Quantifying and modelling spatio-temporal flood-mitigation, drought-resilience, and water-quality benefits provided by grassland interventions in the Eden Catchment (North-West England, UK)

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ABSTRACT

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Overland flow and rapid shallow-subsurface flow are hydrological pathways capable of quickly transporting precipitation to the stream network. These rapid pathways can greatly increase the flood-risk within a catchment, as well as cause water-quality degradation through the transport of harmful pollutants and pathogens, alongside encouraging other serious issues such as soil erosion, damage to infrastructure, mass siltation, and reduced agronomic efficiency. Given the economic and humanitarian costs associated with flooding and water-quality deterioration throughout the globe, efforts are needed to effectively control and limit such hydrological pathways.

Grasslands, specifically those used for intensive agriculture such as improved-pastures and meadows, encompass a large percentage of the land-use in many regions of the world. Many improved-pasture- and meadow-dominated catchments are prone to both flooding and water-quality issues, with concerns also growing regarding their resilience to drought. With such widespread international prevalence, incorporating grasslands within a land-use management framework could well be an effective method of mitigating against floods, droughts and water-quality deterioration in such areas. Despite such an extensive presence globally, the hydrological understanding of grasslands is extremely underdeveloped, with the understanding of surface

hydrodynamics and their controls on overland flow and rapid shallow-subsurface flow extremely limited.

The aim of this thesis was to quantify the observable hydrological change for several widespread grassland farming features and practices (interventions) that likely alter the spatio-temporal dynamics of overland flow and rapid shallow-subsurface flow and their hydrological controls. This was conducted by paired-plot experimentation, controlled experiments, and subsequent statistical and physics-based analysis and modelling of three improved-pasture- and meadow-dominated sub-catchments of the Eden catchment, North-West England, United Kingdom. The four investigated interventions were semi-natural grasslands (Chapter 4), blade-aeration (Chapter 5), hedgerow wild-margins (Chapter 6), and dry-stone walls (Chapter 7).

In Chapter 4, surface moisture patterns were compared between a semi-natural grassland and a bordering improved-pasture/silage field. Converting semi-natural grasslands into improved-pasture was shown to significantly reduce the natural diversity in surface soil moisture patterns, causing substantially more uniform responses to hydrological stresses. Improved-pastures were shown to naturally dry faster than neighbouring semi-natural grasslands in spring, although slurry applications were shown to offset this drying in summer and improve drought resilience. Slurry wetted improved-pasture became wetter than the semi-natural grassland at the beginning of the autumnal rains however, heightening the likelihood of overland flow and rapid shallow-subsurface flows. During sampling within a storm, both semi-natural grassland and improved-pasture were shown to visibly produce overland flow.

In Chapter 5, blade aeration was conducted on subsections of two improved-pasture and silage fields. Blade aeration was shown to significantly improve the topsoil permeability of an improved-pasture/silage field and significantly improve the penetration resistance between 5 cm – 15 cm, as well as substantially reduce the likelihood of infiltration-excess overland flow. Improvements were only seen in one of the two investigated sites however, with the alternate location showing no notable changes to permeability or infiltration-excess overland flow likelihood, and a significant increase in penetration resistance at 10 cm.

In Chapter 6, hedge-margin overland flow plots were compared against overland flow plots within an immediately adjacent improved-pasture. Hedge-margins were shown to significantly improve topsoil permeability and soil physico-chemical properties compared to the adjoining improved-pasture. Hedge-margins were slower to produce overland flow, requiring an equal or increased amount of saturation than the improved-pasture, and resultantly produced a lower total overland flow volume. Hedge-margins were also found to release more nitrate, nitrate-nitrite and loose sediment in comparison to the improved-pastures in a 'wash-off' experiment, and therefore may store more potential contaminants on the surface and possibly offer water-quality benefits.

In Chapter 7, the effect that dry-stone walls have on topsoil wetness was assessed by measuring soil volumetric wetness during saturated and near-saturated conditions above and below several dry-stone walls throughout the landscape within a large number of sloped improved-pastures. Dry-stone walls were shown to predominantly have an inconsistent and insignificant effect on soil volumetric wetness, although a possible rain shadow effect on a very localised scale (up to 3 m from the dry-stone wall) was observed.

The thesis has successfully quantified the role of four widespread grassland interventions present in improved-pasture and meadow dominated catchments throughout the globe in relation to how they alter surface hydrodynamics, specifically overland flow and rapid shallow-subsurface flow. This research has resultantly improved the understanding of the hydrological functioning of agricultural grasslands specifically in relation to flood-risk and water-quality, with some advances in drought-resilience also made. The thesis finally highlights several key areas for future hydrological research in relation to furthering the understanding of grassland hydrology, as well as offers improvements and advice for both hydrometric observations, experimental designs, as well as broader hydrological modelling, and hydrological and environmental science.

DECLARATION

I declare that this thesis is my own work, and has not been submitted in substantially the same form for the award of a higher degree elsewhere. Any sections of the thesis which have been published, or submitted for a higher degree elsewhere, have been clearly identified. The total regulated word count for the thesis is: 88,135.

Please find below a list of publications included within this thesis with information regarding my contributions to each of the publications. Manuscripts submitted for publication as well as unpublished manuscripts intended for future publication have also been included. There are minor modifications between thesis manuscripts and published manuscripts as per the viva voce examination process.

Published journal articles

1) Wallace, E.E. and Chappell, N.A. (2019). Blade Aeration Effects on Near-Surface Permeability and Overland Flow Likelihood on Two Stagnosol Pastures in Cumbria, UK. *Journal of Environmental Quality*, 48(6), pp.1766–1774. Doi: 10.2134/jeq2019.05.0182. [CHAPTER 5]

2) Wallace, E.E. and Chappell, N.A. (2020). A statistical comparison of spatio-temporal surface moisture patterns beneath a semi-natural grassland and permanent pasture: From drought to saturation. *Hydrological Processes*, 34(13), pp.3000–3020. Doi: 10.1002/hyp.13774. [CHAPTER 4]

3) Wallace, E.E., McShane, G., Tych, W., Kretzschmar, A., McCann, T. and Chappell, N.A. (2021). The effect of hedgerow wild-margins on topsoil hydraulic properties, and overland-flow incidence, magnitude and water-

quality. *Hydrological Processes*, 35(3). Doi: 10.1002/hyp.14098. [CHAPTER 6]

Published conference papers

4) Wallace, E.E. and Chappell, N.A. (2020). The spatiotemporal dynamics of surface soil moisture within upland grassland ecosystems. In: *Spatio-temporal and/or (geo) statistical analysis of hydrological events, floods, extremes, and related hazards*. EGU General Assembly. EGU. Doi: 10.5194/egusphere-egu2020-8798. [CHAPTER 4]

Unpublished/unsubmitted journal articles

5) Wallace, E. E., Kretzschmar, A., Hankin, B., Page, T. J. C. and Chappell, N. A. (2021). The effect of dry-stone walls on localised hydrological functioning. – With a future plan to submit to either: Agricultural Water Management, Hydrological Processes, Journal of Hydrology, or similar journals. [CHAPTER 7]

Paper 1) Leading up to the first paper, myself and NAC theorised the research question, and I created the experimental design and chose the research site with support from the Eden Rivers Trust (ERT) following work at several preliminary sites (please see appendices). I was additionally responsible for all field hydrometry, soil sampling and laboratory measurements. I conducted the majority of the statistical analysis and modelling of the results, as well as formulated the overall paper manuscript. NAC assisted in result interpretation within the discussion and with general manuscript revisions, as well as in providing field equipment.

Paper 2) Leading up to the second paper, myself and NAC theorised the research question, and I created the experimental design and selected the study site with support from ERT following work at a preliminary site (please see appendices). I conducted all field hydrometry, soil sampling, ecological/taxonomic measurements, and the topographic and laboratory analysis. I was additionally responsible for the majority of the statistical analysis and modelling, and the overall drafting of the paper. NAC supported the result interpretations and discussion of the paper, and also significantly assisted with the drafting of the implications and conclusions, as well as in general manuscript revisions.

Paper 3) Leading up the third (conference) paper, I was responsible for the drafting of the conference abstract, with NAC reviewing the abstract. I solely presented the conference paper.

Paper 4) Leading up to the fourth paper, myself and NAC jointly theorised the research question, and myself, NAC and GM jointly decided upon the experimental design and site selection, with support from the Lowther Estate Trust, ERT and JNQ following preliminary work (please see appendices). I and GM were responsible for plot construction. I conducted all field hydrometry, field-experiments, soil sampling, ecological/taxonomic measurements and laboratory analysis, with some support from TM regarding field-experiments and laboratory analysis and ERT regarding ecological/taxonomic interpretations. I and GM both conducted the topographic analysis. WT provided substantial support with the control-engineering analysis and interpretation, with AK also providing additional support. The discussion was written by myself, with feedback from all the authors regarding result interpretation.

Paper 5) Myself and NAC theorised the research question, which was advanced upon by all authors, with some input from ERT. I conducted all of the field hydrometry, with some assistance from AK. I conducted all statistical analysis and topographic analysis, with AK assisting with topographic analysis. I drafted up the manuscript, with feedback provided by NAC (and Prof. Keith Beven and Dr. Sim Reaney).

At all stages, NAC, WT and JNQ provided support relating to the experimental designs, use of laboratories, and fieldwork equipment.

Agreed and dated:

Ethan Wallace

Ethan Wallace – 1st February 2021

Nick Chappell

Nick Chappell – 1st February 2021

Barry Hankin

Barry Hankin – 2nd February 2021

Ann Kretzschmar

Ann Kretzschmar – 2nd February 2021

Thomas McCann

Thomas McCann – 2nd February 2021

Gareth McShane

Gareth McShane – 4th February 2021

Trevor Page

Trevor Page – 2nd February 2021

John Quinton

John Quinton – 2nd February 2021

Wlodek Tych

Wlodek Tych – 2nd February 2021

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PROJECT OUTPUTS

Journal articles

1) Wallace, E.E. and Chappell, N.A. (2019). Blade Aeration Effects on Near-Surface Permeability and Overland Flow Likelihood on Two Stagnosol Pastures in Cumbria, UK. *Journal of Environmental Quality*, 48(6), pp.1766–1774. Doi: 10.2134/jeq2019.05.0182.

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Conference papers

1) Wallace, E.E. and Chappell, N.A. (2020). The spatiotemporal dynamics of surface soil moisture within upland grassland ecosystems. In: *Spatio-temporal and/or (geo) statistical analysis of hydrological events, floods, extremes, and related hazards*. EGU General Assembly. EGU. Doi: 10.5194/egusphere-egu2020-8798.

Conference posters

1) Wallace, E. E., Chappell, N. A., Tych, W., Quinton, J. N., Dyer A., Kidd, S. and Dahl, L. (2019). *Quantifying flood mitigation and water-quality benefits provided by pasture interventions within the Leith, Lowther and Petteril catchments.*

[poster]. Natural Flood Management: Does it work? [British Hydrological Society]. Lancaster University, 24th April 2019. Doi: 10.13140/RG.2.2.18511.02729.

Quantifying flood-mitigation and water-quality benefits provided by pasture interventions within the Leith, Lowther and Petteril catchments (Eden Catchment, Cumbria, UK).

Ethan E. Wallace (e.wallace@lancaster.ac.uk), Nick A. Chappell, (Lancaster University) Wlodek Tych (LU), John N. Quinton (LU),
Andy Dyer (Eden Rivers Trust), Sarah Kidd (ERT), Lev Dahl (ERT)

1) Introduction

As part of agri-environmental payment schemes, British livestock farmers are being asked to incorporate pasture interventions into their farming systems in an effort to reduce downstream flood-risk and water-quality degradation.

Very little data exists regarding the effectiveness of these interventions, which makes hydrological modelling highly uncertain.

2) Aim and objectives

Aim: To quantify runoff reduction (overland flow and streamflow) caused by the following interventions:

Objectives

1) Grassland aeration

2) Rough-grazing

3) Stone-walls

4) Hedgerow buffer-strips

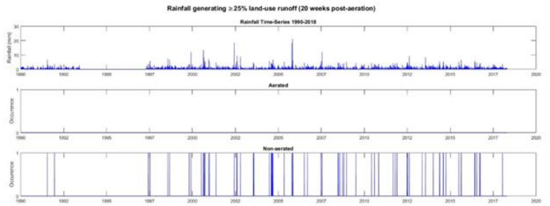
5) Channel re-alignment

All objectives involve field monitoring, with interventions contrasted against neighbouring pasture. Results are statistically analysed and modelled at plot, sub-catchment and full-catchment scales using systems engineering and physics-based modelling tools to quantify flood-risk and water-quality improvements.

3) Current results and discussion

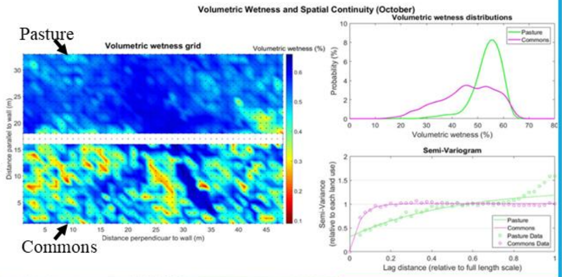
3.1 Grassland aeration

- Aeration improved permeability for at least 20 weeks, and reduced simulated overland flow by up to 100%.



3.2 Rough-grazing

- During early storm season prior to saturation (late Oct), grazed commons had significantly reduced ($p<0.001$) soil volumetric moisture content ($\tilde{x}=46.6\%$) compared to permanent pasture ($\tilde{x}=54.6\%$).



3.3 Stone-walls

- 33% of sampled stone walls significantly ($p<0.05$) held up soil moisture during saturated conditions, when measuring over 1/4 miles of stone-walls at one meter resolution (16 meters above and below the wall).

A (broken) stone-wall slowing flow at Gaythorne Hall in a neighbouring catchment of the Lowther.

Lancaster University

Centre for Global Eco-Innovation

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Eden Rivers Trust

COMMONLY USED ABBREVIATIONS

Agriculturally-improved pasture – AIP

Agriculturally-improved pasture plot one – P1

Agriculturally-improved pasture plot two – P2

Anderson-Darling – AD

Antecedent Precipitation Index – API

Backward shift operator – z^{-1}

Before-After-Control-Impact – BACI

Brown-Forsythe – BF

Catchment Hydrology and Sustainable Management – CHASM

Computer Aided Program for Time-series Analysis and the Identification of Noisy systems – CAPTAIN

Data-based Mechanistic Modelling – DBM

Dissolved organic carbon – DOC

Dissolved total phosphorus – DTP

Eden Demonstration Test Catchment – Eden DTC

Extreme value theory – EVT

Field Pasture 1 – FP1

Field Pasture 2 – FP2

Field Pasture 1A: Aerated section – FP1A

Field Pasture 1N: Non-aerated section – FP1N

Field Pasture 2A: Aerated section – FP2A

Field Pasture 2N: Non-aerated section – FP2N

Garbage-In-Garbage-Out – GIGO

Geographic Information System – GIS

Hedgerow Wild-Margin – HWM

Hedgerow wild-margin plot one – M1

Hedgerow wild-margin plot two – M2

Hydrologic Engineering Core – Hydrologic Modeling System – HEC-HMS

Infiltration-excess overland flow – IOF

Kolmogorov-Smirnov – KS

Mann-Whitney-Wilcoxon – MWW

Mass of dry soil – m_s

Mass of water – m_w

Maximum observed rainfall intensity – MORI

Natural Flood-Risk-Management – NFRM

Nitrate-Nitrite – $NO_3^-NO_2^-$

Overland flow and rapid shallow-subsurface flow – OFRSSF

Overland flow plot – OFP

Particulate total phosphorus – PTP

Peaks-Over-Thresholds – POT

Permanent Pasture – PP

Pure time-delay – δ

Refined Instrumental Variable – RIV

Saturated hydraulic conductivity – K_{sat}

Saturation-excess overland flow – SOF

Semi-natural grassland – SNG

Simplified time-domain reflectometry – sTDR

Soil dry bulk-density – ρ_b

Soil organic matter – SOM

Soil penetration resistance – PR

Soil porosity – η

Soil volumetric wetness – θ_V

Soluble reactive phosphorus – SRP

Steady-state-gain – SSG

Système Hydrologique Européen – SHE

Time Varying Parameter – TVP

Time constant – TC

Time-domain reflectometry – TDR

Topographic Model – TOPMODEL

Total sediment – TS

Transfer function – TF

Volume of air – V_a

Volume of soil – V_s

Volume of water – V_w

Water Framework Directive – WFD

Youngs Information Criterion – YIC

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1 INTRODUCTION

1.1 Hydrological processes and hydrological hazards

Hydrological processes are an integral part of earth system science that regulates the global hydrological cycle. These processes determine flood, drought and water-quality hazards throughout the globe, which resultantly can have substantial consequences for human life. Essentially all regions of the world are affected by hydrological hazards and associated disasters in some manner, and there is growing concern that anthropogenic emissions and natural climate variability may be modifying and potentially amplifying such hazards (Dankers and Feyen, 2008; Whitehead et al., 2009a; 2009b; Pall et al., 2011; Watts et al., 2015; Roudier et al., 2016; Schaller et al., 2016; Blöschl et al., 2017; 2019a). This increased threat is intensified by increasing pressures upon environmental systems and changing land-uses, as well as an increasing at-risk global population (Güneralp et al., 2015; Arnell and Gosling, 2016; Stevens et al., 2016). Consequentially, hydrological hazards are a serious contemporary and future concern for most of the world.

Flooding is a significant threat throughout much of the globe, and is generally considered the leading hydrological hazard in temperate climates. Flooding can come in many forms such as fluvial (riverine/channel) flooding, lacustrine (lake) flooding, coastal flooding, groundwater flooding, sewer flooding, and pluvial flooding (flooding in the absence of an overflowing water body/watercourse, sometimes termed surface-water flooding), with floods of any origin that appear suddenly often described as flash floods. Within Europe, a recent series of widespread and destructive floods have occurred in the last three decades, including: autumn 2000 (Barredo, 2007; Kundzewicz et al., 2013), summer 2002 (Ulbrich et al., 2003a, 2003b; Barredo, 2007;

Blöschl et al., 2013a), summer 2005 (Barredo, 2007; Kundzewicz et al., 2013), spring-summer 2006 (Mikhailov et al., 2008), summer 2009 (Mendoza-Tinoco et al., 2020), spring-summer 2010 (Bissolli et al., 2011; Kundzewicz et al., 2013; Romanescu et al., 2018), summer 2013 (Blöschl et al., 2013a; Grams et al., 2014), spring 2014 (Stadtherr et al., 2016), and spring-summer 2016 (Bronstert et al., 2018).

Within the UK, flooding is largely considered the primary environmental concern, with £200 billion of assets currently at-risk (GOS, 2004; CCC, 2017). Recent major and widespread UK floods include: October – December 2000 (Kelman, 2001; Barredo, 2007), December 2002 – January 2003 (Marsh, 2004), August 2004 (Golding et al., 2005; Knight and Samuels, 2007), January 2005 (Knight and Samuels, 2007), June – July 2007 (Marsh and Hannaford, 2007), November 2009 (Miller et al., 2013), April – December 2012 (Parry et al., 2013), December 2013 – February 2014 (Muchan et al., 2015), and December 2015 – January 2016 (Barker et al., 2016). Flood damage in the UK is estimated to be £1.3-1.4 billion a^{-1}, with an additional £800 million a^{-1} on erecting/maintaining flood-defences (GOS, 2004). Flooding also causes considerable levels of social damage nationally (Munro et al., 2017); however, the number of direct, flood-related fatalities remains very low. Extreme rainfall and river discharges as well as rising sea levels are predicted to occur more frequently and with considerably higher magnitude across much of the UK in the future, meaning flooding remains a serious and increasing economic threat (Fowler et al., 2007; Haigh et al., 2011; Prudhomme et al., 2012; Lavers et al., 2013; Kendon et al., 2014; Watts et al., 2015; Collet et al., 2018).

Water-quality deterioration is another significant hydrological hazard affecting much of the world. This deterioration can come in various forms from the eutrophication, acidification and degradation of natural ecosystems, to the siltation of key waterbodies, increased water temperatures, or outbreaks of water-borne infectious disease (Kay et al., 2009; Semenza and Menne, 2009; Collins et al., 2010; Howden et al., 2010; Fewtrell et al., 2011; Brown and Murray, 2013; Hannah and Garner, 2015). Across Europe and within the UK, water-quality is generally moderate-high, although industrial, domestic and agricultural activities can often put increased pressure on natural water systems. Shifting weather patterns, extreme events (both floods and droughts), as well as changing land-use are all likely to have substantial impacts upon a range of water-quality parameters (Kay et al., 2009; Whitehead et al., 2009a; 2009b; Watts et al., 2015). Given that water-quality is essential for domestic consumption, agriculture, numerous industries, as well as being key to the health of biological ecosystems; maintaining high water-quality and preventing water-quality hazards or deterioration is a priority throughout the globe (Collins et al., 2010).

Drought is the final major hydrological hazard that exists worldwide. Droughts can cause substantial damage such as crop failures, reduced water supply/abstractions for domestic, agricultural and industrial purposes, as well as deteriorated water-quality through the concentration of pollutants (Mosley, 2015; Webber et al., 2018; Salmoral et al., 2019; Bussi et al., 2020). Recent widespread and economically costly droughts have struck across Europe in recent years such as 1992 (Spinoni et al., 2015), 2003 (Rebetez et al., 2006; Spinoni et al., 2015), 2007-2008 (Spinoni et al., 2015), 2011-12 (Spinoni et al., 2015; Zahradníček et al., 2015), 2015 (Ionita et al., 2016; Hoy et al., 2017) and 2018 (Masante et al., 2018; Buras et al., 2020).

Although generally seen as subordinate in importance to flooding (and water-quality), droughts are a growing concern in the UK, potentially becoming very severe in the future (Prudhomme et al., 2012; EA, 2013; Rahiz and New, 2013; Watts et al., 2015; CCC, 2017; Collet et al., 2018; Spinoni et al., 2018). The future risk of drought is amplified as relatively large and densely populated sections of the country are already considered water-scarce (EA, 2013; EEA, 2018). Recent widespread/damaging UK droughts include: 1995-1997 (Parry et al., 2012; Spinoni et al., 2015; Barker et al., 2019), 2004-2006 (Spinoni et al., 2015; Barker et al., 2019), 2010-12 (Kendon et al., 2013; Todd et al., 2013; Spinoni et al., 2015; Barker et al., 2019) and 2018 (Buras et al., 2020).

Hydrological hazards are evidently both frequent and widespread across Europe and the UK, and can be economically, environmentally and humanitarianly costly throughout the globe. It is therefore beneficial to reduce the risks that these hazards pose whenever possible. In order to reduce these risks, an increased understanding of the underlying scientific processes controlling such hazards is needed. An increased understanding of hydrological processes is therefore necessary to help alleviate the risks posed by hydrological hazards. From here on, the primary focus will be upon flood-risk within a UK, and to a lesser extent, European and global context, with secondary focus provided to both water-quality and drought-risk.

1.2 Hydrological processes generating streamflow and stormflow

The hydrological catchment is the area that contributes all the water that passes through a given cross-section of a stream, with the catchment boundary known as the watershed or divide. Essentially all water enters the land phase of the hydrological cycle by being deposited as precipitation within the catchment. It is the following hydrological processes, which convert this precipitation into streamflow, with particular emphasis on stormflow (elevated streamflow), that are described here.

Streamflow is a catchment-scale phenomenon that represents all the end products of the hydrological cycle, and consists of a series of streamflow peaks (stormflows) between periods of steady, much lower streamflow (baseflows). Stormflows (or flood hydrographs) are the streamflow immediately during and after a significant rain event, and is often the main interest of hydrologists as stormflows often determines the incidence and extent of flooding. The characteristics of each precipitation event alongside each individual catchment controls the paths and rates of water movement to the outlet, and therefore the magnitude and timing of stormflows (Dingman, 1994). The water volume and velocity within these hydrological pathways is therefore important in determining stormflows, and hence flood-risk, as well as for the rate of solute and sediment transport, and hence water-quality. Understanding which hydrological processes are occurring is therefore crucial in order to moderate stormflow and therefore mitigate against flooding and water-quality degradation, and to some extent, drought.

1.2.1 Direct channel precipitation

Direct channel precipitation is precipitation that falls into waterbodies within a catchment (streams, rivers, lakes, reservoirs etc.) without making contact with the land surface. Direct channel precipitation can either be precipitation directly entering a waterbody or as throughfall following contact with a vegetation canopy. Direct channel precipitation clearly contributes to stormflow, although direct channel precipitation in the UK alone cannot account for all of stormflow, and therefore water must also be being transferred from the land surface to the channel. It is thought that the importance of direct channel precipitation is spatially-temporally variable and is linked to the size of the catchment area consisting of open-bodies of water and surrounding saturated-zones (Crayosky et al., 1999), and is important in areas with low runoff coefficients (Beven, 2012).

1.2.2 Overland flow pathways

Overland flow is water that flows across the surface of a catchment, including flow within the vegetation litter-layer. This can occur as a thin film over large areas (sheet flow), or more commonly concentrating in minor incisional channels on the soil surface known as rills (rill flow) or gullies (gully flow), also known as concentrated flow paths (Stevens and Quinton, 2008). Overland flow tends to be an extremely fast hydrological pathway with recorded velocities well in excess of 1000 metres per hour during both field and laboratory experiments (Emmett, 1970). Overland flow pathways are therefore capable of rapidly transferring precipitation to the stream network, thereby contributing towards stormflow (Anderson and Burt, 1990; Brutsaert, 2005). Overland flow additionally contributes towards water-quality degradation as it rapidly mobilises and transports solutes and pollutants (see Section 1.3).

There are four key mechanisms for overland flow in the UK. One of the most common is saturation-excess overland flow (SOF), also known as Dunne/Dunneian overland flow (Cappus, 1960; Dunne and Black, 1970). Saturation-excess overland flow occurs when a soil (or rock/regolith) is fully saturated so that no additional rainfall (irrespective of intensity) can infiltrate into the ground and is temporarily stored on, or flows off, the soil surface. This occurs when all possible pore-space (i.e., storage) within the soil has been completely filled by water (see Sections 2.3.2 and 2.4.4). Saturation-excess overland flow is most common during winter when soils are largely saturated, and can occur even with very light rainfall intensities if conditions are suitable (Barker et al., 2016). Saturation-excess overland flow is believed to originate from specific subareas of a catchment that are prone to saturation, known as the partial area concept, often surrounding streams and at the base of large hillslopes, as well as in areas of convergence (Ragan, 1968; Anderson and Burt, 1978). This SOF generating area is spatio-temporally dynamic, both within and between storms, often termed the ‘variable-source area’ concept (Hewlett and Hibbert, 1967).

Return-flow is another relatively common form of overland flow, and is the exfiltration of sub-surface water onto the soil surface before reaching the stream network (Cook, 1946). Exfiltration often occurs in areas of low transmissivity or soil moisture storage, within areas of decreasing gradient (e.g., at the base of hillslopes), within highly saturated areas (e.g., around the banks of channels and waterbodies), and within areas where flow lines converge into plan form (Anderson and Burt, 1990; Bracken, 2010; Shaw et al., 2011). Return-flow is fairly common for shallow soils overlaying impermeable subsoils, as well as at the base of long, low-gradient hillslopes and in areas where the streamside is particularly flat and/or wide (Shaw et al., 2011; Beven, 2012). Return-flow can be thought of as SOF caused by a non-rainfall mechanism. Return-flow is largely responsible for overland flow when rainfall is not immediately present.

Infiltration-excess overland flow (IOF) or Hortonian overland flow is another form of overland flow (Horton, 1933). Infiltration-excess overland flow occurs when the rainfall-intensity exceeds the infiltration-capacity of the soil. Infiltration-capacity is defined as the rate at which water can enter into the ground surface under unit cross-sectional area and unit hydraulic gradient (i.e., the saturated hydraulic conductivity – see Section 2.3.3). Infiltration-excess overland flow can occur at any soil saturation, and is usually associated with intense, convective downpours that primarily occur in summer (Dingman, 1994). It is believed IOF is fairly rare in the UK due to low rainfall intensities that rarely exceed soil infiltration-capacities, although soil infiltration-capacity can be reduced to the extent that IOF is possible, such as in the wheel-tracks of farm machinery (Silgram et al., 2015). Soil infiltration-capacities are spatially-temporally variable, so IOF may only be generated from certain sections of a catchment (Betson, 1964). When IOF does occur, it tends to not travel far, as overland flow generated on one section of land may infiltrate as it traverses nearby more-permeable areas, so called run-on-run-off phenomena (Yair and Lavee, 1976; Bonell and Williams, 1986; Bergkamp, 1998).

The final overland flow mechanism possible in the UK is hydrophobic overland flow. Hydrophobic overland flow is when the soil, leaf-litter and/or vegetation root-mat becomes excessively dry and results in the accumulation of certain waxy, organic compounds from plants, fungi and microorganisms which facilitate hydrophobicity (Chan, 1992; Dingman, 1994; Doerr et al., 2000; Martínez-Zavala and Jordán-López, 2009; Young et al., 2012). This results in water repellency, reduced infiltration, and consequential overland flow (Doerr et al., 2000). Wildfires and controlled burns can vaporize these waxy, organic compounds and can transport this hydrophobic layer below the soil surface (DeBano, 2000a, 2000b). Certain plants, soil conditions and land uses can also facilitate soil hydrophobicity (Bond, 1964; Doerr et al., 1998; de Jonge et al., 1999; Doerr et al., 2000; Martínez-Zavala, and Jordán-López, 2009). Increasing the organic content of soil may also encourage soil hydrophobicity (Doerr et al., 2000), such as via slurry application. Temperature (de Jonge et al., 1999), relative humidity (Doerr et al., 2002) and water content (Dekker and Ritsema, 1994; de Jonge et al., 1999) can all also affect soil hydrophobicity.

Combining the volume of water from direct channel precipitation, and overland flow (which is frequently absent during the majority of precipitation events in the UK), shows that these hydrological pathways are insufficient to produce observed stormflows (Davie, 2008). Therefore, stormflow must also involve hydrological pathways below the surface (Burt, 1989). These subsurface pathways which contribute to stormflow are primarily shallower subsurface flow paths, which also partially controls the variable-source areas for SOF (Anderson and Burt, 1982).

1.2.3 Rapid subsurface flow pathways

Shallow-subsurface flow is water that flows relatively close to the soil-surface and contains many rapid hydrological pathways that contributes towards the majority of stormflow (Dingman, 1994; King et al., 2014). Pipeflow is preferential flow within large conduits within the soil profile such as natural pipes and large soil fractures, as well as within man-made field-drains (Jones, 1971; Hatch et al., 2002; Holden and Burt, 2002; Holden, 2005). Natural conduits tend to be formed, enlarged and maintained by shear stresses of water-flow and by liquefaction. Flow within pipes can equal or even exceed overland flow velocities, especially if orientated downslope. The area, length, and continuity of conduits are extremely important in determining the speed of flow, as is the amount of water both within and surrounding the conduits.

Macropore flow is essentially pipeflow but in smaller, often less continuous conduits (structural pores) such as animal or earthworm burrows, root channels, and small-scale soil cracks, although there is no strict distinction between pipeflow and macropore flow (Anderson and Burt, 1990; Brutsaert, 2005). Macropores can transfer water downslope considerable distances through otherwise unsaturated soils – known as bypass/preferential flow (Mosley, 1979; Beven and Germann, 1982; Germann, 1986; 1990; Beven and Germann, 2013). Although generally slower than pipeflow, macropore flow can equal or exceed the transport speed of overland flow in some scenarios (Whipkey and Kirkby, 1978).

Another rapid shallow-subsurface hydrological pathway is flow above layers of impeding soil, rock or regolith (Chappell et al., 1990; Chappell and Sherlock, 2005). This occurs when infiltrated water cannot percolate vertically and is forced to travel laterally. This can be caused by naturally impermeable soils or geologies (surficial or bedrock), causing flow along the bedrock-soil interface, such as those found at MaiMai in New Zealand, Panola in the USA or Fudoji in Japan (McDonnell, 2003). Flow above layers of impeding soil, rock or regolith can also be due to anthropogenic impacts such as plough pans.

Pressure-waves in the shallow subsurface can also cause rapid responses at the hillslope scale (Chappell et al., 1998). When fully saturated conditions exist in the subsurface, the effects of inputs to the saturated zone can be transmitted as a pressure/propagation wave, at a wave velocity or celerity which will travel at a higher flow velocity than water (Beven, 2012). This celerity is controlled by how quickly the input can fill the profile of soil moisture deficit above the saturated zone (Beven, 2012). This pressure-wave displaces 'pre-event' or 'old' water during storms and causes stormflow to have a chemical signature more similar to 'pre-event' water rather than 'new' or 'event' water (Sklash and Farvolden, 1979).

1.2.4 Slower subsurface pathways

Slow shallow-subsurface flow is termed matrix flow, which is water that flows through the entire soil profile within micropores between individual soil particles rather than within conduits that bypass the bulk soil volume (i.e., it is non-preferential flow). Matrix flow can contain areas of preferential flow which bypasses sections of the soil matrix, known as bypass flow or finger flow, which has been evidenced by many tracer experiments (see Beven and Germann, 2013). Bypass flow may have some crossover with macropore flow as both are preferential flow pathways, although the latter may be considered to contain more continuous pathways (Gerke, 2011). Matrix flow is generally much slower than the above-mentioned hydrological pathways and has a much lower impact upon stormflow or flood-risk (Germann, 1990). Increased saturation and lightly textured soils tend to increase the speed of matrix flow, although this pathway contributes relatively little towards flood hydrographs. Within unsaturated soil, tracer studies suggest that retention times of matrix water can be much longer than the time scale of the flood hydrograph, even if some of that water might be displaced from the slope storage within a flood hydrograph (Harman et al., 2015).

Water that percolates below shallow-subsurface hydrological pathways is assumed to enter into deeper, very slow hydrological pathways such as flow within subsurface regolith and rock aquifers. These hydrological pathways have residence times so large that short-term pulses of input are damped out, and these act more as storage components such as groundwater reservoirs rather than a route for rapid transport to the stream network. These pathways are additionally more strongly associated with baseflows rather than stormflows (Whipkey and Kirkby, 1978).

1.3 Hazards posed by key stormflow hydrological pathways

Most stormflow in the UK is generated from overland flow (if present) and rapid shallow-subsurface hydrological pathways, hereby termed OFRSSF (Anderson and Burt, 1990; Perks et al., 2015). The partitioning of rainfall into OFRSSF and other hydrological pathways and stores is therefore a significant driver of fluvial flood-risk given the rapid transport properties of OFRSSF (Dingman, 1994; Deasy et al., 2009; Pattison and Lane, 2011). Overland flow and rapid shallow-subsurface pathways are also often associated with pluvial flooding. Reducing the occurrence and magnitude of OFRSSF, especially in key locations, is therefore a potential method to mitigate against flood-risk.

The rapid transport properties of OFRSSF also suggest that it is a significant determinant of water-quality, especially during stormflows. This is because OFRSSF tends to be highly erosive due to concentrated, high-velocity flow; meaning soil is easily entrained and carried into stream networks (Deasy et al., 2009; Bracken, 2010; Verachtert et al., 2011; Perks et al., 2015). Furthermore, most biological and chemical contaminants tend to be located either at or very close to the soil-surface, and are therefore easily entrained within OFRSSF either as a solute or directly attached to soil particles (Snyder and Woolhiser, 1985; Wallach, 1991; Shi et al., 2011). The rapid transport properties of OFRSSF additionally restricts biological uptake or physico-chemical degradation of contaminants, meaning pollutants may still be active when reaching the hydrological network.

Overland flow and rapid shallow-subsurface flows are clearly important in relation to hydrological hazards, but can also contribute towards related and serious issues such as soil erosion and reduced agronomic efficiency (Govers et al., 1996; Perks et al., 2015; Szönyi et al., 2016; Evans et al., 2020). Methods of mitigating OFRSSF are often limited, as catchment hydrological responses are largely determined by factors often at least partially beyond human modification. Factors determining the hydrological response of a catchment include precipitation amount, intensity and spatio-temporal distribution, alongside catchment geology, soils, topography, antecedent conditions and land-use. Out of all these influential factors, land-use is the most easily controlled, and therefore land-use modification could offer methods to control OFRSSF.

Controlling land-use is largely the only plausible (cost-effective) method of mitigating flood-risk in areas with a large number of dispersed settlements at risk of flooding (Forbes et al., 2016). This is because constructing and maintaining hard-engineered flood defences across multiple channels is often not economically viable (villages often do not qualify via cost-benefit approaches). Furthermore, land-use approaches allow distributed mitigation and can target specific locations susceptible for OFRSSF. It is also relatively easy to financially incentive certain land-uses, including specific practices conducted within such land-uses, facilitating widespread adoption.

1.4 Grassland and improved-pasture hydrology

Of the approximately 104 million km^2 of habitable land on Earth, 51 million km^2 is devoted to agriculture (Ritchie and Roser, 2019). Of this 51 million km^2, 40 million km^2 (77 % of agricultural land, >38 % of total habitable land) is devoted to rearing livestock for meat and dairy production (Ritchie and Roser, 2019). In many regions of the world, the majority of these livestock rearing areas are some variation of grassland, meaning grasslands encompass a considerable percentage of the Earth's land surface (Blair et al., 2014; Ritchie and Roser, 2019).

Given that they encompass such a large global area, grasslands are key for global food production, particularly in areas less favourable for alternative farming practices (Ritchie and Roser, 2019). Grasslands are resultantly often economically valuable, as well as socially, culturally and politically important (Lamarque et al., 2011; Bengtsson et al., 2019; Morse, 2019). Environmentally, it is well established that grasslands play a key role in earth system functioning, such as in ecology and the biosphere, regarding air-quality, carbon emissions and atmospheric science, and relating to soil functioning, nutrient-cycling and the pedosphere (Reaney et al., 2011; Blair et al., 2014; Bengtsson et al., 2019). Logically, therefore, improved-pastures are likely to play a considerable role within the hydrosphere.

Grasslands in the UK account for 60 % of the total agricultural area, particularly within Northern England, Wales, and sections of Scotland (DEFRA, 2019: See Section 3.1.2). These areas are typically managed to produce food for livestock, predominantly sheep and cows, with many of these areas additionally at high-risk of flooding and water-quality issues. When intensively managed these grasslands are often enclosed, with grassland that is grazed by livestock but is not cut for conservation fodder known as agriculturally-improved (or simply improved-, as well as permanent-) pasture, and grassland that is both cut and grazed known as hay/silage fields or meadows (although both are regularly termed (improved-)pastures). From here on, both pastures and meadows will simply be termed improved-pasture or permanent pasture, with reference to cutting practices stated when this is present.

Improved-pastures in the UK are predominantly found in valley bottoms and on lower valley sides where favourable growing conditions exist for grassland vegetation, although they also exist along hillslopes and on hilltops under suitable settings (Jerram and Backshall, 2001). Grassland that is not enclosed or intensively managed for agriculture but is still grazed by livestock is often termed unimproved pasture or (semi-)natural grassland, as well as alternative names e.g., heathlands, moor(land) etc. Semi-natural grasslands tend to be located in exposed, remote and less productive regions, and are therefore less economically viable for improved-pasture establishment (Sansom, 1999).

Modern and historical agricultural management interventions taking place in grasslands have likely altered hydrological processes and pathways in numerous ways, and in some cases, significantly (Figure 1.1: Wheater et al., 2008). As a result,

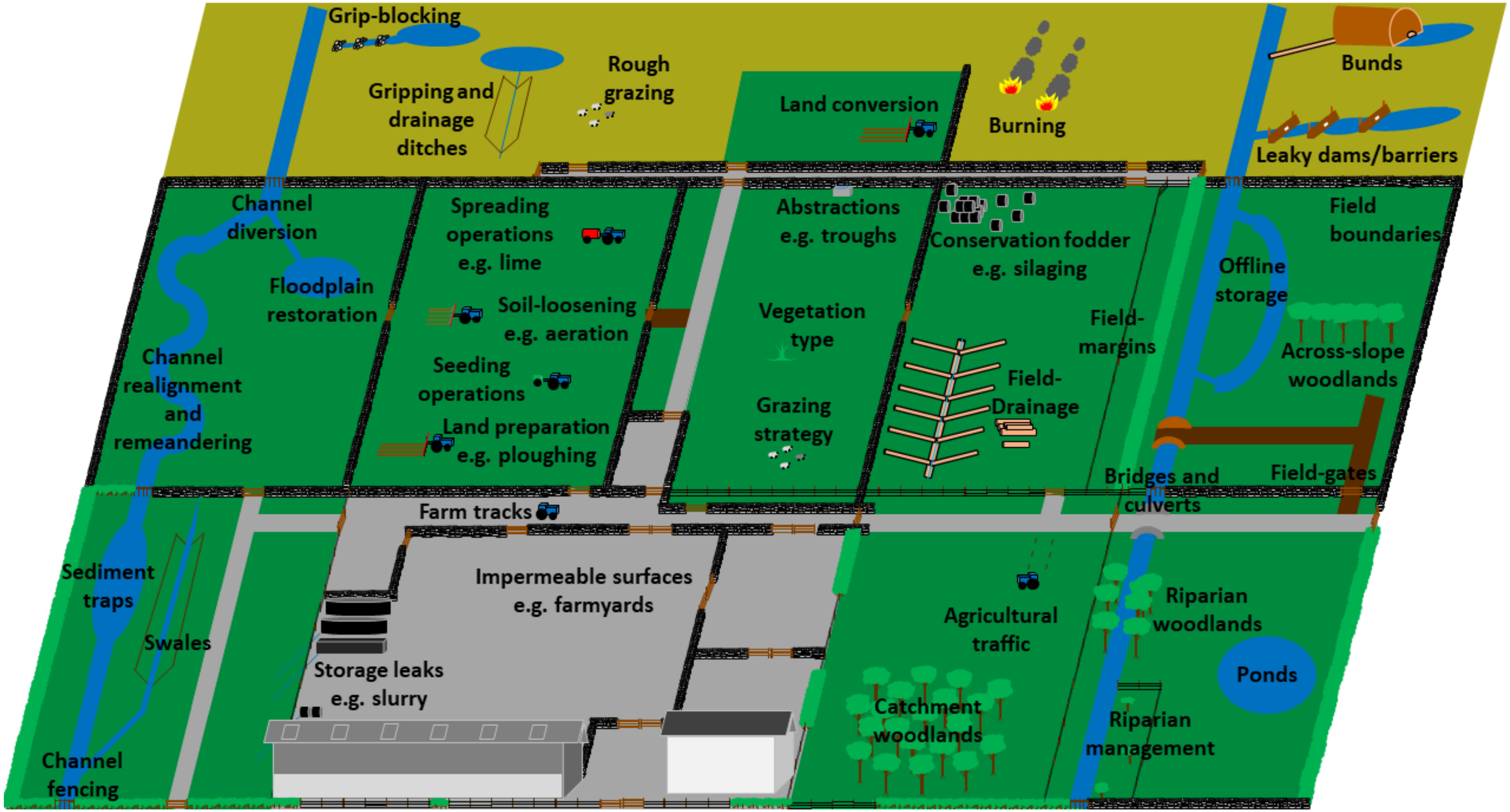

Figure 1.1: A schematic diagram outlining common pastoral farming practices and features (interventions) that likely alter the hydrological functioning of improved-pastures. Several widespread farming practices immediately upslope of the improved pastures such as on (semi-)natural grasslands have also been included.

management may greatly influence flood-risk, water-quality and drought-resilience for a region across a range of scales (Kay et al., 2009). In areas with large percentages of grassland, these management practices are likely to be one of, if not the, most dominant control on hydrological functioning and rainfall-runoff processes. Consequentially, incorporating grasslands within land-use management is potentially a feasible approach to managing flood-risk, water-quality and drought-resilience. Using grasslands for hydrological management is therefore an adaption of soft-engineered flood-risk management approaches such as Natural Flood-Risk Management.

Given that improved-pastures are intensively managed almost continually, there is great scope to use them within a hydrological framework. This may provide hydrological (and perhaps additional) ecosystem services, whilst the land remains highly productive. The following section provides a brief outline of common grassland management practices and features (hereby-termed interventions) in respect to improved-pastures occurring in the UK, and how these may influence hydrological processes and pathways. Most of these grassland interventions exist throughout the globe, particularly within temperate regions, albeit in modified forms. Essentially all interventions currently present in UK grasslands are adopted in the interest of maximising income for farms either through agricultural output or agri-environmental payments.

1.5 How has improved-pasture management altered key stormflow pathways?

Essentially all improved-pastures enclose land with the use of dry-stone walls, hedgerows, barbed wire and/or fences to define boundaries and to control the movement of livestock (Carey et al., 2008). Several of these boundary methods are very likely to have an influence on hydrological processes, as they may prevent, divert or accelerate the transfer and movement of water and solutes between sections of farmland, as well as influence interception and evapotranspiration, alongside soil hydraulic properties (Mérot and Bruneau, 1993; Herbst et al., 2006; 2007; Tayefi et al., 2007; Ghazavi et al., 2008; Yu and Lane, 2010; Ghazavi et al., 2011; Benhamou et al., 2013; Coates and Pattison, 2015; Holden et al., 2019). These grassland boundaries are also likely to influence water-quality through the abscission of deciduous vegetation, and the biogeochemical weathering and general deterioration of boundary materials; with their influence also altering soil moisture regimes and therefore drought-resilience (Ghazavi et al., 2008; Albéric et al. 2009; Grimaldi et al., 2009; Ghazavi et al., 2011; Holden et al., 2019).

Enclosed improved-pastures are often ploughed to loosen the topsoil before first applying seeds, as well as to redistribute nutrients within the topsoil and to disturb unwanted weeds, thatch or crop residues. Ploughed land can then be harrowed or tilled (sometimes instead of ploughing) to allow the surface to be more receptive to seeding by disrupting soil clodding, sods and root mats as well as to further disrupt weed species and to flatten ant or mole hills (Jerram and Backshall, 2001). These land preparation interventions undoubtedly change the hydraulic, physico-chemical and biological properties of the soil (Estavillo et al., 2002), and therefore certainly influence flood-risk, water-quality and drought resilience at least on a small-scale.

Once suitably prepared, improved-pastures are (re)seeded with grass species to create highly productive swards to sustain livestock. The most common UK seeding species are perennial ryegrass (*Lolium perenne*), Italian ryegrass (*Lolium multiflorum*), and various hybrid ryegrasses, often supplemented with legumes to increase nitrogen fixation and by additional species to improve livestock nutrition (AHDB, 2015; Guy et al., 2018). Sometimes grasslands are also overseeded, where new seeds are added to existing improved-pasture, as well as cut to store fodder during winter (AHDB, 2015). The process of (re)seeding, overseeding and cutting, including the agricultural machinery deployed, likely changes surface roughness, soil hydraulic properties, evapotranspiration rates, as well as soil-chemistry/biology through changes in vegetation and compaction, ultimately influencing hydrological functioning (Macleod et al., 2013; AHDB 2015; 2016; Alaoui et al., 2018; Bond et al., 2020). Changes in vegetation species and cover also likely alters soil erosion, chemistry, microbiology and ecology, and therefore influences local water-quality, alongside changes water usage rates, and therefore drought-resilience (Macleod et al., 2013; Jung et al., 2020).

To maintain and increase the productivity of established improved-pastures, surface applicants are often added in the form of lime, slurry, farmyard manure, compound fertilisers and pesticides (Vogt et al., 2015; Holland et al., 2018; Walsh et al., 2018). Agricultural machinery such as aerators, sward-lifters and subsoilers are also often used to reduce compaction and improve productivity within certain grasslands (Davies et al., 1989; Harrison et al., 1994; Douglas et al., 1995; Drewry and Paton, 2000; Bhogal et al., 2011; Newell-Price et al., 2015). In wetter regions with slowly permeable soils and geologies, agricultural (field-)drainage is also often installed (Robinson and Beven, 1983; Robinson et al., 1985; Robinson, 1990; Heathwaite et al., 2006; Wheater et al., 2008). All of these interventions clearly affect soil physico-chemical properties and soil biology as evidenced by sward improvements, and therefore likely causes some changes to hydraulic functioning, and consequentially flood-risk, water-quality and drought-resilience.

Sheep alongside beef and dairy cattle are the dominant livestock in most improved-pastures in the UK. The grazing regime of livestock i.e., livestock type, stocking densities, grazing strategy etc. can all substantially influence hydrological processes in a region, as can grazing unimproved pastures (Heathwaite et al., 1989, 1990; Sansom, 1999; Drewry et al., 2000a; Meijles et al., 2006; Orr and Carling, 2006; Meijles et al., 2015). The trafficking of machinery across grasslands as part of grassland management also feasibly alters hydrological processes by changing soil hydraulic properties and functioning, ultimately influencing flood-risk, water-quality and drought-resilience (Dingman, 1994; Alaoui et al., 2018).

As evidenced, numerous pastoral farming interventions exist that clearly alter sward productivity, and by extension, soil physico-chemical and biological properties. These practices are consequently likely to influence hydrological processes, including OFRSSF (Table 1.1). Thus, these interventions likely alter hydraulic functioning, and by extension, flood-risk, drought-resilience and water-quality. Despite the widespread occurrence of such interventions both in the UK and overseas, relatively little scientific research has been conducted on how certain pasture management practices influences hydrological processes (either theoretical studies, modelling studies, field experiments, or laboratory experiments: Considerably restricting confidence in Table 1.1; Bhogal et al., 2011; Blackwell et al., 2018). Several pastoral interventions that are fairly common throughout the globe have seldom been hydrologically studied, e.g., dry-stone wall boundaries. Dedicated grassland research institutes such as the Institute of Grassland and Environmental Research (IGER) tend to prioritise agronomy and animal-rearing science over hydrological science (although hydrological and interdisciplinary studies of grasslands have been conducted here; with research

Table 1.1: A list of common pastoral farming interventions present in improved-pastures and on immediately adjacent semi-natural grasslands (see Figure 1.1), and if they theoretically alter hydrological parameters strongly associated with overland flow and rapid shallow-subsurface flow (OFRSSF) likelihood. Interventions are either very likely (Y), possibly (?) or unlikely (-) to alter each hydrological parameter. The overall impact on OFRSSF, and the confidence in this estimate, is given as high (H), medium (M) or low (L).

Hydrological parameter / **Intervention**	**Infiltration and permeability**	**Soil wetness**	**Overland flow velocity**	**Surface roughness**	**Interception**	**Evapo-transpiration**	**OFRSSF water-quality**	**Theoretical impact on OFRSSF**	**Confidence**
Abstractions	-	-	-	-	-	-	-	L	L
Across-slope woodlands	Y	Y	Y	Y	Y	Y	?	H	L
Agricultural traffic	Y	-	?	?	-	-	-	M	M
Bridges and culverts	-	-	-	-	-	-	-	L	L
Bunds	-	Y	-	-	-	-	-	L	L
Burning	Y	?	Y	Y	Y	Y	Y	H	M
Catchment woodlands	Y	Y	Y	Y	Y	Y	?	H	M
Channel diversion	-	-	-	-	-	-	-	L	L
Channel fencing	-	-	-	-	-	-	-	L	L
Channel realignment and remeandering	-	?	-	-	-	-	-	L	L

Table 1.1 (continued):

Hydrological parameter / Intervention	Infiltration and permeability	Soil wetness	Overland flow velocity	Surface roughness	Interception	Evapo-transpiration	OFRSSF water-quality	Theoretical impact on OFRSSF	Confidence
Conservation fodder (cutting)	Y	-	?	Y	?	?	?	M	L
Farm tracks	Y	Y	Y	Y	-	?	?	H	M
Field-boundaries	?	?	Y	Y	Y	?	?	M	L
Field-drainage	-	Y	-	-	-	-	-	M	L
Field-gates	Y	?	?	?	-	-	?	M	L
Field-margins	?	?	Y	Y	?	?	?	M	L
Floodplain Restoration	-	Y	-	-	-	-	?	L	L
Grazing strategy	Y	Y	Y	Y	Y	Y	Y	H	M
Grip-blocking	-	Y	-	-	-	-	-	L	L
Gripping and drainage ditches	-	Y	-	-	-	-	-	L	L
Impermeable surfaces	Y	Y	Y	Y	-	Y	?	H	L
Land conversion	Y	?	?	Y	?	?	?	H	M
Land preparation	Y	?	?	Y	?	?	?	M	M
Leaky dams/barriers	-	Y	-	-	-	-	-	L	L

Table 1.1 (continued):

Hydrological parameter / Intervention	Infiltration and permeability	Soil wetness	Overland flow velocity	Surface roughness	Interception	Evapo-transpiration	OFRSSF water-quality	Theoretical impact on OFRSSF	Confidence
Offline storage	-	Y	-	-	-	Y	-	L	L
Ponds	-	Y	-	-	-	Y	-	L	L
Riparian management	Y	Y	Y	Y	Y	Y	?	H	L
Riparian woodlands	Y	Y	Y	Y	Y	Y	?	H	L
Rough grazing	Y	Y	Y	Y	Y	Y	Y	H	M
Sediment traps	-	-	-	-	-	-	-	L	L
Seeding operations	Y	-	?	?	-	-	-	M	L
Soil-loosening	?	?	?	?	?	?	?	M	L
Spreading operations	Y	Y	?	?	-	-	Y	M	L
Storage leaks	-	Y	-	-	-	-	Y	L	L
Swales	-	Y	Y	?	-	-	?	M	L
Vegetation type	Y	Y	Y	Y	Y	Y	Y	H	M

focuses shifting; Blackwell et al., 2018). Indeed, the effects of land-use changes on hydrological response is understudied globally (Rogger et al., 2017).

This therefore underlines a comparatively understudied area of hydrology, specifically a subset of ‘grassland hydrology’, hereby referred to as ‘pasture hydrology’ (this term has been used previously by other authors e.g., Davies-Colley, 1997; Bilotta et al., 2007; Southwell et al., 2008; Pilon et al., 2017; Kussainova et al., 2020, amongst others, but to the author’s knowledge, has never been clearly defined in detail as done here). Pasture hydrology is therefore defined as the hydrological science in relation to grasslands (specifically pastures), including the interlinkages with practices and features directly present within such environments and landscapes. Increased understanding of this scientific discipline will compliment more well-studied areas of land-use hydrology such as ‘forest hydrology’ and ‘arid hydrology’, as well as broader hydrological and environmental science, in addition to agronomy and related scientific fields.

Given the threat posed by hydrological hazards in the UK and indeed internationally, understanding how improved-pasture interventions alter hydrological processes and pathways could allow more informed land-use decisions to be made, and therefore offer a distributed yet heavily managed form of land-use management. This new and intermediary approach to land-use management may hold substantial benefits to communities affected by hydrological hazards, as well as to landowners/farmers who supervise interventions through justification of advanced subsidies (Hayhow et al., 2019; Morse, 2019). Investigating improved-pasture interventions that maintain land productivity in terms of food production, whilst improving flood-risk, drought-resilience and water-quality, and perhaps provide additional ecosystem services (e.g., to air-quality, ecology, cultural heritage etc.) is therefore a much-needed area of research.

1.6 The need for both data and models

In order to quantify changes to hydrological processes caused by grassland interventions with scientific rigour, there is a need for detailed experimental evidence i.e., observations, of how interventions behave hydrologically whilst undisturbed within the landscape (Beven, 2019; Beven et al., 2020). These observations are necessary for any scientific analysis and therefore interpretation of hydrological processes to be credible, especially given that hydrological processes are extremely spatio-temporally dynamic and scale-dependent, and the governing processes on this variability are still not (and may never) be fully understood (Freeze, 1975; Dagan, 1997; Beven, 2000; Blöschl et al., 2007; Merz et al., 2009; 2011; Pattison and Lane, 2011; Rogger et al., 2017; Beven, 2019). This is particularly important if conclusions from the study are to be used within hydrological models and land-use management that operates beyond the observational sites or catchments (grasslands may cover ~40 % of the Earth's land surface), and if uncertainties are to be adequately acknowledged (Beven and Binley, 1992; Gelhar, 1993; Beven, 2000).

To achieve this, intensive fieldwork, which utilises the most hydrometrically accurate instruments is needed to derive measurements of field hydraulic properties (see Section 2.3). Samples taken during observations also require accurate laboratory (soil and water bio-physio-chemical properties) and taxonomic (vegetation) techniques to further analyse data, to contextualise findings, and to further improve the understanding of changes to hydrological processes (see Sections 2.3-2.5). To analyse observations in detail, advanced physics-based and statistical models are required to ensure observations and conclusions have sound scientific underpinnings (see Sections 2.6-2.11). Finally, to interpret observations hydrologically, observations must be assessed for how the pasture interventions have altered hydrological processes, specifically in relation to OFRSSF, and therefore be incorporated within a form of physically-based hydrological model (see Sections 2.6-2.11). This combined approach (see Chapter 2) is essential to scientifically determining and quantifying how pastoral interventions have altered hydrological processes that control OFRSSF, and consequentially how they influence flood-risk, and to a lesser extent water-quality and drought-resilience.

This thesis investigates in detail four widespread grassland (improved-pasture) interventions within the Leith, Lowther and Petteril sub-catchments of the Eden catchment in Cumbria, North-West England, with each intervention the focus of each individual chapter. Each chapter involved conducting intensive fieldwork to collect detailed observations of the hydrological properties of the intervention. Field-samples of supporting physico-chemical and biological properties of soil and vegetation were also collected for laboratory and taxonomic analysis to improve holistic interpretations. Observations of each intervention underwent detailed statistical analysis to quantify the changes to hydrological processes, as well as to justify conclusions. Finally, and most importantly, individual interventions were interpreted hydrologically with support from all of the above methods, in order to investigate and quantify how each intervention alters hydrological processes surrounding OFRSSF generation. All monitored interventions exist in some form across the UK and Europe, and often further afield, and were specifically selected due to their widespread presence.

1.7 Aims and objectives

1.7.1 Aim

The fundamental aim of the research is to *quantify* the magnitude of observable hydrological change for a variety of pastoral farming features and practices (interventions), each delivering regulatory watershed services (flood-risk, water-quality and drought-resilience benefits), within three neighbouring Cumbrian basins from local intervention scales to larger scales. Each chapter compares an intervention against 'typically-managed' neighbouring improved-pasture, and must not inhibit farm profitability. Each intervention must be adoptable into the majority of Cumbrian farming systems without inhibiting productivity (and ideally into UK and international farming systems), and provide ecosystem services beyond purely hydrological to further justify increased agri-environmental payments. *This will address a significant gap in experimental evidence, scientific understanding and hydrological modelling which explores the potential impact of farmland interventions within a grassland-rich landscape*, and demonstrate links between academic disciplines of scientific hydrology, meteorology, soil science, and agricultural sciences, and will be supported by advanced statistical and physics-based modelling techniques.

1.7.2 Objectives

Objective 1) Quantify the effect of a grazed semi-natural grassland as opposed to a grazed agriculturally-intensive improved-pasture and silage field on spatio-temporal soil volumetric wetness (θ_v), from extremes of drought to fully saturated conditions. Soil volumetric wetness will be correlated to local topography and local vegetation to assess their influence. Soil volumetric wetness will be used to infer the sources and amounts of saturation-excess overland flow during flood events, as well as the extent of drying during drought events. – CHAPTER 4.

Objective 2) Quantify the effect of blade aeration within several agriculturally-intensive improved-pasture and silage fields on temporal saturated hydraulic conductivity (K_{sat}) and related soil penetration resistance (PR) of the topsoil. Saturated hydraulic conductivity will be used to investigate changes in the amount of infiltration-excess overland flow during extreme precipitation events generated from aerated and unaerated improved-pasture, by contrasting a high-frequency precipitation time-series with K_{sat} values. – CHAPTER 5.

Objective 3) Quantify the effect of hedgerows/hedge-margins within agriculturally-intensive improved-pastures in controlling changes in the amount of overland flow and rapid shallow-subsurface flow during floods and extreme precipitation events via direct measurement. This will be supported by causal factors of saturated hydraulic conductivity (K_{sat}) and soil volumetric wetness (θ_v). Overland flow will be analysed for targeted co-benefits to water-quality (total sediment (TS), nitrate (NO_3^-), nitrate-nitrite ($NO_3^-NO_2^-$), soluble reactive phosphorus (SRP), particulate total phosphorus (PTP), dissolved total phosphorus (DTP), and dissolved organic carbon (DOC)). – CHAPTER 6.

Objective 4) Quantify the effect of dry-stone wall boundaries within sloped areas of agriculturally-intensive improved-pastures upon changes to soil volumetric wetness (θ_v) above and below the barrier, and therefore infer changes to overland flow and rapid shallow-subsurface flow likelihood by assessing the impedance of θ_v transfer downslope. – CHAPTER 7.

Objective 5) Using statistical and physics-based modelling techniques, create physically-based models with interpretable parameters to project measurements to demonstrate the value of existing interventions on overland flow and rapid shallow-subsurface flow production from plot-scales (and possibly larger scales). – CHAPTERS 4-7.

Objective 6) Share findings of the study with the farming community in the Leith, Lowther and Petteril basins, and wider UK stakeholders (both academic and practitioners). – CHAPTERS 4-8.

2 METHODOLOGY

2.1 Introduction to methods

This chapter is aimed at providing a background and justification to the methodologies and experimental designs used within Chapters 4-7. The chapter should be taken as a basic overview rather than a comprehensive review of methods, and is split into four major themes. The first theme highlights the specific question(s) that each research chapter and objective of the thesis aims to address, and provides a short justification of the methods used to address these questions (Section 2.2). The second theme deals with field and laboratory techniques in relation to hydrometry, soil science, and water-quality, with reference to literature therein for further information (Sections 2.3-2.5).

The third theme deals with the analysis and interpretation of collected data (Sections 2.6-2.11). This theme provides a broad overview of hydrological modelling approaches and techniques, and the main associated uncertainties, and attempts to justify the Data-Based Mechanistic modelling approach used throughout the thesis (Sections 2.7-2.8). The background to a range of statistical analytical and modelling techniques used within Chapters 4-7 is also provided (Sections 2.7-2.11). As before, reference to further literature is provided throughout.

The fourth and final theme of the methodology (Sections 2.12-2.13) deals with the precise experimental design and data analysis conducted within Chapters 4-7, and links the prior sections together (Section 2.12). A conclusory section and brief summary (Section 2.13) then pairs the experimental designs with the objectives and lays out the approaches taken in each research chapter to satisfy the respective objectives. Throughout the chapter, abbreviations have been kept to a minimum to maximise clarity.

2.2 Methods required to satisfy each objective

The specific rationales/research question(s) for each research chapter objective (Objectives 1-4/Chapters 4-7) is as follows:

Objective 1 – Chapter 4) How has converting 'natural' grassland ecosystems into 'agriculturally-intensive' grassland ecosystems altered their hydrological regime, specifically in relation to surface moisture patterns? This question has implications for the generation of overland flow and rapid shallow-subsurface flow (particularly saturation-excess overland flow), and resultantly flood-risk and water-quality, but also will divulge information regarding their resilience to drought. Understanding these 'baseline' conditions will also help to understand if modern agricultural practices have modified the hydrological responses of grasslands. It would additionally be logical to acknowledge the spatial arrangement of the hydrological regime, and the potential local control(s) on this. To investigate this question, a (semi-)natural grassland and an agriculturally-intensive grassland that are immediately adjacent and were identical prior to land conversion need to be compared to reduce natural variability between plots (which may influence hydrological functioning), as well as for logistical reasons.

Objective 2 – Chapter 5) Given that improved-pastures cover such a wide range of areas and are extremely heterogeneous, it would be very useful for an intervention to be capable of being deployed in problematic subareas that are highly vulnerable to overland flow and rapid shallow-subsurface flow. Soil loosening devices such as aerators, sward-lifters and subsoilers are all interventions that can be deployed uniformly across an improved-pasture or can target problematic subsections if needed. Furthermore, this equipment is freely available from the Eden Rivers Trust for local farmers. Blade aerators are particularly well suited to upland UK improved-pastures as they can operate in dense superficial geology, do not pose risks to (often unmapped and century-old) drainage systems, and can be pulled by smaller engined vehicles. Quantifying the effectiveness of blade aeration would be very useful in relation to improvements to infiltration and permeability, which can then be linked to infiltration-excess overland flow likelihood when paired with a precipitation time-series. As before, measurements of aerated and unaerated improved-pasture need to be adjacent to reduce natural variability.

Objectives 3 – Chapter 6) Almost invariably, improved-pastures are enclosed by boundary features such as hedgerows, dry-stone walls, fences, barbed wire etc. Given the sheer number of improved-pastures, there is clearly an extremely large number of boundary features. Hedgerows (and hedge-margins) are extremely common boundary features both in the UK and internationally, and therefore they are a suitable intervention for investigation. These features are additionally present within securely enclosed improved-pasture; therefore providing added security and privacy to protect expensive and sensitive equipment should overland flow be directly recorded here. Furthermore, hedgerows likely alter soil (hydro)chemistry, and therefore overland flow water-quality could be incorporated within this chapter. It would therefore be useful to directly monitor overland flow (volume and water-quality) from a hedgerow(-margin), as well as the controls on overland flow such as permeability and soil volumetric wetness. Results from the hedgerow(-margin) should be directly compared to an immediately adjacent improved-pasture to understand ‘baseline’ conditions, but also to reduce natural variability in soils and topography.

Objective 4 – Chapter 7) Similarly to hedgerows, dry-stone wall boundaries are extremely common across the UK and internationally. These features extend below the soil surface to some depth and often sit on top of the subsoil. Many of these features are well over a century old and are relatively stationary in comparison to hedgerows (they do not grow, get cut, need laying, die etc.), and many are likely to remain well into the future. It would therefore be useful to know if these features prevent or slow the transfer of water between improved-pastures, and if they influence soil volumetric wetness. Determining if dry-stone walls influence soil volumetric wetness will resultantly improve the understanding of overland flow and rapid shallow-subsurface flow pathways in improved-pastures. To answer this research question, soil volumetric wetness above and below dry-stone wall boundaries that are parallel with hillslope contours needs to be measured. This information can be used to understand the spatial extent of change in soil volumetric wetness. It will be key that the improved-pastures above and below the boundary are as similar as possible in terms of soil, management practices, vegetation etc.

There are also general considerations to be made for the research objectives:

- Directly measuring overland flow and rapid shallow-subsurface flow for every chapter is well beyond the financial constraints of this project. That being said, at least one chapter should include direct measurements to confirm that overland flow and/or rapid shallow-subsurface flow is indeed present in the studied sub-catchments. Given that such measurements will use a large percentage of the project resources, it will be best to deploy measurement equipment next to a feature (rather than in association with a practice) to ensure that the intervention is applied and the resources are not wasted. It is also best to select a site where overland flow has been observed or measurements suggest that overland flow is likely present to ensure resources are used effectively.
- Given that overland flow is rare, relatively long time-series of overland flow observations are presumably going to be needed to capture enough data to build

suitable models. It is best if overland flow plots are installed fairly quickly for this reason. It is also important to build redundancy into the system as overland flow events may only occur during extreme conditions. Extreme conditions are when equipment is most likely to fail, malfunction or be damaged; therefore, resources should be spread between more than one plot and more than one rain-gauge for this reason.

- It is only a small extension to include water-quality once the infrastructure for overland flow has been installed. Therefore, only interventions that likely alter soil (hydro-)chemistry, and by extension, overland flow water-quality, will be considered for overland flow infrastructure. It will additionally be best to prioritise water-quality parameters of interest to the Eden Rivers Trust, such as nitrates, phosphates, sediment etc.

To achieve all of these objectives, clearly a wide range of hydrometric measurements are needed (see Section 2.3). Understanding baseline soil conditions as well as changes to these would also be beneficial to the hydrological interpretation of each intervention, as well as to help categorise findings (Section 2.4). Given that water-quality is an additional important aspect of overland flow and rapid shallow-subsurface flow, water-quality measurements should also be included (Section 2.5). To interpret results (both at and beyond the plot-scale), hydrological and statistical modelling techniques are needed (Sections 2.6-2.11). To specifically satisfy the research questions for objectives 1 and 4 regarding the spatial arrangement of surface moisture patterns, geostatistical approaches are needed (Section 2.10). To understand the localised controls on surface moisture patterns in objective 1, a linear mixed-effect regression approach is an appropriate option (Section 2.11). To link the permeability and infiltration measurements with infiltration-excess overland flow likelihood in objective 2, a peaks-over-thresholds analysis within the branch of extreme value theory is very well suited (Section 2.9). To satisfy hydrological time-series observations of overland flow, a transfer function modelling approach is a suitable analysis option (Section 2.8).

2.3 Hydrometry

Hydrometry is the monitoring of the components of the hydrological cycle, and resultantly was the main method of sample collection. As the primary objective of the research is to directly observe and quantify improvements caused by grassland interventions, largely in relation to overland and rapid shallow-subsurface flow, hydrometry of the topsoil and supporting hydrological components such as river flow (discharge) and precipitation were monitored. Deeper sub-surface hydrology and hydrogeology are largely avoided, as these pathways are believed to have less influence upon stormflow or surface hydrological pathways, alongside being considerably more difficult and expensive to observe.

2.3.1 Overland flow and rapid shallow-subsurface flows

A range of methods exist for monitoring and measuring OFRSSF. Crest stage recorders are sheltered receptacles buried within the topsoil. These are useful in determining both the incidence and spatial-extent of overland flow (and shallow-subsurface flow if buried deep enough), and are very low cost and require minimal labour (Shaw et al., 2011). Crest stage recorders however cannot capture flow dynamics, are subject to large errors (e.g., overtopping, unknown source areas, soil smearing during installation), and are easily disturbed or damaged. Having non-sealed storage containers also limits subsequent water-quality analysis. Within intensively-managed agricultural catchments, crest stage recorders have very limited application.

Embedded plots/networks are the most robust method of measuring OFRSSF within agricultural catchments. Embedded plots directly capture OFRSSF via a (usually steel) ‘Gerlach’ collection-trough (see Gerlach, 1967). This approach has the advantage that steel troughs are extremely durable, can be buried within the soil and hammered into the soil profile (below root-mats for example), and their peripheries can be relatively easily sealed (preventing leakage). The collection-trough can be connected to a pipe network and route flow through flumes or large tipping-bucket devices, which can accurately capture flow dynamics if combined with dataloggers (Shaw et al., 2011). Pipe networks can be buried for added protection, and can route flow to other locations where sensitive monitoring equipment can be safely and securely stored, as well as into storage for water-quality analysis.

Embedded networks additionally benefit by enclosing and defining a source area with impermeable materials such as steel or plastic sheeting, which can close the water-balance and allow the quantification of runoff coefficients. Having a defined source-area can also reduce the likelihood of the system being overwhelmed and can allow a more informed allocation of resources (Shaw et al., 2011). Lastly, embedded networks can largely be left in the field and require fairly minimal maintenance once installed.

Downsides of using embedded networks are that they are often two or more orders of magnitude more expensive than using crest stage recorders. Installing embedded networks (including piping and drainage ditches) are extremely labour intensive, and in some situations cannot use machinery, as the site may be inaccessible or easily disturbed. The nature of embedded plots and a well-defined source area can also limit the applicability of such plots, as they may not be suitable for travelling through boundaries e.g., hedgerows, in areas with high levels of agricultural traffic, in very uneven terrain, or in highly erodible soils.

Overland flow and shallow-subsurface flows in this thesis have been directly measured using embedded plots, with specific details of plot installation given in Chapter 6. Overland flow and shallow-subsurface flows were directly captured and analysed between an agriculturally-intensive improved-pasture and a hedgerow wild-margin in Chapter 6. The difficulty in installing the embedded plot and drainage ditch directly within the hedgerow, as well as the risk of damaging the hedgerow, was the reason a wild-margin was used rather than the hedgerow itself. Chapter 4 contains visual confirmation of overland flow presence, and measures soil volumetric wetness (and degree of saturation) in an attempt to predict the likelihood of saturation-excess overland flow between a semi-natural grassland and an agriculturally-intensive improved-pasture. Chapter 7 similarly measures soil volumetric wetness to predict the likelihood of saturation-excess overland flow and rapid shallow sub-surface flow above and below the dry-stone wall boundaries. Chapter 5 contains a large number of permeability values, which are linked to infiltration-excess overland flow likelihood by comparing observed permeability with observed precipitation intensities between aerated and unaerated improved-pasture/silage fields.

2.3.2 Soil volumetric wetness

Soil moisture can be expressed as the mass of water within a mass of dry soil. This can be given as Equation 2.1:

$$\theta_m = \frac{m_w}{m_s} \tag{2.1}$$

Where m_w is the mass of water (g), m_s is the mass of dry soil (g), and θ_m (g g^{-1}) is the mass wetness. Soil moisture is offset by natural variations in the mass of the soil however i.e., the dry bulk-density of the soil (g cm^{-3}: See Section 2.4.2). Consequentially, the soil volumetric wetness (θ_v) is the most commonly expressed measurement of soil moisture content (Dingman, 1994; Shaw et al., 2011). Soil volumetric wetness accounts for variations in m_s by multiplying m_w with the soil dry bulk-density. Given that 1 cm^3 of water is 1 g, θ_v is therefore equivalent to the volume of water (V_w) within an undisturbed volume of soil (V_s). Soil volumetric wetness is expressed in Equation 2.2:

$$\theta_v = \frac{m_w \, m_s}{m_s \, V_s} = \frac{V_w}{V_s} \tag{2.2}$$

Soil volumetric wetness was measured using time-domain reflectometry (TDR) with simplified sensors (sometimes termed simplified time-domain reflectometry or sTDR). With sTDR, moisture-probes are buried in the soil-surface and emit a continuous 100 MHz outgoing electromagnetic signal down a wave-guide and a reflection is generated at the base of the wave-guide. A composite standing wave is produced by the interaction of the outgoing and reflecting waves. The ratio of the outgoing wave and the composite standing wave is dependent on the dielectric constant of the wave-guides, which is given by Equation 2.3:

$$\varepsilon = \left(\frac{ct}{L}\right)^2 \tag{2.3}$$

Where ε is the dielectric constant (dimensionless), c is the speed of light (3×10^8 m s^{-1}), t is the travel time (s), and L is the length of the wave-guide (m). The dielectric constant is equivalent to the relative permittivity of the material surrounding the wave-guides, which is largely determined by θ_v surrounding the wave-guides (Gaskin and Miller, 1996). Advantages of sTDR over higher-powered TDR is that the equipment is cheaper, more robust, more mobile, and provides more rapid measurements, although the uncertainty of measurements is slightly increased (Gaskin and Miller, 1996; Shaw et al., 2011). Certain site conditions can also amplify the uncertainty within measurements, such as differing soil organic matter contents or soil textures.

Knowledge of the θ_v distribution is very useful as it can determine SOF likelihood, especially if soil porosities are known (Western and Grayson, 2000: See Section 2.4.4). The spatial-structure of θ_v is also crucial, as it can highlight contributory area connectivity (Grayson and Blöschl, 2000; Zehe and Blöschl, 2004). Soil volumetric wetness can also be useful in determining plant stresses and agronomic output during drought, as well as wildfire frequency (Albertson et al., 2009; Schulte et al., 2012). Furthermore, θ_v can heavily influence soil biology, as well as soil and soil water chemistry. As a result, θ_v is clearly a key variable in the hydrological cycle, and has a strong influence upon hydrological functioning (Anderson and Burt, 1990).

Soil volumetric wetness measurements were taken with an ML3 'theta' moisture-probe (Delta-T Devices Ltd.), with details given in Chapters 4, 6 and 7 (see Whalley, 1993; Gaskin and Miller, 1996; and Miller et al., 1997; for further details). Soil volumetric wetness measurements were taken extensively in Chapter 4, in order to quantify θ_v patterns between a semi-natural grassland and an agriculturally improved-pasture. Similarly, in Chapter 7, θ_v was measured extensively above and below multiple dry-stone walls. In Chapter 6, θ_v was taken during the artificial-rainfall experiment to quantify plot saturation and to infer possible hydrological (saturation) pathways and overland flow mechanisms. Chapter 5 did not include θ_v, although this is discussed further in Chapters 5 and 8.

2.3.3 Permeability

Permeability is the rate at which a fluid can pass through a porous media. The saturated hydraulic conductivity (K_{sat}) is the rate of water percolation through fully saturated material (soil or regolith) under unit cross-sectional area and unit hydraulic gradient, and is synonymous with the coefficient of permeability. Darcy's Law (Darcy, 1856; Equation 2.4.1) demonstrates how subsurface flow in saturated ground is directly proportional to the hydraulic gradient (dH/L):

$$Q_{subsurface} = k_{sat} A \frac{dH}{L} \tag{2.4.1}$$

Where $Q_{subsurface}$ (m s^{-1}) is the volumetric discharge of subsurface water, A (m^2) is the cross-sectional area through which the water is flowing, L (m) is the length of the flow path, and dH (m) is the change in hydraulic head between each end of the flow path (Shaw et al., 2011). Algebraically rearranged, the same equation can define the saturated hydraulic conductivity, giving Equation 2.4.2:

$$k_{sat} = \frac{Q_{subsurface}\, L}{A\, dH} \tag{2.4.2}$$

Permeability is highly variable both spatially and temporally, and can vary by several orders of magnitude even within the same geological formation (Chappell and Ternan, 1992; Dingman, 1994; Dagan, 1997). Permeability is the parameter that the majority of physically-based distributed hydrological models are the most sensitive to, and therefore is a key component within hydrology (Shaw et al., 2011). Permeability was chosen as a key parameter within several chapters as permeability is directly quantifiable between different land-uses (intervention versus non-intervention), and has a significant impact upon both overland-land flow likelihood, shallow-subsurface flows, and hydrological modelling.

Measurements of K_{sat} were conducted via a Talsma ring permeameter. The Talsma ring permeameter is a core-based permeametry method that makes use of a constant head permeameter device. Detailed operation and error analysis of the Talsma ring permeameter is given in Chappell and Ternan (1997).

Talsma ring permeametry was the selected permeametry method for several reasons. Ring permeametry can take in-field measurements, saving transportation time and preventing disturbances such as drying which occurs during transport (a serious issue on the organic-rich/swelling soils of Cumbria – see Section 3.1.5). In-field measurements also benefit from being repeatable, with any suspect results flagged and tests repeated. The ring permeametry method utilises a large soil core (7000 cm^3: 11x larger than standard laboratory cores), minimising sampling and extraction disturbances (Chappell and Ternan, 1997). The large core allows the edges of the core to be more easily sealed, which is crucial for the accurate measurement of clay-rich soils (Chappell and Lancaster, 2007). Core permeametry additionally reduces smearing that auger-based and borehole-based permeameter methods suffer from, which are especially problematic within clay-rich soil (Chappell and Ternan, 1997). It is generally given that with proper operation, the Talsma ring permeameter is the most accurate permeametry method currently available.

Several negatives of using Talsma ring permeametry exist however. The largest issue with the Talsma ring permeameter is that measurements are very physically intensive. Each measurement requires the use of a sledgehammer and the lifting of large, fully-saturated soil cores. The ring permeameter also requires a relatively large amount of supplementary field-equipment that often needs to be manually transported. In-field measurements additionally require large volumes of locally-sourced water, with 40 litres of water not uncommon to fully saturate and conduct a single measurement on drier, highly permeable soils. This volume of water not only adds to the physical effort of sampling, but can restrict the location of sampling due to water resource limitations, particularly during times of drought.

Three notable additional downsides of using the Talsma ring permeameter also exist (many other core permeametry methods also suffer from these limitations). The first is that the large soil core requires a large volume of soil suitable for sampling. Although large soil volumes/cores are rarely problematic directly within improved-pasture, this can be problematic when sampling near/within grassland features such as hedgerows or woodlands with shallow roots (particularly common in coniferous forests), as it is often problematic due to the difficulty in finding a sampling site that does not contain thick tree roots (considerably difficult to hammer through and often causes core collapse if conducting *ex-situ* measurements). Secondly, ring permeametry is unsuitable within confined spaces due to the use of the sledgehammer, for example, taking measurements directly within a hedgerow. Lastly, core permeametry takes considerably longer to conduct measurements than more rapid techniques, such as auger-based methods, although the increased accuracy of measurements is often preferable to an increased number of samples.

Permeability samples were collected and analysed extensively in Chapter 5 in order to quantify changes to the saturated hydraulic conductivity caused by the blade aerator, as well as to quantify changes to infiltration-excess overland flow likelihood. Chapter 6 also conducted a considerable number of permeability samples to aid in the interpretation of results. The difficulty of conducting permeability measurements directly within the hedgerow was a contributing factor for focusing upon the hedgerow wild-margin rather than the hedgerow itself within Chapter 6. Chapter 4 did not conduct permeability measurements due to time constraints and the difficulty in acquiring local water resources, especially during the dry conditions. Chapter 7 did not involve permeability measurements, as permeability was assumed identical above and below the dry-stone walls as per the experimental design (see Section 2.12.4).

2.3.4 River discharge

River discharge is the volume of water flowing within a river channel, and is conventionally measured in cubic metres per second (m^3 s^{-1}), where 1 m^3 is equivalent to 1000 litres. The term river discharge or river flow is synonymous with stream discharge or streamflow, and is primarily a subjective reference to the size of the channel. River discharge is useful in defining a catchment's response to rainfall (rainfall-runoff modelling), as well as being used as a proxy for the level of integrated catchment saturation, assuming the majority of flow routes through the channel. River discharge that exceeds channel capacity causes over-bank flow (fluvial/riverine flooding) and is a major concern across much of the UK and indeed internationally. Very low river discharges can also concentrate pollutants and negatively affect biodiversity, as well as limit water abstractions and the use of waterways.

River discharge can be monitored through a variety of methods (see Herschy, 2009; Shaw et al., 2011). River discharge for the larger waterbodies of the studied catchments (see Chapter 3) such as the River Lowther was provided by the Environment Agency and was calculated via the velocity-area method. The velocity-area method is calculated by splitting the cross-section of the channel into sub-sections, and measuring the velocity and area of each sub-section. The area and mean velocity of each sub-section is then multiplied together to get the discharge for each sub-section. A summation of the sub-section discharges therefore gives the total discharge of the river or stream. The velocity-area method is expressed as Equation 2.5:

$$Q_{river} = \sum_{i=1}^{n} \bar{v}_i a_i \tag{2.5}$$

Where Q_{river} is river discharge ($m^3\ s^{-1}$), v_i is the mean of the velocity measurements for each sub-section ($m\ s^{-1}$), a is the area of each sub-section (m^2), i is each individual sub-section, and n is the total number of sub-sections.

River discharge of smaller streams such as Bessy Gill and Back Greenriggs was directly calculated by installing structures, specifically FRPB style flumes. These greatly improve the estimation of stream discharge by having a rigid, highly durable structure of standardized shape and characteristics. The basic hydraulic mechanism applied to gauging via structural methods (which includes weirs) is by causing critical flow within the structure. Critical flow is a type of energy criterion. For a certain discharge, the energy of the flow is a function of its depth and velocity, which is characterized by the Froude number (Equation 2.6.1):

$$Fr = \frac{v}{\sqrt{(d\ g)}} \tag{2.6.1}$$

Where Fr is the Froude number (dimensionless), v is the flow velocity ($m\ s^{-1}$), d is the flow depth (m), and g is gravitational acceleration ($m\ s^{-2}$). Froude values <1 have flow which is subcritical (slow and laminar), and Froude values >1 have flow which is supercritical (fast and turbulent). When the Froude value is 1, flow is said to be critical. At the point of critical flow for a given discharge, there is a unique relationship between velocity and the discharge (Equation 2.6.2):

$$v = \sqrt{(d\ g)} \tag{2.6.2}$$

Therefore, only depth is needed to accurately determine discharge when critical flow is occurring. Critical flow is created within structures through either a constriction of the width of the channel (flume), or the depth of the channel (weir).

FRPB flumes were chosen to monitor stream discharge as they offer high-frequency, highly-accurate measurements of streamflow that can be conducted remotely and recorded via a data-logger and a telemetry system. Flumes tend to be very resilient against fine-sediment, larger boulders and biofilm/vegetation growth, and require fairly minimal maintenance if situated in suitable locations. Flumes also require less-specific installation conditions than weirs, and can often be installed without machinery. Flumes tend to be good at calculating low flows, and have correction equations (with considerably reduced accuracy) if overtopped, which is an issue for extreme flood magnitude events (usually avoided as flumes usually have a specific catchment area range – although this range can be complicated by permeable geologies, which may cause underflow). FRPB flumes are suited to catchments up to 1 km^2, and therefore were deemed suitable for the given streams (especially given the permeable geologies – see Section 3.1.6). Flumes also do not tend to inhibit fish migration or general stream biodiversity.

Downsides of monitoring discharge via structural methods includes being financially expensive in terms of equipment and materials, and requiring a large degree of time and physical effort to install. A telemetry system is also needed for each flume, which adds to the network expense, although these can often be paired with meteorological equipment or water-quality monitoring systems. Flumes also have a clear upper limit (overtopping) and lose a considerable degree of accuracy above this point, and therefore may not be suitable for the most extreme of flood-events. Flumes can also encourage over-bank flow by consuming space within the channel, and therefore should be constructed in locations with this in mind (i.e., not immediately adjacent to housing). Complex geologies can also cause underflow i.e., flow in the porous space immediately beneath a stream which is not recorded by the flume, as well as can complicate calculations (flumes/weirs work off the assumption that flow in leaving the catchment via the channel).

Three flumes were installed to monitor stream discharge, all within small streams within the Leith catchment. Two flumes were installed within Back Greenriggs coming from Trantrams woods and the Back Greenriggs hamlet, and a single flume within Bessy Gill next to Bessy Gill Wood (see later Chapters 3 and 6). Originally, all three flumes were planned to be used as part of a collaborative channel re-alignment research paper as part of a collaboration with NERC grant NE/R004722/1, but due to delays in modifying the channel, flumes were instead used to monitor discharge and infer catchment integrated saturation for Chapter 6. Each flume is attached to a telemetry system, and therefore also provides meteorological information, and these provided precipitation time-series for Chapter 6.

For Chapter 4, daily discharge at all time steps from the River Lowther (gauged at Eamont Bridge) was calculated and provided by the Environmental Agency using a rating curve, which is established via the velocity-area method using a cableway (NRFA, 2021). Daily discharge was used to infer catchment saturation via an antecedent precipitation index. River discharge was not included within Chapter 5 as only infiltration-excess overland flow, rather than streamflow, was considered in the chapter. Chapter 7 also did not include river discharge as only extremely localised soil volumetric wetness was of interest.

2.3.5 Precipitation

Precipitation, predominantly rainfall, is the driving force behind essentially all hydrological processes operating in the UK. Rainfall as a function of the landscape is responsible for riverine (fluvial) flooding across the UK, but flooding directly from rainfall (pluvial flooding) is fairly common in urban (and sometimes rural) regions. All research chapters of the thesis used precipitation in some manner.

Precipitation throughout the studied catchments was collected via automated tipping-bucket rain-gauges. Automated tipping-buckets work by collecting rainfall over a known area, and funnelling this rainfall into a bucket of fixed volume. Once a critical volume within the bucket has been reached, the bucket tips and drains, and a twin-adjoining bucket takes its place. At each tip, a magnet attached to the connecting pivot closes an electrical circuit and the ensuing electrical pulse is recorded by a datalogger. Data can then be stored by the datalogger within/linked to the rain-gauge, and either downloaded at a later data or emailed via a telemetry system. Most tipping-bucket rain-gauges are enclosed for protection, and contain drainage holes below each tipping bucket to drain precipitation.

Tipping-bucket rain-gauges have the notable advantage over storage gauges in that they can capture rainfall dynamics. Tipping-bucket rain-gauges additionally self-empty and can be fitted with screens to reduce the intrusion of debris or insects within the buckets or mechanisms, meaning they often require minimal supervision or maintenance. Tipping-bucket rain-gauges contain fewer moving parts and a lower energy demand than tipping-syphon gauges, further reducing maintenance. These advantages have meant that tipping-bucket rain-gauges are widely deployed throughout many regions of the world, second only to storage gauges, and were suitable for providing information for Chapters 4-6.

Disadvantages of the tipping-bucket rain-gauge are that the tipping mechanism can freeze in cold weather, which can be a problem in upper, exposed areas during winter. Tipping-bucket rain-gauges also contain notable uncertainties regarding precipitation dynamics when snowfall occurs due to the melting of the snow (although other rain-gauges also have this issue), which is again an issue in more upland, exposed regions. Uncertainties in rainfall dynamics also exist with high rainfall intensities and at high rainfall resolutions (Habib et al., 2001). Components within the rain-gauge can jam/fail, and external cables of the rain-gauge can also be damaged or become detached, and without telemetry, this problem is often undetectable until a download is complete. The data from tipping-bucket rain-gauges is also subject to errors and uncertainties due to the effects of wind, wetting, evaporation and splashing (Sevruk, 1996; Fankhauser, 1998; Nešpor and Sevruk, 1999; Habib et al., 2001). There can also be non-conformance of the bucket-size with the constant calibration volume required by the manufacturer, and the calibration of the device can often be time-consuming and complex (Humphrey et al., 1997; Habib et al., 2001). Tipping-buckets are also more expensive than standard storage gauges, although capturing precipitation dynamics is usually worth this additional expense.

A Kalyx tipping-bucket rain-gauge was installed as part of Chapter 6 within the lower improved-pasture overland flow plot. This tipping-bucket rain-gauge was used to infer immediate rainfall conditions affecting the overland flow plots, especially to capture any convective events as these can generate very localised precipitation extremes, and therefore may cause localised IOF. The Kalyx tipping-bucket rain-gauge developed an electrical fault due to a loose cable during Storm Ciara (8^{th} – 9^{th} February 2020), possibly as a result of excessive wind or falling debris, and therefore rainfall data from this rain-gauge was largely excluded for quality-control, and has been included in the supplementary materials in Chapter 6 (see later Supplemental Figure 6.S1).

Within Chapter 4, Gaugemap provided precipitation information from the Shap weather station (Wet Sleddale) for the whole of 2018 at a daily resolution. This data was key in producing an Antecedent Precipitation Index (API) to infer the level of integrated catchment saturation for the Lowther catchment. For Chapter 5, NERC grant NE/R004722/1 provided rainfall from Skelton rain-gauge for 1990-2018 (excluding July 1993-March 1997) at a 15-minute resolution, as part of a collaboration between projects. This data was utilized to predict infiltration-excess overland flow likelihood by overlaying this with surface saturated hydraulic conductivity for aerated and unaerated treatments. Within Chapter 6, NE/R004722/1 provided rainfall from the Back Greenriggs flume rain-gauge at a 5-minute resolution as part of a collaboration between projects. This data was key in providing the input to the transfer function model, given that the overland flow plot rain-gauge failed mid-storm, as well as in producing an API for the site. Chapter 7 did not include precipitation information due to only visiting each site a single time.

2.4 Hydropedology

Soil type, structure, biology, and physico-chemical properties are substantial controls on the hydraulic functioning of soils, and thus influence OFRSSF, and hence, flood-risk and water-quality, as well as drought-resilience to some degree. As a result, soil samples were collected extensively for Chapters 4-6. These samples underwent laboratory analysis for the determination of certain physico-chemical soil properties, which were deemed the most influential in the generation of OFRSSF. Supporting soil types and surficial and bedrock geological information were collected from a variety of sources for supplementary site information. Further information about catchment soils is given in Section 3.1.5, surficial geologies in Section 3.1.6.1, and bedrock geologies in Section 3.1.6.2. Chapter 7 did not involve soil analysis as per the experimental design (see Section 2.12.4).

2.4.1 Particle size distribution

Soil particle size distribution is a method to determine the inorganic mineral proportion of the soil according to particle size, which is also known as the texture of a soil. Soil texture is important as it can strongly influence soil and water chemistry, and provides some very basic/generalised information surrounding hydraulic conductivities and soil saturation rates (Vogt et al., 2015). Soil texture is therefore very useful if paired with additional soil bio-physico-chemical and hydrological properties as it helps to contextualise findings.

Laser diffraction with a Beckman Coulter LS-13-320 was used to determine soil particle size distributions, with specific details for each chapter given in Chapters 4-6. In certain chapters, soil samples underwent hydrogen peroxide treatment or loss-on-ignition tests before laser diffraction if cementing agent content (i.e., organic matter) was determined to be sufficiently high to skew laser diffraction readings. Sodium polymetaphosphate and manual aggregate breaking was also deployed to further separate aggregates to minimise errors associated with laser diffraction. Alternative methods to determine soil particle size distributions include wet sieving, dry sieving, the pipette method, the hydrometer method, and hand texturing (Sarkar and Halder, 2005). All of these alternate methods are slower, and have less precision than laser diffraction, and were thus avoided.

2.4.2 Soil dry bulk-density

Soil dry bulk-density (ρ_b) is defined as the mass weight of a unit volume of dry soil (Equation 2.7):

$$\rho_b = \frac{W_s}{V_s} \tag{2.7}$$

Where ρ_b is the dry bulk-density of the soil (g cm^{-3}), W_s is the weight of the soil (g), and V_s is the volume of the soil (cm^3).

The soil volume therefore includes both soil particles and pore/void space (Dingman, 1994; Davie, 2008). Soil ρ_b is useful in determining the level of compactness for soil structure, and is therefore useful in calculating the pore space of soils, which may influence hydrological functioning, as well as soil biota, chemistry and root penetration (Beven and Germann, 1982; Sarkar and Halder, 2005; Vogt et al., 2015). Soil ρ_b values typically contain high levels of both spatial and temporal variability due to natural variations within the soil. Soil ρ_b is heavily influenced by land management, soil texture, and vegetation, as well as by stone-content (notable in areas of glacial drift) and loss of organic material through volatilization (most notable in organic-rich soil), although the impact of the latter three can be minimised through proper technique and procedures. Generally, clay-rich soils tend to have lower ρ_b in comparison to coarse textures soils due to the loose packing of the clay particles (Sarkar and Halder, 2005).

Soil dry bulk-density was determined for Chapters 4-6 by collecting topsoil in a 221 cm^3 bulk-density metal sampling tin, which was hammered into the soil surface. The cylindrical sampling tin was then extracted from the topsoil by digging around the tin with a spade. Excess soil that overhung from the tin was then cut away, and any vegetation removed, with an effort made to minimize any soil lost. Samples were then transported back to the laboratory, where soils were sieved to 2000 microns and any stones/gravels removed, and any visible vegetation further removed. Soils were then oven-dried at 105 °C for 24 hours to remove any water and then weighed to determine ρ_b. Higher temperatures were not used so as to reduce the volatilization of any organic material contained within the sample, which is likely an issue in organic-rich soils.

2.4.3 Soil organic matter content

The percentage of organic matter within a soil sample is defined as the weight of material containing carbon-based compounds, divided by the total weight of the soil sample. Soil organic matter content can be defined by Equation 2.8:

$$SOM = \frac{W_o}{W_s} \tag{2.8}$$

Where Wo is the weight of the organic material (g), Ws is the weight of the soil (g), and SOM is the organic matter content (g g^{-1}).

The organic component of mineral soils has a great effect on soil properties, including physical, biological and chemical functioning, and thus, can significantly influence hydrological functioning. Soil organic matter (SOM) can increase the water holding capacity of soils, as well as support the creation of stable aggregates which can increase soil (macro-)porosity and hence, aeration and water movement (Chaney and Swift, 1984; Chandler and Chappell, 2008). Soil organic matter can additionally darken soil and decreases its albedo, leading to higher evaporation losses and hence a drier moisture state (Vogt et al., 2015). High levels of SOM may additionally suggest that there are abundant food supplies for soil biota.

Soil organic matter content was determined via the loss-on-ignition method for Chapters 4-6. Determining soil organic matter content follows the exact same principles as ρ_b, only that once ρ_b is determined, the soil is then placed into a furnace and further heated to 550 °C for 6 hours to volatilize any organic material. The soil is then reweighed and this determines what percentage of the sample consisted of organic rather than mineral components. The loss of material is termed organic matter rather than organic carbon, as carbon plus volatile elements associated with carbon (hydrogen, nitrogen, oxygen, sulphur) will also have been lost (Vogt et al., 2015).

2.4.4 Soil porosity

The porosity (η) of the soil is the undisturbed unit volume of porous space within a soil (volume of air and water) divided by the undisturbed unit volume of the soil (Dingman, 1994), defined by Equation 2.9:

$$\eta = \frac{V_a + V_w}{V_s} = \frac{V_p}{V_s} \tag{2.9}$$

Where η is the porosity (g g^{-1} cm^3 cm^{-3}), V_a is the volume of air (g cm^3), V_w is the volume of water (g cm^3), V_p is the volume of pores (g cm^3), and V_s is the volume of the soil (g cm^3).

The η of a soil is determined by the orientation, compaction and structure of the solid soil particles, and therefore is often partially correlated with ρ_b. Porosity substantially influences the hydrological functioning of soils as it determines the maximum degree of saturation within a soil. When a soil is fully saturated (i.e., all pore-space is filled by water, $V_a + V_w = V_w$), θ_v will be equal to η. The size and connectivity of both pores and macropores can substantially influence topsoil permeability and drainage rates, which can have an impact upon the soil volumetric wetness (Beven and Germann, 1982). Porosity is also very influential upon soil biota and soil (hydro-)chemistry.

Porosity was determined once samples were returned to the laboratory by gradually saturating each soil core whilst still in the bulk-density sampling tin with deionized and deaired water over 48 hours in an attempt to allow all pore-spaces to fill with water. Porosity was then determined with the soil moisture-probe by averaging a large number of measurements of the soil core. Soil remained in the sampling tins to avoid changing the structure, and therefore the η of the soil, therefore biasing readings. Porosity was calculated for Chapters 4-6, and was useful in determining the maximum θ_v before saturation-excess overland flow would be expected to occur.

2.4.5 Soil penetration resistance

Soil penetration resistance (PR) is the physical resistance of a soil to penetration by a standardized, steel cone, measured in kilonewtons (kN), and is a useful indicator of compaction. Soil PR is very useful in determining the mechanical impedance towards root growth, and therefore is primarily used in agronomy and soil science to highlight potential restrictions upon agricultural output. The link between soil PR and hydrology is much less studied however, although soil PR likely influences hydrology in several ways: Firstly, soil with high PR values may lack a developed or deep root system, which may prevent water and contaminants being transferred to depth. This likely leads to a higher risk of OFRSSF and resultantly increases both flood-risk and water-quality issues. Secondly, limited root growth due to elevated PR likely reduces nutrient uptake and inhibits plant growth, thereby reducing evapotranspiration and interception because of reduced foliage and biomass, as well as allowing nutrients to concentrate within the soil. Third and lastly, high PR suggests more compacted soil, which often has reduced permeability and aeration, affecting bio-physico-chemical functioning as well as soil micro-fauna.

Soil PR was measured by an SC900 Field Scout soil compaction meter in Chapters 4-5. The device measures penetration resistance via a 12.8 mm diameter cone attached to an internal-load cell. Specific methods of measurements are given within each chapter. Chapters 6 did not involve PR measurements as the depth of measurements may have disturbed the overland flow plots.

2.4.6 Soil acidity

The hydrogen ion activity or pH of a soil is a key element in its hydrological and chemical functioning. Soil acidity can affect flood-risk as acidic soils tend to be sensitive to disturbance, and therefore less capable of supporting macropores in larger numbers (Chappell et al., 1996). Soil acidity is also very important for soil chemistry and hence water-quality by affecting the solubility and availability of many nutrients and pollutants (Vogt et al., 2015). Vegetation is also strongly affected by soil acidity, which consequentially has a large impact upon flood-risk, water-quality and drought-resilience, alongside agricultural output.

Soil acidity was determined using a Mettler Toledo benchtop pH meter (Mettler Toledo) in Chapters 4-6. Soil acidity determination involved placing ~10g of fresh soil into a beaker. For every gram of soil, 2.5 ml of distilled water was added to the beaker (~25 ml total) and the water was shaken for 30 minutes on an orbital shaker to disrupt soil aggregates. The pH at the soil water interface was then measured using the pH meter. Each pH measurement was taken several times, and the probe fully recalibrated with pH standards and temperature corrected after each measurement to ensure results were as accurate as possible.

2.5 Water-quality measurements

The primary interest of the thesis is changes to flood-risk caused by grassland interventions; however, water is also the primary transport mechanism for sediments and solutes to reach stream channels. As water-quality is strongly influenced by chemical and biological reactions occurring as water moves through a catchment; managing water-quality therefore requires a solid understanding of hydrological pathways by which water input becomes streamflow (Dingman, 1994). As detailed hydrometric and hydropedological measurements were collected, water-quality parameters of local and indeed national importance were included in Chapter 6 as this is only a small extension to the experimental design (see Section 2.12.3 and Chapter 6). This subsection provides a basic overview of investigated water-quality parameters and laboratory techniques used for their detection.

2.5.1 Nitrogenous compounds

Nitrogen (N) is a major plant nutrient, and is often applied through fertilisers to agricultural land to improve/maintain yields. Improved-pastures are regularly supplemented by legume plant species, particularly clover, due to their association with dinitrogen (N_2) fixing bacteria which increases soil nitrogen reserves (Guy et al., 2018). Other major sources of nitrogen in Cumbrian grasslands include spreading operations, manure and slurry storage, septic systems, wastewaters, decaying organic material, groundwaters, and atmospheric deposition. The conversion efficiency of nitrogen to saleable product is expected to be around 20 % in livestock-grazing systems, 3-4x lower than arable systems (Hatch et al., 2002).

During the breakdown of organic material or fertilisers containing inorganic nitrogen, large amounts of plant-available nitrogen such as nitrate (NO_3^-) is released

(Whitehead, 1990; Hatch et al., 2002). Plants and microbial organisms take up this readily available nitrogen for the synthesis of organic nitrogenous compounds. Any excess nitrate is prone to being lost from the surface layers of the soil. This loss is principally via solution as nitrate is highly soluble and does not remain fixed to organic material or clay particles due to its negative charge (Davie, 2008). Resultantly, NO_3^- is very susceptible to losses via leaching, field-drainage and OFRSSF (Hatch et al., 2002; El-Sadek, 2007). As nitrate is the stable, unreactive form of nitrogen in oxygenated systems, it tends to contaminate waters and soils for extended periods. Nitrate is especially problematic in groundwater due to a lack of nearby growing vegetation for nitrate uptake and largely aerobic condition, which limits denitrification.

Excess terrestrial nitrates can change heathlands to grasslands, as well as reduce species biodiversity (Southon et al., 2013). Certain nitrogenous compounds commonly associated with agriculture such as nitrite (NO_2^-) and ammonia (NH_3) are directly toxic to aquatic life (Davie, 2008). Nitrates that enter aquatic ecosystems additionally increase eutrophication, whereby there is excessive growth of cyanobacteria and other phytoplankton species; often negatively influencing biodiversity, amenity value and navigation, as well as increasing water purification and filtration costs (Dingman, 1994; Leinweber et al., 2002; Ritter, 2010; Bennion et al., 2012; Williams et al., 2015).

Concentrations of nitrate, nitrite and ammonia in drinking water is regulated by the Water Framework Directive (WFD). The primary health concern of high-levels of nitrate/nitrite is methaemoglobinemia (‘blue baby syndrome’), which can negatively affect pregnancy, young infants, or those with rare medical conditions by interfering with the oxygen transport of haemoglobin (WHO, 2011). Although debated, several studies suggest that nitrates/nitrites could possibly be carcinogenic to humans (Hatch et al., 2002; WHO, 2011).

The nitrogenous compounds nitrate and nitrate-nitrite ($NO_3^-NO_2^-$) were measured in OFRSSF water-samples for Chapter 6 via a SEAL AQ2 Discrete Analyzer (Seal-analytical). This automated analyser uses a robotic sampling arm and a stepper-motor driven syringe to aspirate, dispense and mix precise quantities of sample and a reagent into reaction wells (miniaturized test tubes). Samples and reagents are then incubated within the reaction wells until chemical reactions are complete and in steady-state. An aliquot of the mixture is transferred to an optical-quality glass cuvette and the absorbance is read. See Section 6.5.5. for further details regarding water-sample collection and storage.

2.5.2 Phosphoric compounds

Phosphorus (P) is another key element within biological systems and is crucial for crop production and the growth, productivity and fertility of livestock. Major sources of phosphorus in Cumbrian grasslands include spreading operations, decaying organic material, wastewaters, slurry and manure stores, dietary supplements for livestock, and geochemical weathering. Areas of intensive livestock production generally have large surpluses of phosphorus (Leinweber et al., 2002).

Inorganic phosphate (PO_4^{3-}) is the phosphoric form that is readily taken up by vegetation. This is either dihydrogen phosphate ($H_2PO_4^-$) under acidic conditions, or hydrogen phosphate (HPO_4^{2-}) under neutral and alkaline conditions (Leinweber et al., 2002). As soluble and reactive forms of phosphorus are the only forms available for plant uptake, these phosphoric compounds are often combined and termed soluble reactive phosphorus (SRP). Total phosphorus is also usually separated further into dissolved (dissolved total phosphorus: DTP), and particulate (particulate total phosphorus: PTP) fractions.

Inorganic phosphorus tends to be strongly bound to soil, and therefore tends to be more strongly linked with soil erosion rather than leaching. In neutral and acidic soils such as those in the Eden (see Section 3.1.5 and Chapters 4-7), the dynamics of inorganic phosphorus is largely dominated by phosphate sorption onto clay minerals and pedogenic iron and aluminium oxides (Leinweber et al., 2002). In grassland catchments, phosphorus tends to accumulate near the soil surface due to the permanent, often shallow tilled/untilled nature of improved-pastures alongside agricultural inputs and excretal returns (Haygarth et al., 1998). This makes phosphorus a particular concern for near-surface hydrological pathways such as OFRSSF, especially field-drainage (Grant et al., 1996; Haygarth et al., 1998; Dils and Heathwaite, 1999; Heathwaite et al., 2006).

Excess phosphorus can migrate into receptor waterbodies, often whilst still physically attached to soil particles. Phosphorus enrichment thereby facilitates the growth of undesirable algal blooms and a general deterioration in water-quality, and the consequential negative environmental, economic and health effects. Phosphorus can be a long-term issue in waterbodies, particularly lakes, as phosphorus-rich sediments can be buried and re-emerge following disturbance events.

Phosphoric compounds SRP, TDP and PDP were measured in Chapter 6 via the SEAL AQ2 Discrete Analyzer (see Section 2.5.1). It should be noted that SRP detects reactive phosphorus in soluble form rather than purely inorganic phosphorus (Haygarth and Sharpley, 2000). See Section 6.5.5. for further details regarding water-sample collection and storage.

2.5.3 Dissolved Organic Carbon

Dissolved organic carbon (DOC) is a general measurement of the organic material dissolved within water, and mostly originates from the decomposition and exudation of plant, soil microorganism or animal matter. Agricultural, industrial and domestic human activities can elevate the concentrations of DOC in natural waters (Ritson et al., 2014). Generally, DOC is predominantly made from complex, high molecular weight compounds collectively termed humic substances, with a small percentage of identifiable, low molecular weight compounds such as carbohydrates and amino acids (Evans, et al., 2005). Hydrological pathways can be a key transport mechanism for DOC.

The concentration of DOC strongly influences within-stream hydrochemistry (Evans, et al., 2005; Thacker et al., 2008). Stream DOC concentrations control the transport and fate of heavy metals and organic pollutants, initiates photochemical reactions, contributes towards ionic strength, and influences particulate and colloid chemistry. The concentration of DOC is particularly important within the treatment of water for domestic consumption and industrial supply, as elevated DOC dramatically increase water treatment costs (Ritson et al., 2014). When DOC-enriched water is chlorinated for disinfection, high formation rates of disinfection by-products such as organo-chlorine compounds (the most well-known being trihalomethane compounds) are formed which are known to be carcinogenic (Alarcón-Herrera et al., 1994). Potable water containing high levels of DOC can also cause unpleasant tastes, odours and colours, as well as indicate microbial contamination (Evans et al., 2005).

Dissolved organic carbon was determined in Chapter 6 by absorption of a sample by two wavelengths of electromagnetic radiation using a model 7315 spectrophotometer

(Jenway), as demonstrated in Tipping et al. (2009). Thacker et al. (2008) determined the specific absorbance and specific absorptivity for multiple surface waters in Northern England. The study highlighted a monotonic increase in the absorbance at the wavelength ratio 340 nm / 254 nm, and absorbance at 340 nm. This conclusion suggests that DOC can be determined from absorbance measurements alone. Using this method, differences between calculated and modelled values of DOC were observed to be < 2 % (Tipping et al., 2009). The level of nitrate within each sample was determined before DOC was calculated to confirm the method had a suitably small uncertainty band (see Tipping et al., 2009). See Section 6.5.5. for further details regarding water-sample collection and storage.

2.5.4 Total sediment

Total sediment (TS) in a water-quality sense is any material either directly held within suspension or that settles out of a liquid. Sediment entering Cumbrian stream networks are primarily mobilised by soil erosion through hydrological mechanisms, although other sources include direct livestock excretions into streams, decaying organic material, mineral extraction sites, geological erosion, aeolian scouring, landslips, and construction works.

Soil degradation often reduces the infiltration-capacity of the soil, increasing the likelihood of OFRSSF and therefore pathways for enhanced transport of turbid flow and a resultant loss of soil (Harrod and Theurer, 2002). Sediment can be mobilised by and entrained within OFRSSF relatively easily. Within overland flow, concentrated flow pathways such as rill flow and gully flow tend to efficiently increase soil erosion. Large amounts of sediment can also be carried within rapid shallow-subsurface flows such as within pipes and macropores, and through field-drains (Haygarth et al., 1998;

Harrod and Theurer, 2002). Sediment can also be mobilised via additional hydrological mechanisms e.g., stream bank erosion, deeper flow pathways.

Mobilised sediment poses a significant risk to the aquatic environment. Sediment, particularly clays and organic matter, tend to harbour chemically active materials and are important vectors for nutrients, pollutants and microbes (Harrod and Theurer, 2002; Vogt et al., 2015). Large quantities of sediment can entirely block or occupy considerable volumes within a channel, increasing flood-risk and reducing navigability (Barker et al., 2016). This sediment can make receiving water turbid, reducing aesthetic and amenity value. Sediment can bury terrestrial and aquatic vegetation as well as fish eggs, alongside absorbing and scattering light, negatively impacting biodiversity. Sediment additionally increases water treatment costs, particularly those involved with filtration and purification processes.

Total sediment was calculated in Chapter 6 by collecting a known volume of sample into a container and heating the container at 105 °C until all water was evaporated (see Chapter 6 for further details). This method removes water from the sample itself, as well as water trapped within sediments, following the sample principle as ρ_b determination (see Section 2.4.2). A temperature of 105 °C was chosen as this reduces the volatilization of organic material (Shaw et al., 2011). The weight of the residue was then recorded. Dividing the weight of the residue by the volume of the sample can therefore determine the TS concentration within a water-sample (Equation 2.10):

$$TS = \frac{W_{sed}}{V_{solution}} \tag{2.10}$$

Where TS is the total sediment concentration (g L^{-1}), W_{sed} is the weight of the sediment residue (g), and $V_{solution}$ is the volume of the solution (L). See Section 6.5.5. for further details regarding water-sample collection and storage.

2.5.5 Electrical conductivity

The electrical conductivity (specific conductance) is a measure of the ionic strength of an aqueous solution, and is therefore a measurement of the ability of an aqueous solution to conduct an electrical current. Electrical conductivity is dependent on the total concentration, mobility and valence of ions present, as well as the temperature of the solution (Ritter, 2010). Major positively charged ions that primarily determine electrical conductivity are sodium (Na^+), potassium (K^+), magnesium (Mg^{2+}) and calcium (Ca^{2+}). Major negatively charged ions that affect electrical conductivity are carbonate (CO_3^{2-}), bicarbonate (HCO_3^-), sulphate (SO_4^{2-}) and chloride (Cl^-). Nitrate and phosphate play a more minor role in determining electrical conductivity. The sources of the ions that affect electrical conductivity in the Eden are largely from geology, soils, agricultural activities, industrial wastes, septic tanks, and road salting.

Most inorganic compounds dissolved within a solution are relatively good conductors of an electrical current, and therefore electrical conductivity is useful in determining their presence. Electrical conductivity is therefore a useful method of estimating total dissolved ionic solids within a substance, which is a good initial and general indicator of pollution. Electrical conductivity does not highlight which specific pollutants are present in the solution however, which requires much more expensive and time-consuming chemistry, and was thus avoided.

Electrical conductivity was measured via a 340i electrical conductivity multimeter (Geotech Environmental Equipment) in Chapter 6. This conductivity multimeter was pre-calibrated using a single point calibration standard. The probe was placed in 50 ml of sample held within a glass beaker with a magnetic stirrer and read once the measurement was stabilized. Two electrodes at a fixed distance were placed in the glass beaker and an alternating electrical current is passed through the electrodes. The resistance is then measured by the conductivity meter and corrected to 25 °C.

2.6 Hydrological models and modelling techniques

There is a need to model within hydrology (and indeed all science) in order to demonstrate an understanding of the science and its underlying phenomena within a hydrological system (Kohler, 1969, cited in Beven, 2012; Beven, 2012). This is especially important within hydrology, as hydrometric measurements are both spatially and temporally very limited, particularly in regards to subsurface processes and states (Dingman, 1994). A hydrological model is therefore a necessary approximation and representation that allows the investigation of the properties, outputs and internal processes of the hydrological system, possibly with predictive or forecasting potential. Models are therefore necessary to further investigate certain hydrological processes, and to estimate how catchments will respond to changes such as land-use, or to more extreme events such those posed by natural-variability and/or climate-change. That being said, all hydrological, and indeed other models, are wrong and are known to be wrong, although some may serve a useful purpose (Box, 1976; Morton, 1993; Beven, 2012; See Section 2.6.9).

There are an extremely wide variety of hydrological models that serve various purposes. These range from parametrically and computationally efficient simple statistical models attempting to understand a single hydrological process at a very local scale at a single point in time; to extremely complex, state-of-the-art rainfall-runoff models attempting to incorporate all spatially-distributed hydrological (and sometimes additional) processes occurring within an entire catchment and extrapolating findings beyond observational records in both space and time. Each approach to hydrological modelling has associated strengths and weaknesses and is suited to different objectives, and there are various methodologies to groups these types of modelling approaches (most of which are not exclusive).

This thesis primarily deals with the former statistical approach to modelling in order to improve the physical interpretation of hydrological processes controlling OFRSSF likelihood at plot and field/hillslope scales. A brief overview of hydrological model classifications and uncertainties is given for improved contextualisation and for a justification of the methodological underpinnings of the thesis. This overview is in no way exhaustive, and a much more detailed review of hydrological modelling, particularly for rainfall-runoff purposes is provided in Beven (2012), and uncertainty estimation techniques in Beven and Binley (1992) and Beven (2009).

2.6.1 Black-box hydrological models

Black-box modelling within hydrology attempts to link an input (e.g., rainfall) with an output (e.g., discharge or overland flow). Black-box modelling operates within a purely analytical framework and therefore black-box models lack internal structure. Predominantly statistical and systems techniques are applied with this approach. Black-box modelling is also commonly known as induction, top-down, downward, metric, empirical, or data-based modelling (Young, 2002; 2003; Beven, 2012; Liu et al., 2016). A black-box model with a feasible physical interpretation of the model structure for the given locality has become known as Data-Based Mechanistic modelling (and is technically no longer a black-box model, but is closer to a grey-box model; see Section 2.6.3), and is the adopted modelling approach within the thesis, which is described in detail in Section 2.7.

Benefits of the black-box modelling approach are that models are purely based on observations with a priori assumptions about the data or model structure largely removed, and therefore modeller subjectivity is minimised. Black-box models are computationally and parametrically efficient (parsimonious), minimising computational requirements. Black-box models can also be used effectively without extensive knowledge or observations of the catchment under study, providing input and output data are available, and are applicable across very different catchments.

Drawbacks of black-box modelling include the lack of an internal structure potentially causing little-no physical interpretation of model parameters or structure. This lack of internal structure can lead to either none or very different (and sometimes conflicting) conclusions being drawn regarding the internal processes occurring within the modelled system. Both input and output data are also necessary to drive the model, and this data must also be of good quality to expect reasonable modelling outputs (so called Garbage-In Garbage-Out, GIGO, modelling; see Section 2.6.9). Modelling ungauged catchments or scenario assessment is often difficult with this approach.

Despite potential issues with physical interpretations, black-box models are often extremely functional and are widely and successfully implemented throughout hydrology (Dingman, 1994; Davie, 2008). They are particularly well suited to situations in which internal processes are not prized as a research objective. Regression models, antecedent precipitation/saturation index models, time-series models, frequency analysis models, artificial neural networks, neuro-fuzzy and fuzzy-logic type models, and the unit hydrograph, are all commonly used black-box modelling methodologies within hydrology (Davie, 2008; Beven, 2012; Xu et al., 2019).

2.6.2 White-box hydrological models

White-box models are the opposite of black-box models in that the internal structure of the model is entirely defined (mainly through theoretical approaches). As the priority of this modelling approach is in representing each individual hydrological process occurring within the catchment, they are commonly termed physics-based, process-based, hypothetico-deduction, theoretical, bottom-up or upward models (Young, 2003). This modelling is often seen as speculative simulation modelling which represents the current, state-of-the-art of the scientific understanding of the hydrological system (Young, 2018).

The main benefit of white-box modelling is that there is some physical basis for physical interpretation of model parameters and structure, and therefore increased potential for improvements to the understanding of specific hydrological processes (Ambroise et al., 1996). This modelling approach additionally supports scenario modelling e.g., different land-uses or meteorological conditions. White-box models are therefore well-suited make predictions in areas where data is absent e.g., ungauged basins. As it is heavily process orientated, white-box modelling can also become paired with other environmental processes, such as the Système Hydrologique Européen (SHE) model with the SHE TRAN variant for water-quality and SHE SED variant for sediment transport (Abbott et al., 1986a, 1986b; Wicks and Bathurst, 1996; Ewen et al., 2000).

Disadvantages of the white-box approach include that models often contain extremely complex structures with a large number of parameters, meaning they are regularly subject to overparameterization and overfitting, and often contain considerable degrees of subjectivity due to assumptions that derive from current hydrological paradigms (Young, 1998a; Perrin et al., 2001; Young, 2006; 2018). This complexity also often means they are extremely computationally intensive to run (Young, 2002; 2018). Furthermore, the physics within white-box models are also frequently disputed, which is amplified by the difficulty of conducting planned and confirmatory experiments within environmental systems (Garnier and Young, 2014; Semenova and Beven, 2015; Young, 2018). The complex internal structures can also mean that white-box models are highly specific to certain (sub-)catchments or extremely similar catchments, and is at least a contributing factor as to why so many physics-based hydrological models exist (Beven, 2012).

The first white-box hydrological model was theorised by Freeze and Harlan (1969), with early white-box models in practice including the SHE model (Abbott et al., 1986a, 1986b). Due to the extreme complexity and associated uncertainties with white-box modelling, grey-box and hybrid models are also widely used.

2.6.3 Grey-box and hybrid hydrological models

It is common for relatively few key processes to dominate within a white-box model so arguments exist regarding them being overly complex (Beven, 1989; Young, 2003). When refined to a smaller number of dominant hydrological processes, these are known as grey-box models (Liu et al., 2016). When grey-models are partially theoretical with some pre-determined structure, and partially data-based, with some parameters estimated from observational data through parameter calibration, they are often termed hybrid models, although the terminology is often interchangeable.

Benefits of grey-box/hybrid modelling is that models can retain the benefits of white-box models but can become substantially less computationally and parametrically intensive. Hybrid-models that use data to influence parameter calibration and model structure also benefit from using actual observations from the (sub-)catchment under study to ensure the model structure and parameters are at least operationally feasible (Chen et al., 2016; Huang and Bardossy, 2020). Furthermore, hybrid-models can often make relatively good use of Geographical Information System (GIS) datasets in regards to topography, soils, geology etc. which are often much more extensive than hydrological measurements (Ambroise et al., 1996).

Grey-box/hybrid models retain many white-box modelling issues however (see Sections 2.6.2 and 2.6.9), even if these are somewhat reduced. Additional negatives of using grey-box/hybrid-models are that there are often insufficient observations to support robust parameter optimisation or to confirm model structure. This is particularly an issue in relation to subsurface parameters or if parameters are optimised solely through catchment-integrated observed discharge. The parameter calibration process can be complex and can also cause the model to become highly specified to certain (sub-)catchments.

Grey-box and hybrid models have, to a degree, superseded white-box models. The most well-known grey-box and hybrid-models derive from TOPMODEL (Topographic Model) and its many variants which are based on very specific assumptions about the nature of the hydrological processes (e.g., Beven and Kirkby, 1979; Kirkby, 1997; Beven and Freer, 2001; Beven, 2012; Gao et al., 2015; Metcalfe et al., 2015). The Stanford Watershed Model is another early and well-known model of this type which is based on purely conceptual elements (Crawford and Lindsey, 1966; Beven, 2012).

2.6.4 Lumped hydrological models

Lumped models treat the catchment as a single unit with state variables that are averaged across the entire area of interest (usually a catchment or sub-catchment). They therefore contain no spatial discretisation and do not account for the response of individual regions or units within the modelled (sub-)catchment. As a result, most black-box models are lumped models.

Benefits of using lumped hydrological models are that they are parametrically and computationally efficient, do not require spatial data, are applicable across a range of catchments, and have reduced computational requirements. Resultantly, lumped models are considerably simpler, cheaper and faster to operate than many alternatives.

The main negative of lumped modelling is that the spatial heterogeneity is entirely lost within the system response. Resultantly, scenario-models such as land-use change are very basic and/or difficult with this modelling approach. The lack of spatial dimensions also means assessing hydrochemical or sediment transport from point sources can be limited. Lumped models are also often only able to predict discharge (or other modelling outputs) at a single outlet.

Despite the omission of spatial heterogeneity, lumped models can produce very good hydrological outputs. When spatial heterogeneity is not a primary investigatory objective, lumped models are often the preferred approach. Examples of lumped models include most Data-Based Mechanistic transfer function models (see Section 2.7), IHACRES (Identification of unit Hydrographs and Component flows from Rainfall, Evaporation and Streamflow data: Jakeman et al., 1990; Jakeman and Hornberger, 1993), the Soil Conservation Service Curve Number (SCS-CN) methodology (Mishra and Singh, 2003), and the unit hydrograph (Davie, 2008).

2.6.5 Distributed hydrological models

Distributed hydrological models (also known as fully-distributed hydrological models) are spatially distributed, with state variables that are locally averaged by discretising the catchment into grids (elements) and solving the state variables equations for each of these grids. Parameter values must be specified for each of these individual grids, and therefore distributed models can be seen as lumped models operating at the grid-scale (Beven, 1989). Distributed models are usually white-box, grey-box or hybrid models, although black-box and Data-Based Mechanistic varieties do exist (e.g., distributed transfer function models, also called runoff-runoff or runoff-routing models).

Distributed models benefit from being capable of modelling spatially explicit processes and are therefore very useful for land-use management and scenario modelling at large scales. Spatially determined processes are also very useful when modelling sediment or contaminant transport, particularly if point sources are of interest, or when including variable rainfall inputs. Additionally, distributed models can be paired relatively easily with GIS layers and topography, and can utilise local expert opinion (Chen, 2016).

Negatives of distributed modelling include that they are often extremely computationally and parametrically intensive, and are extremely data demanding as model parameters must be provided for every grid element in the flow domain and boundary and initial conditions must be specified for every length and area within the domain (Beven, 2012). The spatial resolution (grid or element size) is very important, as different resolutions can reveal quite different inferences in relation to the governing hydrological processes, and can produce different outputs with identical driving data. It is also impossible to physically measure all parameters used within a distributed model, which is amplified by the incommensurate nature of hydrological data (see Section 2.6.9). Furthermore, locally averaged parameters at the grid/element scale do not well reproduce system heterogeneities in system response (Beven, 1989). Many distributed models also retain lumped modelling elements, such as assuming uniform rainfall inputs.

Despite the additional complexity, distributed models often do not outperform simpler lumped or semi-distributed models (see Sections 2.6.4 and 2.6.6: Beven, 2012). Examples of distributed models include: the SHE model and its later variants (Abbott et al., 1986a; 1986b; Bathurst et al., 1995; Refsgaard and Storm, 1995; Refsgaard et al., 2010), the Institute of Hydrology Distributed Model (IHDM: Calver and Wood; 1995), and the TETIS model (Francés et al., 2007; Ocio et al., 2019).

2.6.6 Semi-distributed hydrological models

Semi-distributed models are distributed models that do not make calculations for every point in a catchment, but partially divide the (sub-)catchment into smaller units. These units then often use probability distribution models which produce a distribution function of characteristics to represent the spatially variability of runoff generation within the unit such as in TOPMODEL (Beven, 2012). The distribution function is an acknowledgement that not all of the catchment can be expected to respond in an identical fashion (Beven, 2012). Some models allow this distribution function to be mapped back into space e.g., TOPMODEL, although this is only a spatial approximation. Alternatively, semi-distributed models can use a kinematic wave-theory approach for each unit, such as in HEC-HMS (Hydrologic Engineering

Core – Hydrologic Modeling System). Semi-distributed models are often white-box, grey-box or hybrid-models.

Semi-distributed models allow spatially explicit processes to be investigated, but with a reduced number of specified parameters and lower computational requirements than distributed models. Many semi-distributed models can still include sediment or contaminant transport despite reduced discretisation (Page et al., 2006). Semi-distributed models can still often utilise external sources of information e.g., GIS.

Semi-distributed models tend to be more computationally intensive than lumped models however, and also have large data requirements for specifying parameters. The division of the catchment into smaller units can also limit spatial analysis. Predictions may also only be possible at specific points in the (sub-)catchment, depending on how the areal units were created.

Semi-distributed models are widely used in hydrology as they take benefits of both the lumped and distributed modelling approaches, whilst partially reducing some negatives of each approach. Examples of semi-distributed models include TOPMODEL, HEC-HMS, SWMM (Storm Water Management Model) and HBV (Hydrologiska Byrans Vattenavdelning: Bergström, 1976, 1992; Harlin and Kung, 1992).

2.6.7 Deterministic hydrological models

Deterministic hydrological models do not consider randomness in the inputs or outputs. Resultantly, these models will produce the same output from a single set of input data. Deterministic models can be used to estimate prediction uncertainties (e.g., via Monte Carlo methods, or with ensemble inputs in estimating the effects of future change).

2.6.8 Stochastic hydrological models

Stochastic hydrological models contain some randomness and therefore the outputs (and sometimes inputs) acknowledge a degree of uncertainty, and therefore a single set of input data can produce different outputs, often with a probability assigned to each outcome (Beven, 2012). Uncertainty in such models can come in the form of epistemic/system uncertainty which is a lack of knowledge of the system under study (Beven, 2012). Uncertainty can also come in the form of aleatory/statistical uncertainty which is uncertainty due to random variability which needs to be represented by a statistical model (Beven, 2012). Mounting evidence supports the inclusion of uncertainty within hydrological (and indeed many environmental) models (see Section 2.6.9), with the interest in stochastic modelling growing in recent decades.

2.6.9 Sources of uncertainties within hydrological modelling

The hydrological system is an admixture of complicated assemblages of interacting processes, consequentially, no hydrological model is a true reflection of the processes involved; even highly complex white-box distributed hydrological models are extreme simplifications of reality (Beven, 2012). Hydrological processes are largely nonlinear, nonstationary, spatio-temporally variable, complex, auto correlated, interconnected, partially depend upon antecedent conditions, not fully understood, contain threshold-enabled phenomena, and many are unobservable (meteorological driving data can also share these issues). Logically therefore, there is considerable uncertainty within any conceptual approximation of such a complex system, and therefore models should acknowledge both assumptions made about the underlying system, as well as any uncertainty associated with any modelling outputs. As a result, all hydrological models are intrinsically uncertain, but many still serve useful purposes. This subsection briefly outlines some of the major uncertainty issues associated with hydrological modelling to further justify the Data-Based Mechanistic approached used in the thesis.

2.6.9.1 Lack of data

In many hydrological modelling exercises and applications, there is insufficient data regarding hydrological processes operating within the area of interest (Irving et al., 2018; Huang and Bardossy, 2020). Many hydrological models (particularly white-box and distributed models) require detailed internal state observations for model calibration and validation, and therefore many hydrological models are heavily data-dependent (Grayson et al., 2002). Insufficient data on initial and boundary conditions is equally common (Blöschl et al., 2019b). Even with detailed catchment information such as geology, soil, or vegetation, these are usually insufficient to accurately estimate the parameters required to run a hydrological model. This is because catchments with seemingly similar geologies and soils (permeable, impermeable), antecedent conditions (saturated, unsaturated), land-uses (urban, rural), topographies (flat, steep), shapes (fan, elongated, composite), vegetation (vegetated, unvegetated), drainage density etc. can give very dissimilar hydrological responses to seemingly identical precipitation inputs (Beven, 2000; 2012). This is because of the spatio-temporal uniqueness of individual catchments and the unique spatio-temporal distribution of precipitation intensity within each individual precipitation event (Beven, 2000; Blöschl et al., 2019b).

Investment in intensive fieldwork campaigns or highly hydrometrically telemetered catchments can reduce but cannot overcome this lack of data. This is because: a) many hydrological processes are difficult to observable such as many subsurface hydrological pathways (e.g., Blöschl et al., 2019b), b) hydrological processes are extremely spatially and temporally variable at a wide-range of scales (e.g., Chappell and Ternan, 1992), c) the nature and interconnectedness of several hydrological processes are not fully understood (see Blöschl et al., 2019b), d) observations that do exist often contain relatively high levels of instrumental uncertainty (e.g., Chappell and Lancaster, 2007), and e) direct hydrological observations are usually incommensurate with model parameters due to scale (see Section 2.6.9.3). Parameter optimisation is a possible way of reducing the impact of this lack of observational data (Pokhrel et al., 2012; Chen, 2016; Chen et al., 2016); although this can give a range of plausible parameter sets/values (see Section 2.6.9.5).

Hydrology is therefore fundamentally limited as a science by both data availability and measurement techniques, as well as by the temporal length of records and their spatial coverage (Dingman, 1994; Beven, 2019; Huang and Bardossy, 2020). This issue is worsened by widespread violation in stationarity of existing hydrological (and meteorological) data due to natural (e.g., seasonality) and anthropogenic (e.g., greenhouse-gas emissions, land-use changes) trends and variability (e.g., Blöschl et al., 2017; 2019a; Faulkner et al., 2019; Hesarkazzazi et al., 2021). Furthermore, hydrological observations that do exist may not be directly related to the parameter(s) or variable(s) of interest within a model (Section 2.6.9.3).

2.6.9.2 Hydrometric data uncertainty

Hydrometric measurements (both inputs, outputs and state parameters/variables) can contain relatively large levels of uncertainty (instrumental uncertainty: Chappell and Lancaster, 2007). Given the data dependency of many hydrological models, if the uncertainty in measurements is especially large, this uncertainty can lead to GIGO modelling (Beven, 2012; McMillan et al., 2018). This is especially problematic when measuring hydrological extremes e.g., abnormally high or low flows, extreme rainfall etc. as this is often when instrumental uncertainty is maximised. It is therefore critical to evaluate the quality of hydrological observations before proceeding to base models or predictions upon them, including an acknowledgement of such uncertainties (Beven, 2012).

Hydrological data can be so highly uncertain that the information can be disinformative and therefore harm the operation of a model (Beven and Westerberg, 2011; Beven et al., 2011; Beven and Smith, 2014). For example, observations can record events with runoff coefficients >1, or streamflow responses in the absence of rainfall, both of which are due to deficiencies in data (often rainfall coverage) rather than the underlying model structure (Beven and Westerberg, 2011; Beven and Smith, 2014). Non-stationarity in hydrological data can also cause disinformative hydrological data (Beven and Westerberg, 2011). Beven and Westerberg (2011) argue that nearly all datasets used for rainfall-runoff modelling contain some periods of disinformative data. Distinguishing between usefully unusual and un-usefully unusual events can be considerably difficult (Beven and Smith, 2014).

2.6.9.3 Incommensurate data and scaling issues

Direct hydrological observations are often fairly rare point measurements in both space and time, operating at micro-, plot-, or occasionally field-scales. For example, the Talsma ring permeameter measures the saturated hydraulic conductivity over 7000 cm^3 at a single point in time. Even the most finely discretised models have elements operating at substantially larger scales than such measurements, and physics which represent small-scale processes do not necessarily represent processes operating at much larger scales.

The scale of observed hydrological parameters and these identical parameters within hydrological models are therefore incommensurable (they may have identical names but are very different entities). Resultantly, hydrological observations may have unclear relationships with model variables or parameters, and therefore the model parameters can be poorly defined without a clear interpretation (Young, 2018). It is therefore difficult for models to match internal state measurements and physics of the system to ensure that the model is predicting the correct response for the correct reasons (Ebel and Loague, 2006).

The problems of scale and incommensurate data causes issues with grid integration and interconnectedness, as well as problems with dimensionality in model calibration (Beven, 1989). Furthermore, there is often a wide distribution of observed hydrometric values within a single modelling grid/element. A considerable problem in distributed predictions is the use of a single parameter to represent this entire element (the lumped at the element-scale argument), although the use of a distribution of parameters has been shown to be more useful (Beven, 2000; 2012).

2.6.9.4 Model structural uncertainty

Model structural uncertainty is another limitation caused by the complexity and number of equations relative to the heterogeneous reality of the hydrological system, with even the most complex, highly parameterised model not capable of accurately simulating such behaviour (Klemeš, 1986). Attempts to capture and recreate all this complexity results in overparameterized models with poorly-defined parameters. This extreme system heterogeneity has contributed to the need for so many hydrological models for seemingly similar areas, and the difficulty in applying certain models to other areas (Beven, 2012).

Further to the issue of overparameterization, hydrological models may contain structural errors which are errors within the underlying physics or representation of such processes (Sherlock and Chappell, 2005; Semenova and Beven, 2015). These come in various forms such as: incorrect representation of processes; incorrectly representing characteristics of these processes; ignored processes (both knowingly and unknowingly); and implementation errors (time and space discretisation, numerical algorithms) that may make effective parameter values scale and numerical algorithm dependent in ways that are not clearly understood within dynamic and nonlinear systems (Beven, 2005; 2012; Beven and Young, 2013; Beven and Lamb, 2014). Assessing and correcting model structural errors is generally difficult due to a lack of observations upon the area of interest (Garnier and Young, 2014: See Sections 2.6.9.1-2.6.9.2).

Another major uncertainty within the model structure is that model parameters interact, an issue that is magnified exponentially by having a large number of parameters. Given that parameters can be ill-defined and incommensurate, it is therefore often not clear how parameters are interacting, and interactions themselves can become very complex. This parameter interaction can cause substantial issues with model calibration due to the numerous combinations of parameters that give feasible outputs (equifinality: Section 2.6.9.5).

2.6.9.5 Equifinality

The extremely complex nature of overparametrized hydrological models (predominantly white-box and distributed models) can lead to many different combinations of parameters and structures producing essentially identical hydrographs or other model outputs. This can lead to numerous seemingly plausible parameter sets satisfying the modelling objective, which is known as the equifinality concept (Beven, 1993; 2006). Model equifinality is worsened by identifiability problems caused by poorly-defined parameters, as this can often mean it is hard to discount certain parameter values or combinations. There is also sufficient interaction between the components of the system so that unless detailed characteristics of these components can be specified independently, many combinations or representations of these parameters can be acceptable (Beven and Freer, 2001).

As a result of equifinality, any model parameter set or model structural combination that predicts the variable(s) of interest must be considered equally likely as a simulator of the system (Beven, 1989). As many hydrological flow pathways are essentially unobservable and therefore unquantifiable, it is often difficult to discern which combination of parameters (if any) are both suitable and correct (Beven and Freer, 2001). The comparison of predicted and observed hydrographs is therefore an insufficient method to ensure that the model adequately simulates the true internal processes occurring within a catchment (Beven, 1989; Ebel and Loague, 2006). Resultantly, it is extremely difficult to find an optimal parameter set and model structure within the high dimensional parameter space of many distributed hydrological models (Beven and Binley, 1992; Beven and Freer, 2001; Beven and Binley, 2014). The higher the order of the model and the increased number of associated parameters tends to increase the difficulty in finding optimal parameter sets. One way of accounting for equifinality is to calculate the likelihood or uncertainty surrounding each parameter set (see Whitehead and Young, 1979; Beven and Binley, 1992; Parkinson and Young, 1998; Young, 1998a, 2006, Beven, 2012; Beven and Binley, 2014; Young, 2018).

2.7 Data-Based Mechanistic (DBM) modelling

The concept of Data-Based Mechanistic (DBM) modelling was first applied to hydrology in Young and Beck (1974) during an application of modelling water-quality in rivers, and later to river flows in Young (1974). The DBM term did not originate until its first usage in Young and Lees (1993) however. The DBM philosophy has been developed and refined over many decades and has been applied to areas as diverse as ecology, economics and environmental data, and has been extensively applied within hydrology (Young, 1974; 1993; Young and Beven, 1994; Young 1998a; 2011; Jones et al., 2014; Chappell et al., 2017). This section provides a basic overview of the DBM modelling approach, which is the modelling methodology used within the thesis, interested readers are advised to see the references therein for a more comprehensive overview.

The DBM approach is built on the assumption that wherever possible, the dynamic modelling of environmental systems should not be based solely upon theoretical simulation models, and that preconceptions should be kept to a minimum by allowing the data to define the model structure and parameters (Young, 2006). The first step of DBM modelling is data-based modelling and is identical to inductive black-box modelling approaches, whereby a model is fit to data using statistical analysis or systems techniques (Young, 2002; 2003). Theoretical preconceptions about the model complexity, form, structure or parameters should be minimised at this initial stage of analysis. This is in order to avoid prejudicial imposition of untested perceptions about the nature of the system in order to satisfy the modelling objectives. Wherever possible, model structure should be inferred directly and entirely from observational data (Young, 1998a; 2003).

Following this step, models undergo mechanistic interpretation, whereby models without physically meaningful and feasible interpretations of the internal structure in respect to the system under analysis and the modelling objective(s) are rejected (Young, 1998a; Young et al., 2004; Young, 2006; Young and Garnier, 2006). Considerations for scale are also essential at this point within the modelling philosophy (Young, 2003). This physical-interpretation is especially important given that multiple, very different models can fit the data equally well as defined by the equifinality concept, and is a method of refining plausible models (Beven, 2006; 2012: See Section 2.6.9.5). This physical interpretation differentiates DBM modelling from black-box modelling approaches and moves it closer to the grey-box or hybrid modelling categories. Because of this approach, DBM modelling is heavily dependent upon the quality of both input and output data (Young, 1998a; 2003; 2006).

There are many benefits of using the DBM approach for hydrological modelling, many of which involve reducing the significant uncertainties highlighted throughout Section 2.6. Firstly, DBM models are parametrically efficient (parsimonious), low-order models that require relatively little catchment information (Box and Jenkins, 1970; Beven, 2012; Young, 2018). This means that there is substantially reduced computational requirements when running a DBM model in comparison to white-box models, and DBM models can be used for real-time and adaptive forecasting (Young, 2002). Parsimonious models are also unlikely to become overly complex or overparameterized (Young, 2018).

Secondly, DBM models can extract the dominants modes of a system through Dominant Mode Analysis (DMA), which is a description of the core mechanisms that control the system’s behaviour (Young, 1998a; 2006). These dominant modes can often explain data extremely well with coefficients of determination exceeding 0.99, although this is unlikely when applied to rainfall-runoff modelling due to uncertainties in the data (Young, 1998a). This analysis can be used in conjunction with more physics-based modelling approaches to infer processes occurring within the area of interest, and therefore support model order reduction and decrease unnecessary complexity within hydrological models (Young, 1998a; 2006). Results and interpretations found from the DBM modelling can then be used to inform the complex simulation models which represent the current state-of-the-art of largely complex distributed hydrological simulation models (Young, 1998a).

Thirdly, DBM models are essentially Bayesian in concept and inherently stochastic, and therefore are appreciative of the uncertainties associated with modelling complex environmental systems (Young, 2002; Garnier and Young, 2014; Young, 2018). This is because the associated parameters are assumed inherently uncertain and can therefore only suitably be characterised in a stochastic format, such as through probability distributions (Young, 2006). Many DBM models can therefore emulate more complex distributed hydrological models and convert deterministic simulation equations derived from such models into stochastic formats (Young, 2006).

Fourthly, DBM models are based on actual observations and therefore are much more credible and operationally feasible when applied to individual catchments than purely theoretical modelling approaches. The physical, mechanistic interpretation of the model also makes them substantially more functional than black-box models. This combination of approaches tends to extract the benefits of both modelling approaches.

There are negatives of using the DBM modelling philosophy that should be considered however. These are mainly in relation to possible misinterpretation of the physical mechanisms/processes identified from the model structure and parameter values, as well as several hydrological processes possibly being combined into a single parameter within the model structure. With DBM models it is also relatively difficult to run scenario assessments or model ungauged catchments, although with synthetic transfer functions and model parameter regionalisation this is possible, e.g., Littlewood and Jakeman (1991).

Any model class where model structure can be identified purely from the data is acceptable within the DBM philosophy, providing physical interpretations are present and feasible (Young 1993). Consequentially, all models within the thesis satisfy the DBM modelling philosophy, though not all utilise the traditional transfer function format. Modelling approaches within the thesis include transfer function models (Section 2.8: Chapter 6), extreme value theory models (Section 2.9: Chapter 5), geostatistical models (Section 2.10: Chapter 4 and Chapter 7), and linear mixed-effects regression models (Section 2.11: Chapter 4).

2.8 Transfer function modelling

2.8.1 The transfer function concept and hydraulic interpretation of a linear store

Transfer functions (TFs) are rational polynomial functions of complex operator (usually backward shift (z^{-1}) operator in discrete-time, or Laplace differential (s) operator in continuous-time), broadly mathematically equivalent to systems of linear differential or difference equations which are commonly used to describe mass or heat transfers and chemical reactions, but defined in the frequency domain. They originate within signal and control engineering, but due to their efficient parametrisation and the existence of general and effective estimation methods have become a ubiquitous tool in many disciplines, including hydrology (Young, 1998a; Young and Garnier, 2006).

A TF model can be defined in both discrete- or continuous-time. Hydrological and environmental time-series are usually sampled at discrete-time intervals, and resultantly, discrete-time TF models are the most widely used (Jakeman et al., 1990; Young, 2003; Young and Garnier, 2006; Garnier and Young, 2014; Young, 2018). Continuous-time TF formulations have the advantage over discrete-time models in that the model's parameters are not related to the sampling-interval and may better approximate systems containing very rapid dynamics (Åström, 1969; Young and Garnier, 2006; Young, 2010). Comparisons of continuous-time and discrete-time TF models applied to the same data sets are provided in Young (2008; 2011).

The discrete-time TF is expressed in Equation 2.11.1:

$$Y = \frac{B(z^{-1})}{A(z^{-1})} \tag{2.11.1}$$

In the time domain, Y=Y(k), k=1,…,N is the observed output at sample k (of the total N observations), U=U(k), k=1,…,N is the input.

The polynomial $A(z^{-1})$ is a constant-coefficient polynomial defined as Equation 2.11.2:

$$A(z^{-1}) = 1 + a_1 z^{-1} + a_2 z^{-2} + \ldots + a_n z^{-n} \tag{2.11.2}$$

And the polynomial $B(z^{-1})$ is a constant-coefficient polynomial defined as Equation 2.11.3:

$$B(z^{-1}) = b_o + b_1 z^{-1} + b_2 z^{-2} + \ldots + b_m z^{-m} \tag{2.11.3}$$

with z^{-1} the backward shift operator, defined in the time domain by $z^{-i}\, y(k) = y(k-i)$. Pure time-delay can be included in the model by the initial δ of B polynomial terms taking values of zero.

Fitting a TF model to data is one of the possible approaches within the Data-Based Mechanistic methodology, estimating a range of models characterised by the structure parameters: order of the denominator polynomial A n (also – order of the system: the highest derivative order of the equivalent set of differential equations with a potential interpretation of the number of stores in the modelled system), order of the numerator polynomial B m, and the pure time-delay (δ). Together this structure is often formulated as the triad:

$$[n, m, \delta]$$

Most TF models within hydrology have n and m value of three or less, as higher order models are difficult to interpret within the Data-Based Mechanistic philosophy (Young, 2002). This triad is sometimes supplemented by the coefficient α, which is the parameter of the non-linear (usually power) function mapping the current rainfall and the variable chosen as the surrogate of the catchment saturation, usually either past discharge or a form of antecedent precipitation/saturation index:

$$[n, m, \delta]^{\alpha}$$

The nature of this nonlinearity is consistent with the well-known hydrological concept of a ‘dynamic contributing area’ described in Section 1.2 (Young, 2003; Beven, 2012).

For systems which can be interpreted within the mass transfer paradigm, the TF models can be presented as a combination of serial, parallel or feedback connections of first-order TF blocks, each of them describing the transport or storage dynamic model. This can be interpreted as a hydraulic vessel analogy – each first-order block represents a leaky vessel (see later Figure 2.1).

The equivalent continuous-time TF, which is directly and parametrically linked to differential equations, has a simple interpretation in terms of gains (equilibrium equivalent losses, long-term runoff coefficient) and time constants (recession-time) of each of the identified vessels, characterising the system's response respectively in the magnitude and temporal dimensions (Equation 2.12.1).

$$Y\,(s) = \frac{B(s)}{A(s)} e^{-s\tau} U(s) \tag{2.12.1}$$

A(s) and B(s) are defined as before, but in terms of the Laplace derivative operator s, and are defined in general as Equation 2.12.2:

$$A(\mathrm{s}) = \mathrm{s}^{\mathrm{n}} + \mathrm{a}_1\mathrm{s}^{\mathrm{n}-1} + \mathrm{a}_2\mathrm{s}^{\mathrm{n}-2} + \cdots + \mathrm{a}_{\mathrm{n}}\mathrm{s}^0 \tag{2.12.2}$$

And Equation 2.12.3:

$$\mathrm{B}(\mathrm{s}) = \mathrm{s}^{\mathrm{m}} + \mathrm{b}_1\mathrm{s}^{\mathrm{m}-1} + \mathrm{b}_2\mathrm{s}^{\mathrm{m}-2} + \cdots + \mathrm{b}_{\mathrm{m}}\mathrm{s}^0 \tag{2.12.3}$$

$e^{-s\tau}$ is the Laplace transformation of the pure time-delay.

As before, in the transport and storage context, these polynomials can be split through partial fractions expansion into a combination of sums and products of first-order TFs, equivalent to a combination of serial, parallel or feedback connections of vessels or stores. Each of these vessel models can be shown to be parameterised by steady-state-gain (SSG) and time constant (TC) as shown in Equation 2.13:

$$Y = \frac{b_0}{s + a_1} e^{-s\tau} U = \frac{SSG}{sTC + 1} e^{-s\tau} U \tag{2.13}$$

Therefore, for example, a parallel connection of two vessels can be modelled in continuous-time as Equation 2.14:

$$Y = \frac{b_0 s + b_1}{s^2 + a_1 s + a_2} e^{-s\tau} U \tag{2.14}$$

Very similar, analogous consideration can be made for a parallel connection of two equivalent discrete-time first-order stores models. Given that this is a parallel connection of two vessels or linear stores, the denominator polynomial in continuous-time has two real roots equal to the negatives of reciprocals of the two TCs, and the two gain parameters SSG are non-negative (Equation 2.15.1):

$$Y = \frac{SSG_1\left(s + \frac{1}{TC_2}\right) + SSG_2\left(s + \frac{1}{TC_1}\right)}{\left(s + \frac{1}{TC_1}\right)\left(s + \frac{1}{TC_2}\right)} e^{-s\tau} U \qquad (2.15.1)$$

Leading to Equation 2.15.2:

$$Y = \left(\frac{SSG_1}{1 + TC_1 s} + \frac{SSG_2}{1 + TC_2 s}\right) e^{-s\tau} U \qquad (2.15.2)$$

The total SSG (SSG_t) of the system is therefore defined as Equation 2.16:

$$SSG_t = SSG_1 + SSG_2 \qquad (2.16)$$

Time constants in this context provide the temporal scaling of the exponentially decaying recession curves. In terms of physical interpretation, they relate to mean residence time of the hydraulic vessel – the simple linear store (Figure 2.1).

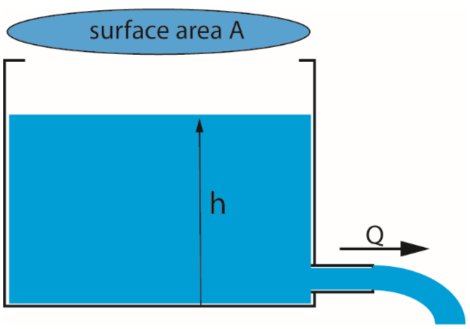

Figure 2.1: The hydrological interpretation of the draining of a hydraulic vessel – a simple linear store.

This simple linear store analogy can derive the following equation describing the change in head (h) given no input over time (t) as Equation 2.17.1:

$$\frac{dh}{dt} = -\frac{\rho_w g}{AR} h = -\frac{h}{TC} \tag{2.17.1}$$

with ρ_w being water density, g the gravitational constant, A the tank surface area, R the flow resistance, and TC the time constant. This differential equation leads to the solution for the recession curve (Equation 2.17.2):

$$h(t) = h_0 e^{-\frac{t}{TC}} \tag{2.17.2}$$

with h_0 being the initial condition.

When TC_1 and TC_2 are considerably different i.e., the system's dynamics are characterised by a combination of fast and slow dynamic behaviour, the system is said to be 'stiff' (Garnier and Young, 2014). This infers that the system contains both a fast pathway and a slow pathway (physically interpreted as the two hydraulic vessels emptying at different rates), potentially separating OFRSSF from deeper/slower hydrological pathways (Young, 2002).

This simple dynamic-linear analogy has some constraints, but it is clear how TC may relate to the properties of a hydrological system, potentially involving storage, hydraulic conductivity, saturation, travel-time etc. It is also straightforward to use this analogy to produce parallel flow models, reaches feeding into one another (serial connection) and also to introduce feedbacks and threshold phenomena. Equations 2.17.1 and 2.17.2 can be easily modified to include inflow into the store/vessel, either as precipitation (directly in units of height or head) or as inflow from another vessel. The effect of this inflow can be quantified as SSG and the equations translate directly into a TF.

2.8.2 Data-Based Mechanistic (DBM) transfer function identification and model structure

Model identification within a Data-Based Mechanistic transfer function framework is defined as the process of identifying a shortlist of structures expressed as $[n, m, \delta]^{\alpha}$ triads through estimating a range of model orders with n, m, δ and α changing within specified ranges. The estimation results are then quantified using objective statistical criteria such as the coefficient of determination (also known as Nash-Sutcliffe Efficiency – NSE, see Nash and Sutcliffe, 1970). For "sensible" models this criterion can be interpreted as the proportion of variance of the observed output y explained by the model output ŷ, as expressed in Equation 2.18.1:

$$R_t^2 = \frac{\sigma_{\hat{y}}^2}{\sigma_y^2} \tag{2.18.1}$$

Assuming that the residuals are uncorrelated with the model output, this becomes Equation 2.18.2:

$$R_t^2 = \frac{\sigma_y^2 - \sigma_e^2}{\sigma_y^2} \tag{2.18.2}$$

where e are the model residuals. For a model with perfect fit, R_t^2 has a value of 1 (zero residuals), however, for very poor or unstable models R_t^2 can take negative values, with the model variance being larger than the measured output variance (Blöschl et al., 2013b). Being variance based, R_t^2 is sensitive to the highest values of residuals. Values over zero indicate the level of data explanation by the model is better than a simple output average (when the numerator of the R_t^2 equation is zero).

Another criterion is Young's Information Criterion (YIC), which is a heuristic measure derived from a combination of Akaike's Information Criterion and the measure of entropy using the Fisher Information Matrix used in the Least Squares estimation process (Young, 1990; 2011). The YIC is defined as Equation 2.19:

$$YIC = ln\frac{\sigma_e^2}{\sigma_y^2} + ln\{NEVN\} \tag{2.19}$$

Where NEVN is the Normalised Error Variance Norm, expressed as Equation 2.20:

$$NEVN = \frac{\sigma_e^2}{n_p}\sum_{i=1}^{i=n_p}\frac{\hat{P}_{ii}}{\hat{\alpha}_i^2} \tag{2.20}$$

where $\hat{\alpha}_i^2$, i = 1,…np are estimated parameters of the TF and $\hat{P}_{ii}$, i = 1,…np are the diagonal elements of the estimated covariance matrix of these parameters.

The YIC quantifies the combination of a part of the coefficient of determination with a form of combined coefficients of variation of each estimated parameter. The second part of YIC is a measure of relative (normalised) uncertainty of parameters. In over-parameterised models this uncertainty grows fast and this is penalised when minimising the YIC. Thus, the combination of YIC and R_t^2 provide a balance, producing a "league table" of models, listing the best fitting and best-defined models (Young et al., 1996). This is done automatically within the Computer Aided Program for Time-series Analysis and the Identification of Noisy systems (CAPTAIN) Toolbox identification procedures within MATLAB (see Taylor et al., 2007).

Note that the TFs used in this thesis are linear, and thus the Refined Instrumental Variable (RIV) and Simplified RIV (SRIV) provide a robust approach to model identification and estimation (Young and Jakeman, 1979, 1980; Young 1985; Young et al., 1996; Young, 1998a). When relationships are considerably nonlinear, Time Varying Parameter (TVP) or State Dependent Parameter (SDP) estimation procedures can be used to identify the nature of the nonlinearity (Young, 1998a; 2003; Young and

Garnier, 2006). This combines Fixed Interval Smoothing (FIS) algorithms which provide methods of non-parametric estimation, with the TVP which defines the non-parametric relationship, which can then be interpreted in state-dependent terms (Young, 1993; Young and Beven, 1994; Young, 1998b; 2003; Young and Garnier, 2006).

Following identification, the models undergo mechanistic interpretation as part of the Data-Based Mechanistic philosophy. Models that do not have a physical interpretation, in this case, the ones that do not factorise into physically feasible combinations of linear stores, are rejected, usually leaving a clear model candidate with well-defined parameters and a structure that has a physical interpretation.

A TF modelling approach was used in two applications for Chapter 6 to compare an agriculturally-intensive improved-pasture with a hedgerow wild-margin (described in detail in Chapter 6). The first was a controlled experiment in which a constant rainfall rate from a laboratory-made rainfall generator provides a constant input, and the soil saturation time-series is monitored (output). This TF model of soil saturation then undergoes decomposition into simple first-order models to infer different hydrological response components, which determine θ_v between the two aforementioned land-uses. In the event that overland flow occurs, an in-series TF model can be created, with the soil saturation time-series possibly acting as an input for the overland flow time-series (note that overland flow was not generated during the experiment). In the second instance, natural (and therefore dynamic) precipitation is used as an input to the TF model in order to predict overland flow (output). This is again decomposed into first-order TF models to suggest the influence of land-use and evapotranspiration/drainage upon overland flow.

2.9 Extreme value theory and peaks-over-thresholds analysis

Extreme value theory (EVT) is the branch of statistics that deals with the stochastic behaviour of extreme deviations from the median of a standard probability distribution, including the study of the asymptotic distribution of extremes. This is critically important for risk assessment/analysis, and resultantly, EVT is commonly used throughout hydrology (Longin, 2016). The most common application of EVT in hydrology includes the forecasting of large return (flood) events that surpass observational records (e.g., the 1-in-100-year flood event). Other common hydro(meteoro-)logical applications of EVT include estimating probable maximum precipitation intensities, and in estimating the length of drought. The methods of pre-processing data for such analyses are very useful in extracting extremes from a dataset.

There are two widely used methods within EVT which are applicable in extracting extreme values. The first is the ‘block maxima (minima)’ approach (Gumbel, 1958). This approach involves dividing data (predominantly temporal) into ‘blocks’ of uniform length and selecting the maximum (minimum) observation within each block. Extracted extremes then create a new parametric distribution, with block maxima (minima) tending to fit the Generalised Extreme Value (GEV) distribution. This new distribution can then undergo further distribution analysis (for example, when transformed into the Gumbel, Fréchet or Weibull distributions [Type I, II and III, respectively]: Gumbel, 1958; Collet et al., 2017). Alternatively, extracted values can undergo statistical analysis and scientific interpretation.

The largest weakness of the block maxima (minima) approach is that ‘block’ length – year, month, week etc. can substantially bias extracted values, as extreme events often cluster together both spatially and temporally. This therefore can easily cause extreme events to be excluded, as well as include observations not typically associated with extremes (Ferreira and de Haan, 2013).

The second EVT method for extracting extremes is the Pickands-Balkema-de Hann theorem (Balkema and de Haan, 1974; Pickands, 1975) – the ‘peaks-over-thresholds’ (POT) approach. The POT method involves measuring the frequency and/or magnitude of events that surpass a given threshold over a full dataset (also usually temporal), and extracting these extremes. Extracted events are then used to create a new parametric distribution, with POT data tending to fit a General Pareto Distribution (GPD: Longlin, 2016). This new distribution can then be analysed as described above.

One of the largest drawbacks associated with the POT method is that the selected threshold can be seen as an arbitrary value. There are however a host of environmental thresholds in hydrology e.g., channel capacity, soil infiltration-capacity, or legislation thresholds e.g., WFD/World Health Organisation pollutant concentrations. Resultantly, POT analysis is particularly well-paired with hydrology. The POT method is also substantially more robust against clusters of extreme events compared to the block maxima (minima) approach, and therefore often makes better use of the data (Ferreira and de Haan, 2013). For these reasons, the POT approach was the chosen method to extract extreme values within the thesis.

There are several drawbacks associated with all EVT analyses that should be considered however. Firstly, EVT distributions are determined from the underlying parent distribution, with observations that are assumed both independent and identically distributed (i.e., stationary). The independence of environmental observations is frequently violated, with extreme events often clustered together

(Ferreira and de Haan, 2013; Cooley et al., 2017). Assuming stationarity in the underlying parent distribution is additionally unlikely to be an accurate assumption when working with environmental data, particularly with longer timeframes, due to the presence of trends or seasonality (Longlin, 2016; Collet et al., 2017). If there is clear violation in the independent and identically distributed assumption, for example, seasonal periodicity or short-range/term dependency, the block maxima (minima) method may be a more robust approach than POT (Ferreira and de Haan, 2013).

The second major uncertainty which compounds on the former issue is that statistical inference about extreme events can only be deduced from those observations which are extreme in some sense (Longin, 2016). Observations on such extreme events will be sparse by definition, such that the high magnitude tail of the distribution of events will necessarily be associated with increased uncertainty. Given the very limited coverage of hydrological data-series (both temporally and spatially), this substantially increases the uncertainty of such predictions (Watts et al., 2015; Barker et al., 2016; Cooley et al., 2017). This is further amplified as extreme hydrological events are often difficult to accurately measure and quantify hydrometrically, and are often associated with considerable uncertainties (e.g., overtopping of the FRBP-type flumes). This lack of data surrounding extreme events combined with increased uncertainty within existing data associated with the extremes complicates resultant analysis and interpretations, as well as potential modelling.

An EVT approach was applied to Chapter 5, with a POT statistical model pairing topsoil infiltration-capacity with precipitation-intensity. This paired data was then used to determine the incidence of infiltration-excess overland flow. This is explained in further detail in Chapter 5.

2.10 Geostatistical modelling

Geostatistics is the branch of statistics that analyses and interprets spatial (and sometimes spatio-temporal) data and patterns of spatial continuity. Geostatistics therefore analyses the variability of the difference between two observations of a stochastic process that are a given displacement apart (often called lag distance), usually in space, in order to quantify their spatial-structure (Chilès and Delfiner, 2012). These techniques primarily operate upon stochastic processes operating within (at least) two dimensions, although certain geostatistical tools can use one-dimensional statistics. This information is then used to quantify the spatial-structure of the process such as through variography e.g., (semi-)variogram and correlogram models, or to interpolate spatial-structure through gaussian process regression, most often kriging or spline functions (Chilès and Delfiner, 2012).

(Semi-)variogram (and to a lesser extent correlogram) models are commonly used to summarise spatial auto correlation (see Armstrong, 1998; Chilès and Delfiner, 2012). The variogram is a graphical representation of the magnitude of the variance of the difference as a function of displacement for every observation. Spatial variance is measured as the average of a squared difference (to account for negative values) for every pair of observations for a given displacement to create the variogram, and this value is then halved to become the semi-variance, which produces the semi-variogram. When based on observations rather than theory, the semi-variogram is termed the empirical semi-variogram, as it is impossible to measure all possible pairs for an infinite number of displacements.

In most stochastic processes with an observable spatial-structure, observations with small values of displacement tend to display high spatial correlation. Spatial correlation then tends to decay with increasing displacement (distance decay), causing the semi-variance to increase (Tobler's Law, see Tobler, 1970: Diggle and Giorgi, 2019). Plotting a graph of semi-variance against displacement creates a cloud of observations, which an empirical semi-variogram model can be estimated from (Diggle and Giorgi, 2019).

The defined parameters within most empirical semi-variogram models are the range, the sill, the nugget, and the partial sill (shown in Figure 2.2). The range is the displacement beyond which there is no longer spatial auto correlation present. The range can be difficult to accurately determine in many situations (and may not exist in the multifractal and non-stationarity case), as there are usually fewer observations at increasing displacements. Ideally an experimental design should measure displacement well beyond the empirical semi-variogram range to improve its derivation, although it is very difficult to estimate the range of a stochastic process without first sampling for and/or having very good prior knowledge of the underlying phenomenon.

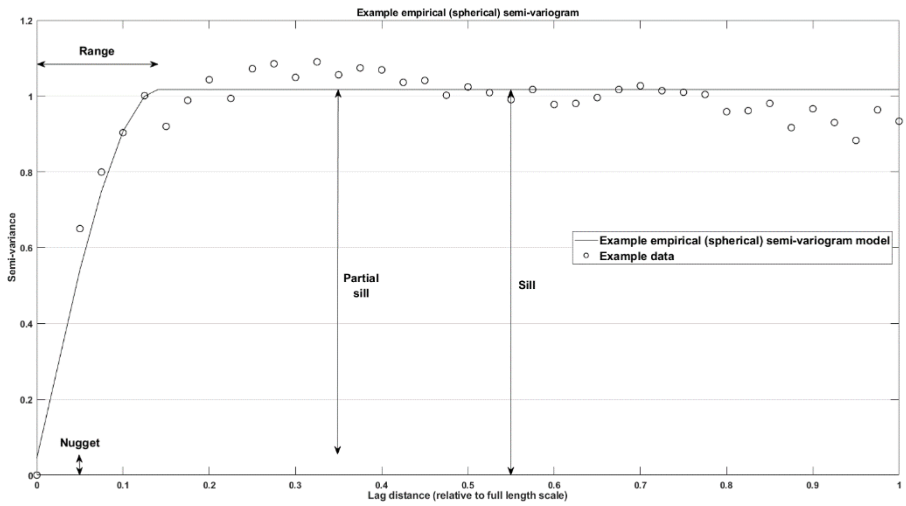

Figure 2.2: An example empirical (spherical) semi-variogram model, showing the nugget, range, sill, and partial-sill.

The nugget (or nugget variance) is the variance without displacement, and theoretically should be null. In most instances however, the nugget variance is not null, and this is caused by a combination of measurement and instrumentation error, and microscale variability, which is variance below the minimum displacement during sampling (variance exists at infinitely small displacements).

The sill is the point at which the semi-variance of the process plateaus (i.e., the semi-variance as lag distance approaches infinity), and in most semi-variogram models it is close to one (Chu, 2017). The sill is usually approximately one as this is when the semi-variance of observations is equal to the semi-variance of the stochastic process being studied (and hence is usually closely tied with the range). In instances when the sill is substantially larger than one, it indicates that the observations have not adequately captured all of the spatial structure present in the stochastic process. The partial sill is also sometimes included as a model parameter, and is simply the nugget variance subtracted from the sill.

There are numerous semi-variogram models, the most common being spherical, exponential and Gaussian models (Biswas and Si, 2013; Appel Neto et al., 2018). Spherical models show a progressive decrease of spatial auto correlation with displacement until a key displacement is reached, whereby there is no longer spatial auto correlation present (i.e., the range). In an exponential model, spatial auto correlation decreases exponentially with increasing displacement, meaning that theoretically spatial auto correlation continues to an infinite distance (SAS, 2019). The Gaussian model resembles the cumulative distribution function of the Gaussian (normal) distribution, and is used when the stochastic process has very strong auto correlation at small displacements before a progressive increase in semi-variance. There are also many other possible semi-variogram models of varying complexities, including combined (nested) semi-variogram models (see Stein, 1999; Guttorp and Gneiting, 2006; Chilès and Delfiner, 2012; Batistella Pasini et al., 2014; Appel Neto et al., 2018; Diggle and Giorgi, 2019; SAS, 2019).

Semi-variogram models make several notable assumptions. The first is the assumption of stationarity in the mean, which assumes that the mean is constant between samples and is independent of location (ArcGIS, 2012). The second is the assumption of intrinsic stationarity, which assumes that the variance of the difference is the same between any two points that have identical displacements, irrespective of samples or location (ArcGIS, 2012). Many semi-variogram models additionally assume that the underlying stochastic process operates within an isotropic plane, and therefore the semi-variogram is omnidirectional and has an identical spatial-structure in all directions for a given displacement, irrespective of location or angular orientation (Diggle and Giorgi, 2019). More complex semi-variograms can contain anisotropy, whereby spatial-structure is measured in respect to both displacement and angular orientation (SAS, 2019).

Variography was incorporated within Chapter 4 to quantify soil volumetric wetness spatial-structures between the semi-natural grassland and the improved-pasture for both drought, saturated, and two intermediary conditions. This variography was conducted using the EasyKrig 3.0 geostatistical package in MATLAB (Chu, 2017), and is explained further in Chapter 4. Both the semi-variogram model structure and model parameters were also analysed in detail for each sampling date for each land-use. Semi-variograms were assumed isotropic for simplicity, and semi-variogram models were determined to be either spherical, exponential or Gaussian. Chapter 7 involved a very basic form of geostatistical analysis regarding soil volumetric wetness at varying distances from the dry-stone wall boundary.

2.11 Linear mixed-effects regression modelling

Linear mixed-effects regression modelling is an extension of the simple and multiple linear regression modelling procedures, highly suited to complicated or detailed sampling designs, particularly nested sampling, repeat measurements, or multiple sampling sites (Faraway, 2016). Within simple linear regression, a dependent variable (y) is predicted according to an independent or 'predictor' variable (x_1), with a slope parameter (β_1), alongside an intercept (c). This single independent variable is unlikely to explain all the variance observed within the dependent variable, so a random error term (ε) is included within the model to account for unexplained variance caused by variables that are not directly measured. This produces the standard (simple) linear regression model (Equation 2.21):

$$y = c + \beta_1 x_1 + \varepsilon \tag{2.21}$$

Multiple linear regression involves multiple independent variables (x_1, $x_{2\ldots}$ x_v), with their respective slope parameters (β_1, $\beta_{2\ldots}$ β_v), up to the maximum number of independent variables, x_v, which are used to predict the dependent variable. This produces the multiple linear regression model (Equation 2.22):

$$y = c + \beta_1 x_1 + \beta_2 x_2 + \ldots \beta_v x_v + \varepsilon \tag{2.22}$$

When including multiple independent variables, often the effect size of an independent variable is dependent upon the value of other independent variables, which is termed an interaction effect. This interaction effect can be included in the model formula, and is displayed in Equation 2.23, which uses two independent parameters with an interaction term as an example:

$$y = c + \beta_1 x_1 + \beta_2 x_2 + \beta_3 x_1 x_2 + \varepsilon \tag{2.23}$$

Where $\beta_3 x_1 x_2$ is the interaction effect of the x_1 and x_2 independent variables.

Both single and multiple linear regression are best suited to situations when observations come from a single, homogenous group without any underlying structure. When observations are from multiple sites, contain repeat measures, and/or belong to nested or hierarchical subgroups within the population, a mixed-effects model is often a more suitable modelling approach. Linear mixed-effects regression models can include random effects within the model, which incorporates underlying structures that would otherwise be lost within the ε term. An increased amount of variance may therefore be explained by these subgroups, which could improve the overall model fit.

Linear mixed-effects regression models include both fixed effects and random effects (Zuur et al., 2009; Faraway, 2016). A fixed effect is identical to the independent variable (described above) within both simple and multiple linear regression, with investigations primarily interested in how fixed effects influence the dependent variable (Faraway, 2016). Fixed effects are therefore expected to have a systematic and predictable influence upon the dependent variable. Conversely, random effects are expected to have a stochastic rather than predictable influence upon the dependent variable. These are largely variables of secondary-no interest to the investigation, although these may still explain variance in the dependent variable (Faraway, 2016). It is common for samples regarding the random effect(s) to be a small subset of a wide range of possible values (they do not exhaust the population of possible values), and/or variables that are believed to be specific to the data or sampling site (i.e., highly localised or not consistent across the sampling population). Commonly, individual subjects, sampling site or broad-landscape factors are often suitable random effects.

There are three basic forms of linear mixed-effects regression model. A random intercept(s) model is a linear mixed-effects model, which assumes a different intercept value for each specific value within the random effect (Zuur et al., 2009). Random intercept models can contain multiple random intercepts (Winter and Grawunder, 2012). A random intercept model with multiple random intercepts is shown in Equation 2.24:

$$y = c + c_{r1} + c_{r2} + \ldots c_{rv} + \beta_1 x_1 + \beta_2 x_2 + \ldots \beta_v x_v + \varepsilon \tag{2.24}$$

Where c_r is the random intercept and c_{rv} is the number of random intercepts in the model. The general intercept for all the data (c) is still retained, although this intercept is combined with the value(s) of the random intercept(s) to further explain variance.

A random slope(s) model is another form of linear mixed-effects regression model. This model is where the slope parameter for a fixed effect is added to a slope value generated for each of the specific values of the random-effect (Zuur et al., 2009). This is expressed in Equation 2.25:

$$y = c + (\beta_{r1} + \beta_1)x_1 + (\beta_{r2} + \beta_2)x_2 + \ldots (\beta_{rv} + \beta_v)x_v + \varepsilon \tag{2.25}$$

Where β_r is the random slope and β_{rv} is the number of random slopes in the model.

The final type of linear mixed-effects model is a random intercept(s) and random slope(s) model. This model is simply a linear mixed-effects model containing both a random intercept(s) and a random slope(s) (Zuur et al., 2009), and is a combination of Equation 2.24 and Equation 2.25, given by Equation 2.26:

$$y = c + c_{r1} + c_{r2} + \ldots c_{rv} + (\beta_{r1} + \beta_1)x_1 + (\beta_{r2} + \beta_2)x_2 + \ldots (\beta_{rv} + \beta_v)x_v + \varepsilon \tag{2.26}$$

Note that all linear mixed-effects models retain the ε term, as unexplained variance is still likely present.

Several modelling assumptions must be satisfied or at least acknowledged before deeming a linear-mixed effects regression model a suitable model of the data (all assumptions are synonymous with multiple linear regression). Firstly, model residuals should contain no clear trend or pattern, as this indicates a violation of linearity. Secondly, the model residuals should be normally distributed. Thirdly, the model residuals should be homoscedastic by having approximately equal variance across the range of predicted values within the model. Fourthly, fixed-effects/independent variables should not be strongly correlated with each other as this leads to instability in model interpretation. Fifthly, there should be no extremely influential data points that substantially influence the model outputs. Lastly, the model assumes that each data-point is independent of all other data points, unless strictly specified within the model structure.

A linear mixed-effects regression model was used to investigate and explain the variance in soil volumetric wetness between the semi-natural grassland and the improved-pasture, and is explained in detail in Chapter 4. Fixed-effects of land-use, vegetation and month were used, alongside interaction effects of month:vegetation and month:land-use to account for temporal variability in the effect of vegetation and land-use upon soil volumetric wetness (note that a colon is standard notation for an interaction effect in statistical literature). Random-effects were taken as the intercept of elevation, with by-elevation random slopes for the effect of vegetation and month. Elevation was taken as a random effect due to the limited range of elevation measured given its possible values across the landscape, and due to the fact that it is a repeat measure but can diverge more information than simply referring to it as an individual subject. All linear mixed-effects model assumptions were satisfied following model calibration to ensure model suitability.

2.12 Experimental designs and data analyses

2.12.1 Experimental design and data analysis of Chapter 4

2.12.1.1 Chapter 4 experimental design

To reiterate: *Objective 1)* Quantify the effect of a grazed semi-natural grassland as opposed to a grazed agriculturally-intensive improved-pasture and silage field on spatio-temporal soil volumetric wetness (θ_v), from extremes of drought to fully saturated conditions. Soil volumetric wetness will be correlated to local topography and local vegetation to assess their influence. Soil volumetric wetness will be used to infer the sources and amounts of saturation-excess overland flow during flood events, as well as the extent of drying during drought events.

Chapter 4 (Objective 1) will involve selecting a (semi-)natural grassland and an adjoining improved-pasture, with the latter an approved Eden Rivers Trust site that is possibly involved with the Farming Facilitation Fund. Both sites must be identical besides the land-use, as well as be 'typically' managed for the area e.g., not abnormally heavily grazed, to avoid skewing findings. No other large features should be included within the sampling grid that could skew measurements and resultant analysis e.g., no trees. The site should be easily accessible given that equipment will have to be manually carried, as well as for safety reasons. Ideally, phone signals should also be available and houses nearby, also for safety reasons. Permission from both landowners needs to be granted before accessing land, and the required permits obtained.

Soil volumetric wetness will be compared between the two land-uses at several points throughout the year, ranging from dry (ideally drought), to saturated (ideally storm) conditions, with intermediary conditions also included. Measurements of local vegetation and topography will be taken at the same scale of soil volumetric wetness so that these can be easily and directly compared as local controls on soil volumetric wetness (as inputs for the linear mixed-effects regression model). Both topography and vegetation are assumed stationary during the study so only require measuring once. Vegetation will be measured in spring/summer to facilitate species identification. Soil samples will be taken randomly throughout each land-use to categorise site conditions and to help interpret data. An antecedent precipitation index based upon local rainfall will also be created for the site to assess prior conditions influencing soil volumetric wetness at each of the sampling dates.

A sampling grid that encompasses both land-uses equally will be deployed. The grid will be 32 m x 48 m in total (16 m x 48 m each), giving 1536 total measurements (768 measurements each). The grid will be orientated with the longest sides directly along the boundary to keep measurements as close as possible, and soil volumetric wetness will be taken at a 1 m resolution within the sampling grid. The maximum number of measurements possible will be taken to support statistical analysis (and geostatistical modelling), although it is crucial that the soil volumetric wetness does not change during measurements. To ensure soil volumetric wetness does not change during measurements, the first several measurements will be repeated following the final measurements to ensure no change in soil volumetric wetness has occurred beyond instrumental uncertainty. Fixed markers e.g., stones and sticks, will be placed on several points of the grid to allow for repeat analysis of the same points. Following measurements, the grid will be removed to allow grazing and farming practices to continue unhindered.

Local topography will be taken using a total station and a differential GPS. This approach can allow the site to be precisely mapped by using the fixed markers. Vegetation will be determined by detailed measurements within each 1 m grid.

2.12.1.2 Chapter 4 data analysis

Soil samples will be measured for particle size distribution, dry bulk-density, organic matter content, porosity, penetration resistance and pH. Non-parametric statistical tests will be used to compare soil variables between land-uses due to small samples sizes.

Soil volumetric wetness will be quantitatively compared between land-uses through statistical tests of central tendency (mean or median depending upon normal distribution tests) and variance. Skew and kurtosis measurements will also be included to further divulge distribution information given the sample sizes. Geostatistical models will be used to analyse spatial dependency between the two land-uses and their parameters and structure directly compared given identical grid sizes/shapes and immediate vicinity. A linear mixed-effects regression model will be used to assess and quantify the influence of vegetation, topography, as well as land-use and sampling date on soil volumetric wetness variance at a 1 m scale.

2.12.2 Experimental design and data analysis of Chapter 5

2.12.2.1 Chapter 5 experimental design

To reiterate: *Objective 2)* Quantify the effect of blade aeration within several agriculturally-intensive improved-pasture and silage fields on temporal saturated hydraulic conductivity (K_{sat}) and related soil penetration resistance (PR) of the topsoil. Saturated hydraulic conductivity will be used to investigate changes in the amount of infiltration-excess overland flow during extreme precipitation events generated from aerated and unaerated improved-pasture, by contrasting a high-frequency precipitation time-series with K_{sat} values.

Chapter 5 (Objective 2) will involve selecting an improved-pasture that is being partially treated with blade aeration. The site must be Eden Rivers Trust approved and possibly ran by a member of the Farming Facilitation Fund. Both treated and untreated sections of the improved-pasture must be as similar as possible and within the same field. The site should be easily accessible given that equipment will have to be manually carried, as well as for safety reasons. Ideally, phone signals should also be available and houses nearby, also for safety reasons. Permission from the landowner needs to be granted before accessing land. Given that permeability measurements require large volumes of water, a large amount of accessible water is essential (this does not need to be potable).

The saturated hydraulic conductivity and soil penetration resistance will be measured between both aerated and non-aerated sections of improved-pasture. Specific locations for measurements will be selected randomly within each treatment, and if slits remain, the soil penetrometer will not be intentionally placed into them (the permeability ring will also be placed randomly i.e., not intentionally over slits if slits are visible). The penetrometer will also not be used on sites previously used for saturated hydraulic conductivity measurements (and vice versa). Measurements will be repeated in several-week intervals to assess the duration of possible improvements caused by the blade aerator. Soil samples will be collected at varying depths from soil pits dug (at a random location) within each treatment to investigate what is occurring below the immediate soil surface, as well as for improved site categorization. Samples of grass species will also be collected for site categorisation and to improve holistic interpretations. A precipitation time-series from a local rain-gauge will also be collected.

Permeability measurements will be conducted whilst cores remain within the ground (*in-situ*). This is to ensure that permeability measurements are capturing potential compaction and reduced permeability below the depth of the aerator blades i.e., the core will not be removed from the ground as the limiting factor on permeability may be below the depth of the ring, and therefore the removal of the core will artificially inflate permeability values. Global Positioning System (GPS) locations of each permeability measurement will be taken to assess for clear trends in data and to avoid repeat measurements at a sampling location.

2.12.2.2 Chapter 5 data analysis

Soil samples will be measured for particle size distribution, dry bulk-density, organic matter content, porosity, and pH. Non-parametric statistical tests will be used to quantitatively compare soil variables between land-uses due to small samples sizes. Qualitative descriptions of the soil profile, supported by quantitative soil properties wherever possible, will also be used.

Soil penetrometer results will be quantitatively compared by contrasting the number of successful measurements, the mean compaction, and the median compaction, for each of the measured depths. The median value will then be compared for each depth with the Mann-Whitney-Wilcoxon test. The permeability measurements will be quantitatively compared with summary statistics, as well as via the Mann-Whitney-Wilcoxon test. A peaks-over-thresholds statistical model will pair the summary statistics of the measured permeability with the precipitation time-series, to assess the likelihood of infiltration-excess overland flow between the treated (aerated) and control (unaerated) land-uses.

2.12.3 Experimental design and data analysis of Chapter 6

2.12.3.1 Chapter 6 experimental design

To reiterate: *Objective 3)* Quantify the effect of hedgerows/hedge-margins within agriculturally-intensive improved-pastures in controlling changes in the amount of overland flow and rapid shallow-subsurface flow during floods and extreme precipitation events via direct measurement. This will be supported by causal factors of saturated hydraulic conductivity (K_{sat}) and soil volumetric wetness (θ_v). Overland flow will be analysed for targeted co-benefits to water-quality (total sediment (TS), nitrate (NO_3^-), nitrate-nitrite ($NO_3^-NO_2^-$), soluble reactive phosphorus (SRP), particulate total phosphorus (PTP), dissolved total phosphorus (DTP), and dissolved organic carbon (DOC)).

Chapter 6 (Objective 3) will involve selecting an improved-pasture and an adjoining hedge/hedge-margin. The site must be Eden Rivers Trust approved, possibly involving the Farming Facilitation Fund. The sites must be immediately adjacent and have identical soil types, geologies etc. The site should be easily accessible as using installation machinery will likely invalidate findings due to site disturbance, and therefore installation will be manual wherever possible. Phone signals should be nearby for both safety and possible telemetry reasons. Permission from the landowner needs to be granted before accessing land, and the required permits obtained for land access, the installation of infrastructure, and the building of small, temporary structures.

Overland flow plots will be installed in both the improved-pasture and the hedge-margin. They will be installed as close as feasibly possible to reduce natural soil variation, but not close enough to be unduly affected by edge effects. Overland flow plots, particularly the troughs, need to be protected from livestock trampling and general disturbance, and this will be done by fencing (note that this fencing is only immediately surrounding the troughs and is only cattle-proof, with smaller sheep and lambs still having easy access to the trough to graze right up to the collection system). The steel sheeting used to bound the plots needs to be extremely durable, capable of sustaining heavy trampling and possibly even smaller agricultural traffic. The overland flow plots additionally need to be a size that will not overwhelm the system in an extreme event. The trough needs to be fully-sealed and leak-proof, and known volumes of water needs to be passed through the system and accurately recorded. The overland flow plot system also needs to be self-draining to prevent water backlogs damaging equipment, and the pipe network wide enough to allow material to pass through without causing a blockage. The plots also need to channel water to storage for water-quality analysis. The full overland flow plots need to be entirely capable of working under gravity alone, as well as require minimal maintenance.

Overland flow will be monitored at the plots until several events have been recorded. More than one overland flow plot for each land-use will be constructed to insulate against equipment redundancy, as well as to capture natural variation in overland flow. A rain-gauge will be installed within the improved-pasture to capture highly localised rainfall e.g., convective events, which may generate highly localised overland flow, particularly IOF. A secondary rain-gauge needs to be located close by, also to insulate against equipment redundancy. A flume will also be installed on the local stream to compare/correlate overland flow occurrence and river discharge, and to infer the saturation-state of the sub-catchment.

Permeability will be measured surrounding the plots to improve the interpretation of overland flow measurements. Topography of the plots will also be taken using a total station to further understand natural slope variability between plots and to help categorise findings. Soil samples will be taking immediately surrounding the overland flow plots to help compare land-uses, to add to site categorisation, and to help improve interpretations. A detailed survey of local vegetation within each plot will also be taken to improve both site categorisation and hydrological interpretations.

An artificial-rainfall experiment and a wash-off experiment will also be conducted within the overland flow plots. The artificial-rainfall experiment will involve a soil moisture-probe placed in the overland flow plots during the experiment to quantify plot saturation, and therefore to increase the understanding of the saturation process preceding overland flow generation. This experiment will operate using plausible precipitation intensities to help to understand the saturation mechanisms prior to SOF, and possibly the required precipitation intensities required for IOF. It is also possible that the artificial-rainfall experiment could help to understand hydrophobic overland flow given site conditions. The artificial-rainfall experiment may also generate overland flow from each land-use, which can be volumetrically compared, as well as analysed for water-quality. The artificial-rainfall experiment will therefore require considerable volumes of deionized water.

Wash-off experiments will be a secondary option if the artificial-rainfall experiment fails to generate overland flow. Wash-off experiments will ensure overland flow is generated and can be chemically analysed, providing a failsafe for both the artificial-rainfall experiment and the natural overland flow time-series. Due to extremely high rainfall-intensities associated with wash-off experiments, overland flow will be captured once it has entered the overland flow pipe network but before it reaches the overland flow gauge as the high volume may damage the equipment and flood the enclosure (including the expensive data-logging equipment). As before, a considerable volume of deionised water will be necessary for this experiment.

2.12.3.2 Chapter 6 data analysis

Soil samples will be measured for particle size distribution, dry bulk-density, organic matter content, porosity, and pH. Non-parametric statistical tests will be used to quantitatively compare soil variables between land-uses due to small samples sizes. Qualitative descriptions of the soil profile, supported by quantitative soil properties wherever possible, will also be used.

The permeabilities will be statistically assessed for normality, and then quantitatively contrasted via statistical tests for median and/or mean. The overland flow time-series generated by natural precipitation, the overland flow generated by the artificial rainfall-experiment, and the soil volumetric wetness time-series generated by the artificial-rainfall experiment will be quantitatively compared using a transfer function modelling approach. The overland flow water-quality will only be quantitatively compared only.

2.12.4 Experimental design and data analysis of Chapter 7

2.12.4.1 Chapter 7 experimental designs

To reiterate: *Objective 4)* Quantify the effect of dry-stone wall boundaries within sloped areas of agriculturally-intensive improved-pastures upon changes to soil volumetric wetness (θ_v) above and below the barrier, and therefore infer changes to overland flow and rapid shallow-subsurface flow likelihood by assessing the impedance of θ_v transfer downslope.

Chapter 7 (Objective 4) will involve taking soil volumetric measurements above and below multiple dry-stone wall boundaries on several sloped improved-pastures. Each site must be Eden Rivers Trust approved and possibly ran by a member of the Farming Facilitation Fund. The specific management practices above and below the wall must be as similar as possible, and both sites must have as similar as possible soil types, surficial geologies and vegetation. Any non-uniformity must be clearly highlighted as this suggests a possible external influence beyond that of the intervention. The sites should have phone signals and should be nearby to houses for safety reasons. Given the minimal amount of required equipment, easy access is not essential. Permission from the landowner needs to be granted before accessing land.

Soil volumetric wetness measurements will be conducted above and below the dry-stone wall barriers in (primarily) 16 m by 16 m grids. This grid size correlates well with Chapter 4, can be conducted extremely rapidly, and does not extend excessively above or below the boundary. This grid size may be modified if this will avoid external (non-intervention) influences being recorded in the data e.g., a farm track or a change in vegetation. Any external influence on soil volumetric wetness must be clearly identified for each site. Efforts will be made to minimise the impact of stones when taking measurements immediately adjacent to the wall, which could skew soil volumetric wetness readings. Each wall will be measured once, although a considerable number of walls will be measured throughout the landscape. Measurements will only be conducted in the winter half of the year to improve the likelihood of (near-)saturated conditions occurring and therefore the chance to record measurements during periods of high likelihood of moisture transfer downslope. Soil samples will not be needed, as the land-use is assumed identical above and below the wall.

2.12.4.2 Chapter 7 data analysis

Soil volumetric wetness will be statistically and quantitatively compared above and below the dry-stone walls by central tendency tests of mean or median (depending upon normality tests). These measurements will be conducted over the full soil volumetric wetness grid, as well as at varying distance to the boundary. Qualitative descriptions of the soil volumetric wetness will also be included where necessary, possibly supported by visual soil volumetric wetness grids.

2.12.5 General considerations for experimental designs and data analyses

The experimental designs and data analyses adopted in this thesis and outlined above can be deployed throughout the UK and much further afield. These can include the monitoring of identical interventions elsewhere (both in the UK and overseas), for example, on different soils or on replicates of those studied here. Alternatively, these could be deployed on different interventions not measured in this thesis (see Table 1.1: Figure 1.1: for examples). It is also possible to modify and adopt the experimental designs and data analyses for alternative investigations, both hydrological or otherwise.

2.13 Summary of methods in relation to objectives

Objective 1 (Chapter 4) will be achieved by measuring soil volumetric wetness (Section 2.3.2) and soil hydropedological properties (Section 2.4), supported by precipitation (Section 2.3.5), to further the understanding of the influence of land conversion on overland flow and rapid shallow-subsurface flows (Section 2.3.1), as well as droughts. The effect of land conversion on surface moisture patterns will be analysed through geostatistical modelling (Section 2.10) and linear mixed-effects regression modelling (Section 2.11), in a hydrological modelling framework (Section 2.6-2.7).

Objective 2 (Chapter 5) will be achieved by measuring permeability (Section 2.3.3) and soil hydropedological properties (Section 2.4), supported by precipitation (Section 2.3.5), to further understand the influence of aeration on overland flow and shallow-subsurface flows (Section 2.3.1). This will be analysed through an extreme value theory and peaks-over-thresholds analysis (Section 2.9), as before within a hydrological modelling framework (Sections 2.6-2.7).

Objective 3 (Chapter 6) will be achieved by measuring all highlighted hydrometric properties (Section 2.3), hydropedological properties (Section 2.4) and water-quality properties (Section 2.5). The impact of hedge-margins on overland flow and shallow-subsurface flow (Section 2.3.1) will then be analysed by deploying various transfer function modelling approaches within a hydrological modelling framework (Sections 2.6-2.8).

Objective 4 (Chapter 7) will be accomplished by measuring soil volumetric wetness (Section 2.3.2) above and below each dry-stone wall. This will be used to infer if dry-stone walls prevent or slow the transfer of water downslope, and therefore can infer changes to overland flow and shallow-subsurface flow pathways and likelihoods (Section 2.3.1). Given the assumed uniformity of land-use, no additional hydrological or hydropedological measurements will be needed. This will then be incorporated within a hydrological modelling framework (Sections 2.6-2.7; Section 2.10).

Objective 5 (Chapters 4-7) will be achieved by deploying the statistical and hydrological modelling techniques highlighted in Sections 2.6-2.11.

Objective 6 (Chapters 4-7) will be achieved by sharing the findings of objectives 1-5 with the Eden Rivers Trust, the farming community, and wider UK and international stakeholders (both academic and practitioner).

3 SITE DESCRIPTION

3.1 Introduction to the River Eden catchment and the Leith, Lowther and Petteril sub-catchments

This chapter briefly introduces the Eden catchment, and describes the Leith, Lowther and Petteril sub-catchments in depth as study sites for where this hydrological research is focused. It provides detailed information regarding land-use, demography, watercourses, soils, geologies, and climate, all in relation to hydrology, and is designed to provide a more instructive background than could be produced within Chapters 4-7. The latter sections of this chapter specifically introduce hydrological issues within the Leith, Lowther and Petteril sub-catchments, as well as highlights hydrological studies conducted with these and neighbouring sub-catchments.

3.1.1 Demography, urbanisation and economy

The Eden catchment is located on the western border of England and Scotland within North-West England, within the United Kingdom of Great Britain and Northern Ireland (Figure 3.1). The Eden catchment is almost entirely within the county of Cumbria, with a small section in the north-east crossing into Northumberland. This catchment contains the only city within Cumbria, Carlisle, as well as the towns of Penrith and Appleby-In-Westmorland. The Leith (60.1 km^2), Lowther (156.5 km^2) and Petteril (160.7 km^2) catchments (Figure 3.2) are the studied sub-catchments of the Eden catchment (2217.9 km^2: EA, 2020a).

The vast majority of settlements in the Leith, Lowther and Petteril are small and moderately isolated villages and hamlets, which have historically been expanded

Figure 3.1: The Eden catchment (2217.9 km^2) from the Sheepmount river gauging station at Carlisle, within a UK outline. Contains Ordnance Survey (OS) data.

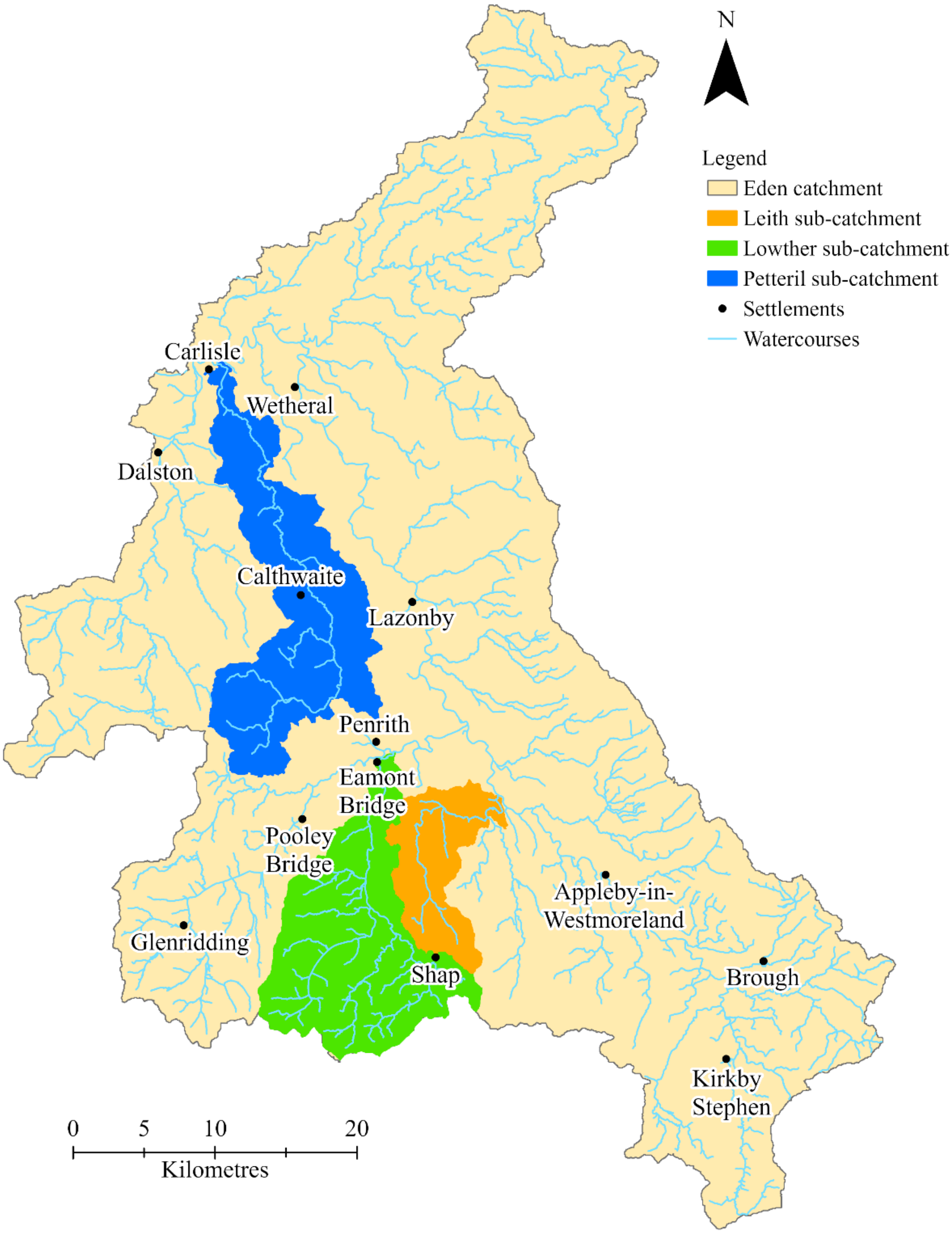

Figure 3.2: The Leith, Lowther and Petteril sub-catchments within the Eden catchment. The major settlements of the Eden catchment are shown, alongside the major watercourses. Contains Environment Agency (EA) and Ordnance Survey (OS) data.

surrounding farmhouses, often as part of the extensive Lowther and Lonsdale estates. Small settlements were also developed around mines and quarries. Many of these villages and farmhouses were settled surrounding the main watercourses due to ease of water abstraction/access, fishing, navigation/transport, waterpower, as well as to access more fertile soils.

This legacy of development in close proximity to the river network has meant that many of these communities are now at substantial risk of flood-damage, and that the watercourses are extremely vulnerable to water-quality degradation. This highly distributed nature of urbanisation has also meant that erecting and maintaining hard-engineered flood defences for each individual locality is not economically feasible, and therefore more distributed interventions (i.e., land-use) are needed.

3.1.2 Land-use

The land-use in the Eden is heavily dominated by improved-pasture, predominantly for sheep, beef and dairy production (Figure 3.3). Arable farming exists within lower regions (see later Figure 3.7 for topography), but is still largely minor in comparison to pastoral farming, and is almost entirely part of mixed-farming systems (both pastoral and arable combined). Common mixed-farming crops include cereals (barley, wheat), oil-yielding crops (oilseed rape), fodder crops (maize, stubble turnips), and energy crops (maize). There is also a considerable amount of (semi-)natural grassland, heathland/moor and peat bog in the upper regions of the Eden, mostly located along the catchment borders. There are additionally patches of urbanised areas, industry and infrastructure, mostly surrounding the major settlements. The two largest waterbodies in the sub-catchment are Ullswater Lake, located between Pooley Bridge and Glenridding, and Haweswater Reservoir, to the south-east of Ullswater.

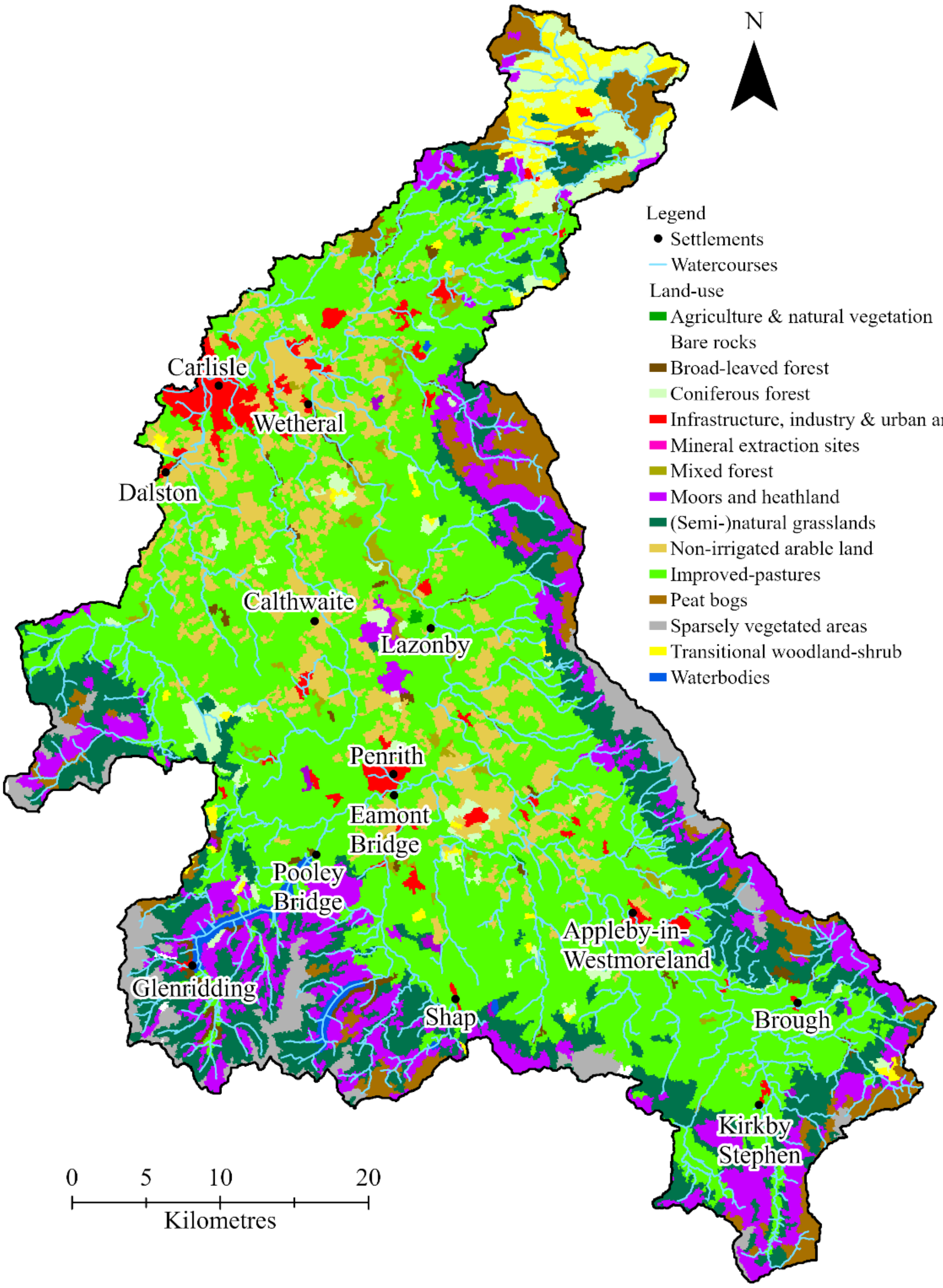

Figure 3.3: The land-use within the Eden catchment in 2018. It is clear that improved-pasture is by far the dominant land-use. Contains Environment Agency (EA), European Environment Agency (EEA) and Ordnance Survey (OS) data.

The land-use in the Leith sub-catchment (Figure 3.4) is almost entirely improved-pasture, and is therefore typical of much of the Eden catchment. There are additionally notable sections of arable land, mixed forest, and coniferous forest, small urban areas, as well as two mineral extraction sites (one of which also created the neighbouring water body). Shap is partially-located within the Leith sub-catchment.

The Lowther sub-catchment largely consists of improved-pastures, (semi-)natural grasslands, moors and heathlands (Figure 3.5). There are additionally considerable areas of peat bogs and sparsely vegetated areas in the more remote and elevated regions of the southern half of the sub-catchment (see later Figure 3.7 for topography). There are also small urbanised areas, mostly in lower sections approaching Penrith. The largest clearly visible water body in the Lowther is Haweswater reservoir, with the much smaller Wet Sleddale reservoir to the south-east.

The land-use in the Petteril is predominantly improved-pasture, with arable land also encompassing a considerable percentage, with the two combining to account for the vast majority of the land-use (Figure 3.6). Other minor land-uses in the area include natural grasslands in the south-west of the sub-catchment, moors and heathland in the east, and urban and infrastructure, mostly in the north surrounding Carlisle.

As evidenced, the land-use in the Leith, Lowther and Petteril sub-catchments, as well as the Eden catchment as a whole, is predominantly improved-pasture. These improved-pastures principally encompass ryegrass, which are used for both grazing and for conservation fodder (predominantly hay/silage) to support livestock during winter. Improved-pastures in the region tend to be intensively managed i.e., contain surface additives, artificial drainage etc. (see Figure 1.1) to improve land productivity, and are therefore often referred to as improved-pasture.

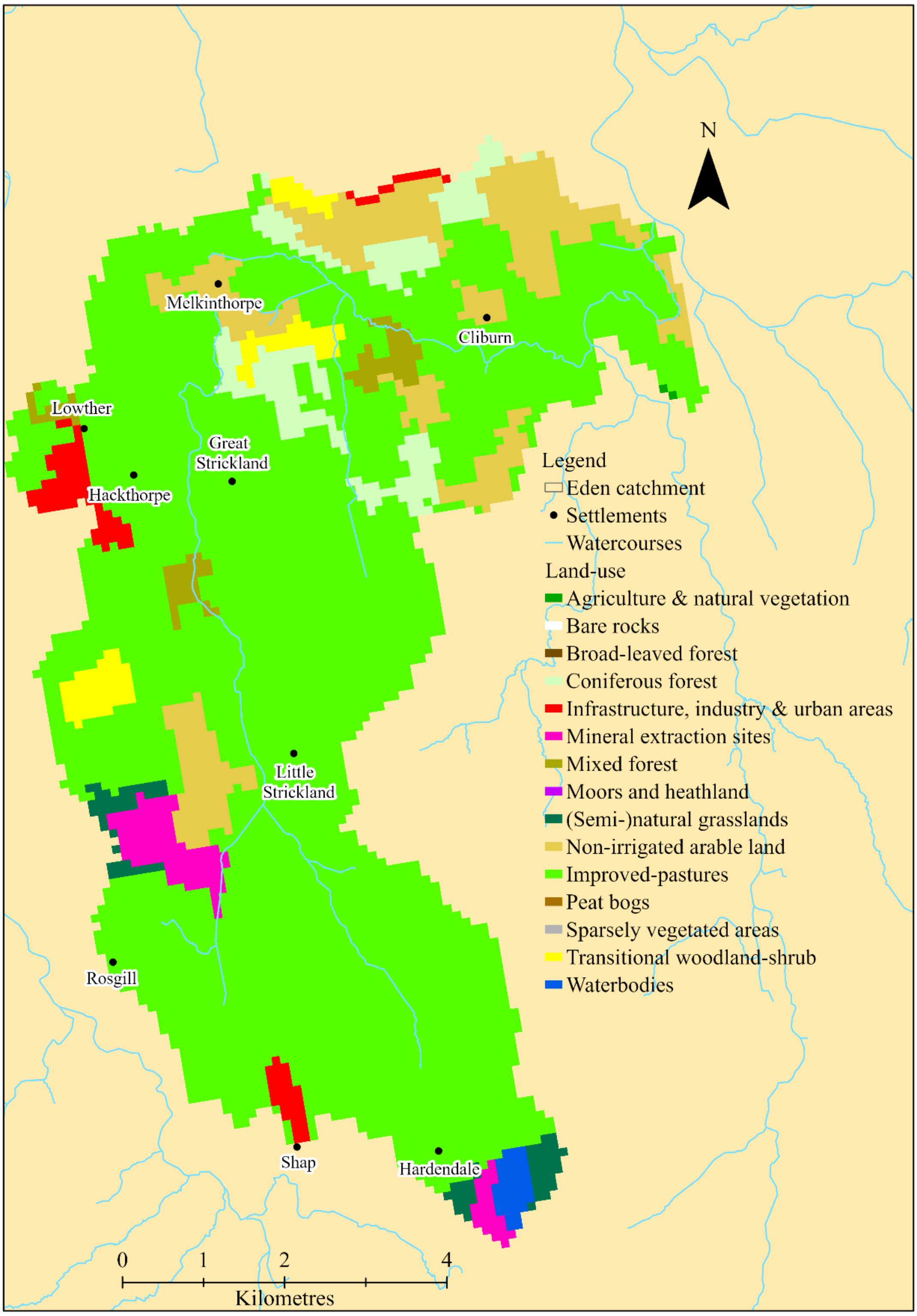

Figure 3.4: The land-use within the Leith sub-catchment in 2018. Improved-pasture is clearly the dominant land-use in the sub-catchment. Contains Environment Agency (EA), European Environment Agency (EEA) and Ordnance Survey (OS) data.

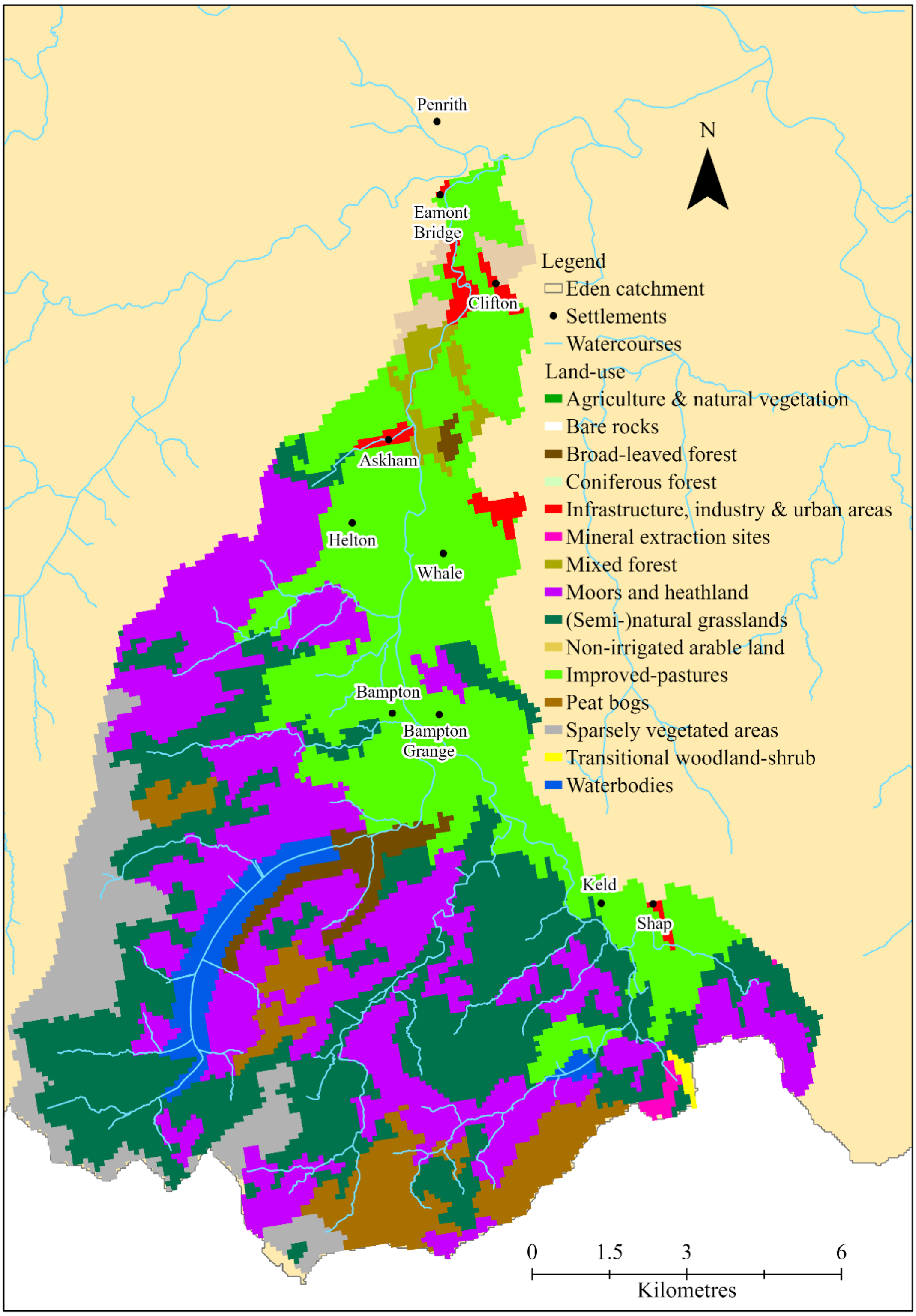

Figure 3.5: The land-use within the Lowther catchment in 2018. Improved-pasture is a major land-use here, although (semi-)natural grasslands, moors and heathland are also substantial land-uses. Contains Environment Agency (EA), European Environment Agency (EEA) and Ordnance Survey (OS) data.

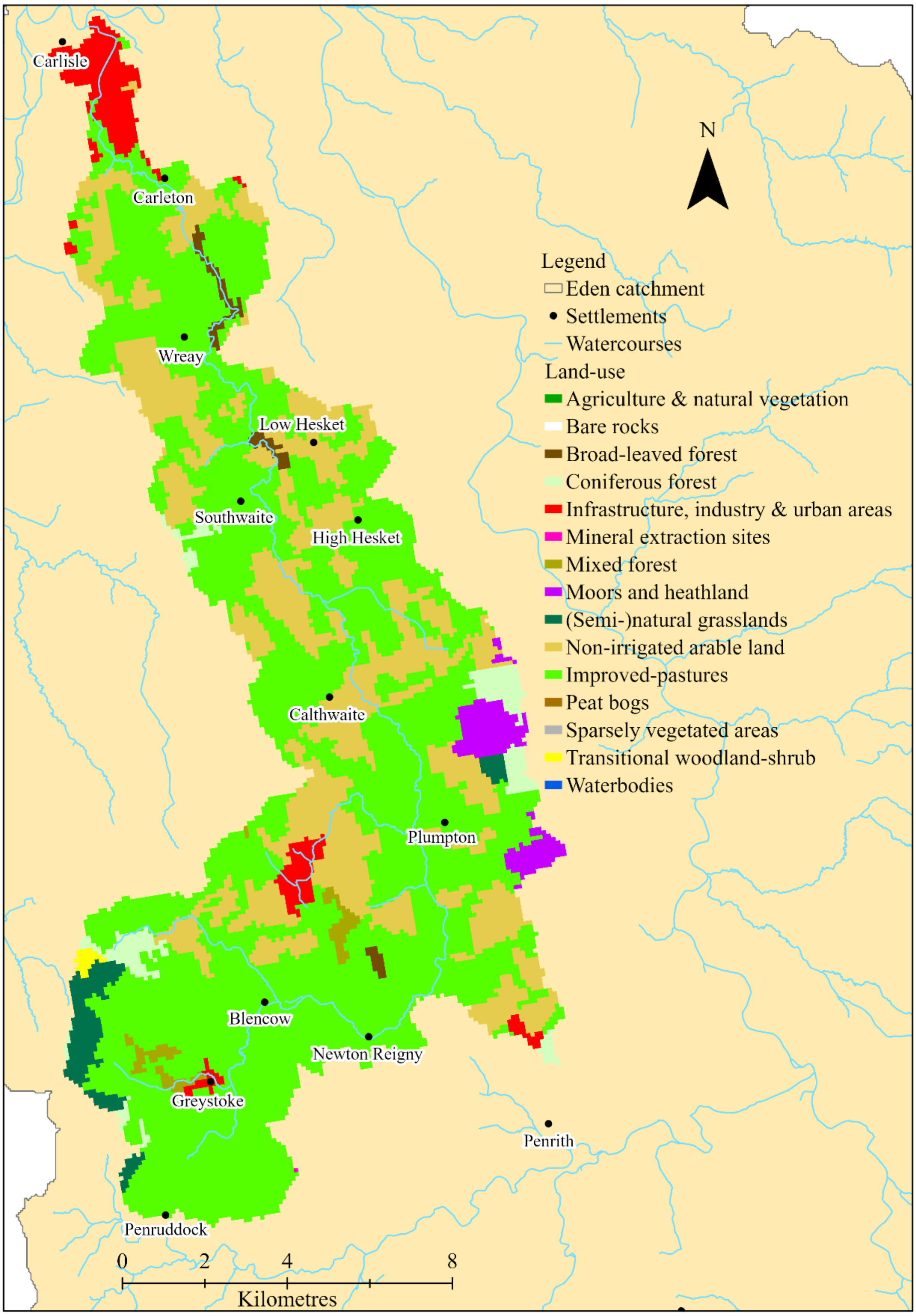

Figure 3.6: The land-use in the Petteril catchment in 2018. Improved-pasture is the dominant land-use here, although non-irrigated arable land also encompasses a considerable percentage. Contains Environment Agency (EA), European Environment Agency (EEA) and Ordnance Survey (OS) data.

Consequentially, investigating interventions that can provide hydrological benefits whilst maintaining the productivity of improved-pastures will be extremely valuable for the Eden and other similar catchments, both in the UK and overseas. As a result, all interventions within the thesis have been specifically selected with this aim in mind.

Many pastoral farms also often graze the natural grasslands, moors and heathlands surrounding their land (this land is often termed semi-natural grassland, or locally as moorland). For example, the Lowther sub-catchment contains considerable amounts of (semi-)natural grassland in the more elevated regions of the south and west such as Bampton Common, Helton Common, Mardale Common and Ralphland Common. These upland ecosystems have fairly minimal management, and are often lightly-moderately grazed through commoners' rights by sheep, although some areas undergo additional management practices. Given that 'rough-grazing' on semi-natural grasslands is a common practice of pastoral farming systems, and that semi-natural grasslands were the original land-use prior to being converted into improved-pasture, these semi-natural grasslands additionally require scientific investigation and comparison to contemporary pasture.

3.1.3 Watercourses

The River Eden is sourced from Mallerstang Common on the Cumbria-Yorkshire border, and travels broadly north-west through Kirkby Stephen and Appleyby-in-Westmoreland, and onto Carlisle where it discharges into the Solway Firth. The River Eden is approximately 145 km in length, and covers 2217.9 km^2. Several rivers within the Eden catchment, including the River Eden itself, flood fairly frequently, causing substantial economic damage (see Section 3.2) throughout the catchment, including within the Leith, Lowther and Petteril sub-catchments (see Section 3.2.1). The Eden catchment is additionally important for water abstraction, as United Utilities extracts approximately 420 megalitres of water d^{-1} from the River Eden and Haweswater Reservoir for use across North-West England, as well as Greater Manchester (ERT, 2020).

The headwaters of the River Leith lie to the north and east of Shap, with the upper reaches of the River Leith often termed Shap Beck. The River Leith heads northwards from its origin through Little Strickland and Great Strickland towards Melkinthorpe, before travelling east through Cliburn and joining the River Lyvennet (which then joins the River Eden approximately 1700 m further to the north-east). The Leith sub-catchment covers 60.1 km^2 and is the smallest of the studied sub-catchments. The largest tributaries of the River Leith are Sanswath Sike and Greenriggs Sike.

The River Lowther is sourced in the fells to the south and south-west of Keld and travels broadly northwards, through Keld and to the west of Shap. The River Lowther then continues northwards through Bampton, Butterwick, Helton and Askham before passing through Eamont Bridge and joining the River Eamont to the south-east of Penrith at Beehive, which then shortly joins the River Eden approximately 5.5 km to the east at Udford. The Lowther sub-catchment covers 156.5 km^2 and includes major tributaries such as Swindale Beck, Haweswater Beck (which drains Haweswater Reservoir), Hawes Beck, and Heltondale Beck. There are also several substantial lakes and reservoirs in the Lowther sub-catchment including Haweswater Reservoir (third largest in the UK), Wet Sleddale Reservoir and Blea Water. The Lowther sub-catchment is approximately 160 % larger than the Leith sub-catchment, but is a very similar size to the Petteril sub-catchment.

The River Petteril originates at Penruddock and Mothersby. The river travels north-east through Greystoke and Little Blencow before travelling east through Newton Reigny and then broadly northwards. The river passes Plumpton, Calthwaite, Southwaite, and into Carlisle, where it directly joins the River Eden. Major tributaries of the River Petteril include Blackrack Beck, the Old Petteril and the North Petteril. The Petteril sub-catchment covers 160.7 km^2.

3.1.4 Topography

The Eden catchment and the studied sub-catchments have a wide range of topographies (Figure 3.7).

The Lowther sub-catchment has the highest topography of the studied sub-catchments, particularly in the south and west that includes some of the highest peaks in the Eden catchment (Figure 3.7). The westernmost sections of the Lowther on the border with the Eamont sub-catchment tend to be the highest, with peaks such as High Street (828 m), High Raise (802 m) and Kidsty Pike (780 m), with several other mountains above 700 m. Several considerable peaks also exist towards the southern peripheries of the Lowther sub-catchment such as Tarn Crag (664 m) and Grey Crag (638 m), as well as notable peaks away from the sub-catchment border to the south of Haweswater such as Branstree/Artlecrag Pike (713 m) and Selside Pike (655 m). Due to this high elevation, improved-pastures are often difficult to establish in the Lowther sub-catchment, and therefore rough grazing of sheep (on semi-natural grasslands, heathlands and moors) is a very common agricultural practice. Lower regions of the Lowther are more economically viable for improved-pasture establishment, and therefore these areas mostly consist of grazed improved-pasture for sheep, beef and occasionally dairy.

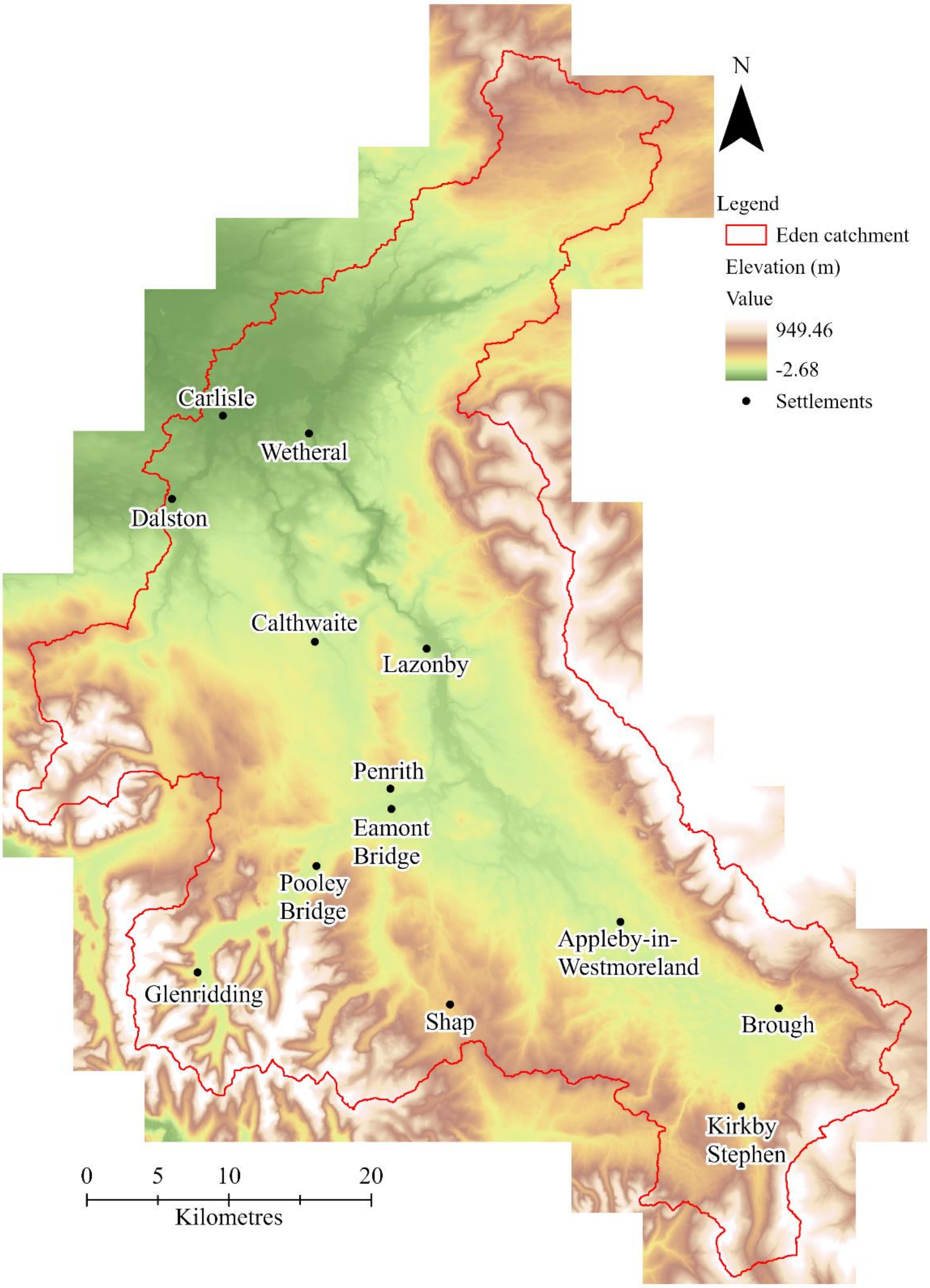

Figure 3.7: The topography of the Eden catchment. The river and lake systems of the Eden catchment are also clearly visible within the topography. Contains EDINA Digimap Ordnance Survey and Ordnance Survey (OS) data.

The Leith sub-catchment does not contain the high elevations found in the Lowther, and therefore consists of predominantly pastoral farming of mostly sheep and beef. Some arable farming is also found in the lowest sections of the Leith (as part of mixed-farming systems). The Petteril sub-catchment is the most low-lying of the three sub-catchments. As a result, this sub-catchment has the mildest climate (see Section 3.1.7), as well as highly fertile soils, and is the most favourable of the three for arable farming (this again is mixed-farming). Due to the gentler topography and lower elevations in the Petteril compared to the Leith and Lowther, farming in this sub-catchment is also considerably more suited towards beef and dairy production rather than sheep.

3.1.5 Soils

The Eden catchment contains a relatively large variety of soils (Figure 3.8), predominantly Stagnosols, Cambisols, Regosols, Histosols, Luvisols and Podzols. Soils in the Leith sub-catchment are predominantly Stangosols north of Little Strickland, and Cambisols south of this point. East of Cliburn where the River Leith meets the River Lyvannet, immediately surrounding the rivers are Gleysols.

The Lowther sub-catchment contains several soil types. In the highest regions these are Histosols, which largely transition into Umbrisols further downslope and then into Stagnosols and Cambisols in lower regions. There is also some Gleysol surrounding the River Lowther around Bampton, also with minor sections of nearby Leptosol to the east of Bampton. Podzols are also present in the west of the sub-catchment, south-west of Helton.

Much like the Leith sub-catchment, the Petteril sub-catchment is primarily Stagnosols and Cambisols. There are also notable areas of Podzols on the eastern edge of the

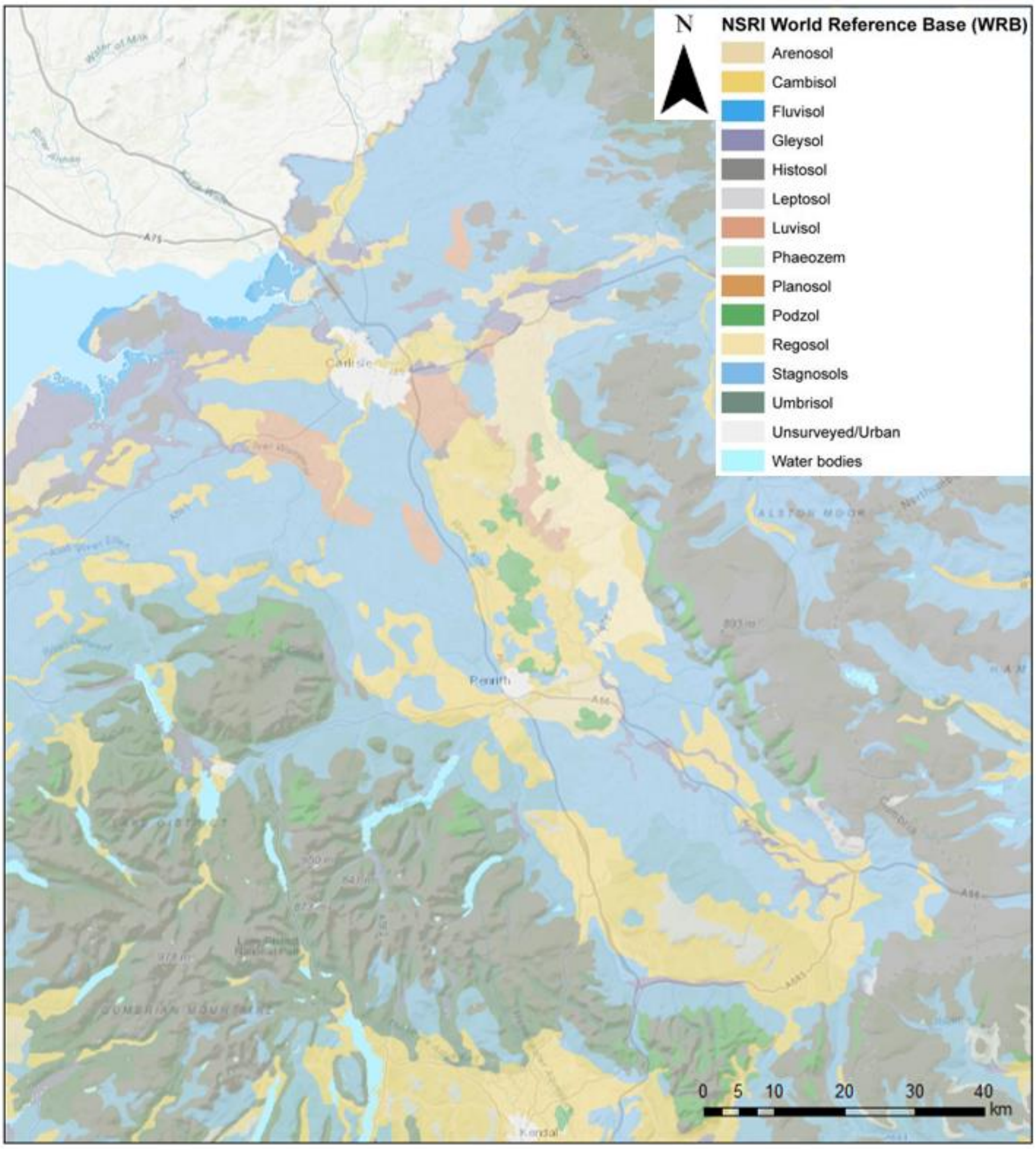

Figure 3.8: A map of the soil types within the Eden catchment. Soil type data is provided by and reproduced with the permission of the British Geological Society (BGS) and the United Kingdom Soil Observatory (UKSO).

sub-catchment, particularly east of Plumpton. A considerable area of Luvisol also exists extending from Carlisle to Wreay, as well as a section of Luvisol north-west of Calthwaite.

All experiments within this thesis were conducted upon Stagnosols due to their consistent and widespread presence in the studied sub-catchments and the wider Eden catchment (as well as the UK as a whole). Stagnosols are additionally considered highly susceptibility to poor drainage and OFRSSF, and therefore were ideal conditions to examine changes to overland flow and rapid shallow-subsurface flow pathways (Jarvis et al., 1984). Conducting experiments within a single soil type also improves comparability between interventions.

3.1.6 Geology

3.1.6.1 Superficial geology

The superficial geology of the Eden catchment as a whole is predominantly till (Figure 3.9), which is also known as glacial drift (or simply drift). Within all three studied sub-catchments, till of variable thickness is the dominant superficial geology. Within the headwater of the Lowther sub-catchment, there is a considerable amount of peat. In the lower sections of both the Lowther and Petteril, notable amounts of alluvium are present surrounding the River Lowther and River Petteril, respectively.

Tills are widespread across the north of the UK as well as much of Northern Europe and North America. Due to the widespread presence of till locally and nationally, all field sites were specifically selected due to containing till. Soils containing till are often slowly permeable (Hankin et al., 2018), and are therefore considered to be highly susceptible to OFRSSF, combining well with the Stagnosol criterion. Till also generally supports the formation of Stagnosol (and similar Gleysol) soils, supporting the multiple criteria for each field-site.

3.1.6.2 Bedrock geology

The Eden catchment consists of an extremely wide range of bedrock geologies that have formed through various mechanisms and over numerous timeframes (Figure 3.10). The bedrock geology of the Leith sub-catchment is predominantly limestone

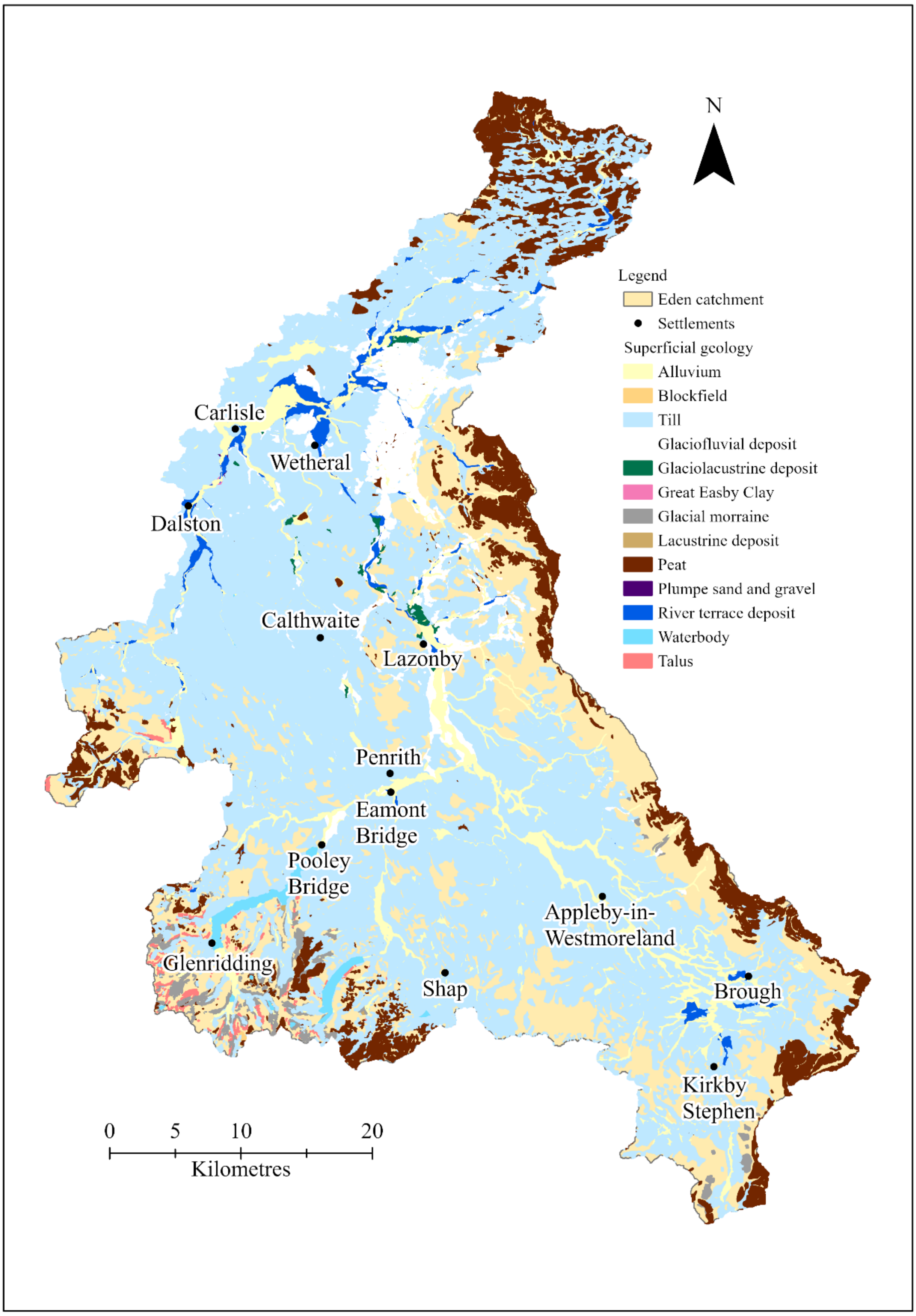

Figure 3.9: The superficial geology of the Eden catchment. Glacial till (also called drift) is clearly the dominant superficial geology. Contains EDINA Digimap Geology, British Geographical Society (BGS) and Ordnance Survey (OS) data.

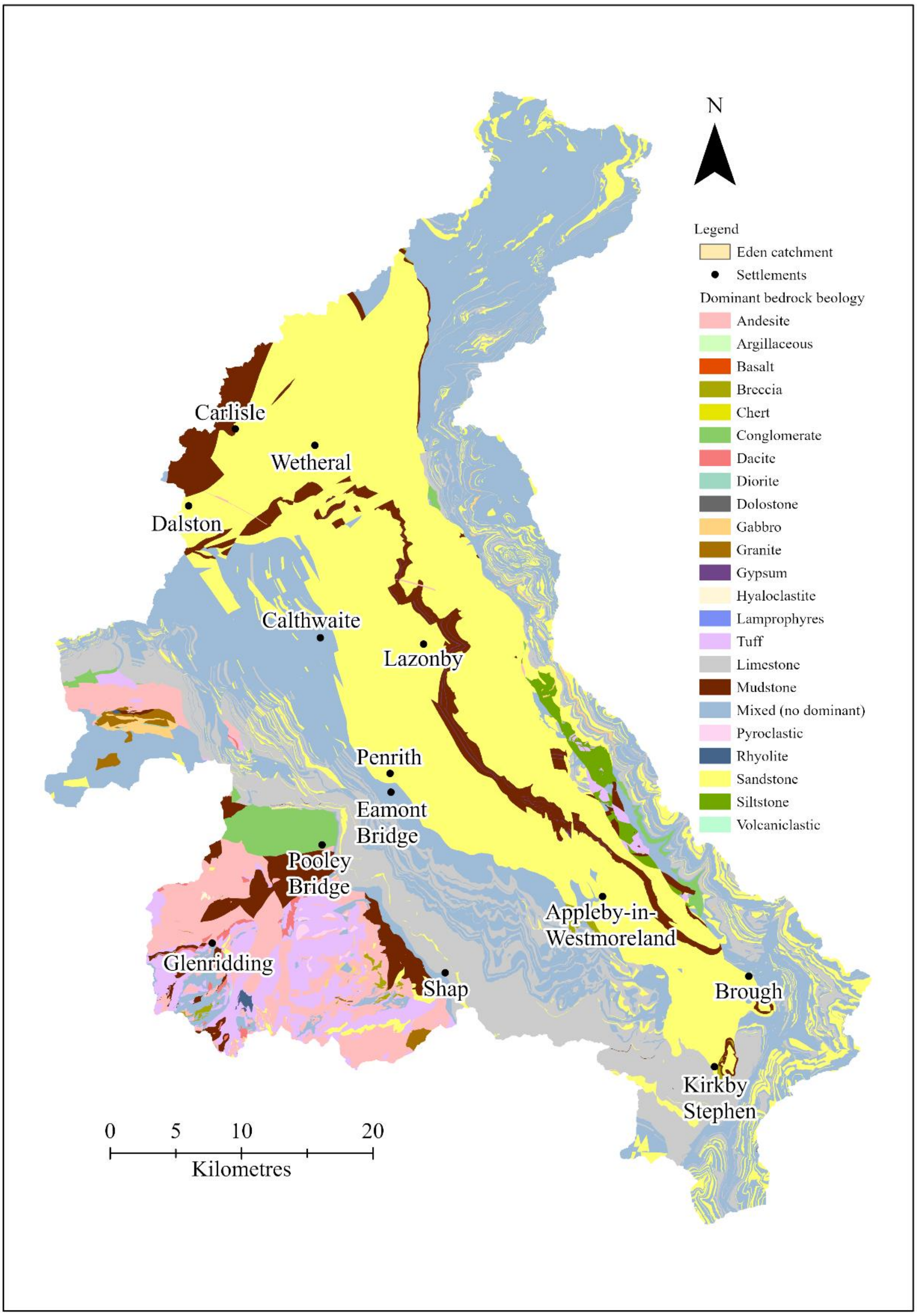

Figure 3.10: The bedrock geology of the Eden catchment. Note that the mixed geology category encompasses a wide variety of geologies. Contains EDINA Digimap Geology, British Geographical Society (BGS) and Ordnance Survey (OS) data.

and mixed geology (mixed being defined as more than one dominant local geology – often being very complex), with some sandstone in the north-east. Within the Lowther sub-catchment there is a range of bedrock geologies, primarily consisting of limestone, andesite, mudstone, tuff and mixed geology. The bedrock geology of the Petteril sub-catchment is almost entirely split between limestone, sandstone, and sections of mixed geology. Bedrock geology can affect the overall hydrological functioning within a catchment; however, bedrock geology is believed to have minimal impact upon OFRSSF, particularly when glacial-till is present (which largely disconnects the surface from the bedrock geology), and therefore no strict criteria was enforced upon bedrock geology when selecting field-sites.

3.1.7 Climate

The climate of the United Kingdom is wet temperate, experiencing both mild winters and mild summers. Cumbria as a whole is the wettest county in England, and one of the wettest counties in the United Kingdom. Of the 30 largest monthly record rainfalls in England (>1000 mm), all thirty are from within Cumbria (Burt, 2016). As a result, Cumbria often experiences hydrological hazards, particularly flooding, but also water-quality issues. A 25.5-yr precipitation time-series (1990–2018, excluding July 1993–March 1997) from within the Petteril catchment at Skelton (54°42′59″ N, 2°52′38″ W: Figure 3.11), gives average annual precipitation (Figure 3.12), average seasonal precipitation (Figure 3.13) and average daily precipitation (Figure 3.14) for the studied sub-catchments. There is additionally growing concern regarding the resilience of the region to drought, particularly due to the regions important in water supply for major settlements. Both anthropogenic climate change, land-use change, as well as natural climate variability may be exacerbating such hydrological hazards.

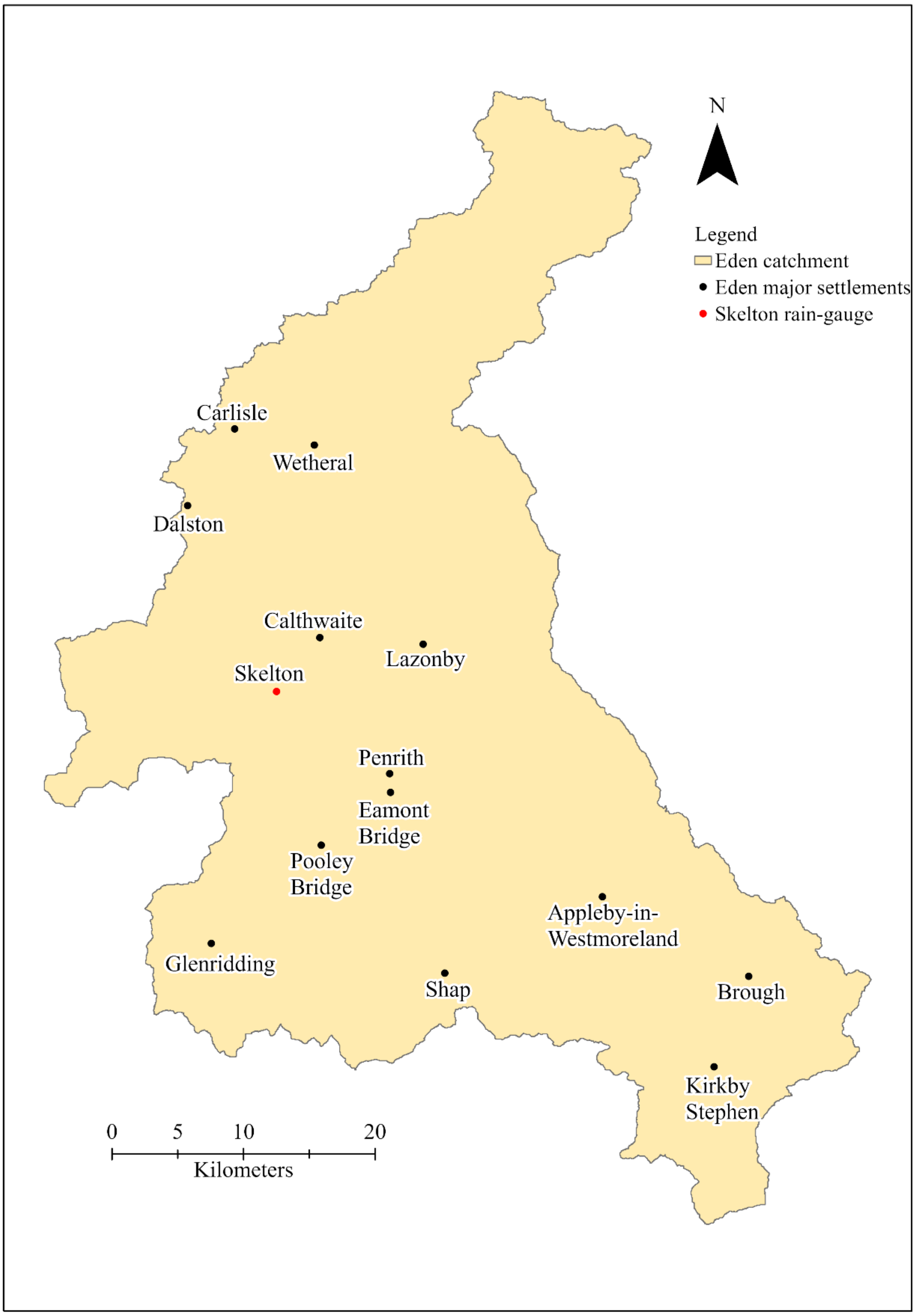

Figure 3.11: The location of the Skelton rain-gauge within the wider Eden catchment.

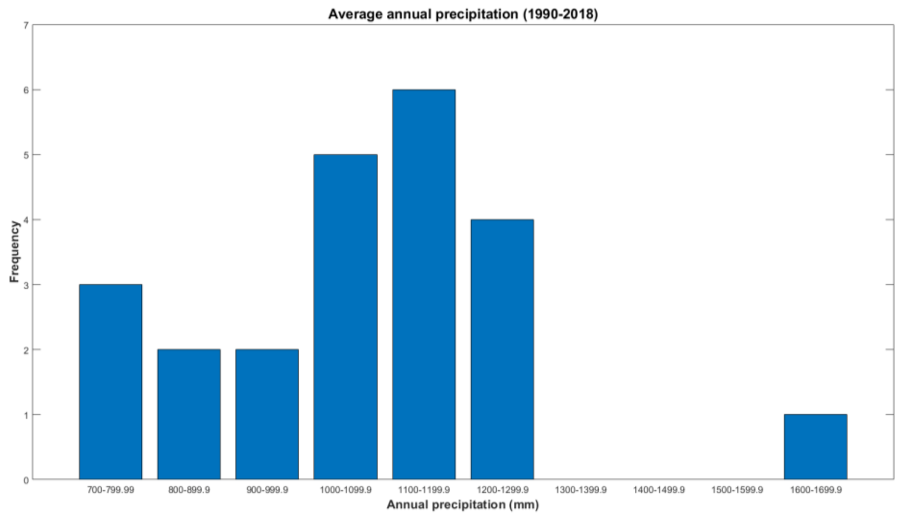

Figure 3.12: The average annual precipitation from Skelton in the Petteril sub-catchment.

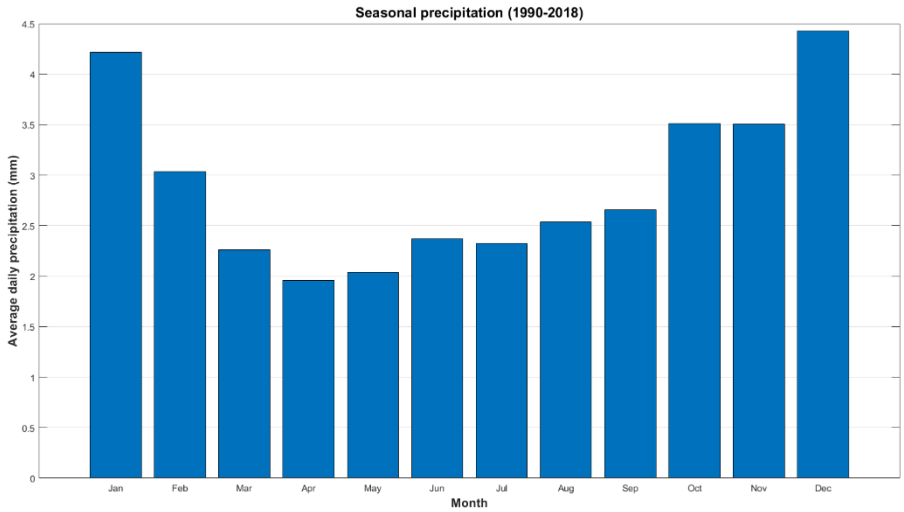

Figure 3.13: The average monthly precipitation from Skelton in the Petteril sub-catchment.

3.2 Hydrological issues in the Eden catchment and the Leith, Lowther and Petteril sub-catchments

3.2.1 Flood-risk in the Eden catchment and the Leith, Lowther and Petteril sub-catchments

Flooding is the major hydrological issue within the Eden catchment and is economically the costliest environmental hazard. Severe flooding has occurred from

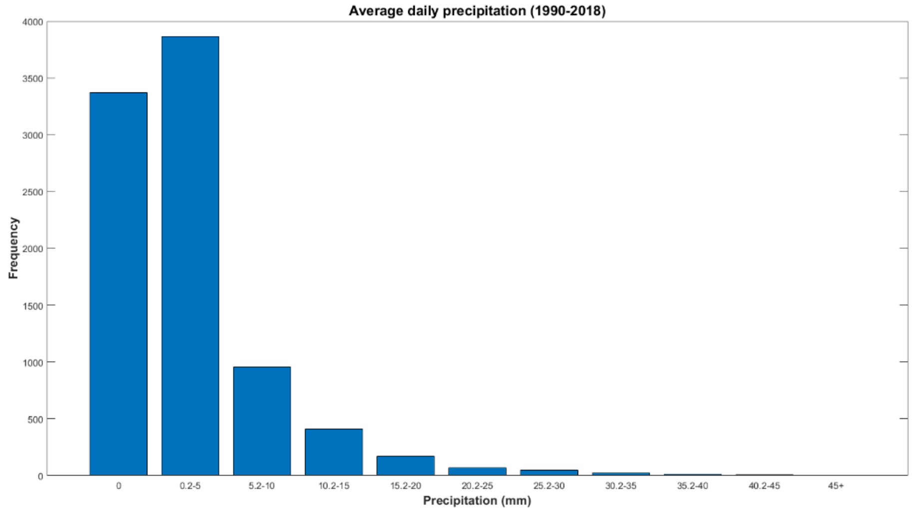

Figure 3.14: The average daily precipitation from Skelton in the Petteril sub-catchment. The five days recording >45 mm of precipitation recorded 54 mm, 55.2 mm, 69.4 mm, 76.8 mm and 135 mm.

frontal rainfall in the Eden catchment at least seven times since c. 1600: 2nd Jan 1752, 13th October 1771, 1st – 3rd Feb 1822, 14th October 1829, 23rd – 24th March 1968, 8th – 9th Jan 2005, December 2015 – January 2016 (Watkins and Whyte, 2008; 2009; Szönyi et al., 2016). Severe flooding from convective events have also occurred in the Eden catchment in 1689, 24th July 1888, 9th August 1894, and 18th June 1930 (Watkins and Whyte, 2008). Areas with the largest number of properties at risk from riverine and pluvial flooding both immediately downstream and within the Leith, Lowther and Petteril sub-catchments include Carlisle (90 % of the at-risk homes in the Eden catchment are here), Penrith, Eamont Bridge and Shap (EA, 2009; Szönyi et al., 2016).

The costliest of these floods has been the recent January 2005 floods at £450 million nationally (contemporary corrected), and the 2015/16 winter floods at £1.6 billion nationally (EA, 2018; Watkins and Whyte, 2009). During Storm Desmond in December 2015, the River Eden equalled the largest river discharge ever to be recorded within England (Burns, 2016; Szönyi et al., 2016), although this discharge estimate is based upon extrapolation of hydraulic rating curves and may be subject to considerable uncertainty. These recent disasters have raised the priority of mitigating both fluvial and pluvial flood-risk in the region (EA, 2009). Coastal flooding is an additional concern for downstream settlements to the west of Carlisle, however neither the Leith, Lowther nor Petteril contain coastlines. Given the distributed nature of the settlements located in the studied sub-catchments and the Eden catchment as a whole, alongside the large amount of grassland, particularly improved-pasture but also semi-natural grassland, this area is therefore a suitable location to conduct hydrological research into grassland and improved-pasture hydrology.

3.2.2 Water-quality in the Eden catchment and the Leith, Lowther and Petteril sub-catchments

Water-quality is another hydrological issue in the Eden catchment, and receives particular attention from the Eden Rivers Trust due to its detrimental effects on biodiversity and overall amenity use. Particular physico-chemical parameters of importance in the Leith, Lowther and Petteril sub-catchments are nitrate, phosphate, ammonia, copper, mercury, colour and temperature. Based on the latest 2019 Environmental Agency detailed surveys, all the Leith, Lowther and Petteril (all nine subdivisions) have moderate overall waterbody status (Figure 3.15), although all nine subdivisions fail the chemical water body classification due to excessive polybrominated diphenyl ethers and excessive mercury and mercurous compounds (Figure 3.16). The ecological status of the whole Leith sub-catchment is good (Figure 3.17). The most headwater sections of the Petteril and Lowther sub-catchments are good, although further downstream the ecological status deteriorates to moderate (see Figure 3.17: EA, 2020a).

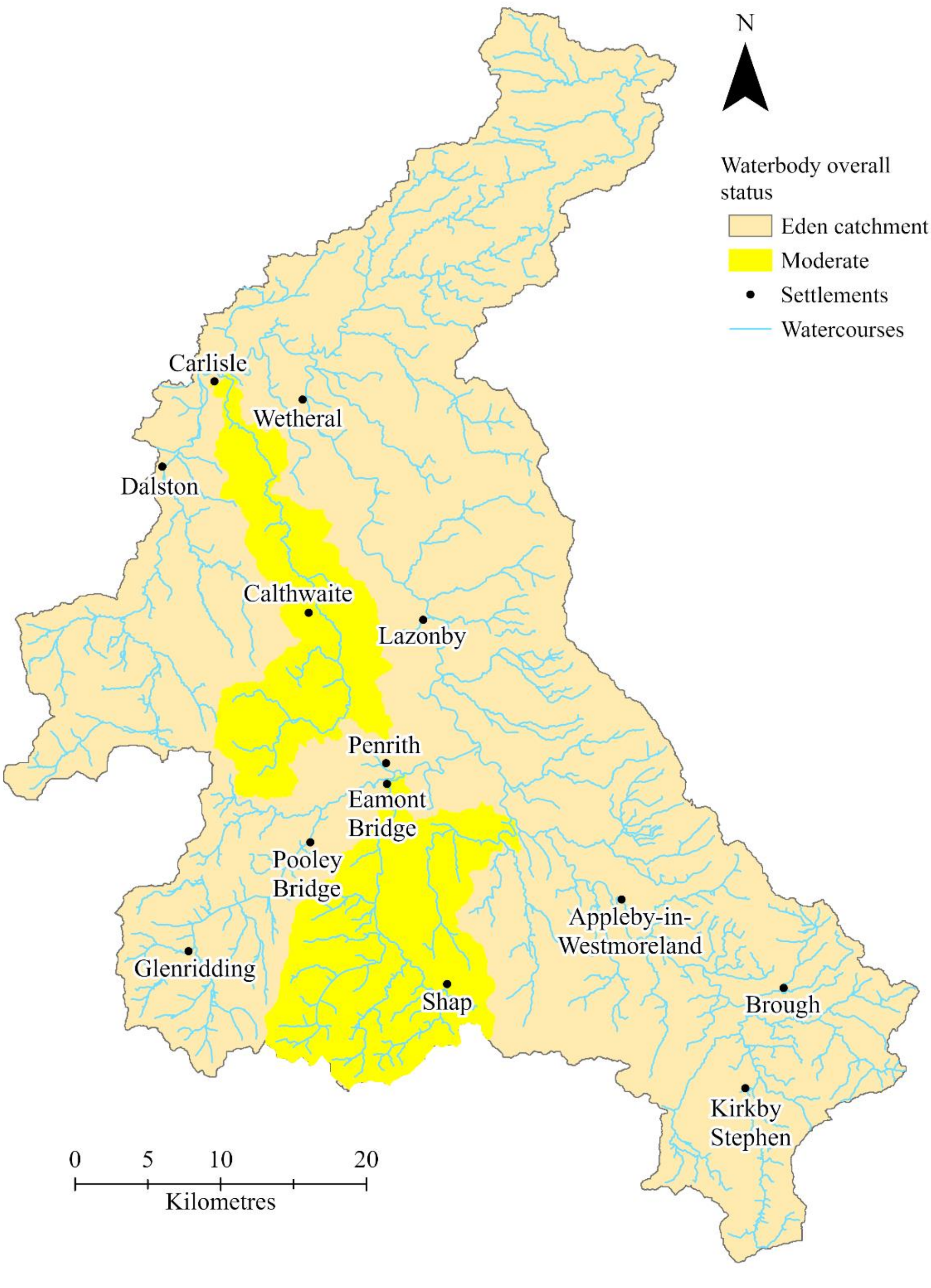

Figure 3.15: The waterbody overall status within the Eden catchment for the Leith, Lowther and Petteril sub-catchments. All studied sub-catchments have moderate overall status. Contains Environment Agency (EA) and Ordnance Survey (OS) data.

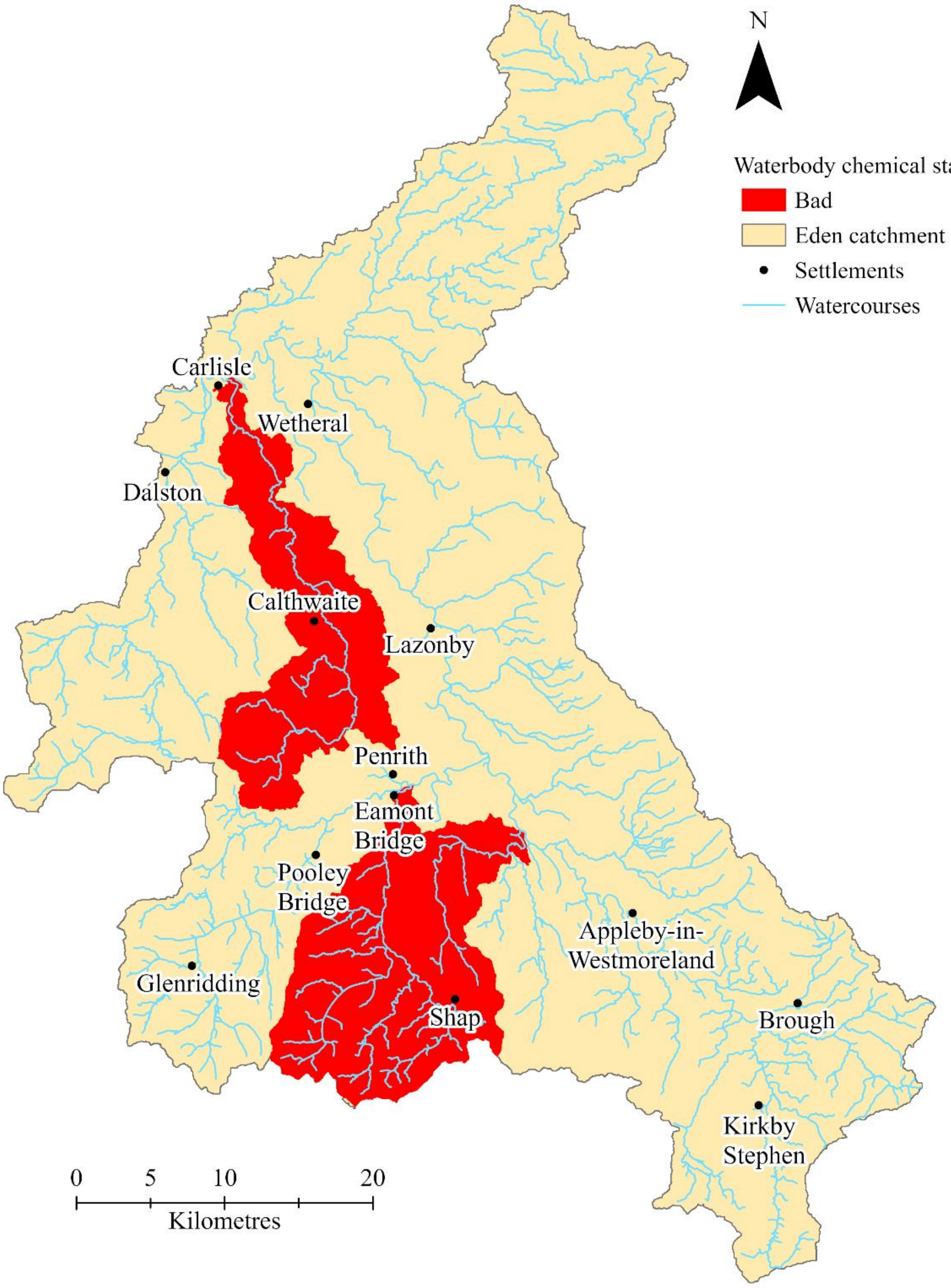

Figure 3.16: The waterbody chemical status within the Eden catchment for the Leith, Lowther and Petteril sub-catchments. All studied sub-catchments have bad chemical status due to elevated levels of mercury/mercurous compounds and polybrominated diphenyl ethers. Contains Environment Agency (EA) and Ordnance Survey (OS) data.

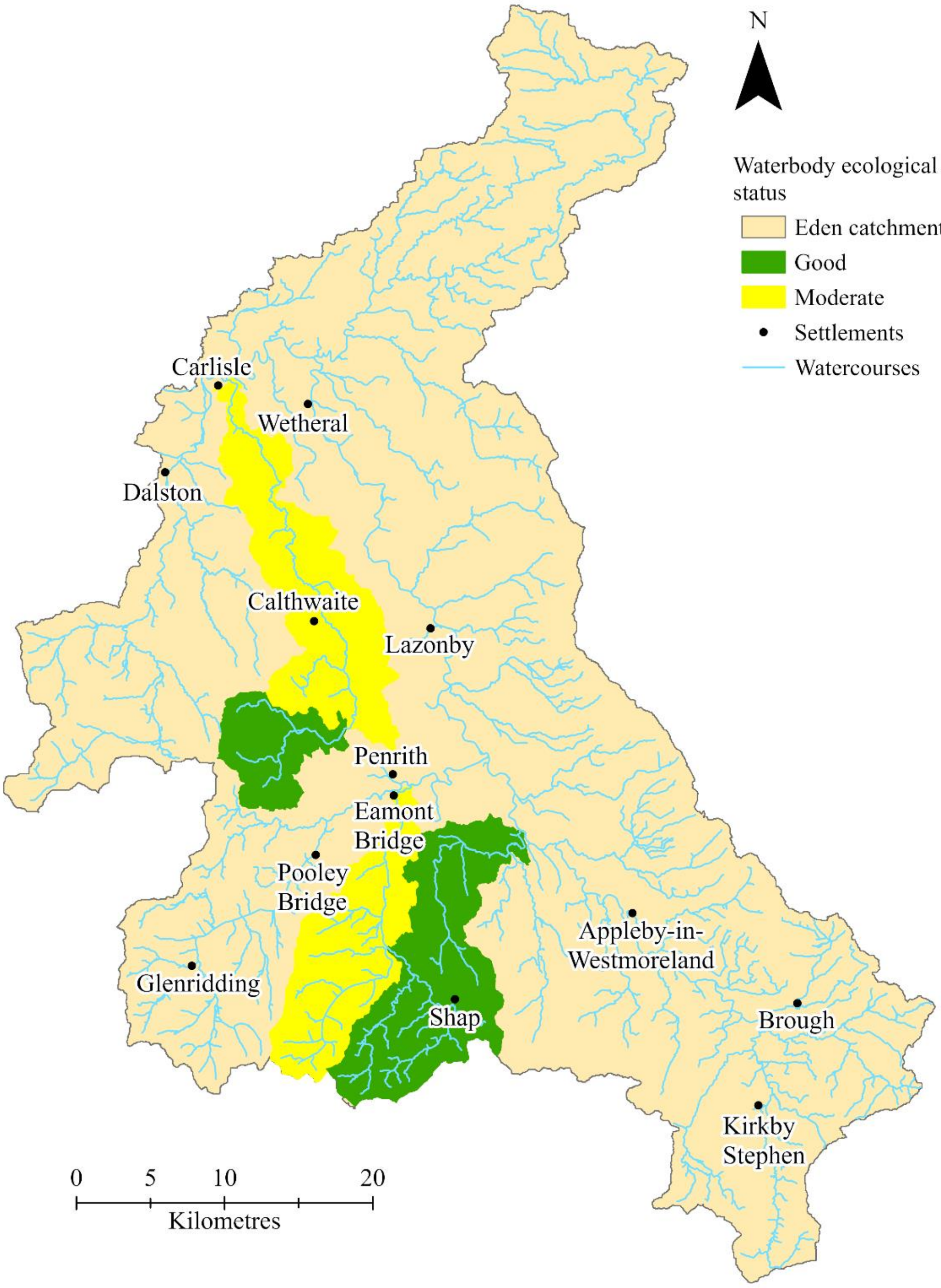

Figure 3.17: The waterbody ecological status within the Eden catchment for the Leith, Lowther and Petteril sub-catchments. The ecological status ranges from good to moderate. Contains Environment Agency (EA) and Ordnance Survey (OS) data.

The entire Leith and Lowther sub-catchments, and a small area of the Petteril sub-catchment, are within drinking water safeguard zones for surface water-quality, with particular emphasis placed on colour and pesticides (especially dichlorophenoxyacetic acid: 2-4D: EA, 2020b). The Petteril sub-catchment also contains several nitrate vulnerable zones, and also has a small area within a drinking water safeguard zone for groundwater in relation to nitrate contamination (EA, 2020b). Although infrequent, outbreaks of water-borne disease occur within the Eden catchment. These are predominantly cryptosporidium and giardia outbreaks, with North-West England usually amongst the worst affected regions of England for both infections each year (Goh et al., 2004; Sturdee et al., 2007; PHE, 2013; 2018).

Severe point-source pollution events are extremely infrequent in the region although these have occurred previously, such as the accidental release of ammonium fertiliser into the Eden mainstem in 1993 (Shaw et al., 2011), or the discharge of slurry from a dairy unit into Skitwath Beck (Eamont catchment) in September 2015, both of which resulted in substantial fish and invertebrate kills (ERT, 2016). Invasive aquatic plant and animal species transported by the catchment waterbodies (and riparian zones) are a growing concern within the area (ERT, 2016). Recent years have seen progressive improvements to water-quality throughout the Eden catchment, largely due to proactive work from the Eden Rivers Trust, increased community engagement, and more frequent adoption of mitigation measures to safeguard water-quality (EA, 2020a; EAA, 2020).

3.2.3 Drought-risk in the Eden catchment and the Leith, Lowther and Petteril sub-catchments

Drought is the final considerable hydrological hazard in the Eden catchment discussed within the thesis. Numerous and severe droughts in the region have been recorded on numerous occasions since the 18th century (Marsh et al., 2007; Murphy et al., 2020). Drought is a growing concern in the region, largely due to its detrimental effects on agricultural output, alongside causing issues with drinking water supplies as several large reservoirs are present within the sub-catchments. It is possible that droughts in the region will worsen in the future because of climate change and natural climate variability.

3.3 Prior studies in the Eden catchment

Previous hydrological projects have been conducted within the Eden catchment, the largest of which have been the Eden Demonstration Test Catchment (Eden DTC) project and the Catchment Hydrology and Sustainable Management (CHASM) project. The Eden DTC was a DEFRA (Department for Environment, Food and Rural Affairs) Demonstration Test Catchment, with the Eden catchment paired with the Hampshire Avon catchment, Hampshire, UK, and the Wensum catchment, Norfolk, UK. The goal of the Eden DTC was to cost effectively reduce the impact of agricultural diffuse pollution on ecological functioning whilst maintaining food security through the implementation of multiple on-farm measures (e.g., Owen et al., 2012; Adams et al., 2018; Reaney et al., 2019; Snell et al., 2019). The goal of the CHASM project was to support new predictive methodologies for sustainable catchment management across scales in regards to hydrology and ecology, primarily in either poorly gauged or ungauged catchments (O’Connell et al., 2007b; Mills and

Bathurst, 2015; Wilkinson and Bathurst, 2018). There have also been several other hydrological studies within the Eden catchment (e.g., Horritt et al., 2010; Binley et al., 2013; Pattison et al., 2014; Bond et al., 2020; amongst many others). There are also numerous non-hydrological studies in the Eden catchment, such as in biological and ecological fields (Everard and Denny, 1984; Hale and Lurz, 2003; Stevenson et al., 2013; Mayhew et al., 2015), sedimentology (Grayson and Plater, 2008), geology (Musson and Henni, 2002); archaeology and history (Usai, 2001; McCartney et al., 2015), sociology (Convery and Bailey, 2008; Chang, 2010), and numerous further disciplines.

4 SEMI-NATURAL GRASSLANDS

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4.1 Brief introduction to paper

Historically, much of the landscape in the Eden catchment was semi-natural grassland, often described locally as open-moor(land). This environment remains relatively unmanaged and is not used for intensive agriculture, maintaining a wide range of flora and fauna, and consists of large sections of natural, waterlogged and highly organic soils (some classed as peatland or bog, see Section 3.1.2). Within many upland farming systems in the Eden catchment, sections of semi-natural grassland closest to human settlements are grazed by sheep (and occasionally cattle) under commoners' rights.

Semi-natural grasslands have been converted into agricultural grasslands in the region over millennia; with the most rapid expansion during the Enclosure Acts of the 18th and 19th century. Understanding how this land conversion and subsequent management from a quasi-natural ecosystem into an agriculturally-intensive ecosystem has altered the dynamics of surface water can infer how agriculture has influenced the hydrological cycle from its relatively natural baseline conditions.

Maintaining and/or incorporating these semi-natural soils and vegetation within modern farming systems may therefore be seen as a grassland intervention, which could provide hydrological benefits.

Objective 1) Quantify the effect of a grazed semi-natural grassland as opposed to a grazed agriculturally-intensive improved-pasture and silage field on spatio-temporal soil volumetric wetness (θ_v), from extremes of drought to fully saturated conditions. Soil volumetric wetness will be correlated to local topography and local vegetation to assess their influence. Soil volumetric wetness will be used to infer the sources and amounts of saturation-excess overland flow during flood events, as well as the extent of drying during drought events.

4.2 A statistical comparison of spatio-temporal surface moisture patterns beneath a semi-natural grassland and permanent pasture: From drought to saturation.

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4.3 Abstract

Some 60 % of the agricultural land in the UK is grassland. This is mostly located in the wetter uplands of the west and north, with the majority intensively managed as permanent pasture. Despite its extent, there is a lack of knowledge regarding how agricultural practices have altered the hydrological behaviour of the underlying soils relative to the adjacent moorland covered by semi-natural grassland. Near-surface soil moisture content is an expression of the changes that have taken place and is critical in the generation of flood-producing overland flows. This study aims to develop a

pioneering paired-plot approach, producing 1536 moisture measurements at each of the monitoring dates throughout the studied year, that were subsequently analysed by a comparison of frequency distributions, visual-cum-geostatistical investigation of spatial patterns and linear mixed-effects regression modelling.

The analysis demonstrated that the practices taking place in the improved-pasture (ploughing, re-seeding and drainage) reduced the natural diversity in moisture patterns. Compared to adjacent moorland, the topsoil dried much faster in spring with the effects offset by moisture from slurry applications in summer. With the onset of autumn rains, these applications then made the topsoil wetter than the moorland, heightening the likelihood of flood-producing overland flow. During the sampling within one such storm-event, the adjacent moorland was almost as wet as the improved-pasture with both visibly generating overland flow. These contrasts in soil moisture were statistically significant throughout. Further, they highlight the need to scale-up the monitoring with numerous plot-pairs to see if the observed highly dynamic, contrasting behaviour is present at the landscape-scale. Such research is fundamental to designing appropriate agricultural interventions to deliver sustainable sward production for livestock or methods of mitigating overland flow incidence that would otherwise heighten flood-risk or threaten water-quality in rivers.

4.4 Introduction

Grassland accounts for 60 % of the total UK agricultural area, which is proportioned almost equally at 55 % as agriculturally-improved permanent pasture and 45 % rough grazing on semi-natural grasslands (DEFRA, 2019). Both permanent pasture and semi-natural grassland (often referred to as open-moorland or ‘unimproved’ pasture) encompasses a large percentage of the UK uplands, providing sustenance to grazing livestock alongside other ecosystem services (Lamarque et al., 2011; Bengtsson et al., 2019; Hayhow et al., 2019; Morse, 2019). Historically, semi-natural grassland was converted into permanent pasture during the eighteenth and nineteenth century to increase agricultural output (Kain et al., 2004; Whyte, 2006). Gilman (2002), O'Connell et al. (2004), Holden et al. (2007) and Wheater et al. (2008) note the lack of research into the hydrological functioning of semi-natural grassland, with Gilman (2002) directly stating that 'there is little or no experimental evidence to support theoretical studies' relating to the effects of semi-natural grasslands on flood-risk. Consequentially, it remains unknown how converting upland semi-natural grassland into permanent pasture has altered soil moisture regimes that affect flood generation processes and drought-resilience.

Very few upland UK studies have compared semi-natural grasslands to permanent pasture, with research operating at coarse-scale resolution without conducting paired-plot analysis, thus, observations and knowledge of hydrological processes at the plot-hillslope scale is lacking. Orr and Carling (2006) compared catchment-scale flood-risk within North-West England, commenting that transitioning from heather (*Calluna vulgaris*) or scrub vegetation to drained pasture could increase downstream flood-risk. Marshall et al. (2006),Wheater et al. (2008) and McIntyre and Marshall (2010) similarly conclude, through hydrograph assessment, that a semi-natural grassland in mid-Wales, UK (Pontbren experimental site), had a damped flood response compared to improved-pasture. Ockenden and Chappell (2008) noted that a measured plot of semi-natural grassland was significantly drier than a nearby improved-pasture in the River Eden catchment (Cumbria, UK). Gilman (2002) is the only UK study to compare permanent pasture to semi-natural grassland as a primary research objective. The study concluded that pasture reversion could reduce River Severn peak flows by

0.5% – 2 %, and smaller channel peak flows by 2% – 4 %, although acknowledged the lack of supporting studies with which to justify model values used to simulate changes. These studies emphasise a considerable research gap, justifying the need for a hydrological comparison of permanent pasture and semi-natural grassland in an upland UK landscape (Wheater et al., 2008). Indeed, there is a global dearth of studies relating to how livestock production alters the hydrological functioning of natural soils (Magliano et al., 2019).

A significant component of the catchment water budget is the dynamic variation in θ_V, which is the total volume of water present between soil particles divided by the total undisturbed soil volume. Soil θ_V is crucial in regulating hydrological system functioning (Gilman, 2002; Schulte et al., 2012). Antecedent θ_V preceding storm events can dictate rainfall-runoff responses by changing the likelihood of IOF and SOF generation, even from highly permeable soils, so elevating both flood-risk and water-quality degradation (Dunne and Black, 1970; Entekhabi et al., 1996; Marshall et al., 2009; Minet et al., 2011). The precise spatial arrangement of θ_V is fundamental in determining a rainfall-runoff response, as purely using θ_V probability distributions does not capture spatial-structures and therefore contributory area connectivity (Bonell and Williams, 1986; Grayson and Blöschl, 2000; Zehe and Blöschl, 2004; Minet et al., 2011; Meijles et al., 2015). Soil θ_V is equally important during drought, determining water stress for agricultural crops, wildfire frequency etc. (Albertson et al., 2009; Schulte et al., 2012). The spatial arrangement of θ_V during dry conditions can allow targeted irrigation (including slurry application) during water stress. An understanding of differences in spatio-temporal θ_V between semi-natural grassland and permanent pasture would, therefore, provide insights into the hydrological functioning of each land-use, and, therefore, infer how land conversion (or restoration) affects hydrological responses.

The aim of this study is to compare the spatio-temporal dynamics of surface soil volumetric wetness in an area of semi-natural grassland with an adjacent area that has been converted and managed as permanent pasture in the UK uplands. Both the reference and converted plots are adjacent to minimise natural differences. The methodological development aspect of the research aims to quantify the spatial variability of θ_V at the plot-scale, which demands intense measurements. The research specifically measures each plot temporally, rather than the replication of plot-pairs in the landscape, which is beyond the scope of this study. The plot comparison is conducted over a 6-month period (including drought and fully-saturated conditions), to assess non-stationarity in the differences. A high-resolution (1 m^2) soil volumetric wetness grid (1536 m^2) was needed to capture fine-scale spatio-temporal soil moisture variability, which is then compared with localised factors such as land-use, vegetation, season, and elevation, to assess their impact.

Thus, the detailed research objectives are:

I. To develop a statistically robust methodology for the quantification of soil moisture differences between an example 768 m^2 area of semi-natural grassland, and an adjacent area of the same size managed as permanent pasture.

II. To statistically contrast the soil volumetric wetness probability density functions between a permanent pasture and a semi-natural grassland, to quantify soil moisture differences.

III. To compare geostatistically the spatial-structure of soil volumetric wetness between a permanent pasture and a semi-natural grassland, to assess spatially-dependent soil moisture patterns.

IV. To determine which factors significantly influence soil volumetric wetness in the contrasting, adjacent land-uses, to highlight any potential predictors of soil volumetric wetness at this particular locality.

4.5 Materials and methodology

4.5.1 Study site

Measurements were taken within a permanent pasture and a bordering semi-natural grassland located within the Lowther catchment, 3 km north-west of Shap (Cumbria, UK), between April 2018 and May 2019. The improved-pasture (centre 54° 32' 26" N, 2° 43' 44" W) and semi-natural grassland (centre 54° 32' 26" N, 2° 43' 42" W) are immediately adjacent and are separated by a 1.3 m dry-stone wall (Figure 4.1). The dry-stone wall was likely raised between 1838-1855 based on surrounding Enclosure Acts (Kain et al., 2004; Whyte, 2006). Both plots are mapped as Brickfield Soil Association (Jarvis et al., 1984). This equates to an FAO Eutric Stagnosol, or an Aquic soil within several USDA soil orders (USDA, 1999; WRB, 2015). Eutric Stagnosols are widespread throughout the UK uplands, and are highly susceptible to saturation, poor drainage, and overland flow (Jarvis et al., 1984). The study site soils are till-derived and slowly permeable, which overlay Tarn Moor Formation mudstone of the Buttermere and Bitter Beck Formations within the Skiddaw Group (Cooper et al., 1995; Stone, 2007).

The local climate from the Shap weather station (54° 30' 49" N, 2° 40' 40" W: 301 masl: Figure 4.1) is wet temperate, with a mean winter temperature of 4.1 °C, a mean

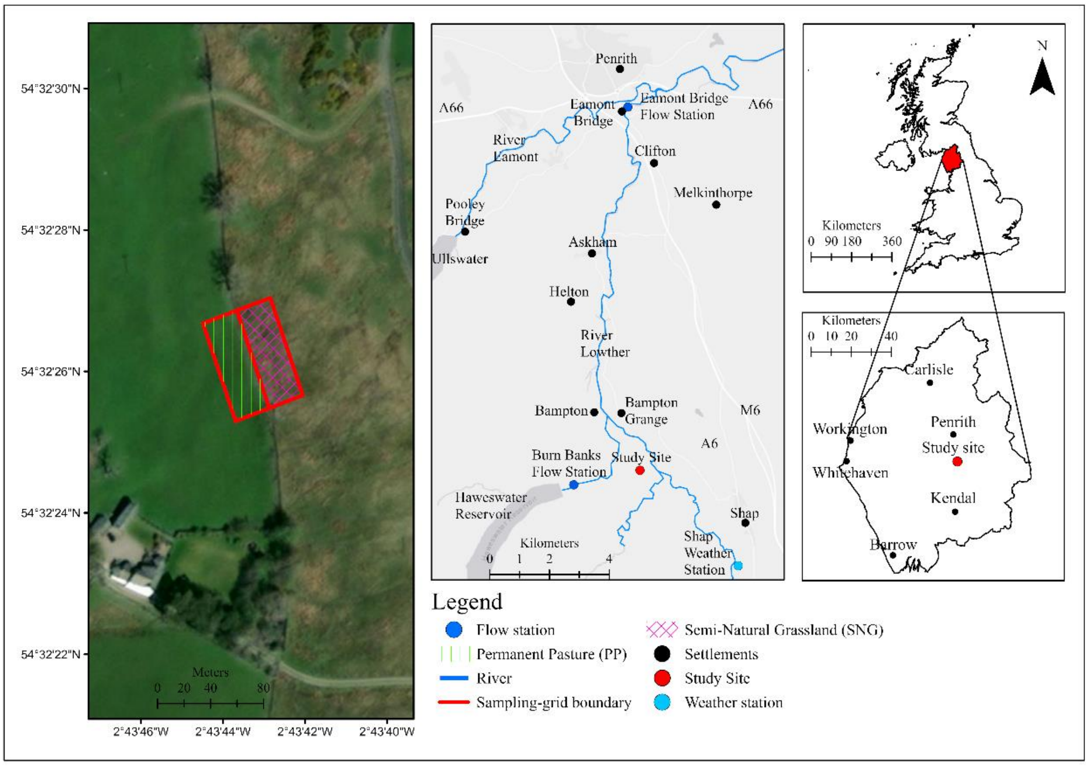

Figure 4.1: The experimental site within a UK, Cumbrian, and local area context. The permanent pasture (PP) and semi-natural grassland (SNG) sites are highlighted in green stripes and pink crosshatch, respectively. Shap weather station, alongside the downstream river gauging station (Eamont Bridge), are shown. Historically, the pasture was semi-natural grassland until being improved during the Inclosure (Enclosure) Acts of the early-mid 19th century, with the wall likely erected between 1838-1855. Contains Ordnance Survey, Gaugemap, Environment Agency (EA), ESRI, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community data.

summer temperature of 11.5 °C, and an annual rainfall average of 1,779 mm (Met Office, 2020). Daily precipitation data alongside downstream River Lowther discharge (gauged at Eamont Bridge; 54° 38' 60" N, 2° 44' 15" W: 119 masl: Figure 4.1) during the study is given in Figure 4.2. An Antecedent Precipitation Index (API) for the study site (Figure 4.2) was calculated according to Equation 4.1:

$$API_t = R_t + \kappa API_{t-1} \tag{4.1}$$

Where API is the antecedent precipitation index, R is the daily precipitation total, and κ is an empirical decay factor below 1. A κ value of 0.99 was chosen for this site as

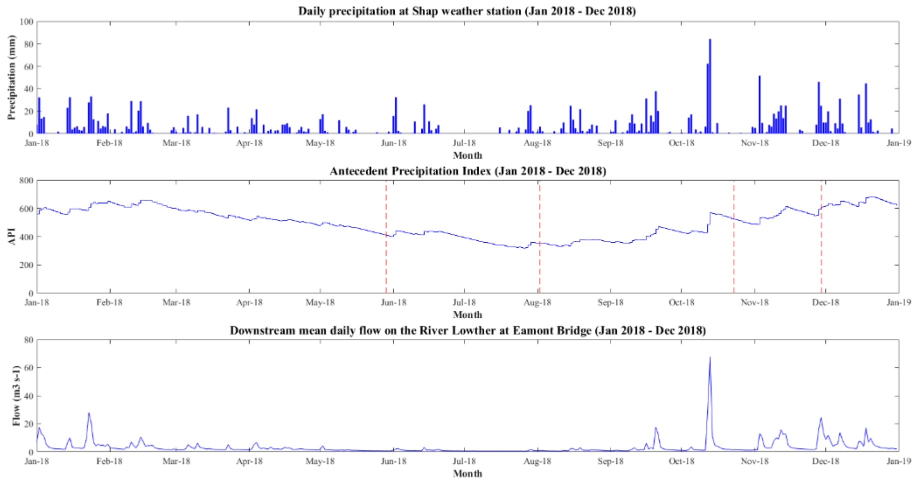

Figure 4.2: The daily precipitation data taken from Shap weather station throughout the experiment, alongside the Antecedent Precipitation Index (API) and the mean daily downstream flow at Eamont Bridge (see Figure 4.1). The sampling dates and API for 29th May (407), 2nd August (355), 23rd October (524) and 29th November (611) are shown (red dash), respectively. Note that the hydrological record covers the 2018 British Isles heatwave that lasted from 22nd June to 7th August. Precipitation data is provided by Gaugemap (Gaugemap, 2019), and discharge data by the Environment Agency (EA, 2019).

catchment conditions change relatively slowly and the API covered almost a full year. An initial condition ($API_{t=0}$) of 450 was approximated for mid-December 2017, which had no affect beyond April 2018, with the experiment beginning in May 2018.

The Permanent Pasture (PP) is a re-seeded agriculturally improved-pasture dominated by ryegrass (*Lolium spp.*) and clover (*Trifolium spp.*), but with ingress of common rush (*Juncus effusus*). PP is moderately grazed by both sheep (7.4 ha^{-1}) and beef cattle (0.5 ha^{-1}), averaging approximately 1.4 grazing livestock units ha^{-1}. The improved-pasture receives sporadic slurry, fertiliser and lime application, as well as infrequent mechanical soil-loosening as part of typical regional farming practice, although the latter did not occur during the experiment. The PP plot is well separated from farm tracks and field gates, and does not receive surplus vehicular passes, or excessive trampling or grazing pressure in comparison to the remaining improved-pasture. Sections of the PP field are drained.

The Semi-Natural Grassland (SNG) plot on Ralphland Common is communally grazed at moderate intensity by sheep (0.6 ha^{-1}), with occasional grazing by a small population of wild red deer (*Cervus elaphus*). The area is a ‘rush pasture’ of predominantly common rush, and includes a wide variety of vegetation species that are primarily controlled by grazing. Management of SNG is minimal, with no evidence of burning, quarrying, peat extraction, drainage, or other intensive management practice. The studied SNG area is separated from local vehicle and walking tracks, only receiving infrequent quad-bike passes during shepherding.

4.5.2 Experimental design

A paired plot experimental design was adopted, as PP and SNG are immediately adjacent Eutric Stagnosols with similar slopes (4% – 4.5 %). Both PP and SNG have virtually identical distributions of topographic wetness (Figure 4.3; Beven and Kirkby, 1979). The topographic wetness index (Kirkby index) is calculated according to Equation 4.2:

$$\text{Topographic wetness index} = \ln\left(\frac{\alpha}{\tan(\beta)}\right) \qquad (4.2)$$

where α is the local upslope area draining through a certain point per unit contour length, and β is the local slope angle in radians.

Both plots were covered by semi-natural grassland until PP was enclosed, likely during the early-mid 19th century (Kain et al., 2004). This experimental design therefore suggests observed differences are due to land conversion and subsequent management as opposed to inherent site dissimilarity. The study site location was

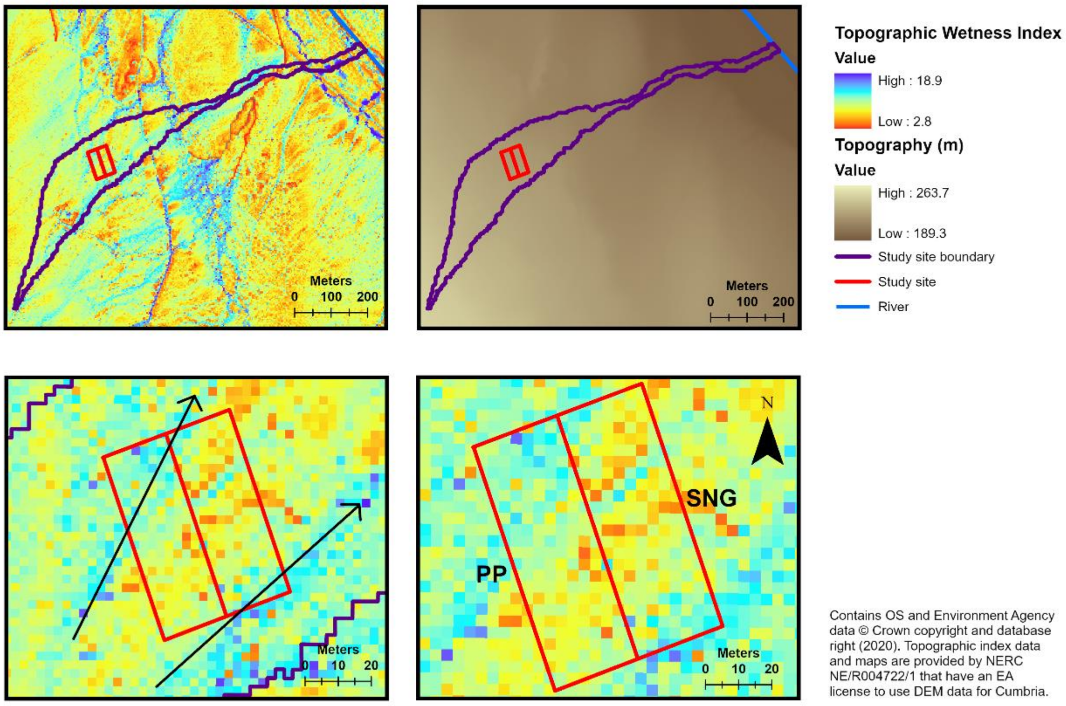

Figure 4.3: The topographic wetness index (Kirkby index: defined in Equation 4.2) for both the Permanent Pasture (PP) and the Semi-Natural Grassland (SNG). Note that both PP and SNG have similar upslope drainage areas, and similar slopes, and hence, relatively similar distributions of topographic wetness. Arrows have been annotated over the bottom-left figure to highlight the dominant flow paths travelling through the paired-plots. Topographic measurements are at a 2 m resolution. Contains Environment Agency data.

appropriate due to PP and SNG belonging to the most common upland soil type in England, with both sites following typical regional pastoral/moorland agricultural practice. Supporting precipitation and discharge information was available to infer site conditions prior to and between sampling, and to aid interpretation of results. Given that frontal rainfall is the dominant precipitation mechanism in the UK; both plots are assumed to have identical hydrometeorological inputs. Sheltering from the dry-stone wall is assumed minimal.

A rectangular 32 m by 48 m (1536 m^2) sampling grid encompassing both land-uses equally was used to measure θ_V at a 1 m resolution. This grid size was deemed the maximum number of point measurements that could be collected without the grid wetting/drying during sampling. The combined 1536 samples were considered sufficient for statistical modelling. Soil θ_V was measured (see Section 4.5.2.1) four times over a six-month period (Table 4.1), spanning from drought to fully-saturated conditions. Fixed markers remained, but the grid of measuring tapes was removed following each sampling date to avoid inhibiting agricultural practices. Following the final θ_V sampling, relative elevation and vegetation composition were recorded at an identical 1 m resolution to support interpretations.

Table 4.1: The timetable for soil volumetric wetness, terrain, soil and vegetation sampling for both the permanent pasture and the semi-natural grassland.

Date	**Activity**
29/05/18	Soil volumetric wetness sampling
02/08/18	Soil volumetric wetness sampling
23/10/18	Soil volumetric wetness sampling
29/11/18	Soil volumetric wetness sampling
12/03/19 – 13/03/19	Terrain sampling
14/05/19	Vegetation and soil sampling

This intensive sampling regime collected more results than any previous plot study of θ_V on grassland in the UK. For example, Meijles et al. (2003) recorded moisture at 151 locations over a 12,000 m^2 plot, and Ockenden and Chappell (2008) used a range of sampling intensities from 101 locations over a 525 m^2 plot to 546 locations over a 4000 m^2 plot. The greater sampling intensity in this study was considered important to ensure accurate frequency distributions and so accurate summary statistics; visual representation of soil moisture patterns and quantitative estimates of the spatial-structure (via empirical semi-variograms and models of these); and credible comparison of the moisture patterns with some of the potential controlling factors.

4.5.2.1 Surface volumetric wetness measurements (0-6 cm)

Topsoil volumetric wetness was measured *in-situ* in the field during each sampling date. A moisture-probe (ML3 'Theta-probe': Delta-T Devices Ltd) gave 768 readings per land-use for each sampling date (1536 total). The device consists of four 6 cm wave-guides arranged in a trefoil formation (attached to a probe-body) that were fully inserted into the soil surface. This moisture-probe measures θ_V (m^3/m^{-3}) using sTDR. In brief, the moisture-probe emits a continuous 100 MHz outgoing wave and records the reflection of this wave to produce a composite standing wave. The outgoing and standing wave ratio is dependent upon the dielectric constant of the soil surrounding the wave-guides, which is largely controlled by θ_V (see Gaskin and Miller, 1996). Simplified TDR was the selected experimental method due to it being a rapid and repeatable in-field technique (see Gaskin and Miller, 1996), which is necessary given the number of samples required to get an accurate measurement of θ_V given its variability (see Hills and Reynolds, 1969).

Following Whalley (1993), the moisture-probe reported in-field measurements in millivolts (mV), which were converted to θ_V post-measurement via the calibration Equation 4.3:

$$\theta_V = \frac{[1.07 + 6.4mV - 6.4mV^2 + 4.7mV^3] + \alpha_0}{\alpha_1} \quad (4.3)$$

where α_0 and α_1 are soil coefficients, and were taken as -1.6, and +8.4, respectively, due to the experiment primarily involving mineral soils (Whalley, 1993). The moisture-probe is accurate to +/- 2 % θ_V, and averages θ_V over the full length of the wave-guides, primarily around the central wave-guide (Whalley, 1993; Gaskin and Miller, 1996). The same moisture-probe was used for all measurements to account for any unknown instrument bias. Gaskin and Miller (1996) and Miller et al. (1997) give detailed information regarding the design, operation, calibration and uncertainty of moisture-probe measurements. At each sampling date no evidence of prior sTDR measurements e.g., holes in the soil surface, was present.

4.5.2.2 Reference topsoil physico-chemical properties that may influence soil volumetric wetness

Several large-scale studies have shown that soil physico-chemical properties can substantially influence soil θ_V (e.g., Pan et al., 2012). Soil texture (Wallace and Chappell, 2019), η (Beven and Germann, 1982), acidity (Holland et al., 2018), ρ_b (Drewry et al., 2000a), PR (Wallace and Chappell, 2019), and SOM (Beven and Germann, 1982), can all affect soil structural stability and functioning, and therefore permeability and water retention. To determine such properties, topsoil was extracted from the surface 10 cm. Soil samples were taken using 221 cm^3 bulk-density tins across sixteen randomly selected locations throughout PP and SNG (Figure 4.4). Random sampling was chosen as it eliminated sampling bias.

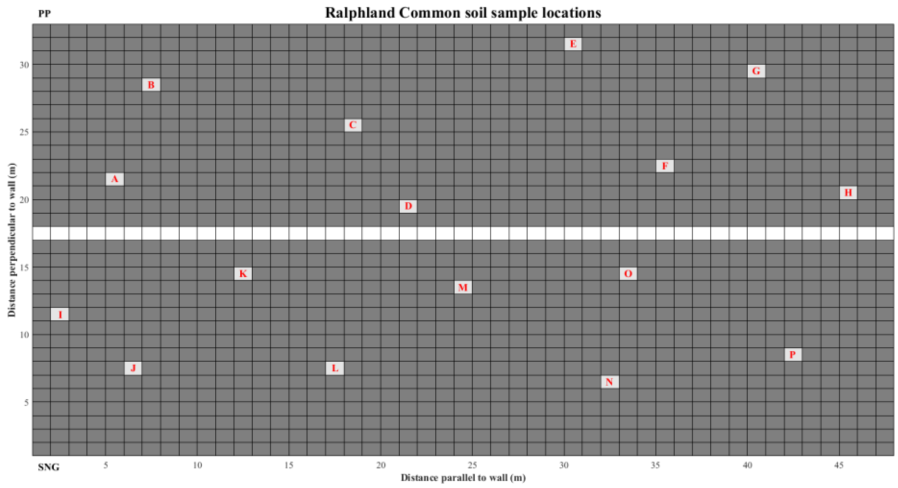

Figure 4.4: Labelled topsoil sample locations within the Permanent Pasture (PP: top) and Semi-Natural Grassland (SNG: bottom) taken on the 14th May 2019. The dry-stone wall is also shown to separate the land-uses.

Four soil samples per location were taken, with three undergoing an initial 48-hr air dry. The first air-dried sample was used to measure soil pH. The second air-dried sample was oven-dried at 105 °C for 24 hr for ρ_b calculation, and then underwent a 6-hr 550 °C loss-on-ignition test to calculate OM. The third air-dried sample underwent particle-size analysis. Particle-size analysis involved sieving oven-dried soil through a 2000-μm sieve, before mixing the sample with 1 % sodium polymetaphosphate for 24 hr to separate aggregates. The soil then underwent hydrogen peroxide treatment to remove organic material. Finally, samples underwent manual aggregate breaking and high-power sonication for five min, before laser diffraction (Beckman Coulter, LS-13-320). The final sample was gradually submerged for 48 hours with de-ionised water, and was then measured with the moisture-probe to determine η (i.e., maximum soil volumetric wetness) to remain consistent with field measurements. Soil penetration resistance was measured *in-situ* adjacent to soil sampling locations, using an SC900 Field Scout (Spectrum Technologies) penetrometer using a 12.8-mm-diameter cone. The device measures soil penetration resistance via an internal load cell, and uses an ultrasonic depth sensor to record depth in 2.5 cm steps for up to 7.5 cm.

4.5.2.3 Correlating terrain and vegetation properties with soil volumetric wetness

A 1 m resolution topographic survey of the site was undertaken four-months after the θ_V sampling, as topography may influence soil θ_V patterns (following Meijles et al., 2006; Minet et al., 2011; Meijles et al., 2015). The terrain survey combined a differential GPS (Trimble, R8-Integrated-GNSS) with a total station (Trimble, Robotic-S6), giving both the co-ordinates and elevation of each point. A vegetation survey followed the terrain analysis to allow θ_V to be compared against floristic composition. Vegetation potentially explains a significant amount of θ_V variance within areas of (semi-)natural vegetation (Chappell and Ternan, 1992; Meijles et al., 2003). The genera/species that encompassed the majority of the above-surface biomass in each square metre was recorded, even if several were present. Each square metre was centred around a sTDR measurement point. The vegetation survey occurred six months following θ_V sampling during mid-spring to aid vegetation identification. No taxonomic shifts were evident throughout the experiment, and thus, vegetation communities were assumed stationary.

4.5.3 Statistically analysing and modelling soil volumetric wetness

4.5.3.1 Comparing soil volumetric wetness distributions (Objective I)

Soil θ_V distributions at each sampling date were assessed for normality via Kolmogorov-Smirnov (KS) and Anderson-Darling (AD) tests. Theoretically, θ_V distributions cannot satisfy normality due to the bounding effect of η (which distorts frequency distributions into substantial negative skews), although normality can be an adequate practical assumption (Western et al., 2002). Soil θ_V distributions underwent Box-Cox transformations for further normality assessment.

Untransformed θ_V distributions were compared between land-uses via the non-parametric Mann-Whitney-Wilcoxon (MWW) test to avoid normality assumptions. Homogeneity of variances was tested via the non-parametric Brown-Forsythe (BF) test (Brown and Forsythe, 1974). The non-parametric statistical approaches justify the need for such an intensive sampling regime. Significance levels were taken as $p \leq 0.05$, $p \leq 0.01$, and $p \leq 0.001$.

4.5.3.2 Soil volumetric wetness spatial-structure (Objective II)

The GLOBEC geostatistical package in MATLAB (Chu, 2017) was used to generate empirical semi-variograms from θ_V observations, to assess PP and SNG spatial-structure. Semi-variogram models were derived using the GLOBEC least-squares-fit function. A range of models were fitted to the empirical semi-variogram data, including exponential (Equation 4.4), Gaussian (Equation 4.5), and spherical (Equations 4.6.1-4.6.2) models:

$$\gamma_{\mathrm{h}_{exp}} = P_0\left(1 - e^{\left(\frac{-h}{L}\right)}\right) + \gamma_0 \tag{4.4}$$

$$\gamma_{\mathrm{h}_{gau}} = P_0\left(1 - e^{\left(\frac{-h}{L}\right)^2}\right) + \gamma_0 \tag{4.5}$$

$$\gamma_{\mathrm{h}_{sph}} = P_0\left(\frac{1.5h}{L} - \frac{0.5h^3}{L^3}\right) + \gamma_0\,, 0 < h \leq L \tag{4.6.1}$$

$$\gamma_{\mathrm{h}_{sph}} = P_0 + \gamma_0\,, h > L \tag{4.6.2}$$

where γ_h is semi-variance at each lag distance, P_0 is the partial sill, h is the lag distance, L is the length scale, and γ_0 is the nugget effect. A residual sum of squares gave a goodness of fit for each semi-variogram model.

4.5.3.3 Relative influence of potential controlling variables on observed soil volumetric wetness patterns (Objective III)

To correlate elevation with soil θ_V, the corrcoef function (Pearson correlation coefficient) in MATLAB was used. The influence of flora was assessed by comparing dominant genera/species with soil θ_V values. To assess if each recorded variable significantly influences θ_V, a linear mixed-effects regression model was developed using the *lmer* function within the *lme4* package (Bates et al., 2012) using R statistical programming (R Core Team, 2018). A linear mixed-effects regression approach was necessary due to the hierarchical, highly auto-correlated, non-independent nature of the investigated variables, alongside the possibility of including temporal changes and repeat measurements within the model. Stepwise bidirectional elimination utilising a combination of Akaike and Bayesian Information Criteria created the most parsimonious model to account for soil θ_V variance. Soil θ_V was predicted according to fixed effects of land-use, month, and vegetation, with interaction effects of month-vegetation and month-land use. Random effects were taken as the intercept of elevation, with by-elevation random slopes for the effect of month and vegetation. Elevation was taken as a random effect as a limited elevation range for the local area was taken, and elevation weakly correlated with θ_V. An analysis of variance (ANOVA) test using the *anova* function was applied to the model summary to outline each explanatory factor significance (R Core Team, 2018). All linear mixed-effects regression model assumptions were satisfied following model calibration to ensure model suitability.

4.6 Results and discussion

4.6.1 Reference site conditions which may influence soil volumetric wetness

4.6.1.1 Physio-chemical topsoil properties

Topsoil samples from PP and SNG (Figure 4.4) were compared to investigate physico-chemical properties, which may influence soil θ_V, and to contextualise findings (Table 4.2). Textural analysis reveals statistically similar distributions for all particle-size fractions (≤ 2 μm, 2-20 μm, 20-60 μm, 60-200 μm, 200-2000 μm). The improved-pasture topsoil (Figure 4.5) was predominantly silty-clay loam (62.5 %), with some silt loam (25 %) and silty-clay (12.5 %). The semi-natural grassland topsoil (Figure 4.5) was mostly silty-clay loam (37.5 %) and loam (37.5 %), with some silt loam (12.5 %) and clay loam (12.5 %).

Table 4.2: The topsoil physio-chemical properties taken within the Permanent Pasture (PP: Sites A-H) and the Semi-Natural Grassland (SNG: Sites I-P) on the 14th May 2019 (see also Figures 4.4-4.5). Variables included particle-size distribution, soil texture, soil volumetric wetness (θ_V), surface penetration resistance, pH, soil dry bulk-density (ρ_b), soil organic matter (SOM), and soil porosity (η). Variable medians ($\tilde{x}$) are statistically compared via the non-parametric Mann-Whitney-Wilcoxon (MWW) test.

Site	**Particle-Size Distribution (%)**					**Soil Texture**	**θ_V (%)**	**Surface Penetration Resistance (kN)**				**pH**	**ρ_b (g cm^{-3})**	**SOM (% weight)**	**η (%)**
	≤ 2 µm	**2-20 µm**	**20-60 µm**	**60-200 µm**	**200-2000 µm**			**0cm**	**2.5cm**	**5cm**	**7.5cm**				
A	41.6	47.7	6.9	3.8	0.0	Silty Clay	37.4	104	207	1104	1138	5.48	0.92	15.3	57.3
B	36.0	41.6	9.1	13.3	0.1	Silty Clay Loam	34.3	586	1311	794	932	6.08	1.00	13.6	58.5
C	23.9	36.3	14.0	14.6	11.3	Silt Loam	36.9	0	34	1173	932	6.02	0.77	23.9	58.2
D	32.7	50.9	10.3	6.1	0.0	Silty Clay Loam	31.9	242	1346	1346	1138	5.91	0.77	18.1	57.4
E	30.3	43.6	12.1	13.4	0.5	Silty Clay Loam	34.0	287	391	1035	886	6.19	0.58	17.8	57.7
F	20.0	39.9	17.0	15.1	8.4	Silt Loam	35.7	207	862	862	862	6.25	0.81	13.9	58.3
G	33.3	52.0	10.1	4.6	0.0	Silty Clay Loam	33.9	518	1276	1414	1484	5.97	0.69	13.6	58.2
H	28.3	42.8	12.8	15.4	0.8	Silty Clay Loam	36.8	207	1380	1208	1173	5.85	0.95	13.6	58.5
I	35.4	55.3	7.2	2.1	0.0	Silty Clay Loam	38.7	34	138	518	1000	5.07	0.83	11.9	58.2
J	25.5	44.1	16.0	14.2	0.3	Silt Loam	39.1	0	104	276	276	4.94	0.76	12.7	59.8
K	32.6	50.6	12.4	4.4	0.0	Silty Clay Loam	47.4	69	69	1035	380	4.37	0.77	23.6	60.7

Table 4.2 (continued):

Site	Particle-Size Distribution (%)					Soil Texture	θ_V (%)	Surface Penetration Resistance (kN)				pH	ρ_b (g cm^{-3})	SOM (% weight)	η (%)
	≤ 2 µm	2-20 µm	20-60 µm	60-200 µm	200-2000 µm			0cm	2.5cm	5cm	7.5cm				
L	24.7	37.2	11.1	19.7	7.3	Loam	50.7	518	380	345	276	5.24	0.44	27.8	59.9
M	14.6	24.2	15.8	27.5	17.9	Loam	54.6	104	86	112	233	5.15	0.17	23.5	58.2
N	16.7	24.0	15.2	28.9	15.2	Loam	58.0	34	104	207	69	5.42	0.60	62.3	61.1
O	28.7	37.3	14.0	15.8	4.3	Silty Clay Loam	46.5	0	0	932	862	5.66	0.36	28.1	59.9
P	29.4	33.8	13.9	14.5	8.4	Clay Loam	42.6	0	0	1484	1449	4.69	0.53	16.9	58.3
PP $\tilde{x}$	31.5	43.2	11.2	13.4	0.3	NA	35.0	225	1069	1139	1035	6.00	0.79	14.6	58.2
SNG $\tilde{x}$	27.1	37.3	14.0	15.2	5.8	NA	47.0	34	95	432	328	5.11	0.57	23.6	59.9
PP σ	6.8	5.4	3.1	5	4.5	NA	1.9	197	562	217	209	0.24	0.14	3.6	0.5
SNG σ	7.3	11.3	2.9	9.6	6.9	NA	7	175	120	485	481	0.41	0.23	16	1.1
MWW (*p*)	0.235	0.279	0.246	0.279	0.333	NA	<0.001***	0.038*	0.006**	0.041*	0.027*	<0.001***	0.028*	0.278	0.022*

** Significant at the 0.05 probability level.*

*** Significant at the 0.01 probability level.*

**** Significant at the 0.001 probability level*

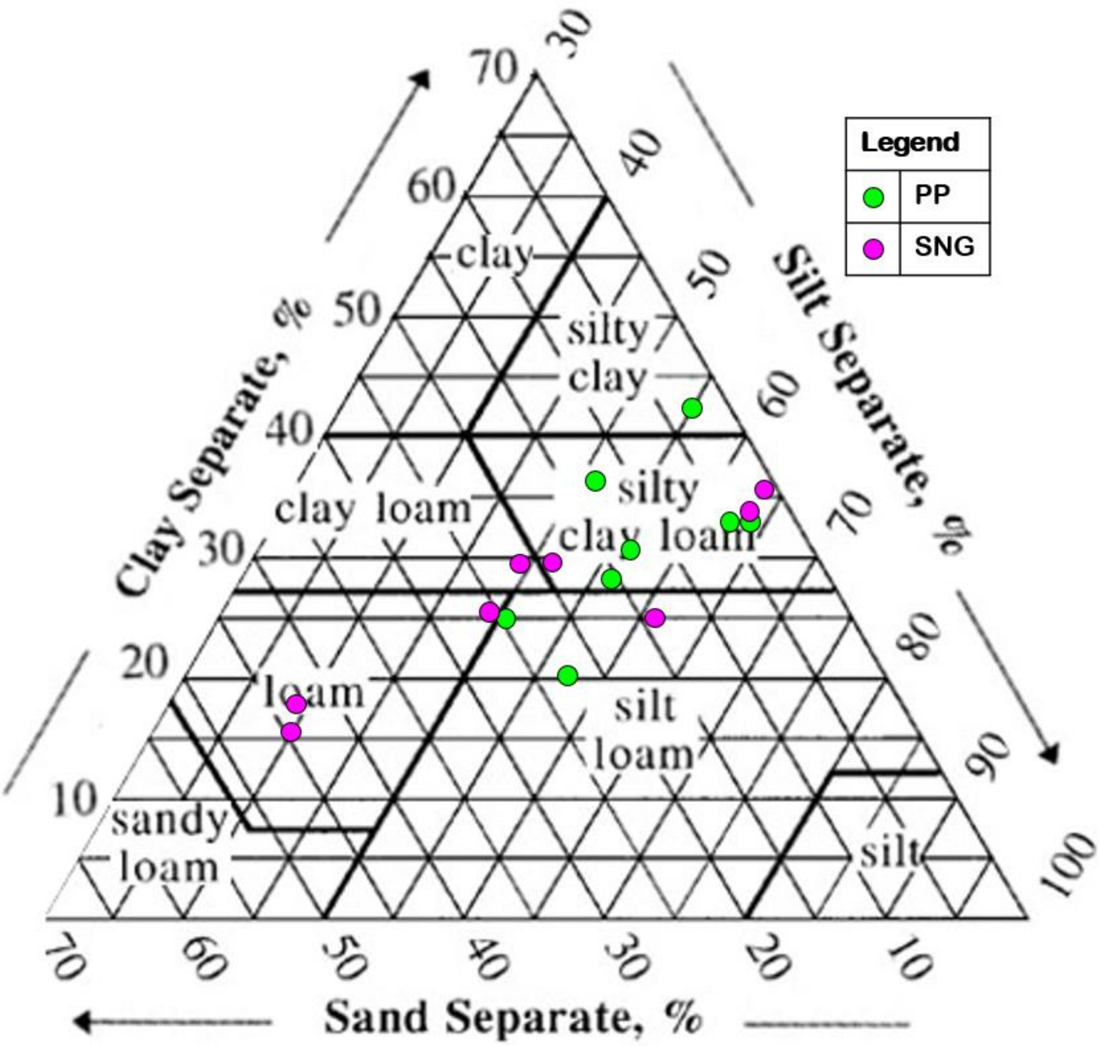

Figure 4.5: The soil particle-size distribution analysis for the Permanent Pasture (PP) and the Semi-Natural Grassland (SNG).

During topsoil sampling (Table 4.2), PP was significantly drier than SNG ($p \leq 0.001$), with a median θ_V of 35 % as opposed to 47 %. The improved-pasture also had significantly higher PR ($p \leq 0.006$-0.041) at all recorded depths (0 cm, 2.5 cm, 5 cm, and 7.5 cm), partly due to the drier soil (Wallace and Chappell, 2019). Bulk-density was significantly higher ($p \leq 0.028$) within the PP plot compared to the SNG plot, with η significantly lower ($p \leq 0.022$), possibly indicating that vegetation differences may have some influence upon soil properties (Macleod et al., 2013), or that agricultural practices had compacted the improved-pasture and potentially reduced the infiltration-capacity (Drewry et al., 2000a; Gilman, 2002; Pan et al., 2012).

Both plots had statistically similar ($p \leq 0.278$) SOM levels (Table 4.2). This may be because of slurry additions to the improved-pasture that are maintaining the naturally high levels of SOM seen in the soils beneath semi-natural grassland. Soil organic matter content was, however, considerably more variable within the SNG plot, likely due to the presence of localised carbon-rich ‘rush flushes’ within the Eutric Stagnosol (Chappell and Ternan, 1992). Indeed, sample point ‘N’ in SNG (Figure 4.4) contained more than 2.5 times the SOM of the most organic PP sample. The PP plot was significantly less acidic than the SNG plot ($p \leq 0.001$), probably due to liming (Holland et al., 2018).

4.6.1.2 Topography and elevation survey

The detailed topographic survey (Figure 4.6) showed that PP was marginally higher than SNG, with an arithmetic mean of 222.5 masl as opposed to 221.8 masl. Topography shows PP and SNG have similar elevation profiles. Average gradients perpendicular to the drystone-wall are 4.3 % for PP and 4.5 % for SNG, whilst average gradients parallel to the drystone-wall are 4 % for PP and 4.1 % for SNG.

Linear drainage features following a south-by-south-west to north by north-east (SSW-NNE) trajectory are visible in some moisture plots (refer to later Figures 4.9 and 4.11). The SSW-NNE connectivity of the near-surface drainage is most visible within the SNG plot, with agricultural interventions making this less clear within the PP plot. Both plots have a virtually identical topographic wetness (Figure 4.3), suggesting both plots should have similar θ_V values, and therefore differences in saturation should be predominantly due to land-use as opposed to landscape factors.

The foundations of the drystone-wall may impede any shallow drainage of moisture from the PP plot to the SNG plot, giving accumulation upslope of the wall within PP.

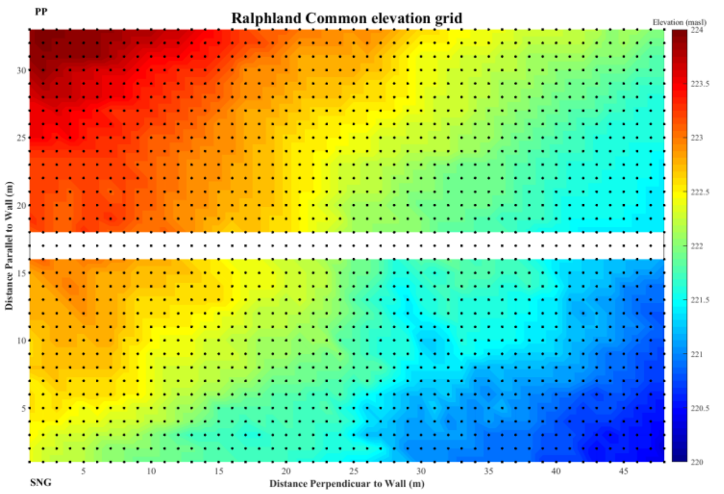

Figure 4.6: The elevation profile at the study site, within the Permanent Pasture (PP: top) and Semi-Natural Grassland (SNG: bottom). The wall separating the plots is approximately along a NNW-SSE axis. Note the depression at the boundary within SNG at approximately 28 m distance parallel to the wall, which extends further into SNG. It is likely this local depression will remain wetter than the surrounding areas throughout the experiment (see Figure 4.3).

This effect is however, not seen in the surface moisture measurements (refer to later Figures 4.9-4.12); perhaps indicating the walls foundation is permeable. Figure 4.6 highlights a shallow depression within the SNG plot beginning at the drystone-wall at approximately 28 m north-west along the wall, heading approximately north-east. This depression is likely to retain moisture and may have contributed to the increased SOM soil content as observed in sample point ‘N’ (Figure 4.4).

4.6.1.3 Vegetation survey

The taxonomic survey (Figure 4.7) reveals a small number of dominant genera/species within each square metre. The PP plot was almost entirely dominated by ryegrass (*Lolium spp.*), encompassing 94.7 % of the sampling grid, with pockets of common rush covering only 4.4 %. Stinging nettle (*Urtica dioica*) and broad-leafed dock (*Rumex obtusfolious*) were present against the dry-stone wall, at 0.7 % and 0.3 % of the area, respectively. The improved-pasture contained significant clover (*Trifolium*

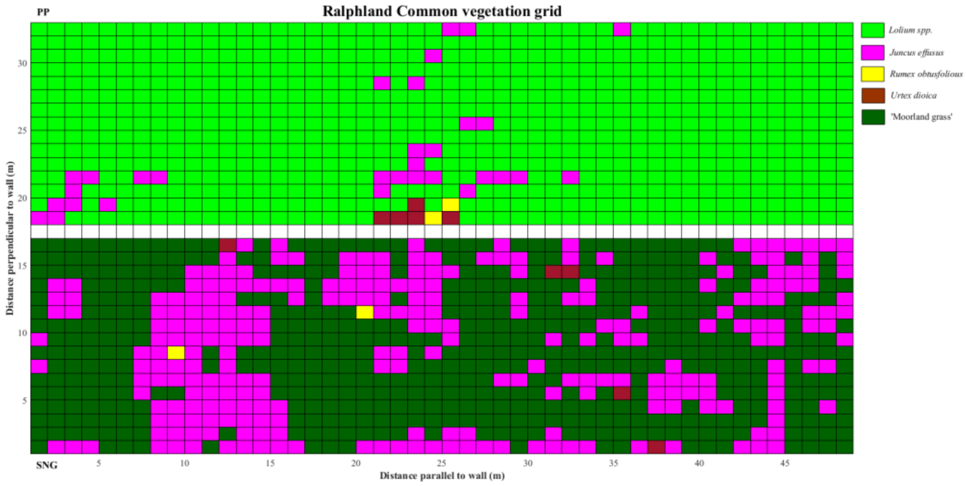

Figure 4.7: The dominant vegetation within each square metre, with the Permanent Pasture (PP: top) and Semi-Natural Grassland (SNG: bottom) highlighted. Note that SNG contains substantially more common rush, and that the moorland grass is an admixture of several different grass species. The stinging nettle and broad-leafed dock within PP border the dry-stone wall and are amassed around a small tree stump.

spp.), buttercup (*Ranunculus spp.*) and ox-eye daisy (*Leucanthemum vulgare*) populations, likely from re-seeding mixtures, although these never dominated a grid cell.

Vegetation within SNG (Figure 4.7) was predominantly a mixture of the *Pooideae* subfamily of grass species (65.4 %), primarily consisting of common bent (*Agrostis capillaris*), creeping bent (*Agrostis stolonifera*), mat-grass (*Nardus stricta*), and ryegrass. This grass mixture is hereby referred to as ‘moorland grass’. Common rush was also very common at 33. 7 %. Stinging nettle and broad-leafed dock occurred at 0.7 % and 0.3 % incidence, respectively. A large number of additional vegetation species were recorded within SNG, particularly plume thistles (*Cirsium spp.*), and *Sphagnum* mosses (*Sphagnum spp.*), with no grid containing fewer than three species.

Ryegrass dominance within PP is due to improved-pasture re-seeding to maintain sward levels and prevent reversion (Gilman, 2002), with the significant clover and ox-eye daisy population further supporting re-seeding. Common rush prevalence in both land-uses is likely caused by locally poor drainage in a high rainfall environment, both highly suited to *Juncus spp.* proliferation (McCorry and Renou, 2003; AHDB, 2013). Gilman (2002) notes that *Juncus spp.* will often be the first moorland species to colonise ryegrass/clover swards in high rainfall upland environments of the UK. Nearby upland catchment studies have also noted considerable *Juncus spp.* and *Nardus spp.* compositions, both being typical of upland soils (Gilman, 2002; Orr and Carling, 2006).

To reemphasise, minimal research has compared the differences between moorland and improved-pasture vegetation that may be correlated with changes in hydrological properties. The conversion of semi-natural grassland into permanent pasture likely reduced vegetation height, biomass and root depth, thereby potentially reducing evapotranspiration in PP due to reduced wet-canopy evaporation, as well as reducing η in PP (Sansom, 1999; Gilman, 2002; Orr and Carling, 2006). Sansom (1999) and Gilman (2002) postulate that moorland conversion reduces infiltration rates, hydraulic conductivity, surface roughness, and evapotranspiration, ultimately causing increased overland flow and elevated flood-risk.

4.6.2 Soil volumetric wetness probability distributions (Objective I)

Tables 4.3-4.4, highlight that no PP or SNG probability distribution satisfied normality (KS or AD tests), justifying the use of non-parametric statistical tests. May and August distributions had predominantly weak-positive skews, whilst October and November had more extreme negative skews (Figure 4.8). Distributions were principally leptokurtic (displaying excess kurtosis), especially November (Tables 4.3-4.4). Box-Cox transformations could only normalise (AD tests only) May and August PP distributions, further justifying the non-parametric approach.

Table 4.3: The Kolmogorov-Smirnov (KS) and Anderson-Darling (AD) statistical distribution tests applied to the Permanent Pasture (PP) and Semi-Natural Grassland (SNG) volumetric-wetness probability distributions. Note that most tests are extremely significant, indicating that they significantly differ from the Gaussian distribution. The statistical tests are paired with excess kurtosis and skewness values to infer why distributions may violate normality. Box-Cox transformed distributions using the maximum log-likelihood function are shown adjacent to the raw data to highlight the extent of non-normality, with further supporting kurtosis and skewness values.

Sampling Date	**Land-use**	**Raw Data**				**Box-Cox Transformed**			
		KS	**AD**	**Excess Kurtosis**	**Skewness**	**KS**	**AD**	**Excess Kurtosis**	**Skewness**
29th May 2018	**PP**	<0.001***	<0.001***	+0.86	+0.75	<0.001***	0.839	+0.02	-0.00
	SNG	<0.001***	<0.001***	-1.00	-0.14	<0.001***	<0.001***	+1.02	-0.11
2nd August 2018	**PP**	<0.001***	<0.037*	-0.04	+0.25	<0.001***	0.887	-0.08	-0.00
	SNG	<0.001***	<0.001***	+3.20	+1.39	<0.001***	0.007**	+0.47	-0.01
23rd October 2018	**PP**	<0.001***	<0.001***	+4.37	-1.68	<0.001***	0.013*	-0.49	-0.14
	SNG	<0.001***	<0.001***	-0.53	-0.44	<0.001***	<0.001***	-0.86	-0.13
29th November 2018	**PP**	<0.001***	<0.001***	+22.58	-3.72	<0.001***	0.002**	+0.64	+0.00
	SNG	<0.001***	<0.001***	+7.26	-2.42	<0.001***	<0.001***	-0.70	-0.49

Table 4.4: The summary statistics, including arithmetic mean ($\bar{x}$), median ($\tilde{x}$), and coefficient of variation (CV), for the soil volumetric wetness measurements (θ_V) taken at each sampling date within the Permanent Pasture (PP) and Semi-Natural Grassland (SNG). The statistical tests for central tendency (Mann-Whitney-Wilcoxon, MWW) and variation (Brown-Forsythe, BF) are also given. Note that all statistical tests are extremely significant, excluding the November MWW test.

Sampling Date	**Land-use**	**$\bar{x}$ (θ_V %)**	**Geometric Mean (θ_V %)**	**Min (θ_V %)**	**Lower Quartile (θ_V %)**	**$\tilde{x}$ (θ_V %)**	**Upper Quartile (θ_V %)**	**Max (θ_V %)**	**CV (%)**	**MWW**	**BF**
29th May 2018	**PP**	28.2	27.8	15.9	24.6	27.6	31.2	50.2	18.5	<0.001***	<0.001***
	SNG	42.2	40.7	17.4	33.6	42.7	51.0	63.1	25.9		
2nd August 2018	**PP**	30.7	30.1	12.9	26.6	30.4	34.5	51.5	18.9	<0.001***	<0.001***
	SNG	24.7	23.6	9.3	19.6	23.4	28.3	62.4	32.1		
23rd October 2018	**PP**	53.7	53.4	20.4	51.4	54.6	57.6	61.9	10.4	<0.001***	<0.001***
	SNG	45.9	44.5	14.8	38.8	46.6	54.4	65.6	23.2		
29th November 2018	**PP**	60.9	60.9	47.7	60.5	61.1	61.7	63.6	2.5	0.757	<0.001***
	SNG	59.2	58.9	27.3	58.3	61.0	62.5	64.7	9.2		

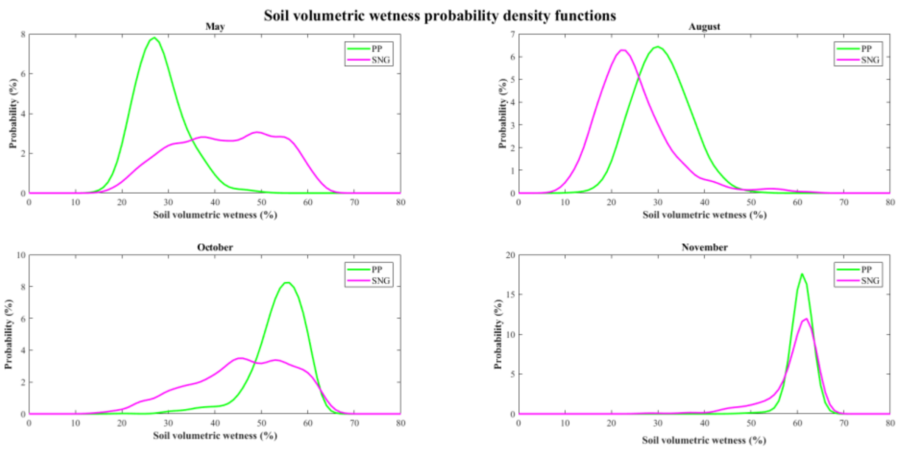

Figure 4.8: The kernel-generated probability density functions for soil volumetric wetness during each sampling date. Each distribution was generated according to 768 samples based on the respective land-use. Note that these are statistically tested for normality in Table 4.3, and the central tendency and variation is statistically compared between land-uses in Table 4.4.

4.6.2.1 May dataset

During the May sampling date (Figure 4.9), PP was significantly drier than SNG ($p \leq 0.001$), with a median θ_V of 27.6 % as opposed to 42.7 % (Table 4.4: Figure 4.8). Soil volumetric wetness differences could be because of higher evapotranspiration within PP, due to the rapidly growing, dense ryegrass sward (Hall, 1987; Cox et al., 1988). Evapotranspiration is generally assumed to be greater from semi-natural grassland compared to improved-pasture; although this relationship is dependent on vegetation growth and is primarily based off studies involving heather (which almost certainly presents greater roughness) as opposed to 'rush pasture' (Miranda et al., 1984; Hall and Harding, 1993; Gilman, 2002; Orr and Carling, 2006). During May, the SNG plot had several unvegetated soil patches, the moorland grass was heavily grazed, and the common rush was withered with minimal foliage; all implying low transpiration rates. Furthermore, the PP plot could additionally contain fewer pockets of impermeable soil as local agricultural practices encourage drainage, reducing θ_V (Wallace and Chappell, 2019).

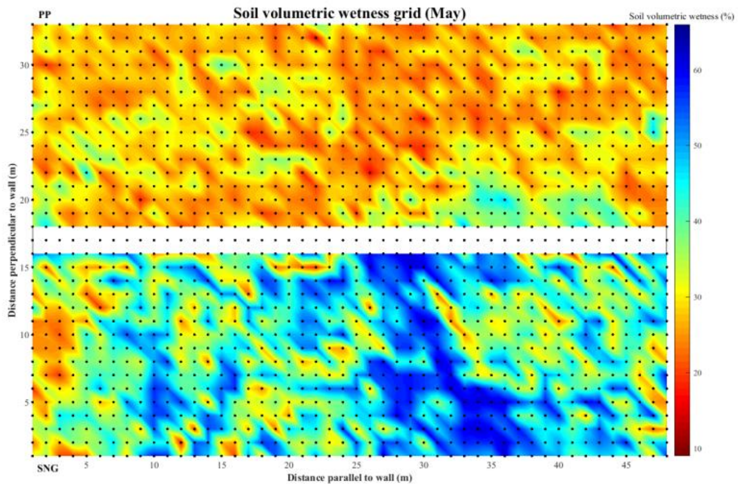

Figure 4.9: The soil volumetric wetness grid taken on the 29th May 2018. Note that the Permanent Pasture (PP) is at the top of the figure, and the Semi-Natural Grassland (SNG) is at the bottom, with the dry-stone wall shown to separate the land-uses. The permanent pasture was significantly drier than the semi-natural grassland, and contained significantly less variation. Linear features draining from the south-west to north-east according to the ‘regional’ topographic highpoint are also evident, primarily within SNG.

Removing θ_V variability within an improved-pasture is a central objective of ploughing prior to re-seeding, in order to generate an even grass sward (Schulte et al., 2012). This pioneering study has shown that the PP plot did indeed contain significantly less variation in θ_V ($p \leq 0.001$) than observed in the SNG plot (Table 4.4: Figure 4.8). If the ecological status and functioning of permanent pastures were to be restored to behave more like semi-natural grassland, then the diversity in moisture patterns would need to be re-introduced. As this first sampling date (spring 2018) shows the improved-pasture to be drier than the semi-natural grassland, if representative, this may suggest that permanent pastures dry faster and thus, are more sensitive to water stress with the onset of droughts, a potential concern for livestock production.

4.6.2.2 August dataset

Between May and August 2018, the semi-natural grassland saw median θ_V fall from 42.7 % to 23.4 % (Table 4.4, Figures 4.8 and 4.10). At the 2nd August 2018 sampling date, the API was close to the lowest value for the whole of 2018 (Figure 4.2), indicating that the sampling programme had observed soil near its driest state in 2018. The high degree of drying was due to relatively high levels of solar radiation over the summer months and moderate rainfall since the previous measurement (only 192 mm in 64 days), with the 40 days prior to sampling recording 56 % of the long-term average rainfall for this period at this locality (Met Office, 2020). In some contrast, the median θ_V in the PP plot was maintained over the same period, increasing slightly from 27.6 % to 30.4 % (Table 4.4, Figure 4.8). As a result, the SNG plot became significantly drier than the PP plot ($p \leq 0.001$). As before, the SNG plot contained significantly more variance than the improved-pasture ($p \leq 0.001$).

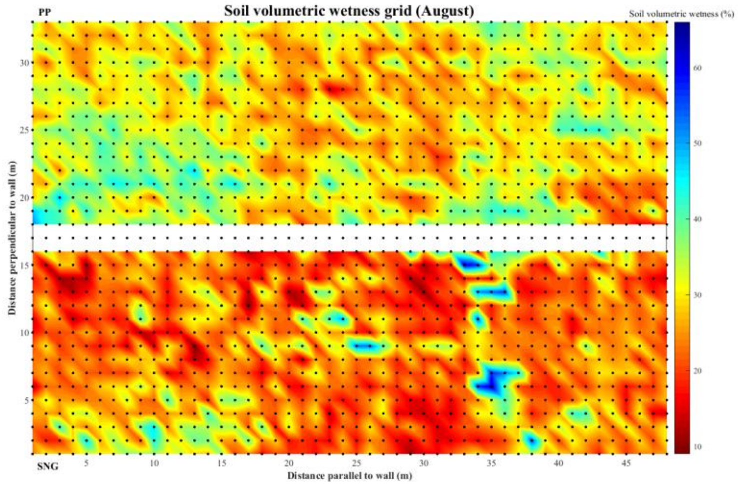

Figure 4.10: The soil volumetric wetness grid taken on the 2nd August 2018 during the British Isles heatwave. Note that the Permanent Pasture (PP) is at the top of the figure, and the Semi-Natural Grassland (SNG) is at the bottom, with the dry-stone wall shown to separate the land-uses. The permanent pasture was significantly wetter and significantly less varied than the semi-natural grassland during August sampling. Some linear features are still observable in both land-uses despite the dryness.

The additional drying effects of higher radiation and lower rainfall was more than offset by artificial moisture additions in the form of slurry to the PP field. This indicates that while the PP plot initially dried faster than the SNG plot, agricultural interventions could offset these effects. Slurry additions to improved-pastures do have a negative impact on the water-quality of adjacent streams however (Hunter et al., 1999). Consequently, if such additions were not permitted, then the permanent pasture would lose its artificial moisture input during a drought, and from the May results, could be in a drier state than the semi-natural grassland when the drought is most severe. Withholding slurry during these periods would therefore likely cause substantial sward damage (Schulte et al., 2012).

4.6.2.3 October dataset

Over the 81 days between the 2nd August and 23rd October 2018 sampling dates, 504 mm of rain was recorded (Figure 4.2). As a result, the SNG plot became much wetter, increasing to a median θ_V of 46.6 % (Table 4.4, Figures 4.8 and 4.11). The PP plot became wetter still; increasing to 54.6 % (Table 4.4, Figure 4.8) and remaining statistically wetter than the SNG plot ($p \leq 0.001$).

Interestingly, θ_V within both plots increased by similar amounts (+24.2 % for PP and +23.2 % for SNG). Identically to previous sampling dates, the SNG plot contained significantly higher variance ($p \leq 0.001$). The October θ_V data shows that improved-pastures with summer slurry additions can be wetter than semi-natural grasslands at the onset of autumn rains. Indeed, the median improved-pasture θ_V is only 3.6 % below the median η, suggesting that most of the improved-pasture is near saturation and could quickly saturate during storm events. The semi-natural grassland θ_V is 13.3 % below median η, suggesting some remaining storage capacity before SOF

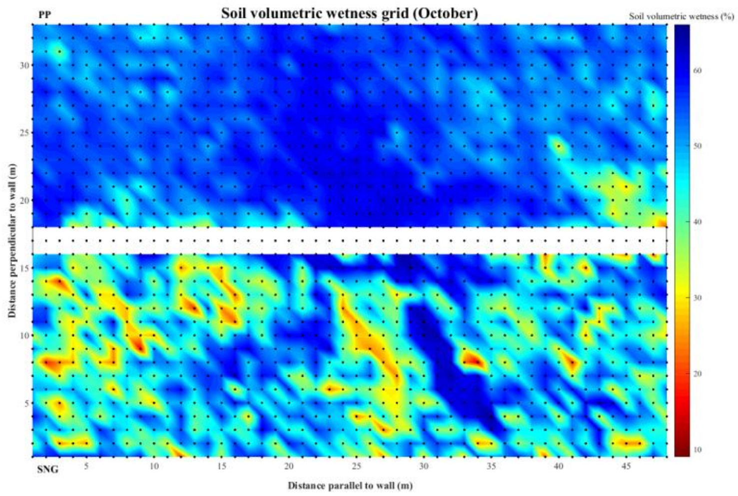

Figure 4.11: The soil volumetric wetness grid taken during on the 23rd October 2018. Note that the Permanent Pasture (PP) is at the top of the figure, and the Semi-Natural Grassland (SNG) is at the bottom. The dry-stone wall is shown to separate the land-uses. The pasture was significantly wetter than the semi-natural grassland at the time of sampling, and contained significantly less variation. Linear features are still clearly observable within the semi-natural grassland, although these are slightly masked in the permanent pasture due to the level of saturation.

generation. At moisture plots 20 km to the east of those in this study, Ockenden and Chappell (2008) also observed that their single permanent pasture plot was wetter than semi-natural grassland plots during autumnal monitoring.

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4.6.2.4 November dataset

November sampling (Figure 4.12) occurred during Storm Diana (28th – 29th November 2018) when surface ponding and overland flow was observed throughout both land-uses. Within the PP plot, overland flow predominantly flowed north by north-east, and did not appear to be moving onto SNG (Figure 4.3). Within the SNG plot, overland flow aligned with the linear drainage patterns and generally headed north-east. Both land-uses contained almost identical θ_V medians ($p \leq 0.757$), with 61.1 % and 61.0 %, respectively (Table 4.4: Figure 4.8). Variance remained significantly higher in SNG however ($p \leq 0.001$: Table 4.4). Ockenden and Chappell (2008) working at the sites previously mentioned, similarly observed larger θ_V variation within semi-natural grasslands compared to improved-pastures for monitoring dates including the winter.

Between October and November sampling dates, 265 mm of precipitation fell in 36 days, and this was reflected in a very high API on the sampling date (Figure 4.2). The strong negative skew within both frequency distributions suggests that moisture content at most places in both plots was approaching the upper limit of topsoil wetness, i.e., η (Tables 4.2 and 4.4: Figure 4.8: Western et al., 2002). These findings suggest that during large storm events, even semi-natural grasslands may generate SOF and so heighten local flood-risk. Thus, attempting to re-establish semi-natural grasslands and associated soils in areas of permanent pasture may not necessarily reduce the incidence of SOF as part of so-called NFRM.

4.6.3 Soil volumetric wetness spatial-structure (Objective II)

The geostatistical analysis shows that the spatial-structure of θ_V within the SNG plot remained similar (i.e., relatively stationary) from May to November 2018 and is described well by exponential/spherical models (Figure 4.13; Table 4.5). Meijles et al.

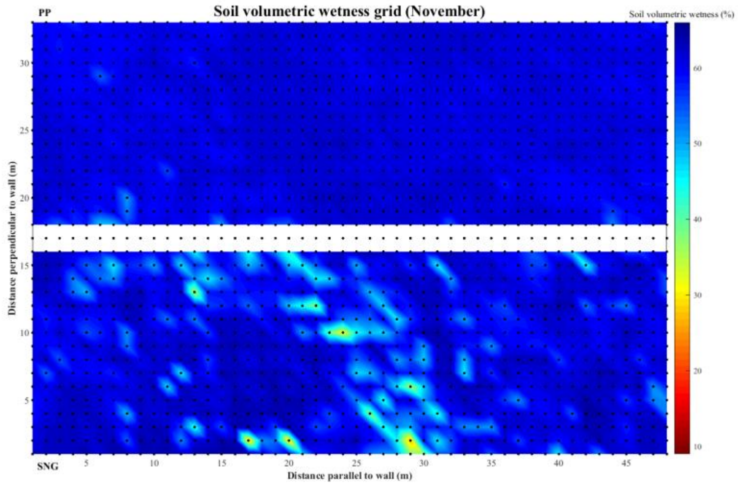

Figure 4.12: The soil volumetric wetness grid taken on the 29th November 2018 during Storm Diana. Note that the Permanent Pasture (PP) is at the top of the figure, and that the Semi-Natural Grassland (SNG) is at the bottom. The wall is shown to separate the land-uses. Both land-uses have statistically similar medians during these extremely saturated conditions, although SNG remained significantly more varied. Linear features are weakly observable within the semi-natural grassland, even though most of the land-use is at saturation.

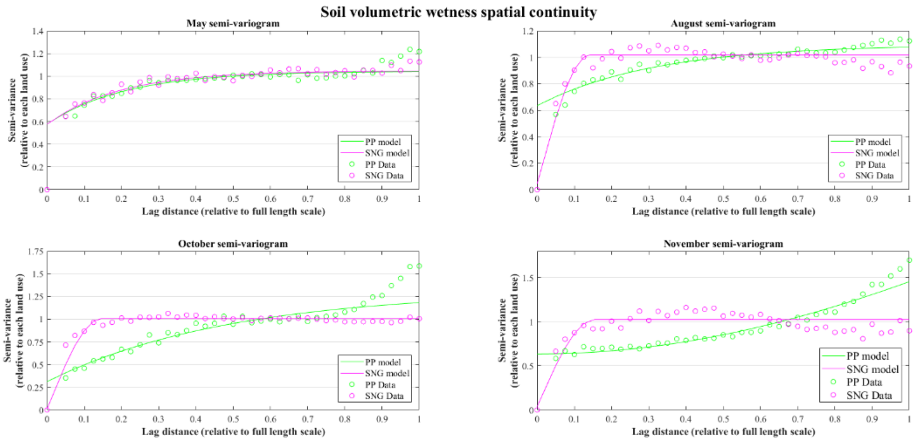

Figure 4.13: The GLOBEC generated empirical semi-variograms for all sampling dates. All models are fitted with the least-squares fit function within GLOBEC, using the 768 samples in each land-use. Note that empirical semi-variograms become progressively dissimilar as the experiment proceeded, and that the full-length scale lag distance is 50.6 m.

Table 4.5: The model parameters from GLOBEC generated empirical semi-variogram models for each sampling date within the Permanent Pasture (PP) and the Semi-Natural Grassland (SNG), alongside the Antecedent Precipitation Index (API) for each of the respective dates. The actual range in metres is given below the effective range. Note that the sill is the nugget plus the partial sill.

Sampling date	API	Land-use	Nugget effect (γ_0)	Sill ($\gamma_0 + P_0$)	Effective range	Model	Residual Sum of Squares
29th May 2018	**407**	**SNG**	0.574	1.044	0.196 (9.9m)	Exp	0.377
		PP	0.576	1.044	0.210 (10.6m)	Exp	0.464
2nd August 2018	**355**	**SNG**	0.045	1.017	0.141 (7.1m)	Sph	0.111
		PP	0.635	1.097	0.317 (16.0m)	Exp	0.453
23rd October 2018	**524**	**SNG**	0.011	1.008	0.148 (7.5m)	Sph	0.085
		PP	0.312	1.312	0.488 (24.7m)[a]	Exp	0.643
29th November 2018	**611**	**SNG**	0.039	1.026	0.154 (7.8m)	Sph	0.370
		PP	0.635	6.914	2.672 (135.2m)	Gau	0.575

[a] *Note that the model fit is reduced at larger lag distances and, therefore, that the true model effective range is, therefore, highly uncertain.*

(2003) identically found semi-natural grassland at Dartmoor, UK, to have exponential or spherical semi-variogram models.

The spatial-structure of θ_V within PP was similar to that of the SNG plot during the relatively dry conditions of May. Slurry additions and rainfall gradually shifted the spatial-structure from exponential to a Gaussian relationship, whereby the auto correlation continued beyond the size of the experimental plot (Figure 4.13). Selected models suitably fit the improved-pasture data, although the October semi-variogram has noticeable residuals at large lags, probably because of the transitioning spatial-structure.

The sill is the point at which the semi-variance plateaus within a model (i.e., the semi-variance as lag-distance approaches infinity). Most semi-variogram models (Figure

4.13; Table 4.5) have a sill marginally above 1, with October PP having a slightly higher sill of 1.31, and November PP (Gaussian model) having a sill at 6.91. The elevated sills outline higher spatial variance of two distantly separated points as the improved-pasture saturated; which was unobserved within SNG (Grayson and Blöschl, 2000).

The effective range is the distance from zero lag to the onset of the sill (95 % in exponential models, 100 % in spherical and Gaussian models), and can be interpreted as correlation length (i.e., the point beyond which there is no spatial auto correlation). The effective range within SNG remained essentially stationary throughout the study, suggesting θ_V spatial auto correlation is independent of the level of saturation. With increased saturation, PP contained considerably larger effective ranges than SNG (Figure 4.13: Table 4.5). Agricultural interventions within PP likely homogenised soil variation and facilitated moisture redistribution. As soils saturated, the lack of heterogeneity exerts a greater control on soil moisture redistribution at decimetre scales rather than the metre scales seen with the natural soils under the semi-natural grassland, and thus, amplifies spatial auto correlation. This was also seen within the decreasing coefficient of variation as PP saturated (Table 4.4). Ockenden and Chappell (2008) similarly found shorter correlation lengths for semi-natural grassland when compared with those of a single improved-pasture plot. Meijles et al. (2003) found correlation length in semi-natural grassland to vary with saturation, an unobserved process in this study.

The nugget variance is the model semi-variance at zero lag, and is generally interpreted as a combination of sampling/instrument error and spatial variation below the minimum sample spacing (i.e., < 1 m variation). Within both PP and SNG plots, the nugget variance reached approximately half that of the sill variance (Figure 4.13).

This nugget variance would indicate that there is significant variation in θ_V at distances shorter than the 1 m sampling grid. This suggests that future studies that are able to collect more than the 1536 (i.e., 768 x 2) values of θ_V across a paired-plot on a sampling day should do so over an even finer sampling resolution (e.g., 10 cm grid). This would confirm whether deterministic spatial-structure is present at sub-metre scales or whether other factors such as instrument-related uncertainty in θ_V measurements are responsible. A simulated semi-variogram with 2 % θ_V error (uniformly distributed) for a plot-scale grid gave a nugget variance of approximately 0.05, suggesting instrument-related error is minimal (see Appendix B4.0 for Supplemental Figure 4.S1 and Supplemental Table 4.S1). The slightly higher nugget variance in PP compared to SNG suggests increased fine-scale θ_V variation within the improved-pasture, further implying decimetre-scale moisture redistribution.

4.6.4 Predictor variables of soil volumetric wetness (Objective III)

The final objective of this study is to determine for this particular plot-pair, the relative strength of the relationships between soil moisture content and the potential predictors of land-use, elevation, vegetation species, and season. This was assessed via correlation coefficients and linear mixed-effects regression modelling.

Table 4.6 shows the correlation coefficients (r) of θ_V and elevation for both individual and combined PP and SNG plots (Figure 4.6). Weak correlations between topsoil moisture and elevation for the two plots, suggests elevation is not acting as a dominant control on θ_V (see Figure 4.3). Combining the weak relationships with the limited topographic range justifies the use of elevation as a random effect in the linear mixed-effects regression model.

Beneath the complex vegetation communities of the semi-natural grassland,

Table 4.6: The correlations of soil volumetric wetness with elevation for both the Permanent Pasture (PP) and Semi-Natural Grassland (SNG), as well as when combined, expressed in terms of the correlation coefficient (r). Note that inverse correlations imply that higher elevations tend to be drier.

Month	Land-use	Elevation (Land-use)	Elevation (Combined)
May	PP	-0.107	-0.453
	SNG	-0.248	
August	PP	0.124	0.223
	SNG	-0.048	
October	PP	0.140	0.194
	SNG	-0.120	
November	PP	-0.099	0.000
	SNG	0.000	

differences in θ_V between common rush and grass species were not apparent (Table 4.7). This lack of apparent difference may be because of weak vegetation differentiation within the SNG plot, with many sampling grids containing both rush and grass species. Other studies have more successfully differentiated semi-natural grassland vegetation species, with Meijles et al. (2015) outlining that moorland grasses saturate faster than heather or bracken (*Pteridium aquilinum*). Conversely to the SNG plot, soil beneath common rushes in PP was wetter than ryegrass during August and October sampling dates, although similar in May and November (Table 4.7). While the mechanism is unclear, the dense root network beneath common rush apparently retains more moisture (from drainage or transpiration) following slurry or rainfall compared to soil beneath ryegrass.

As a measure of the relative importance of temporal changes (i.e., across the four sampling dates), spatial differences due to land-use (i.e., semi-natural grassland versus agriculturally-improved permanent pasture), and vegetation (i.e., common rush versus grass species); linear mixed-effects regression modelling was undertaken and the results presented in Table 4.8. Both the sampling month ($p \leq 0.001$) and the vegetation classification ($p \leq 0.002$) significantly benefit θ_V prediction.

Table 4.7: The arithmetic mean ($\bar{x}$) and median ($\tilde{x}$) soil volumetric wetness (θ_v) for grasses and common rush within the Permanent Pasture (PP) and Semi-Natural Grassland (SNG) for each sampling date. Both averages are presented due to non-normality in the θ_v distributions.

Land-use	PP (θ_V %)						SNG (θ_V %)					
	Average		Ryegrass		Common rush		Average		Moorland grass		Common rush	
Month	$\bar{x}$	$\tilde{x}$	$\bar{x}$	$\tilde{x}$	$\bar{x}$	$\tilde{x}$	$\bar{x}$	$\tilde{x}$	$\bar{x}$	$\tilde{x}$	$\bar{x}$	$\tilde{x}$
May	28.2	27.6	28.3	27.6	27.9	27.4	42.2	42.7	41.7	42.6	43.0	43.2
August	30.7	30.4	30.5	30.2	35.0	34.2	24.7	23.4	24.3	23.0	25.6	24.1
October	53.7	54.6	53.5	54.5	57.2	58.8	45.9	46.6	45.8	46.4	45.9	47.0
November	60.9	61.1	60.9	61.1	61.5	61.7	59.2	61.0	59.2	61.0	59.3	61.3

Table 4.8: The analysis of variance (ANOVA) output tables given the most parsimonious linear mixed-effects regression model according to a combination of Akaike and Bayesian Information Criteria using bidirectional elimination. Note that this includes all fixed effects of the model, where Month:Land-use and Month:Veg indicate interaction terms.

ANOVA	Df	Sum Sq	Mean Sq	F-Value	P-Value
Month	3	399661	133220	2758.4	<0.001***
Land-use	1	28	28	0.585	0.444
Veg	4	841	210	4.35	0.002**
Month:Land-use	3	78782	26261	543.7	<0.001***
Month:Veg	12	1257	105	2.1687	0.011*
Residual	6094	-	-	-	-

The interactions of month-land-use ($p \leq 0.001$) and month-vegetation ($p \leq 0.011$) also significantly improve θ_V prediction. The interaction terms in the linear mixed-effects regression model demonstrate that sampling date (i.e., month), significantly affects how land-use and vegetation influence θ_V. Land-use alone does not significantly improve prediction because its importance is already captured within the interaction terms. Modelling results compare well with Meijles et al. (2006) and Meijles et al. (2015), who concluded that vegetation was the dominant control on soil wetness during 'dry' conditions at Dartmoor (semi-natural grassland).

The regression model output (Table 4.9) demonstrates a conditional R^2 of 78.3 %, meaning that the predictor variables can explain more than three quarters of θ_V variance at a 1 m resolution. Including elevation only adds a very small additional

Table 4.9: The regression model output giving variance (σ^2), slope effects of elevation:month ($\tau_{00\ Elev:Month}$) and elevation:vegetation ($\tau_{00\ Elev:Vegetation}$), the intra-class correlation (ICC), the number of elevation values (N_{Elev}), the total number of observations (N_{Obs}), marginal and condition R^2 values, the Akaike Information Criteria (AIC), and the Bayesian Information Criteria (BIC). The marginal R-squared shows the model fit purely using the fixed-effects, and the conditional R-squared shows the model fit using mixed-effects.

θ_V Linear Mixed-Effects Regression Model	
Predictors	**Estimates**
Intercept	25.93
σ^2	48.30
$\tau_{00\ Elev:Month}$	2.55
$\tau_{00\ Elev:Vegetation}$	1.65
ICC	0.03
N_{Elev}	38
N_{Obs}	6144
Marginal R^2	0.777
Conditional R^2	0.783
AIC	41538
BIC	41874

amount of explained variance (0.6 %: the difference between R^2 values). The degree of interaction between predictor variables seen (i.e., multicollinearity), justifies the use of the mixed-effects regression model. In particular, the findings highlight the very strong temporal dependency in the predictor variables and the θ_V patterns. The model accuracy is relatable to Meijles et al. (2003) who used slope, topographic index and vegetation as predictor variables in their Dartmoor soil moisture model, explaining 84 % and 82 % of soil moisture variance for ‘dry’ and ‘wet’ states, respectively.

4.7 Implications and conclusions

Grasslands cover 60 % of the agricultural area of the United Kingdom. Some 55 % of this area is covered by permanent pasture that has a history of ploughing, re-seeding and artificial inputs (e.g., slurry, fertiliser and/or lime). The other grasslands are semi-natural grasslands (in the UK uplands described as ‘moorland’) grazed with sheep and cattle but with near-natural soils largely unaffected by ploughing, re-seeding or artificial inputs.

Despite the areal extent of grasslands in the UK, there is virtually no research contrasting the soil moisture differences between permanent pastures and the less intensively managed semi-natural grasslands. *This study, while focused on one experimental plot-pair in Cumbria (upland UK), has demonstrated the intensity of moisture measurements required to highlight the new research needed to explain the contrasting behaviour between improved-pasture and semi-natural grassland if the whole landscape is considered.*

The key findings were:

- The contrast in soil moisture patterns between the paired-plots changed markedly throughout the monitoring period, as did the interactions between the potential controlling variables. During spring sampling (29th May 2018) the improved-pasture was significantly drier than the semi-natural grassland, making the vegetation more sensitive to water stress. With the reduced rainfall and higher transpiration of summer, the moisture content of the semi-natural grassland plot reduced to only 23 %. In some contrast, moisture added in the form of cattle slurry maintained topsoil moisture at ~30 % in the improved-pasture, underlining an overlooked agronomic benefit of slurry. As these slurry additions have consequences for water-quality, a desire to restore wildlife habitats could see this practice barred. As the improved-pasture was significantly drier than the semi-natural grassland prior to slurry additions, such a change could amplify the drying of improved-pasture soils during drought conditions. Research is needed to demonstrate how dry improved-pastures could become without slurry additions, and whether soil restoration techniques could successfully moderate such conditions. The smaller dataset of dry bulk-density and porosity did indicate that the improved-pasture might have been compacted by agricultural practices. Such new research might therefore, include intensive sampling of soil properties, and subsequent modelling of the combined factors to show their role in the moisture status of improved-pastures during droughts (and indeed during floods). Additional hydrological variables such as infiltration-capacity, evapotranspiration, and hydraulic roughness, would further improve system interpretation.

- With the onset of autumn storms, the slurry-wetted improved-pasture continued to be wetter than the semi-natural grassland, being much closer to saturation (i.e., 3.6 % versus 13.3 % below saturation, respectively). These wetter antecedent conditions could mean that such improved-pastures saturate quicker and consequentially produce more of the rapidly moving saturation-excess overland flow, and so heighten downstream flood-risk. Experimental research is needed to quantify if a greater mean wetness of slurry-managed improved-pasture soils in the autumn does translate into a greater incidence, magnitude and speed of SOF. Given the extensive nature of permanent pastures in the high rainfall areas of upland UK, experimental research into measures that would reduce the incidence, magnitude and speed of SOF (arising from saturated topsoil conditions) on or immediately downslope of improved-pastures is also needed. Such work would need to combine paired moisture plots and volumetric overland flow measurements. Visual observations during a storm event during high antecedent moisture conditions (29th Nov 2018) did show that the largely saturated semi-natural grassland did generate overland flow. This underlines the importance of new detailed measurements of soil moisture and overland flow in both improved-pasture and adjacent moorland conditions, so as not to assume that moorland restoration will completely remove overland flow incidence as part of 'Natural

Flood-Risk Management' interventions (see Kirkby and Morgan (1980) for overland flow quantification).

- The high intensity of soil moisture sampling in the paired-plots highlighted the linear connectivity of zones of wetter soils in the semi-natural grassland (in a SSW-NNE direction) that was 'smeared out' within the improved-pasture as a result of the history of ploughing, re-seeding, land drainage etc. This may explain the much longer correlation lengths and stronger spatial-structure observed for the improved-pasture, especially at increased moisture contents. This highlights how farming and changes in floristic/faunal composition have altered the hydrological diversity of an area. The high proportion of semi-variance at the smallest measurement separation (i.e., 1 m) demands further research conducted at even smaller separation distances (e.g., 0.1 m). This research would determine if the cause is the presence of deterministic spatial-structure at sub-1 m distances or intrinsic errors in the moisture measurement technique. Significantly increasing the sampling intensity above that used in this study (i.e., 1536 moisture measurement per plot-pair) would make it difficult to sample the whole area without the average moisture content having changed over the duration of the sampling period. Potentially, multiple moisture-probes would need to be used synchronously for each plot-pair, and each probe used cross-calibrated.

The paired-plot experimental design with a dense grid of soil moisture measurements has provided clear evidence for a contrasting soil moisture regime between intensively managed improved-pasture versus the semi-natural soil-vegetation conditions prevailing at the studied upland locality in Cumbria (UK); and that the relationship changes markedly through the year. The detailed story of within-plot behaviour acts as a basis for detailed replication to understand plot-scale variability across the landscape. The challenge is to replicate this work at numerous locations in this mountainous region to understand which of the contrasts observed in one plot-pair dominates in this landscape. Of equal importance is the extensive replication of the work in other very different locations of permanent pasture in the UK and overseas. Given the limited viability of plot-scale representativeness at the landscape-scale, a body of further research is required before results and conclusions can be applied to regional-scale models.

4.8 Conflict of interest

The authors declare no conflict of interest.

4.9 Acknowledgements

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4.10 Data availability

The data that supports the findings of this study are available at: https://dx.doi.org/10.17635/lancaster/researchdata/331.

4.11 The spatiotemporal dynamics of surface soil moisture within upland grassland ecosystems

Wallace, E.E. and Chappell, N.A. (2020). The spatiotemporal dynamics of surface soil moisture within upland grassland ecosystems, EGU General Assembly 2020, Online, 4th – 8th May 2020, EGU2020-8798, https://doi.org/10.5194/egusphere-egu2020-8798, 202.

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4.12 Brief introduction to paper

This paper was presented at the European Geosciences Union (EGU) on the 6th May 2020. The conference was hosted in Vienna, Austria, although presented remotely due to travel restrictions imposed by the Covid-19 outbreak. The paper was presented as part of the Hydrological Sciences (HS) division, at the session titled: Spatio-temporal and/or (geo) statistical analysis of hydrological events, floods, extremes, and related hazards.

4.13 The spatiotemporal dynamics of surface soil moisture within upland grassland ecosystems (conference paper)

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Land conversion from semi-natural grassland to intensively managed improved-pasture (for sheep, beef and dairy production) has altered the near-surface, soil moisture regime across much of the uplands of Europe. This widespread conversion has modified both the temporal distributions and spatial-structure of surface volumetric wetness, thus affecting the incidence of flood-producing overland flow and resilience of the grasslands to drought stresses. In order to investigate these spatiotemporal dynamics, an intensive fieldwork campaign captured high-resolution (1 m^2) surface volumetric wetness from a 1536 m^2 paired-plot monitored over a year including both drought and fully saturated conditions. The measurements and combined statistical and geostatistical analyses form part of integrated studies into the hydrological effects of agricultural interventions to mitigate floods in the Cumbrian mountains of the UK.

The intensive monitoring highlighted significant temporal variations between land-uses. The improved-pasture dried faster than the semi-natural grassland with the onset of a severe drought, but these effects were more than offset by the application of livestock slurry. This artificial wetting did however produce a more rapid build-up of moisture in the improved-pasture with autumn storms. The large rain-event of Storm Diana (28^{th} – 29^{th} Nov 2019) did, however fully saturate both the improved-pasture and semi-natural grassland to generate visible saturation-excess overland flow. Seasonal changes in the spatial patterns of soil volumetric wetness were equally evident. The semi-natural grassland contained significantly larger variation within soil moisture statistical distributions and substantially larger coefficients of variation compared to the improved-pasture throughout the study. Very weak spatial-structure was observed within the semi-natural grassland. Conversely, a relatively strong spatial-structure was observed within the improved-pasture plot, which intensified with saturation, suggesting farming practices (ploughing, reseeding, artificial inputs, etc.) have removed natural soil moisture variability and encouraged moisture redistribution. A geostatistical model showed that the weak semi-natural grassland spatial-structure remained relatively stationary, whereas the improved-pasture showed extreme non-stationarity, with increasing saturation causing a gradual transition from an exponential to a gaussian geostatistical relationship.

The work highlights the complexity of spatiotemporal soil moisture dynamics taking place at the metre- to decimetre-scales through wetting-and-drying cycles and the strong impact of improved-pasture management upon this. It justifies the need for both intensive soil moisture sampling at experimental sites, sampling across seasons, and the need for combined statistical and geostatistical analyses. Further such analyses in the uplands of Europe are needed if we are to better understand the effects of grassland management on flood and drought hydrology, and to use this knowledge to mitigate our impacts on floods and droughts.

5 BLADE AERATION

Wallace, E.E. and Chappell, N.A. (2019). Blade Aeration Effects on Near-Surface Permeability and Overland Flow Likelihood on Two Stagnosol Pastures in Cumbria, UK. *Journal of Environmental Quality*, 48(6), pp.1766-1774. Doi: 10.2134/jeq2019.05.0182.

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5.1 Brief introduction to paper

As seen in Chapter 4, semi-natural grassland conversion and subsequent management has enabled improved-pastures to become widespread throughout the United Kingdom, including within the Eden catchment. These pastoral soils are typically under intensive agricultural pressures such as drainage, grazing, slurrying, agricultural traffic etc. These practices often combine with the harsh climate and natural soil conditions to degrade the soil surface over time, reducing soil macroporosity and hence the infiltration-capacity and permeability of such areas. This can increase the likelihood of overland flow, heightening both flood-risk and the threat to local water-quality.

Soil-loosening techniques for ameliorating compaction between 15 cm – 35 cm (sward-lifting) or deeper (sub-soiling) do exist, however, these are unsuitable for the near-surface compaction as usually seen in livestock systems following over-grazing or grazing/trafficking in wet conditions. The legs of deep tillage equipment can often be damaged by the till-rich subsurface of Cumbrian soils, alongside bringing stones to the improved-pasture surface, which hinders future silaging. Deep tillage equipment additionally provides a hazard to historic and largely unmapped field-drainage systems (tile-drains, mole-drains, etc.).

Aerators are extremely robust tillage machinery that target near-surface (0-15 cm) soil degradation within grasslands, with minimal risk to field-drainage infrastructure. Aerators are already widely used within many mechanised agricultural settings, including within Cumbria, in order to encourage sward yield by penetrating grass root-mats, which encourages new root growth and may simultaneously facilitate the infiltration of both rainfall and slurry. Aerators may therefore be seen as a potential improved-pasture intervention for flood-risk and water-quality improvements. These devices additionally can be applied both uniformly throughout an improved-pasture, as well as can target specific problematic subsections of a field.

Objective 2) Quantify the effect of blade aeration within several agriculturally-intensive improved-pasture and silage fields on temporal saturated hydraulic conductivity (K_{sat}) and related soil penetration resistance (PR) of the topsoil. Saturated hydraulic conductivity will be used to investigate changes in the amount of infiltration-excess overland flow during extreme precipitation events generated from aerated and unaerated improved-pasture, by contrasting a high-frequency precipitation time-series with K_{sat} values.

5.2 Blade aeration effects on near-surface permeability and overland flow likelihood on two Stagnosol pastures in Cumbria, UK

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5.2.1 Core ideas

- Aeration can significantly increase topsoil permeability and reduce compaction.
- Aeration can substantially lower the likelihood of infiltration-excess overland flow.
- Aeration may be ineffective on impermeable subsoils or highly compacted sites.
- *Ex-situ* permeability results may have limited application within aeration research.
- Combined before-after-control-impact and paired-plot approaches are advised for future aeration studies.

5.3 Abstract

Overland flow from permanent pastures is believed to be a rapid pathway to the drainage network and potentially contributes to flooding within numerous grassland regions of the world. Studies investigating whether aeration can reduce observed overland flow have revealed mixed findings. To improve process interpretation within these studies, topsoil saturated hydraulic conductivity (K_{sat}) and penetration resistance (PR) were measured at two permanent Stagnosol (Aquic soil) improved-pastures (FP1 and FP2) within Cumbria, UK, after blade aeration to 10 cm. Results were measured 2-, 6-, 13-, and 21-weeks post-aeration and compared with the local rainfall record to assess the impact on IOF likelihood (when rainfall intensity exceeds soil infiltration-capacity). Within FP1, aeration significantly increased K_{sat} by up to a factor of 7.5 and caused several significant reductions in PR between 5 and 15 cm. Aeration decreased the IOF likelihood during the 13- and 21-week sampling dates, reducing IOF likelihood from up to 11.4 % of rainfall periods pre-aeration to 0.0926 % of rainfall periods post-aeration. Aeration within FP2 revealed no significant increases in K_{sat}, and no PR change besides a significant increase at 10 cm. The IOF likelihood was virtually identical between aerated and unaerated treatments within FP2. The study

highlights that aeration can significantly improve K_{sat} and PR, as well as substantially reduce the likelihood of IOF generation, although benefits can be site specific.

5.4 Introduction

Extensive soil compaction is hypothesised to increase flood-risk across numerous regions of the globe (Alaoui et al., 2018). Within the United Kingdom, 60 % of managed improved-pasture in England and Wales exhibits signs of topsoil compaction and/or surface capping (AHDB, 2016). Topsoil compaction can severely impede water infiltration and drainage due to reduced soil pore volumes, thereby altering the distribution, frequency, and continuity of water-transmitting macropores within the soil matrix (Kuncoro et al., 2014). This pore network restructuring can increase the likelihood of IOF during precipitation events. Infiltration-excess overland flow is generated when rainfall intensity exceeds soil infiltration-capacity. Infiltration-capacity is the flow of water into saturated soil under unit cross-sectional area and unit hydraulic gradient. Infiltration-excess overland flow is often a rapid drainage pathway and increases the likelihood of channel capacity being exceeded, creating flooding (see Horton, 1933).

Topsoil compaction reduces improved-pasture productivity by restricting sward root aeration (Davies et al., 1989; Douglas et al., 1995). This compaction is often caused by livestock grazing in wet conditions (see Drewry et al., 2000a), as well as by farm traffic (see Bhogal et al., 2011). Slit aeration to 10 to 15 cm using a blade aerator is a practice commonly adopted by UK livestock farmers to aerate improved-pasture for increased sward production (Davies et al., 1989; Bhogal et al., 2011). This practice has the potential co-benefit of enhancing topsoil permeability (Davies et al., 1989; Crawford and Douglas, 1993; Douglas et al., 1995). Enhanced permeability (infiltration-capacity) within improved-pastures can potentially minimize IOF, thus reducing the fast drainage pathway (O'Connell et al., 2007a), alongside reducing agrochemical losses carried within surface flows (Van Vliet et al., 2006). Mechanical slit aeration (blades or tines) has been studied in relation to changes in overland flow within the United States (Shah et al., 2004; Franklin et al., 2006, 2007; Butler et al., 2008; De Koff et al., 2011) and Canada (Van Vliet et al., 2006) with mixed results (Table 5.1). Shah et al. (2004) found aeration did not significantly reduce rainfall-induced overland flow, although significant reductions were found when combined with liquid dairy manure application. Franklin et al. (2006) found no significant overland flow reductions after aeration when incorporating inorganic fertilizers and broiler litter. Van Vliet et al. (2006) found that annual winter overland flow from aerated plots significantly decreased by 47 to 81 % compared with unaerated plots over a 4-yr study in British Columbia, Canada. Franklin et al. (2007) found that aeration significantly reduced overland flow, although effects were soil dependent. Butler et al. (2008) found that aeration failed to significantly alter overland flow under natural soil conditions, after various fertilizer application methods, and after artificial compaction. De Koff et al. (2011) highlight aeration to occasionally decrease overland flow volumes significantly, with some significant increases in infiltration rate (based on subtracting overland flow from rainfall). These conflicting findings (Table 5.1) highlight the need for a greater understanding of the processes governing overland flow generation after slit aeration. No research directly examines how slit aeration alters topsoil permeability (infiltration-capacity), which is the pivotal IOF controlling parameter. This study addresses this key evidence gap at two nearby improved-pastures, having a soil type that is considered highly susceptible to compaction, restricted aeration, and overland flow, namely, a Clifton Association Stagnosol.

Table 5.1: A list of available studies that have investigated the effect of mechanical slit aeration on changes in overland flow volumes.

Author	Study site	Soil type and texture	Rainfall mechanism	Rainfall depth	Rainfall intensity	Absolute overland flow change	Magnitude of overland flow change	Study notes
Shah et al., 2004	West Virginia, USA	Ultic Hapludalf (silt loam)	Simulated (6 events)	28 mm to 65 mm per event	120 mm hr^{-1}	-3 mm to +5 mm (aeration) -7 mm to +5 mm (aeration and aeration with liquid dairy manure)	-23 % to +45 % (aeration) -50 % to +133 % (aeration and aeration with liquid dairy manure)	
Franklin et al., 2006	Georgia, USA	Aquic Hapludult (sandy-loam)	Simulated (2 events)	Not given	50 mm hr^{-1}	-6.2 mm to +1.9 mm (aeration with broiler litter and inorganic fertiliser)	-32 % to +10 % (aeration with broiler litter and inorganic fertiliser)	
Van Vliet et al., 2006	British Columbia, Canada	Aquic Dystroxerept (silt loam-sandy loam)	Natural (56 events)	1185 mm y^{-1} average (4740 mm total)	Not given	-67.2 mm to -0.76 mm annually (aeration with liquid dairy and swine manure)	-81 % to -47 % (aeration with liquid dairy and swine manure)	OF grouped from October-April over four years
Franklin et al., 2007	Georgia, USA	Typic Kanhapludults, Aquic Hapludults, Aquultic Hapludalfs (sandy loams)	Natural (133-203 events)	1037 mm y^{-1} unaerated average, 1051 mm y^{-1} aerated average.	Up to ~100 mm per event.	Not given.	-35 % to +3 % (aeration)	Before and after rather than paired plot approach

Table 5.1 (continued):

Author	Study site	Soil type and texture	Rainfall mechanism	Rainfall depth	Rainfall intensity	Absolute overland flow change	Magnitude of overland flow change	Study notes
Butler et al., 2008	Georgia, USA	Typic Kanhapludult (sandy loam)	Simulated (2 events)	Not given	85 mm hr^{-1}	Not given	-20 % to +18 % (aeration) -31 % to +25 % (aeration with broiler litter and slurry manure)	Pre and post compaction OF experiments
De Koff et al., 2011	Arkansas, USA	Typic Hapludults and Glossaquic Fragiudults (silt loams)	Simulated (11 events)	Not given	70 mm hr^{-1}	-36.7 mm to -4.6 mm (aeration). -50.3 mm to +17.4 mm (aeration with poultry litter and swine slurry)	-74 % to -8 % (aeration) -75 % to +16 % (aeration with poultry litter and swine slurry)	

The objectives of this study were:

I) To ascertain if blade aeration reduces soil penetration resistance and increases topsoil permeability in two nearby FAO Stagnosol permanent pasture replicates.

II) To assess blade aeration effects on the likelihood of IOF generation by comparing statistical distributions of topsoil permeability with 847,320 values that comprise a 25-yr record of 15-min observed local rainfall intensity.

5.5 Methodology

5.5.1 Study site

Measurements were taken within two reseeded permanent pastures dominated by perennial ryegrass (*Lolium spp.*), situated 9 km north of Penrith in Cumbria, UK, between June and November 2018. The local climate is wet temperate, with a mean winter temperature of 4.9 °C and a mean summer temperature of 12.1 °C (Met Office, 2016). The 1990 to 2018 average annual rainfall is 1050 mm at Skelton, located 4.5 km south-west of the experimental site.

The experimental site has plots within two nearby fields (Field FP1 and Field FP2: for field pasture; Figure 5.1). The centre of the FP1 plot is ~600 m from the centre of the FP2 plot (54°44′00″ N, 2°49′00″ W, and 54°44′09″ N, 2°48′36″ W, respectively). Both fields are mapped regionally as comprising the same broad soil type, namely the 711n Clifton Soil Association (Jarvis et al., 1984). This equates to the FAO Stagnosol soil group (WRB, 2015), which is a soil with Aquic properties within several USDA soil orders (USDA, 1999).

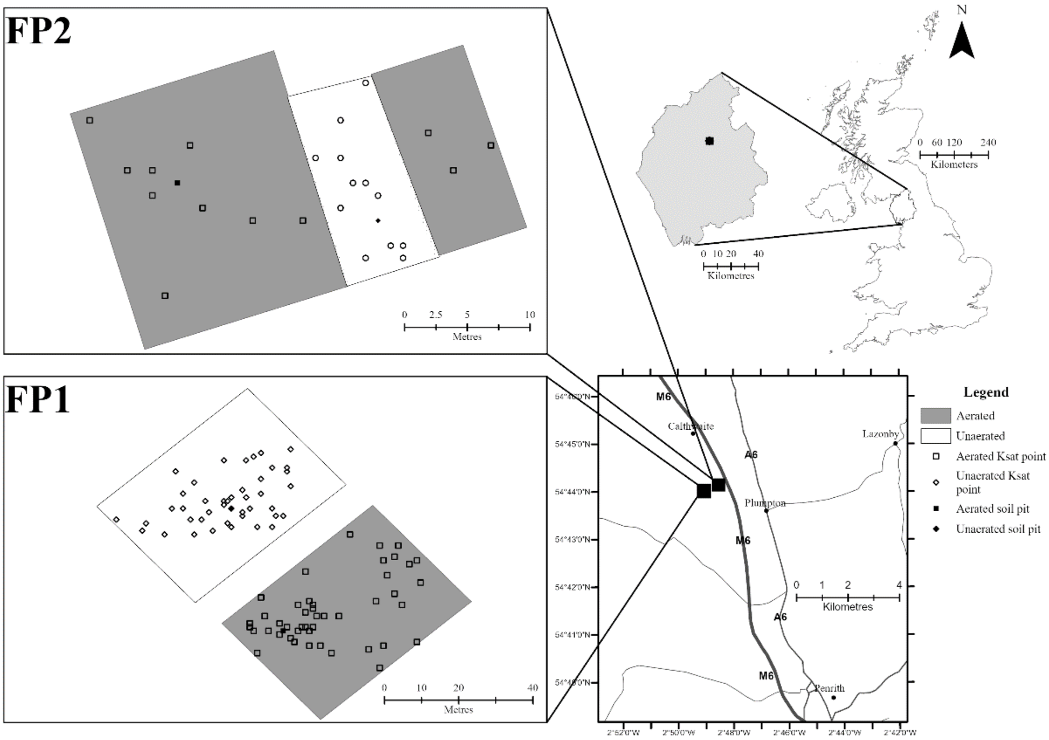


Figure 5.1: The location of the study site within a UK, Cumbria and local area context. The study site is within the Petteril sub-catchment of the Eden catchment. The aerated and unaerated regions of Field Pasture 1 (FP1) and Field Pasture 2 (FP2) are shown, with georeferenced locations of permeability (K_{sat}) and each soil pit. Contains Ordnance Survey (OS) data.

Field plots FP1 (456 m^2) and FP2 (232 m^2) are permanently grazed improved-pasture and silage fields and receive heavy vehicular passes during silage cutting and slurry application. The land manager stated that neither has been ploughed or aerated in recent years. Both sites have 4 % slopes, although FP1 is at the base of a slope and FP2 is near a summit. Both fields belong to similar pastoral management systems and were continually grazed throughout the experiment, with FP2 grazed by sheep (8 ha^{-1}) and dairy cattle (0.75 ha^{-1}), and FP1 solely sheep grazed (15 ha^{-1}). As a result, the sampling sites were selected due to being mapped as the same soil type and having fairly similar management practices and are considered replicates within the study.

During the final sampling experiment, 0.6-m-deep soil pits were manually excavated at random locations within aerated (FP1A and FP2A) and unaerated plots (FP1N and

FP2N, Figure 5.1), for the determination of reference soil properties. Soil was extracted using a 221-cm^3 bulk-density cylindrical ring at 5-cm increments from the soil surface to 20 cm, totalling four soil samples per reference pit. Samples were oven dried at 105 °C for 24 h for ρ_b calculation. Soil organic matter content was determined from the bulk-density cores via a 550 °C 6-h loss-on-ignition test. Particle size analysis involved sieving oven-dried soil through a 2000-μm sieve, mixing the sample with 1 % sodium polymetaphosphate for 24 h, followed by high-power sonication for 3 min and laser diffraction (Beckman Coulter, LS-13-320).

5.5.2 Aeration treatment

The experiment began in June 2018 during an atypically dry summer (Supplemental Table 5.S1). Each of the two replicates (FP1 and FP2) was randomly divided into two areas – one with blade aeration to a depth of 10 cm (denoted as sub-area “A” in Field Pasture names), and a control (unaerated, denoted as sub-area “N” in Field Pasture names). Aeration was applied on 11th June 2018, using a Ritchie 863G 3M (Ritchie Agricultural) blade aerator. The aerator operates two in-series rotor shafts; each with nine rotatable discs that individually have three blades angled 120° apart. Discs are spaced 23 cm apart within each rotor shaft, with a 17.5-cm gap between rotors. The 475-kg aerator was fully ballasted (with an additional 700 kg) during operation to increase blade ground penetration and traversed the replicates at an approximate rate of 1 ha h^{-1}. No markings or blade insertion paths were visible on the sward or soil surface 2 weeks post-treatment.

5.5.3 Field measurements

A total of 1366 PR and 114 K_{sat} measurements were taken in the plots FP1A, FP1N, FP2A, and FP2N via random sampling throughout the experiment. Samples were taken 2, 6, 13, and 21 weeks post-aeration (Supplemental Table 5.S1; Supplemental Figure 5.S1).

Soil penetration resistance was measured using an SC900 Field Scout soil compaction meter using a 12.8-mm-diameter cone. The device measures PR via an internal load cell and has a maximum load capacity of 9000 kN. An ultrasonic depth sensor recorded measurement depths at 2.5-cm increments to a depth of 15 cm. The PR samples were taken randomly throughout each replicate, with efforts made to avoid disturbed soil or aeration slits. Measurements that exceeded meter capacity were also recorded.

Topsoil permeability was measured using a Talsma ring permeameter (see Talsma, 1960; Bonell et al., 1983; Chappell and Ternan, 1997). This constant-head technique gives measurements of the coefficient of permeability, also called the saturated hydraulic conductivity. Permeability was calculated via Darcy's law (Equation 2.4.1) once steady-state was achieved through the soil core. The procedures detailed in Chappell and Ternan (1997) were followed exactly except that the 10-cm-deep soil core was tested while inserted into the ground. This modification to the technique was so that vertical water percolation out of any 10-cm-long slits within the core was into underlying soil (see Sherlock et al., 2000).

5.5.4 Statistical analysis and modelling

The K_{sat} frequency distributions are expected to be strongly positively skewed (Baker, 1978; Bonell et al., 1983; Zhai and Benson, 2006). Consequently, it was likely that a nonparametric statistical test was needed (i.e., MWW), and a parametric approach would only be adopted if results satisfied normality. The MWW tests were conducted within the MATLAB programming environment (Mathworks), using the *ranksum* function with significance levels of p≤0.05, p≤0.01, and p≤0.001.

Statistical comparisons of the permeability frequency distributions from each replicate were made with local rainfall intensities to estimate IOF likelihood to create a POT statistical model. A 25.5-yr precipitation time-series recorded at a 15-min resolution from the Skelton rain-gauge (54°42′59′′ N, 2°52′38′′ W) was compared with the summary statistics (minimum, 10th percentile, lower quartile, median, and upper quartile) of K_{sat}, to account for climatic variability within IOF likelihoods. The rain-gauge is 4.3 km south-west of FP1 and 4.8 km south-west of FP2. Both replicates are considered to have identical rainfall inputs given the prevailing frontal systems.

5.6 Results and discussion

5.6.1 Reference soil properties

Soil pits were excavated in both fields for visible soil characterisation and sampling to determine reference soil properties. At the FP1A pit (Figure 5.1), an O-horizon extended to 3 cm. An A-horizon existed between 3 and 12 cm but was weakly defined from the B-horizon. The B-horizon extended to 45 cm and contained redoximorphic features, before a well-defined sandy C-horizon. The FP1N pit similarly had an O-horizon to 3 cm and an A-horizon from 3 to 10 cm. The B-horizon was visibly denser and stonier, extended to 40 to 45 cm, and contained redoximorphic features. The FP2A and FP2N pits had an O-horizon to 5 cm and an A-horizon between 5 and 10 to 12 cm. The B-horizon extended to 35 to 45 cm and was visibly heavier and stonier than the topsoil.

From the physico-chemical analysis, the aerated plots in both replicates had substantially greater medium to very coarse sand contents (40 % – 43 %) in the upper subsoil (15 cm – 20 cm, Tables 5.2-5.3) compared with the unaerated plots (5 % – 12 %). Greater SOM content at the 0- to 10-cm depth where roots are commonly found was observed in the aerated plots, with averages of 9.5 % and 9.5 % for FP1A and FP2A, and 8.4 % and 7.0 % for FP1N and FP2N, respectively. Overall, however, the unaerated plots in FP1 and FP2 had similar reference characteristics, as did aerated plots, although minor differences were apparent between adjacent pits within the same replicate.

5.6.2 Soil penetration resistance differences between aerated and unaerated plots (0–15 cm: Objective I)

A total of 1366 PR tests were undertaken across all replicates. Some 45 % of measurements were too dense (i.e., PR exceeded 7000 kPa) to give a reading (Table 5.4). The two replicates were very different in the numbers of tests that exceeded 7000 kPa. In FP2, some 76 % of tests on aerated topsoil and 75 % on unaerated soil exceeded 7000 kPa during the first measurement date, whereas this was only 35 % of aerated topsoil tests and 53 % of unaerated topsoil tests in FP1 (Table 5.4). This implies that FP2 had more pockets of either (i) compacted topsoil or (ii) drier topsoil than FP1.

As the experiment progressed, PR failures and measurable PR generally decreased as soils likely became increasingly saturated (Table 5.4; Supplemental Figure 5.S1; Cotching and Belbin 2007). Pasture FP1A had noticeably fewer test failures and lower PR than FP1N throughout the experiment, whereas PR values and test failures within FP2 were almost identical between treatments.

Measurable PR highlights significant differences in FP1 between treatments 2 weeks post-aeration (Table 5.4), at 5 ($p \leq 0.001$), 7.5 ($p \leq 0.004$), and 10 cm ($p \leq 0.007$). The

Table 5.2: Physico-chemical properties of the two soil pits in Field Pasture 1 (FP1). Variables included particle-size distribution, soil texture, soil pH, soil organic matter (SOM), and soil dry bulk-density (ρ_b).

Land Use	Depth (cm)	Particle Size Distribution (%)					Soil Texture	pH	SOM (%)	ρ_b (g cm^{-3})
		≤ 2 µm	2-20 µm	20-60 µm	60-200 µm	200-2000 µm				
FP1 (A)	**0-5**	2.7	11.0	17.6	40.8	27.8	Sandy loam	5.87	11.4	1.01
FP1 (A)	**5-10**	4.6	18.2	22.6	38.6	15.9	Sandy loam	5.82	7.7	1.15
FP1 (A)	**10-15**	4.0	17.3	21.5	34.3	22.8	Sandy loam	5.97	7.7	1.04
FP1 (A)	**15-20**	2.6	10.3	13.5	30.8	42.8	Sandy loam	6.12	7.0	1.32
FP1 (N)	**0-5**	4.3	17.9	17.9	27.8	32.1	Sandy loam	5.91	9.1	1.09
FP1 (N)	**5-10**	3.2	12.6	11.3	19.8	53.0	Sandy loam	5.96	7.1	1.11
FP1 (N)	**10-15**	11.3	41.6	23.7	22.0	1.3	Silt loam	6.12	4.7	1.87
FP1 (N)	**15-20**	8.5	34.2	25.3	27.4	4.6	Silt loam	5.67	4.8	1.33

Table 5.3: Physico-chemical properties of the two soil pits in Field Pasture 2 (FP2). Variables included particle-size distribution, soil texture, soil pH, soil organic matter (SOM), and soil dry bulk-density (ρ_b).

Land Use	Depth (cm)	Particle Size Distribution (%)					Soil Texture	pH	SOM (%)	ρ_b (g cm^{-3})
		≤ 2 µm	2-20 µm	20-60 µm	60-200 µm	200-2000 µm				
FP2 (A)	**0-5**	2.7	11.7	15.6	40.1	29.8	Sandy loam	5.92	12.8	0.79
FP2 (A)	**5-10**	4.6	20.4	24.5	38.7	11.8	Sandy loam	5.98	6.2	1.42
FP2 (A)	**10-15**	6.6	29.0	29.2	31.7	3.5	Silt loam	6.23	4.2	1.09
FP2 (A)	**15-20**	4.7	19.1	14.1	22.3	39.8	Sandy loam	6.73	4.3	1.67
FP2 (N)	**0-5**	5.7	24.1	20.7	24	25.5	Sandy loam	6.43	9.2	1.17
FP2 (N)	**5-10**	3.5	16.3	19.4	36.4	24.3	Sandy loam	6.83	4.8	1.18
FP2 (N)	**10-15**	4.1	17.3	21.3	35.5	21.8	Sandy loam	6.94	5.9	1.18
FP2 (N)	**15-20**	5.7	23.5	24.5	33.9	12.4	Sandy loam	6.94	3.3	1.52

Table 5.4: Soil penetration resistance statistics, including the percentage of successful measurements, median and mean penetration resistance, and the Mann-Whitney-Wilcoxon p-values between treatments in Field Pasture 1 and Field Pasture 2. Note that the median values are compared in the Mann-Whitney-Wilcoxon tests.

Site and date	**Statistics**	**Treatment**	**n**	**Depth (cm)**						
				0	**2.5**	**5**	**7.5**	**10**	**12.5**	**15**
FP1 (Week 2)	Successful Measurements (%)	Aerated	110	65	65	65	65	65	51	35
		Unaerated	110	47	47	47	47	44	18	9
	$\tilde{x}$ (kN)	Aerated		225	1242	1932	2312	2605	2450	2881
		Unaerated		276	1622	2518	2726	3105	2967	2881
	$\bar{x}$ (kN)	Aerated		612	1290	1876	2283	2545	2573	2887
		Unaerated		662	1504	2384	2690	3060	3071	3215
	MWW (p)			*0.818*	*0.195*	*0.001****	*0.004***	*0.007***	*0.144*	*0.521*
FP1 (Week 6)	Successful Measurements (%)	Aerated	143	74	74	74	74	73	66	52
		Unaerated	142	44	44	44	44	43	25	13
	$\tilde{x}$ (kN)	Aerated		552	1035	1087	1138	1346	1690	1828
		Unaerated		742	1138	1518	1828	2380	2674	2967
	$\bar{x}$ (kN)	Aerated		604	983	1101	1220	1475	1754	1925
		Unaerated		765	1111	1560	1934	2406	2817	3047
	MWW (p)			*0.156*	*0.104*	*0.001****	*0.001****	*0.001****	*0.001****	*0.001****
FP1 (Week 13)	Successful Measurements (%)	Aerated	150	77	77	77	77	77	75	73
		Unaerated	130	67	67	67	67	66	49	44
	$\tilde{x}$ (kN)	Aerated		345	552	552	552	621	690	724
		Unaerated		207	656	724	897	1329	1622	1690
	$\bar{x}$ (kN)	Aerated		374	563	558	566	634	740	812
		Unaerated		366	676	751	941	1382	1689	1755
	MWW (p)			*0.522*	*0.006***	*0.001****	*0.001****	*0.001****	*0.001****	*0.001****

Table 5.4 (continued):

Site and date	Statistics	Treatment	n	Depth (cm)						
				0	**2.5**	**5**	**7.5**	**10**	**12.5**	**15**
FP1 (Week 21)	Successful Measurements (%)	Aerated	103	94	94	94	94	94	91	83
		Unaerated	93	71	71	71	71	71	58	47
	$\tilde{x}$ (kN)	Aerated		207	380	448	483	552	621	724
		Unaerated		310	448	380	448	621	897	1070
	$\bar{x}$ (kN)	Aerated		242	389	437	474	575	713	792
		Unaerated		314	448	409	516	822	1039	1086
	MWW (p)			*0.031**	*0.105*	*0.232*	*0.750*	*0.003***	*0.001****	*0.001****
FP2 (Week 2)	Successful Measurements (%)	Aerated	192	24	24	24	24	22	14	2
		Unaerated	193	25	25	25	25	23	13	3
	$\tilde{x}$ (kN)	Aerated		69	310	2380	3140	3864	4106	3812
		Unaerated		121	880	2070	2933	3191	2932	3847
	$\bar{x}$ (kN)	Aerated		553	1021	2107	2871	3558	3782	3829
		Unaerated		515	965	2120	2693	2963	3021	3565
	MWW (p)			*0.315*	*0.990*	*0.442*	*0.263*	*0.014**	*0.067*	*0.914*

aerated site had a lower average PR (1876 vs. 2384 kN, 2283 vs. 2690 kN, and 2545 vs. 3060 kN) compared with FP1N, for 5, 7.5, and 10 cm, respectively. The lower PR within FP1A persisted with repeated monitoring, with. significant differences at 5 to 15 cm in Week 6, 2.5 to 15 cm in Week 13, and 0 cm and 10 to 15 cm in Week 21. Contrastingly, within FP2, FP2A had either equivalent or slightly higher PR in comparison with FP2N, with the aerated plot being significantly more compacted ($p \leq 0.014$) at the 10-cm depth.

5.6.2.1 Interpretation and explanation 1

Two alternative explanations are proposed to explain PR findings, although other interpretations are possible. The first explanation (Explanation 1) is that aeration caused PR improvements within FP1 but was ineffective within FP2. Slit aeration is believed to alleviate soil compaction through the soil-loosening effects of the rolling blades/tines (Davies et al., 1989; Douglas et al., 1995), so it may have lowered PR within FP1. Slits may produce preferential infiltration (both rainfall and slurry: Crawford and Douglas, 1993; Douglas et al., 1995), which may preferentially wet soil around slits and reduce density within aerated plots (see Cotching and Belbin, 2007). Aeration can additionally disrupt dense and established root mats (Bhogal et al., 2011); this encourages new root growth, which could potentially lower PR. The observed SOM differences in the 0- to 10-cm soil layer between treatments may indicate that aeration increased root growth at both replicates (see Section 5.6.1).

The soil loosening, preferential infiltration (particularly during dry conditions), and root mat disruption and root growth may combine to create a favourable earthworm environment, enhancing earthworm activity and reducing PR. Eggleton et al. (2009), in a UK study, showed strong declines in earthworms during dry periods, and relative increases during saturated conditions, showing earthworm abundance and resultant bioactivity is strongly linked to soil moisture. Furthermore, Capowiez et al. (2009) demonstrate that by adopting reduced-compaction agricultural practices, earthworm colonisation can increase by an average of 20 %.

These albeit untested hypotheses complement Douglas et al. (1995), who found blade aeration to reduce topsoil ρ_b in Scotland. Alternative soil loosening devices are capable of reducing improved-pasture density. These include subsoilers, which operate at 35 to 50 cm to remove deep compacted layers (Harrison et al., 1994), and sward lifters, which operate at 15 to 35 cm (Newell Price et al., 2015). Subsoilers and sward lifters target deeper compaction than blade aerators and are mostly used to relieve compaction from heavy machinery (Bhogal et al., 2011).

The limited PR difference within FP2 could be due to ineffective aeration, soil recompaction (possibly caused by cattle), soil textural disparity, no established root mat, and/or a sparser or less mobile earthworm population. Crawford and Douglas (1993) demonstrate that progressively drier soil causes shallower and less effective aeration. It was not apparent during treatment that replicates had inherently different soil moisture contents, and consequentially soil moisture measurements were not undertaken. It is possible, however, that replicates were at different saturations, and future researchers are advised to record soil moisture during aeration. Pasture FP2 may have been drier than FP1 due to being toward the summit as opposed to the base of a slope, and more recent slurry wetting within FP1 may have increased antecedent soil moisture. This potentially lower antecedent soil moisture content could have reduced blade penetration within FP2.

The higher PR baseline (FP1N vs. FP2N) and denser B-horizon within FP2 may additionally reduce aerator penetration depth (see Davies et al., 1989; Douglas et al., 1995). The higher PR baseline within FP2 could be caused by dairy cattle, which may compact the topsoil so it is resistant to aeration, and/or rapidly remove aeration improvements. Drewry et al. (2000a) found that dairy cattle significantly increased topsoil ρ_b by one third when comparing 97 sheep farms with 87 dairy farms in New Zealand, and also noted that dairy farms had significantly lower K_{sat} at 0 to 5 and 10 to 15 cm. It is also possible that FP2 improvements were not apparent because FP2A was intrinsically more compact due to greater silt concentrations (see Tables 5.2-5.3).

Indirectly observed factors could also have prevented PR improvement in FP2. Higher observed SOM content within FP2A (9.5 %), as opposed to FP2N (7 %), suggests that although new root growth may be occurring (not directly measured in this study), this has not reduced PR. This is potentially due to the lack of a dense root mat preceding aeration, as shown by the low SOM within FP2N. The higher PR baseline within FP2 may additionally inhibit earthworm motility and therefore their ability to reduce soil density and resultant PR (Capowiez et al., 2009; 2014). Results support studies such as Van Vliet et al. (2006), who found no improvements to ρ_b following blade aeration.

5.6.2.2 Interpretation and explanation 2

The alternate explanation (Explanation 2) is that a combination of the sampling method and natural soil variation falsely indicated aeration to have reduced PR within FP1, when in reality it was effective in neither. The sampling method could have caused lower PR readings within the aerated region of FP1, as the penetrometer may have entered into the aeration slits. However, the lack of visible slits 2 weeks post-aeration makes this an untested hypothesis. The suggested ineffective aeration within FP2 would also explain why PR is highly comparable between treatments, as the aerator may have failed to generate slits within FP2, and the penetrometer therefore may have had either no slits or very shallow slits to enter.

Natural soil variation may also have falsely indicated aeration to have been highly effective in FP1. The denser, stone-rich layer within the B-horizon of FP1N compared with FP1A may have inferred aeration to reduce PR in FP1, as fewer PR measurements within the aerated plot would fail due to stone contact (Davies et al., 1989). Uniform stone coverage throughout FP2 suggests similar PR profiles as was observed. Textural analysis also reveals disparity at the 10- to 15-cm depth between treatments, which may have caused or contributed to the observed differences, at least for deeper measurements.

5.6.3 Topsoil permeability difference between aerated and unaerated plots (Objective I)

A total of 114 K_{sat} tests were undertaken across the replicates (Table 5.5). Two weeks post-aeration in FP1 (Table 5.5), the aerated topsoil permeability was a factor of 3.4 times higher than unaerated topsoil. The difference was statistically significant at $p \leq 0.004$ (Table 5.6). These results contrasted with FP2, where the aerated and unaerated plot had similar K_{sat} values ($p \leq 0.894$).

Repeat measurements (at random locations within the plots) within FP1 shows FP1A to have a larger K_{sat} in Week 6 by a factor of 3.9 ($p \leq 0.002$), in Week 13 by a factor of 5.7 ($p \leq 0.001$), and in Week 21 by a factor of 7.5 ($p \leq 0.024$). Repeated measurements within the same treatment in FP1 showed statistically significant higher K_{sat} between the first (Weeks 2 and 6) and latter two sampling dates (Weeks 13 and 21, Table 5.6). The K_{sat} values derived from the Talsma ring permeametry tests are conducted under a constant applied head and percolation rate through the analysed

Table 5.5: Saturated hydraulic conductivity summary statistics for Field Pasture 1 and Field Pasture 2.

Field Pasture 1									
Sampling Period	**Min (mm/hr)**	**Q10 (mm/hr)**	**Q25 (mm/hr)**	**Median (mm/hr)**	**Q75 (mm/hr)**	**Max (mm/hr)**	**Geometric Mean (mm/hr)**	**Coefficient of Variation**	**n**
Week 2 (A)	304.2	472.7	955.7	1872.8	2746.2	7356.1	1587.4	88.5%	11
Week 2 (N)	143.1	155.0	214.8	459.8	754.0	2107.0	461.5	89.2%	11
Week 6 (A)	1331.6	1572.0	1742.4	2794.1	4300.4	7361.7	2885.0	55.4%	12
Week 6 (N)	226.5	265.0	327.5	501.0	1678.0	4273.8	748.4	105.7%	11
Week 13 (A)	178.4	241.7	458.6	658.6	1188.2	2512.7	672.7	73.5%	13
Week 13 (N)	10.4	37.8	54.4	170.8	331.2	427.2	118.3	82.1%	13
Week 21 (A)	16.8	60.6	206.6	316.0	779.1	886.4	300.8	71.4%	9
Week 21 (N)	3.3	6.7	12.6	21.0	107.9	3633.3	40.0	269.9%	9
Field Pasture 2									
Week 2 (A)	40.5	75.8	132.2	306.8	588.8	1365.5	263.6	93.7%	12
Week 2 (N)	132.0	138.7	172.9	321.6	520.9	928.7	311.4	62.7%	13

Table 5.6: Saturated hydraulic conductivity Mann-Whitney-Wilcoxon tests between aerated and unaerated treatments in Field Pasture 1 and Field Pasture 2.

Field Pasture 1								
Mann-Whitney-Wilcoxon test (p)	**Week 2 (A)**	**Week 6 (A)**	**Week 13 (A)**	**Week 21 (A)**	**Week 2 (N)**	**Week 6 (N)**	**Week 13 (N)**	**Week 21 (N)**
Week 2 (A)	-	0.091	0.013*	0.002**	0.004**			
Week 6 (A)		-	0.001***	0.001***		0.002**		
Week 13 (A)			-	0.144			0.001***	
Week 21 (A)				-				0.024*
Week 2 (N)					-	0.365	0.006**	0.006**
Week 6 (N)						-	0.001***	0.002**
Week 13 (N)							-	0.083
Week 21 (N)								-
Field Pasture 2								
Mann-Whitney-Wilcoxon test (p)	**Week 2 (A)**				**Week 2 (N)**			
Week 2 (A)	-				0.894			
Week 2 (N)					-			

core, only once equilibrium is achieved (i.e., core and below-core conditions are fully saturated). The temporal K_{sat} variation in FP1 may imply, therefore, that soil cracking had developed due to the prolonged dry period preceding treatment (see Supplemental Figure 5.S1); these cracks gradually closed with repeated soil wetting in the subsequent autumn (see Bouma and Dekker, 1978). Topsoil K_{sat} is highly sensitive to crack structure (i.e., secondary porosity or macroporosity) and is known to change as Stagnosols and Gleysols dry or conversely rewet (Chappell and Lancaster, 2007). The marked K_{sat} changes over a 21-week period within FP1N despite no artificial intervention, indicates that before-and-after (before–after–control–impact [BACI]) measurements would have been unsuitable to detect intervention improvements. Results suggest that parallel measurements of unaerated against adjacent aerated plots (paired-plot design) are needed, alongside BACI measurements of aerated plots. Combining the paired-plot and BACI approach in this study would have been very beneficial to the interpretation of results and is therefore recommended for future research.

The two previously proposed explanations can explain why aeration may have increased permeability within FP1 and not FP2, although other interpretations are again possible.

Explanation 1 proposes that aeration was effective within FP1, yet ineffective within FP2. Aeration may have increased FP1 permeability due to a combination of soil loosening, root mat disruption, and/or enhanced soil bioactivity. The soil loosening effect of the blades may contribute to altering soil macroporosity to increase K_{sat}. In addition, the perforation of a root mat potentially enhances water percolation, especially if it is well-established (Bhogal et al., 2011). The proposed improved earthworm environment (see above; Edwards and Lofty, 1977) may also have increased earthworm colonisation within FP1A and improved permeability (see Capowiez et al., 2009, 2014). Results support other published research suggesting that aeration may increase infiltration rates (de Koff et al., 2011) and reduce overland flow (Van Vliet et al., 2006; de Koff et al., 2011). Similarly to PR, sward lifters (Drewry and Paton, 2000; Newell Price et al., 2015) and subsoilers (Harrison et al., 1994) are capable of increasing improved-pasture K_{sat}, although these operate at different depths to blade aeration.

The failure of aeration to increase permeability within FP2 may be due to either ineffective aeration, soil re-compaction, no disruptable root mat, reduced earthworm abundance and activity (see above), or impermeable subsoil nullifying improvements. Impermeable subsoil could restrict K_{sat} improvements within FP2, as subsoil may nullify topsoil improvements if it is the limiting K_{sat} factor. This questions the practical applications of *ex-situ* permeability tests that are commonly adopted during related studies (e.g., Drewry and Paton, 2000; Drewry et al., 2000b), if done on sites with impermeable subsoil (see Chappell and Ternan, 1997; Sherlock et al., 2000). For aeration to reduce flood-risk, it is likely necessary for infiltrated water to percolate vertically through the subsoil rather than follow rapid near-surface flow pathways. Thus, aeration on slowly permeable topsoils that overlie permeable subsoils may produce the greatest flood-mitigation benefit.

Explanation 2 proposes that aeration was entirely ineffective, and natural soil variation falsely indicated aeration to improve K_{sat} within FP1. Stagnosols are typically slowly draining (Jarvis et al., 1984), so slight variation in macrostructure between FP1 treatments could influence readings (see Bouma and Dekker, 1978). However, soil data from FP1 (Table 5.2) suggest only minor textural difference. Furthermore, FP1 plot boundaries are only 10 m apart, with plot centres only 50 m apart (Figure 5.1).

Thus, the supporting soil data and plot proximity dispute this, but it remains possible. Explanation 2 supports several related studies that found aerators to negligibly reduce overland flow (Franklin et al., 2006; Butler et al., 2008). The proposed BACI-paired-plot approach for K_{sat} measurements would test this hypothesis and is therefore recommended for future research.

5.6.4 Permeability comparison with local precipitation intensity (Objective II)

A recent 25.5-yr record (1990–2018, excluding July 1993–March 1997) for the Skelton rain-gauge, comprising of 847,320 values sampled at 15-min intervals, shows that rain occurred during 66,985 of those intervals (7.91 % of the time). The maximum observed rainfall intensity (MORI) for this period was 21.2 mm 15 min^{-1}. Converting the K_{sat} data into millimetres per 15 min and overlaying this with rainfall generates IOF likelihood (where rainfall intensities exceed the topsoil K_{sat}; Supplemental Tables 5.S2 and 5.S3; Horton, 1933).

During Week 2 in FP1 (Supplemental Table 5.S2), the minimum observed K_{sat} for both aerated (76 mm 15 min^{-1}) and unaerated (35.8 mm 15 min^{-1}) plots, exceeds the MORI. This suggests little to no potential for IOF generation at FP1 in either treatment. For Week 2 in FP2 (Supplemental Table 5.S3), six intervals (0.00896 % of rainfall periods) surpass the K_{sat} minimum in the aerated site (10 mm 15 min^{-1}), and one interval (0.00149 % of rainfall periods) surpasses the 10th percentile of the aerated site (19 mm 15 min^{-1}), whereas the minimum K_{sat} within the unaerated region (33 mm 15 min^{-1}) exceeds the MORI. This suggests aeration to potentially cause very minor increases in IOF likelihood within FP2.

Repeat sampling at FP1 (Supplemental Table 5.S2) in Week 6 highlights that virtually no IOF would likely be generated for both treatments, with an aerated minimum K_{sat} of 333 mm 15 min^{-1}, and an unaerated minimum K_{sat} of 56.8 mm 15 min^{-1}. Week 13 in FP1 demonstrates the aerated minimum K_{sat} (44.5 mm 15 min^{-1}) to exceed the MORI, yet 234 intervals (0.349 % of rainfall periods) exceed the unaerated minimum K_{sat} (2.5 mm 15 min^{-1}), six (0.00896 % of rainfall periods) exceed the 10th percentile (9.5 mm 15 min^{-1}), and three (0.00448 % of rainfall periods) exceed the lower quartile (13.5 mm 15 min^{-1}). This highlights aeration's potential to substantially reduce IOF likelihood 13 weeks post-treatment. During Week 21, in FP1, rainfall intensities exceed some K_{sat} threshold in both treatments, with aeration causing substantial reductions in IOF likelihood at the minimum (7546 fewer intervals, 11.3 % of rainfall periods), 10th percentile (726 fewer intervals, 1.08 % of rainfall periods), lower quartile (107 fewer intervals, 0.160 % of rainfall periods), and median (40 fewer intervals, 0.0597 % of rainfall periods).

Without direct overland flow measurements or resulting streamflow, it is not possible to state if IOF likelihood changes can make a noticeable difference at whole-field or stream micro-catchment scales. This is because IOF generated on micropatches of topsoil may infiltrate as it traverses adjacent micropatches of more permeable soil, so called "runoff–runon phenomena" (Bonell and Williams, 1986). Assuming Explanation 1 to be true, Weeks 13 and 21 at FP1 imply that aeration may reduce flood-risk (at least on a sub-field scale), as ≥25 and ≥50 %, respectively, of the FP1N plot area could be generating IOF for a considerable number of events (for 107 and 40 15-min intervals, respectively). In contrast, only three 15-min intervals had the potential to generate IOF on ≥10 % of the FP1A plot area, and no recorded rainfall intensity had the potential to generate IOF on ≥25 % of FP1A. The very minor likelihood of increase in IOF due to aeration in FP2 is unlikely to cause a noticeable difference in flood-risk, as it affected ≥25 % of FP2A for only a single 15-min period, and ≥10 % of FP2A for only six 15-min periods. This deduction implies that blade aeration may have the potential to reduce flood-risk in regions where topsoil permeability conditions mean that IOF is a frequent flood-generating mechanism. Pasture FP1 findings support Van Vliet et al. (2006) and de Koff et al. (2011), who found aeration to reduce overland flow. Pasture FP2 results support the negligible changes observed in Shah et al. (2004), Franklin et al. (2006), and Butler et al. (2008).

5.7 Summary and conclusions

Overland flow potentially amplifies flood-risk across various regions of the world, yet previous research investigating if aeration reduces overland flow has revealed mixed findings. To improve process interpretation, two highly similar UK Stagnosol improved-pastures (FP1 and FP2) underwent blade aeration and subsequent topsoil penetration resistance and permeability measurements over 21 weeks. Permeability and precipitation information gathered from each replicate was used to generate IOF likelihood, to assess if blade aeration could reduce this fast drainage pathway.

Blade aeration significantly reduced PR for at least 21 weeks post-aeration in FP1, although FP2 showed no significant changes to penetration resistance, with a significant increase at 10 cm. The permeability results highlight that aeration significantly improved topsoil permeability for at least 21 weeks post-treatment in FP1, although no permeability improvement was observed within FP2. Proposed reasons for increases in permeability and decreases in PR are blade-induced soil loosening, preferential infiltration, root mat disruption, and/or increased soil bioactivity. The FP1 peaks-over-thresholds analysis highlights that aeration can substantially reduce IOF likelihood from up to 11.4 % of rainfall periods pre-aeration to 0.0926 % of rainfall periods post-aeration, although aeration within FP2 caused no change in IOF likelihood. Results highlight that aeration has the potential to reduce flood-risk in areas with elevated IOF likelihood, although improvements can be site specific.

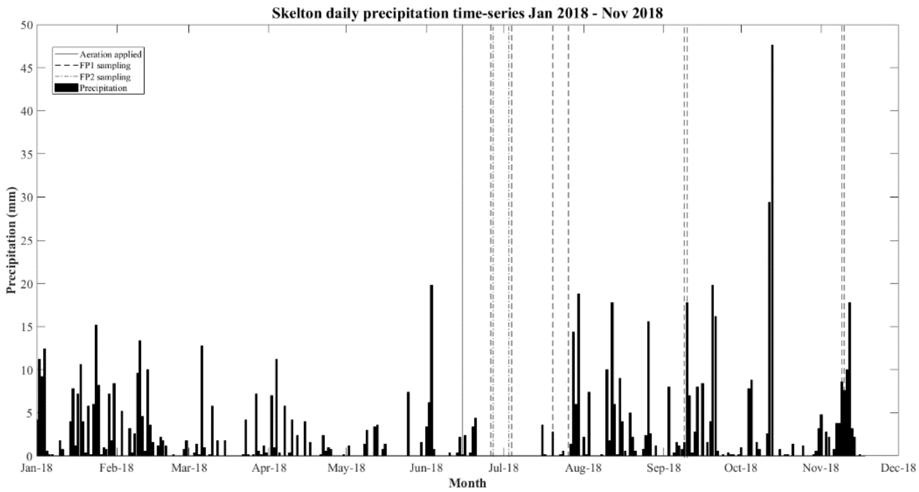
Future aeration researchers are advised to include root density and earthworm diversity for improved system interpretation. Researchers should additionally include soil moisture measurements and blade penetration depth during aeration to validate aerator effectiveness. X-ray tomography after aeration would also determine the effects of aeration on soil macroporosity. Future studies are advised that if blade insertion points and paths cannot be determined (either visually or via georeferencing), soil penetrometers may enter slits and bias measurements of penetration resistance. Similarly, studies are invited to question the suitability of *ex-situ* permeametry for their study sites. Finally, future research should consider combining the paired-plot and BACI approaches, to rule out natural soil variation causing observed differences.

5.8 Supplemental materials

The supplemental material consists of the treatment and sampling timetable for both improved-pastures (Supplemental Table 5.S1). A daily precipitation time-series taken from Skelton is provided to infer site conditions prior to treatment, as well as during and between sampling dates (Supplemental Figure 5.S1). The statistical peaks-over-thresholds model is shown for Field Pasture 1 (Supplemental Table 5.S2) and Field Pasture 2 (Supplemental Table 5.S3). The peaks-over-thresholds statistical model highlights the number of precipitation events that surpass each K_{sat} summary statistic (minimum, 10th percentile, lower quartile, median, and upper quartile), and therefore the implied reduction in IOF events.

Supplemental Table 5.S1: Treatment application and sampling timetable of Field Pasture 1 (FP1) and Field Pasture 2 (FP2).

Date	**Action**
11/06/18	Aeration undertaken in plots FP1A and FP2A
Week 2 (26/06/18-04/07/18)	FP1 and FP2 sampling
Week 6 (09/08/18-10/08/18)	FP1 sampling
Week 13 (20/09/18-26/09/18)	FP1 sampling
Week 21 (08/11/18-09/11/18)	FP1 sampling; FP1 and FP2 soil sampling



Supplemental Figure 5.S1: The Skelton daily precipitation time-series from January 2018-November 2018, with the treatment (blade aeration) application date, and subsequent permeability sampling dates in Field Pasture 1 (FP1) and Field Pasture 2 (FP2).

Supplemental Table 5.S2: The peaks-over-thresholds analysis comparing the saturated hydraulic conductivity (K_{sat}) with local rainfall intensities, highlighting the difference in infiltration-excess overland flow exceedance between aerated (A) and unaerated (N) regions of Field Pasture 1 for each sampling period.

Field Pasture 1										
Sampling Period	**K_{sat} Min Threshold (mm 15 mins^{-1})**	**Min Exceedance (15 mins)**	**K_{sat} Q10 Threshold (mm 15 mins^{-1})**	**Q10 Exceedance (15 mins)**	**K_{sat} Q25 Threshold (mm 15 mins^{-1})**	**Q25 Exceedance (15 mins)**	**K_{sat} Median Threshold (mm 15 mins^{-1})**	**Median Exceedance (15 mins)**	**K_{sat} Q75 Threshold (mm 15 mins^{-1})**	**Q75 Exceedance (15 mins)**
Week 2 A	76	0	118.3	0	239	0	468.3	0	686.5	0
Week 2 N	35.8	0	38.8	0	53.8	0	115	0	188.5	0
Overland Flow Reduction (15mins)		*0*		*0*		*0*		*0*		*0*
Week 6 A	333	0	393	0	435.5	0	698.5	0	1075	0
Week 6 N	56.8	0	66.3	0	82	0	125.3	0	419.5	0
Overland Flow Reduction (15mins)		*0*		*0*		*0*		*0*		*0*
Week 13 A	44.5	0	60.5	0	114.8	0	164.8	0	297	0
Week 13 N	2.5	234	9.5	6	13.5	3	42.8	0	82.8	0
Overland Flow Reduction (15mins)		*-234*		*-6*		*-3*		*0*		*0*

Supplemental Table 5.S2 (continued):

Field Pasture 1										
Sampling Period	**K_{sat} Min Threshold (mm 15 mins^{-1})**	**Min Exceedance (15 mins)**	**K_{sat} Q10 Threshold (mm 15 mins^{-1})**	**Q10 Exceedance (15 mins)**	**K_{sat} Q25 Threshold (mm 15 mins^{-1})**	**Q25 Exceedance (15 mins)**	**K_{sat} Median Threshold (mm 15 mins^{-1})**	**Median Exceedance (15 mins)**	**K_{sat} Q75 Threshold (mm 15 mins^{-1})**	**Q75 Exceedance (15 mins)**
Week 21 A	4.3	62	15.3	3	51.8	0	79	0	194.8	0
Week 21 N	0.75	7608	1.75	729	3.3	107	5.3	40	27	0
Overland Flow Reduction (15mins)		*-7546*		*-726*		*-107*		*-40*		*0*

Supplemental Table 5.S3: The peaks-over-thresholds analysis comparing the saturated hydraulic conductivity (K_{sat}) with local rainfall intensities, highlighting the difference in infiltration-excess overland flow exceedance between aerated (A) and unaerated (N) regions of Field Pasture 2 for each sampling period.

Field Pasture 2										
Sampling Period	**K_{sat} Min Threshold (mm 15 mins^{-1})**	**Min Exceedance (15 mins)**	**K_{sat} Q10 Threshold (mm 15 mins^{-1})**	**Q10 Exceedance (15 mins)**	**K_{sat} Q25 Threshold (mm 15 mins^{-1})**	**Q25 Exceedance (15 mins)**	**K_{sat} Median Threshold (mm 15 mins^{-1})**	**Median Exceedance (15 mins)**	**K_{sat} Q75 Threshold (mm 15 mins^{-1})**	**Q75 Exceedance (15 mins)**
Week 2 A	10	6	19	1	33	0	76.8	0	147.3	0
Week 2 N	33	0	34.8	0	43.3	0	80.5	0	130.3	0
Overland Flow Reduction (15mins)		*+6*		*+1*		*0*		*0*		*0*

5.9 Conflict of interest

The authors declare no conflict of interest.

5.10 Acknowledgements

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5.11 Data availability

The data that supports the findings of this study are available from the corresponding author upon reasonable request.

6 HEDGEROW WILD-MARGINS

Wallace, E.E., McShane, G., Tych, W., Kretzschmar, A., McCann, T. and Chappell, N.A. (2021). The effect of hedgerow wild-margins on topsoil hydraulic properties, and overland-flow incidence, magnitude and water-quality. *Hydrological Processes*, 35(3). Doi: 10.1002/hyp.14098.

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6.1 Brief introduction to paper

Improved-pastures throughout the globe, including within the Eden catchment, are often separated by hedgerows; principally in more low-lying and sheltered regions where milder conditions and more fertile soil prevail to encourage the growth of woody vegetation. Hedgerows in Cumbria (and the United Kingdom as a whole) are typically dominated by hawthorn (Crataegus monogyna), and often contain a considerable percentage of blackthorn (Prunus spinosa). Local agricultural hedgerows in the Eden catchment are usually supplemented by additional species to satisfy agri-environmental schemes, the most common of which being dogs rose (*Rosa canina*), field-maple (*Acer campestre*), guelder rose (*Viburnum opulus*), and hazel (*Corylus avallana*).

It is well documented that hedgerows offer a range of agricultural and ecosystem services, some of these key services being:

- The designation of property boundaries.
- The restriction of livestock movement.

- The provision of materials (timber, firewood, fruits, sloes) and livestock forage.
- The sheltering of livestock from excessive wind, rain, snow and sunlight.
- An increase in local biodiversity, including the hosting of pollinator and pest-controlling insect species.
- Aesthetic properties and historical/cultural heritage.
- The uptake of excess agrochemicals and slurry.
- The retention of soil.
- Carbon capture and storage, and climate regulation.

As a result of these ecosystem services, hedgerows are frequently included within environmental stewardship schemes. Hedgerows may also offer regulatory ecosystem services in relation to hydrology (flood-risk, water-quality and drought-resilience). Installing and maintaining hedgerows (and their features) could therefore be considered an improved-pasture intervention to reduce flood-risk and protect water-quality, and possibly improve drought-resilience.

Objective 3) Quantify the effect of hedgerows/hedge-margins within agriculturally-intensive improved-pastures in controlling changes in the amount of overland flow and rapid shallow-subsurface flow during floods and extreme precipitation events via direct measurement. This will be supported by causal factors of saturated hydraulic conductivity (K_{sat}) and soil volumetric wetness (θ_v). Overland flow will be analysed for targeted co-benefits to water-quality (total sediment (TS), nitrate (NO_3^-), nitrate-nitrite ($NO_3^-NO_2^-$), soluble reactive phosphorus (SRP), particulate total phosphorus (PTP), dissolved total phosphorus (DTP), and dissolved organic carbon (DOC)).

6.2 The effect of hedgerow wild-margins on topsoil hydraulic properties, and overland flow incidence, magnitude and water-quality

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6.3 Abstract

Overland flow and rapid shallow-subsurface flow from agricultural catchments are believed to contribute towards flood-risk and water-quality degradation across the globe. Hedgerows are commonplace agricultural features that may disrupt these rapid hydrological pathways. Research into the hydrological functioning of hedgerows is very limited however, with no field-based quantitative comparison of overland flows within hedgerows versus other land-uses. This research is the first globally to observe changes in overland flow incidence, volume and water-quality, alongside topsoil hydraulic and physico-chemical properties, induced by a hedgerow and adjoining wild-margin within a grassland landscape. Observations were conducted within two replicated paired-plots between a hedgerow wild-margin and a bordering improved-pasture, within Cumbria, UK.

Compared to adjacent improved-pasture, hedge-margins significantly reduced topsoil bulk-density and increased porosity, and significantly increased the topsoil median permeability by a factor of 22-27. Overland flow models, based on direct observations, highlight that hedge-margins are slower to produce overland flows than improved-pastures, requiring an equal or greater amount of saturation before the onset of overland flow generation. Hedge-margins resultantly produced less overland flow

volume, likely due to increased infiltration, percolation and/or evapotranspiration. Soil saturation models, also based on direct observations, confirm improved-pastures saturate faster than hedge-margins, with hedge-margins having extremely variable dynamics in relation to precipitation, whereas improved-pastures have more moderate and consistent dynamics. Overland flow water-quality from ‘wash-off’ experiments highlight that hedge-margins may store substantially more nitrate (70 % – 260 %), nitrate-nitrite (640 % – 650 %), and loose sediment (540 % – 3970 %) on the ground surface compared to improved-pastures; although further experimentation is needed to determine contaminant mobilisation potential.

6.4 Introduction

Hedgerows are commonplace landscape features that provide a wide range of ecosystem and agricultural services (Baudry et al., 2000; Wolton et al., 2014; Blanuša et al., 2019). Post Second World War agricultural mechanisation, alongside land consolidation and occasional land abandonment, has lowered the prevalence of managed hedgerows throughout Western Europe (Burel and Baudry, 1990; Petit et al., 2003; Deckers et al., 2005; Sánchez et al., 2009; van Apeldoorn et al., 2013; Arnaiz-Schmitz et al., 2018), and indeed further afield (Schmucki et al., 2002; Sklenicka et al., 2009). In 1945, England and Wales contained approximately 1.4 million km of hedgerows, with the latest 2007 estimate at 456,000 km (O’Connell et al., 2004; Carey et al., 2008).

Hedgerow removal within temperate climates of Europe has been associated perceptually with increasing flood frequency and magnitude, but very limited experimental evidence shows the moderating effect of hedgerows on flood-generation processes or water-quality (Wolton et al., 2014). Consequentially, the understanding of the hydrological functioning of hedgerows remains incomplete and requires greater attention (Supplemental Table 6.S1: Carluer and De Marsily, 2004), with the current state of knowledge summarised below:

6.4.1 Wet-canopy evaporation, interception and transpiration of hedgerows

Hedgerows support enhanced evapotranspiration due to extensive root networks which enables considerable water-uptake, low stomatal resistance to facilitate transpiration, naturally low albedo causing high net-radiation, intensive air turbulence that supports air exchanges, and dense foliage to encourage wet-canopy evaporation (Ryszkowski and Kędziora, 1987; 1993; Herbst et al., 2007; Ghazavi et al., 2008; Grimaldi et al., 2012). Within Western France, Ghazavi et al. (2008) quantified hedgerow trees to intercept 2.4 % of rainfall without foliage, and 5.6 % when in leaf. Within Southern England, Herbst et al. (2006) quantified hedgerow interception storage as 2.6 mm when leaved, and 1.2 mm when leafless, an amount comparable to many forest types. Herbst et al. (2006) additionally quantified hedgerow wet-canopy evaporation at 24 % with foliage, and 19 % when leafless. Further, Herbst et al. (2007) showed that unit-area hedgerow transpiration rates in Southern England are higher than many temperate woodlands. In a modelling study within Western France, Benhamou et al. (2013) simulated hedgerow trees to increase grid-scale (20 m^2) evapotranspiration by 20 %, with a lightly-hedged scenario increasing catchment-scale (5 km^2) evapotranspiration by 3.3 %. In Northern England, Coates (2019) found a 7 % – 56 % reduction in precipitation on the leeward side of a hedgerow, and a 5 % – 29 % increase in leaf wetness on the windward side. Coates (2019) additionally found the hedgerow to intercept 67 % – 79 % of gross precipitation, with a lower interception percentage during winter.

6.4.2 The effect of hedgerows on streamflow

Prior studies (all in Western France) suggest that hedgerows/hedgerow trees may affect flood-risk and discharge at the micro/small-catchment scale. In a paired-catchment study (each 0.32 km^2), Mérot (1978), cited in Mérot (1999), concluded that a hedged catchment could reduce peak flows by 33 % – 50 %, and runoff coefficients from 15 % to 5 %, compared to an unhedged adjacent catchment. Viaud et al. (2005) employed a water-balance modelling approach for a 5 km^2 catchment, outlining that a heavily-hedged catchment (200 m ha^{-1}) could halve discharge compared to a hedgeless catchment in drier years. Benhamou et al. (2013) modelled a 5 km^2 catchment with and without 1.5 % hedgerow cover, highlighting that hedgerows could lower channel discharges by 4.5 %. Within three 15 km^2 catchments, Viel et al. (2014) using a 'hydrological connectivity model', suggested that despite high hedgerow density, slopes may not be fully-partitioned and may still be hydrologically connected, although hedgerows could potentially store or direct overland flow into the soil.

6.4.3 The effect of hedgerows on soil drying

Hedgerow roots can range 10 m beyond their peripheries and to considerable depths (Caubel-Forget et al., 2001; Caubel et al., 2003; Ghazavi et al., 2008). This expansive root network enables substantial water-uptake to support transpiration, and can cause localised soil drying (Thomas et al., 2008; Albéric et al., 2009; Thomas et al., 2012; Hao et al., 2014). If situated close to the water-table, roots can constantly access water, facilitating enhanced transpiration (Thomas et al., 2008; 2012).

Root water-uptake by the hedgerow dries the soil in spring and summer, possibly delaying autumnal rewetting (Caubel et al., 2003; Ghazavi et al., 2008; 2011). Hedgerow-induced soil dryness may impede shallow-subsurface flow, reducing slope connectivity and increasing pollutant residence-times (Ghazavi et al., 2008; 2011). Utilising the Kirkby Topographic Index (Beven and Kirkby, 1979), Mérot and Bruneau (1993) showed that hedgerow banks may influence the distribution of saturated areas.

6.4.4 The effect of hedgerows on water-quality

Several studies throughout France have suggested hedgerows may influence local soil hydrochemistry. Comparing hedgerows to improved-pastures, Albéric et al. (2009) highlights soil-water beneath hedgerows to contain higher concentrations of major ions; likely caused by drying cycles and a lack of dilution from precipitation. Grimaldi et al. (2009) similarly highlight soil-water beneath hedgerows contains higher chloride concentrations due to soil drying.

Hedgerows can strongly influence the distribution of nitrates within groundwaters (Caubel-Forget et al., 2001; Thomas and Abbott, 2018). Hedgerows are highly effective at nitrate removal from shallow groundwater when compared with improved-pasture vegetation or arable crops, and could potentially ameliorate groundwater contamination and transport to streams (Caubel-Forget et al., 2001; Grimaldi et al., 2012; Thomas and Abbott, 2018). Effective nitrate removal is augmented by a combination of plant water-uptake, and denitrification due to increased organic carbon, heterogeneous redox conditions, and heightened microbial processes (Caubel-Forget et al., 2001; Grimaldi et al., 2012; Thomas and Abbott, 2018). Furthermore, Thomas et al. (2016) demonstrate how hedgerow density influences stream hydrochemistry and nitrate fluxes within three headwater catchments in Western France (2.3 km^2 – 10.8 km^2). Benhamou et al. (2013) modelled a potential 3.3 % drop in streamflow NO_3-N due to the hedgerow presence.

6.4.5 The effect of hedgerows on surface hydrology

Despite these studies, substantial knowledge gaps remain around the hydrological functioning of hedgerows within temperate regions, with quantitative studies particularly lacking (Hao et al., 2014; Wolton et al., 2014; Blanuša and Hadley, 2019; Holden et al., 2019: Supplemental Table 6.S1). A significant component of this is how hedgerows influence surface and near-surface hydrology, including infiltration-capacity, overland flow and water-quality. Carluer and De Marsily (2004) hypothesise that hedgerows could significantly increase infiltration, although highlight a lack of observations to support this assumption and subsequent modelling. Thomas et al. (2008) similarly acknowledge the lack of data regarding how hedgerows alter hydraulic conductivities.

Holden et al. (2019) is the only published study to directly measure hedgerow topsoil permeability, noting a hedgerow in North Yorkshire, UK, was significantly more permeable than nearby improved-pasture or arable fields (Supplemental Table 6.S1). Coates (2019) in a PhD thesis also observed higher permeability values in close proximity to a hedgerow in North Yorkshire, UK, with lower rates of permeability with increasing distance from the hedgerow. Blanuša and Hadley (2019) are the first to demonstrate that common temperate hedgerow species such as Hawthorn (*Crataegus monogyna*) significantly delay and reduce runoff in plant-pot and model trench experiments (Supplemental Table 6.S1). These studies underline that no field-based observations exist regarding how hedgerows alter overland flow, with this information dearth substantially hindering modelling studies, and consequentially limiting the hydrological understanding of hedgerows.

The aim of this paper is to contrast topsoil hydrological properties and overland flow (contaminant concentrations and flow) between a hedgerow/hedge-margin and an adjoining agriculturally-improved permanent pasture in an upland UK setting. Thus, the objectives of this study are:

I) To compare soil and topographic properties that may influence the topsoil saturated hydraulic conductivity, moisture status, and overland flow occurrence between permanent pasture plots and plots in an adjacent hedgerow wild-margin.

II) To contrast the measured topsoil saturated hydraulic conductivities between the two aforementioned land-uses.

III) To compare the incidence and magnitude of overland flow produced from natural precipitation events between the two aforementioned land-uses.

IV) To compare the hydrological response of the two aforementioned land-uses to an artificial-rainfall experiment in relation to soil moisture, surface hydrological pathways and overland flow incidence and magnitude.

V) To compare the water-quality of overland flow produced from the two aforementioned land-uses following a 'wash-off' experiment.

6.5 Materials and methodology

6.5.1 Study site

A study site (centre 54°35'16.38"N, 2°42'53.43"W) was established in the River Leith catchment (Cumbria, UK), 8 km SSE of Penrith and 7 km NNW of Shap (Figure

6.1). The local climate from Shap weather station (54°30'49"N, 2°40'40"W: 301 masl) is wet temperate, with annual average maximum and minimum temperatures of 11. 5 °C and 4.1 °C, respectfully, and a long-term (1981-2010) annual rainfall average of 1,779 mm (Met Office, 2020). Average potential evapotranspiration for the experimental site (1981-2010) is 1.29 mm d^{-1}, with a summer average of 2.43 mm d^{-1} and a winter average of 0.33 mm d^{-1} (Robinson et al., 2016).

The study site belongs to the 713g Brickfield soil-series, equivalent to an FAO Eutric Stagnosol, or an Aquic soil within USDA soil taxonomy (Jarvis et al., 1984; USDA, 1999; WRB, 2015). Brickfield soils are slowly permeable with clay subsurface horizons impeding drainage (wetness class IV/V), increasing overland and shallow-surface flow susceptibility (Jarvis et al., 1984). The soil profile is developed from slowly permeable glacial drift (Hankin et al., 2018). The solid geology beneath the site is the Yoredale Group (Alston Formation), consisting of bioclastic limestone, mudstone, siltstone and sandstone (Arthurton and Wadge, 1981).

6.5.2 Plot-pairing

The study site comprises two replicated paired-plots spanning an agriculturally-improved permanent pasture and an adjacent uncultivated hedgerow wild-margin, running perpendicular to the hillslope contours (Figure 6.2; see later Figure 6.3). The two agriculturally-improved pasture (AIP) replicates are termed pasture plot one (P1) and pasture plot two (P2). Plots P1 and P2 are perennial ryegrass (*Lolium perenne*) monocultures, with a root mat extending 3 cm – 6 cm below the surface. Some stinging nettle (*Urtica dioica*) is well-established within P1. During the study the improved-pasture was stocked with ewes (~6 ha^{-1}), lambs (~11 ha^{-1}), and occasionally heifer beef cattle (~1 ha^{-1}), averaging ~1.8 livestock units ha^{-1}. Intensive agricultural

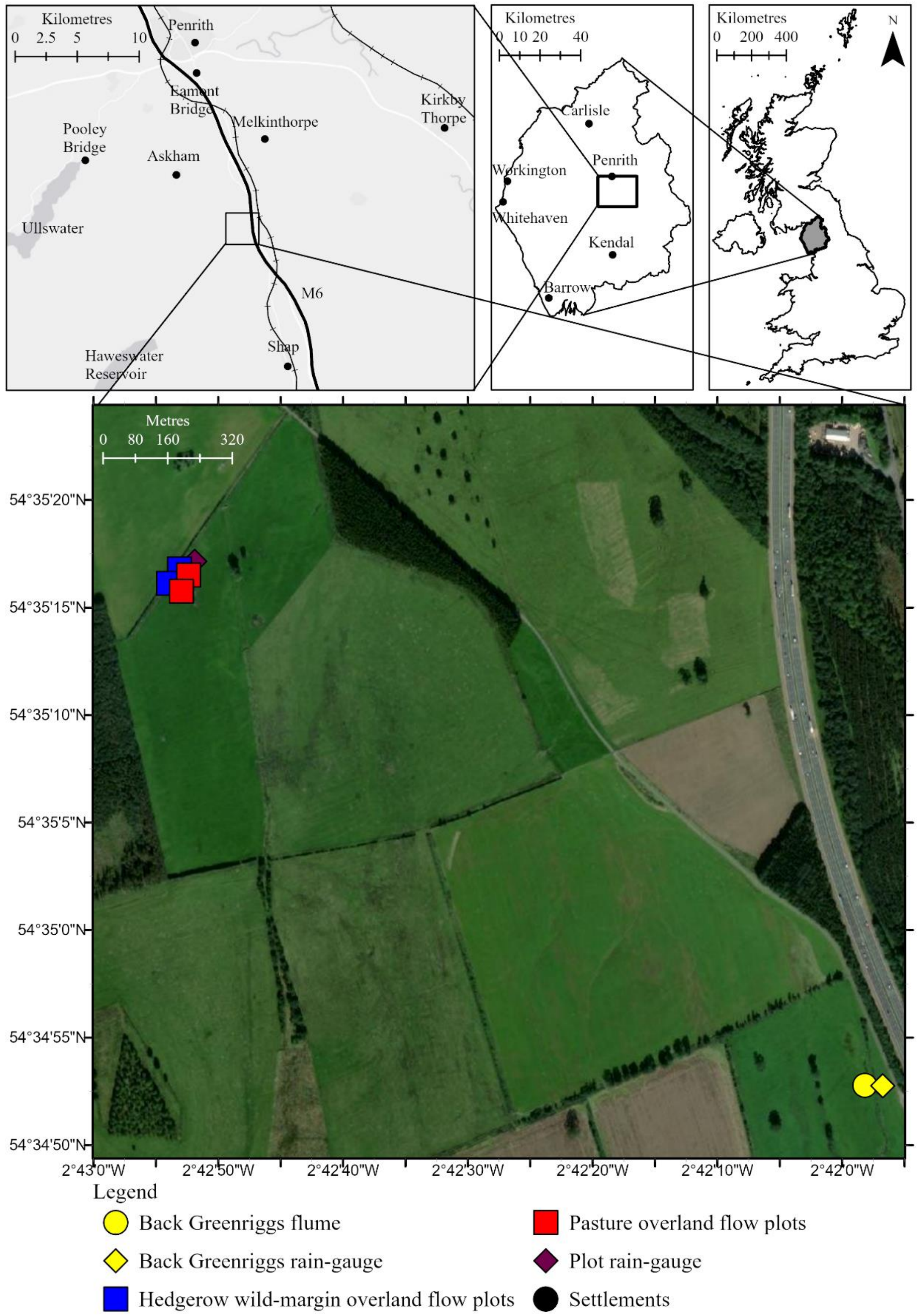


Figure 6.1: The experimental site in a UK, Cumbria, and local area context. The experimental site is entirely within the Leith sub-catchment. The location of the pasture and hedge wild-margin overland flow plots are highlighted, alongside the supporting Back Greenriggs flume and Back Greenriggs rain-gauge, as well as the overland flow plot rain-gauge. Contains Ordnance Survey (OS), ESRI, HERE,

Garmin, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, openstreet map contributor and the GIS User Community data.

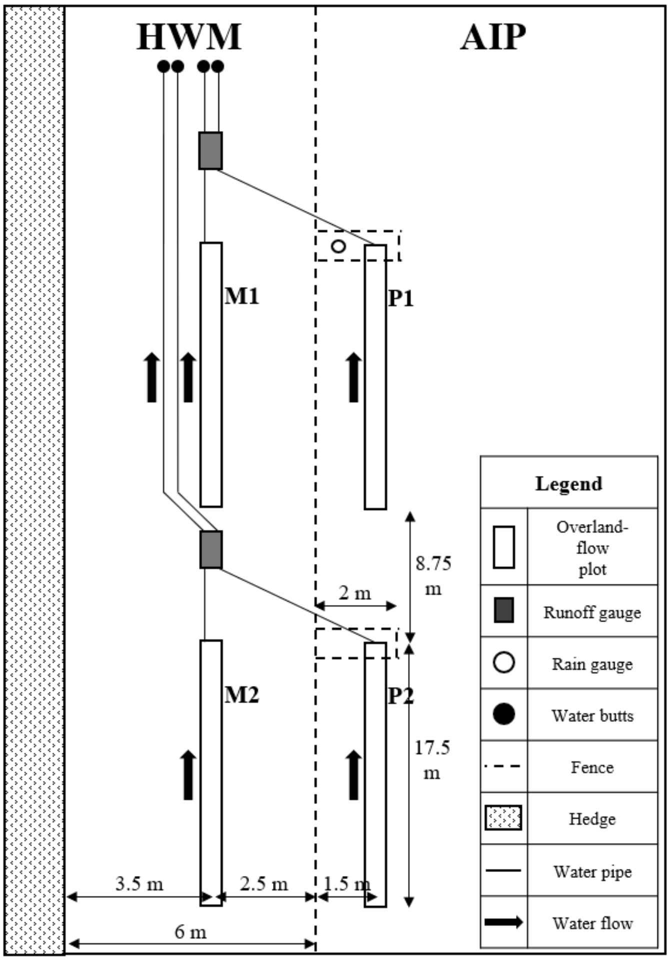


Figure 6.2: A schematic diagram of the experimental site, with the agriculturally improved-pasture (AIP) and hedgerow wild-margin (HWM) land-uses labelled. The four replicated plots: Pasture plot one (P1), pasture plot two (P2), hedgerow wild-margin plot one (M1), and hedgerow wild-margin plot two (M2), as well as Overland Flow Plots (OFPs), the plot rain-gauge, the water flow direction, overland flow gauges, water pipes, storage water butts, and fences, are all highlighted.

practices were previously applied to AIP (ploughing, slurrying, reseeding etc.), although none have occurred since August 2018.

The hedgerow wild-margin (HWM) comprises the alternate plots. HWM includes a hedgerow planted in 2003, consisting of 50 % hawthorn, 40 % blackthorn (*Prunus spinosa*), with the remainder mostly dog rose (*Rosa canina*) and hazel (*Corylus avellana*), as well as an adjacent 6-m wide wild-margin (grazing exclusion zone) which was fenced off from the improved-pasture. This wild-margin was sparsely planted with trees in 2013 as part of an environmental higher-level stewardship agreement.

The first HWM replicate is wild-margin plot one (M1) which has a 0.6 m – 0.9 m scrub layer of perennial ryegrass, stinging nettle, and various brambles (*Rubus spp.*). Two immature hazel trees are directly within M1, with an immature hazel and an immature holly (*Ilex aquifolium*) immediately adjacent. Wild-margin plot two (M2) has a very dense 1.2 m –1.5 m scrub layer of stinging nettle, brambles, creeping thistle (*Cirsium arvense*), and immature blackthorn. Directly within M2 are three mature blackthorn trees and one mature hawthorn tree. Immediately adjacent to M2 are sixteen mature blackthorn trees, one immature hazel and one immature oak (*Quercus robur*).

Paired-plots are separated by approximately 4 m across the slope, with AIP plots positioned 1.5 m into the improved-pasture and HWM plots positioned 2.5 m into the hedgerow wild-margin (Figure 6.2). The HWM plots are located approximately 3.5 m from the hedgerow. Upper and lower plots are separated by 8.75 m.

6.5.3 Overland flow plots

Four replicated overland flow plots (OFPs) were established within AIP and HWM (Figure 6.2). Each OFP involved embedding 30 cm of galvanised steel lawn edging in a 17.5 m by 0.5 m rectangle (8.75 m^2), with 5 cm remaining above the surface to retain overland flow. At the terminus of each OFP, a 0.5 m wide ‘Gerlach (collection)-trough’ was installed to collect overland flow (Gerlach, 1967). The trough lip was hammered into the topsoil to 8 cm depth (just below observable root mats) to ensure all overland flow was captured. Specifically, the OFPs capture both overland flow (i.e., water flowing across the soil surface and within the litter-layer), as well as near-surface lateral flows within the topsoil. Collection-troughs were sealed with Montmorillionite clay to prevent leakage and each AIP trough was fenced off for protection. Care was taken during installation to avoid plot disturbance.

The collection-troughs were connected to a gravity-fed underground pipe network (4 cm diameter) to funnel overland flow into enclosures buried within HWM. Each enclosure contained two KIPP100 tipping-bucket counters (METER group, USA), each having a 100 cm^3 tip volume. The tipping-bucket counters were connected to a H21 HOBO datalogger (Onset computer corporation, USA) to record the timing of each tip. Pipes exit the enclosures into four sealed water butts for subsequent water-quality analyses. To ensure no leakage (or addition) of water, known volumes of deionised water were passed through the system.

The overland flow gauges recorded the incidence and volume of overland flow between 10th April 2019 – 10th March 2020. A rain-gauge 2 km to the south-east (Back Greenriggs rain-gauge) provided precipitation data, with a plot rain-gauge providing supplementary precipitation data between 10th April 2019 – 7th February 2020 before equipment malfunction (Figure 6.1; Supplemental Figure 6.S1).

6.5.4 Artificial-rainfall experiments

Standardised, artificial-rainfall experiments were conducted on the 11th (P1 and M1) and the 15th April 2019 (P2 and M2). Prior to each experiment, the collection system was flushed with deionised water and electrical conductivity tests confirmed no contaminants remained in the collection network. Each experiment involved a laboratory-made rainfall generator that delivered a constant 22.5 mm hr^{-1} rainfall intensity. This rainfall intensity is marginally above the MORI from a local rainfall record (21.2 mm hr^{-1}: see Wallace and Chappell, 2019), and is therefore plausible for extreme conditions at this locality.

Immediately upslope of the collection-troughs, 1 m^2 of each OFP was enclosed with the rainfall-generator positioned centrally (Figure 6.S2). The artificial-rainfall experiment ran for two consecutive hours in an attempt to generate overland flow. Throughout the experiment, a simplified Time-Domain Reflectometer probe (Delta Ltd ML3 Theta-probe: Gaskin and Miller, 1996) was positioned centrally 10 cm upslope of the collection-trough, to quantify θ_V over the surface 0-6 cm of the litter layer and upper topsoil (and by combining with η the degree of saturation), and therefore the likelihood of SOF incidence. The moisture-probe was additionally used to assist in the identification of near-surface hydrological pathways, and therefore to improve the understanding of the saturation process(es) occurring between treatments prior to overland flow generation.

The goal of the artificial-rainfall experiment was: a) to determine if IOF could be generated from each OFP at plausible precipitation intensities, b) to quantify at what degree of saturation did overland flow occur from each OFP (possibly highlighting SOF when combined with η), and c) to determine the required elapsed time of extreme rainfall before overland flow occurred from each OFP given the naturally variable baseline conditions. The tipping-bucket counters would also record the volumes of overland flow.

6.5.5 Wash-off experiments

Following each artificial-rainfall experiment, a ‘wash-off’ experiment was undertaken by applying a 20-litre pulse of water in 30 seconds (2400 mm hr^{-1}) to generate overland flow on the surface of each plot to transport potential contaminants from the plot surface into the collection system. This precipitation intensity is well beyond observed rates. The aim of the experiment was not to reproduce natural 'wash-off' events, but to show what contaminants are present on the plot surfaces.

Water-samples taken during the experiment were taken from the overland flow pipe system and underwent laboratory analysis for the determination of physico-chemical properties. Samples were stored in a cool-room from approximately 90 minutes after each experiment, with all chemical analysis conducted within 24 hours. Total sediment concentrations were determined by evaporating 750 ml of sample at 105 °C and weighing the residue. Dissolved organic carbon (DOC) concentrations were estimated via ultraviolet-visible spectroscopy using a Jenway 7315 spectrophotometer, with absorbance at 254 nm and 400 nm (Tipping et al., 2009). Nitrate (NO_3^-), NO_3^- NO_2^-, SRP, dissolved total phosphorus (DTP) and particulate total phosphorus (PTP) were measured using a Seal AQ2 Discrete Analyzer. Electrical conductivity was measured on-site using a WTW 340i electrical conductivity meter.

6.5.6 Supporting hydrological, pedological and topographic measurements

Topsoil permeability measurements were taken surrounding the four OFPs on the 29th – 30th April 2019. Permeability was determined via a Talsma ring permeameter (Talsma, 1960; Chappell and Ternan, 1997). The ring permeameter is a constant-head device that provides measurements of the coefficient of permeability, also known as the saturated hydraulic conductivity. Permeability was determined via Darcy's law once equilibrium was attained (i.e., the core was fully saturated). The procedure detailed in Chappell and Ternan (1997) was followed exactly, except that measurements were conducted whilst cores remained in the ground to account for underlying soil properties (Sherlock et al., 2000; Wallace and Chappell, 2019).

To determine soil physico-chemical properties, eight topsoil samples surrounding each OFPs was extracted in 221 cm^3 bulk-density cores on the 25th November 2019. Samples were sieved to 2 mm and oven-dried at 105 °C for 24-h for ρ_b calculation. Soil organic matter content was determined from oven-dried soil via a 550 °C 6-h loss-on-ignition test. Particle-size analysis involved mixing furnace-dried soil with 1 % sodium polymetaphosphate for 48-h, followed by manual aggregate breaking. Samples then underwent high-power sonication for 5-min and laser diffraction (Beckman Coulter, LS-13-320). Porosity determination involved gradually submerging each soil core with deionised and de-aired water over 48 hours, before measuring saturation with the moisture-probe. The topography of each OFP was measured using a total station (Trimble, Robotic-S6) to derive slope angles.

6.5.7 Statistical analysis and Data-Based Mechanistic analysis

Soil properties (excluding texture) were contrasted via the MWW test. The MWW is a robust, non-parametric statistical test suited to small sample sizes (eight per plot), and was therefore deemed appropriate. To contrast K_{sat} values, permeability measurements first underwent AD and Shapiro-Wilk normality tests. Permeability distributions were then contrasted via the two-sample t-test due to satisfying normality assumptions, as well as the MWW test due to limited and unequal sample sizes (minimum of seven per plot). All analysis was conducted within Matlab (The Mathsworks, Inc) using the *ranksum*, *adtest*, *swtest* (Bensaïda, 2019), and *ttest* functions, with significance levels of $p \leq 0.05$, $p \leq 0.01$, and $p \leq 0.001$.

A TF modelling approach within the DBM modelling framework was applied to both the artificial-rainfall experiment and the overland flow response to natural-rainfall events (see Young, 2011). Transfer functions – expressed as ratios of polynomials in complex variables – are a useful alternative form of differential or difference equation with linear dynamics. Their convenience comes from the ease of interpretation of the complex variables as either a Laplace operator (derivative in time-domain) or a backward shift operator (easily expressing finite differences). Transfer functions may be manipulated algebraically with several well-established methods of identifying their orders and estimating their parameters. As functions in the time-domain, they map an input data-series into the modelled response – the output data-series.

During the artificial-rainfall experiments, rainfall (input) was used to predict the observed θ_v response (output). In these experiments, the artificial-rainfall failed to generate overland flow, and thus, overland flow could not act as system output. The TF for each OFP during each experiment could be separated into different response components, which may represent different hydrological components (Ockenden and Chappell, 2011; Jones et al., 2014; Chappell et al., 2017).

A separate TF model containing a non-linearity was used to simulate the observed overland flow (output) from observed natural rain-events (input) for each OFP (Supplemental Figure 6.S3). Streamflow from the Back Greenriggs flume (Figure 6.1) was used as a surrogate measure of latent catchment saturation that controls the non-linearity of the overland flow response (this performed better than antecedent precipitation indices – see later Figure 6.4). The non-linearity in the generation of overland flow, due to the need for topsoil saturation to develop, was represented by Equation 6.1:

$$P_e(k) = P_o(k)\ (ASI(k-l)^{\alpha}) \tag{6.1}$$

Where P_e is the effective rainfall, P_o is the observed rainfall, α is the non-linearity, k is the sample, l is the sampling-lag, and ASI is the Antecedent Saturation Index, defined by Equation 6.2:

$$ASI = \max(0, Q_o(k) - Q_c(k)) \tag{6.2}$$

Where Q_o is the observed streamflow and Q_c is a streamflow-threshold.

Different streamflow-thresholds were required for each OFP to produce the best fitting models, highlighting that different levels of catchment-integrated saturation were necessary for overland flow generation for each OFP. Stronger non-linearity suggests increased influence of streamflow on overland flow production (Young and Beven, 1994). The ASI and non-linearity were then combined with the Back Greenriggs precipitation time-series to produce effective rainfall (i.e., rainfall that produced plot-scale overland flow: see e.g., Kretzschmar et al., 2016). Sampling-lags (up to an hour) were included to incorporate the potential timing differences between OFP-response and the streamflow-response.

The Data-Based Mechanistic modelling philosophy makes no *a priori* assumptions about the system under analysis, and states that observations must dictate the identified structure of the modelled system processes (Young, 1999). The identified structures must however, have real-world interpretations and be feasible at the studied locality. Transfer function models of the system's dynamics were identified and estimated using the RIV functions within CAPTAIN (Computer Aided Program for Time-Series Analysis and Identification of Noisy systems) toolbox (Taylor et al., 2007), in either continuous or discrete-time, as appropriate (Young and Jakeman, 1979; 1980; Young and Garnier, 2006). At the model identification stage a range of different candidate model structures are evaluated, and the statistically optimal model selected according to a combination of model fit measured using R_t^2 (a simplified Nash-Sutcliffe criterion), and YIC, an information criterion taking into account the residuals' form and a measure of variation coefficients of the estimated parameters (Young and Jakeman, 1979; Young, 2011). The main dynamic characteristics: TCs, δs, and SSGs, of each identified component were extracted from the selected models (Young, 2011; Jones et al., 2014; Chappell et al., 2017).

6.6 Results

6.6.1 Soil and topographic properties that may influence hydraulic properties

The topography for each OFP is given in Figure 6.3. Slopes for P1, M1, P2 and M2 are 8.9 %, 9.4 %, 13.1 %, and 12.6 %, respectively. Slopes remain fairly-uniform throughout each OFP; although the upper plot-pairing is notably steeper than the lower plot-pairing.

All four soil profiles below OFPs lacked defined soil horizons, with topsoil properties given in Table 6.1. Plot P1 was 100 % silt loam, whilst P2 was silty clay loam (62.5 %) and silt loam (37.5 %). Plot M1 was mostly silt loam (75 %), with some loam (25 %); whilst M2 was 100 % silt loam. All OFPs have similar soil textures, although AIP contains slightly more clay and slightly less sand than HWM.

Statistical tests of topsoil properties are given in Table 6.2. Soil ρ_b was significantly higher in AIP than HWM, overall and in both replicates ($p \leq 0.001$). Pasture plots had statistically similar ρ_b ($p \leq 0.130$), although wild-margin plots were statistically dissimilar ($p \leq 0.001$). Soil η was significantly higher in HWM than AIP within both

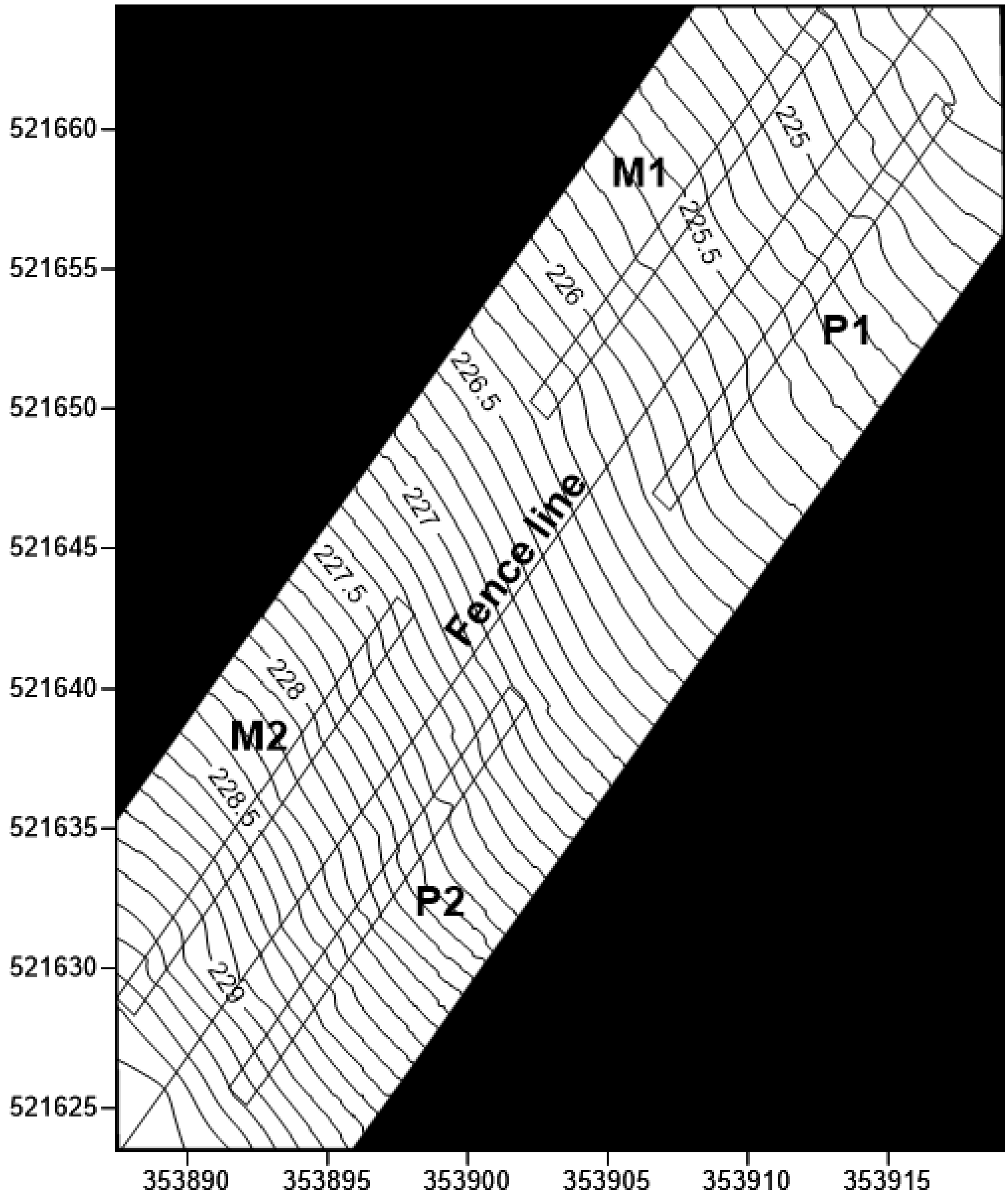


Figure 6.3: A detailed topographic map of the experimental site, outlining gradient contours (in metres) within and surrounding the overland flow plots (OFPs) for pasture plot one (P1), pasture plot two (P2), hedgerow wild-margin plot one (M1), and hedgerow wild margin plot two (M2), as well as the separation fence. Slopes of similar inclinations are extremely common in the region and across much of Great Britain and Ireland.

Table 6.1: Arithmetic mean values and standard deviation (in brackets) of soil dry bulk-density (ρ_b), porosity (η), soil organic matter (SOM), pH, and particle-size distribution for pasture plot one (P1), pasture plot two (P2), hedgerow wild-margin plot one (M1), and hedgerow wild-margin plot two (M2), alongside combined Agriculturally-Improved Pasture (AIP) and Hedgerow Wild-Margin (HWM) values, and the number of samples (N).

Site	**ρ_b (g cm^{-3})**	**η (%)**	**SOM (%)**	**pH**	**Particle Size Distribution (%)**					**N**
					>2 µm	**2-20 µm**	**20-60 µm**	**60-200 µm**	**200-2000 µm**	
Pasture plot 1	9.3×10^{-1} (2.5×10^{-2})	58.0 (0.2)	10.4 (0.7)	5.8 (0.1)	16.7 (2.6)	35.1 (6.5)	27.6 (2.4)	20.3 (8.7)	0.3 (0.5)	8
Pasture plot 2	8.8×10^{-1} (5.8×10^{-2})	59.1 (0.2)	11.5 (1.6)	5.8 (0.3)	27.8 (7.7)	49.8 (7.3)	17.0 (7.3)	5.4 (7.3)	0.0 (0.0)	8
Agriculturally Improved-Pasture	9.0×10^{-1} (5.1×10^{-2})	58.5 (0.6)	10.9 (1.3)	5.8 (0.2)	22.2 (8.0)	42.4 (10.1)	22.3 (7.6)	12.8 (11.0)	0.2 (0.4)	16
Wild-margin plot 1	5.6×10^{-1} (1.1×10^{-1})	59.2 (0.2)	16.5 (5.6)	5.9 (0.3)	13.8 (3.4)	33.2 (9.5)	26.0 (3.5)	23.9 (10.2)	3.1 (3.3)	8
Wild-margin plot 2	7.6×10^{-1} (3.8×10^{-2})	60.7 (0.4)	14.8 (7.0)	5.6 (0.2)	16.8 (4.5)	40.9 (13.0)	25.1 (6.2)	16.8 (12.3)	0.5 (1.3)	8
Hedgerow Wild-Margin	6.6×10^{-1} (1.3×10^{-1})	59.9 (0.8)	15.6 (6.2)	5.7 (0.3)	15.3 (4.1)	37.0 (11.7)	25.6 (4.9)	20.3 (11.5)	1.8 (2.8)	16

Table 6.2: Statistical comparisons of soil dry bulk-density (ρ_b), soil porosity (η), soil organic matter (SOM), soil pH, and saturated hydraulic conductivity (K_{sat}) for pasture plot one (P1), pasture plot two (P2), hedgerow wild- margin plot one (M1), and hedgerow wild-margin plot two (M2), alongside combined Agriculturally-Improved Pasture (AIP) and Hedgerow Wild-Margin (HWM) values.

Mann-Whitney-Wilcoxon Tests				
ρ_b				
Site	**Pasture plot 1**	**Wild-margin plot 1**	**Wild-margin plot 2**	**Hedgerow Wild-Margin**
Pasture plot 1	-	0.001***		
Pasture plot 2	0.130		0.001***	
Agriculturally-Improved Pasture				0.001***
Wild-margin plot 2		0.001***	-	
η				
Site	**Pasture plot 1**	**Wild-margin plot 1**	**Wild-margin plot 2**	**Hedgerow Wild-Margin**
Pasture plot 1	-	0.001***		
Pasture plot 2	0.001***		0.001***	
Agriculturally-Improved Pasture				0.001***
Wild-margin plot 2		0.001***	-	
SOM				
Site	**Pasture plot 1**	**Wild-margin plot 1**	**Wild-margin plot 2**	**Hedgerow Wild-Margin**
Pasture plot 1	-	0.001***		
Pasture plot 2	0.050*		0.328	
Agriculturally-Improved Pasture				0.001***
Wild-margin plot 2		0.328	-	

Table 6.2 (continued):

pH				
Site	**Pasture plot 1**	**Wild-margin plot 1**	**Wild-margin plot 2**	**Hedgerow Wild-Margin**
Pasture plot 1	-	0.590		
Pasture plot 2	0.898		0.205	
Agriculturally-Improved Pasture				0.157
Wild-margin plot 2		0.195	-	
K_{sat}				
	Pasture plot 1	**Wild-margin plot 1**	**Wild-margin plot 2**	**Hedgerow Wild-Margin**
Pasture plot 1	-	0.001***		
Pasture plot 2	0.613		0.001***	
Agriculturally-Improved Pasture				0.001***
Wild-margin plot 2		0.955	-	
Two sample t-test				
K_{sat}				
Site	**Pasture plot 1**	**Wild-margin plot 1**	**Wild-margin plot 2**	**Hedgerow Wild-Margin**
Pasture plot 1	-	0.001***		
Pasture plot 2			0.002**	
Agriculturally-Improved Pasture				0.001***

replicates and overall ($p \leq 0.001$), with η significantly different between all plots ($p \leq 0.001$). Soil organic matter content was significantly higher in HWM than AIP overall ($p \leq 0.001$), and significantly higher in M1 compared to P1 ($p \leq 0.001$), although statistically similar between P2 and M2 ($p \leq 0.328$). Soil organic matter content between improved-pasture plots was statistically different ($p \leq 0.050$), although wild-margin plots were statistically similar ($p \leq 0.328$). All plots had a statistically similar pH.

6.6.2 Topsoil permeability

Statistical normality tests highlight K_{sat} surrounding the OFPs satisfy normality (Table 6.3). Mann-Whitney-Wilcoxon and two-sample t-tests (Table 6.2) highlight the topsoil surrounding wild-margin OFPs is significantly more permeable than by the improved-pasture OFPs (least significant result $\rho \leq 0.002$). Median K_{sat} is 2,700 % larger in M1 than P1, and 2,200 % larger in M2 than P2 (Table 6.3). Improved-pasture plots have similar K_{sat} ($\rho \leq 0.613$), as do the wild-margins ($\rho \leq 0.995$).

6.6.3 Overland flow generated by natural precipitation events

Between 10th April 2019 – 10th March 2020, the Back Greenriggs rain-gauge recorded 1,357 mm of rainfall (Figure 6.4). The maximum 5-minute rainfall-intensity during monitoring was 5.6 mm (equivalent to 67.2 mm hr^{-1}). In November 2019, the P2 plot was damaged by agricultural traffic, and thus, overland flow results after this date have been excluded for this plot.

Only four overland flow events were generated during the whole monitoring period (Figure 6.5: Supplemental Table 6.S2). The first event occurred on the 24th June 2019, and produced a small amount of overland flow in a single 5-minute period from P2, M1 and M2. The three-remaining overland flow events occurred on 10th December 2019 (following Storm Atiyah, classified in the UK), 8th – 9th February 2020 (Storm Ciara), and 15th – 16th February 2020 (Storm Dennis). The latter three storms produced hydrographs suitable for TF modelling with the Back Greenriggs rain-gauge.

Storm Atiyah produced 38 mm of rainfall, which generated 0.57 mm (1.5 % of rainfall), 0.00 mm (0 % of rainfall) and 1.10 mm (2.9 % of rainfall) of overland flow from plots P1, M1 and M2, respectively. Storm Ciara produced 98.2 mm of rainfall,

Table 6.3: Permeability (K_{sat}) averages, minimum and maximum values, coefficients of variation (CoV), the standard deviation (σ), the number of samples (N), and the factor of difference for pasture plots (P1), pasture plot two (P2), hedgerow wild-margin plot one (M1), and hedgerow wild-margin plot two (M2). Anderson-Darling (AD) and Shapiro-Wilk (SW) normality test are also given.

Parameter	P1	M1[b]	P2	M2[b]	P1:M1 Ratio	P2:M2 Ratio	P1:P2 Ratio	M1:M2 Ratio
K_{sat} geometric mean (mm hr^{-1})	3.43×10^{2}	7.65×10^{3}	2.82×10^{2}	7.64×10^{3}	1:22.3	1:27.1	1:0.822	1:1.00
K_{sat} arithmetic mean (mm hr^{-1})	4.09×10^{2}	8.72×10^{3}	3.18×10^{2}	8.46×10^{3}	1:21.3	1:26.6	1:0.778	1:0.970
K_{sat} median (mm hr^{-1})	3.17×10^{2}	8.78×10^{3}	3.24×10^{2}	7.29×10^{3}	1:27.7	1:22.5	1:1.02	1:0.830
K_{sat} minimum (mm hr^{-1})	1.68×10^{2}	3.27×10^{3}	1.19×10^{2}	4.02×10^{3}	1:19.5	1:33.8	1:0.708	1:1.23
K_{sat} maximum (mm hr^{-1})	8.92×10^{2}	1.78×10^{4}	4.75×10^{2}	1.35×10^{4}	1:20.0	1:28.4	1:0.533	1:0.758
σ (mm hr^{-1})	2.63×10^{2}	4.57×10^{3}	1.42×10^{2}	3.98×10^{3}	1:17.4	1:28.0	1:0.540	1:0.871
CoV. (%)	64.2	52.4	44.8	47.0	1:0.816	1:1.05	1:0.698	1:0.897
N	8	8	7	7	1:1	1:1	1:0.875	1:0.875
AD (ρ)	0.229	0.309	0.229	0.252	NA	NA	NA	NA
SW (ρ)	0.178	0.223	0.201	0.207	NA	NA	NA	NA

[b] *Due to the rapid emptying of the permeameter, and potential violation of Darcy's Law, these are approximate values only.*

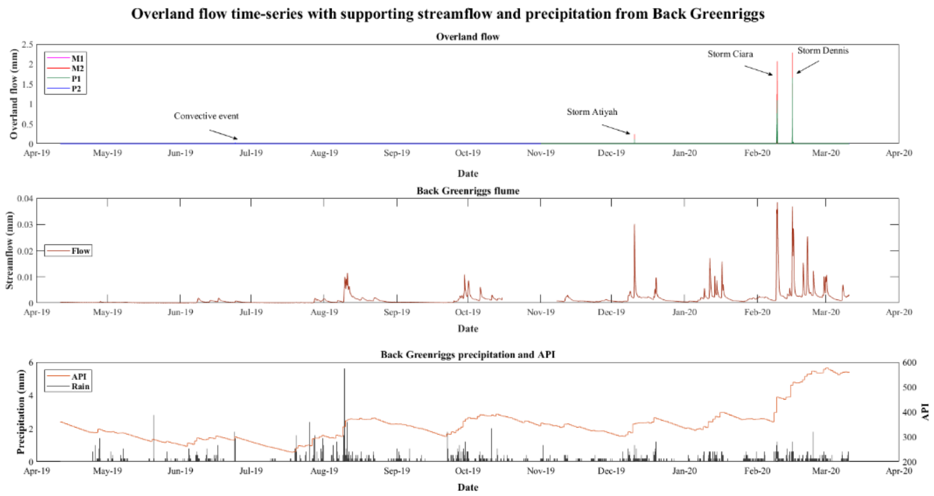


Figure 6.4: The time-series of overland flow from pasture plot one (P1), pasture plot two (P2), hedgerow wild-margin plot one (M1), and hedgerow wild-margin plot two (M2). This is combined with streamflow from the nearby Back Greenriggs flume, and rainfall from the Back Greenriggs rain-gauge (see Figure 6.1). An Antecedent Precipitation Index (API) which started in January 2019 with a 0.99 decay factor (see Wallace and Chappell, 2020a: Equation 4.1: Figure 4.2) is also given, with initial conditions having no effect from the beginning of the monitoring period.

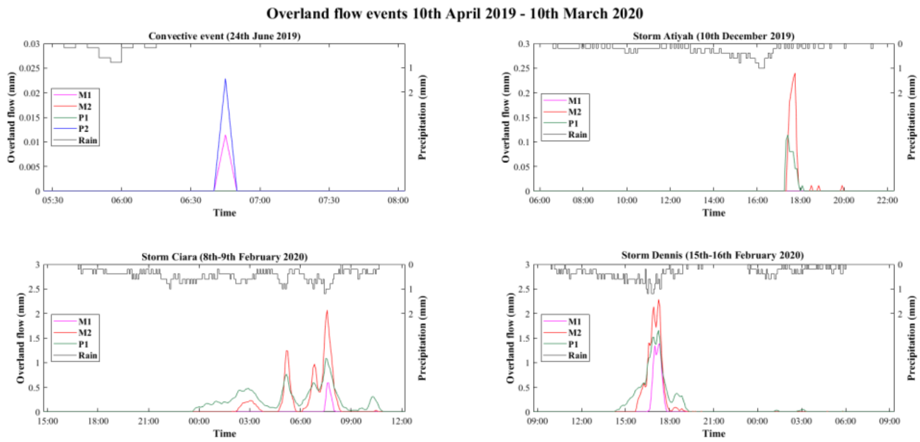


Figure 6.5: The four overland flow events from pasture plot one (P1), pasture plot two (P2), hedgerow wild-margin plot one (M1), and hedgerow wild-margin plot two (M2). The rainfall from the Back Greenriggs rain-gauge is also given for each individual storm. Note that for the convective event on the 24th June 2019, the P2 and M2 hydrographs are identical, and therefore overlap. Also note that P2 monitoring failed in November 2019, and therefore results after this date have been omitted.

which generated 37.63 mm (38.3 % of rainfall), 2.45 mm (2.5 % of rainfall) and 27.52 mm (28 % of rainfall) of overland flow from plots P1, M1 and M2, respectively. Storm Dennis produced 67.6 mm of rainfall, which generated 31.68 mm (46.9 % of rainfall), 11.21 mm (16.6 % of rainfall) and 27.51 mm (40.7 % of rainfall) of overland flow from plots P1, M1 and M2, respectively. Over the whole 11-month record, overland flow in P1 amounted to 5.2 % of total rainfall, while the adjacent hedge-margin plot (M1) produced a much smaller 1 % of total rainfall. Overland flow in the wild-margin plot further upslope (M2) accounted for 4.1 % of total rainfall.

In order to get an understanding of the rainfall to overland flow system, the RIV algorithm was used to identify a single parameter set capable of simulating the three overland flow events. The optimal simulation for P1 is presented in Figure 6.6, for M1 in Figure 6.7, and for M2 in Figure 6.8, with model parameters in Table 6.4. Overall simulation performance is weak for M1 (36.7 %), moderate for M2 (51.0 %) and good for P1 (77.2 %). The standardised unit step response provides an illustration of the simulated water balance for each model (Figure 6.9).

6.6.4 Identifying hydrological responses in the litter layer and upper topsoil to artificial-rainfall

The artificial-rainfall experiment delivered a constant rainfall rate of 22.5 mm hr^{-1} for two consecutive hours (45 mm total) in an attempt to generate overland flow on the four OFPs. Despite the high short-term rainfall intensity, neither IOF nor SOF could be generated within any OFP. During the relatively dry conditions of 11^{th} – 15^{th} April 2019 (Figure 6.4), the moisture-probe data confirmed that the topsoil in each OFP did not reach saturation (i.e., where θ_v equals η) during the artificial-rainfall experiments (Figure 6.10; Table 6.1).

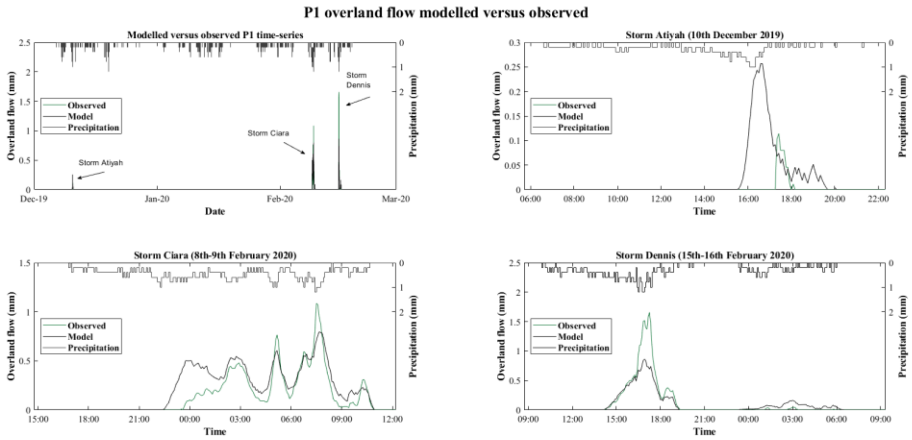


Figure 6.6: The observed and modelled overland flow time-series from pasture plot one (P1), for the full time-series and for individual storms. The P1 model had the best Rt^2 of all overland flow transfer-function models. The modelled magnitudes of overland flow are close to the observed, although Storm Dennis is under predicted and Storm Atiyah is over predicted. The timing of the model responses is very good for Storm Ciara and Storm Dennis, although poor for Storm Atiyah.

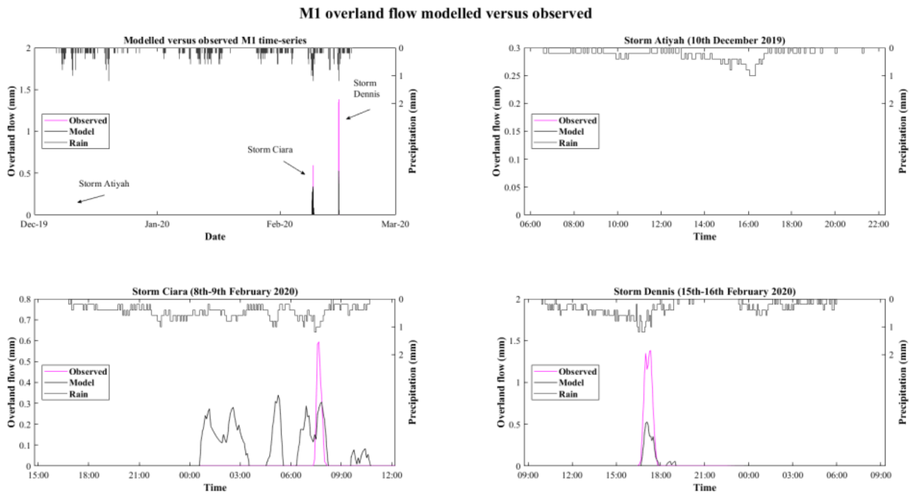


Figure 6.7: The observed and modelled overland flow time-series from hedgerow wild-margin plot one (M1), for the full time series and for individual storms. The model correctly identifies no overland flow for Storm Atiyah. The model relatively closely matches the magnitude of Storm Ciara, but both starts early and finishes late. The model under predicts the peak magnitude of Storm Desmond but is very good regarding timing.

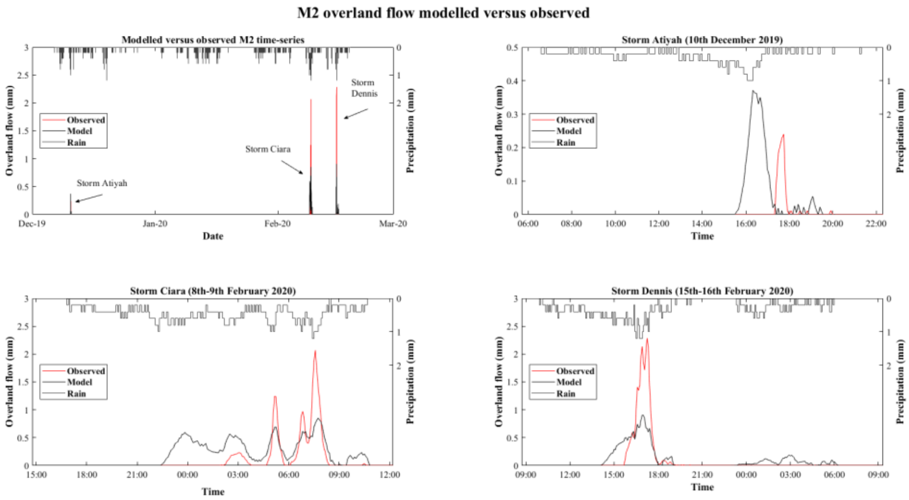


Figure 6.8: The observed and modelled overland flow time-series from hedgerow wild-margin plot two (M2), for the full time series and for individual storms. The modelled magnitudes of overland flow are notably under predicted for both Storm Ciara and Storm Dennis, although Storm Atiyah is slightly over predicted. The M2 model is relatively good with timing regarding Storm Ciara and Dennis, although predicts the Storm Atiyah event early however.

Table 6.4: The Refined Instrumental Variable (RIV) model parameters when predicting overland flow from natural precipitation for pasture plot one (P1), pasture plot two (P2), hedgerow wild-margin plot one (M1), and hedgerow wild-margin plot two (M2). The orders of numerators (n), denominators (m) and pure time-delay (δ) is shown, alongside saturation-threshold (Q_c) and non-linearity (α) for each model. The time constant (TC) and steady-state-gain (SSG) for each component within each model is also given. Model fit is given by the Rt^2 and the Youngs Information Criterion (YIC).

Overland Flow plot	**Response component**	**Model structure [n, m, δ]**	**δ (mins)**	**TC (mins)**	**SSG (mm/ mm)**	**Q_c (mm)**	**α**	**Rt^2**	**YIC**
P1	Gain	[1, 1, 0]	0	33.2	121.4	0.017	1	0.772	-9.188
	Loss	[1, 1, 0]	0	61.1	-80.8				
M1	Gain	[1, 1, 2]	10	17.9	3.4	0.031	0.1	0.367	-7.575
	Loss	[1, 1, 2]	10	34.4	-3.2				
M2	Gain	[1, 1, 1]	5	28.8	230.6	0.017	0.9	0.510	-7.571
	Loss	[1, 1, 1]	5	37.3	-206.4				

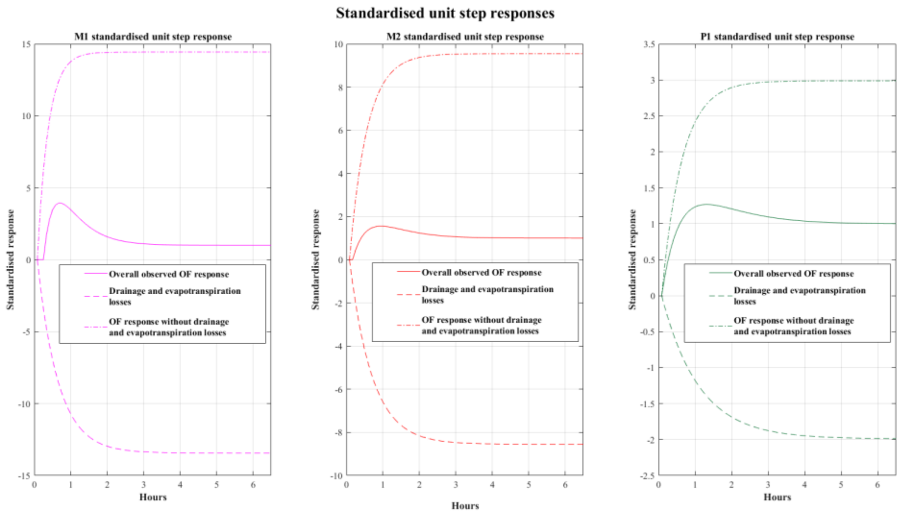


Figure 6.9: The standardised unit step responses for pasture plot one (P1), hedgerow wild-margin plot one (M1), and hedgerow wild-margin plot two (M2) overland flow (OF) models. The overall observed OF response is given, alongside drainage and evapotranspiration losses, and the OF response with these losses removed. These step responses are only applicable above the saturation-threshold of each individual OFP due to the system nonlinearity and threshold enabled phenomenon. Note that the y-axis is standardised and therefore dimensionless.

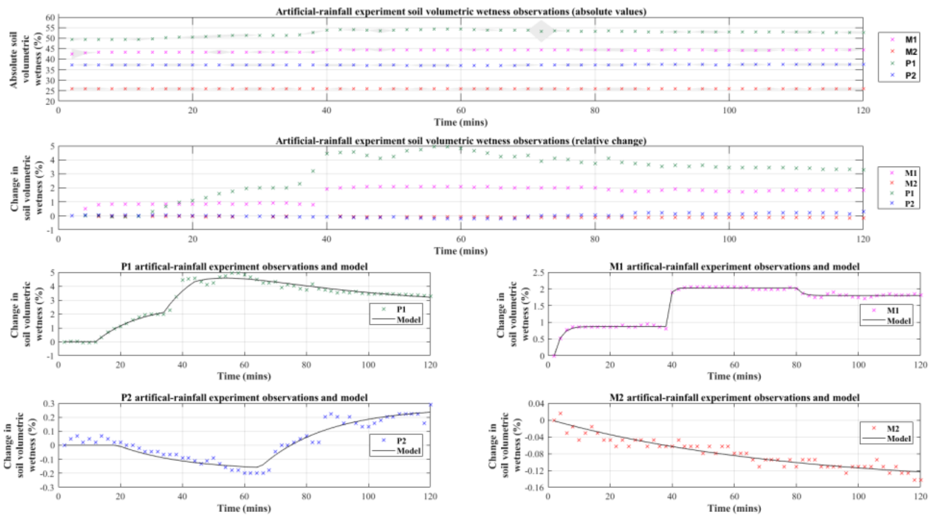


Figure 6.10: The soil volumetric wetness observations from the artificial-rainfall experiment, showing both absolute difference (top plot: with standard error shaded), and the relative difference with the initial soil volumetric wetness baseline removed (all remaining plots). The soil saturation models for pasture plot one (P1), pasture plot two (P2), hedgerow wild-margin plot one (M1), and hedgerow wild-margin plot two (M2) are also given. This figure is paired with Table 6.5.

Whilst no OFP reached saturation in the litter layer and upper topsoil (strictly over 0-6 cm as measured by the moisture-probe), the θ_v in each OFP responded differently to the same artificial-rainfall input (Figure 6.10). For each plot-pair, the initial θ_v was lowest within HWM. Plot M1 had a higher initial θ_v than P2 however, although this may be due to evapotranspiration between experiments (four clear days with moderate temperatures).

The P1 and M1 artificial-rainfall experiments were conducted on the 11th April 2019 (Figure 6.10). The RIV functions identified three responses (hydrological response components in the litter layer and upper topsoil) for each OFP – which were modelled as three first-order models in parallel – and independently from the natural overland flow time-series. Models fit both P1 and M1 OFP time-series extremely well (minimum $Rt^2 > 0.99$), with the YIC indicating that the models are parsimonious, i.e., are not overfitting the data (Table 6.5). Each identified response was represented by a single hydrological component (rather than a combination of components); thus, all components were identified as first-order. Hydrological components were assumed identical between P1 and M1 given plot-proximity and response timing.

The P2 and M2 artificial-rainfall experiments were conducted on the 15th April 2019 (Figure 6.10). Plot M2 did not respond to the precipitation stimulus, with θ_v declining throughout the experiment. Plot P2 responded similarly to M2 for approximately half of the experiment, before a gradual increase began. The above response(s) were identified and models produced very good fits (minimum $Rt^2 > 0.91$: Table 6.5). As before, all components were represented by individual, first-order components.

6.6.5 Overland flow water-quality from wash-off experiments

Physico-chemical properties of overland flow collected from the 'wash-off' experiment are presented in Table 6.6. Soluble reactive phosphorus within water-samples was almost identical between paired-plots, and roughly double in the upslope

Table 6.5: The Refined Instrumental Variable (RIV) model parameters when predicting overland flow (soil volumetric wetness) from the artificial-rainfall experiment for pasture plot one (P1), pasture plot two (P2), hedgerow wild-margin plot one (M1), and hedgerow wild-margin plot two (M2). The orders of numerators (n), denominators (m) and pure time-delay (δ) is shown for each model. The time constant (TC) and steady-state-gain (SSG) for each component within each model is also given. Model fit is given by the Rt^2 and the Youngs Information Criterion (YIC).

Overland Flow Plot	Response component	Model structure [n, m, δ]	δ (mins)	TC (mins)	SSG (m^3 m^{-3} mm^{-1} hr^{-1})	Rt^2	YIC
P1 (Continuous-time)	1 - Macropore	[1, 1, 5]	10	14.7	0.12	0.993	-8.881
	2 - Wetting front	[1, 1, 16]	32	12.1	3.41		
	3 - Loss	[1, 1, 21]	42	42.6	3.51		
M1 (Discrete-time)	1 - Macropore	[1, 1, 1]	2	2.1c	0.04	0.999	-7.482
	2 - Wetting front	[1, 1, 19]	38	0.9[c]	1.15		
	3 - Loss	[1, 1, 40]	80	1.7[c]	0.23		
P2 (Continuous-time)	1 - Loss	[1, 1, 9]	18	22.7	0.01	0.913	-5.291
	2 - Wetting front	[1, 1, 32]	64	20.1	0.42		
M2 (Continuous-time	1 - Loss	[1, 1, 0]	0	150.0	0.01	0.964	-7.454

[c] *As the sampling frequency for soil volumetric wetness during the experiment was every two minutes, having TCs equal to or faster than the sampling frequency suggests that a higher sampling frequency was needed to capture these dynamics. These TCs therefore contain a relatively high level of uncertainty, and should only be taken as approximate values. It is generally suggested to have a sampling interval several times faster than the fastest TC.*

plots (P2 and M2) that of downslope plots (P1 and M1). Within the downslope plot-pair, DTP was roughly four times higher and PTP roughly a quarter higher in P1. For the upslope plot-pair, DTP in M2 was double that of P2, with PTP three-quarters higher within M2.

Nitrogenous compounds, DOC, TS and electrical conductivity were higher/more concentrated in HWM than AIP. Nitrate was 260 % higher in M1 than P1, and 70 % higher in M2 than P2. Nitrate-nitrite was 650 % larger in M1 than P1, and 640 % larger in M2 than P2. ‘Wash-off’ overland flow DOC was 125 % higher in M1 than P1, and 100 % higher in M2 than P2. ‘Wash-off’ overland flow TS was 3970 % higher

Table 6.6: Physico-chemical properties of overland flow samples taken during the 'wash-off' experiment for pasture plot one (P1), pasture plot two (P2), hedgerow wild-margin plot one (M1), and hedgerow wild-margin plot two (M2). Parameters include soluble reactive phosphorus (SRP), dissolved total phosphorus (DTP), particulate total phosphorus (PTP), nitrate (NO_3^-), nitrate-nitrite (NO_3^- NO_2^-), dissolved organic carbon (DOC), total sediment (TS), and electrical conductivity (EC).

Parameter	P1	M1	P2	M2	P1:M1 Ratio	P2:M2 Ratio	P1:P2 Ratio	M1:M2 Ratio
SRP (mg P L^{-1})	3.1×10^{-2}	3.6×10^{-2}	7.1×10^{-2}	7.2×10^{-2}	1:1.2	1:1	1:2.3	1:2
DTP (mg P L^{-1})	1.17×10^{-1}	3.1×10^{-2}	1.00×10^{-1}	2.00×10^{-1}	1:0.26	1:2	1:0.855	1:6.5
PTP (mg P L^{-1})	3.04×10^{-1}	2.38×10^{-1}	1.50×10^{-1}	2.62×10^{-1}	1:0.783	1:1.75	1:0.493	1:1.10
NO_3^- (mg N L^{-1})	1.43	5.14	2.20	3.75	1:3.59	1:1.70	1:1.54	1:0.730
NO_3^- NO_2^- (mg N L^{-1})	6.6×10^{-2}	4.96×10^{-1}	7.2×10^{-2}	5.36×10^{-1}	1:7.5	1:7.4	1:1.1	1:1.08
DOC (mg L^{-1})	4.65	1.04×10^{1}	5.00	1.02×10^{1}	1:2.24	1:2.04	1:1.08	1:0.981
TS (mg L^{-1})	3.29×10^{2}	1.34×10^{4}	8.57×10^{2}	5.51×10^{3}	1:40.7	1:6.43	1:2.60	1:0.411
EC (S m^{-1})	2.37×10^{1}	8.37×10^{1}	NA	NA	1:3.53	NA	NA	NA

within M1 than P1 and 540 % higher within M2 than P2. Electrical conductivity was recorded for the downslope plots only, and was 250 % larger within M1.

6.7 Discussion

6.7.1 Soil and topographic properties that may influence hydraulic properties (Objective I)

Plot-pairings are shown to have similar topographies, and therefore topography should not bias differences between land-uses (Figure 6.3). The variety in slopes between upper and lower plot-pairings captures some natural variation in gradient, and possibly some natural variability in hydrological functioning. Similarities in soil texture, type, pH and profiles, as well as gradient and geologies, justifies the study site as being identical prior to intervention.

Higher ρ_b and lower η in AIP is likely due to topsoil compaction caused by livestock grazing and agricultural machinery within the improved-pasture (Heathwaite et al., 1989; 1990; Drewry et al., 2000; Alaoui et al., 2018). The wild-margins are sheltered from these pressures and vegetation growth in HWM creates an extensive root system that likely reduces topsoil compaction. Walter et al. (2003) similarly observed lower ρ_b values around hedgerows, and Coates (2019) observed a (insignificant) decrease in ρ_b values with increasing distance from a hedgerow. Holden et al. (2019) found hedgerow topsoil to have significantly lower ρ_b than improved-pastures, but found similar ρ_b between hedge-margins and improved-pastures. Elevated ρ_b and reduced η within AIP may negatively influence soil macroporosity in particular, thus lowering topsoil permeability and potentially increasing the likelihood of overland flow generation (Beven and Germann, 1982).

Increased SOM overall within HWM compared to AIP may be due to higher leaf-litter inputs (Hongve, 1999; Bernacki, 2003; Walter et al., 2003). Soil organic matter likely increases aggregate and structural stability, thereby increasing permeability and reducing overland flow likelihood, as well as increasing water-holding capacity (Chaney and Swift, 1984; Chandler and Chappell, 2008). Elevated SOM in HWM may additionally suggest an abundant food supply for soil fauna such as earthworms; which can strongly influence soil hydraulic properties (Capowiez et al., 2009; 2014). Hof and Bright (2010) noted that arable fields with grassy margins had higher earthworm abundance than fields without margins. Holden et al. (2019) did not observe significant differences in earthworm biomass and density between hedge-margins and improved-pastures however.

Highly similar soil pH is likely because of similar soils, as well as plot-proximity and identical historical management. The fairly neutral conditions (Table 6.1) are likely because of historic liming (Holland et al., 2018; Wallace and Chappell, 2020a). Acidic soils have increased sensitivity to disturbance, and therefore a greater likelihood of reduced topsoil permeability and increased overland flow likelihood (Chappell et al., 1996).

6.7.2 Topsoil permeability (Objective II)

Permeability was substantially higher in HWM than AIP, with all summary statistics at least 20 times higher (Table 6.3). Permeability measurements within HWM are approximate values, as at such a high permeability flow through macropores may become turbulent and so not accurately represented by Darcy’s Law (Chappell and Ternan, 1997). Holden et al. (2019) observed similar K_{sat} values between hedge-margins and improved-pastures, although higher K_{sat} directly within hedgerows compared to improved-pastures. Coates (2019) observed higher K_{sat} in very close proximity to a hedgerow which decreased with increasing distance from the feature.

Four provisional mechanisms are proposed to explain the significantly higher K_{sat} observed within HWM. Firstly, extensive hedgerow/shrub root networks can promote the creation of macropores within the soil-matrix, even following root death (Beven and Germann, 1982). Soil biota that improve macroporosity could also differ between land-uses (Capowiez et al., 2009; Hof and Bright, 2010; Capowiez et al., 2014). Holden et al. (2019) observed similar macropore-flow distributions between improved-pasture and hedge-margin soils however. Secondly, HWM is largely sheltered from compacting forces such as agricultural traffic and livestock (Alaoui et al., 2018). This is further supported by ρ_b and η (Tables 6.1-6.2). Fewer compacting forces may suggest that macropores/structural-cracks within HWM are slower to re-seal, and thus, permeability is maintained (Bouma and Dekker, 1978).

Thirdly, extensive root systems within HWM support substantial water uptake to facilitate transpiration, with consequential soil drying. This soil drying is likely amplified by rain sheltering and wet-canopy evaporation provided by the hedgerow (Herbst et al., 2006; Ghazavi et al., 2008; Coates, 2019). Soil desiccation encourages structural-cracks, increasing macroporosity (Bouma and Dekker, 1978). Soil cracking is frequently observed on desiccated Gleysols/Stagnosols during extended warm weather (Beven and Germann, 1982; Chappell and Lancaster, 2007), and has been observed for nearby catchments during this period (Wallace and Chappell, 2019). Holden et al. (2019) observed significantly drier soil in the winter underneath hedge-margins compared to improved-pastures, with Ghazavi et al. (2008) and Holden et al. (2019) observing the same process for hedgerows versus improved-pastures in both summer and winter.

Lastly, higher SOM concentrations in HWM soils may improve aggregate structure (Tables 6.1-6.2), with stable soils capable of supporting macropores in larger numbers (Chaney and Swift, 1984). Follain et al. (2007) observed hedgerow soils to contain high levels of organic carbon. The resultant structural improvement may increase K_{sat} (Baffet, 1984, cited in Walter et al., 2003).

6.7.3 Overland flow generated by natural precipitation events (Objective III)

During the 11-month monitoring period, only four overland flow events were observed, suggesting overland flow is a rare phenomenon and only occurs during very large rain-events. Overland flow occurred surrounding three named mid-latitude cyclones, which resulted in the highest streamflows of the monitoring period (although not necessarily the highest rainfall intensities), as well as during a probable convective event on the 24th June 2019. The latter event was likely convective as no rainfall was observed at the Back Greenriggs rain-gauge during this period, although the plot rain-gauge observed short, intense rainfall (Supplemental Figure 6.S1). Overland flow from this event was possibly linked to the development of hydrophobicity of the vegetation root-mat and litter-layer following a dry period (Burch et al., 1989; Mao et al., 2016).

All plots confirm that streamflow generation is primarily sourced from subsurface flow. During periods of elevated streamflow (a proxy for catchment wetness) however, overland flow can become an active hydrological pathway and contribute water to lower slopes. The importance of the antecedent saturation conditions on overland flow is emphasised given that Storm Dennis was a considerably smaller rain-event than Storm Ciara, yet produced much higher proportions of overland flow in respect to the rainfall input. The overland flow data for this site is of considerable value, as so few experimental sites have such measurements, and this is the first site globally to capture field-based observations of overland flow from hedgerows/wild-margins. Results underline that considerably more overland flow may be produced from improved-pastures in comparison to wild-margins, and that overland flow can be an active hydrological component during large rain-events in upland UK.

Single-parameter sets for the three overland flow events highlight that while the P1 performance is reasonable, the magnitude and timing of the smaller rainstorm (Storm Atiyah) is missed completely (Figure 6.6). This finding could indicate that the representation of the non-linearity in the rainfall to overland flow dynamics is poorly represented by the state of the Back Greenriggs streamflow. Representing the build-up of saturation in the litter layer and topsoil to generate SOF may need a better index of topsoil wetness than catchment-integrated streamflow generated primarily by deeper flow pathways (Ockenden and Chappell, 2011).

The streamflow-threshold for the onset of overland flow in the modelling is identical between P1 and M2 plots, outlining effective rainfall begins at the same degree of catchment-integrated saturation (streamflow) for both OFPs. Both P1 and M2 also demonstrated strong non-linearity in the rainfall to overland flow response, implying a strong influence of catchment-integrated saturation upon overland flow generation. Plot M1 had a much higher streamflow-threshold, implying a higher degree of saturation was required before overland flow occurred, although M1 also contained a weaker non-linearity.

The TCs of the overland flow response (or 'residence-times') for each plot once saturated is comparable at 33 mins (P1), 18 mins (M1) and 29 mins (M2; Table 6.4). Such flashy overland flow responses are slower than those observed on tropical hillslopes on Borneo (Chappell et al., 2004), but much faster than micro-catchment responses dominated by subsurface flow observed elsewhere in the UK (e.g., Jones and Chappell, 2014) – highlighting differences in overland flow response due to scale and place. Model δs are the delay between effective rainfall and the onset of overland flow. Only δs up to 10 minutes were considered due to computational limitations. Hedgerow wild-margin plots had longer δs than P1 (Table 6.4), suggesting they are slower to produce overland flow once the streamflow-threshold was reached.

The SSG is the ratio of output to input at steady-state, and therefore infers the conversion of precipitation into overland flow once the saturation-threshold has been reached. The relative SSG of the loss and gain components within HWM plots were very similar, suggesting fairly small amounts of overland flow production, likely because of high drainage and evapotranspiration rates (Table 6.4: Figure 6.9). The larger gain relative to the loss component in P1 suggests that improved-pastures produce a much higher proportion of overland flow compared to wild-margins, as was observed (Figure 6.9). Due to the system being non-linear with threshold-enabled phenomenon, Figure 6.9 is only interpretable above the specified saturation-thresholds. These piecewise linear models provide both proof and quantification of the conceptual model (Supplemental Figure 6.S3).

6.7.4 Identifying hydrological responses in the litter layer and upper topsoil to artificial-rainfall (Objective IV)

The artificial-rainfall experiments (modelled separately to the natural overland flow time-series) support the natural precipitation overland flow time-series by demonstrating that overland flow is not easily generated even with sustained extreme precipitation intensities, further suggesting that saturation is a key component in overland flow generation. During the P1 and M1 artificial-rainfall experiments, the initial gain in θ_v (component 1) is possibly macropore-flow, as this is likely the fastest hydrological pathway present in the OFPs (although further work is needed to investigate each hydrological pathway within each OFP such as via tracer experiments). The instantaneous M1 response suggests a well-developed macropore structure capable of rapid transport; with the delayed P1 response suggesting a less effective macropore system. Both of these findings are strongly supported by K_{sat} measurements, and to a lesser extent by ρ_b and η (Tables 6.1-6.3). The second θ_v gain (component 2) within both plots is likely a slower, soil-matrix wetting front. This gain component occurs almost simultaneously between OFPs, suggesting similar travel times, and could be explained by a wetting front resulting from the similar soils. The final response (component 3) indicates a θ_v loss, and is possibly evapotranspiration as the drainage rate is assumed quasi-constant throughout the experiment. Artificial-rainfall experiments were conducted on warm, clear days with moderate winds, which may facilitate evapotranspiration. It is likely that uncompensated temperature drift and inherent uncertainty (±2 % θ_v) within the moisture-probe contributed towards this loss component (see Gaskin and Miller, 1996), especially given this loss pathway is so small in comparison to the instrumental uncertainty (Figure 6.10).

The TCs for M1 were very small throughout the artificial-rainfall experiment, highlighting fast system dynamics which concurs with observed responses from the natural overland flow time-series (admittedly at probable different θ_v contents). Some of the TCs for M1 were below the artificial-rainfall experiment moisture-monitoring frequency, indicating that faster sampling (ideally 10-15 seconds) is necessary to accurately quantify hydrological dynamics near the surface. Plot P1 came relatively close to saturation (see Table 6.1 for porosities) and showed much larger TCs, and therefore slower dynamics, which additionally concurs with observations from the natural overland flow time-series (broadly comparable TC to Table 6.4). Differences in TCs between P1 and M1 suggest that wild-margin soils respond more rapidly to precipitation stimuli than improved-pasture soils (Table 6.5).

The SSG in this instance infers the conversion of precipitation into θ_v. Smaller SSGs for M1 compared to P1 suggest that less rainfall is being converted into θ_v accumulation in the litter layer and upper topsoil (Table 6.5). This finding suggests that although wild-margins may respond more rapidly than improved-pastures to precipitation stimuli (smaller TCs), they are likely to be slower to saturate and produce overland flow (smaller SSGs), further confirming overland flow observations.

Within M2 and P2, the θ_v loss component is likely evapotranspiration, as meteorological conditions were similar to previous experiments. As before, inherent uncertainty and uncompensated temperature drift within the moisture-probe likely contributed towards this (Gaskin and Miller, 1996), especially since observed θ_v changes are so small (Figure 6.10). The gain component within P2 is possibly a soil-matrix wetting front, as P2 is the most clay-rich OFP (Table 6.1), and therefore may have slower matrix flow than other OFPs. This gain component could be macropore-flow; however Tables 6.2-6.3 highlight similar permeability between improved-pastures, although unsaturated flow is not accurately represented by K_{sat}. As above, further work is needed to investigate each hydrological pathway within each OFP.

The TC (longer than the experiment) and δ (non-existent) values for M2 underline no observable response to the artificial-rainfall, as θ_v actually declined throughout the experiment. This lack of an observable response is reinforced by small SSG values, which suggests that no input (i.e., rainfall) is being converted into output (i.e., θ_v). The complete lack of response in M2 during the artificial-rainfall experiment is likely caused by the dry initial conditions (see Figure 6.10), and therefore underlines the importance of including non-linearity within the overland flow models. It is possible that the rainfall input was absorbed by the OFP upslope of the soil moisture-probe, or that water was transferred vertically rather than horizontally within the OFP, both of which would prevent the soil moisture-probe from detecting the input.

The SSG of the loss component was identical between P2 and M2. The P2 gain dynamics were roughly half as fast as the P1 gain dynamics, and much slower than M1. These results suggest that improved-pastures have more consistent dynamics in relation to precipitation compared to hedgerow wild-margins, which have both extremely fast (M1) and extremely slow (M2) dynamics, possibly due to more homogenous vegetation and land management. The improved-pastures contained notably different response (latency) times however (different δ values), with P2 having a considerably longer delay and therefore requiring a longer time to saturate. This difference in latency time was possibly caused by differing initial θ_v, alongside soil and topographic variations between OFPs.

6.7.5 Overland flow water-quality from wash-off experiments (Objective V)

The similar SRP concentrations in overland flow between plot-pairs suggests losses are identical between land-uses. Reduced SRP concentrations in the lower paired-plots implies that it is less available downslope. Total phosphorus concentrations (DTP and PTP) revealed inconsistent patterns however.

Higher nitrate and NO_3^- NO_2^- in overland flow from HWM highlights that the surface of wild-margins can release considerably larger quantities of nitrogenous compounds in a flush of overland flow compared to improved-pastures following rewetting. This finding is substantial given that improved-pastures are prone to nitrate flushing (Gordon et al., 2008; Mian et al., 2008). Nitrates are known to accumulate within hedgerow wood and leaf-litter, with soil dryness within HWM decreasing denitrification, SOM mineralization, and microbial immobilisation, all of which may increase nitrate flushing (Bernacki, 2003; El-Sadek, 2007; Gordon et al., 2008; Grimaldi et al., 2012; Benhamou et al., 2013). Higher DOC within HWM overland flow is possibly due to leaf-litter accumulation, as well as increased SOM (Table 6.1; Hongve, 1999). Higher DOC in overland flow alongside higher SOM may suggest higher mineralization within HWM soils, which could further amplify nitrate flushing (Holden et al., 2019).

Higher sediment concentrations within HWM overland flow suggests that wild-margins contain substantially more loose material on the soil surface than improved-pastures, possibly because of aeolian sheltering, increased soil binding, and reduced overland flow (Angima et al., 2002; Walter et al., 2003). The dense grass sward may also have protected the underlying soils within AIP. This sediment disparity was not detectable with the phosphorus chemistry however, suggesting HWM soils to be much less phosphorus-rich. Higher electrical conductivity in HWM overland flow highlights a greater concentration of ions, and therefore supports that an increased number of potential pollutants are stored on the surface of wild-margins in comparison to improved-pastures.

The wash-off experiments highlight that overland flow water-quality is considerably worse from HWM in comparison to AIP, particularly in relation to nitrate, NO_3^- NO_2^- and sediment. The impact of hedge-margins on water-quality during real events is much less clear-cut however, given that natural precipitation intensities (including the artificial-rainfall experiment) rarely generated overland flow, and HWM produced considerably less overland flow than AIP. Further experimentation under more natural rainfall conditions is needed to determine the true mobilisation potential of contaminants, and therefore to determine if hedge-margins may act as sinks or sources of potential contaminants in agricultural catchments. Further experimentation is also needed to determine how long the hydrochemical/water-quality signature is maintained for from each land-use during the foul flush, and how long these potential contaminants take to reaccumulate.

6.8 Implications and conclusions

Hedgerows are commonplace landscape features that provide an array of ecosystem and agricultural services. Despite their widespread presence, minimal data exists regarding how hedgerows influence near-surface hydrology, with no field-based studies directly observing changes to overland flow (and entrained contaminants) induced by hedgerows. This study has quantified changes to topsoil hydraulic properties and overland flow incidence, magnitude and water-quality following the conversion of an improved-pasture to a hedgerow wild-margin after only a relatively short time-period.

The key findings of the research were:

- Hedgerow wild-margins had significantly lower soil dry bulk-density and significantly higher porosity than the improved-pastures. This conclusion implies that introducing wild-margins can fairly rapidly improve soil physico-chemical properties.
- Wild-margins had significantly higher permeability than the improved-pastures (median 2,200 % – 2,700 % higher). Wild-margins may therefore provide flood-mitigation benefits for agricultural catchments, particularly where soils are less

permeable (see Wallace and Chappell, 2019). This finding supports prior modelling assumptions and provides quantitative permeability estimates for future studies.

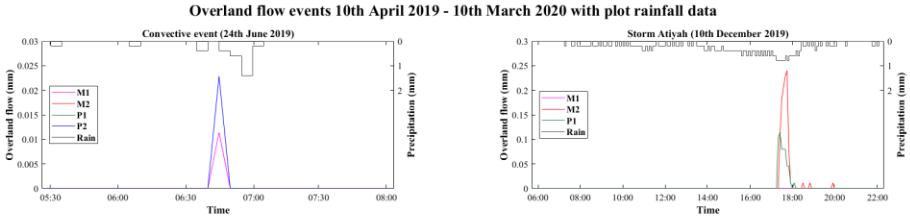
- Overland flow observations and models based on natural precipitation highlight that overland flow only becomes an active hydrological pathway during large rain-events in an example upland UK catchment, almost entirely occurring during periods of peak streamflow. Wild-margins were found to generate considerably less overland flow than improved-pastures, possibly due to increased drainage and/or evapotranspiration. The systems modelling produced a proof and quantification of model concept, outlining that wild-margins require an equal or greater threshold of streamflow (catchment-integrated saturation) for the onset of overland flow in comparison to improved-pastures, and wild-margins were slower to produce overland flow once this threshold had been reached. Systems modelling additionally highlights rapid dynamics within overland flow responses. Future work could repeat this analysis on individual events to assess stationarity in model structure/parameters, as well as contrast overland flow and streamflow hydrographs. Improved representation of the non-linearity due to topsoil saturation is also required (possibly derived from individual precipitation events or artificial-rainfall experiments).
- The artificial-rainfall experiments confirm that improved-pastures saturate faster than hedge-margins, highlighting wild-margins to have extremely variable dynamics in response to precipitation, whereas improved-pastures have more moderate and consistent (but still variable) dynamics. Variable dynamics in both land-uses was likely caused by non-linearity due to differing initial moisture conditions and/or spatial-variability between plots. Future work could repeat this

experiment with pre-saturated plots and differing precipitation intensities/durations.

- Water-quality results from the 'wash-off' experiment outline that wild-margins may contain considerably larger quantities of nitrogenous compounds and sediment on the ground surface compared to improved-pastures. Further experimentation is needed to determine contaminant mobilisation potential during natural rainfall events. Future researchers conducting similar experiments are advised to incorporate soil chemistry/microbiology for improved interpretations of water-quality. These experiments could include measurements of soil nitrogenous compounds or nitrogen fixing bacteria, or other water-quality parameters of interest such as heavy metals or faecal indicator organisms.

The experimental design and findings presented in this manuscript demonstrates the potential value of studies of the hydrological effects of hedgerows on near-surface hydrology. Widespread plot replication upon different soils, within dissimilar farming systems and contrasting climates, and with differing hedgerows in respect to species, age and management is needed to further the hydrological understanding of hedgerows and associated features. Observing overland flow directly from a hedgerow (rather than a wild-margin) would add to understanding, as would the monitoring of hedgerows parallel with hillslope contours. The spatial extent of hydrological improvements caused by hedgerows also requires investigation. Given the limited viability of plot-scale representativeness, a considerable volume of further evidence is required before hedgerows/hedge-margins can be suitably incorporated within larger-scale hydrological models.

6.9 Supplemental materials



Supplemental Figure 6.S1: The two overland flow events from pasture plot one (P1), pasture plot two (P2), hedgerow wild-margin plot one (M1), and hedgerow wild-margin plot two (M2) where plot-rainfall is available. Note that the P2 and M2 hydrographs are identical during the 24th June 2019 convective event, and therefore overlap.

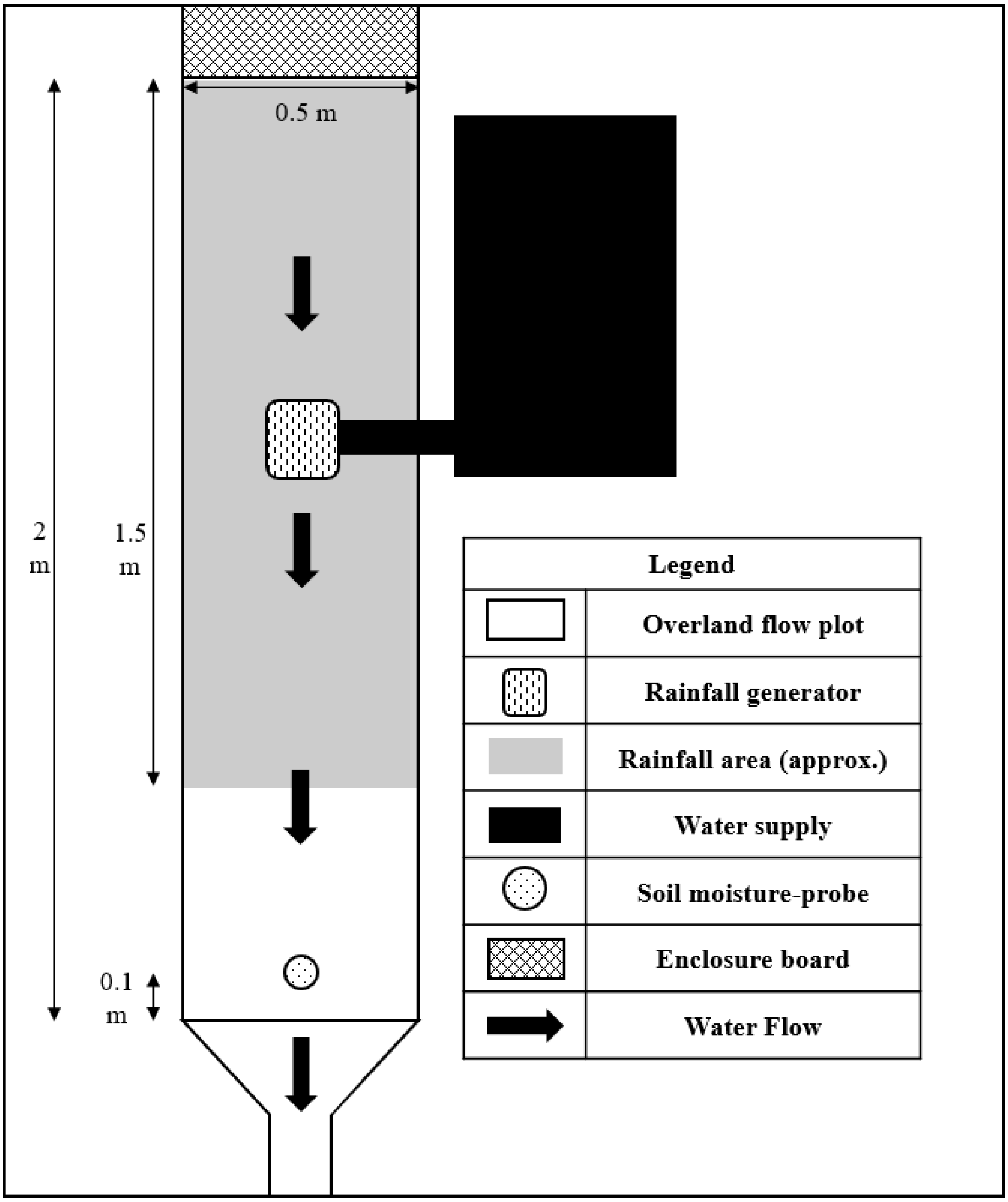
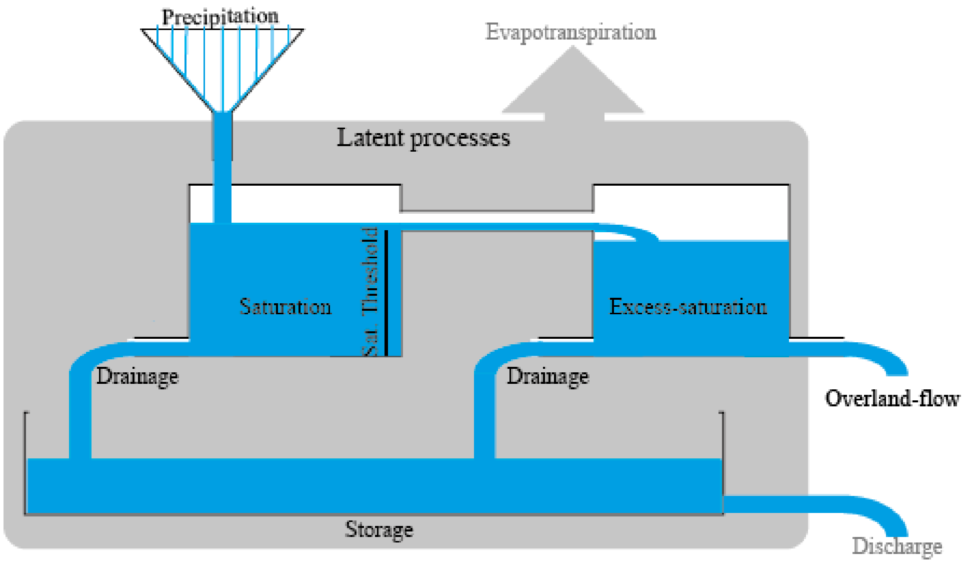


Figure 6.S2: The experimental setup for the artificial-rainfall experiment showing the overland flow plot, the rainfall generator, approximate rainfall area, the water supply, the soil moisture-probe, the enclosure board, and the approximate water flow direction due to gradient.



Supplemental Figure 6.S3: A schematic diagram of the transfer function modelling approach used for the overland flow modelling applied to the natural precipitation time-series. The Back Greenriggs flume provided streamflow data, which was used as a surrogate for the state of the first vessel (to infer saturation of the catchment). A saturation-threshold (Sat. Threshold), equivalent to Q_c in Equation 6.2 is given, highlighting that different Overland Flow Plots (OFPs) required different levels of saturation before effective rainfall occurred and the potential for overland flow was generated. The discharge and evapotranspiration of the system was not implicitly modelled, but is shown to close the water balance.

Supplemental Table 6.S1: Available published literature (excluding theses) with quantitative surface hydrological measurements of author termed 'hedgerows'.

Author	Location	Site and hedgerow description	Surface hydrological observations	Absolute change	Magnitude of change/ relative change
Blanuša and Hadley, 2019	Berkshire, UK	Greenhouse experiments of individual hedge plants and model hedge troughs (primarily urban hedge species), with *Crategus monogyna* (hawthorn) values extracted	Runoff delay	21 minutes of 28 mm hr^{-1} rainfall (9800 ml total) until runoff from fully saturated trough	377 % increased runoff delay compared to bare substrate
				38.7 minutes of 28 mm hr^{-1} of rainfall (18060 ml total) until runoff from unsaturated trough	117 % increased runoff delay compared to bare substrate
			Runoff volume	15 ml of runoff after 40 minutes of 6 mm hr^{-1} rainfall (4030 ml total)	0.4 % of rainfall converted to runoff
				739 ml of runoff after 1 hour of 28 mm hr^{-1} rainfall (28000 ml total)	2.6 % of rainfall converted to runoff
Ghazavi et al., 2008	Brittany, France	Oak (*Quercus robur*) hedgerow trees, 50+ years old, 25 metres tall	Interception	NA	28 % per event for leafed period, 12 % per event for leafless period

Table 6.S1 (continued):

Author	Location	Site and hedgerow description	Surface hydrological observations	Absolute change	Magnitude of change/ relative change
Herbst et al., 2006	Wiltshire, UK	Predominantly *Crategus monogyna* (hawthorn) hedgerow, with some *Acer campestre* (field maple), 3.3-4 m tall	Stemflow	1.68 mm of rainfall (168 ml) as stemflow over summer, 2.54 mm of rainfall (254 ml) as stemflow over winter	0.2 % of total rainfall in summer, 0.5 % total rainfall in winter
			Wet-canopy evaporation (interception loss)	NA	52 % of gross rainfall over canopy area, or 23 % of gross rainfall over affected area
			Interception storage capacity	2.6 mm when leaved, 1.2 mm when leafless	NA
Herbst et al., 2007	Wiltshire, UK	Predominantly *Crategus monogyna* (hawthorn) hedgerow, with some *Acer campestre* (field maple), 3.3-4 m tall	Transpiration rates	8 mm (800 ml) d^{-1} at peak 3 mm (300 ml) – 3.5 mm (350 ml) d^{-1} on average	337 % more transpiration than average daily rainfall 64 % - 91 % more transpiration than average daily rainfall
			Stomatal conductance	Hawthorn maximum of 335 mmol m^{-2} s^{-1} Field maple maximum of 260 mmol m^{-2} s^{-1}	NA NA

Table 6.S1 (continued):

Author	Location	Site and hedgerow description	Surface hydrological observations	Absolute change	Magnitude of change/ relative change
Holden et al., 2019	North Yorkshire, UK	Hedgerow was predominantly *Crategus monogyna* (hawthorn), with *Sambucus nigra* (elder) and *Ilex aquifolium* (holly). Site was upon a well-drained loamy calcareous brown earth from the Aberford series of Calcaric Endoleptic Cambisols. Hedgerows (6) were compared against nearby hedgerow-margins (6), arable fields (3) and improved-pasture fields (3)	Manual soil volumetric wetness (0-6 cm)	Annual mean volumetric wetness ~25 %	~7.5 % drier than arable, ~10 % drier than hedgerow margin, ~12.5 % drier than improved-pasture
				Summer mean volumetric wetness ~17.5 %	~7.5 % drier than arable, ~10 % drier than hedgerow margins or improved-pasture
				Winter mean volumetric wetness ~30 %	~7.5 % drier than arable, ~10 % drier than hedgerow margin, ~15 % drier than improved-pasture
			Time from peak rainfall intensity to peak soil volumetric wetness	3.5 h mean time	1 hour slower than arable (2.5 h), 1.3 hours slower than improved-pasture (2.2 h)
			Time from start of rainfall to peak soil volumetric wetness	5 h mean time	2 hours slower than arable (3.0 h), 1.2 hours slower than improved-pasture (3.8 h)
			Saturated hydraulic conductivity (0-10 cm)	Geometric mean ~21.6 mm hr^{-1}	~880 % more permeable than margin (~2.2 mm hr^{-1}), ~5,300 % more permeable than improved-pasture (~0.4 mm hr^{-1}), ~31,000 % more permeable than arable (~0.07 mm hr^{-1})
			Saturated hydraulic conductivity (10-20 cm)	Geometric mean ~72 mm hr^{-1}	~400 % more permeable than margin (~14.4 mm hr^{-1}), ~1,200 % more permeable than improved-pasture (~5.4 mm hr^{-1}), ~36,000 % more permeable than arable (~0.2 mm hr^{-1})

Supplemental Table 6.S2: The total overland flow volume in mm equivalent per event for pasture plot one (P1), pasture plot two (P2), hedgerow wild-margin plot one (M1), and hedgerow wild-margin plot two (M2). Note that the P2 plot was damaged in November 2019, and data following this date have been excluded for this plot.

Overland flow volume	**P1**	**P2**	**M1**	**M2**	**Overland flow volume**
24th June 2019 – Convective event (mm)	0.00	0.02	0.01	0.02	**24th June 2019 – Convective event (mm)**
10th December 2019 – Atiyah (mm)	0.57	NA	0.00	1.10	**10th December 2019 – Atiyah (mm)**
8th – 9th February 2020 – Ciara (mm)	37.63	NA	2.45	27.52	**8th – 9th February 2020 – Ciara (mm)**
15th – 16th February 2020 – Dennis (mm)	31.68	NA	11.21	27.51	**15th – 16th February 2020 – Dennis (mm)**
Cumulative total overland flow volume (mm)	69.88	0.02[d]	13.67	56.15	**Cumulative total overland flow volume (mm)**

[d] *This is only representative of the 24th June 2019 event due to damage to the P2 overland flow plot.*

6.10 Conflict of interest statement

The authors declare no conflicts of interest.

6.11 Acknowledgements

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6.12 Data availability

The data that supports the findings of this study are available from the corresponding author upon reasonable request.

7 DRY-STONE WALLS

Wallace, E.E., Kretzschmar, A., Hankin, B., Page, T.J.C. and Chappell, N.A. (2021). The effect of dry-stone walls on localised hydrological functioning.

7.1 Brief introduction to paper

In many upland and colder areas of the United Kingdom, harsher climates often combine with poorer soils to inhibit the growth of woody vegetation, making the creation of a biotic stock-proof barrier of any reasonable length very difficult, even with the hardiest of hedgerow species. In many such regions, the most common barrier is the dry-stone wall. These barriers are constructed without mortar or wet adhesive materials, often directly onto the subsoil, and are predominantly made from locally sourced materials found during the ploughing and clearance of nearby fields, or were quarried from nearby sources (as well as occasionally extracted from rivers and streams).

Dry-stone walls share (but also exclude) many benefits of hedgerows in relation to both ecosystem and agricultural services (see Williamson, 2002), with additional benefits of:

- Requiring less frequent maintenance.
- Are largely unaffected by soil type or meteorological conditions.
- Are naturally longer lasting and more stock-proof.
- Offer an immediate stock-proof barrier.
- Provide increased shelter to livestock.
- Are resistant against fire, disease and rot.
- Can more clearly define property boundaries (e.g., wall heads).

- Can incorporate stiles as a minor alteration to the typical construction process.
- Reduce the risk of injury/infection to livestock.
- Can more easily be relocated/reoriented.

For these reasons (alongside socioeconomic and political reasons in relation to the Enclosure Acts), dry-stone walls are extremely prevalent throughout upland UK catchments, including the Eden catchment, and are the dominant boundary in more upland regions such as the western and southern sections of the Lowther catchment.

Due to the presence of their foundation, dry-stone wall boundaries parallel to hillslope contours may alter surface hydrodynamics by acting as a physical barrier against overland flow and shallow-subsurface flows. Dry-stone walls may therefore directly disrupt these hydrological pathways, potentially reducing slope hydrological connectivity and therefore both flood-risk and water-quality degradation by restricting water movement downslope, as well as potentially offering rain-shadowing effects. Constructing and maintaining these barriers may therefore be seen as an improved-pasture intervention that is adoptable within many upland farming systems in both the UK and overseas.

Objective 4) Quantify the effect of dry-stone wall boundaries within sloped areas of agriculturally-intensive improved-pastures upon changes to soil volumetric wetness (θ_v) above and below the barrier, and therefore infer changes to overland flow and rapid shallow-subsurface flow likelihood by assessing the impedance of θ_v transfer downslope.

7.2 The effect of dry-stone walls on localised hydrological functioning

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7.3 Abstract

Dry-stone walls (distinct from (dry-)stone terraces) are an extremely common agricultural boundary feature in many regions of the world. Despite such a widespread global presence, to the knowledge of the authors, no study has ever investigated the effect of dry-stone walls upon hydrological functioning at the plot-field-scale through hydrological observations. This study assessed whether dry-stone walls were impeding soil moisture transport downslope at 23 improved-pasture sites throughout Cumbria, UK, through field-measurements of soil volumetric wetness above and below each dry-stone wall, and subsequent statistical analysis.

Results highlight very high spatial-variability in soil volumetric wetness up to 15 m from the dry-stone wall (in mostly 16 m x 16 m sampling grids), and that soil volumetric wetness distributions within improved-pastures are likely non-normally distributed during saturated and near-saturated conditions. The effect of the dry-stone walls on soil volumetric wetness was very inconsistent at the full-grid (mostly 16 x 16 m) scale, suggesting a relatively minor impact of the dry-stone wall at such scales. At smaller scales, soil volumetric wetness was predominantly not significantly different above or below the dry-stone wall, although particularly at the 0 m – 3 m scale, upslope of the dry-stone walls was shown to be significantly wetter than downslope of the dry-stone walls in a considerable minority of scenarios, implying some very localised influence of the dry-stone wall on soil volumetric wetness. The orientation of the dry-stone wall appeared to have some control in determining soil volumetric wetness up to 3 m from the boundary, highlighting a possible rain-shadow effect.

7.4 Introduction

Dry-stone walls are stone barriers constructed throughout many agricultural regions of the world. These barriers are created from local materials (often superficial geology) without the use of adhesive components such as mortar, cement, or clay, and have a durability of many centuries. Within pastoral systems, dry-stone walls are primarily used as enclosures for livestock management and to highlight property boundaries; being common within Great Britain, Ireland, Scandinavia, France, Spain, Switzerland, Italy, Slovenia, Croatia, Greece, Cyprus, the United States of America, Canada, Australia, New Zealand and further afield (Carey et al., 2008; Norton et al., 2012; Collier, 2013; Johnson and Ouimet, 2014; 2016; Bunce et al., 2018; UNESCO, 2018). Within Great Britain there are approximately 174,000 km of wall features, the majority of which being dry-stone wall (Carey et al., 2008; Norton et al., 2012). Within upland England, Wales and Scotland, these boundaries encompass 53.9 %, 46.7 % and 43.3 % of all linear features, respectively, making them a dominant landscape feature in such regions (Bunce et al., 2018).

Dry-stone walls are distinct features from (dry-)stone terraces (which are sometimes also termed 'dry-stone walls' due to lacking adhesive components); as dry-stone walls are not buried to create a stepped, landscape effect as commonly seen with (dry-)stone terraces in mountainous regions, especially in the Mediterranean and in parts of Asia, Africa and Latin America. Dry-stone walls are more similar to the parats and parets seen in the Balearic Islands (see Grimalt and Rossello, 2018), although these features are seldom built within large stream channels. Dry-stone walls also have similarities to the mounded 'stone hedgerows' of Central Europe (see Kovář et al., 2011), although dry-stone walls are constructed of much larger, interlocking stones.

(Dry-)stone terraces have been widely shown to influence hydrology (Gallart et al., 1994; Arnáez et al., 2015; Kovář et al., 2016; Calsamiglia et al., 2018; Preti et al., 2018; Mesfin et al., 2019; Moreno-de-las-Heras et al., 2019; Ran et al., 2020), as well as sedimentology (Lesschen et al., 2008; Arnáez et al., 2015; Kovář et al., 2016; Calsamiglia et al., 2018; Camera et al., 2018; Moreno-de-las-Heras et al., 2019; Pijl et al., 2020), ecology (Manenti et al., 2014; Assandri et al., 2018), and culture (Assandri et al., 2018; UNESCO, 2018), throughout many regions of the world (see e.g., Arnáez et al., 2015; Deng et al., 2021). It is therefore plausible that dry-stone walls could similarly influence such ecosystem services, with a further need to understand these effects given how internationally widespread dry-stone walls are (Marshall and Moonen, 2002; Collier, 2013; UNESCO, 2018; Grove et al., 2020; Hollingsworth and Collier, 2020).

Despite such a significant global presence, very few studies have directly investigated the effect of dry-stone walls on hydrology. Through hydraulic modelling approaches of the Wharfe catchment (North England, UK), Tayefi et al. (2007) and Yu and Lane (2010) both suggest that dry-stone walls are largely impermeable to flow (excluding for the location of gates), and can be an important control on flood routing. Yu and Lane (2010) additionally suggest that dry-stone walls may be capable of constraining flow into compartments (the pasture enclosures) during flood events. Similarly, through hydraulic modelling of a flash flood event in the Girona catchment (Alicante Province, Spain), Segura-Beltrán et al. (2016) concur that ‘agricultural walls’ influence flood routing. Tayefi et al. (2007), Yu and Lane (2010), and Segura-Beltrán et al. (2016) all therefore conclude that water is slowed/ stopped by the presence of dry-stone/ agricultural walls, and potentially that such features are substantially influencing hydrological functioning within a catchment, particularly during flood conditions.

Prior hydrological studies involving dry-stone walls have been focused towards hydraulic modelling approaches (i.e., Tayefi et al., 2007; Yu and Lane, 2010, Segura-Beltrán et al., 2016); consequently, there is a lack of direct observations of how (and if) dry-stone walls influence hydrological/ hydraulic processes at the plot-field scale. There is a need for such observations as these can support/oppose assumptions made about the hydrological processes associated with dry-stone walls, which can further improve and/or validate hydrological/hydraulic models. This research would ultimately facilitate improved hydrological understanding of such widespread, yet understudied agricultural features.

Common non-lithic agricultural boundaries such as hedgerows and their associated features have been shown to significantly influence surface hydrological pathways and processes at the plot-field scale (e.g., Blanuša and Hadley, 2019; Holden et al., 2019; Wallace et al., 2021). (Dry-)stone wall terraces have also been shown to influence surface hydrological pathways and processes (Arnáez et al., 2015; Ran et al., 2020; Deng et al., 2021). It is therefore plausible that dry-stone walls may also influence surface hydrological pathways and processes at the plot-field scale. Coates and Pattison (2015) hypothesise that dry-stone walls could reduce hydrological connectivity and may possibly change soil moisture levels by altering soil structure and porosity, thus influencing localised hydrological processes and possibly wider hydrological functioning. It is therefore possible that dry-stone walls are (partially-)impermeable and are reducing water transfer across the boundary, potentially being particularly useful if located on slopes or immediately adjacent to streams (Tayefi et al., 2007; Yu and Lane, 2010).

To investigate the influence of dry-stone walls on hydrological processes, specifically (near-)surface saturation and hydrological connectivity, an intensive fieldwork campaign was undertaken during winter-spring of 2017/2018 and the autumn of 2018, within Cumbria, UK. Extensive soil volumetric wetness measurements were taken above and below 23 dry-stone walls present on slopes throughout the landscape, to assess if the dry-stone walls were acting as barriers to moisture transfer downslope during predominantly saturated and near-saturated conditions. Soil volumetric wetness distributions were then statistically assessed and compared at varying distances from the boundary, as well as with respect to orientation, to assess for possible rain-shadowing effects of the feature.

Thus, the objectives of this study are:

I) To statistically contrast soil volumetric wetness distributions at varying distances above and below dry-stone wall boundaries during (near-)saturated conditions.

II) To statistically assess the impact of orientation of the dry-stone wall on soil volumetric wetness above and below dry-stone wall boundaries during (near-)saturated conditions.

7.5 Site description, dry-stone wall design, and methodology

7.5.1 Study sites

Twenty-three dry-stone walls that travelled approximately parallel with hillslope contours were selected throughout the River Lowther catchment, Cumbria, UK, (a sub-catchment of the River Eden catchment: Figures 7.1-7.2). The number of dry-stone walls was deemed a reasonable trade-off between the number of walls measured given the size of measurement grids (see Sections 7.5.3-7.5.4 for details), and financial and time constraints. All dry-stone walls were selected as being constructed upon Stagnosol soils and glacial drift, both of which can facilitate the generation of overland flow and rapid shallow-subsurface flows, and resultantly increase flood-risk and water-quality degradation (Jarvis et al., 1984; Hankin et al., 2018). Cumbria is particularly well suited to this investigation given that there are over 15,000 km of dry-stone wall in the county alone, the second highest length in any English county, as well having a very high density (3,070 m^2 km^{-1}) of dry-stone walls (Land Use Consultants, 2007). The Lowther catchment was specifically chosen due to the number of dry-stone walls locally present (Figure 7.2), as well as the catchment (and downstream catchments) sensitivity to flooding and water-quality degradation.

The local climate from Shap weather station (54°30'49"N, 2°40'40"W: 301 masl) is wet temperate, with annual average maximum and minimum temperatures of 11.5 °C and 4.1 °C, respectfully (Met Office, 2020). The long-term (1981-2010) annual rainfall average is 1,779 mm (Met Office, 2020).

7.5.2 Local dry-stone wall design

Most dry-stone walls in northern England were constructed in the late 18th – mid 19th century following local Inclosure (Enclosure) Acts, although the precise date of construction is unknown for most boundaries (Kain et al., 2004; Whyte, 2006). The construction style and materials as well as the dimensions of dry-stone walls varies in different regions, as well as over time and with local conditions, and a relatively broad generalisation for Cumbrian dry-stone walling design is given here (see Rollinson

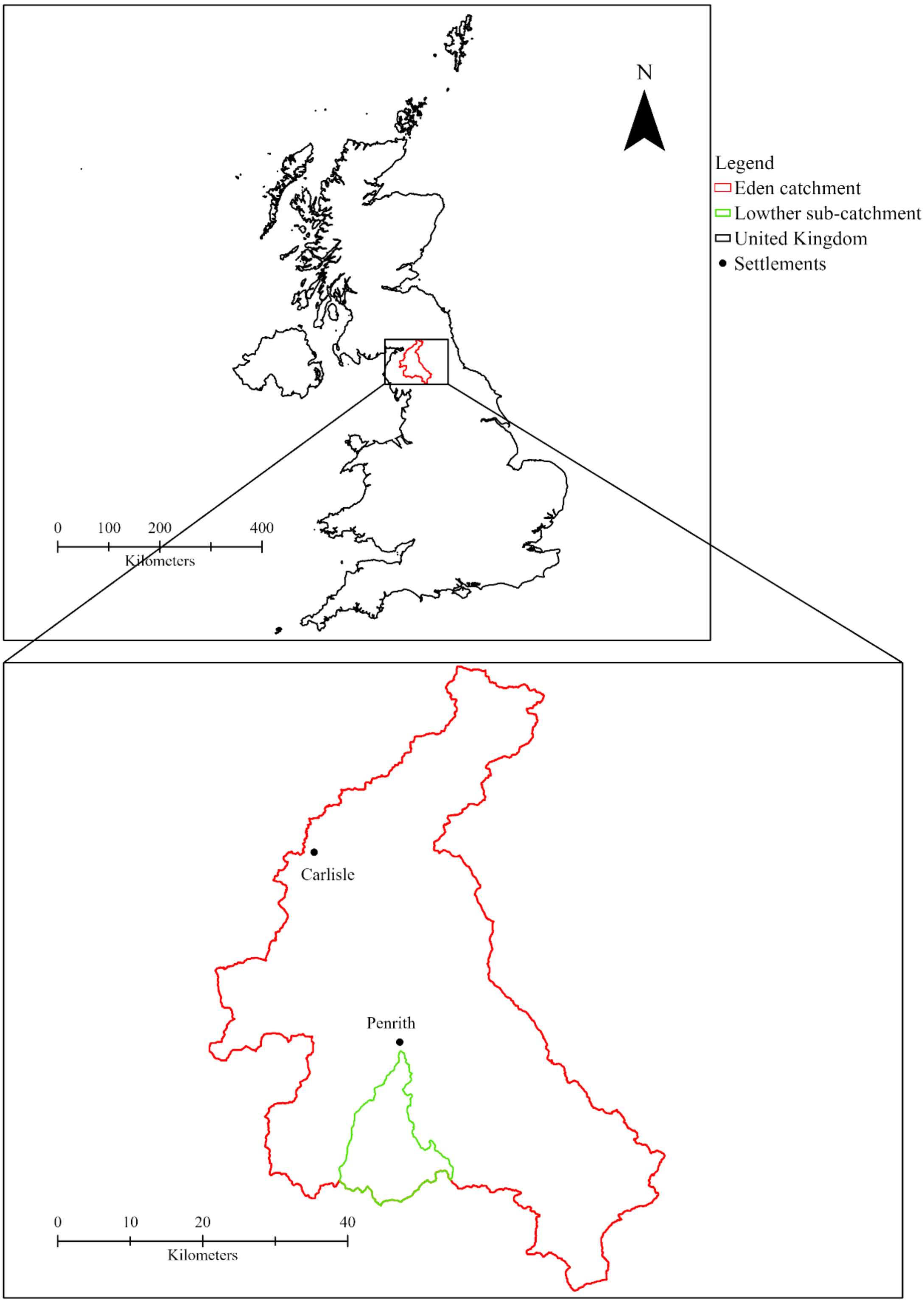

Figure 7.1: The location of the Eden catchment and Lowther sub-catchment in a UK context, alongside major settlements in the Eden catchment.

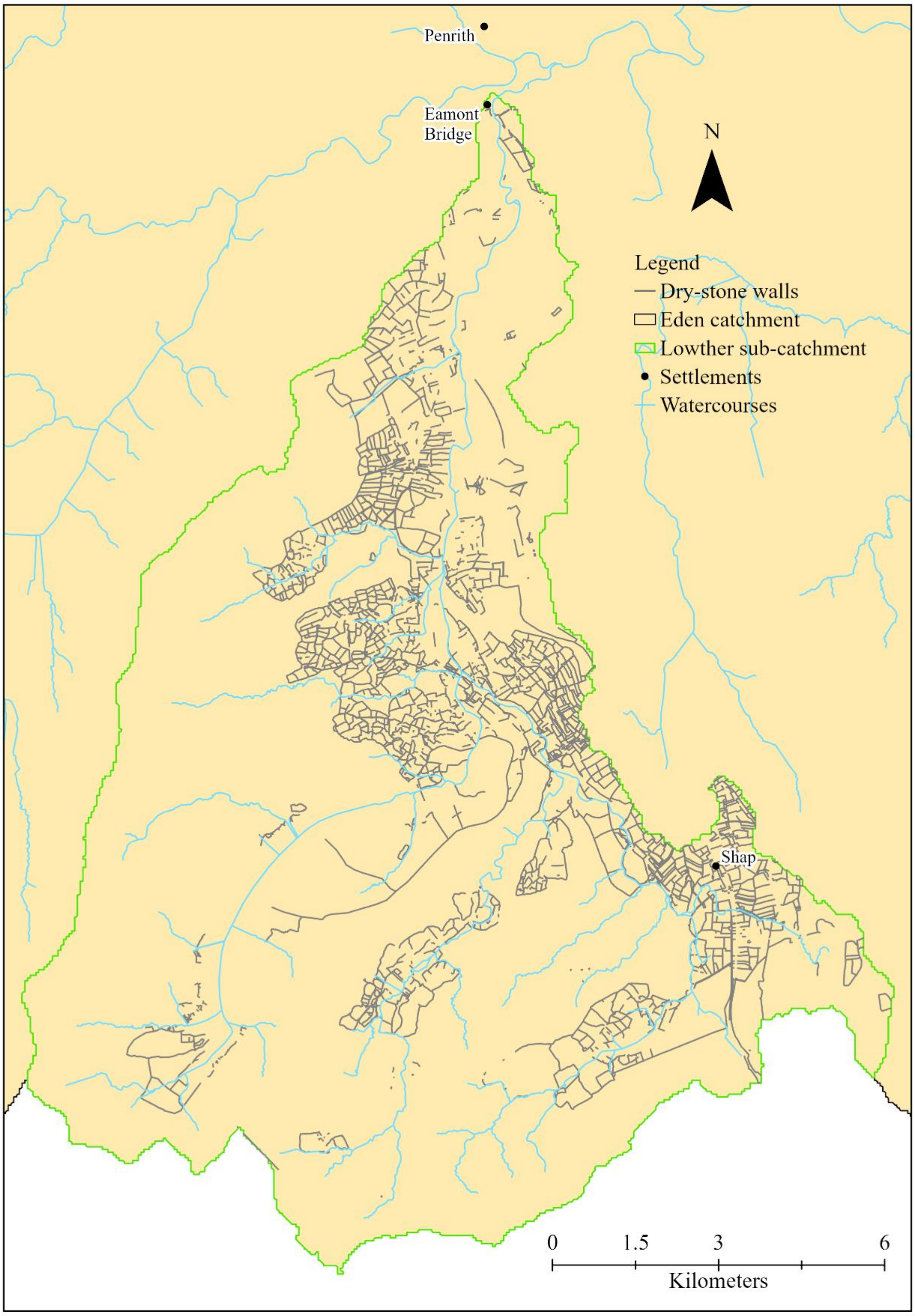

Figure 7.2: The manually calculated prevalence and extent of dry-stone walls within the Lowther sub-catchment.

1969; Bodman, 1984; Williamson, 2002; Garner, 2007; Winchester, 2016; for more in-depth dry-stone wall design).

The initial stage of construction of a dry-stone wall involves digging a trench and removing turf and topsoil until a firm-bedding is found, often on top of either subsoil or bedrock. This trench is levelled and large and relatively square foundation (also known as footing) stones are then added in two rows (a double-line) on either side of the trench. The widest and flattest face of each foundation stone is laid facing downwards so that the foundation will not sink under the weight of the overlying stones. For added strength, the longest side of the stone is orientated into (rather than along) the wall. The gap between the double-line of foundation stones is then filled with smaller, irregular shaped 'packing' stones (also called fill or hearting). The dimensions of the base of the wall vary due to several factors, with the foundation typically being ~ 70 cm – 80 cm in width and ~ 15 cm in depth.

Slightly smaller stones are then added on top of the initial foundation stones, with further packing stones added. These outer stones are laid either flat or sloped slightly downwards, to avoid funnelling rainwater into the centre of the wall where frost expansion can rapidly damage the wall. This process is then repeated, with stones gradually reducing in size as the height of the wall grows. Each new layer of stones is positioned to overlay joints in the stones below (termed breaking or covering the joints), and each new layer brings the double-line closer together, with this angle known as the taper or batter. At certain intervals, the wall will be levelled off and throughstones added which are large stones that connect the double-line of stones together for added strength, with the sections between throughstone insertions known as lifts.

Once acceptable dimensions have been reached (typically ~ 1.15 m in height and ~ 36 cm – 41 cm in width), relatively large and often flat stones called copestones (also called camstones or combers) are added on the top of the wall to further connect the double-line of stones together and to add weight to the wall, as well as to deter sheep. The final outcome is a dry-stone wall typically 1.4 m – 1.5 m tall (Figure 7.3), which is usually adequate to control livestock and can last multiple centuries with sufficient maintenance.

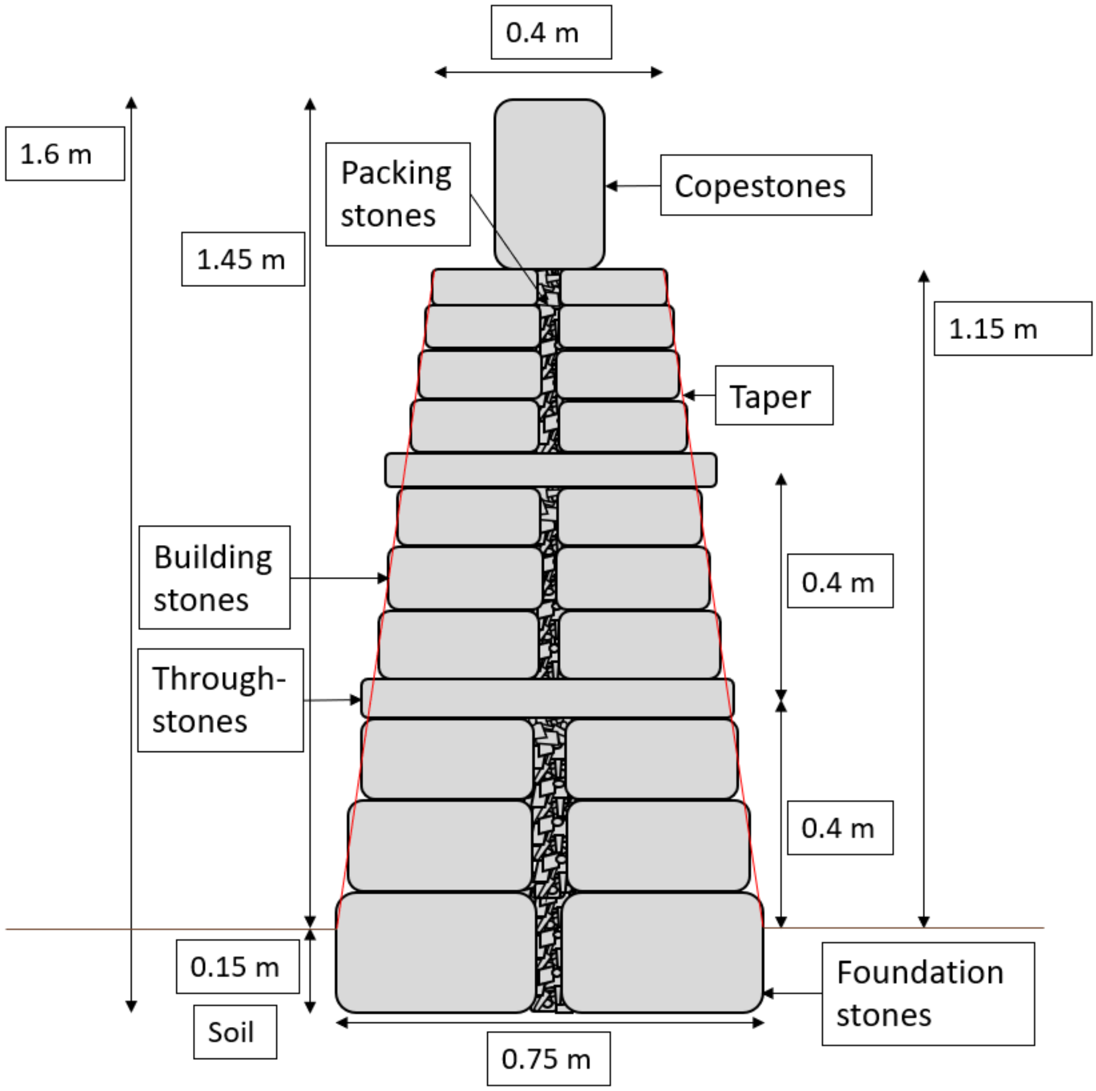

Figure 7.3: The generalised components and dimensions of a dry-stone wall typically seen in Cumbria.

7.5.3 The potential impact of dry-stone walls upon hydrological processes

Given the width of the dry-stone wall as well as the depth of the foundation, it is therefore plausible that these features could impede water transport downslope (Figure 7.4). This impedance could be in the form of direct disruption to overland flow pathways, as well as interfering with relatively near-surface hydrological pathways. Yu and Lane (2010) directly suggest that dry-stone walls are impermeable to overbank flow, only losing effectiveness when these features become submerged (or at the location of gates).

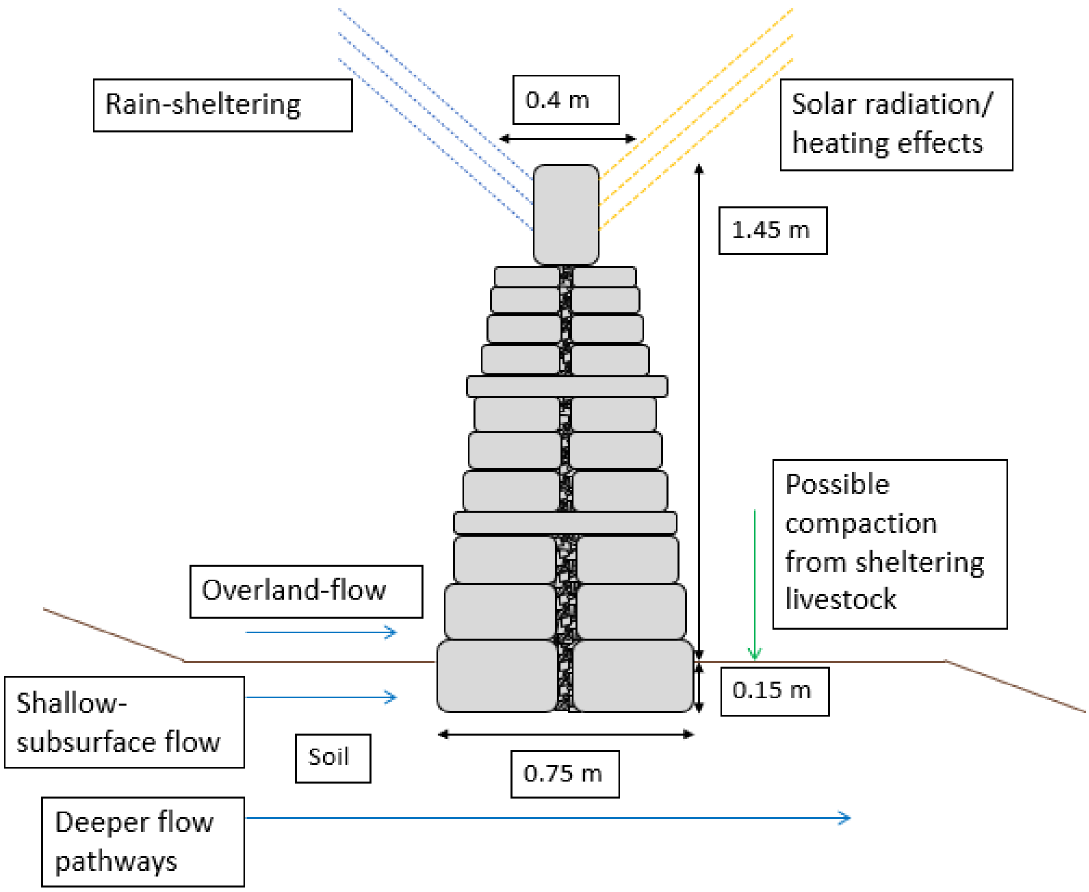

Figure 7.4: The possible direct and indirect impacts of the dry-stone wall on hydrological processes.

Dry-stone walls may both directly and indirectly alter soil hydraulic functioning and broader hydrological processes (Coates and Pattison, 2015). The walls themselves may directly compress underlying soil due to their weight, as well as increase localised soil compaction by providing shelter to livestock during precipitation or storm events, as well as from sunlight during warmer periods. The frequent, qualitative observation of livestock sheltering behind dry-stone walls additionally suggests that walls offer some rain and/or wind shadowing effects depending upon its orientation and the prevailing wind directions, as well as potentially affecting solar radiation receipt in respect to aspect of the wall. The walls may also indirectly alter soil chemistry, biota and vegetation (and resultant water-quality) through concentrated excretal returns of these sheltering livestock, as well as though biogeochemical weathering of the wall materials (particularly limestone).

7.5.4 Experimental design

A total of 23 dry-stone wall sites were selected throughout the Lowther catchment that were situated within improved-pastures, and had identical management practices above and below the barrier. The number of sites monitored was deemed the maximum number of walls that come be measured given time and financial constraints, and also captured some degree of spatial variability. Measurements were taken at each wall once during autumn-winter or winter-spring conditions (sampling dates of 1st Feb 2018; 26th Feb 2018; 10th May 2018; 11th May 2018; 13th November 2018; 14th November 2018; 21st November 2018; 22nd November 2018) to increase the likelihood of saturated and near-saturated conditions occurring, which is when local flood-risk is elevated and surface hydrological pathways e.g., SOF, are likely.

Both upper and lower grids had to be mapped as containing identical surficial geologies and soil types, as well as have highly similar vegetation (*Lolium spp.*). This acceptance criteria were to support that observed differences could be attributed to the presence of the intervention (dry-stone wall), rather than inherent land-use differences or site dissimilarities. Any site that contained a non-uniformity feature such as a farm track or slight variation in vegetation but was retained in analysis is clearly highlighted and this feature described.

7.5.5 Soil volumetric wetness sampling

Topsoil volumetric wetness was measured *in-situ* in the field in predominantly 16 m x 16 m grids above and below the dry-stone wall at each site at a 1 m resolution (with U in site names denoting upper (above-wall) grids, and L in site names denoting lower (below-wall) grids). Occasionally grid sizes were reduced to avoid areas of non-uniformity e.g., a clear change in vegetation. Within each sampling grid measurements began immediately adjacent to the walls (denoted as 0 m), and continued until the extent of the measurement grid (usually 15 m), thereby creating predominantly 16 m x 16 m sampling grids.

A soil moisture-probe (ML3 ‘Theta-probe’: Delta-T Devices Ltd) gave 256 readings (at $1m^2$ resolution) per standard upper and lower grid for each monitored dry-stone wall (512 total measurements per standard paired-grid). The soil moisture-probe consists of four 6 cm wave-guides arranged in a trefoil formation (attached to a probe-body) that were fully inserted into the soil surface. This moisture-probe measures θ_V (m^3/m^{-3}) using sTDR. Briefly, the moisture-probe emits a continuous 100 MHz outgoing wave and records the reflection of this wave to produce a composite standing wave. The outgoing and standing wave ratio is dependent upon the dielectric constant of the soil surrounding the wave-guides, which is largely controlled by θ_V (see Gaskin and Miller, 1996). Due to being both rapid and repeatable, sTDR was the selected experimental method (see Gaskin and Miller, 1996).

Following Whalley (1993), the moisture-probe reported in-field measurements in mV, which were converted to θ_V post-measurement via Equation 4.2, due to the experiment primarily involving mineral soils (Whalley, 1993). The moisture-probe is accurate to +/- 2 % θ_V, and averages θ_V over the full length of the wave-guides, primarily around the central wave-guide (Whalley, 1993; Gaskin and Miller, 1996). The same moisture-probe was used for all measurements to account for any unknown instrument bias. Gaskin and Miller (1996), Miller et al. (1997) and Wallace and Chappell (2020a) give detailed information regarding the design, operation, calibration and uncertainty of soil moisture-probe measurements.

It should be noted that θ_V measurements immediately surrounding the dry-stone wall were occasionally difficult to fully insert the probe due to the presence of both surface and subsurface stones. This difficulty therefore underlines that the measurements immediately surrounding the dry-stone wall (both above and below the feature) likely contain a higher margin of error than measurements elsewhere in the sampling-grids, and are potentially biased towards slightly drier conditions. This sampling bias should be relatively uniform between upper and lower grids however. It should additionally be noted that the first θ_V sample in each grid is taken as close to the wall as possible, although may not necessarily be precisely at the wall for the above reasons.

7.5.6 Statistical analysis

Topsoil volumetric wetness at each upper and lower grid was first assessed for normality via KS and AD tests. Due to near-universal violation of normality in θ_V

distributions, θ_V between upper and lower grids was contrasted via the non-parametric MWW tests for all sites. Following these statistical tests, upper and lower grids were further subdivided regarding their distance above and below the dry-stone wall, and this subdivided data was statistically contrasted via MWW tests due to small sample sizes, as well as to avoid normality assumptions. All statistical analysis was conducted within MATLAB (The Mathsworks, Inc) using the *kstest*, *adtest*, and *ranksum* functions, respectively, with significance levels of $p \leq 0.05$, $p \leq 0.01$, and $p \leq 0.001$.

7.6 Results and discussion

7.6.1 The influence of dry-stone walls on soil volumetric wetness over the full (16 m x 16 m) sampling grids

A total of 363 m of dry-stone walls were measured, giving 23-paired sampling grids (producing 11,490 total θ_V measurements over 11, 490 m^2). Of these paired-grids, twenty-one pairs were the standard 16 m x 16 m grids, one pair was a 15 m x 15 m grid, and one pair was a 12 m x 12 m grid. Soil volumetric wetness data over the full sampling grids (Table 7.1) show that θ_V fails to satisfy normality assumptions in all upper and lower sampling grids (no site satisfied both KS and AD test), and therefore justifies the adopted non-parametric approach (MWW tests). Non-normality within improved-pasture θ_V distributions has also been observed in nearby studies across a range of conditions (Wallace and Chappell, 2020a). Soil volumetric wetness sampling likely occurred during saturated and near-saturated conditions, with studies such as Wallace and Chappell (2020a) highlighting median pasture porosities of 58.2 %, and Wallace et al. (2021) highlighting arithmetic mean pasture porosities of 58.0 % – 59.1 %, both in nearby locations. Median and arithmetic mean θ_V of several sampling-grids in Table 7.1 are very similar to these values, highlighting saturated conditions, with

Table 7.1: The total number of samples (N) of soil volumetric wetness (θ_V) within each paired sampling-grid, as well as the median ($\tilde{x}$) and mean ($\bar{x}$) θ_V for individual upper and lower sampling-grid. Kolmogorov-Smirnov (KS) and Anderson-Darling (AD) normality tests are shown for each individual upper and lower sampling-grid, as well as the Mann-Whitney-Wilcoxon (MWW) statistical tests of central tendency, with a plain-language interpretation of these results. Notes on any non-uniformities are also included. Note that upper sites have the notation (U), and that lower sites have the notation (L).

Site	**N**	**Orientation**	**Angle (nearest 10°)**	**$\bar{x}$ ($\theta_{V\,\%}$)**	**$\tilde{x}$ ($\theta_{V\,\%}$)**	**KS**	**AD**	**MWW**	**Non-uniformity notes**
WF1U	450	N to S Downslope to the E	10°	56.3	56.4	0.001***	0.001***	0.001*** Wetter downslope	WF1U/WF1L: Some ingress of common rush (*Juncus effusus*) at periphery furthest from wall.
WF1L				56.2	57.2	0.001***	0.001***		
WF2U	288	N to S Downslope to the E	10°	57.7	57.7	0.001***	0.001***	0.005** Wetter downslope	WF2U/WF2L: Some ingress of common rush (*Juncus effusus*) at periphery furthest from wall.
WF2L				57.3	59.2	0.001***	0.001***		
WF5U	512	N to S Downslope to the E	10°	54.0	54.5	0.001***	0.001***	0.001*** Wetter upslope	WF5U/WF5L: Some form of blocked passage (possible disused creep-hole) at 12 m. Toddle gutter adjacent on left-hand side. WF5L: Tree adjacent on the right-hand side.
WF5L				48.9	49.9	0.001***	0.001***		
HH1U	512	W to E Downslope to the N	270°	58.3	59.6	0.001***	0.001***	0.001*** Wetter upslope	HH1U/HH1L: Tree to left of grid at wall. HH1U: Very minor ingress of common rush (*Juncus effusus*) at periphery furthest from wall.
HH1L				53.8	54.3	0.001***	0.001***		

Table 7.1 (continued):

Site	**N**	**Orientation**	**Angle (nearest 10°)**	**$\bar{x}$ ($\theta_{V\,\%}$)**	**$\tilde{x}$ ($\theta_{V\,\%}$)**	**KS**	**AD**	**MWW**	**Non-uniformity notes**
HH2U	512	WNW to ESE Downslope to NNE	300°	56.7	58.7	0.001***	0.001***	0.001*** Wetter upslope	HH2U/HH2L: Minor stream passes through wall at ~3 m. HH2L: Water post at wall at ~7 m in standing water.
HH2L				55.4	56.7	0.001***	0.001***		
HH3U	512	NW to SE Downslope to the NE	320°	55.5	59.3	0.001***	0.001***	0.434 No difference	HH3U/HH3L: Minor stream passes through wall at ~10 m HH3U: Tree present on right-hand at ~15 m along the wall, ~4 m upslope of wall. HH3L: Water post at ~7 m in standing water.
HH3L				57.2	58.7	0.001***	0.001***		
HH4U	512	NNW to SSE Downslope to the ENE	330°	59.2	60.9	0.001***	0.001***	0.331 No difference	HH4U: Small tree present on left hand side ~7 m along the wall, ~5 m upslope from the wall.
HH4L				59.3	60.7	0.001***	0.001***		
HH5U	512	NNW to SSE Downslope to the ENE	330°	50.8	51.0	0.001***	0.002**	0.001*** Wetter downslope	-
HH5L				52.5	53.6	0.001***	0.001***		

Table 7.1 (continued):

Site	N	Orientation	Angle (nearest 10°)	$\bar{x}$ ($\theta_{V\ \%}$)	$\tilde{x}$ ($\theta_{V\ \%}$)	KS	AD	MWW	Non-uniformity notes
HH7U	512	WNW to ESE	300°	54.8	55.5	0.001***	0.001***	0.001*** Wetter upslope	HH7U/HH7L: Tree to left of grid at wall.
HH7L		Downslope to NNE		52.0	53.1	0.001***	0.001***		
WW1U	512	NE To SW	50°	52.4	53.6	0.001***	0.001***	0.065 No difference	WW1U: Faint quad-tyre tracks immediately next to the wall.
WW1L		Downslope to the SE		53.4	54.1	0.001***	0.001***		
WW2U	512	NE To SW	50°	59.0	59.7	0.001***	0.001***	0.001*** Wetter upslope	-
WW2L		Downslope to the SE		50.2	50.7	0.001***	0.001***		
WW3U	512	ENE to WSW	60°	45.2	45.5	0.001***	0.178	0.003** Wetter upslope	-
WW3L		Downslope to the SSE		43.6	43.9	0.001***	0.011*		

Table 7.1 (continued):

Site	**N**	**Orientation**	**Angle (nearest 10°)**	**$\bar{x}$ ($\theta_{V\,\%}$)**	**$\tilde{x}$ ($\theta_{V\,\%}$)**	**KS**	**AD**	**MWW**	**Non-uniformity notes**
WW4U	512	ENE to WSW Downslope to the SSE	60°	44.6	44.8	0.001***	0.011*	0.464 No difference	-
WW4L				44.2	44.8	0.001***	0.006**		
WW5U	512	ENE to WSW Downslope to the SSE	70°	52.5	51.7	0.001***	0.001***	0.001*** Wetter upslope	-
WW5L				49.2	49.5	0.001***	0.048*		
WW6U	512	ENE to WSW Downslope to the SSE	70°	51.3	51.3	0.001***	0.329	0.001*** Wetter upslope	-
WW6L				48.9	49.4	0.001***	0.001***		
WW7U	512	ENE to WSW Downslope to the SSE	70°	49.4	49.3	0.001***	0.011*	0.070 No difference	-
WW7L				48.4	48.8	0.001***	0.001***		

Table 7.1 (continued):

Site	N	Orientation	Angle (nearest 10°)	$\bar{x}$ ($\theta_{V\,\%}$)	$\tilde{x}$ ($\theta_{V\,\%}$)	KS	AD	MWW	Non-uniformity notes
WW8U	512	ENE to WSW Downslope to the SSE	60°	49.1	49.3	0.001***	0.014*	0.475 No difference	-
WW8L				48.9	49.0	0.001***	0.389		
WW9U	512	N to S Downslope to the E	10°	50.3	50.9	0.001***	0.001***	0.002** Wetter upslope	-
WW9L				49.2	50.0	0.001***	0.001***		
WW10U	512	N to S Downslope to the E	10°	50.3	50.6	0.001***	0.044*	0.001*** Wetter upslope	-
WW10L				46.9	47.8	0.001***	0.001***		
WW11U	512	N to S Downslope to the E	10°	45.5	45.8	0.001***	0.008**	0.001*** Wetter downslope	-
WW11L				47.0	47.3	0.001***	0.001***		

Table 7.1 (continued):

Site	N	Orientation	Angle (nearest 10°)	$\bar{x}$ ($\theta_{V\ \%}$)	$\tilde{x}$ ($\theta_{V\ \%}$)	KS	AD	MWW	Non-uniformity notes
WW12U	512	N to S Downslope to the E	10°	45.6	46.1	0.001***	0.001***	0.001*** Wetter downslope	-
WW12L				49.1	49.2	0.001***	0.071		
WW13U	512	N to S Downslope to the E	10°	46.8	46.9	0.001***	0.073	0.001*** Wetter downslope	-
WW13L				49.1	49.4	0.001***	0.001***		
WW14U	512	N to S Downslope to the E	10°	47.4	47.6	0.001***	0.260	0.798 No difference	-
WW14L				47.1	47.5	0.001***	0.001***		

slightly drier sites highlighting near-saturated conditions. Over the full scale 23 sampling-grids, θ_V was significantly wetter downslope at 6 sites, significantly wetter upslope at 10 sites, and statistically similar at 7 sites (Table 7.1).

Table 7.1 highlights considerable spatial variability of θ_V within seemingly similar and adjacent improved-pastures, as has been observed and suggested elsewhere (Wallace and Chappell, 2019; 2020a; 2020b; Wallace et al., 2021), and reveals unclear and inconsistent patterns regarding the influence of the dry-stone wall over the full-grid scale (mostly 16 m above and below the boundary). Significantly wetter conditions upslope of the boundary may be highlighting that the dry-stone wall is impeding θ_V transport downslope, causing water accumulation upslope of the wall and a resultant increase in θ_V (Figure 7.4). Wetter conditions upslope of the dry-stone walls potentially suggests that such boundary features have flood-mitigation benefits, and supports Tayefi et al. (2007), Yu and Lane (2010), and Segura-Beltrán et al. (2016) regarding the wall being (semi-)impermeable and affecting flow routing. Conversely, significantly wetter conditions below the dry-stone wall may be suggesting that the boundary is permeable and is transferring water downslope rather than holding up the water (potentially amplifying flood-risk), and disagreeing with Tayefi et al. (2007), Yu and Lane (2010), and Segura-Beltrán et al. (2016). Sites with statistically similar conditions above and below the wall may suggest that the dry-stone wall is not having a significant impact on the transport of soil moisture.

Three (non-exclusive) reasons are given for the inconsistent results observed over the mostly 16 x 16 m full-sized grids, which resultantly have implications regarding the influence of the dry-stone wall. The first reason for such result variation is that inherent site dissimilarity in the improved-pastures above and below the boundary could be causing observed θ_V differences. The experimental design adopted in the study attempted to minimise site differences however as sites were specifically selected for their geological, pedological, vegetative and management uniformity (with sites removed that potentially violated this criteria – and any retained minor violation of uniformity clearly highlighted in Table 7.1). Furthermore, the considerable number of sites adopted with the intensive sampling regime provides evidence against this hypothesis, as site dissimilarity is unlikely to be the case in all sampling grid-pairs.

The second proposed reason for the observed result variation may be that only certain dry-stone walls are actively reducing slope hydrological connectivity and retaining water upslope (or conversely, only certain dry-stone walls are actively increasing slope hydrological connectivity and causing wetter conditions downslope). This explanation may be due to variations in the permeability of the foundations of each dry-stone wall and can be seen visually in Figures 7.5-7.10, possibly caused by variations in age or construction styles of each dry-stone wall, or possibly due to local site conditions such as soil or unmapped man-made drainage. This explanation suggests that some dry-stone walls may be retaining water on the upslope section, however, other dry-stone walls may be accelerating water transport downslope.

The third and final proposed reason for the inconsistency in results is that seemingly identical and adjacent improved-pastures are extremely spatially variable regarding θ_V (and other hydrological variables such as permeability), and are potentially influenced more heavily by factors other than the presence of the dry-stone wall at the monitored scale (12 m – 16 m distance from the boundary). It is possible however that the dry-stone wall does have a substantial influence upon θ_V, but perhaps at more localised scales surrounding the boundary.

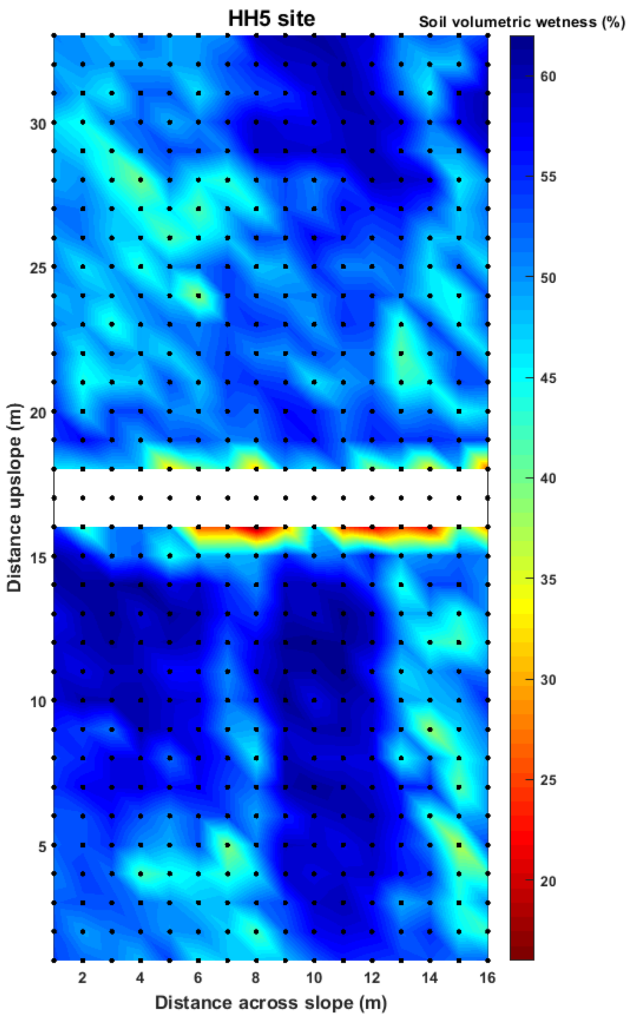

Figure 7.5: The soil volumetric wetness grid from HH5 (note that the bottom half of the grid indicates downslope, and the top half of the grid indicates upslope, with the dry-stone wall shown to separate the two sites). The soil volumetric wetness grid shows that the dry-stone wall may be effective in reducing soil volumetric wetness downslope in certain areas but not others, based on the soil volumetric wetness measurements immediately at the dry-stone wall. Also note that this site aligned relatively well with the dominant wind direction and may contain a rain-shadow effect. The HH5 site is significantly wetter downslope from 4 m – 15 m from the dry-stone wall (with no statistically significant results at smaller scales).

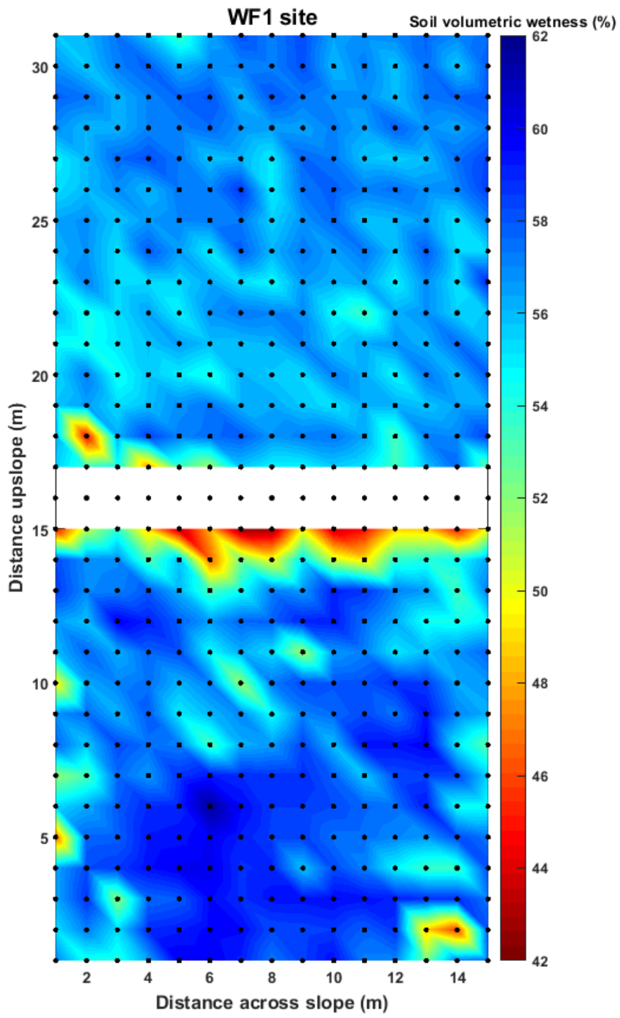

Figure 7.6: The soil volumetric wetness grid from WF1 (note that the bottom half of the grid indicates downslope, and the top half of the grid indicates upslope, with the dry-stone wall shown to separate the two sites). The soil volumetric wetness grid shows that the dry-stone wall may be effective at small scales as shown by significantly drier conditions downslope 0 m – 2 m from the dry-stone wall, which are also clearly visible. This site additionally observed significantly wetter conditions downslope 9 m – 14 m from the dry-stone wall (note that this grid was 15 m x 15 m, rather than the standard 16 m x 16 m, and therefore extended 14 m from the dry-stone wall).

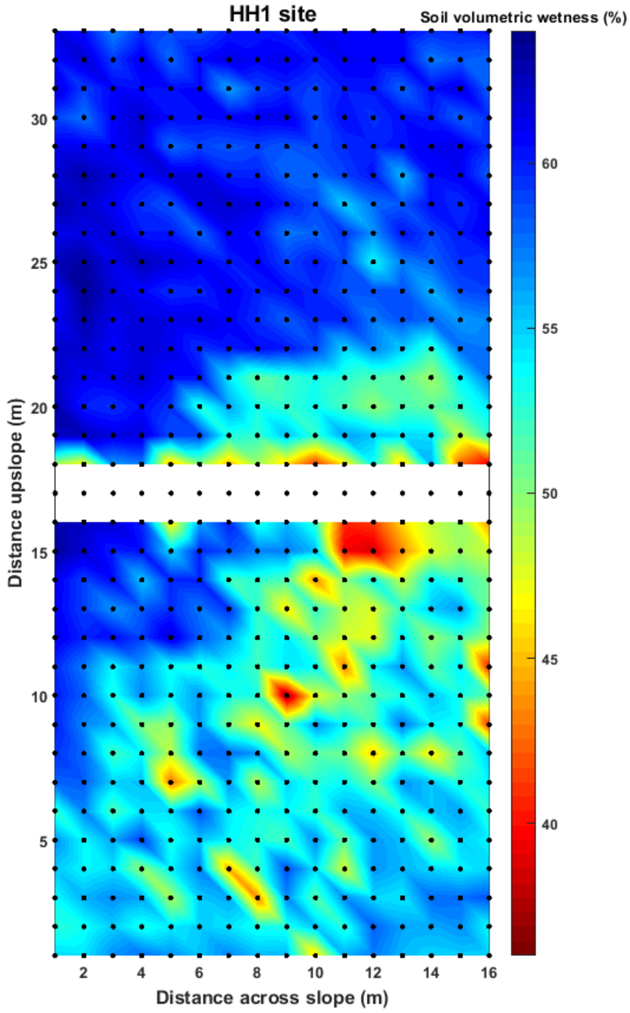

Figure 7.7: The soil volumetric wetness grid from HH1 (note that the bottom half of the grid indicates downslope, and the top half of the grid indicates upslope, with the dry-stone wall shown to separate the two sites). The grid shows very variable soil volumetric wetness, particularly on the downslope section of the dry-stone wall. It is also shown that the right-hand portion of the grid may be drier than the left-hand portion of the grid (particularly downslope of the dry-stone wall). The HH1 site is significantly wetter upslope from 7 m – 15 m from the dry-stone wall.

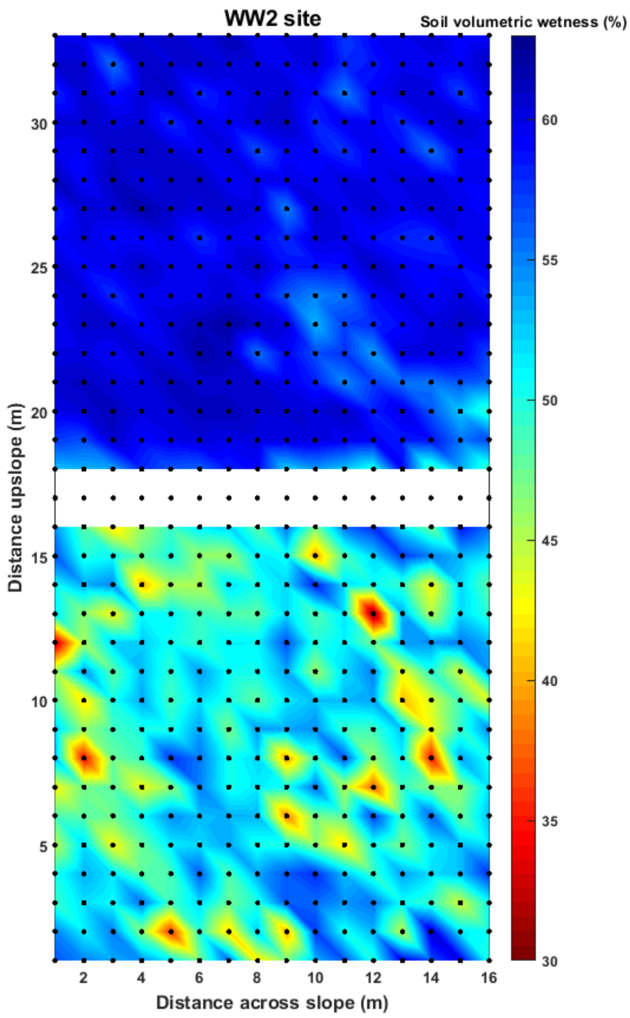

Figure 7.8: The soil volumetric wetness grid from WW2 (note that the bottom half of the grid indicates downslope, and the top half of the grid indicates upslope, with the dry-stone wall shown to separate the two sites). The soil volumetric wetness grid shows a marked difference above and below the dry-stone wall. Results showed WW2 to be significantly wetter upslope 1 m – 15 m from the dry-stone wall.

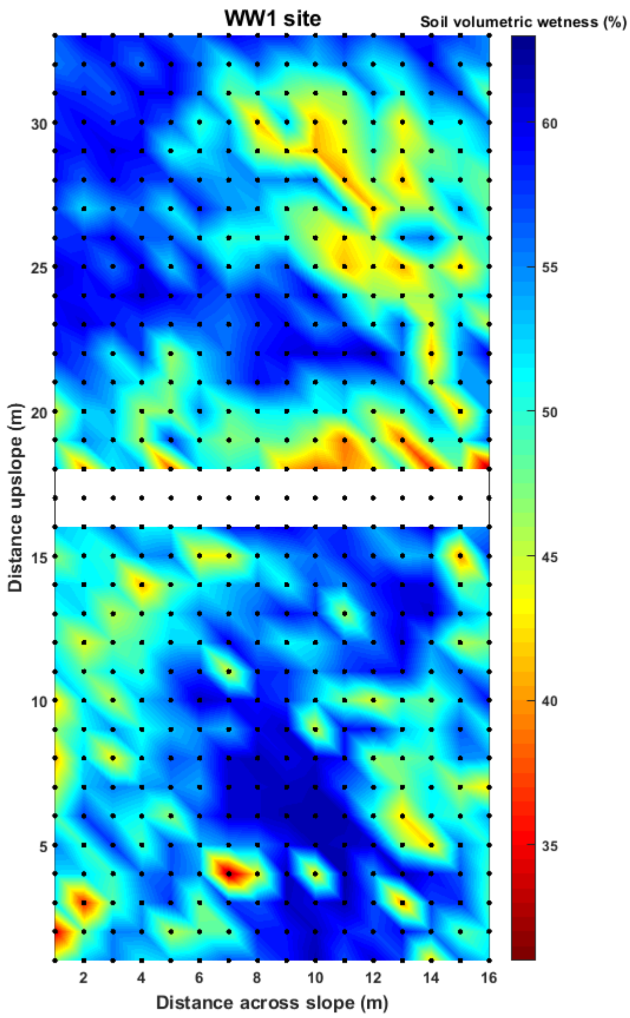

Figure 7.9: The soil volumetric wetness grid from WW1 (note that the bottom half of the grid indicates downslope, and the top half of the grid indicates upslope, with the dry-stone wall shown to separate the two sites). The soil volumetric wetness grid shows that the dry-stone wall may be causing wetter conditions downslope rather than upslope, and is perhaps accelerating water transport (possibly as a result of man-made drainage and/or high permeability of the dry-stone wall foundation, although both hypotheses are untested). Results showed WW1 to be significantly wetter downslope 0 m – 5 m, and at 9 m from the dry-stone wall.

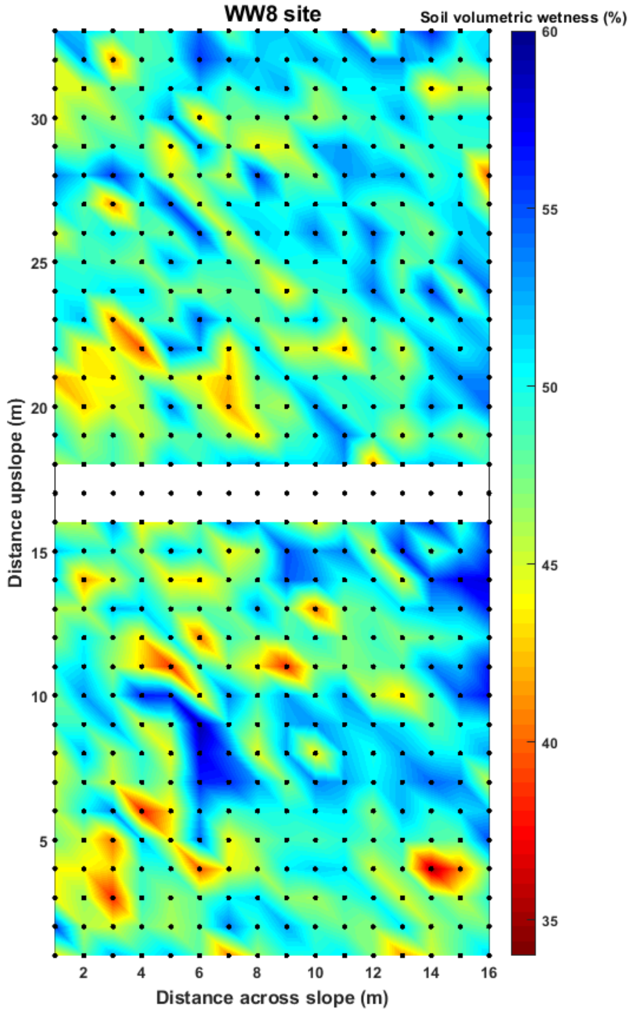

Figure 7.10: The soil volumetric wetness grid from WW8 (note that the bottom half of the grid indicates downslope, and the top half of the grid indicates upslope, with the dry-stone wall shown to separate the two sites). The soil volumetric wetness grid shows very high levels of spatial variation in soil volumetric wetness, and that there is no visible influence of the dry-stone wall. No statistically significant results were observed for WW8 over any distance from the dry-stone wall.

7.6.2 The influence of dry-stone walls on soil volumetric wetness at varying distances from the wall

Given the unclear patterns revealed using the full θ_V grids (Table 7.1), θ_V was analysed at varying distances to the wall in 1 m increments (the sampling resolution: see Table 7.2: Figures 7.5-7.10). At a 0 m scale (i.e., immediately adjacent to the dry-

Table 7.2: The Mann-Whitney-Wilcoxon (MWW) statistical significance tests of soil volumetric wetness at varying distance above and below the dry-stone wall boundary for each field-site, with the angle and downslope direction given. Stars represent the level of significance, highlighting wetter conditions. Note that upper sites have the notation (U), and that lower sites have the notation (L). The final column represents the ratio of insignificant results vs significant results in upper grids vs significant results in lower grids.

Site	**WF1**	**WF2**	**WF5**	**WW 9**	**WW 10**	**WW 11**	**WW 12**	**WW 13**	**WW 14**	**WW 1**	**WW 2**	**WW 3**	**WW 4**	**WW 8**	**WW 5**	**WW 6**	**WW 7**	**HH1**	**HH2**	**HH7**	**HH3**	**HH4**	**HH5**	**Insig vs Usig vs Lsig results**
Angle	**10°**	**10°**	**10°**	**10°**	**10°**	**10°**	**10°**	**10°**	**10°**	**50°**	**50°**	**60°**	**60°**	**60°**	**70°**	**70°**	**70°**	**270°**	**300°**	**300°**	**320°**	**330°**	**330°**	
Downslope direction	**E**	**E**	**E**	**E**	**E**	**E**	**E**	**E**	**E**	**SE**	**SE**	**SSE**	**SSE**	**SSE**	**SSE**	**SSE**	**SSE**	**N**	**NNE**	**NNE**	**NE**	**ENE**	**ENE**	
Distance																								
15mU	NA	NA	***	**	***						***	**			***	***		***	***	***				7:10:4
14mU		NA	***	***	***						***	**			***	***		***		***				7:9:6
13mU		NA	***	***	***						***	**			***	***		***	*	***				6:10:6
12mU		NA	***	***	***						***	***			***	***		***		***				7:9:6
11mU			***	***	***						***	***			**	***		***		***				7:9:7
10mU			***	***	***						***	***				***		***		***				7:8:8
9mU			***	***	***						***	**				**		***		***				9:8:6
8mU			***	***	***						***	**				*		***		***				11:8:4
7mU			***	***	***						***	**						***		***				12:7:4
6mU		*	***	***	***						***	**								***				13:7:3
5mU		***	***	***	***						***	*								***				12:7:4
4mU		***	***	***	***						***									***				13:6:4
3mU		***	*	***	***						***					*				***				14:7:2
2mU	***	***	*	***	***						***					*				*		*		12:9:2
1mU	***	***		***	***						***	*	*									**		14:8:1
0mU	***	***		**	***				*													*		16:6:1
Wall	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
0mL										***														
1mL										***														
2mL							**			***														

Table 7.2 (continued):

Site Angle Downslope direction	WF1 10° E	WF2 10° E	WF5 10° E	WW 9 10° E	WW 10 10° E	WW 11 10° E	WW 12 10° E	WW 13 10° E	WW 14 10° E	WW 1 50° SE	WW 2 50° SE	WW 3 60° SSE	WW 4 60° SSE	WW 8 60° SSE	WW 5 70° SSE	WW 6 70° SSE	WW 7 70° SSE	HH1 270° N	HH2 300° NNE	HH7 300° NNE	HH3 320° NE	HH4 330° ENE	HH5 330° ENE	Insig vs Usig vs Lsig results
Distance																								
3mL							***			***														
4mL							***	*		**													**	
5mL							***	*		*													***	
6mL							***	**															***	
7mL							***	**														*	***	
8mL							***	***														*	***	
9mL	*						***	***		*												*	***	
10mL	**	*				*	***	***														*	***	
11mL	**	**				**	***	***															***	
12mL	***	NA				***	***	***															***	
13mL	***	NA				***	***	***															***	
14mL	***	NA				***	***	***															***	
15mL	NA	NA				***	***	***															***	

stone wall), 16 of the 23 sites showed no significant difference between upper and lower plots, 6 of the 23 sites were significantly wetter upslope than downslope, and 1 of the 23 sites was significantly wetter downslope. These findings highlight that in the vast majority of instances there was no significant difference in θ_V above or below the dry-stone wall at the 0 m scale, potentially suggesting that dry-stone walls are not having a statistically significant effect on θ_V even at the smallest sampling resolution (0 m) scale – presumably when the dry-stone wall is having the largest possible influence. The number of significant wetter upslope plots (6) in comparison to significantly wetter downslope plots (1), may however, highlight that in certain circumstances, dry-stone walls are potentially having an effect on θ_V, with a considerable ratio of significantly wetter upslope plots to significantly wetter downslope plots of 6:1.

At a 1 m scale (i.e., combining 0 m and 1 m measurements), 14 sites showed no significant difference between upslope and downslope plots, 8 sites were significantly wetter upslope compared to downslope, and 1 site was significantly wetter downslope. Similarly to above, these results highlight that in the majority of instances the dry-stone walls are not having a statistically significant effect on θ_V at a 1 m scale; although the ratio of statistically significant wetter upslope plots in comparison to statistically significant wetter downslope plots is 8:1. This is the largest ratio of statistically significant results between upslope and downslope plots for a given distance from the dry-stone wall, potentially suggesting that the dry-stone walls may be having some effect of causing upslope plots to be wetter than downslope plots at this very small scale in a considerable proportion of scenarios, although still in a minority of scenarios.

At the 2 m scale, 12 sites showed no significant difference between upslope and downslope plots, 9 sites were significantly wetter upslope compared to downslope, and 2 sites were significantly wetter downslope. As above, these findings highlight that in the majority of cases there is no statistically significant difference above and below the boundary, although the ratio of statistically wetter upslope plots in comparison to downslope plots is still notable at 4.5:1, and also covers a considerable number of minority cases.

The trend of the majority of sites revealing insignificant θ_V results continued up until 10 m from the boundary, although the ratio between statistically insignificant sites, sites significantly wetter upslope, and sites significantly wetter downslope, became much more even at $\geq$ 9 m. The continued dominance of insignificant θ_V differences above and below the dry-stone wall, combined with relatively even ratios of insignificant results, significantly wetter upslope grids, and significantly wetter downslope grids at $\geq$ 9 m scales, suggests that the wall is not having a substantial effect on θ_V in the majority of cases, and that θ_V is perhaps more strongly determined by other factors rather than the dry-stone wall across the range of monitored scales. This finding is further highlighted as there are slightly fewer statistically significant results (either upslope or downslope being wetter) at the smaller scales, presumably when the dry-stone wall is having the largest effect, in comparison to the number of statistically significant results at larger scales, further suggesting that the dry-stone wall is not having a significant effect on θ_V.

The ratio of significantly wetter upslope plots in comparison to downslope plots was very high at short distances, being 6:1 at 0 m, 8:1 at 1 m, 4.5:1 at 2 m, and 3.5:1 at 3 m. These results suggest that although in the majority of instances there is no statistically significant difference between upslope and downslope plots, upslope plots are considerably more likely to be significantly wetter than downslope plots. At slightly increased distances, the ratio of significantly wetter upslope plots to significantly wetter downslope plots decreased to 1.5:1 at 4 m, 1.75:1 at 5 m, 2.33:1 at 6 m, 1.75:1 at 7 m, 2:1 at 8 m, and 1.5:1 at 9 m. Results similarly highlights that upslope plots are more likely to be significantly wetter than downslope plots at these scales, although this is less likely than at ≤ 3 m scales. At distances beyond 9 m the ratio of significantly wetter upslope plots in comparison to downslope plots fluctuates between 1:1 and 1:67, although at the 15 m extent (i.e., the full sampling 16 m x 16 m grid-scale) this increases to 2.5:1. These ratios therefore conclude that upslope plots are more likely to be wetter than downslope plots, particularly at small (≤ 3 m) distances, although in the majority of instances the difference between upslope and downslope plots is statistically insignificant, especially at scales ≤ 8 m.

7.6.3 The influence of orientation of the dry-stone walls on soil volumetric wetness

The angle and orientation of each dry-stone wall site is given in Tables 7.2-7.3. Over the full-grid scale (Tables 7.1–7.3), orientation and angle may be playing some form of role in determining if dry-stone walls are significantly wetter downslope, as all six significantly wetter downslope sites appeared to have a similar orientation of between 330° and 10°, resulting in having the downslope to the East-North East to East. This orientation is against the predominant wind-direction in the United Kingdom of south-west, and against the locally dominant wind direction from Penrith of west (Weather Spark, 2021), and is perhaps slightly counterintuitive given that downslope grids may be receiving some rain-sheltering effects from the dry-stone wall, and therefore should potentially be drier in comparison to the unsheltered upslope locations. The sampling-grids with a 10° orientation (downslope to the East) also capture numerous instances

Table 7.3: The angle and orientation of each dry-stone wall, with the total number of dry-stone walls, the number of significantly wetter upslope and significantly wetter downslope sites, alongside sites that were statistically similar.

Angle	Orientation	No. of dry-stone walls	Significantly wetter downslope	Insignificant difference	Significantly wetter upslope
10°	Downslope to the E	9	5	1	3
50°	Downslope to the SE	2		1	1
60°	Downslope to the SSE	3		2	1
70°	Downslope to the SSE	3		1	2
270°	Downslope to the N	1			1
300°	Downslope to NNE	2			2
320°	Downslope to the NE	1		1	
330°	Downslope to the ENE	2	1	1	

of significantly wetter upslope conditions however, directly contradicting this trend and suggesting that this relationship is inconsistent. The lack of a clear trend at the full grid-scale is likely due to the fact that rain-shadowing occurs on a much smaller scale than the full-size sampling-grids.

Assuming that the dry-stone walls are ~ 1.5 m on average, the rain-shadow distance of the dry-stone wall is likely two times the height, so ~ 3 m (i.e., up to the 3 m measurements). At the 0 m scale (Table 7.2), all dry-stone walls that are significantly wetter upslope are those with the downslope to the ENE-E, implying that rain-shadowing effects may be present on this small scale, causing statistically significant differences in 6 of 11 possible sites with such an orientation. The single site at the 0 m scale to have significantly wetter conditions downslope was orientated with the downslope to the SE. Several additional dry-stone walls that relatively oppose the predominant wind direction i.e., those with a downslope to NNE to NE, showed insignificant differences above or below the dry-stone wall at this 0 m scale.

At the 1 m scale, 5 of the 11 sites with the downslope to the ENE-E are significantly wetter upslope, with 3 additional sites also significantly wetter upslope, with these three latter sites all having a downslope to the SE-SSE. The single site at the 1 m scale to have significantly wetter conditions downslope was orientated with the downslope to the SE. The 1 m results also suggest a possible rain-shadow effect of the dry-stone wall. As with the 0 m results, the three sites with the downslope to the NNE-NE revealed insignificant results.

At the 2 m scale, 6 of the 11 sites with the downslope to the ENE-E are significantly wetter upslope, with the 3 remaining sites with significantly wetter conditions upslope being orientated with downslopes to the SE, to the SSE, and to the NNE. At the 2 m scale, 1 site orientated with the downslope to the E had significantly wetter conditions downslope (WW12), and 1 site orientated with the downslope to the SE also had significantly wetter conditions downslope. This reinforces the 0 m and 1 m observations of a rain-shadow effect of the dry-stone wall, although this is less consistent given that various orientations not directly opposed to the dominant wind direction are drier downslope now, and there are now also instances of wetter conditions downslope when this is sheltered from the dominant wind direction. Results at the 3 m scale, likely at the limit of the rain-shadowing effect, revealed almost identical conclusions to the 2 m results, with the only difference being 4 (rather than 6) of 11 sites with the downslope to the ENE-E being significantly wetter upslope.

Results suggest that at the 0 m scale, the 1 m scale, and to a lesser extent the 2 m scale and 3 m scale, the dry-stone wall orientation may be playing some role in determining if upslope or downslope conditions are significantly wetter. The numerous instances of significantly drier conditions downslope in comparison to upslope when dry-stone walls were approximately opposing the predominant wind-direction, suggests that a rain-shadowing effect may be being provided by dry-stone walls. Interestingly, dry-stone walls that provide a possible rain-shadowing effect from the prominent wind direction at the 0 m – 2 m scales and result in statistically significantly drier conditions downslope can also be significantly wetter downslope at larger-scales e.g., WF1, WF2 and HH4, perhaps underlining the influence of the rain-shadow effect in comparison to other hydrological processes operating at larger (up to 15 m from the dry-stone wall over the 16 m x 16 m sampling grids) scales.

7.7 Implications and conclusions

Dry-stone walls are widespread agricultural boundary features in many parts of the world, often used in pastoral systems as enclosures and to highlight property boundaries. In many agricultural regions these are the dominant linear boundary feature. Despite such a significant global presence, very few studies have investigated their hydrological influence, with no field-based observations of how such features influence hydrological processes (to the knowledge of the authors). The aim of this study was to assess if dry-stone walls slow the transport of near-surface water downslope within improved-pastures, as well as their broader influence upon soil volumetric wetness, and therefore to assess their potential hydrological influence in relation to overland flow and rapid shallow-subsurface flow generation caused by excessive saturation. This aim was achieved via an intensive sampling program of soil volumetric wetness above and below multiple dry-stone walls and subsequent statistical analyses.

Results highlight that improved-pastures have very high spatial-variability in soil volumetric wetness distributions and these distributions are likely non-normally distributed during saturated and near-saturated conditions. Over the full grid-scales (12 m x 12 m – 16 m x 16 m), the effect of the dry-stone wall on soil volumetric wetness was very inconsistent, suggesting that the boundary does not have a considerable (or consistent) effect at such scales. At smaller scales, soil volumetric wetness differences were predominantly insignificant above and below the dry-stone walls, although particularly at the 0 m – 3 m scale, upslope of the dry-stone walls were shown to be significantly wetter than downslope of the dry-stone walls in a considerable minority of scenarios.

The orientation of the dry-stone wall appeared to be playing some role in causing significant soil volumetric wetness differences, with multiple sites approximately orientated with the downslope sheltered from the prevailing wind-direction showing significantly drier conditions downslope at the 0 m and 1 m scales (and to a lesser extent, the 2 m scale and 3 m scale). This finding implies a possible rain-shadow effects of the dry-stone wall, although this rain-shadow was not detectable over the full-grid scales. Findings conclude that dry-stone walls may have very localised impacts on soil volumetric wetness distributions on sloped improved-pastures, although results are highly spatially variable as well as often insignificant in the majority of scenarios.

To the knowledge of the authors, this research is the first study globally to investigate the hydrological functioning of dry-stone walls at the plot-field scale; resultantly, there are abundant opportunities for future hydrological research regarding this agricultural and landscape feature. This new research could include but is not limited to studies relating to deeper subsurface hydrological flow pathways, the effect of dry-stone walls at larger scales through rainfall-runoff modelling, the effect of slope or solar radiation receipt on the functioning of dry-stone walls, and potential drought-resilience benefits of dry-stone walls. A comparison of dry-stone walls with other stone features such as (dry-)stone terraces or stone hedgerows is also justifiable, as is a comparison with biotic hedgerows. Future investigations could additionally expand into disciplines outside of hydrology, most notably into pedology e.g., the possible soil-conservation benefits of dry-stone walls, given the interest in this field in relation to (dry-)stone terraces.

Future researchers conducting similar experiments are strongly advised to consider the potentially high spatial-variability of soil volumetric wetness within improved-pastures in their experimental designs. Possibly repeating this experimental design on improved-pastures in the absence of a dry-stone wall may be useful to investigate the inherent level of spatial-variability at each field site. Repeat measurements at a single site may also be very useful to investigate the stationarity of the changes caused by the dry-stone wall, and could include drier conditions, as well as possibly pre- and post-disturbance e.g., following rainfall events or slurry application.

7.8 Conflict of interest statement

The authors declare no conflicts of interest.

7.9 Acknowledgements

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7.10 Data availability

The data that supports the findings of this study are available from the corresponding author upon reasonable request.

8 SYNTHESIS AND CONCLUSIONS

8.1 Introduction to synthesis and conclusions

This thesis has successfully quantified the role of four widespread grassland interventions present in pastoral farming systems in the Leith, Lowther and Petteril sub-catchments in relation to how they alter soil hydraulic properties and aspects of overland flow and rapid shallow-subsurface flow generation. This research is expected to have consequences for flood-risk, water-quality, and drought-resilience in the region as well (inter-)nationally, and therefore can be used to inform land-use decisions to control such hazards. Each intervention within this thesis was specifically selected as a measure that was either already present or is adoptable within the majority of grassland farming systems across much of the UK, Europe, and further afield (Figure 1.1). Each intervention additionally does not inhibit farming productivity, and likely provides ecosystem services beyond purely hydrological, therefore further justifying agri-environmental payments for farmers/landowners adopting such features and practices. This chapter briefly: discusses the key findings of each paper (Section 8.2), hydrologically compares interventions (Section 8.3), discusses the cross-cutting themes between chapters (Section 8.4), assesses widespread application of the interventions (Section 8.5), assesses the impact of interventions beyond hydrology (Section 8.6), reviews the thesis in both hindsight and with additional resources (Section 8.7), highlights future research questions raised by the thesis (Section 8.8), and concludes with some closing remarks (Section 8.9).

8.2 Key findings

Chapters 4-7 revealed very important and novel findings regarding how grassland interventions modify hydrology, with specific focus upon OFRSSF, and includes several advances in knowledge such as: a) an improved understanding of hydrological processes (particularly surface hydrodynamics) operating in grassland landscapes; b) examples of robust and widely applicable experimental designs (both field and analytical); c) advice for future research in regards to experimental designs and hydrometric measurement techniques; and d) multiple options for future hydrological research and hydrological modelling, as well as non-hydrological scientific studies.

8.2.1 Key findings of Chapter 4 (semi-natural grasslands)

Chapter 4 highlights that in the absence of intensive agricultural practices, semi-natural grasslands still suffer from hydrological hazards, including drought, as well as overland flow, and potentially the associated flood-risk and water-quality issues. This key finding underlines a very important (and sometimes overlooked) conclusion that hydrological hazards impact (semi-)natural ecosystems with fairly minimal human modification, and that interventions and Natural Flood-Risk Management may reduce, but are unlikely to entirely remove such hazards. This emphasises that grassland agricultural interventions (and Natural Flood-Risk Management interventions) are likely to require widespread adoption to be effective, and that pairing interventions with hard-engineered flood defences (where possible) could be a feasible approach to mitigating against hydrological hazards.

Chapter 4 additionally highlights that converting semi-natural grasslands to improved-pasture/silage fields can substantially alter the soil moisture regime. This change to the soil moisture regime includes substantially reducing the natural variability in soil moisture, which was evidenced by geostatistical models and significant differences in variation throughout, potentially having significant impacts upon local ecology, hydrological connectivity and hydrological modelling. The considerable micro-scale variability (nugget variance) highlighted by the geostatistics was another considerable finding and underlines the importance of including scale in measurements and the understanding of hydrological processes. The semi-natural grassland was also found to saturate to a statistically similar extent to the neighbouring improved-pasture during a storm (although the semi-natural grassland remained statistically more variable). Furthermore, Chapter 4 outlines that the sampling date and the localised type of vegetation significantly affects the soil moisture regime, and that the sampling date specifically influences how the land-use and vegetation influences the soil moisture regime.

The final key finding of Chapter 4 is that current agricultural practices taking place in improved-pastures, some of which may have assumed negative environmental connotations, may also be capable of reducing the (localised) risks of certain hydrological hazards even below the levels occurring in semi-natural systems. This finding was evidenced by the drought resilience benefits provided to the improved-pasture by certain agricultural practices/ features (possibly the application of livestock slurry), whilst the semi-natural grassland dried to such a degree that would likely have inhibited sward production had the improved-pasture dried to this extent. This finding emphasises three very important points: 1) The need to think holistically in hydrology (and in all the environmental, and indeed other, sciences), as something which may potentially increase the risk of one hydrological hazard may also decrease another (e.g., the role of slurry for water-quality and drought-resilience, respectively); 2) to avoid assumptions that certain agricultural practices are wholly-negative upon environmental systems without sufficient evidence, which leads to the most important conclusion; 3) the need for a substantial body of further hydrological investigations into ongoing agricultural practices occurring within grassland landscapes, both in the UK and overseas.

8.2.2 Key findings of Chapter 5 (blade aeration)

In a similar sense to Chapter 4, Chapter 5 demonstrates that hydrological pathways associated with hydrological hazards, in this case infiltration-excess overland flow, is likely to still be present in both intervention and baseline (non-intervention) conditions, if somewhat reduced. This finding further emphasises that the interventions studied in this thesis are unlikely to entirely remove such hydrological pathways and nullify hazards, but may have some mitigation properties. Again, this conclusion underlines that widespread adoption of interventions is likely necessary for them to noticeably reduce hydrological hazards at larger scales.

Chapter 5 highlights that blade-aerating improved-pastures could significantly increase the saturated hydraulic conductivity and significantly reduce soil penetration resistance, as well as substantially reduce the incidence of infiltration-excess overland flow. Improvements to these hydrological properties was observed at only one of the two monitored improved-pastures however. Several further studies (both hydrological and in other sciences) are needed to help understand the hydrological processes that determine why blade aeration was only effective in one improved-pasture and not within the other.

Another key finding of Chapter 5 is in relation to hydrometric measurement techniques. The *in-situ* permeametry method adopted throughout Chapter 5 (and Chapter 6) is specifically to account for underlying soil properties that may be the determining factor on permeability, in order to understand the dominant factor causing the observed permeability change. In agricultural environments, compaction can exist well-below the surface (e.g., plough pans caused by heavy machinery, such as those commonly found in intensive arable systems), sometimes in the absence of surface compaction. *Ex-situ* permeametry methods are a common field-measurement technique for ring permeametry, however, the extraction of the core from the ground could potentially be removing the effect of this naturally limiting factor on permeability, artificially inflating permeability measurements and influencing subsequent analysis and modelling. This observation underlines the importance of thinking about the interconnection of hydrological processes operating in the area of interest, and in how to accurately quantify them.

The final key finding of Chapter 5 relates to the experimental designs of hydrological investigations. Chapter 5 adopted a paired-plot approach (as did all Chapters 4-7), and successfully quantified the difference between aerated and non-aerated improved-pasture. However, this approach should ideally be combined with a before-after-control-intervention experimental design to discount natural soil variability and other factors causing observed differences, improving overall interpretations and confidence in the results. A solely before-after-control-impact approach would have been unsuccessful given the non-stationarity in permeability however. Combining before-after-control-impact and paired-plot approaches is therefore likely to be the optimum design for the majority of future hydrological field-investigations. This combined approach however often adds to the time and expense of sampling programs, and given the time-taken for the establishment of certain interventions, and the limited resources of certain projects, this combined approach may only be suitable for more substantial projects.

8.2.3 Key findings of Chapter 6 (hedgerow wild-margins)

Building upon Chapters 4-5, Chapter 6 highlights that establishing interventions does indeed not entirely remove rapid hydrological pathways such as OFRSSF. Overland flow and rapid shallow-subsurface flow was observed from both intervention and non-intervention land-uses during natural-rainfall conditions (during/surrounding storm conditions), and transfer function models of the artificial-rainfall experiment highlight the possible presence of rapid subsurface flow (possible macropore-flow) in one of the two hedgerows. These findings therefore highlight that interventions do not remove such hydrological flow pathways, but can potentially reduce their occurrence and the risks that they pose. As before, this highlights that interventions likely require adoption at large-scales to have considerable hydrological benefit.

Chapter 6 additionally highlighted that hedgerow-margins, when installed in a pastoral landscape, can significantly increase the permeability of the topsoil. Hedge-margins were also found to store more potential contaminants of hydrological importance on the soil surface in comparisons to improved-pastures (particularly nitrogenous compounds and loose sediment), although the true hydrological mobilisation of such contaminants (i.e., under natural rainfall conditions) remains unquantified. As before, Chapter 6 highlights that a substantial number of further studies are needed to further the understanding of hydrological processes occurring in agricultural grasslands.

8.2.4 Key findings of Chapter 7 (dry-stone walls)

Chapter 7 highlights that dry-stone walls do not have a consistent effect on soil volumetric wetness over the (mostly) 16 m x 16 m grid scales. At smaller scales, upslope of dry-stone walls was shown to be significantly drier than downslope in a notable minority of scenarios, and was possibly attributed to a localised rain shadow effect. The very localised, inconsistent and often statistically insignificant effect of the dry-stone walls suggest that soil volumetric wetness is possibly more heavily influenced by factors other than the dry-stone walls at the monitored scales.

Chapter 7 additionally monitored and analysed soil volumetric wetness over considerably large grids (mostly 16 m x 16 m) over a considerable number (46) of improved-pastures sites. All 46 grids highlight that soil volumetric wetness within improved-pastures (around dry-stone-wall boundaries) is significantly non-normally-distributed during saturated and near-saturated conditions through Kolmogorov-Smirnov tests, and 42 of the 46 grids highlight significant non-normality using Anderson-Darling tests. This finding is also replicated with the improved-pasture site within the soil volumetric wetness measurements in Chapter 4. This is an important discovery and can be used to help inform future hydrological modelling studies involving improved-pastures, as well as the experimental design of future hydrological field-studies and subsequent statistical analyses.

8.2.5 Additional key findings

The thesis additionally highlights the importance of detailed and accurate field measurements in hydrological investigations, as well as the importance and usefulness of subsequent detailed statistical analyses. This approach was taken entirely within a Data-Based Mechanistic framework, and both helps to investigate processes and properties of the complex hydrological system, as well as credibly interpret them with sound scientific and mathematical underpinnings. Adopting a purely dynamic modelling approach with assumed magnitudes of parameters and processes would likely miss such advancements, and may lack credibility, reliability or applicability, particularly in the understudied area of pasture hydrology.

8.2.6 Plain language summary of major key findings

1) Care needs to be taken with hydrometric measurements (especially within agricultural areas) due to the risk of measurement errors and biases, as well as due to the risk of damage to the scientific equipment.

2) Hydrological research should strongly consider combining before-after-control-intervention (BACI) and paired-plot experimental designs when operationally feasible, as this may improve interpretations by more clearly highlighting the impact of natural variability upon measurements, as well as being more resilient against non-stationarity (a stochastic process is stationary if its probability distribution does not significantly change over time).

3) Agricultural features and practices (interventions) in grassland landscapes, by enlarge, do indeed alter the hydraulic functioning of improved-pastures in relation to rapid surface and near-surface hydrological pathways. These hydrological pathways include overland flow i.e., water that flows (rapidly) over the ground surface, and rapid shallow-subsurface flow i.e., water that flows quickly below (but in close proximity to) the soil surface. These hydrological pathways may be strongly linked to flood-risk due to (possibly) rapidly transporting precipitation to stream networks, as well as linked to water-quality degradation due to agricultural contaminants often being located either on or near the soil surface, and therefore may be transported by such pathways. It is additionally possible that hydrological parameters that control such pathways, may additionally infer drought-resilience e.g., soil volumetric water content. These findings therefore highlights that certain agricultural interventions in grassland landscapes may influence flood-risk, water-quality and drought resilience.

4) Agricultural interventions in grassland landscapes are unlikely to completely prevent overland flow and rapid shallow-subsurface flow pathways. They can however, possibly offer some mitigation against them, thereby potentially reducing flood-risk and water-quality issues (and potentially drought-risk). This finding underlines that agricultural interventions are likely needed to be widespread to have considerable hydrological benefits at larger scales. It is possible that agricultural interventions may need to be paired with other approaches such as hard-engineered flood-defences or restrictions upon certain agricultural practices to be most effective.
5) Agricultural interventions in pastoral systems that are often associated with environmental disbenefits (e.g., possible water-quality degradation from slurry), may also have environmental benefits that may need to be acknowledged (e.g., possible drought-resilience benefits of slurry). This includes (dis)benefits both within and beyond hydrology e.g., ecology, sedimentology, atmospheric science, culture etc.
6) There is a substantial body of further work required to further the understanding of the hydrological processes taking place in grasslands (both improved-pastures, but also alternative grasslands). This evidently is true on a national but also a global scale, with certain grassland interventions receiving minimal hydrological study, especially in relation to surface-hydrology.

8.3 Hydrologically comparing interventions

Each research chapter of the thesis has been a comparison of an intervention (occasionally this intervention being the initial semi-natural grassland conditions) against ‘typically-managed’ neighbouring grassland (improved-pasture) in a pastoral, agricultural setting. This approach has been useful in that it compares the intervention against a highly localised baseline, and therefore allows the quantification of hydrological changes with reasonable confidence given natural soil variability (although see Chapters 5 and 7). An issue with this approach however, is that all improved-pastures are considerably different, and the hydraulic (and other) properties of a single improved-pasture can vary both spatially and temporally (see Chapters 4-7), often significantly. This approach therefore creates a relatively large degree of uncertainty when using results collected in one area and applying them to a separate location, even within the same soil type, (surficial-)geology, vegetation, climate and land-use (sites were intentionally selected for such similarity to improve comparisons). This spatio-temporal variability can make comparing interventions considerably difficult if they are not immediately adjacent, and substantially increases uncertainty when modelling the landscape hydrologically. As a result, interventions cannot be credibly compared with good degrees of accuracy or high levels of certainty without substantial further research (both experimental and modelling).

A potential way of addressing this issue in future research is comparing interventions that are immediately adjacent, e.g., aerating pasture immediately next to a hedgerow or dry-stone wall. Providing that a baseline is retained (i.e., improved-pasture that does not contain an intervention), this approach can credibly compare interventions, particularly if this is incorporated within a combined paired-plot and before-after-control-intervention experimental design. If a comparison is needed, it will also be important to monitor the same hydrological parameter for each intervention (e.g., soil volumetric wetness), an approach that was not given priority within this research.

It was found that both aeration and hedgerow wild-margins significantly increased the permeability of improved-pastures. Hedgerow wild-margins were found to significantly, substantially and consistently increase permeability, whereas the results from the blade aeration were inconsistent, and showed smaller absolute and relative magnitudes of improvement. These findings suggests that hedgerow wild-margins are perhaps the better intervention in situations where improving the permeability is required, although the application of each intervention is substantially different i.e., hedgerows require years to be established and cannot be applied uniformly across an improved-pasture, whereas blade aeration can be applied both rapidly, uniformly and locally. The impact that semi-natural grasslands and dry-stone walls have on permeability was not recorded and is a necessary future research question for an accurate comparison of interventions.

Extensive soil volumetric wetness measurements were included for the semi-natural grassland, as well as the dry-stone wall interventions. Both are extremely different interventions and thus are difficult to compare, with both showing flood-risk reduction benefits, although semi-natural grasslands were shown to influence a much larger spatial area than the extremely localised (and largely insignificant) influence of the dry-stone wall. Slurry, an intervention not directly measured in the thesis, was also shown to potentially benefit soil volumetric wetness during drought within the improved-pasture of Chapter 4, although slurry may also have amplified saturation and flood-risk with the onset of autumnal rains. Semi-natural grasslands were additionally shown to better maintain the natural diversity in soil volumetric wetness patterns compared to slurry-wetted pasture (although other management practices were also likely present). The effect that hedgerow wild-margins and blade aeration have on soil volumetric wetness is another area for future research.

Overland flow (and rapid shallow-subsurface flow) volume and water-quality was only directly recorded within the hedgerow wild-margins chapter (Chapter 6). As a result, it is difficult to credibly compare overland flow between interventions. The very high permeability observed from the hedgerow wild-margin intervention compared to the blade aeration intervention (compare Table 5.5 and Table 6.3), given relatively comparable baseline conditions (see Chapters 5-6), suggests that hedgerow wild-margins could perhaps further remove the simulated infiltration-excess overland flow pathway (see Section 5.6.4). Given the non-stationarity in the baseline permeability observed in the blade aeration chapter however (Table 5.6), alongside the lack of direct overland flow observations, reduces the confidence in this conclusion. It is likely that direct overland flow measurements of both volume and water-quality are likely needed from all interventions for a reliable comparison.

Hedgerow wild-margins documented hydrological benefits to flood-risk and possibly to water-quality. Semi-natural grasslands had potential hydrological benefits to flood-risk i.e., being slower to saturate than the slurry-wetted improved-pastures, but also contain hydrological drawbacks i.e., sensitivity to drought. Blade aeration showed mixed-results, with one site showing no hydrological change, and the alternative site showing a reduction in simulated infiltration-excess overland flow and hence a benefit to flood-risk (and potentially water-quality). Dry-stone walls highlighted occasional benefits to soil volumetric wetness and hence lower flood-risk (and potentially water-quality benefits), although results were extremely localised even at the plot-scale and were very inconsistent and mostly statistically insignificant.

8.4 Cross-cutting themes

It is possible that monitored interventions (alongside other interventions) can be used concurrently to deliver hydrological services, and that interventions can be combined into mitigation schemes against flood-risk, drought-risk, or water-quality issues. The changes to hydrological functioning caused by each intervention may also extent beyond targeting a single hydrological hazard, for example, a substantial reduction in the volume of overland flow produced may not only reduce flood-risk, but may also improve local water-quality (e.g., with the introduction of a hedgerow wild-margin or blade aeration). Such improvements could be directly due to hydrological changes e.g., a hedgerow wild-margin improving water-quality by reducing overland flow volume, or such improvement may be indirect to hydrological changes e.g., a hedgerow wild-margin improving water-quality by extracting potentially mobile nitrogenous compounds from the soil. Mitigation schemes may additionally reduce hydrological hazards and simultaneously provide ecosystem services beyond hydrology e.g., improvements to biodiversity (see Section 8.6). It is additionally possible that combining such features may result in benefits beyond the sum of their parts.

It is also possible that combining interventions can conflict and increase other hydrological hazards however e.g., a combined semi-natural grassland and hedgerow wild-margin scheme may reduce flood-risk but could decrease drought resilience. Complimentary measures could be in the form of increased permeability provided by blade aeration, combined with increased permeability provided by the hedgerow wild-margin. These two interventions could work coherently together as hedgerow wild-margins can only be used in certain situations that can sustain a living hedgerow on the peripheries of improved-pastures, whereas blade aeration can be used in small impermeable sections of improved-pasture. Equally, there is a limit to how far a hedgerow wild-margin can credibly extend and the land still be used effectively as improved-pasture. Complimentary measures could also target different hydrological parameters e.g., using blade aeration to reduce flood-risk alongside a hedgerow wild-margin to reduce water-quality deterioration. Although not present in the thesis, combined interventions could directly conflict e.g., one intervention could improve drought-resilience whereas a second intervention could reduce drought-resilience.

Based on the nature of each intervention, all interventions could be used concurrently at a single location within a single mitigation scheme. The only conflict between selecting a specific intervention in a given situation will be deciding between a hedgerow (wild-margins) and a dry-stone wall, likely as the boundary for an improved-pasture. The findings from the thesis preliminary suggest that the hedgerow (wild-margin) may be the optimum feature given it has more obvious hydrological effects regarding soil volumetric wetness, although hedgerows and hedgerow wild-margins are only applicable given certain local conditions (soil, climate etc.), whereas a dry-stone wall is almost universally applicable. Further work is needed to better compare interventions however (see Section 8.3).

8.5 Are interventions across the landscape feasible?

Each intervention was selected so that it did not inhibit the productivity of a farm and was suitable across most pastoral systems operating in the UK (and to a lesser extent, European and global pastoral systems), and therefore facilitates the potential of widespread adoption of the intervention. The possibility of widespread adoption is further evidenced given that all interventions were already present/in-use in the farming systems prior to monitoring. It is very possible for a single small farm to contain all of the studied interventions. Another requirement of each intervention was that they provided ecosystem services beyond hydrology. This requirement therefore offers a stronger argument for increasing agri-environmental payments towards farmers/landowners for services provided by such interventions, but also provides a relatively strong level of insulation against changing legislation and environmental priorities (e.g., a shift from flood-prevention to biodiversity improvements). All of these requirements therefore encourage the widespread inclusion of interventions within pastoral farming systems. Interventions investigated within this thesis could therefore very realistically be incorporated within land-use management schemes in regions largely consisting of grasslands (up to 38 % of total habitable land: Ritchie and Roser, 2019).

The willingness of farmers and landowners to implement such interventions will clearly determine their rate of adoption, and therefore the overall influence these interventions will have on flood-risk, water-quality and drought-resilience. The rate of adoption of the interventions will likely be based on the farmers perceived benefits of the intervention, most likely in terms of direct financial gain (i.e., agri-environment payments), indirect financial gain (e.g., improved resource-use efficiency), or agricultural service (e.g., designation of property boundaries). The time, effort and cost of implementing and maintaining interventions will also be a contributory factor, as will the longevity of the perceived benefits. It is additionally likely that certain interventions may not be suitable for given situations, and therefore it is crucial that a wide range of interventions are investigated if they are to become widespread throughout the landscape (both in the UK and internationally).

8.6 The effect of interventions beyond hydrology

Interventions in this thesis have been investigated in terms of hydrology-related ecosystem services: flood-risk, drought-resilience and water-quality (although a host of studies is still needed). These interventions have had some research directed towards them in relation to primary production and food production, as these were clearly directly beneficial to farmers in terms of output. Many interventions however could also offer soil retention, soil-quality, soil-formation, air-quality, carbon storage, nutrient cycling and biodiversity ecosystem services, as well as a host of cultural services. It is additionally possible that interventions can also be used together to provide different hydrological (and other ecosystem services) services, and that combining interventions may result in benefits beyond the sum of their parts. These additional ecosystem services need to be investigated and quantified if they are to be suitably incorporated within agri-environmental payment schemes, and if farmers/landowners are to be fairly compensated for such services provided by the interventions.

8.7 The thesis in hindsight and with additional resources

With the benefit of hindsight and with additional resources (both financial and time), if the thesis was to be repeated, some elements would be slightly altered:

8.7.1 Chapter 4 (semi-natural grasslands) in hindsight and with additional resources

In Chapter 4, keeping a logbook for the farmer to note the specific dates (and times) of slurrying and the approximate quantity could improve interpretations. Vegetation sampling would have been conducted closer to the soil moisture sampling dates, as this would reduce the chances of any ecological shifts affecting interpretations and the output of the linear mixed-effects model. Although no obvious large-scale ecological shifts were evident, small changes were undoubtedly present (although possibly not detectable at the 1 m^2 scale). The soil samples would similarly be collected shortly before the start of the soil moisture sampling regime to reduce the time between soil moisture sampling dates and a measurement of the soil bio-physico-chemical properties.

Increased time/financial resources could have facilitated additional hydrological measurements to be conducted such as infiltration-capacity, permeability, evapotranspiration and surface roughness (although water access issues existed at the site). Additional hydrological measurements could have improved hydrological interpretations for each land-use. Additional financial resources could also have allowed further bio-physico-chemical comparisons of the two soils.

A substantial increase in time and financial resources could have supported an increased number of monitoring dates (both at higher frequency and over multiple years). Further monitoring dates could capture differences across a wider range of saturations and possibly repeat measurements in similar conditions e.g., during two storms/ droughts or on the same day pre- and post- disturbance (e.g., rainfall or slurry application). The experimental design could also be repeated across multiple sites. The analysis of Chapter 4 could also possibly be expanded to multi-level modelling (and possibly compared with linear mixed-effects modelling).

8.7.2 Chapter 5 (blade aeration) in hindsight and with additional resources

In Chapter 5, determining the initial soil conditions (particularly saturation and compaction) prior to aeration could substantially improve interpretations regarding the effectiveness of the aerator. These antecedent measurements would subsequently improve the understanding of changes caused to the monitored hydrological and pedological parameters, and potentially could dismiss natural soil variation causing observed differences. As aeration was applied prior to monitoring, combining the paired-plot and BACI approach was not possible (which had been adopted at preliminary sites, see Appendix 0.5). Both of these issues are highlighted for future researchers to be aware of in Chapter 5. Soil samples would also have been collected closer to the monitoring dates to avoid shifts in soil bio-physico-chemical properties.

Increased financial and time resources could have facilitated further soil bio-physico-chemical comparison between the two-land uses (increased number of samples, multiple soil pits), including additional parameters such as soil wetness and (macro-)porosity (via X-ray tomography), and therefore an improved understanding of natural soil variability. Additional resources could also reduce the physical requirements of sampling (e.g., quad-bike, automatic press, multiple field-assistants) which would be necessary for further sample collection.

A large increase in resources could have extended the sampling program to further investigate the duration of permeability and soil penetration improvements. Extended monitoring may have continually captured natural soil variation between the land-uses however, and this was a contributing reason towards the decision to conclude the sampling program.

8.7.3 Chapter 6 (hedgerow wild-margins) in hindsight and with additional resources

In Chapter 6, the soil moisture sampling frequency during the artificial rainfall experiment would have been increased to try to more accurately capture the rapid dynamics within the hedgerow wild-margins. Two-minutes was chosen as a reasonable trade-off between refilling the rainfall generator, operating the sTDR moisture-probe, scribing the results, and collecting overland flow samples (the latter was not needed), for the duration of the two-hour experiment. Given the limits of the sTDR however (time taken per sample) questions if the 10-15 second proposed required sampling rate (see Chapter 6) is within the capabilities of the monitoring equipment and experimental design. Possibly combining two sTDR probes and cross-calibrating them post-experiment could resolve this issue. Soil samples would also have been collected closer to the permeability and artificial-rainfall experiment dates.

Further financial and time resources could have expanded Chapter 6 in several manners by including the following: Firstly, a more resilient rain-gauge (preferably one for each overland flow plot) to improve the predicted inputs to each plot, rather than (or alongside) using a rain-gauge not immediately attached to the plots. These additional rain-gauges may also assist in determining if precipitation was convective.

Secondly, equipment to monitor soil-saturation temporally and remotely which may more accurately condition the non-linearity in the transfer function model for the overland flow time-series from natural precipitation (rather than using river discharge as a saturation surrogate). This soil-saturation time-series could undergo transfer function modelling without overland flow occurring, and could extend analysis to include dry periods. Soil saturation would also help determine precise threshold(s) of saturation when overland flow occurs, possibly inferring the overland flow mechanism.

Thirdly, wider analysis of water-quality parameters during the wash-off experiment, most notably expanding into biological analysis or the analysis of heavy metals. The Leith sub-catchment has bad chemical water-quality due to mercurous compounds and polybrominated diphenyl ethers (see Section 3.2.2) which could be investigated. Similarly, nematodes were visible in many waterbodies in the Leith and Petteril sub-catchments which could be investigated. If a telemetered system was included, water-quality analysis could extend to overland flow events from natural precipitation.

Analysis of nitrogenous and phosphoric compounds in soils could also support water-quality interpretations.

Lastly, a multi-year sampling program to record additional overland flow and precipitation measurements to better condition the transfer function models. For the lower hedge-margin (M1), only two natural overland flows were available to condition the model, meaning results are uncertain and unlikely to capture all present dynamics. More time could also have allowed several controlled artificial-rainfall experiments over a wider range of initial conditions.

8.7.4 Chapter 7 (dry-stone walls) in hindsight and with additional resources

In Chapter 7, more thorough initial scoping of the dry-stone wall sites would have been conducted to earlier identify violations in uniformity above and below the boundaries. Occasionally non-uniformity was detected once soil volumetric measurements were underway (often after having already completed an upper/lower grid), causing certain sites to be removed from analysis. The preliminary method of visual inspecting sites regarding land-use, vegetation and slope, as well as consulting geological, historical and pedological maps and conversing with local farmers and the Eden Rivers Trust, was clearly inadequate to detect such non-uniformity. Physically measuring soil volumetric wetness and soil penetration resistance using a course grid (e.g., 20 measurements in each standard 16 x 16 m grid) could potentially rectify this.

With increased time and financial resources, further dry-stone walls could have been monitored. Further monitoring could include repeat measurements at the same site, possibly pre- and post- disturbance. Further resources could support the inclusion of additional hydrological and pedological measurements within the experimental design, such as permeability or soil penetration resistance. These additional measurements could improve holistic understandings, as well as further validate land-use uniformity and the overall experimental design.

A substantial increase in resources could extend the sampling program, the methods of sampling, as well as subsequent analysis. Further resources could facilitate extending the sampling program over an increased number of years and over a larger number of sites, and could also measure dry-stone walls during alternate conditions e.g., in summer when soil volumetric wetness is presumably lower. A large amount of resources could enable hydrogeophysical techniques to be used to measure soil volumetric wetness and possible hydrological flow pathways deeper than at the immediate soil surface, and could be very useful given the depth of the dry-stone walls. Further resources, particularly time, could extent the modelling of Chapter 7, such as to include advanced geostatistical techniques or the inclusion of dry-stone walls in some form of physics-based/ topography-based hydraulic model.

8.8 Future research questions

The thesis has highlighted several significant findings and conclusions from the research. These findings resultantly underline several significant research questions for future studies (each of which could realistically form a future chapter, manuscript or potential thesis of their own), with the key questions developed from each chapter as follows:

8.8.1 Future research questions from Chapter 4 (semi-natural grasslands)

- *The role that slurry plays in saturating improved-pastures:* Slurry potentially offers substantial (localised) drought-resilience benefits within improved-pastures (and possibly elsewhere), although remains unquantified both in this thesis and in scientific literature. As slurry is very heterogeneous, quantifying its effects is likely to require a detailed investigation into the various different forms of slurry. This research could additionally pair with investigations into how slurry influences water-quality in grassland landscapes (Hunter et al., 1999) or how it changes soil bio-physico-chemical properties.

- *Measuring soil saturation at very-high resolutions between improved-pastures and semi-natural grassland:* Even using a fine-scale (1 m^2) sampling grid was not at a high-enough resolution to capture all the present spatial-structure in soil volumetric wetness (i.e., substantial nugget variance was still present). This research is likely to require a large number of field-assistants and field-equipment, or considerable advances in soil volumetric wetness hydrometry to be achievable. This additionally requires investigation across a range of soil saturation conditions.
- *Conducting a similar experiment during saturated conditions using a much larger sampling grid:* The spatial-structure range observed in the improved-pasture was much larger than the sampling grid during saturated conditions. Accurately quantifying the spatial-structure of the improved-pasture is important for future hydrological research as saturated conditions and flood-risk are the dominating interests of UK (and largely international) hydrologists. Information surrounding the range could be an important development for hydrology and hydrological modelling by highlighting the true extent of spatial correlation, which could have impacts for physics-based/distributed model discretisation. Given that sampling within the improved-pasture was substantially easier than the semi-natural grassland, this is at least feasible using current sampling methods, although this

approach would likely require reduced resolution and/or multiple field assistants and soil moisture-probes.

- *Widespread replication of the experiment to contrast semi-natural grasslands against improved-pastures throughout the landscape:* This is in order to: a) quantify how each land-use behaves hydrologically across the landscape rather than at a single field-site, and, b) to support a much-needed modelling study of how the quasi-baseline conditions (semi-natural grassland) have been altered by the introduction of improved-pastures, and therefore to quantify the influence of grassland agricultural practices upon the hydrological functioning of the landscape, and therefore if and how this has changed flood-risk, drought-resilience and water-quality. Once further data on this becomes available, this supports both future scenario modelling as well as historical assessment of the hydrological impact of agriculture in grassland regions.

8.8.2 Future research questions from Chapter 5 (blade aeration)

- *The role of initial soil conditions in determining the effectiveness of blade aeration:* This research question is primarily in relation to antecedent soil saturation, but soil compaction and biological/ecological properties such as the presence of root-mats or soil edaphon would also help identify how/if the aerator improved soil hydraulic properties. Such investigations could involve experiments conducted at small-scales with pre-saturated or pre-compacted plots, or apply an aerator across multiple improved-pastures with differing antecedent conditions.
- *The precise reasons for the observed improvements to permeability and compaction:* This investigation could involve X-ray tomography to assess changes to the porosity network within the soil, as well as changes in soil biota or root

structures. This research is likely to require pre- and post-aeration measurements, as well as multiple field-sites.

- *The role of alternative aerators/soil-loosening devices in regards to surface-hydrology:* Different forms of aerators other than blade-aerators could be investigated in relation to how they alter surface-hydrology. Deeper tillage devices (sward-lifters, subsoilers) could also be investigated, providing that the site is suitable for their use. Scientific literature emphasises that this is indeed a substantial research gap and has been for several decades (e.g., Bhogal et al., 2011).
- *Blade-aeration in arable or semi-natural farming systems:* Blade aeration may be suitable across many different scenarios, which is supported given the robustness of the machinery and the wide-variety of conditions in which they can operate. For instance, semi-natural grasslands that undergo grazing and surface compaction may have root-mats that remain undisrupted for decades and possibly centuries, and may benefit from aeration. Blade aerators could also be used in some arable situations. These could be assessed in terms of hydrological improvements, but also agronomic or ecological.

8.8.3 Future research questions from Chapter 6 (hedgerow wild-margins)

- *The soil-saturation response to artificial-rainfall experiments of different rainfall intensities*: Experiments also need to be conducted over varying (preferably longer) durations, as well as at different initial saturation conditions. Pre-wetting each land-use prior to experiments could also be useful in both generating a response and reducing the amount of water needed to pass through the rainfall-generator.

- *Overland flow water-quality from natural rainfall-events:* This research would capture more accurate volumes and concentrations of potential pollutants coming from each land-use as opposed to the wash-off experiments. Unless these samples could be analysed immediately however, the analysis of hydrochemical and biological contaminants may be very limited. The inclusion of a telemetered system may assist with this.
- *The concentrations of soil nitrogen and phosphorus between land-uses:* The water-quality results from the wash-off experiment highlight that the concentration of potential contaminants in overland flow is substantially different between each land-use. Soil chemistry and microbiology may help to explain the causation of this.
- *The orientation of hedge-margins, and the effect of hedgerows themselves:* It would be beneficial to measure the effect that orientation has on the hydrological functioning of hedgerows/hedge-margins. Specifically focusing upon hedgerows rather than hedge-margins would also be an important advancement.
- *A comparison of streamflow dynamics versus overland flow dynamics:* A comparison of the time constants between flood hydrographs and overland flow hydrographs would investigate the dynamic responses of each system. This research could be conducted either for individual events or for a series of events.

8.8.4 Future research questions from Chapter 7 (dry-stone walls)

- *The effects of dry-stone walls on soil moisture transfer at depth*: The soil moisture-probe deployed throughout the thesis was specifically selected as it could quantify the soil volumetric wetness over the surface 0-6 cm. It is plausible that dry-stone walls affect water movement below this given the depth of the boundary. Studies using deeper soil moisture-probes, as well as hydrogeophysical methods e.g., electrical resistivity, is therefore needed to investigate how these alter deeper hydrological pathways.

- *The effects of dry-stone walls on precipitation interception and possibly evapotranspiration*: Dry-stone walls may provide some interception and rain shadowing effects for land immediately downwind of the dry-stone wall as suggested in Chapter 7, as well as qualitatively shown by livestock sheltering behind dry-stone walls during precipitation events. They may also play some role in evapotranspiration. This research question requires quantification and pairs relatively well with prior hedgerow studies (e.g., Herbst et al., 2006; 2007), and could support agricultural scenario assessments.
- *The orientation of dry-stone walls*: Much like the future research question raised in Chapter 6, the effect of dry-stone walls on hydrology and hydrological pathways is likely to be influenced by the orientation of the wall (both topographically i.e., parallel or perpendicular to gradient, and in relation to gradient, but also in terms of prevailing wind direction due to possible sheltering effects).
- *The effect of different styles/conditions of dry-stone walls on hydrology*: Different construction styles and materials of dry-stone wall require investigation e.g., different rock type or ages. It would also be beneficial to assess dry-stone walls in different conditions e.g., stock-proof vs derelict. It would equally be instructive to investigate dry-stone walls that include clearly defined (small-)stream channels or field-drainage pathways as part of their construction.

- *The effect of dry-stone walls on other ecosystem services beyond hydrology*: This research most notably expands into pedology regarding soil-conservation properties of dry-stone walls – especially given the amount of resources directed towards alternative stone structures such as stone terraces (e.g., Arnáez et al., 2015). Given the minimal literature regarding dry-stone walls, a host of other studies is needed to understand how they influence environmental functioning (Collier, 2013).

8.8.5 Additional future research questions

- *Hydrological studies into different grassland interventions:* There is a need for hydrological observations of many more common agricultural practices/features taking place in grasslands, as all interventions are extremely understudied. This research could involve repeating or modifying the experimental designs adopted here, or creating entirely new methodologies. Future interventions could include: Grazing strategy (e.g., rotational grazing vs set stocking), differing pastoral vegetation species, land-preparation effects (e.g., ploughing, harrowing), surface additives (e.g., slurry, lime etc.), trees/woodlands within improved-pastures, agricultural boundaries (fences, barbed wire etc.), the cutting and harvesting of fodder, and many more (see Table 1.1: Figure 1.1). Interventions not currently adopted in pastoral grasslands, but which could be introduced, could also justify investigation.
- *A comparison of the effectiveness of different interventions.* Although this may be difficult given the very different application of certain interventions, some interventions could be directly compared e.g., hedgerows and dry-stone walls, and

could include monitoring of the same hydrological parameters for each intervention.

- *Quantifying the natural spatio-temporal variability of hydrological variables of importance within pastoral systems, both in the presence and absence of interventions.* All chapters highlight considerable spatio-temporal dynamics relating to hydrological variables within pastoral systems. This spatio-temporal variability (and resultant uncertainty) needs to be further investigated. Studies may wish to investigate this in the absence of interventions e.g., adjacent sites to the paired-plot sites in Chapter 7 without a dry-stone wall would help investigate the natural variability in soil volumetric wetness across and down the slope.
- *Ecosystem services beyond hydrology provided by interventions:* This research could involve studies in entirely different scientific fields e.g., atmospheric science, ecology, psychology etc. Investigations could additionally involve pairing hydrology, pedology or agricultural sciences with other sciences.
- *Future (and historic) scenario modelling – climatic, political and agricultural:* Modelling how climate change or political legislation could also alter the agricultural environment of the UK (and overseas), and therefore if certain interventions will no longer be feasible e.g., if climate change or invasive species causes certain hedgerows species to no longer be viable, or certain practices become banned. Various agricultural scenarios under current conditions could also be modelled. This approach may also allow historic assessments. This is important to assess the influence of interventions beyond the plot-field scale as done in this thesis.

8.9 Closing remarks

Grassland agriculture has evolved over millennia to maximise agricultural output rather than to satisfy environmental concerns, although the two need not necessarily be mutually exclusive or in opposition. A balance between maintaining and increasing agricultural output whilst minimising environmental hazards is essential given the rising food demand of a globally increasing population, as well as the growing threat posed by hydrological hazards. This thesis has aimed to do this by investigating interventions that either maintain or improve agricultural output of grassland farming systems, whilst simultaneously quantifying their hydrological role, specifically in relation to overland flow and rapid shallow-subsurface flows. This research can therefore allow improvements to be made to flood and water-quality hazards, and to a lesser extent drought-risk, within grassland landscapes.

This thesis has successfully quantified and modelled the spatio-temporal hydrological benefits of four widespread grassland interventions in the Eden catchment, with similar interventions present throughout much of the UK and temperate Europe, as well as further overseas. Despite the widespread incidence of grasslands globally (up to ~40 % of total land area), very little research exists regarding how they behave hydrologically within the landscape, and how using grasslands for agriculture has altered hydrological processes. Although this thesis has produced several pioneering chapters and publications, this remains a significant future research question, particularly given that many grassland regions are extremely prone to hydrological hazards, and these grasslands are likely to continue to be used for agriculture for decades and centuries to come.

The new approach of using agricultural interventions within a heavily managed landscape such as improved-pastures (improved-pasture hydrology) is also an important output from the work, and is a modified form of Natural Flood-Risk Management. This approach has strong potential to be used within land-use management to offer protection against hydrological hazards, especially given that grasslands are so widespread. The interventions additionally are largely welcomed by farmers, and therefore uptake is unlikely to be a problem if subsidised through agri-environmental (or alternative) payments. The continual and intensive monitoring of interventions by pastoral farmers who access their land regularly means improvements to the hydrological understanding of these features could be extremely rapid in the future if farmers are to be incorporated within the monitoring and management of such features.

Many of these improved-pasture interventions are already present within the majority of pastoral systems, or can be incorporated with them. Any governmental scheme which rewards the provision of ecosystems services or goods provided by farmers/landowners (arable or pastoral), should incorporate these interventions for an improved holistic acknowledgement of these services. Increased monetary provisions should also be made available to farmers for the maintenance or implementation of these interventions that are already included within other agri-environmental schemes (often in relation to alternative ecosystem services).

The largest conclusion from the thesis is that an extremely large range of further hydrological studies are needed to further the understanding of grassland-, and particularly improved-pasture-, hydrology. Further studies may include direct replication (and modification) of the reproducible experimental paired-plot approaches (possibly with the inclusion of before-after-control-impact experiments), statistical designs, and modelling techniques/philosophies, presented in the thesis across the landscape. Alternative experimental, statistical and modelling designs/approaches may also be justified, possibly including suggestions made throughout the thesis. Such further research could be conducted in very different areas to those in the thesis such as internationally, involve different grassland interventions, and ideally should be conducted over longer timeframes. This is largely because plot-scale studies have limited representativeness at the hillslope or catchment-scales (micro-large) due to uniqueness of place and issues of scale. Equally, scenario-based modelling of both studied and unstudied interventions is also likely to be of importance for both future and historic land-use assessment (in terms of hydrology but also other disciplines). A host of additional observations, modelling studies and hydrological investigations are needed to both inform hydrological processes and therefore to support the operation of models to ensure they are credible, as well as to confirm that they acknowledge the extremely complex nature of the hydrological system.

9 APPENDICES

9.1 Appendix A0.0: Preliminary datasets

Several preliminary investigations were conducted relating to grassland interventions and related features. Each of these interventions were not selected as a main intervention and therefore a chapter within the thesis, although the hydrological data for these are presented. Preliminary investigations were useful to gain experience with field hydrometry and experimental designs, as well to test several possible interventions and improve relationships with both the Eden Rivers Trust, the local farming community, as well as local landowners.

9.1.1 Appendix A0.1: The effect of coniferous plantations on topsoil saturated hydraulic conductivity versus permanent pasture (Whinlatter, Cumbria)

A preliminary dataset into the effect of coniferous plantations on topsoil saturated hydraulic conductivity using a Talsma ring permeameter at Whinlatter Forrest Park, Cumbria. Acknowledgments are due to James Slater and the respective landowner(s). This dataset was not carried forward directly into the thesis as a forestry plantation was considered to be land conversion rather than a pasture intervention, as well as the experimental site not being within the ERT area of operation. The appendix data is available at:

https://www.lancaster.ac.uk/lec/sites/qnfm/wallace/Wallace_Thesis_Appendix_A0.1.doc x and

https://www.lancaster.ac.uk/lec/sites/qnfm/wallace/Wallace_Thesis_Appendix_A0.1.xlsx.

9.1.2 Appendix A0.2: The effect of coniferous plantations on topsoil saturated hydraulic conductivity versus permanent pasture (Thrimby, Cumbria)

A preliminary dataset into the effect of coniferous plantations on topsoil saturated hydraulic conductivity using a Talsma ring permeameter at Thrimby, Cumbria. Acknowledgments are due to David Bryan and the respective landowner(s). This is the same improved-pasture as used within Chapter 6, although approximately 225 metres away from the overland flow plots. This dataset was not carried forward directly into the thesis as a forestry plantation was considered to be land conversion rather than a pasture intervention, and the measurements were deemed too far from the overland flow plots to be directly comparable to the hedgerow wild-margins as used in Chapter 6. This appendix data is available at:

https://www.lancaster.ac.uk/lec/sites/qnfm/wallace/Wallace_Thesis_Appendix_A0.2.doc
x and

https://www.lancaster.ac.uk/lec/sites/qnfm/wallace/Wallace_Thesis_Appendix_A0.2.xlsx.

9.1.3 Appendix A0.3: The effect of hedgerows on topsoil saturated hydraulic conductivity versus permanent pasture (Thrimby, Cumbria)

A preliminary dataset into the effect of hedgerows on topsoil saturated hydraulic conductivity using a Guelph permeameter at Thrimby, Cumbria. Acknowledgments are due to Katherine Deeming and the respective landowner(s). This is the same hedgerow

associated with Chapter 6, although located further downslope. This dataset was not carried forward directly into the thesis due to the large inaccuracies associated with the guelph permeametry on the clay-rich soils, as well as the monitoring of the hedgerow rather than the hedge-margin itself as done in Chapter 6. This appendix data is available at:
https://www.lancaster.ac.uk/lec/sites/qnfm/wallace/Wallace_Thesis_Appendix_A0.3.docx and
https://www.lancaster.ac.uk/lec/sites/qnfm/wallace/Wallace_Thesis_Appendix_A0.3.xlsx.

9.1.4 Appendix A0.4: The effect of harrowing, cattle grazing and agricultural traffic on topsoil saturated hydraulic conductivity versus permanent pasture (Clifton, Cumbria)

A preliminary dataset into the effect of harrowing on topsoil saturated hydraulic conductivity in a cow-grazed field and a cow-grazed, heavily trafficked field using a Talsma ring permeameter at Clifton, Cumbria. Acknowledgments are due to Abi Speakman, Anthony Errington at Low Moor Farm, and the respective landowner(s). This dataset was not carried forward directly into the thesis due to the inability to conduct repeat measurements and therefore create a substantial chapter, although data collected here is possibly suitable for a future manuscript, particularly if paired with additional data (see Chapter 5). This appendix data is available at:
https://www.lancaster.ac.uk/lec/sites/qnfm/wallace/Wallace_Thesis_Appendix_A0.4.docx and
https://www.lancaster.ac.uk/lec/sites/qnfm/wallace/Wallace_Thesis_Appendix_A0.4.xlsx.

9.1.5 Appendix A0.5: The effect of subsoiling on topsoil saturated hydraulic conductivity on a permanent pasture receiving heavy agricultural traffic and cattle grazing (Plumpton, Cumbria)

A preliminary dataset into the effect of subsoiling on topsoil saturated hydraulic conductivity in a heavily-trafficked pasture which was also cow-grazed. Acknowledgements are due to Cerys Gregory, Patrick Grimes, John Gibson at Castlesteads Farms, and the respective landowners(s). Numerous baseline topsoil saturated hydraulic conductivity measurements were taken, although the farmer later decided against subsoiling the monitored field. This dataset was not carried forward directly into the thesis due to the lack of an intervention, although this was initially planned to be either paired with Chapter 5, or form a separate chapter. This appendix data is available at:
https://www.lancaster.ac.uk/lec/sites/qnfm/wallace/Wallace_Thesis_Appendix_A0.5.docx and
https://www.lancaster.ac.uk/lec/sites/qnfm/wallace/Wallace_Thesis_Appendix_A0.5.xlsx.

9.1.6 Appendix A0.6: The effect of sheep versus horses versus ploughing in relation to topsoil soil moisture regimes within permanent pasture (Rosgill, Cumbria)

A preliminary dataset into the effect of sheep grazing, horse grazing, and ploughing, on topsoil soil volumetric wetness in grazed improved-pastures. Acknowledgements are due to Elizabeth Botcherby, Richard Carruthers at Rawfoot Farm, and the respective landowners(s). This dataset was not carried forward into the thesis as horse grazing as an intervention was not considered substantially widespread in the Eden catchment. This appendix data is available at: https://www.lancaster.ac.uk/lec/sites/qnfm/wallace/Wallace_Thesis_Appendix_A0.6.docx and https://www.lancaster.ac.uk/lec/sites/qnfm/wallace/Wallace_Thesis_Appendix_A0.6.xlsx.

9.2 Appendix B4.0: Chapter 4 Appendix

The appendix from Chapter 4 (Semi-natural grasslands), including the nugget variance associated with the soil moisture-probe uncertainty.

9.2.1 Appendix B4.1: Nugget variance associated with soil moisture-probe uncertainty

Supplemental Table 4.S1: A simulated semi-variogram with 2 % θ_V error (identical to the soil moisture-probe) was generated over a grid of equal-scale to the plots used in Chapter 4, with the error uniformly distributed. This simulated semi-variogram was used to assess the impact of the moisture-probe uncertainty on semi-variogram model parameters. The actual range in metres is given below the effective range. Note that the sill is the nugget plus the partial sill. This table should be used alongside Supplemental Figure 4.S1, and compared with Table 4.5 and Figure 4.13.

Semi-Variogram	**Nugget effect (γ_0)**	**Sill ($\gamma_0 + P_0$)**	**Effective range**	**Model**	**Residual sum of squares**
Simulated	0.045	1.002	0.072 (3.6m)	Sph	0.028

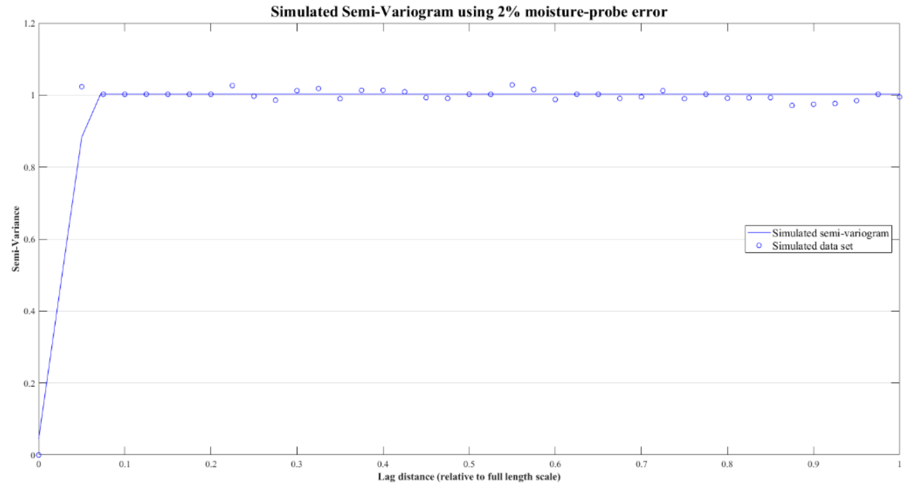

Figure 4.S1: The simulated semi-variogram with 2 % θ_V error (identical to the soil moisture-probe) was generated over a grid of equal-scale to the plots used in Chapter 4, with the error uniformly distributed. This simulated semi-variogram was used to assess the impact of the moisture-probe uncertainty on semi-variogram model parameters (see Table 4.S1).

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