worth noting that the catchment area draining these sites was impacted by the construction Natural Flood Management (NFM) interventions (e.g. storage ponds, arable reversion) from 2018 onwards. These interventions aim to slow and store the flow of water (and pollutants) through the catchment. See Old *et al.* (2019) and Robotham *et al.* (2021) for further details on these catchment interventions.

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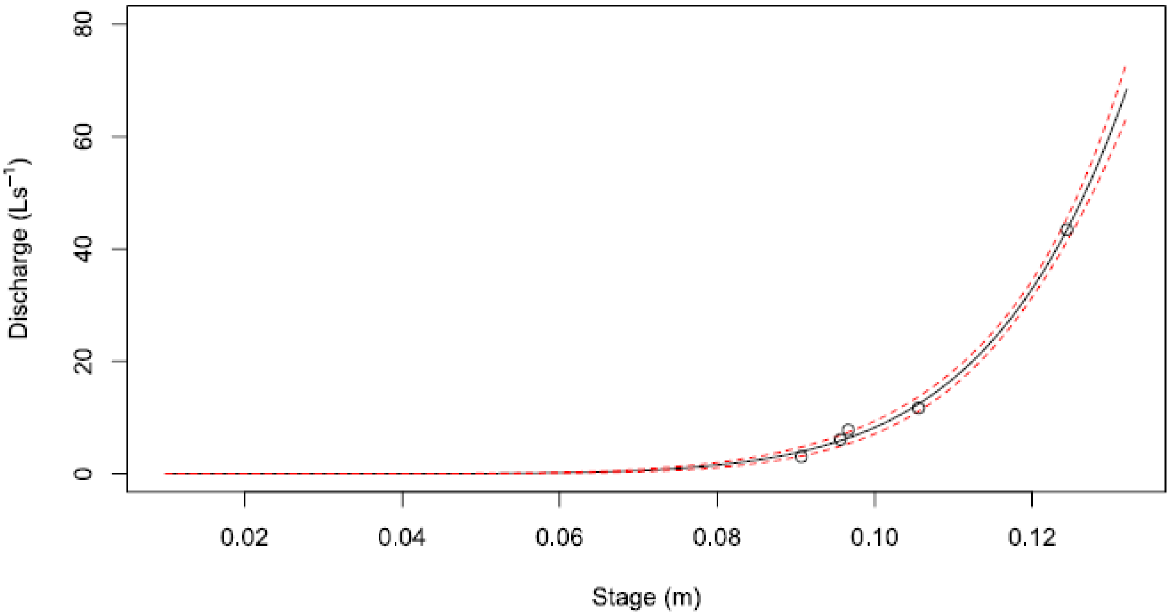
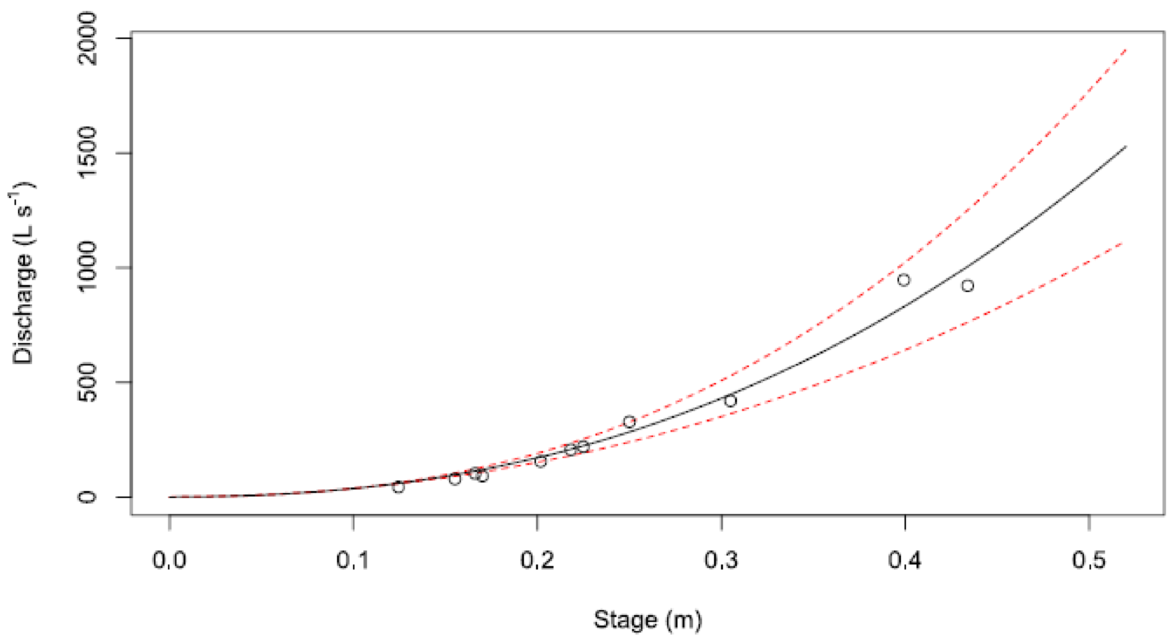
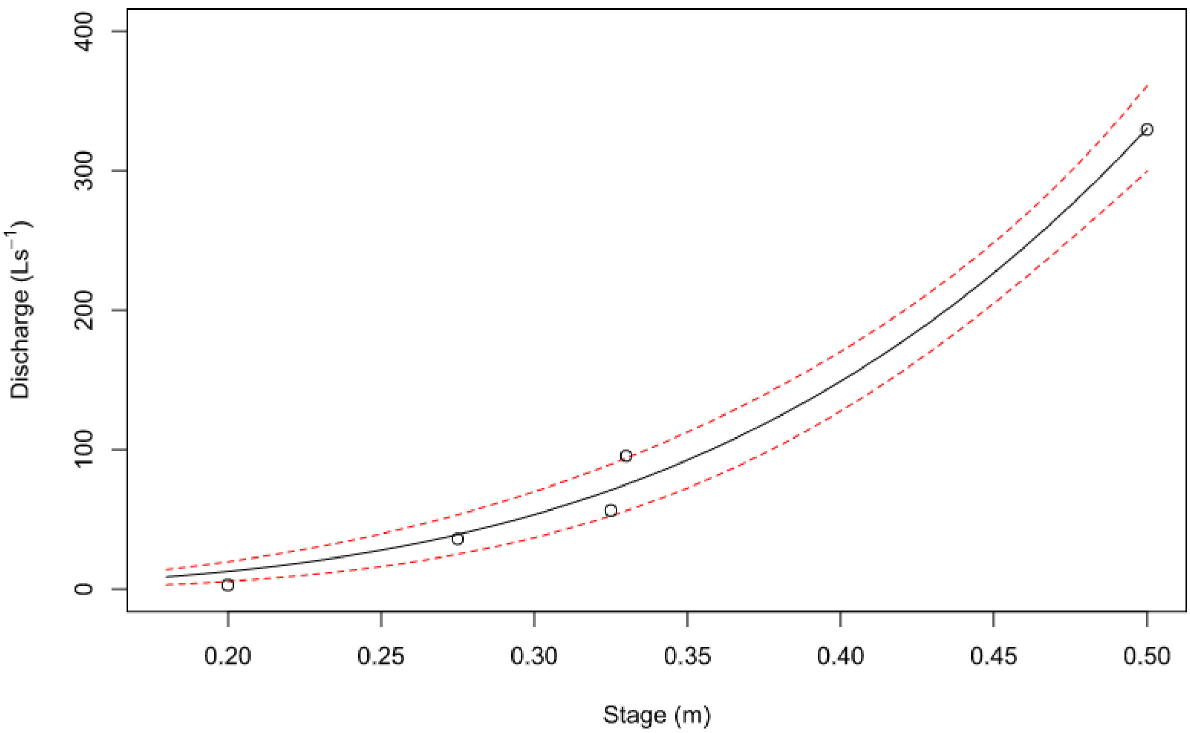
Appendix:



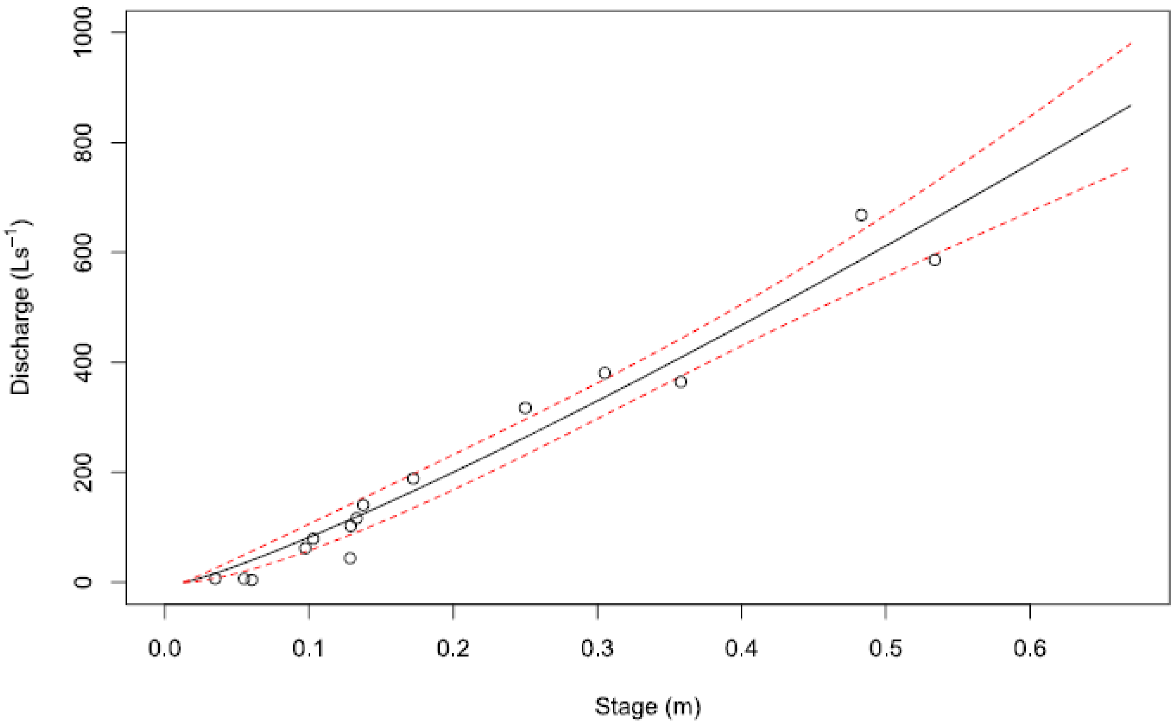
A1: Stage-discharge relationship during low flows (where stage <0.132) at the LSB_The_Heath site. 95% confidence intervals are shown in red.



A2: Stage-discharge relationship during high flows (where stage >0.132) at the LSB_The_Heath site. 95% confidence intervals are shown in red.



A3: Stage-discharge relationship at the LSB_Upstream_The_Heath site. 95% confidence intervals are shown in red.



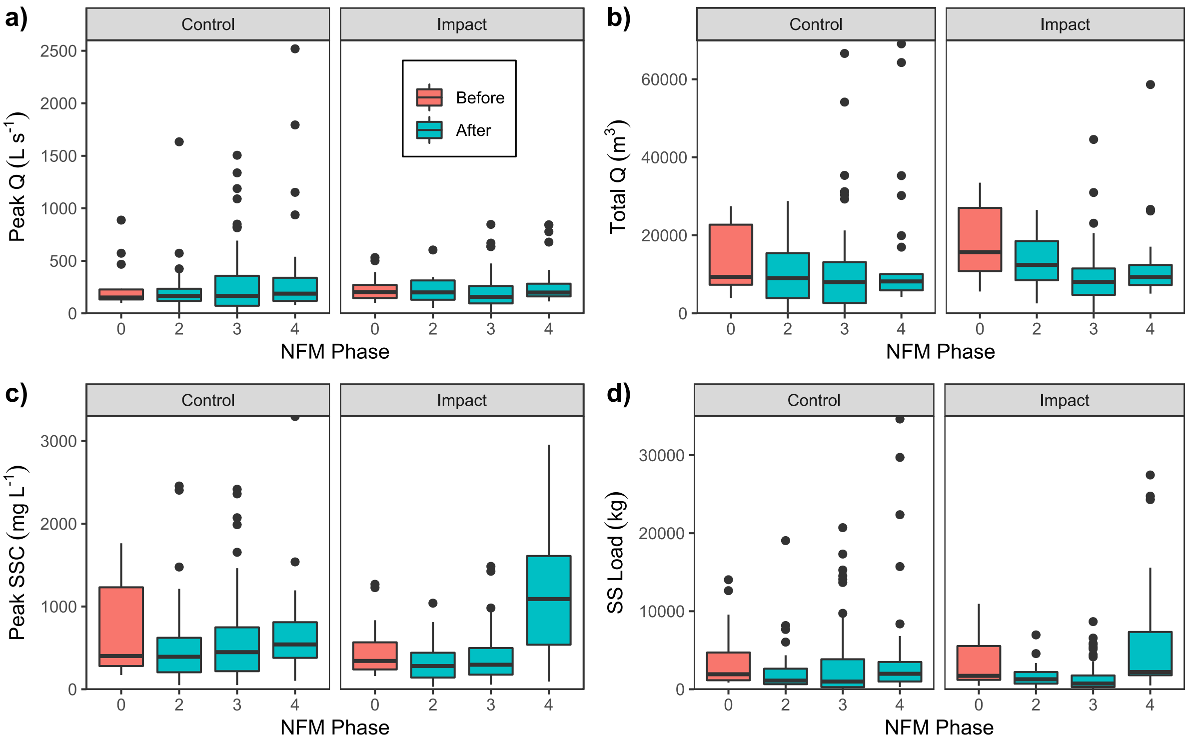
A4: Stage-discharge relationship at the LSB_Church_Meadow site. 95% confidence intervals are shown in red.

Appendix F Supplementary Materials for Chapter 5

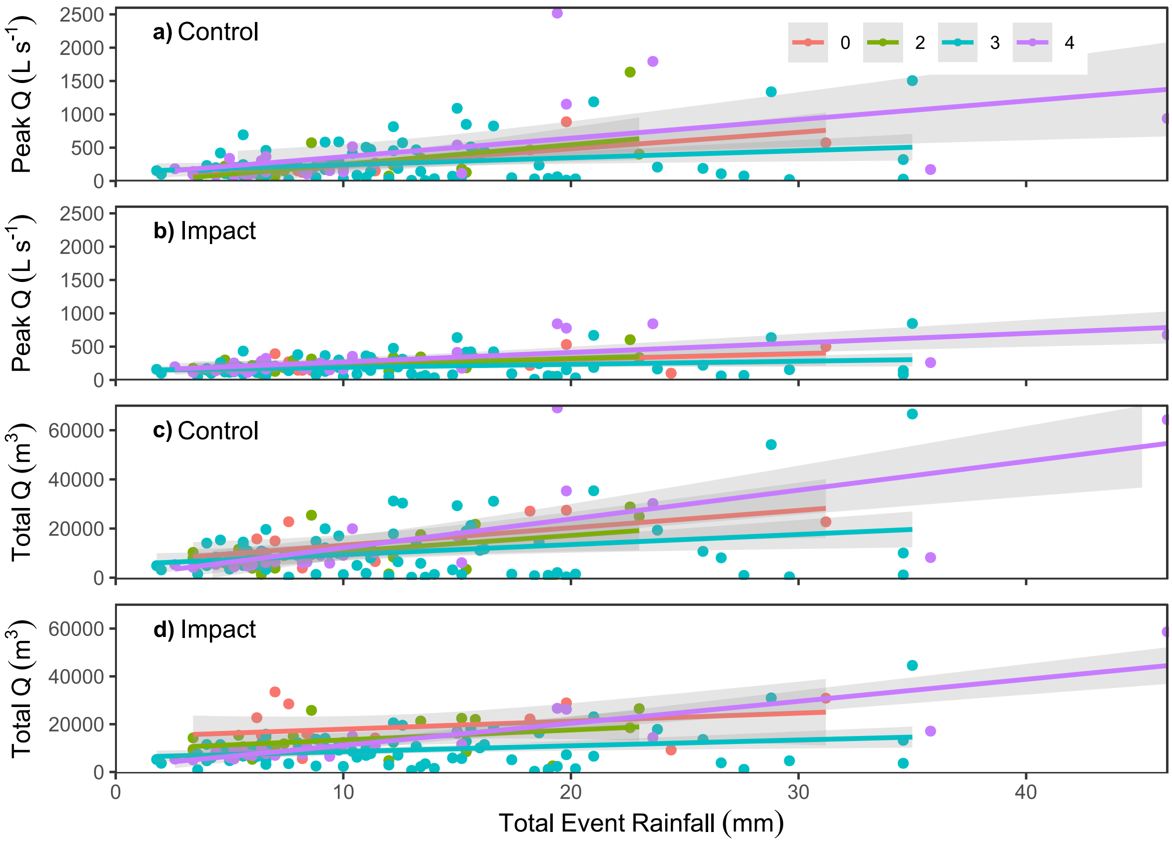
This appendix contains the supplementary materials for Chapter 5 of this PhD thesis.

Monitoring the effects of Natural Flood Management on water quality in an agricultural lowland catchment

Supplementary materials



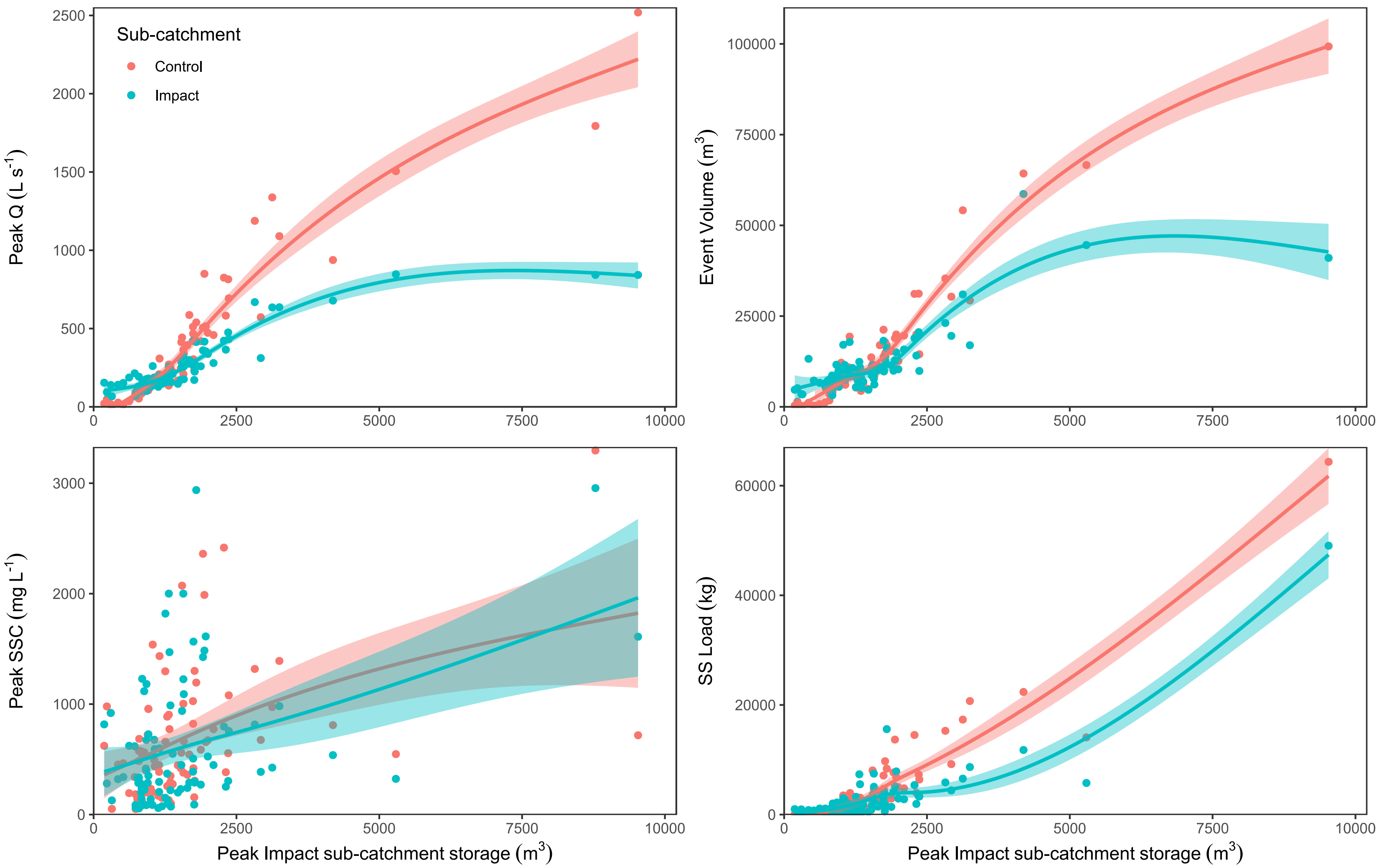
A1: Boxplots comparing the distribution of **a)** Peak Q (L s^{-1}); **b)** Total Q (m^3); **c)** Peak SSC (mg L^{-1}); and **d)** SS Load (kg) in the Control and Impact sub-catchment during each of the NFM Phases. Median values are denoted by the bold horizontal lines, and outlying values as individual points.



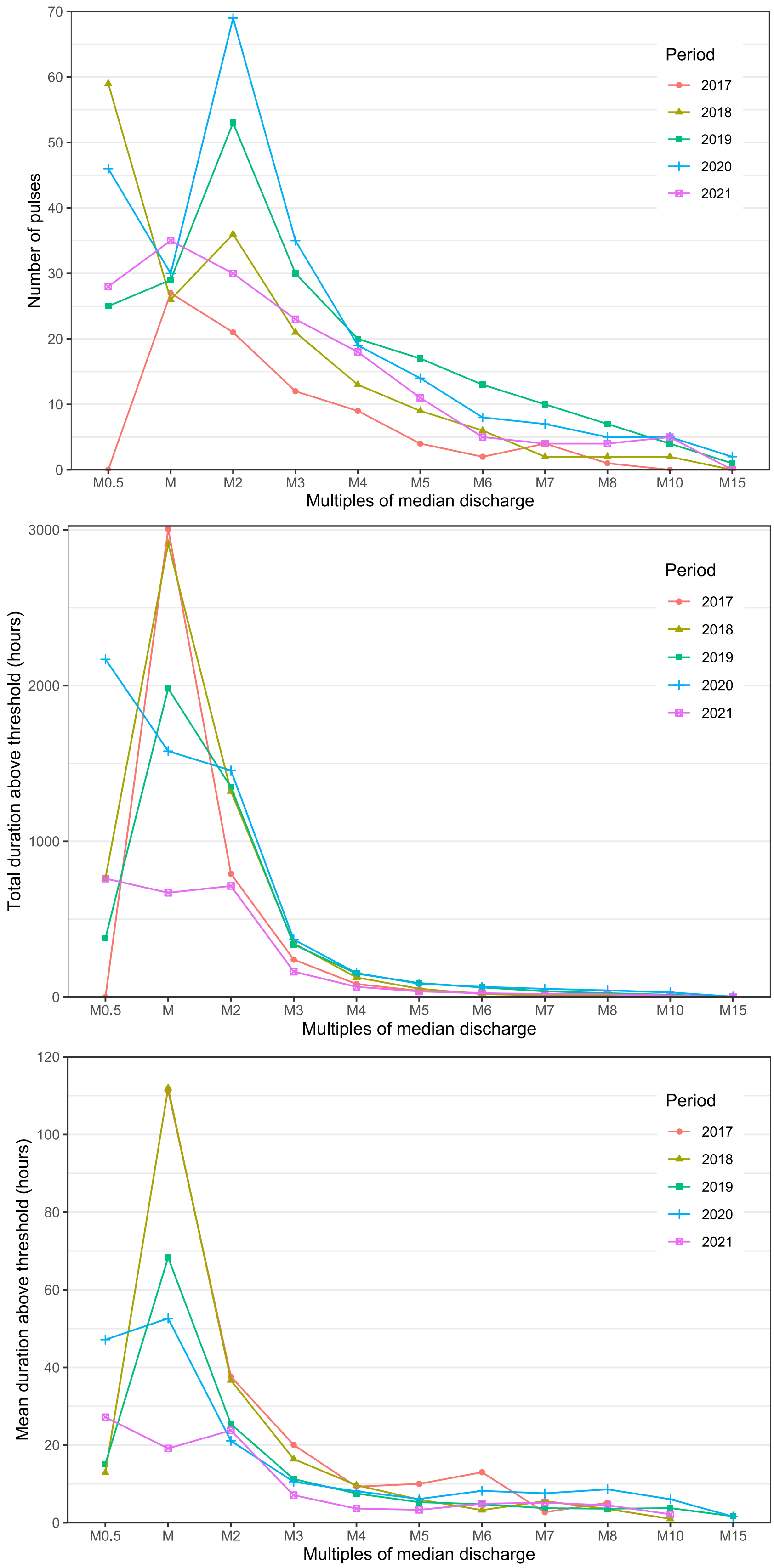
A2: Storm event peak Q (L s^{-1}) and Total Q (m^3) in the Control and Impact sub-catchments as functions of total event rainfall (mm) and the NFM Phase in which each event occurred. NB: Phase 0 events include those that occurred prior to Phase 2 (i.e., pre-NFM).

A3: Significance levels (*** is p<0.001; ** is p<0.01; * is p<0.05) of predictor variables used in the parsimonious GLMs for each response variable. NA is shown where predictor variables were not used as a result of the model selection process.

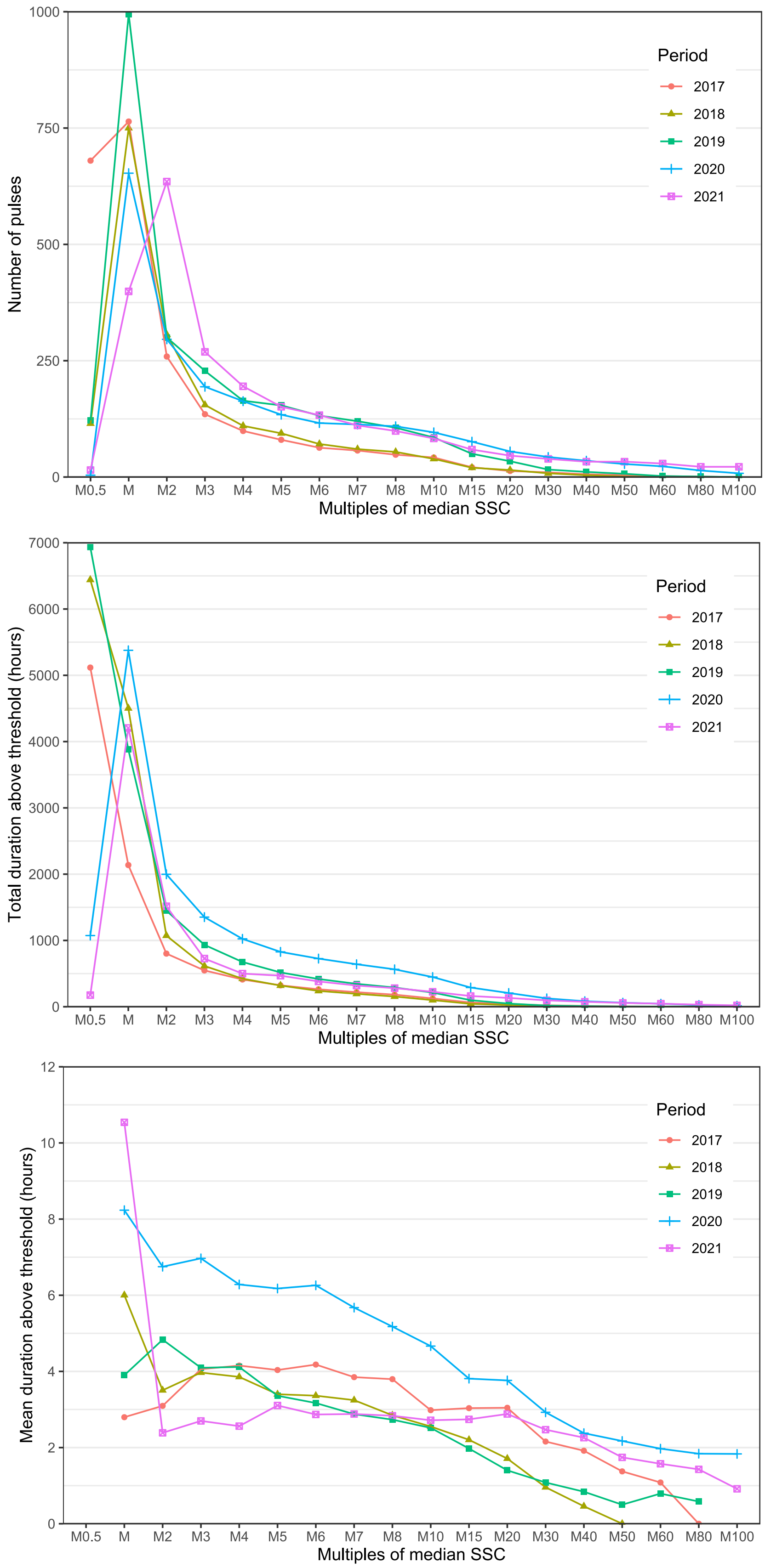
Response variable	Predictor variables							
	Total Rainfall	Mean instantaneous rainfall intensity	Peak Q	API	Exposed Soil	NFM Phase	NFM Phase × sub-catchment	AIC
Peak SSC	NA	***	NA	***	NA		** (Phase 4)	4192
Peak SSC	NA	***	***	***	NA		** (Phase 4)	4103
SS Load	***	*	NA	NA	***			5139
Peak Q	***	NA	NA	***	NA			3743
Total Q	***	NA	NA	***	NA	*** (Phase 3); * (Phase 4)		5958



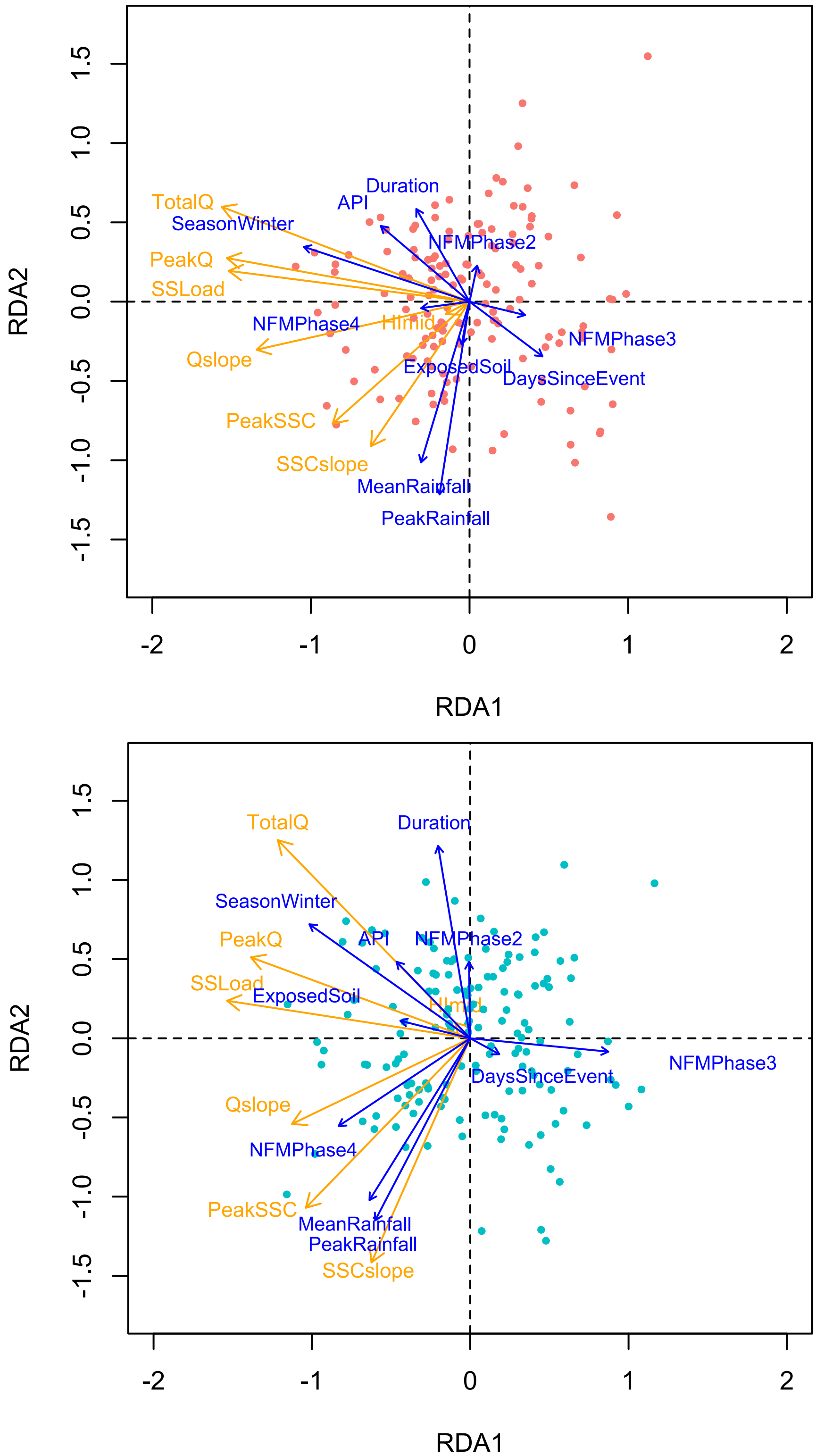
A4: Generalised additive models (GAMs) of storm event peak Q (L s^{-1}), event volume (m^3), peak SSC (mg L^{-1}), and SS load (kg) in the Control (red) and Impact (blue) sub-catchments as functions of peak water storage (m^3) in NFM features within the Impact sub-catchment. Shaded bands represent 95 % confidence intervals.



A5: Number of stream discharge pulses (top); total duration above threshold (hours) (middle); and mean duration above threshold (hours) (bottom) for each median (M) multiple and 12-month period in the Impact sub-catchment.



A6: Number of stream SSC pulses (top); total duration above threshold (hours) (middle); and mean duration above threshold (hours) (bottom) for each median (M) multiple and 12-month period in the Impact sub-catchment. [M0.5 data not shown on bottom plot due to scale]



A7: Triplots of the storm events in the Control (top; red points) and Impact (bottom; blue points) sub-catchments displayed along the first two axes identified by the respective RDAs. Blue and orange vectors represent explanatory and response variables respectively. The cosine of the angle between vectors reflects their linear correlation.

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