What impact do semi-natural woodlands have on flooding in the UK?

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Submitted in accordance with the requirements for the degree of

Doctor of Philosophy

The University of Leeds

School of Earth and Environment

March 2022

Declaration and author contributions

The candidate confirms that the work submitted is her own, except where work which has formed part of jointly authored publications has been included. The contribution of the candidate and the other authors to this work has been explicitly indicated below. The candidate confirms that appropriate credit has been given within the thesis where reference has been made to the work of others.

Chapter 2 has appeared in publication as:

Monger, F., Spracklen, D.V., Kirkby, M, J., Schofield, L. 2021. The impact of semi-natural broadleaf woodland and pasture on soil properties and flood discharge. *Hydrological Processes*, 36: e14453. DOI: 10.1002/hyp.14453.

Contributions: FM led the study design and planning with support from all authors. FM completed all fieldwork with assistance from DS to install field equipment. FM completed all laboratory work and data analysis. FM, DS and MK contributed to the interpretation of data. FM led the manuscript writing and all authors critically assessed the drafts and approved the final version.

Chapter 3 has been accepted for publication in Hydrological Processes as:

Monger, F., Bond, S., Spracklen, D.V., Kirkby, M, J. 2022. Overland flow velocity and soil properties in established semi-natural woodland and wood pasture in an upland catchment. *Hydrological Processes.*

Contributions: This study was designed by FM with the support from all authors. SB developed the flume concept in previous work and provided vital knowledge of the method. FM undertook all data collection with assistance from SB. FM completed all laboratory work and analysis, and led the writing of the manuscript. All authors critically assessed the manuscript and approved the final version.

Chapter 4 is the following manuscript, prepared for submission:

Monger, F., Lane, J.V., Spracklen, D.V., Kirkby, M, J. *In prep*. The influence of land cover and catchment characteristics on active storage within UK river catchments.

Contributions: FM planned the study with support from DS and MK. FM led analysis and modelling of the historical dataset. JL provided modelling and coding support. All authors contributed to the interpretation of results. FM prepared the manuscript, with feedback from all authors and approved the final version.

Chapter 5 is the following manuscript, prepared for submission:

Monger, F., Spracklen, D.V., Kirkby, M, J., Willis, T., *In prep*. Investigating the impact of woodland placement and percentage cover on flood peaks in an upland catchment using distributed TOPMODEL.

Contributions: FM developed model scenarios with guidance from DS and MK. Model calibration, validation, scenario runs and analysis were led by FM. TW produced the ReFH storm event and provided vital knowledge of the model. FM, DS and MK contributed to the interpretation of analysis. FM led the writing of the manuscript and all authors critically assessed drafts.

Thesis by alternative format rationale

This thesis follows the University of Leeds Faculty of Environment protocol for the format and presentation of an alternative style of doctoral thesis including published material. The research questions of the project were investigated using a range of approaches, which made the presentation of the data chapters as four individual manuscripts appropriate.

One of the manuscripts has now been published, one is under consideration for publication, following an initial round of peer review and a request for revisions and the final two manuscripts are prepared for submission to journals.

The main body of the thesis therefore consists of the published and prepared manuscripts. This is preceded by an introduction, which provides background information, reviews relevant literature, and outlines the aims and objectives of the work.

A synthesis chapter, bringing together the findings of the four manuscripts and discussing them in the context of the research questions, concludes the thesis.

Copyright declaration

Acknowledgements

This research was funded by the University of Leeds and the United Bank of Carbon (UBoC).

Firstly, I would like to thank my supervisors, Dom Spracklen and Mike Kirkby, whose guidance, feedback and support has made this thesis possible.

Thank you to United Utilities and RSPB Haweswater for granting land access, permissions, and equipment storage with a special mention to Lee Schofield and Spike Webb, who provided invaluable guidance and support.

Thank you to those in the School of Geography that offered me technical support in the lab and field, including David Ashley, Sarah Hunt and Rachel Gasior. Thanks, also, to Tom Willis for his assistance with SD-TOPMODEL and Jude Lane for her patience and guidance with regards to statistical analysis. Thank you to Steph Bond for offering her support both in the field and via Teams.

Finally, I would like to thank my parents for their unwavering support; thanks to my dad Martin for his relentless determination when assisting with fieldwork and mum Mandy for her endless encouragement and reassurance.

Abstract

Flooding is one of the most costly and widespread climate-related natural hazards, causing substantial risk to communities across the UK. The frequency of flood events has increased and future climate change is expected to cause even more intense rainfall, further increasing the risk of flooding. In response, the interest in natural flood management, in particular woodland creation, has grown as a way of reducing flood risk by storing more water and slowing the flow of water across the land. However, despite well-known connections between woodlands and water there is still low confidence in using woodlands as a flood mitigation method due to limited empirical data, particularly for broadleaf woodlands and at the catchment scale.

This thesis examines the impact of woodland on flooding in the UK using a range of approaches including; field monitoring, laboratory experiments, modelling and data analysis. Field monitoring, implemented in Naddle, Cumbria, found small (< 0.2 km^2) catchments consisting of semi-natural woodland exhibited more muted responses to storm events when compared to pasture catchments. Hillslope-scale overland flow velocity investigations found wood pasture dominated by bracken reduced overland flow velocity when compared with established semi-natural woodland. The extent and spatial distribution of woodland cover impacted peak discharge simulated for a 1 in 50-year storm using SD-TOPMODEL; cross slope and riparian woodland resulted in peak discharge reductions up to 3.93 % for a 10-percentage-point increase in woodland cover compared to catchment woodland. At the catchment-scale, land cover, with the exclusion of woodland, was found to be of significance to catchment active storage when flow data from the NRFA for 418 catchments across the UK was analysed. Although not significant, woodland cover still had high relative importance to catchment active storage. Overall, the findings of this thesis confirm that woodland has an important role in mitigating future flood risk, even at larger scales.

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List of Abbreviations

AICc	Akaike's Information Criterion
ANOVA	Analysis Of Variance
AOD	Above Ordnance Datum
BGS	British Geological Survey
CEH	Centre Of Ecology and Hydrology
CG	Commons Grazing
DEFRA	Department For Environment, Food And Rural Affairs
ELMS	Environmental Land Management Scheme
FRM	Flood Risk Management
GLM	Generalised Linear Model
IDF	Intensity Duration Frequency
Kfs or Ksat	Saturated Hydraulic Conductivity
LG	Low-Density Grazing
LU	Livestock Units
LWD	Large Woody Debris
NFM	Natural Flood Management
NRFA	National River Flow Archive
NSE	Nash-Sutcliffe Efficiency
m	Active Storage Parameter
PAWS	Plantations On Ancient Woodland Sites
QGIS	Quantum Geographic Information System
RSPB	Royal Society for the Protection of Birds
SPD	Specific Peak Discharge
TDR	Time-Domain Reflectometry
UK	United Kingdom
UNDRR	United Nations Office For Disaster Risk Reduction
VIF	Variance Inflation Factors
W	Semi-Natural Woodland
WWNP	Working With Natural Processes

Chapter 1. Introduction

In recent years, the frequency of flood events has increased across the globe, causing a vast amount of environmental and economic damage (Kundzewicz et al., 2014; UNDRR, 2020). The impact of anthropogenic climate change is predicted to further increase the severity and reoccurrence of such events (Tabari, 2020). To try to account for greater flood risk in the future, policy makers and practitioners have looked towards alternative flood mitigation methods rather than depend solely on traditional flood management strategies (Burgess-Gamble et al., 2017). This has included an interest in nature based solutions, such as natural flood management (NFM). In particular, there has been increased discussion surrounding the use of land management, often woodland creation, as a flood management strategy. There is a long history of research in the UK into the impact of woodlands on the hydrological cycle, however it has been mostly limited to the impact of coniferous plantations (Nisbet, 2005).

This introductory chapter provides a background for the research in this thesis and aims to review key areas relating to the practice of NFM; primarily the development of land use as an NFM strategy, with a particular interest in woodland. The past, present and future UK land use will be identified and the current state of knowledge regarding the impact of woodland as a method of flood mitigation will be presented.

In Chapter 2, Chapter 3, Chapter 4 and Chapter 5, I include 4 manuscripts investigating the role of woodland in reducing flood risk. Chapter 2 will introduce the establishment of the correlation catchment study undertaken at RSPB Haweswater, which investigates the impact of semi-natural woodland on soil hydrological properties and flood discharge. Chapter 3 will present the overland flow velocity findings from upland woodland and wood pasture habitats. UK-wide flow data will be used to calculate catchment active storage in Chapter 4, in order to investigate its relationship with land cover. The rainfall-runoff model, SD-TOPMODEL will then be used in Chapter 5; to investigate the impact of woodland cover on peak flow, using findings from Chapters 2 and 3. Chapter 6 will bring together the main findings of this thesis.

1.1. Flooding in the UK

The frequency of flood events have notably increased worldwide over the last three decades (Kundzewicz et al., 2014; Wingfield et al., 2019), and the UK is no exception. In the UK's recent history, there have been a number of detrimental flood events. Years of notable flooding include; 1998, 2000/2001 - the millennium floods, 2007, 2012, 2013/2014 and

2015/2016. The risk from flooding is only expected to increase due the predicted intensification of the hydrological cycle as a result of anthropogenic climate change (Tabari, 2020).

In the past, there has been a reliance on traditional flood management methods, consisting of costly, hard-engineered structures. Traditional interventions such as flood detention reservoirs and flood walls have come under immense pressure in recent years from the persistent reoccurrence of flood events (Dadson et al., 2017). Consequently, there has been a growing international interest in the use of 'soft-engineered' flood mitigation schemes, such as NFM (Stevens et al., 2016).

1.2. NFM

NFM, also known as Working with Natural Processes (WWNP) or nature-based solutions, aims to work with natural processes to enhance the water storage capacity of a catchment and to 'slow the flow'. Approaches may include the development of built water storage (Quinn et al., 2013; Nicholson et al., 2020), river restoration (Dixon et al., 2016), leaky debris dams (Thomas and Nisbet, 2012; Ashbrook, 2020) and land-use management (Spray et al., 2016) among methods shown in Figure 1.1. These methods can also provide ecosystem services such as pollution assimilation, habitat creation and carbon storage (Hankin et al., 2017).

NFM, is becoming increasingly favoured due to its lower costs compared to more traditional flood management methods (Wells et al., 2020) and its ability to benefit the wider ecological, economic and social spheres (Connelly et al., 2020). NFM works well when a number of NFM strategies are implemented throughout a catchment and can also be used as part of a holistic flood management strategy; with NFM measures developed in a catchment's headwaters and engineered flood defences installed downstream (Connelly et al., 2020).

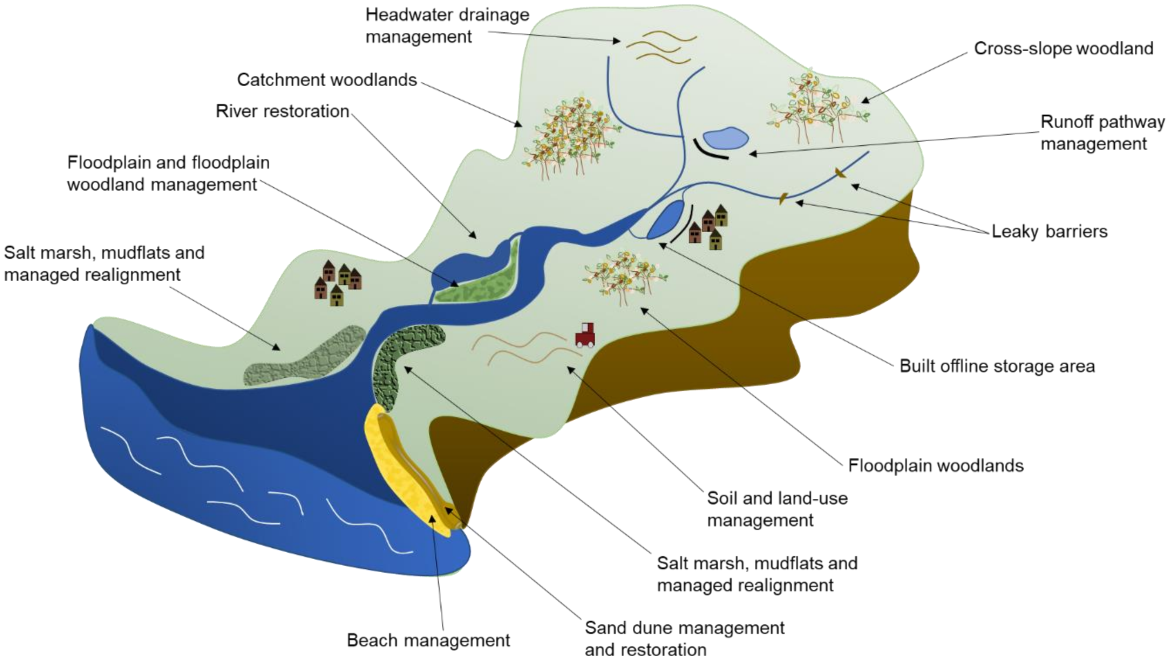

Figure 1.1 Natural flood management approaches throughout a catchment, adapted from (Burgess-Gamble et al., 2017)

Two examples of successful NFM are 'Slowing the Flow at Pickering' and the restoration of Swindale Beck in Cumbria. The 'Slowing the Flow at Pickering' project was a DEFRA funded pilot project which combined a more traditional flood management approach and a number of NFM interventions. A traditional bund was constructed alongside the installation of 129 leaky wooden (LWD) dams, 2 timber bunds, the blocking of moorland drains and woodland planting. These catchment-wide interventions reduced the risk of flooding in the town of Pickering from a 25 % chance of a flood event in any given year to less than 4 % (Nisbet et al., 2015). River restoration of the Swindale Beck involved re-meandering the previous straightened river channel through a partnership between the RSPB, the Environment Agency, United Utilities and Natural England. The restored channel is 140 m longer and around 2 m wider than the old route and now much better connected to the floodplain (Schofield et al., 2017). This has created a buffer against flood events, where the river can spill onto the floodplains and also has improved the habitat for spawning fish.

In the UK, the implementation of NFM measures has been increasingly encouraged as part of a government agenda, most recently through the allocation of £15 million of funding for NFM projects in the '25 Year Environment Plan' (DEFRA, 2018). However, there is still much uncertainty surrounding the effectiveness of NFM interventions (Black et al., 2021).

In recent years, there have been several attempts to review the success of NFM interventions through the analysis of both observational and modelling studies (Black et al., 2021). There is an emphasis that the effectiveness of NFM measures are site-specific, and dependent on many factors (Burgess-Gamble et al., 2017). Where multiple interventions have taken place it can be difficult to disentangle the effects of an individual interventions. There is also a general consensus that the larger the flood event, the smaller the scope for reducing flood risk using NFM measures (Dadson et al., 2017).

1.2.1. Land management for NFM

Presently, there is a particular interest in land management as a method of NFM (Wingfield et al., 2019). It is generally accepted that the management of the land influences hydrological processes (Buytaert et al., 2006; Cheng et al., 2017) through mechanisms such as infiltration, interception, soil storage capacity and runoff attenuation (Bronstert et al., 2002; Carroll et al., 2004; Chandler and Chappell, 2008; Marshall et al., 2014). However, there are conflicting views about the extent to which land management may reduce flood risk, primarily due to lack of data (Ngai et al., 2017). For land use management to develop as an NFM strategy, more certainty is needed surrounding the impact of land use on reducing the risk of flooding.

One topic which needs further research is that of the influence of arable and grassland farming in the UK on flood risk. Both types of farming can contribute to changes in topsoil structure through compaction by livestock or the use of heavy machinery (Holman et al., 2003). This can alter soil bulk density, which in turn affects soil hydrological properties, impacting flood risk (Meyles et al., 2006; O’Connell et al., 2007; Burgess-Gamble et al., 2017). Previous studies have found that the removal of livestock generally allows for soil structure to improve over time, enhancing infiltration and soil storage capacity (Gifford and Hawkins, 1978; Nguyen et al., 1998). Although, recent reviews of the literature highlight that investigations into reduced grazing as an NFM strategy is often at the plot scale (Burgess-Gamble et al., 2017).

A second area of interest is the way in which floodplains are managed for flood risk. Floodplains have been increasingly altered for urban and agricultural development (Wheater and Evans, 2009) causing them to become disconnected from their rivers, increasing flood risk. Fortunately, floodplains are now being restored so they are able to actively store water. The restoration of Padgate Brook as part of the Warrington Flood Risk Management (FRM) Scheme is a successful example of this. The scheme established a more natural river system that allows water to spill onto the floodplain on a regular basis; this has improved the level of flood protection for the surrounding properties to a 1-in-100 year level and has restored 5 ha of reed bed. (Burgess-Gamble et al., 2017).

1.3. Land cover in the UK

In the UK, improved grassland is the dominant land cover type accounting for 29 % of the total land use. The second most abundant land cover is arable (23 %), followed by woodland (both broadleaf and conifer, 13 %), and urban/suburban land cover which makes up 8 % (UKCEH, 2021).

The UK has experienced major changes in land cover over the last century. The agricultural landscape has changed drastically; from small fields surrounded by hedgerows with naturally meandering rivers to the installation of land drains, channelization of rivers, loss of hedgerows and increase in field sizes (Wheater and Evans, 2009). Sheep numbers doubled between 1950 and 1990 as a result of farm support payments (Fuller and Gough, 1999). Such intensification of farming in the UK is thought to have degraded the soil structure (O'Connell et al., 2004; Wheater and Evans, 2009) through the compaction of soil beneath sheep tracks (Wheater, 2006) and modern-day farm machinery wheels (Hamza and Anderson, 2005). These impacts are thought to decrease the levels of infiltration into the soil so that infiltration-excess overland flow becomes more common. It is important to note that since the 1990s changes to the agricultural policy have however led to some de-intensification of arable farming and some reductions in grazing densities (Fuller and Gough, 1999).

Furthermore, attitudes towards woodlands have changed in the UK's recent history. In the early 1900s considerable amounts of ancient woodlands were felled to support industry and defence. Policy introduced in the 1920s and reaffirmed in the 1940s promoted 'rapid blanket afforestation' of non-native coniferous species to boost the country's timber reserves. Afforestation of this type peaked in the 1970s, however due to increasing criticism of its negative impacts to biodiversity, forest policy was directed to the use of forest for not only timber but in order to produce multiple benefits (Nisbet et al., 2011).

A recent investigation into changes in land cover by the UK Centre of Ecology and Hydrology (CEH) calculated a net increase in woodland area of 5,236 km^2 between 1990 and 2015 across Great Britain (UKCEH, 2020). Conversely, there was a net reduction in all types of grassland of 7,668 km^2. With 2,505 km^2 of grassland converted to urban use. Overall, there was net increase of 3,376 km^2 in urban areas, with the majority occurring in England. In

Scotland, woodland cover expanded at the expense of grassland and arable farmland alongside lower levels of urbanisation.

1.3.1. Woodland cover in the UK

At present, it is estimated that there is 32,000 km^2 of woodland within the UK, covering 13 % of the total land area; 10 % in England, 15 % in Wales, 19 % in Scotland and 9 % in Northern Ireland. For this estimate, woodland is defined as 'land under stands of trees with a canopy cover of at least 20% (25% in Northern Ireland), or having the potential to achieve this' (Forestry Commission, 2021). Coniferous woodlands account for 51 % of the UK's wooded area, as low as 26 % in England and as high as 74 % in Scotland (Forestry Commission, 2021).

Ancient woodland covers 2.5 % of the UK's land area (Reid et al., 2021). Ancient woodland describes woods that are thought to have been present since 1600 in England and Wales and 1750 in Scotland (Reid et al., 2021). Ancient woodland can be divided into ancient semi-natural woods; which are woods that have developed naturally, but have been managed over the centuries or plantations on ancient woodland sites (PAWS) which are ancient woods that have been felled and replanted with non-native species (Forestry Commission, 2003).

Semi-natural woodland has sometimes, mistakenly, been used interchangeably with ancient woodlands. With the expectation of plantation woodland, it has been argued that all woodland in the UK is now considered to be 'semi-natural', as 'no areas remain that haven't been touched by people in some way' (Reid et al., 2021). The Forestry Commission define semi-natural woodlands as woods which are composed of predominately native trees and shrubs, of which some have been planted or managed (Forestry Commission, 2003).

Approximately 3.2 % of Britain's land area (data not available for Northern Ireland) consists of 'trees outside woodlands', such as; hedgerows, street trees, trees on farms and along rivers (Reid et al., 2021). Trees in wood pastures are often included in this category. Wood pasture is a mosaic of different successional stages between grassland and woodland (Peringer et al., 2013; Smit et al., 2005; Uytvanck et al., 2008) including the following features: grazing animals, an open ground layer or grassland or heath, shrubs and scrub, veteran trees and decaying wood (Reid et al., 2021).

1.3.2. Future woodland creation

Woodland creation is central to the UK government's target of to reaching net zero emissions by 2050 (UK Government. 2021), as trees are seen as one of the most effective strategies for climate change mitigation and important for carbon sequestration (Bastin et al., 2019; Murphy et al., 2022). Furthermore, woodlands have the potential to improve water quality through reduced soil erosion and limiting the delivery of pollutants to watercourses (Nisbet and Broadmeadow, 2003; UK Government. 2021).

The UK government has committed to creating 30,000 hectares of new woodland per year by the end of this Parliament (2024) (DEFRA, 2018, Jordan and Wentworth, 2021). However, in recent years areas of new planting in the UK only reached a maximum of around 13,000 hectares per year (in 2011-12, 2013-14, 2018-19 and 2019-20) and fell to as low as 6,000 hectares per year (2009-10 and 2015-16). Broadleaf woodland accounted for 43 % of new planting in 2019-20 (Forestry Commission, 2020) and is expected to make up a substantial component of the 11 million trees the UK government committed to planting by 2050 under the UK's 25 Year Environment Plan (DEFRA, 2018). This would increase UK woodland cover to about 18% and it is expected that a large proportion of these new woodlands will be established in the uplands (Murphy et al., 2022).

1.4. Woodland for NFM

It is generally accepted that woodlands can impact flood hydrology and have the potential to influence rainfall-generated flooding (Stratford, 2017). Woodlands interact with the hydrological cycle in a number of ways, see Figure 1.2, such as the processes of interception, evaporation, transpiration (evapotranspiration), soil infiltration and soil storage.

1.4.1. Interception

Trees have the ability to intercept a variable amount of rainfall via their leaves, branches and tree trunks. Trees are usually taller and have greater leaf area than other vegetation types (Nisbet, 2005; Calder and Aylward, 2006) therefore are able exert a greater hydraulic roughness (Confor, 2015). This however, is dependent on trees species. Evergreen species are able to maintain high interception rates all year round and typically intercept 25-45 % of annual rainfall compared to deciduous trees, which are said to intercept on average 10-25 % of annual rainfall (Ahmad-Shah and Rieley, 1989; Nisbet, 2002). Deciduous trees are only in full leaf from June to September therefore interception is limited to tree trunks and branches for a proportion of the year (Roberts and Rosier, 2005). The amount of rainfall intercepted during individual storm events is still debated as the relationship between interception loss and rainfall amount is still not fully understood (Cooper et al., 2021). It is thought that interception is much more effective during light showers, significantly decreasing as the storm intensity increases (Anderson et al., 1976).

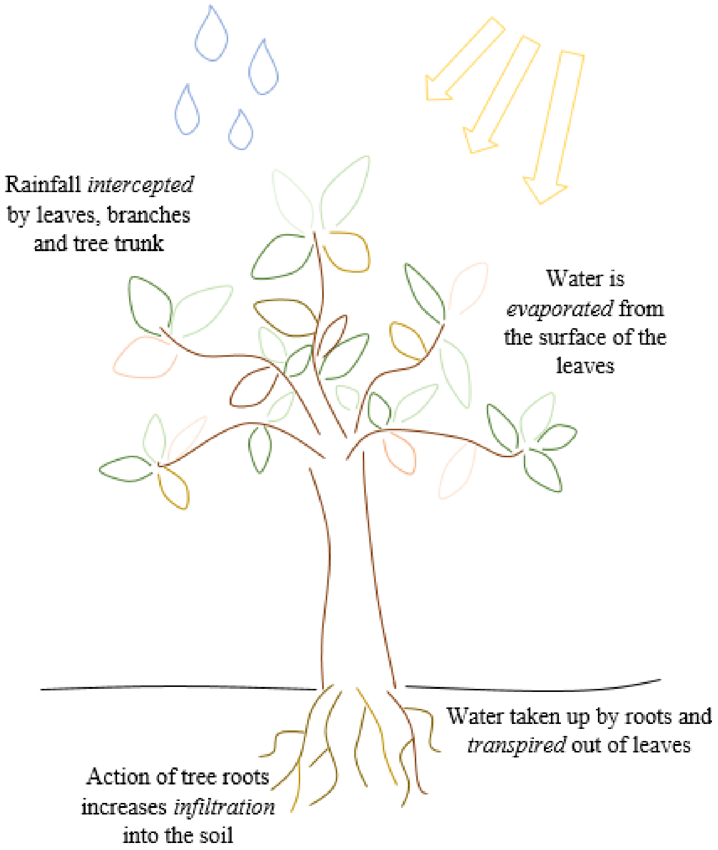

Figure 1.2 Key interactions between trees and the hydrological cycle.

1.4.2. Evapotranspiration

Some of the rainfall intercepted (Forestry Commission, 2018) will be evaporated from the canopy directly or instead drip from the canopy as through fall, run down the branches or trunk as stem flow (Beven, 2011). Rainfall which reaches the ground often infiltrates into the soil, where it is taken up by the roots. In most instances these rain droplets will make their way through the tree to the leaves where they will be transpired out through the stomata (Nisbet, 2005). Increased rates of transpiration in woodlands compared to other land covers facilitates the potential for higher water storage within woodland soils (Murphy et al., 2020).

1.4.3. Soil infiltration & storage

Infiltration rates are often higher in woodland soils when compared with other land cover types (McCulloch and Robinson, 1993; Carroll et al., 2006; Calder et al., 2008; Forestry

Commission, 2017; Mawdsley et al., 2017), as woodland soils commonly exhibit a more open structure as a result of increased organic matter and the action of tree roots (Nisbet and Thomas, 2006). This is commonly referred to as the ‘sponge effect’, as a more porous soil beneath the canopy increases the infiltration capacity and water storage availability, reducing the likelihood of over land flow. It also facilitates recharge to the underlying strata and groundwater, which ensures the continuation of the base flow during drier periods (McCulloch and Robinson, 1993). In recent research by Mawdsley et al. (2017) the ‘sponge effect’ has been shown to develop in a relatively short time-frame, 18 months after tree planting.

1.4.4. Woodland vs pasture

Woodland soils are typically more permeable when compared to pasture soils (Agnese et al, 2011; Archer et al, 2013; Marshall et al, 2009; Mawdsley et al, 2017). This is often attributed to a legacy of soil compaction from long-term over-grazing (Murphy et al., 2020). Grazing alters vegetation structure and composition (Milligan et al., 2016; Orr and Carling, 2006) and has left many pasture soils in poor condition with a loss of connectivity between near-surface and sub-surface macro-pores. This reduces soil water storage and increases runoff, contributing to downstream flood risk (Meyles et al., 2006). Reductions or complete removal of grazing has been found to reverse the long-term soil degradation, however the time frame needed for this to occur is debated. Marrs et al. (2018) and Marrs et al. (2020) suggest it may take 48–62 years however Holden et al. (2007) found that the removal of grazing after just 5 years is enough to allow pasture to recover. The aeration of pasture soils has also been found to reverse some of the impacts of grazing, increasing soil permeability and reducing the formation of overland flow (Wallace and Chappell., 2019).

Woodland creation is an NFM initiative often suggested as a viable soil recovery option for pasture systems especially in the UK uplands (Murphy et al., 2020). Investigations from the Pontbren catchment (Wales, UK) found rapid (<10 years), significant improvements to soil infiltration properties (Carroll et al, 2004; Marshall et al, 2014). Comparatively, Murphy et al., (2020) found a doubling in permeability and reduced surface soil compaction within 15 years of woodland establishment. In addition, individual tree planting within a pasture, rather than woodland creation, have been shown to increase soil permeability up to 13 m from the tree (Chandler and Chappell, 2008).

1.4.5. Conifer vs broadleaf woodlands

In the UK, there has been an historical bias towards investigations into coniferous plantation hydrology (Marshall et al., 2009) with relatively few studies made about semi-natural, broadleaf woodlands. This has resulted in low confidence in broadleaf woodlands as a flood mitigation method due to lack of available data (Burgess-Gamble et al., 2017).

The earliest known catchment study to compare runoff in a forested and un-forested catchment was carried out in the Emmental region in Switzerland in 1900 (McCulloch and Robinson, 1993). The forested catchment produced a lower flood runoff rate, however low flows in the summer appeared to be slightly higher (Engler, 1919). A number of similar investigations followed, well known, are the Wagon Wheel Gap, Coweeta and Hubbard Brook catchment studies in the USA (Warmerdam and Stricker, 2009). In the UK, Law (1956) was the first to suggest an increase in water usage by trees at the Stocks reservoir catchment in Lancashire (Johnson and Whitehead, 1993). Law (1956) found that runoff was 290 mm less in the catchment planted with coniferous trees in comparison with that of an adjacent grassland catchment. These results were originally regarded with scepticism, however they have been corroborated numerous times by other research catchment experiments such as: the Plynlimon experiment, located in Wales (Calder and Newson, 1979; Johnson and Whitehead, 1993; Archer, 2007); Balquhidder, Central Scotland (McCulloch and Robinson 1993) and Coalburn, Northern England (Robinson et al. 1998).

Native, semi-natural, broadleaf woodlands are likely to have different impact on hydrological processes. The most predominate difference is the ability for evergreen conifers to retain leaves all year allowing for year round interception from leaves, whilst broadleaf trees rely on interception by branches and trunks in winter months (Ahmad-Shah and Rieley, 1989; Nisbet and Broadmeadow, 2003; Roberts and Rosier, 2005). Additionally, broadleaf trees typically have deeper root systems and higher soil infiltration rates compared to conifers (Archer et al., 2013).

Human-influences on woodland management are also likely to alter hydrology, where drainage ditches and forest roads are more likely to occur for productive conifer plantations; these processes contribute to increases in downstream peak flows (Stratford, 2017; Bathurst et al., 2018). Furthermore, the occurrence of periodic felling in a productive conifer plantation reduces canopy cover and causes soil disturbance, contributing to an increase in localised flood risk (Nisbet and Thomas, 2006).

Recent studies having tended to focus on identifying the short-term impacts, as soon as 18-months (Mawdsley et al., 2017), of broadleaf woodland creation (Marshall et al., 2014). However, as the areas of woodland planted as part of the UK's 25 Year Environment Plan (DEFRA, 2018) mature, it will be increasingly important to understand how established upland broadleaf woodlands impact both soil properties and streamflow, in order to understand the potential for flood mitigation (Murphy et al., 2020).

1.4.6. Role of surface roughness

Often, investigations into the effectiveness of woodland for NFM have focused on the impact of land cover on soil properties such as infiltration, permeability and porosity (Archer et al., 2013; Murphy et al., 2020), with limited studies on surface roughness. Surface roughness is key to reducing water flow connectivity, therefore slowing the flow and reducing downstream flood peaks. The role of roughness has been well studied regarding channel and bank flow (Medeiros et al., 2012) however, there has been less of a focus on the roughness of vegetation.

1.4.7. Impact of scale

The influence of storm size on the effectiveness of woodlands as a method of NFM is an area of uncertainty. Investigations at the Plynlimon research catchments have found that flow peaks per unit area were smaller in the forested catchment for smaller storms (less than 20 % of mean annual flood) compared to the grassland catchment. During high flood flows there was no significant difference between the two catchments (Dadson et al. 2017). Similar results have been seen in Oregon (Beschta et al., 2000) and the Pyrenees (Gallart and Clotet, 1987; Gallart and Llorens, 2003) with woodlands providing a smaller reduction in peak flow for larger storms and larger catchments. Furthermore, in Coalburn reductions of 5-20 % in peak flows were recorded, however this reduction decreased with an increase in event size. Together these studies suggest that, whilst woodlands are an effective means of reducing peak flow, the mitigating potential becomes less with storm size.

Evidence supporting woodland as a flood mitigation method is greatest at plot and hillslope scale and for small catchments (<10 km^2) (Burgess-Gamble et al., 2017; Rogger et al., 2017). The impact of woodlands on flood peaks is less well understood for larger catchments, as it can become difficult to separate background effects. Furthermore, it is often not feasible to upscale the monitoring equipment required across larger catchments. Hydrological models can instead be used to scale-up woodland cover in larger catchments, using empirical data from smaller scale investigations and simulate streamflow response.

1.5. Catchment modelling

Model-based investigations make up a large number of studies investigating the effectiveness of land cover as a method of NFM (Burgess-Gamble et al., 2017; Black et al., 2021; Cooper et al., 2021). Modelling studies aid practitioners and policy makers in decision making as they are able to assess potential outcomes of multiple NFM interventions before they are implemented. Catchment models can be used to simulate a range of different spatial configurations of land cover in the same catchment (Gao et al., 2016) and predict a catchment's response to varying hydrological scenarios (Singh, 1995). The ability to simulate larger storm events which occur less frequently can be especially useful as they are much harder to document in real time.

Hydrological models usually require several kinds of data including; hydrometeorologic data, geomorphologic data, land cover data, pedologic data and hydrologic data (Hankin et al., 2016). Hydrological models can be classified in a number of ways for example, whether the model is physically or empirically based, or if the model is lumped, semi distributed or fully distributed. Examples of frequently used catchment models include; the HBV model, SWAT model and TOPMODEL. The HBV model is a lumped, conceptual model which gained popularity due to its low data requirements and significant potential for predicting streamflow in ungauged basins (Ouatiki et al., 2020). Comparatively, the SWAT model (Arnold et al., 1998) is a physically-based, semi-distributed hydrological model developed to predict the impact of land management practices over long periods of time on not only runoff, but also impacts to sediment and agricultural chemical yields (Eckhardt et al., 2005). Spatially distributed TOPMODEL (Gao et al., 2015), a fully distributed version of the model originally developed by Beven and Kirkby (1979), was specifically designed so that spatial options for land-cover change could be tested (Gao et al., 2016); factors such as soil infiltration and surface roughness can be fully spatially distributed and informed by empirical data collection. This makes it an ideal choice for investigating the effectiveness of woodland for NFM, modelling spatial configurations of land use as part of a mosaiced landscape.

1.6. Summary and motivation

Policy makers and scientists alike are keen to understand whether land cover, in particular woodlands, have an impact on river flows in a way that can adequately mitigate for future flood risk. Therefore, an increase in empirical data and observations is required. These data can inform parameters in rainfall-runoff models and allow translation of small, field or hillslope findings to cross-landscape investigations (Stratford, 2017).

1.7. Research aims and objectives

The aim of this research is to understand the impact of woodlands, in particular semi-natural, deciduous woodlands on flooding in the UK. To achieve this a number of approaches have been undertaken simultaneously:

i. the creation of a field study to investigate the impact of upland semi-natural woodland and pasture on a hillslope catchment scale,

ii. analysis of overland flow velocity in upland woodland and wood pasture using a novel approach,

iii. a UK-wide approach to identify relationships between catchment wide land cover and catchment storage, and

iv. the use of a rainfall-runoff model to explore the effects of woodland on streamflow within an upland catchment.

In order to accomplish this aim, four main research questions will be addressed in this thesis:

1. What is the impact of semi-natural woodland on soil hydrological properties?
2. How do upland woodland habitats affect overland flow velocity?
3. Does woodland cover within a catchment impact streamflow?
4. Can trends and patterns be identified between land cover and catchment storage?

This thesis will develop our understanding of the impact of woodlands on flooding in the UK.

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Chapter 2. The impact of semi-natural broadleaf woodland and pasture on soil properties and flood discharge

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2.1. Abstract

Woodlands can reduce the risk of rainfall-generated flooding through increased interception, soil infiltration and available storage. Despite growing evidence, there is still low confidence in using woodlands as a flood mitigation method due to limited empirical data, particularly for broadleaf woodlands. We measured soil properties and streamflow for 9 small (< 0.2 km^2) upland catchments and compared mature semi-natural broadleaf woodland where no stock grazing occurs to pasture with varied grazing intensity. We compared streamflow across 28 storm events including a 1 in 10-year event, two 1 in 4-year events and five 1 in 1.5-year events, identified over a 13-month period. We found that semi-natural broadleaf woodlands reduce specific peak discharge by 23-60 % and peak runoff coefficients by 30-60 % compared with pasture. Response to storm events took 14-50 % longer in woodland compared to pasture. These differences in flood response are partly explained by more permeable woodland soils, 11-20 times greater than pasture soil. The more muted response of wooded catchments to storm events is consistent across the storms investigated, including Storm Ciara, a 1 in 10-year event. Our analysis strengthens the argument that semi-natural woodlands can reduce rainfall-generated flooding contributing to the evidence base for natural flood management.

2.2. Introduction

Over the past three decades the frequency of flood events has increased across the UK (Rogger et al., 2017) and worldwide (Hall et al., 2014; Kundzewicz et al., 2014; Wingfield, Macdonald et al., 2019). In England, floods cause damages of £1.1 billion annually with 1 in 6 properties at risk from flooding (Priestley, 2017). This risk is expected to further increase under future climate change (Iacob et al., 2017).

Because of recent floods, there is a growing interest in the use of ‘soft-engineered’ flood mitigation schemes (Dadson et al., 2017; Stevens et al., 2016). Natural Flood Management (NFM), also referred to as Working with Natural Processes (WWNP) or nature-based solutions (Seddon et al., 2020), is an approach to flood management that seeks to work with natural processes to enhance the flood regulatory capacity of a catchment. Often these approaches also provide ecosystem services such as pollution assimilation, habitat creation and carbon storage (Hankin et al., 2017). NFM approaches may include the development of built water storage (Quinn et al., 2013; Nicholson et al., 2020), river restoration (Dixon et al., 2016), leaky debris dams (Ashbrook, 2020; Thomas and Nisbet, 2012) and land-use management (Spray et al., 2016).

Land-use management can influence the generation of overland flow, through hydrological processes such as interception, infiltration into soils, and available water storage, making it a potentially impactful NFM approach (Stratford et al., 2017). However, there is limited empirical data regarding the impact of land-use management as an effective NFM strategy (Burgess-Gamble et al., 2017). Furthermore, the size of a storm event is an important variable influencing the effectiveness of NFM (Archer, 2007; Beschta et al., 2000; Gallart and Clotet, 1987; Kirby et al., 1991).

In the UK and across northern Europe, NFM is often used in headwater catchments of the uplands, which receive high volumes of precipitation and are ideally placed for schemes which aim to 'slow the flow' of water down-slope (Bronstert et al., 2002; Marshall et al., 2009, Wheater and Evans, 2009). These areas are often dominated by grasslands used as permanent pasture (Marshall et al., 2009; Murphy et al., 2020), grazed by livestock, predominantly sheep. Grazing alters vegetation and can lead to soil compaction, loss of macro-porous soil structure and increased flood risk (Alaoui et al., 2018; Holden et al., 2007; Murphy et al., 2020; Palmer and Smith, 2013; Sansom, 1999). Exclusion of livestock has been observed to alter vegetation structure and soil structure, leading to an increase in infiltration rates and a reduction in surface runoff (Gifford and Hawkins, 1978; Greenwood et al., 1997; Marshall et al., 2009; Nguyen et al., 1998; Marshall et al., 2014).

Forested catchments have a different hydrological response compared to un-forested catchments due to greater interception, soil infiltration and available storage. Woodland soils typically have higher permeability rates than other vegetation types (Agnese et al., 2011; Archer et al., 2013; Mawdsley et al., 2017; McCulloch and Robinson, 1993; Zimmermann et al., 2006). This is attributed to a more open structure found in woodland soils as a result of increased organic matter and the action of tree roots (Nisbet and Thomas, 2006). Wooded catchments also have higher evapotranspiration and interception rates compared with other vegetation types, as trees are usually taller and have greater leaf area (Calder and Aylward, 2006; Nisbet, 2005). This means woodlands can produce lower annual runoff compared to other land cover types, as demonstrated in numerous catchment-based studies including Stocks Reservoir (Law, 1956), Plynlimon (Hudson et al., 1997; Kirby et al., 1991); Coalburn (Birkinshaw et al., 2014; Robinson, 1998) and Balquhidder (Johnson, 1995), depending on hydrological regime and climate (Brown et al., 2005; Farley et al., 2005; Zhang et al., 2017). In addition to altering annual runoff, woodland tends to reduce and delay flood peaks (Dadson et al., 2017; Stratford et al., 2017). However, the benefit of woodlands in providing smaller peak flows is typically less for larger storms and larger catchments (Archer, 2007; Beschta et al., 2000; Gallart and Clotet, 1987; Gallart and Llorens, 2003).

Historically, UK catchment-scale hydrological studies have investigated the influence of conifer afforestation (Marshall et al., 2009). UK forest cover increased from 5% in 1920 to 13% in 2020 (Forestry Commission, 2020), largely due to expansion of conifer plantations, which now account for 51% of UK woodland area (Forestry Commission, 2020). Relatively few UK studies have focused on broadleaf woodlands, which are the natural vegetation type in much of the UK. Broadleaf woodlands are likely to have different impact on hydrological processes. For example, evergreen conifers retain leaves all year and intercept 25-40 % of annual rainfall compared with 10-25 % for broadleaf deciduous woodland (Ahmad-Shah and Rieley, 1989; Nisbet and Broadmeadow, 2003, Roberts and Rosier, 2005). In addition, broadleaf trees typically have deeper root systems and higher soil infiltration rates compared with conifers (Archer et al., 2013). Differences in woodland management are also likely to alter hydrology, with the drainage ditches and forest roads that are more likely to be present in a productive conifer plantation, contributing to increases in downstream peak flows (Bathurst et al., 2018; Bathurst et al., 2020; Robinson et al., 2003; Stratford et al., 2017). Furthermore, the occurrence of periodic felling in a productive conifer plantation removes the canopy and causes soil disturbance which may contribute to an increase in localised flood risk (Nisbet and Thomas, 2006) and increased annual flows (Robinson and Dupeyrat, 2005). In a study focused on China, Tembata et al. (2020) found that broadleaf and mixed forests mitigate flooding, but conifer forests do not.

There have been few comparisons between upland permanent pasture and broadleaf woodland particularly studies that have measured both soil properties and streamflow response. Previous studies of broadleaf woodland creation (Marshall et al., 2014) have focused on the short-term impacts, as soon as 18-months after tree planting (Mawdsley et al., 2017). Research at Pontbren, one of the limited broadleaf woodland studies, found median soil infiltration rates were 67 times greater in newly (< 5 years) planted broadleaf woodlands compared with grazed pasture, with runoff volume reduced by 78% (Carroll et al., 2004; Marshall et al., 2014). As areas of newly planted broadleaf woodland mature, it will be increasingly important to understand how established broadleaf woodlands impact both soil properties and streamflow, to better understand the potential for flood mitigation (Murphy et al., 2020).

In this study we report results from a research catchment consisting of mature broadleaf woodland and grazed pasture in the UK uplands. The aims of the study were to:

1. Quantify the impact of pasture and mature semi-natural broadleaf woodland on soil properties.
2. Analyse streamflow response, including peak flow and runoff coefficient, of catchments dominated by pasture and mature semi-natural broadleaf woodland.

Our study is one of the first studies to investigate the impact of a mature broadleaf woodland in the UK, contributing to the evidence base around the benefits of broadleaf woodlands in the UK uplands.

2.3. Methods

2.3.1. Study area

This study took place around Haweswater reservoir (54°31'50.9"N, 2°45'37.3"W) in the Lake District National Park, UK (Figure 2.1). The land is owned by United Utilities and managed in partnership with the Royal Society for the Protection of Birds (RSPB) (RSBP, 2015). Elevations across the study area range from 243 to 720 m. The site experiences mild winters and cool summers (Kenworthy, 2014), with mean monthly temperatures ranging from -0.3°C to 18.3°C. Mean annual precipitation is 1779 mm, with monthly totals ranging from 88 to 231 mm (1981-2010 mean, derived from the Shap weather station at 255 m AoD) (Met Office, 2020). Average potential evapotranspiration (1961–2017) is 1.3 $mm \cdot d^{-1}$, with a summer average of 2.4 $mm \cdot d^{-1}$ and a winter average of 0.3 $mm \cdot d^{-1}$ (Robinson et al., 2020).

Soils in the study area are upland organo-mineral soils, predominately Malvern 611a (Chromic Endoleptic Umbrisol), a free draining acid loamy soil and Bangor 311e (Dystric Epileptic Histosol) soils, ordinarily described as very acid peaty soil underlined by igneous rock (Cranfield University, 2019). Land use includes semi-natural broadleaf woodland and unimproved permanent pasture grazed at a variety of densities. In recent years, United Utilities and RSPB have trialled a number of upland land management strategies, including tree planting, moorland drain blocking and changes to grazing (RSBP, 2015).

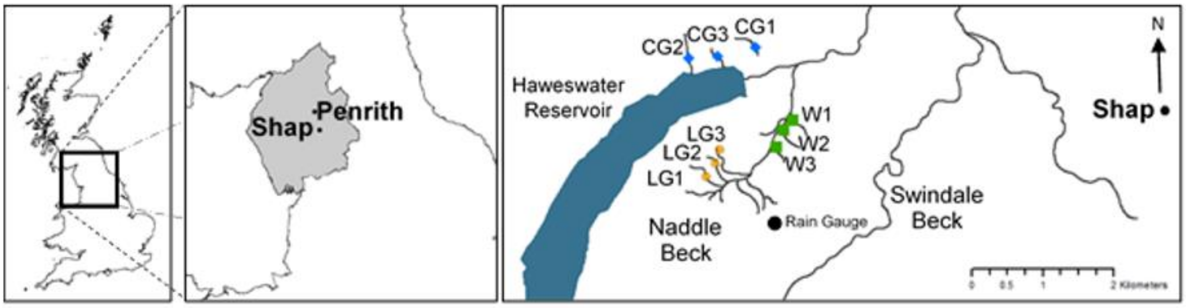

Figure 2.1 Map of field sites RSPB Haweswater within the UK and location of rain gauge and woodland (W), commons grazing (CG) and low-density grazing (LG) pasture.

Study design

We identified 9 small (< 0.2 km^2) catchments with different land covers but similar elevation, slope, geology and soil type (Table 2.1). We compared mature semi-natural woodland (W) and permanent pasture under either commons grazing (CG) or low-density grazing (LG). Woodland catchments consisted of mixed broadleaf species, predominantly oak, ash, alder, birch and hazel, with no stock grazing. Permanent pasture sites were unimproved (i.e., no drainage, ploughing or fertiliser application has occurred). Sheep grazing in CG occurs all year round at a maximum ewe intensity of 0.12 livestock units $LU \cdot ha^{-1}$, whereas grazing intensity in LG never exceeded 0.10 $LU \cdot ha^{-1}$, with no grazing in winter and scattered tree planting. Red and roe deer occurred at all sites.

Table 2.1 Summary information for the nine field catchments. Elevation (m) is recorded where streamflow measurements are taken. LU: Livestock Units.

Land-use		Field Site	Catchment Size (km^2)[a]	Average Elevation (m)	Average Slope (°)	Aspect	Location	Ground Vegetation Cover (Dominant species)	Land Management and Grazing intensity
Pasture	Commons Grazing	CG1	0.05	260	19.3	S-SE	54°32’19 N 2°45’58 W	*Deschampsia flexuosa, Digitalis sp., Galium saxatile, Molinia caerulea, Nardus stricta, Potentilla erecta, Pteridium aquilium, Ranunculus sp. Rubus sp., Sphagnum sp., Ulex europaeus*	All year round grazing at a maximum intensity of 0.12 LU·ha^{-1}.
		CG2	0.08	260	20.2	SE-E	54°32’90 N 2°47’40 W		
		CG3	0.14	270	16.0	S	54°32’14 N 2°46’44 W		
	Low-Density Grazing	LG1	0.10	360	12.0	SE	54°31’12 N 2°46’12 W	*Carduus sp., Cirsium vulgare, Deschampsia flexuosa, Equisetum arvense, Juncus sp, Molinia caerulea, Nardus stricta, Potentilla erecta, Pteridium aquilium, Ranunculus sp., Sphagnum sp., Veronica sp.*	Maximum grazing intensity of 0.10 LU·ha^{-1} with no stock grazing from 1st November - 30th April.
		LG2	0.12	370	8.5	SE-E	54°31’20 N 2°46’39 W		
		LG3	0.11	390	4.6	SW-W	54°31’30 N 2°46’28 W		
Woodland		W1	0.06	280	21.8	NW	54°31’33 N 2°45’39 W	*Deschampsia flexuosa, Euphrasia sp., Mercurialis perennis, Molinia caerulea,Nardus stricta, Pteridium aquilium, Sphagnum sp.,Trifolium repens*	Semi-natural upland woodland (NVC classification W7, W9, W11 – upland mixed woodland and wet woodland) designated as a site of special scientific interest (SSSI). The woodlands are fenced to exclude livestock.
		W2	0.05	310	24.0	N-NW	54°31’30 N 2°45’44 W		
		W3	0.03	270	17.7	NW	54°31’40 N 2°45’33 W		

[a] Calculated as the median of the estimate from OS (Digimap, 2021) and the water balance method (December data assuming zero evapotranspiration).

2.3.3. Soil properties

Soil properties were analysed on a monthly basis during a 12-month period (July 2018 - July 2019) and sampled randomly across each catchment. Soil cores (n=30) were taken at 0-5 cm depth, just below the vegetation layer using Eijelkamp soil sample rings. Top-soil permeability (saturated hydraulic conductivity, Kfs or Ksat) was measured in an Eijkelkamp 25 place Permeameter, from the collected soil cores. Subsoil permeability (Kfs or Ksat, 0.15 m depth) was determined in-field using a constant head well permeameter (CHWP, Engineering Technologies Canada Ltd. (ETC) pask, n=13) (Elrick and Reynolds, 1986; Reynolds, 2008). A pre-wetting phase was included to reduce the time to reach steady state flow and ensure saturation. The Eijelkamp Permeameter was unsuitable for the subsoil measurements due to the rocky nature of the ground. Bulk density (ρ, $g \cdot cm^{-3}$) was calculated after oven drying (105° for a minimum of 16 hours) the soil cores to constant weight. Soil moisture content was measured using a Delta-T Ltd 'theta probe' (n=225). The 'theta probe' uses a simplified Time-Domain Reflectometry (TDR) technique to derive values of volumetric moisture content (Delta-T, 1999).

2.3.4. Hydrological monitoring

Hydrology was monitored over a 13-month period (January 2019 - February 2020). A 90-degree v-notch weir was established within each catchment, with a pressure transducer installed to collect stream depth data every 5 minutes (see Appendix A.1 for details). Flow was calculated using the Kindsvater-Shen equation (Appendix A.2). Locations for data collection within the streams were based on suitability of the channel bed; approximately 1.5 m between channel banks and accessibility for monthly equipment checks. Rainfall data (5 min resolution) was collected using a HOBO RG3 data logging tipping bucket rain gauge (Figure 2.1).

2.3.5. Storm response

Storm events were defined when more than 20 mm of rain occurred during a 24-hr period. The end of the event was defined as 6 hours with no rain. During a 13-month period (January 2019 - February 2020), 28 storms were identified including both winter and summer time storms (Appendix A.3). Storm durations ranged from 9.5 hrs to 96.25 hrs. Storm intensity, defined as total rainfall divided by storm duration, ranged from 0.62 to 5.3 $mm \cdot hr^{-1}$.

We used rainfall data from Wet Sleddale, 8 km from the site, to establish storm return periods as there was insufficient data available from the nearest rain gauge (Naddle). Rainfall at Wet

Sleddale was highly correlated with records from Naddle (see Appendix A.4). An IDF (intensity-duration-frequency) curve was used to calculate return periods.

Catchment response to each storm was analysed in four ways: the specific peak discharge, peak runoff coefficient, volume runoff coefficient and time to flow response. The specific peak discharge (SPD, $mm \cdot hr^{-1}$) indicates the highest discharge during the storm event with the influence of catchment area removed (Appendix A.5). Both the peak runoff coefficient and volume runoff coefficient are dimensionless coefficients relating the amount of runoff to the amount of precipitation received and useful for catchment comparisons to understand how different landscapes impact runoff (Young et al., 2009). A larger runoff coefficient can indicate a catchment with lower infiltration rates that is more susceptible to flooding, whilst a smaller value suggests a more permeable catchment. Peak runoff coefficient (C) is calculated by dividing the peak rate of runoff by the maximum rainfall intensity (Appendix A.5). To determine the volume runoff coefficient (V) for each storm, baseflow was removed from the storm hydrograph and the total storm runoff calculated. The volume runoff coefficient was then calculated as the ratio of total storm runoff to total storm rainfall (Merz et al., 2006). The time to flow response is the time, in hours, between the initiation of rainfall and a significant water level rise (Gonzalez-Sosa et al., 2016), defined in our analysis as three times streamflow at the start of the event.

2.3.6. Influence of storm size and seasonality

We analysed the impact of storm size by two different methods. Firstly, we used the classification of Kirby et al. (1991) with smaller storms classified as discharge peaks < 1 $mm \cdot hr^{-1}$ and larger storms as discharge peaks > 1 $mm \cdot hr^{-1}$, allowing for direct comparison with previous studies (Kirkby et al., 1991; Meyles et al., 2003). Secondly, we divided storms by storm return periods. Storms were divided into those with a return period less than 1.5 years and those with a return period longer than 1.5 years, with this threshold reflecting the average reoccurrence of bank full stage (Wolman and Miller, 1960). Our storm return periods are based on rainfall data, likely leading to longer return periods compared to return periods based on discharge data. There were insufficient large storm events in our study period to fully characterise the impact of land cover as a function of storm size.

We also identified whether the storms occurred during winter or summer and re-calculated the mean SPD, peak runoff coefficient, volume runoff coefficient and time to flow response. Finally we calculated the cumulative sum of flow above two different thresholds (1 and 2 mm·hr^{-1}) between 03.03.2019 and 17.03.2019, a period when data was available at all sites and evapotranspiration can be assumed to be minimal.

2.3.6.1. Hydrograph form and flashiness

We identified the 97.5 % flow threshold and analysed each consecutive period of higher flow. The hydrograph within each of these peak periods was normalised relative to the peak flow in the period, allowing us to compare the relative rates of rise and fall around the peak, thereby providing an indication of the flashiness of the response. This procedure was followed for each measurement site. Since the sites are all close together (within 3.5 km), the incidence of storm events was considered comparable, revealing the inherent differences in flashiness between the sites.

2.3.7. Statistical analysis

Shapiro–Wilks tests were employed to deduce normality of soil properties and storm responses. Non-parametric Kruskal–Wallis and post-hoc tests were used to determine a significant difference (significance determined at $p < 0.05$) between land covers (significant differences between individual catchments can be found in Appendix A.6 and A.8). Statistics were performed using the Python SciPy (Virtanen et al., 2019) and scikit-posthocs (Terpilowski, 2019) packages.

2.4. Results

2.4.1. Soil properties

Woodland sites had significantly ($p < 0.05$) higher topsoil permeability compared with the pasture sites. Median topsoil permeability was 11 times higher in the woodland sites compared with CG sites, and 20 times higher than the LG sites (Figure 2.2a). There was no significant ($p > 0.05$) difference between subsoil permeability at the different sites (Figure 2.2b). The highest mean soil moisture occurred at the woodland sites (49 %), compared with LG (46 %) and CG (33 %) with significant ($p < 0.05$) differences between the soil moisture at the different sites (Figure 2.2c). The lowest bulk density soils were measured at the LG sites (0.36 g·cm^{-3}) compared with CG (0.46 g·cm^{-3}) and W sites (0.50 g·cm^{-3}) (Figure 2.2d). Details of the measurements for the nine catchments are given in Appendix A.6 and A.7.

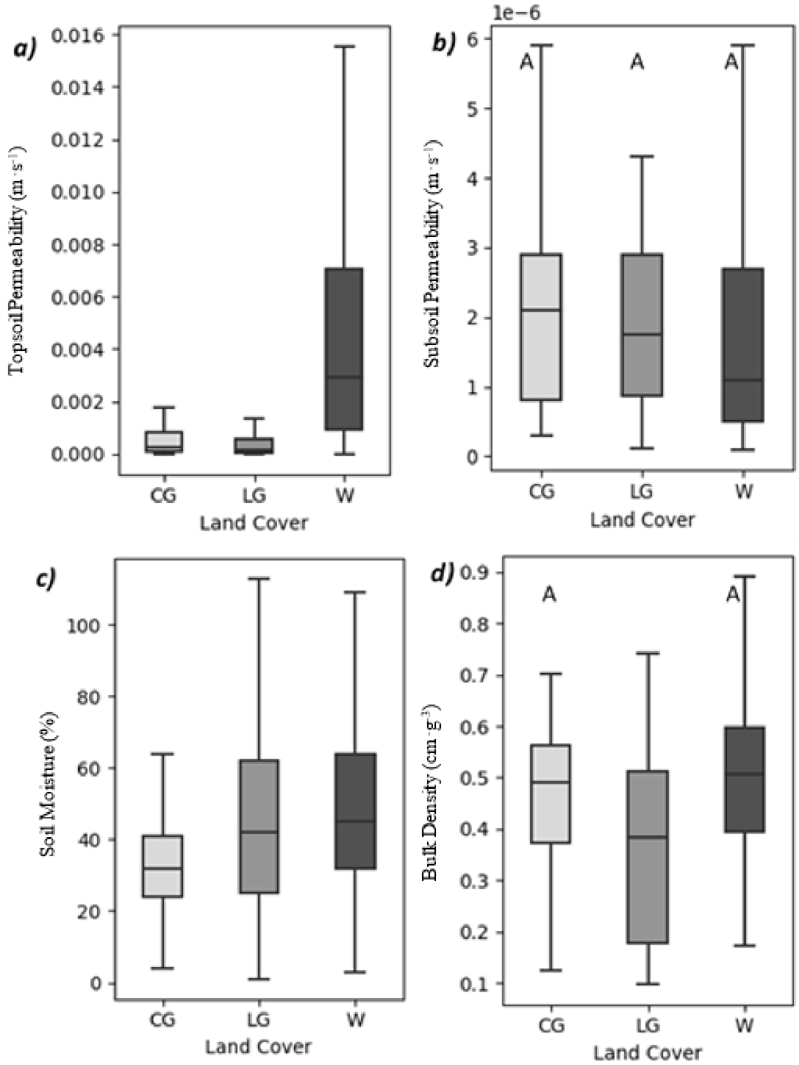

Figure 2.2 Distribution of a) topsoil permeability ($m \cdot s^{-1}$), b) subsoil permeability ($m \cdot s^{-1}$), c) soil moisture (%), d) bulk density ($g \cdot cm^{-3}$) for woodland (W), commons grazing (CG) and low-density grazing (LG) pasture shown as median (line), 25^{th} to 75^{th} percentile (box), and 5^{th} to 95^{th} percentile (whiskers). Sites which are not statistically different share a letter.

2.4.2. Storm response

Figure 2.3 compares SPD, runoff coefficients and time to flow response across all the storms analysed. SPD and peak runoff coefficient were significantly lower at woodland sites compared with CG and LG sites (p <0.05). Woodland sites median SPD was 23 % less than CG and 60 % less than LG sites (Figure 2.3a). Woodland sites had a peak runoff coefficient 30 % less compared with CG sites and 60 % less LG sites (Figure 2.3b). Volume runoff coefficients were not significantly different between land covers (Figure 2.3c). The median time to flow response in woodland was 14 % longer than CG sites and 50 % longer compared with LG sites (Figure 2.3d). Woodland sites had significantly different time taken to flow response compared with the LG ($p < 0.05$), but not CG sites. Details of the measurements for the nine catchments are given in Appendix A.8 and A.9.

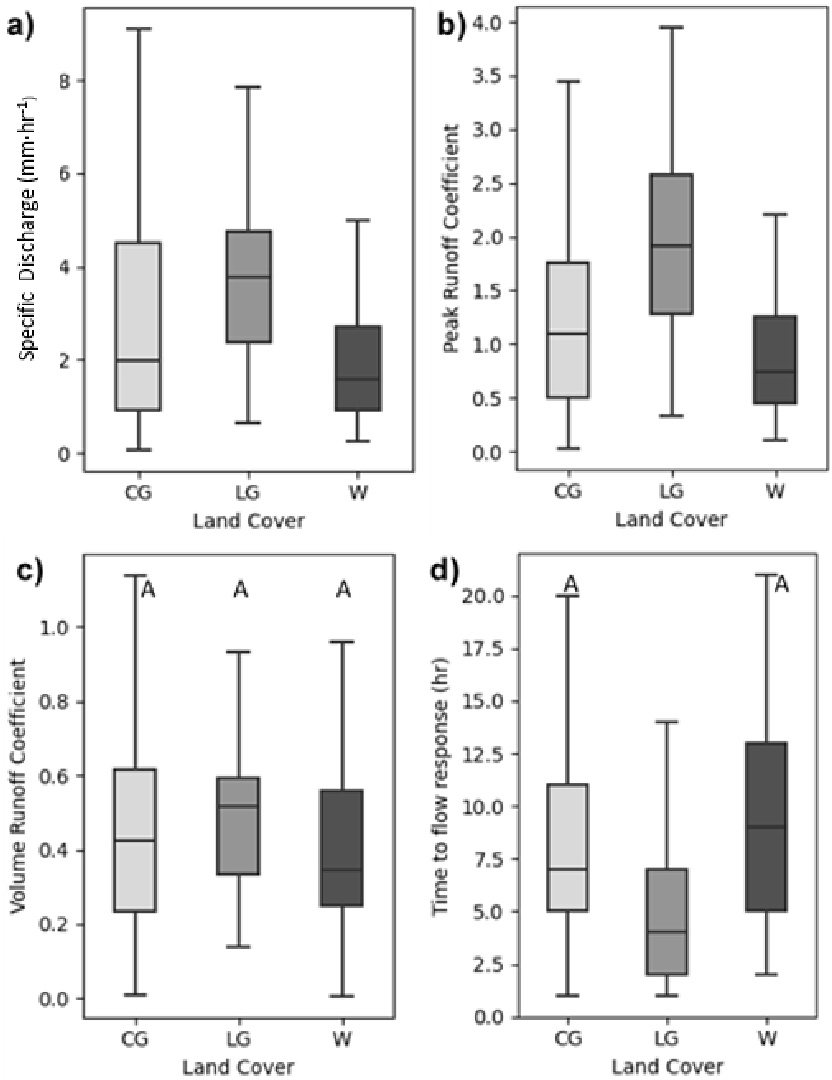

Figure 2.3 Distribution of a) specific peak discharge (mm·hr^{-1}), b) peak runoff coefficient, c) volume runoff coefficient, d) time to flow response (hr) for woodland (W), commons grazing (CG) and low-density grazing (LG) pasture shown as median (line), 25th to 75th percentile (box), and 5th to 95th (whiskers). Sites which are not statistically different share a letter.

2.4.2.1. Influence of storm size and seasonality

We identified 28 storms during our analysis period; including a 1 in 10-year storm event, two 1 in 4-year storm events and five 1 in 1.5-year events (Figure 2.4a). Two of these storms met requirements to be named by the UK Met Office, Storm Ciara (1 in 10-year event) and Storm Dennis (1 in 4-year event) (Parry et al., 2020). Both storms displayed characteristics of storms with longer return periods (Storm Ciara up to a 1 in 50-year event and Storm Dennis a 1 in 10-year event) (Figure 2.4b).

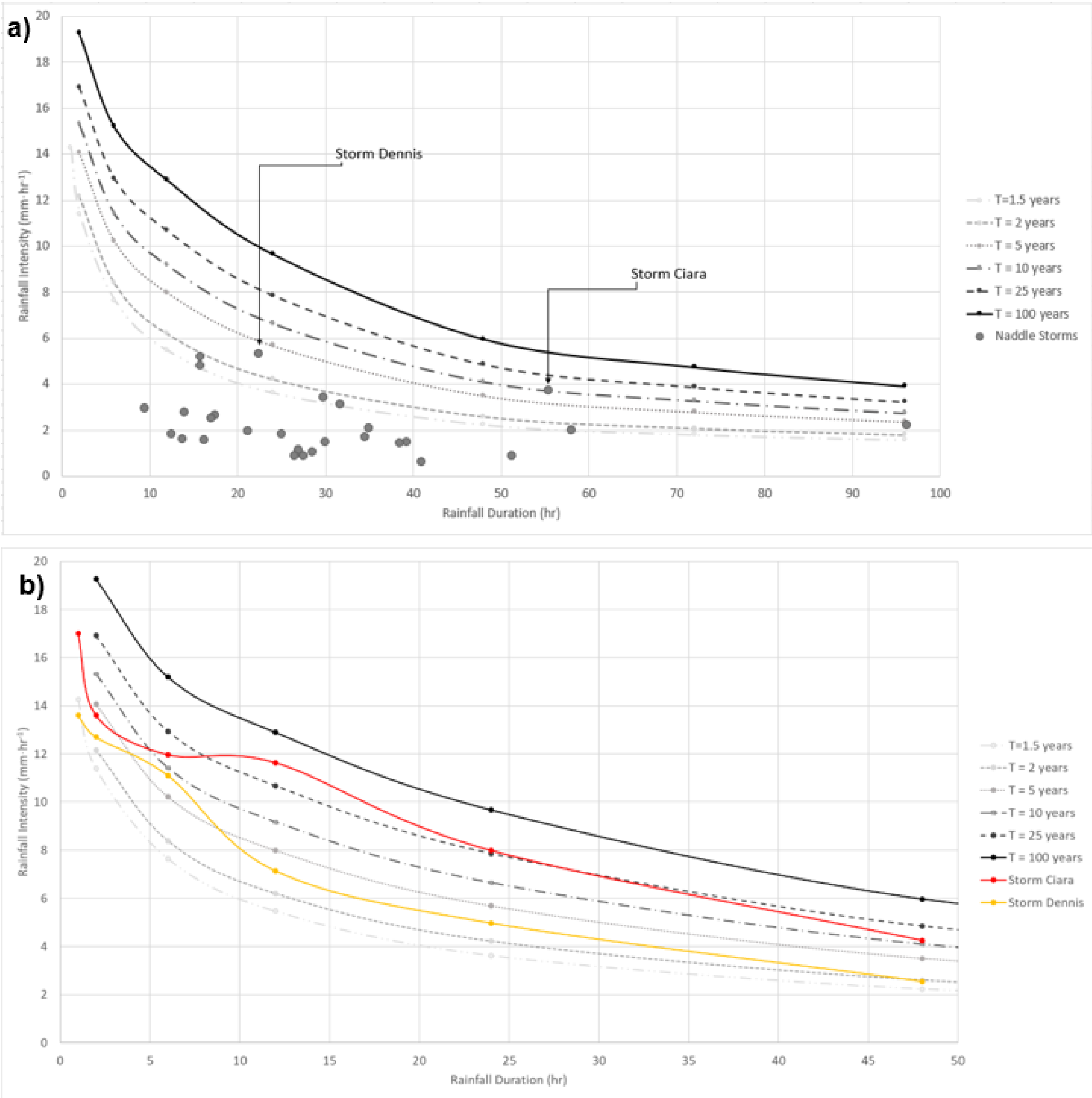

Figure 2.4 Intensity-Duration-Frequency (IDF) curve of Wet Sleddale rainfall data from 1997-2021 for return periods, T = 1.5, 2, 5, 10, 25, 100 years. a) 28 storms identified at the Naddle rain gauge overlaid with grey circles. b) UK Met Office named storms, Ciara and Dennis overlaid to show the range of storm intensities throughout each event.

We divided storms into those with a return period more than 1.5 years (n = 8 storms) and storms with a return period less than 1.5 years (n = 20 storms). For storms with a return period more than 1.5 years, woodlands exhibited significantly different ($p < 0.05$) SPD (Figure 2.5a) and peak runoff coefficient (Figure 2.5b) compared with pasture: median SPD was 53 % lower than for CG sites and 58 % lower than LG sites, peak runoff coefficient was 48 % lower than CG sites and 58 % lower than LG sites. For storms with a return period more than 1.5 years, the median volume runoff coefficient for woodland sites was 26 % lower than CG sites and 41 % lower than LG sites ($p < 0.05$) (Figure 2.5c). Woodland catchments also exhibited lower SPD and peak runoff coefficients than pasture during Storm Ciara (Appendix A.10 and Appendix A.11).

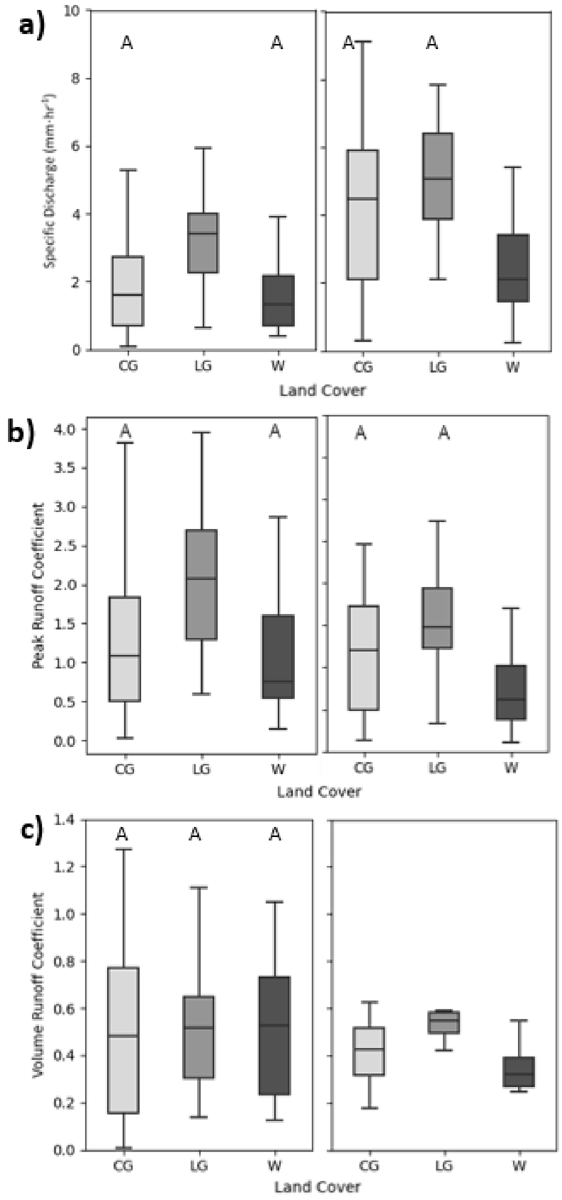

Figure 2.5 Comparison of streamflow for storms with return periods of less than (left hand panels) and more than (right hand panels) 1.5 years for woodland (W), commons grazing (CG) and low-density grazing (LG) pasture. a) Specific peak discharge ($mm \cdot hr^{-1}$), b) Peak runoff coefficient), c) Volume runoff coefficient shown as median (line), 25^{th} to 75^{th} percentile (box), 5^{th} to 95^{th} percentile (whiskers). Sites which are not statistically different share a letter. Appendix A.10 reports tabulated data.

Using the same storm classification as Kirby et al. (1991), we found woodland exhibited a significantly lower mean SPD (2.35 $mm \cdot hr^{-1}$) compared with pasture (3.67 $mm \cdot hr^{-1}$) for larger storms. However, there was no significant difference in mean SPD between woodland (0.68 $mm \cdot hr^{-1}$) and pasture (0.51 $mm \cdot hr^{-1}$) for smaller storms. We found woodlands had lower SPD and runoff coefficients compared with pasture in both summer and winter, with the largest differences in winter (Table 2.2). We found the cumulative flow above a certain threshold during a 14-day period in winter was lower at the woodland sites compared with pasture (Table 2.3).

Table 2.2 Summer and winter streamflow properties for Woodland and Pasture (commons grazing (CG) and low-density grazing (LG) combined due to data availability).

	Specific peak discharge (SPD) ($mm \cdot hr^{-1}$)				Peak runoff coefficient				Volume runoff coefficient				Time to flow response (hr)			
	Summer		Winter		Summer		Winter		Summer		Winter		Summer		Winter	
	n	μ	n	μ	n	μ	n	μ	n	μ	n	μ	n	μ	n	μ
Woodland	12	0.0013	43	0.0019	12	0.80	42	0.89	10	0.50	37	0.42	12	9	45	10
Pasture	36	0.0020	103	0.0035	36	1.15	99	1.64	40	0.37	83	0.61	36	6	106	8

Table 2.3 Cumulative sum of flow above 1 $mm \cdot hr^{-1}$ and 2 $mm \cdot hr^{-1}$ flow thresholds.

Land cover	**Cumulative flow (mm)**	
	1 $mm \cdot hr^{-1}$	2 $mm \cdot hr^{-1}$
Commons Grazing	283	102
Low-density Grazing	270	189
Woodland	131	21

2.4.2.2. Hydrograph form and flashiness

Figure 2.6 shows the medians of normalised hydrograph peaks for all storms exceeding the 97.5% frequency threshold. These medians are derived from individual storm data for each site, shown in full in Appendix A.12. Steeper rising and falling limbs indicate a flashier response, generally associated with more severe flooding from storm rainfall. It is evident that woodland sites are the least flashy, and pasture sites (particularly low density pasture) the most flashy.

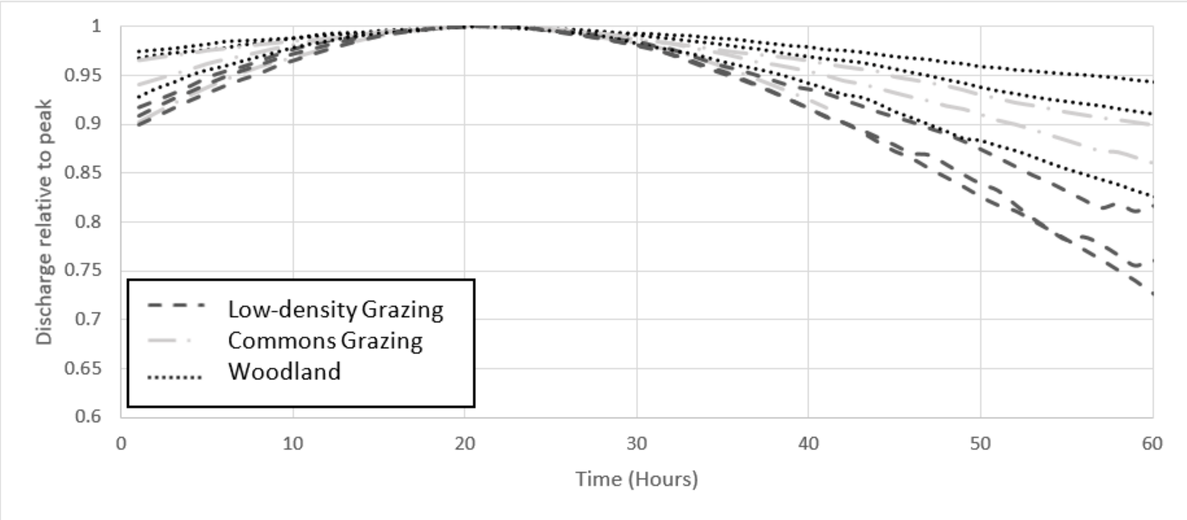

Figure 2.6 Median form of hydrograph peaks for all events exceeding the 97.5% threshold. Data normalised to 100% for the peak flow. Data from all storms shown in Appendix A.12.

2.5. Discussion

Our study provides some of the first information of the impacts of mature semi-natural broadleaf woodlands in the UK on streamflow in small (< 0.2 km^2) catchments. The lower specific peak flow, lower runoff coefficient and longer response time of mature semi-natural broadleaf woodlands compared with pasture will contribute to reduced peak flow downstream. In contrast to some previous studies, we found mature broadleaf woodland can reduce peak flow for larger storms (> 1 mm·hr^{-1}) and for storms with > 1.5-year return periods. Together this demonstrates the effectiveness of mature semi-natural broadleaf woodlands as a NFM method.

2.5.1. Comparison of streamflow response in semi-natural woodland and wood pasture

Across all the storms analysed, we found that woodland sites typically had lower SPD, peak and volume runoff compared with grazed pasture sites by 23-60 %, 30-60 % and 21-35% respectively. Peak runoff coefficients can be strongly influenced by characteristics of the storm event (Figure 2.5b), but in our analysis show consistent behaviour with other streamflow metrics. Sriwongsitanon and Taesombat (2011) reported lower runoff coefficients for a forested area in comparison with an agricultural area. We found that the average time taken for flow to respond to storm events was 14-50 % longer in the woodland compared with pasture sites. Lana-Renault et al. (2011) found a forested catchment took 171 % longer to respond than a formerly agricultural catchment, subsequently left to naturally regenerate.

As in previous studies (e.g., Carroll et al., 2004; Chandler et al., 2018; Mawdsley et al., 2017; Wahl et al., 2003) we compared small catchments which are as similar as possible in all respects except land cover and assume differences in land cover drive difference in hydrological response (McCulloch and Robinson, 1993). Flow response is dependent on antecedent conditions, which will be similar for our sites as they are closely located to each other. We note that it is not possible to identify catchments that are identical in all aspects, so it remains possible that catchment differences drive some of the observed differences in flow response (Lana-Renault et al., 2011; López-Ramírez et al., 2020). Establishing measurements in multiple small catchments (here 3 woodland and 6 pasture catchments) will help to reduce these uncertainties. Future work is needed to track changes in soil properties and streamflow as newly established broadleaf woodlands mature.

2.5.2. Impact of storm size

Previous studies have reported that forests reduce peak discharge during small flood events but not always during larger events (Dadson et al., 2017, Stratford et al., 2017). For example, Kirby et al. (1991) showed lower peak flows in a wooded catchment compared with a grassland catchment during smaller storms (discharge peaks < 1 $mm \cdot hr^{-1}$), but little difference during larger storms (discharge peaks > 1 $mm \cdot hr^{-1}$). Using the same storm classification, we found woodland exhibited a lower mean SPD (2.35 $mm \cdot hr^{-1}$) compared with pasture (3.67 $mm \cdot hr^{-1}$) for larger storms.

We also explored whether land use had different impacts for different storm return periods. Woodlands had a median volume runoff coefficient that was 26-41 % lower and peak runoff coefficient that was 48-58 % lower than pasture for storms with a return period greater than 1.5 years. Our study focused on mature, semi-natural woodlands consisting of native broadleaf tree species without any drainage. In contrast, most previous UK studies have focused on conifer plantation with drains established prior to afforestation (Dadson et al., 2017; Kirby et al., 1991; Stratford et al., 2017) which may partly explain the difference in response to larger storms. Tembata et al (2020) confirms that forest type is important, with broadleaf and mixed forests mitigating flooding whereas conifer forests did not.

Importantly, we found that the response to the largest storm event recorded in our study period, Storm Ciara, remained consistent with the response to other storm events. SPD, peak and volume runoff coefficients were lower in the wooded catchments compared to pasture. The hydrograph response shows the wooded catchments were less flashy during storm Ciara with a slower rising and falling hydrograph with a smaller and later peak (Appendix A.11). However, the impact of land cover on storm response during more extreme storm events, e.g. 100 year return period, remain unquantified and are likely to show lesser effects than those demonstrated here. Longer data records are needed to capture such large storm events and allow for a more detailed analysis of the impacts of land cover as a function of storm size.

2.5.3. Seasonal differences in flood response

We analysed the impacts of land cover during both winter and summertime storms (Table 2.2). Woodlands had lower SPD and runoff coefficients compared with pasture in both summer and winter, with the largest differences in winter. We found the cumulative flow above a certain threshold during a 14-day period in winter was lower at the woodland sites compared with pasture (Table 2.3). An increase in heavy wintertime rainfall across Northern England in recent decades highlights the need for flood management during winter months (Burt and Ferranti, 2012, Orr and Carling, 2006).

2.5.4. Soil properties

The difference in streamflow response between woodland and pasture sites, particularly in winter when differences in evapotranspiration will be more limited (Blyth et al., 2019; Robinson et al., 2020), can in part be explained by differences in their soil properties. Lower peak flows, lower runoff coefficients and longer times to flow response in woodland sites all indicate a more permeable catchment. This is confirmed by differences in topsoil permeability with our woodland sites having a median topsoil permeability 11-20 times greater than the pasture sites. Previous studies have also found woodland catchments to have more permeable soils, with topsoil (< 20 cm) permeability 1.8-8 times greater than that of grazed permanent pasture (Table 2.4). The median topsoil permeability we measured for pasture sites (1.47×10^{-4} - 2.78×10^{-4} $m \cdot s^{-1}$; 529 - 1000 $mm \cdot hr^{-1}$) overlap previously reported values for pasture and field margins in Northern England: Wallace and Chappell (2019) reported median topsoil permeability of 21 – 2794 $mm \cdot hr^{-1}$ whereas Wallace et al., (2021) reported 317 – 8,780 $mm \cdot hr^{-1}$. Hedgerows can also increase permeability, with topsoil permeability 20-30 times higher than pasture (Wallace and Chappell, 2021; Holden et al., 2019). Individual trees within pasture have been shown to increase soil permeability up to 13 m from the tree (Chandler and Chappell, 2008), though we did not observe this effect at our LG pasture sites.

Table 2.4 Ratio between permeability (Kfs) of woodlands and grazed soils , comparing data from previous studies.

Reference	Vegetation	Ratio of Kfs woodland compared to grazed land	Depth (cm)
Agnese et al. (2011)	40-50 year-old broadleaf	3.4	10-20
Archer et al. (2013)	180 year-old broadleaf 500 year-old broadleaf	6 5	4-15 4-15
Mawdsley et al. (2017)	18 month-old saplings	2.3	10-30
Marshall et al., (2009)	7 year-old broadleaf	2.4	18-30
Murphy et al. (2020)	7-15 year-old broadleaf	1.8	6
Zimmermann et al. (2006)	Tropical forest	4 8	12.5 20
López-Ramirez et al. (2020)	Tropical montane	4.8	20
This study	Mature broadleaf	11-20	5

A range of mechanisms have been proposed to explain the greater permeability of woodland and hedgerow soils compared with pasture. The root networks of trees and shrubs can generate macropores within the soil matrix that enhance permeability (Chandler and Chappell, 2008; Wallace et al., 2021). The lower permeability of pasture soils can be due to topsoil compaction caused by livestock grazing (Carroll et al., 2004). Our pasture sites were only lightly grazed and we did not find pasture soils had higher bulk density that would be consistent with compaction. Wallace and Chappell (2019) found that aeration of pasture soils can increase saturated hydraulic conductivity and reduce overland flow. The lower permeability of pasture soils is known to increase runoff and contribute to downstream flooding (Alaoui et al., 2018). Conifer forests soils can have lower permeability compared with both broadleaf woodland and permanent pasture (Chappell et al., 1996; Gonzalez-Sosa et al., 2010), contributing to greater overland flow (Tembata et al., 2020). Many previous studies also found higher subsoil permeability in woodland soils, whereas we found no significant difference in soil permeability at 15 cm depth between woodland and pasture soils. This is likely due to the relatively thin soils in our upland sites, with the action of tree roots in the development of open pores more limited below 15 cm.

We found lower density grazing (LG) pasture and woodland sites had significantly higher soil moisture when compared with commons grazing (CG) pasture sites. The sparse tree planting in the LG may have contributed to the higher soil moisture in this area. Mawdsley et al. (2017) found tree planting can increase soil moisture within 18 months. Furthermore, higher levels of soil moisture are often attributed to lower levels of grazing (Xu et al., 2014).

Wallace and Chappell (2020) found that application of fertiliser and slurry to agriculturally improved pasture resulted in reduced summer soil moisture but increased autumn soil moisture potentially increasing downstream flood risk.

Some previous studies have found woodland soils to have 10-30 % lower bulk density than other vegetation types (Agnese et al., 2011; Sharrow, 2007; Wahren, 2009). In contrast, Upson et al., (2016) found woodland soils had greater bulk density compared with pasture. We found woodlands exhibited the highest bulk density values, with a significant difference between woodland soils and LG soils. Our pasture sites were lightly grazed, possibly explaining the lack of compaction and lower bulk density.

In our study, livestock grazed the pasture sites whereas the woodland sites did not have any livestock grazing. The number of sheep in the UK has changed substantially in recent decades, increasing from 19.7 million in 1950 to 44.5 million in 1990 (Fuller and Gough, 1999), before declining around the turn of the century to 33.5 million in 2019 (DEFRA, 2020). Sheep numbers in the nearby Lune catchment (Cumbria) increased by a factor of five from 100,000 in 1860 to 500,000 in 1990 (Orr and Carling, 2006). These large changes in livestock numbers are likely to have caused substantial impacts (O'Connell et al., 2007). Stock grazing changes vegetation structure and composition (Milligan et al., 2016; Orr and Carling, 2006) and can lead to soil compaction, a reduction in soil permeability (Alaoui et al., 2018) and soil water storage (Meyles et al., 2006). Loss of vegetation and soil compaction can increase flood risk, with simulated flood peaks in a UK upland catchment increased by 33 % under light grazing and 82 % under heavy grazing (Gao et al., 2017). The lower grazing levels found in our LG sites would be anticipated to lead to higher soil permeability and lower peak flow compared with CG sites. In contrast, we found lower rates of permeability and higher SPD and runoff coefficients at the LG compared with CG sites. Overall grazing pressures across both the CG and LG pasture sites in our study were relatively similar at around 0.1 livestock unit per hectare (Table 2.1), with the main difference being less wintertime grazing in the LG sites. Our study does not provide any information on the impacts of higher grazing pressure that is found across much of the UK uplands, exceeding 4 sheep per hectare in some locations (Orr and Carling, 2006). Variability in grazing pressure within a site can result in areas favoured for grazing experiencing more compaction (Orr and Carling, 2006) reducing the downward mixing of organic material, decreasing permeability. The recovery of vegetation after a reduction in grazing should reduce rates of overland flow

(Bond et al., 2020), with impacts on downstream flooding. In our study reduced grazing was introduced fairly recently (~ 7 years ago) and whilst relatively little is known about the effects of reducing stock grazing pressures, it may take 48-62 years to see the benefits of reduced grazing due to the long-term soil degradation caused by intensive sheep-grazing and slow rates of recovery (Marrs et al., 2020; Marrs et al., 2018).

2.5.5. Implications for policy

Our analysis demonstrates the importance of semi-natural broadleaf woodlands in modifying soil properties and reducing flood peaks in small (< 0.2 km^2) catchments, even for larger storm events. A key challenge remains to assess the impacts of woodland at larger scales (Dadson et al., 2017). Data collected here can be used to inform parameter choice in flood prediction models, which can then be used to upscale results to understand the impacts of semi-natural woodlands on downstream flooding in larger catchments (Gao et al., 2017, Jackson et al., 2008, McIntyre et al., 2013). Our analysis includes a number of large storms, including two storms that met the UK Met Office criteria for a named storm (Storm Dennis and Storm Ciara). Storm Ciara resulted in widespread disruption and flooding within Northern England, so our results are relevant for flood risk management. However, the smaller number of recorded peak flows with increasing peak size makes it difficult to demonstrate a statistically significant change in flow response for the largest events (Burgess-Gamble et al., 2017).

We show that semi-natural broadleaf woodland has more permeable soils resulting in lower peak flood discharge compared with pasture grazed by sheep, the dominant land use in much of the UK uplands. All the pasture sites in our study had relatively low grazing intensity (~0.1 sheep / hectare) - the difference in soil permeability and streamflow between woodland and pasture may be even greater for pasture with the higher grazing intensity more typical of the UK uplands (DEFRA, 2020). Our study suggests that restoring or converting upland pasture to semi-natural woodland would help reduce downstream flood risk. Previous studies have found that soil permeability can increase rapidly after tree planting (Mawdsley et al., 2017) so the benefits to reduced flooding could be realised quickly. In contrast, reductions in grazing without tree planting may result in relatively slow changes in soil properties suggesting tree planting may be necessary in many locations if rapid changes are needed.

In the UK, agricultural subsidies have supported upland sheep farming in recent decades (Hardaker, 2018). Planned changes in agricultural subsidy and the need to mitigate climate change and reach net-zero carbon emissions (Paris Agreement, 2015) may increase future interest in woodland creation in the UK uplands. The UK government has committed to creating 30,000 hectares of new woodland per year (DEFRA, 2018, Jordan and Wentworth, 2021), which would increase UK woodland cover to about 18% in 2050. A large proportion of these new woodlands will likely be established in the uplands (Murphy et al., 2022). Broadleaf woodland accounted for 43 % of the new woodland created in the UK in 2019-2020 (Forestry Commission, 2020) and is likely to make up a substantial component of future woodland creation under the UK's 25 Year Environment Plan (DEFRA, 2018). Our work suggests creation of new broadleaf woodlands will help to reduce flood risk. As changes to upland land-use and management occur, it is essential that the influence of those changes to flood risk is monitored and understood (Pender, 2006). An integrated policy perspective combining climate and flood mitigation alongside the additional benefits of woodlands is required to maximise the societal benefits of new woodlands. Future work is needed to identify the most beneficial locations for woodland creation in terms of flood mitigation and to understand how climate and flood mitigation vary for different woodland types.

2.6. Conclusion

Most previous work on the hydrological impacts of forests, especially the impacts on flooding, has been based on conifer forests, typically plantations. The aim of this study was to explore the potential flood mitigation impacts of semi-natural broadleaf woodlands. We established an experimental correlation catchment study in northwest England, to identify differences in streamflow and soil properties between semi-natural broadleaf woodland and permanent pasture. Catchments were selected with similar size, elevation, soil type and geology but different land use.

We found that semi-natural broadleaf woodlands can reduce specific peak discharge by 23-60 %, peak runoff coefficients by 30-60 % and volume runoff coefficient by 21-35 %, compared with pasture. Woodland sites take 14-50 % longer to respond to storm events than do pasture sites. Crucially, we found woodlands reduced runoff for both small and large storms. For storms with a return period of more than 1.5 years, woodlands reduced peak runoff coefficient by 48-58 % and volume runoff coefficient by 26-41 %. Differences in flood response can be explained by the more permeable woodland soils, 11-20 times greater than pasture soil irrespective of the higher bulk density measured.

Our study demonstrates that semi-natural broadleaf woodlands in the uplands can reduce rainfall-generated flooding, strengthening the case for broadleaf woodland creation as a land use management method of NFM. Our study is based on small catchments (< 0.2 km^2) and relatively short (≤ 10 year) storm return periods. Data from our study needs to be used within models to predict the impacts of broadleaf woodlands on downstream flooding in larger catchments and for bigger storm events. Empirical studies are now needed to monitor the long-term impact of reduced grazing levels, tree planting and woodland creation on streamflow and soil properties.

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Chapter 3. Overland flow velocity and soil properties in established semi-natural woodland and wood pasture in an upland catchment

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3.1. Abstract

Management of upland land-use has considerable potential for mitigating flood risk by increasing topsoil storage and slowing overland flow. Recent work has highlighted the potential for vegetation to impact the velocity of saturation-excess overland flow. Woodland creation is widely proposed for Natural Flood Management (NFM), but data on saturation-excess overland flow in woodland habitats is lacking. Here we measure soil properties and overland flow velocities in established broadleaf woodland and wood pasture with an understorey dominated by either grass or bracken. We show that wood pasture dominated by bracken has overland flow velocity 12-20 % lower than established broadleaf woodland and 19-27 % lower than grass-dominated wood pasture. Established woodland soils exhibited 8 times higher saturated hydraulic conductivity than bracken-dominated wood pasture and 80 times higher than grass-dominated wood pasture. We conclude that upland habitats can be managed to reduce flood risk, first by storing storm water in the soil and then by reducing overland flow velocity through rough surface vegetation. These factors combine to reduce floods by delaying the onset of overland flow runoff and slowing its delivery to streams. It is clear than Manning's *n* is far from constant in these shallow overland flows, the development of overland flow datasets is, therefore, also beneficial for improving the theory and practice of hillslope rainfall-runoff modelling.

3.2. Introduction

Flooding has increased across the globe over the past three decades (Kundzewicz et al., 2014; Wingfield et al., 2019) and the frequency of flood events is expected to increase further under future climate change (Blöschl et al., 2019; Iacob et al., 2017). Traditional flood management methods have consisted of expensive, hard-engineered structures. Recently, both researchers and policy makers have shown greater interest in the use of Natural Flood Management (NFM) strategies (Dadson et al., 2017; Stevens et al., 2016). NFM aims to work with natural processes to enhance the water storage capacity of a catchment and to 'slow the flow'. Examples of approaches include the development of built water storage (Nicholson et al., 2020; Quinn et al., 2013), river restoration (Dixon et al., 2016), leaky debris dams (Ashbrook, 2020; Thomas and Nisbet, 2012) and land-use management (Spray et al., 2016).

In recent years land-use management, particularly in the uplands (> 250-300 m above sea level in the UK) has been increasingly debated as an effective method of NFM. The management of upland areas is crucial to managing future flood risk (Murphy et al., 2020) as these regions are experiencing greater increases in precipitation compared to lowland areas (Burt and Holden, 2010; Murphy et al., 2019) and play a principal role in river flow generation (Robinson et al., 2013). Many upland soils are often in poor condition, due to a legacy of soil compaction from long-term over grazing (Holden et al., 2007; Murphy et al., 2020; Sansom, 1999). Soil degradation can increase flood risk due to the loss of macro-porous structures within the soil profile reducing water storage and soil permeability (Alaoui et al., 2018; Murphy et al., 2020; Palmer and Smith, 2013). When implemented in the uplands, NFM land-use management strategies aim to improve the condition of these soils through a number of interventions, including reductions in grazing, peatland restoration and tree planting. However, a lack of empirical data on the impact of NFM interventions on soil and vegetation properties (Bond et al., 2020; Ngai, 2017; Wheater and Evans, 2009) and downstream flooding is a barrier to effective and widespread implementation (Dadson et al., 2017).

There has been a particular interest in tree planting as a method of NFM, as woodland soils are often associated with higher permeability and increased water storage compared to other land covers (Agnese et al., 2011; Archer et al., 2013; Calder et al., 2008; Carroll et al., 2004; Mawdsley et al., 2017; McCulloch and Robinson, 1993; Monger et al., 2021; Murphy et al., 2020; Zimmermann et al., 2006). For example, Archer et al. (2013) found that areas of established woodland exhibited permeability rates 5 to 6 times higher than neighbouring grazed grasslands. Monger et al. (2021) found woodland soils had permeability rates 11–20 times greater than pasture soils. This is often attributed to increased organic matter from leaf litter and the action of tree roots which enhance woodland soils macroporosity and soil structure (Archer et al. 2013; Nisbet and Thomas, 2006). The evidence supporting the benefits of woodland on soil properties has increased in recent years (Burgess-Gamble et al., 2017; Stratford et al, 2017).

Wood pasture are woodland areas grazed by livestock resulting in dynamic systems with a mosaic of different successional stages between grassland and woodland (Peringer et al., 2013; Smit et al., 2005; Uytvanck et al., 2008). Wood pasture has declined drastically in Europe over the last few decades (Smit et al., 2005). Recently there have been several efforts to restore and create wood pasture for conservation and environmental benefits (Uytvanck et al., 2008). Wood pasture offers a compromise to those in land management (including but not exclusively, land owners, land managers and farmers) who want to combine grazing livestock and the benefits of woodland. However, there is limited information available regarding the potential of wood pasture for reducing flood risk (Krašić et al., 2018). Therefore, there is a pressing need to better understand the role of wooded upland habitats in managing flood risk.

The focus on land-use management for NFM has primarily concentrated on the impact of land cover on soil properties through soil permeability and storage (Bronstert et al., 2002; Carroll et al., 2004; Marshall et al., 2014; Monger et al., 2021). Recently, Bond et al. (2020) also highlighted the role of land cover on surface roughness and the importance for flood mitigation. Surface roughness plays a crucial role by reducing the velocity of overland flow and hindering water flow connectivity, 'slowing the flow' and reducing downstream flood peaks. Although the role of roughness has been well studied regarding channel and bank flow (Medeiros et al., 2012), investigations into the impact of vegetation on hillslope roughness are rare (Pan et al., 2016), and measurements of the impact of hillslope vegetation on overland flow have commonly been limited to slopes of less than 1 % and/or have been restricted to semi-arid environments (Dunkerley et al., 2001; Emmett, 1970; Kuhn et al., 2003). An important study by Chow (1959), reported Manning's *n* roughness values for a range of vegetation types on floodplain channels, including cropland and woodland. These values are still commonly used as an estimate of roughness (Burgess-Gamble et al., 2017) but are of limited relevance for shallow overland flow on hillslopes.

Recent studies have started to provide data of hillslope vegetation impacts on overland flow velocity and roughness (Bond et al., 2020; Holden et al., 2008; Wallace et al., 2021). Holden et al. (2008) found the mean overland flow velocity for moss (*Sphagnum*) cover was more than 5 times slower than for a bare peat surface. Gao (2013) used the fully distributed SD-TOPMODEL to estimate that the reduction in overland flow reduced flow peaks by up to 13.4 %. Bond et al. (2020) found overland flow velocity varied by a factor 1.5 between four upland grassland habitats. Grass, leaves, stems and litter all contribute differently to resistance to overland flow (Pan et al., 2016) meaning complex responses to changes in vegetation are likely.

Here we expand on these previous studies and report soil properties and hillslope overland flow velocity for one area of upland established semi-natural woodland and two areas of wood pasture; bracken-dominated wood pasture and grass-dominated wood pasture, for which data is currently lacking. Empirical evidence collected in this study will be useful to inform rainfall-runoff model parameters.

3.3. Methods

3.3.1. Study site

Fieldwork took place in the Naddle catchment, Cumbria, UK (54°31'50.9"N, 2°45'37.3"W) (Figure 3.1a). The area is managed by the RSPB (The Royal Society for the Protection of Birds) on behalf of the landowners, United Utilities. The Naddle catchment consists of a mixture of grazed pasture, grazed wood pasture and un-grazed semi-natural broadleaf woodland. The catchment experiences mild winters and cool summers (Kenworthy, 2014), with mean monthly temperatures ranging from -0.3°C to 18.3°C and mean annual precipitation of 1779 mm, with monthly rainfall ranging from 88 to 231 mm (1981-2010 mean, Shap weather station at 255 m AoD, 5.73km SE (Met Office, 2020)). Soils in the study area are upland organo-mineral soils, predominately Malvern 611a (Chromic Endoleptic Umbrisol), a free draining acid loamy soil (Cranfield University, 2019).

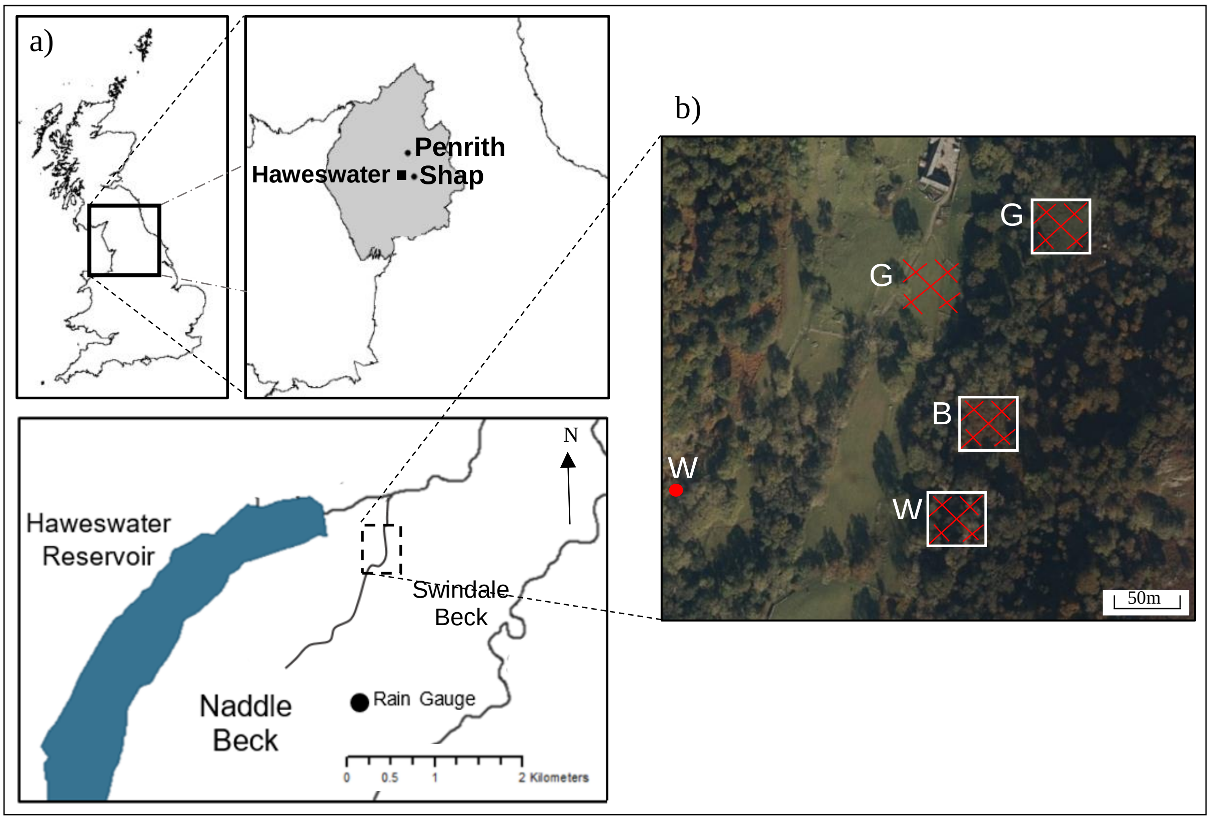

Figure 3.1a) Location of the Naddle Catchment, Lake District, UK. b) Locations for data collection of soil properties (white box) and overland flow velocity measurements (red crosshatch, multiple flume locations or circle, singular flume location). W: established semi-natural woodland. G: grass wood pasture. B: bracken wood pasture.

3.3.2. Data collection

Three UK upland land covers were investigated in this study, 1) an established mature semi-natural broadleaf woodland, 2) wood pasture with an understorey dominated by grass and 3) wood pasture with an understorey dominated by bracken (Table 3.1). Bracken, *Pteridium aquilium,* is a common fern often regarded as a weed species, found on all continents except Antarctica (Rasmussen et al., 2003). Bracken originated as a woodland plant crucial to succession, however it now dominates large tracts of land outside woodland in temperate climates, causing problems for land management (Marrs et al., 2000).

Table 3.1 Land cover descriptions in Naddle Valley. Grazing intensity as LU·ha^{-1}

Land cover	**Dominant Ground Cover Species**	**Management**	**Example site**
Established Semi-natural Woodland	*Nardus stricta, Molinia caerulea, Sphagnum sp., Trifolium repens, Euphrasia sp.*	Deer fenced to exclude livestock and deer grazing. An area of ancient semi-natural upland woodland designated as a site of special scientific interest (SSSI). The NVC classification is W7, W9, W11 – upland mixed woodland and wet woodland.	 
Bracken wood pasture	*Pteridium aquilium, Oxalis sp., Deschampsia flexuosa.*	Intermittent grazing at 0.05 LU·ha^{-1}sheep, with Red and Roe deer. An area of upland wood pasture dominated by bracken, *Pteridium aquilium,* with scattered established trees. Little regeneration due to opportunistic grazing by deer.	 
Grass wood pasture	*Nardus stricta, Molinia caerulea, Anthoxanthum odoratum, Cynosurus cristatus, Festuca spp,Trifolium repens, Rumex acetosa, Ranunculus repens, Achillea millefolium*	All-year round grazing at 0.10 LU·ha^{-1} sheep, with Red and Roe deer. An area of upland wood pasture dominated by grassland, with scattered established trees.	 

Soil sampling and overland flow velocity measurements were completed in October 2019 and October 2020 (Table 3.2). Soil data was collected at 5 m intervals (25 soil sampling sites) across 20 m by 20 m plots established within each of the land covers investigated (Figure 3.1b, white boxes).

Table 3.2 Sampling strategy, number of individual measurements (number of flume locations/soil sampling sites).

Measurement		Established semi-natural woodland	Bracken wood pasture	Grass wood pasture
Overland flow velocity	1.2 l/min	27 (7)	36 (8)	35 (7)
	6 l/min	33 (7)	38 (8)	35 (7)
	12 l/min	35 (7)	40 (8)	35 (7)
Soil properties	Bulk density	25 (25)	25 (25)	25 (25)
	Saturated hydraulic permeability	25 (25)	25 (25)	25 (25)
	Soil moisture	250 (25)	250 (25)	250 (25)
	Organic matter	25 (25)	25 (25)	25 (25)

To investigate overland flow velocity in the established woodland and bracken wood pasture, flume locations were selected at random within approximately 10 m of the plots established for soil sampling. With the exception of one established woodland flume location (Figure 3.1b, red crosshatch, multiple flume locations or circle, singular flume location). Due to changes in land management, the overland flow velocity sampling locations for the grassland wood pasture was moved following the October 2019 field campaign (2 flume locations in 2019, 5 flume locations in 2020). Overland flow velocity measurement sites were restricted to locations with a local gradient of between 11-16 ° and away from field boundaries to reduce edge effects. Sampled elevations range from 250 – 292 m AOD.

3.3.3. Soil properties

We measured soil bulk density, saturated hydraulic conductivity (Ksat), soil moisture and organic matter. To calculate Ksat ($m \cdot s^{-1}$), intact soil cores were taken using Eijelkamp bulk density rings from the upper 5 cm of soil. Any vegetation present was clipped using shears as low as possible (Cresswell and Hamilton, 2002). Soil cores were placed in an Eijelkamp Permeameter and Ksat measurements taken following the method set out by Eijkelkamp (2011). Saturated soil cores were then transferred to pre-weighed containers and dried overnight at 105°C (a minimum of 16 hours) to remove moisture. On removal from the oven, samples were placed in a desiccator to cool and then reweighed to determine bulk density

($g \cdot cm^{-3}$) (Cresswell and Hamilton, 2002). Finally, to calculate the percentage organic matter (%) of the soil, the loss on ignition method at 550°C was used (Dean, 1974).

Soil moisture content (%) was measured in the field using a Delta-T Ltd ‘theta probe’. Approximately 225 readings were taken at each land use. Measurements were taken within a 24 hour period to reduce any potential weather impacts. The ‘theta probe’ uses a simplified Time-Domain Reflectometry (TDR) technique to derive instant values of volumetric moisture content (Delta-T, 1999).

3.3.4. Overland flow velocity

Overland flow velocity was measured using the portable hillslope flume described in detail by Bond et al. (2020). The portable flume was constructed by hammering aluminium side panels into the ground to create a 0.4 m by 2.0 m bounded plot with a z-shape base panel (0.4 m wide with three 0.2 m long faces angled at 60° to form the z-shape) dug into the ground, so that the upper surface of the base panel was level with the soil, downslope of the aluminium side panels (Figure 3.2). The z-shape was driven approximately 2 cm into the soil face to create a seal between the ground surface and z-shape. A plastic funnel was then fitted level to the z-shape on the opposing side using tape to secure and petroleum jelly to make watertight. The funnel collected and subsequently channelled water that had travelled the length of the flume through the attached Seapoint Rhodamine fluorometer. The fluorometer logged the fluorescence of water in SEVolts every one second using a CR220x data logger. A low-concentration tracer, Rhodamine water tracing (WT) dye, was injected at the inlet of the flume to enable automated velocity measurements. Mean overland flow velocity was calculated for three discharge rates (30, 15 and 3 $l \cdot m^{-1} \cdot s^{-1}$), with a minimum of 5 Rhodamine tracer injections recorded at each discharge rate.

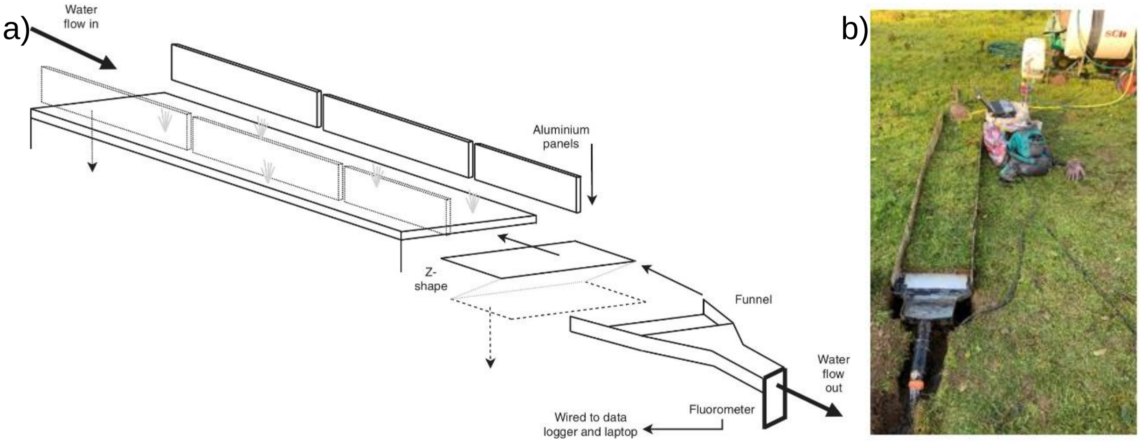

Figure 3.2 a) Overland flow hillslope flume design (Bond et al., 2020) b) Example of portable flume installed in the grass wood pasture habitat.

As set out in Bond et al. (2020), mean overland flow velocity, U (m·s^{-1}) was calculated using an inverse time weighting (i.e. linear in distance) summed over r sequential time steps where:

$$U = \frac{\sum_{i=1}^{r} \frac{I}{t_i v_{q_i}}}{\sum_{i=1}^{r} Vq_i} \tag{3.1}$$

Where:

I is the vegetated flume length (m);

t is the time difference from point of Rhodamine tracer injection (s);

Vq is fluorescence strength.

Further information about these calculations, including a list of abbreviations and examples of breakthrough curves, can be found in Appendices S1 and S3 of Bond et al. (2020).

Manning's n roughness ($s{\cdot}m^{-\frac{1}{3}}$) was calculated as a commonly used measure of roughness:

$$n = \frac{R^{\frac{2}{3}} S^{\frac{1}{2}}}{U} \tag{3.2}$$

Where:

R is the hydraulic radius (m);

S is the slope (radians).

Mean flow depth, d (m) was calculated from mean overland flow velocity, U, the fixed flume width, w (m) and the set discharge rate, Q (m^3·s^{-1}):

$$d = \frac{Q}{wU} \tag{3.3}$$

Mean overland flow velocity was calculated for the three discharge rates (30, 15 and 3 $l \cdot m^{-1} \cdot s^{-1}$), with a minimum of 5 Rhodamine tracer injections recorded at each discharge rate.

3.3.5. Statistical analysis

Shapiro–Wilk tests were used to test the normality of soil properties and overland flow velocity. Normally distributed data was then analysed using ANOVA, followed by Tukey's post hoc tests to identify significant differences ($p < 0.05$) between land covers, whilst non-normal data was analysed using Kruskal-Wallis and Conover's post-hoc tests.

Statistics were performed using the Python SciPy (Virtanen et al., 2019) and scikit-posthocs (Terpilowski, 2019) packages.

3.4. Results

3.4.1. Soil properties

Bulk density data was normally distributed, whilst Ksat, soil moisture and soil organic matter was non-normally distributed. A post-hoc Tukey's test found that the bulk density of established woodland soil was significantly ($p < 0.05$) lower than both bracken wood pasture and grass wood pasture soils (respectively 21 % and 18 % lower) (Figure 3.3a, Table 3.3). However, there was no significant difference between bracken wood pasture and grass wood pasture. Using a post-hoc Conover's test we found that established woodland soils exhibited significantly ($p < 0.05$) higher Ksat compared with bracken wood pasture and grass wood pasture soils, by 8 times and 80 times respectively (Figure 3.3b). Bracken wood pasture Ksat was significantly ($p < 0.05$) (10 times) higher than grass wood pasture. Soil moisture in semi-natural woodland soils (86.5 %) was significantly ($p < 0.05$) greater than bracken wood pasture (75.5 %) and grass wood pasture (78.5 %). Additionally, grass wood pasture soil moisture was significantly ($p < 0.05$) higher than bracken wood pasture. The organic matter of semi-natural woodland soils (24.0%) was significantly ($p < 0.05$) higher than the bracken wood pasture (18.4 %) and grass wood pasture (16.6 %) soils. There was no significant difference between bracken wood pasture and grass wood pasture.

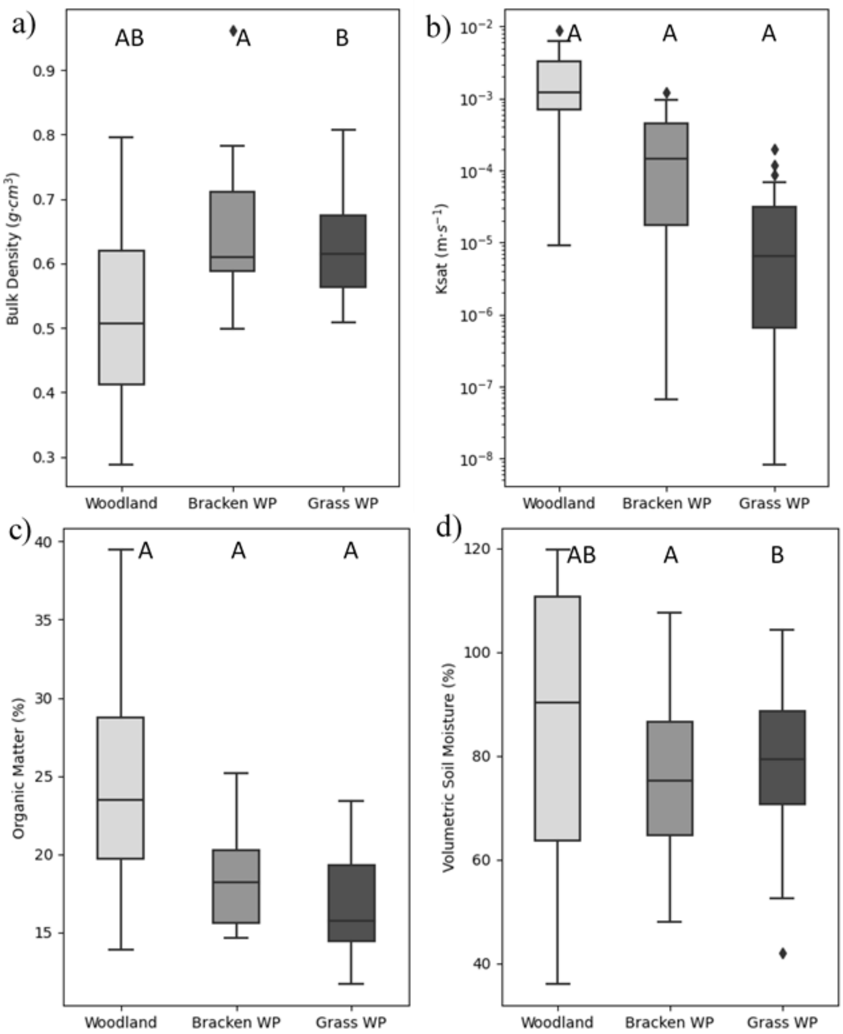

Figure 3.3 Soil properties including a) Bulk density ($g{\cdot}cm^{-3}$), b) Saturated hydraulic conductivity (Ksat, $m{\cdot}s^{-1}$), c) Soil moisture (%), d) Soil organic matter (%) shown as median (line), 25th to 75th percentile (box), 5th to 95th percentile (whiskers). Sites that are statistically different share a letter.

Table 3.3 Median (η), mean (μ) and standard error (SEM) for Bulk Density, Permeability, Soil Moisture and Organic matter at each habitat.

Habitat	**Bulk density ($g{\cdot}cm^{-3}$)**			**Permeability ($m{\cdot}s^{-1}$)**			**Soil moisture (%)**			**Organic matter (%)**		
	η	**μ**	SEM	η	**μ**	SEM	η	**μ**	SEM	η	**μ**	SEM
Established Semi-natural Woodland	0.51	0.51	0.03	$1.23{\times}10^{-3}$	$2.28{\times}10^{-3}$	$4.90{\times}10^{-4}$	92.7	86.5	1.6	23.5	24.0	1.4
Bracken wood pasture	0.61	0.65	0.02	$1.50{\times}10^{-4}$	$2.90{\times}10^{-4}$	$7.00{\times}10^{-5}$	74.6	75.5	0.9	18.2	18.4	0.6
Grass wood pasture	0.61	0.63	0.02	$1.00{\times}10^{-5}$	$3.00{\times}10^{-5}$	$1.00{\times}10^{-5}$	78.7	78.5	0.8	15.7	16.6	0.6

3.4.2. Overland flow

The lowest mean overland flow velocity was found at the bracken wood pasture site under all discharge rates, whilst grass wood pasture site had the highest mean overland velocity (Table 3.4). Mean overland flow at the bracken wood pasture was 12-20 % lower than the semi-natural woodland and 19-27 % lower than the grass-dominated wood pasture (Figure 3.4). Overland flow velocity data collected for all three discharge rates was non-normally distributed. Therefore Kruskal-Wallis followed by Conover's post-hoc tests were used to identify significant differences between land covers. At the lowest discharge rate (3 $l \cdot m^{-1} \cdot s^{-1}$), overland flow velocity at the bracken wood pasture site was significantly ($p < 0.05$) lower than both grass wood pasture and established woodland, with no significant difference between grass wood pasture and established woodland. At both 15 and 30 $l \cdot m^{-1} \cdot s^{-1,}$ overland flow velocity at the bracken wood pasture site was significantly ($p < 0.05$) lower than both grass wood pasture and established woodland. In addition, overland flow velocity at the established woodland site was significantly ($p < 0.05$) lower than the grass wood pasture site.

When converted to Manning's *n* roughness, values ranged between 0.50 – 2.54 $s{\cdot}m^{-\frac{1}{3}}$. Grass wood pasture consistently had the lowest Manning's *n* values across the three discharge rates.

Mean flow depth ranged between 0.006 and 0.013 m across the three discharge rates. At the lowest discharge rate (3 $l \cdot m^{-1} \cdot s^{-1}$), depth was greatest for semi-natural woodland sites. However, for the other two discharge rates, bracken wood pasture had the greatest depth and grass wood pasture the shallowest.

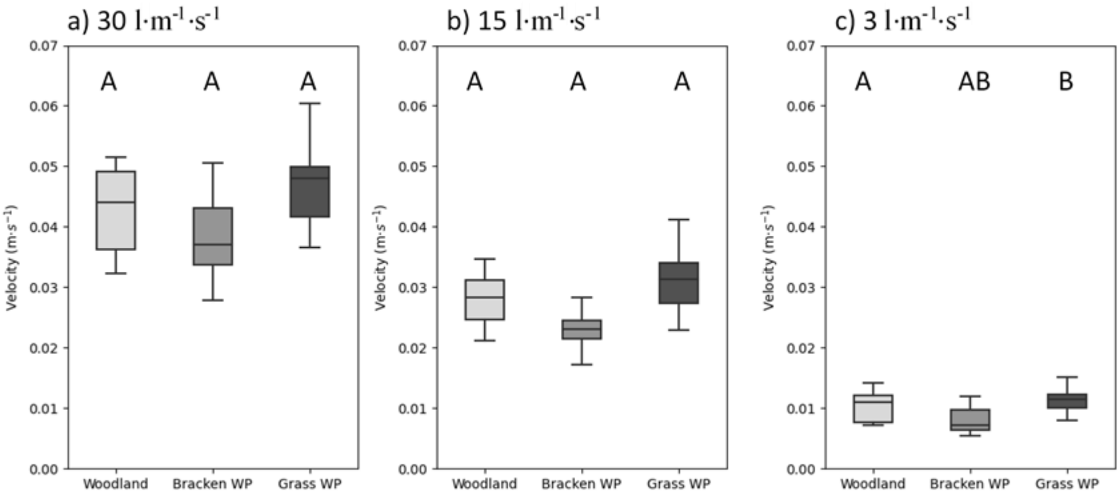

Figure 3.4 Distribution of flow velocity for flow rates of a) 30 $l \cdot m^{-1} \cdot s^{-1}$, b) 15 $l \cdot m^{-1} \cdot s^{-1}$, c) 3 $l \cdot m^{-1} \cdot s^{-1}$, at the three different habitats. Shown as median (line), 25th to 75th percentile (box), 5th to 95th percentile (whiskers). Sites that are statistically different share a letter.

Table 3.4 Overland flow velocity (U) recorded at 30, 15 and 3 $l{\cdot}m^{-1}{\cdot}s^{-1}$. Count represents the number of Rhodamine injections per habitat. For velocity, the mean (μ) and standard error of the mean (SEM) is given. For Slope, the mean (μ) slope in radians is shown.

Habitat	Velocity, U (m.s^{-1})			Manning's n $(s{\cdot}m^{-\frac{1}{3}})$	Slope	Depth (m)
	Count, n	μ	SEM	μ	μ	μ
3 $l{\cdot}m^{-1}{\cdot}s^{-1}$						
Established Semi-natural Woodland	27	0.010	0.0005	2.41	0.24	0.011
Bracken wood pasture	30	0.008	0.0005	2.54	0.22	0.009
Grass wood pasture	33	0.011	0.0003	1.89	0.23	0.009
15 $l{\cdot}m^{-1}{\cdot}s^{-1}$						
Established Semi-natural Woodland	32	0.028	0.0007	0.76	0.24	0.009
Bracken wood pasture	34	0.023	0.0006	1.00	0.22	0.011
Grass wood pasture	33	0.031	0.0008	0.52	0.23	0.006
30 $l{\cdot}m^{-1}{\cdot}s^{-1}$						
Established Semi-natural Woodland	34	0.043	0.001	0.59	0.24	0.012
Bracken wood pasture	38	0.038	0.001	0.67	0.22	0.013
Grass wood pasture	31	0.047	0.001	0.50	0.23	0.011

3.5. Discussion

Our study reports some of the first upland hillslope overland flow velocity measurements for established semi-natural woodland and wood pasture. We demonstrate the importance of trees as a potential NFM strategy whether as part of a woodland or wood pasture habitat.

3.5.1. Comparison of soil properties in established woodland and wood pasture

We found that semi-natural woodland soils exhibited higher Ksat and lower bulk density when compared to other vegetation types, consistent with current understanding (Archer et al., 2013, Calder et al., 2008, Carroll et al., 2004, McCulloch and Robinson, 1993). Higher Ksat is attributed to a more open soil structure, evidenced by lower bulk density and greater organic matter in woodland soils. Semi-natural woodland soils also exhibited significantly higher soil moisture compared to wood pasture soils, possibly due to lack of livestock and lower levels of grazing at the woodland site (Xu et al., 2014). However, the grass wood pasture had higher grazing intensity but also exhibited higher soil moisture compared to the bracken wood pasture. The larger range of soil moisture in woodland soils may be explained by a more porous structure meaning soils can store greater rainfall volumes and so there is potential for higher and lower soil moisture compared to grazed pasture systems. There was no significant difference in bulk density and organic matter between the two wood pasture habitats, yet the soils at the bracken wood pasture site had significantly higher Ksat. Whilst it is well understood that grazing modifies vegetation structure, soil composition (Milligan et al., 2016, Orr and Carling, 2006) and water storage capacity (Meyles et al., 2006), the influence of different grazing patterns and foraging behaviour is less well studied. Therefore, the presence of bracken, which is a less favourable grazing environment than the grassland dominated site, due to its toxicity, may have influenced grazing regimes and therefore impacted soil characteristics.

3.5.2. Comparison of overland flow velocity in established woodland and wood pasture

Land management impacts soil properties and vegetation composition which both influence the generation of overland flow and the potential for downstream flooding (Stratford et al., 2017). We found that woodland soils had greater Ksat and lower bulk density, which may contribute to delays in the formation and reduce the volume of saturation excess overland flow (Monger et al., 2021). Infiltration excess overland flow may be generated when rainfall intensity exceeds infiltration rates or where compaction is high. Regardless of how overland flow is generated, surface roughness plays a dominant role in delaying delivery of water to streams, extending the tail of the hydrograph and reducing the flood risk. Vegetation alters the velocity of overland flow through controlling the roughness of the surface (Bond et al, 2020; Holden et al, 2008). We found that the bracken wood pasture sites had the greatest surface roughness and the lowest overland flow velocity across all three discharge rates analysed. This could be explained by the accumulations of bracken leaf litter, creating friction between the vegetation and overland flow. Contrastingly, grass wood pasture sites had short-cropped vegetation and the highest overland flow velocity. Bracken is generally considered as a problem species and often heavily managed due to its toxicity to livestock (Marrs et al., 2000; Pakeman and Marrs, 1992). Here we identify a positive and largely unrecognized benefit of a bracken understorey for reducing overland flow.

3.5.3. How does woodland and wood pasture compare to other upland habitats?

Table 3.5 compares overland flow velocities measured in our study against those from upland peat (Holden et al., 2008) and grassland (Bond et al., 2020). For a discharge rate of 30 $l \cdot m^{-1} \cdot s^{-1}$, overland flow velocities varied from 0.023 $m \cdot s^{-1}$ for peatland habitats dominated by grass and moss, 0.028 $m \cdot s^{-1}$ for low-density grazed pasture, 0.038 $m \cdot s^{-1}$ for bracken wood pasture, 0.050 $m \cdot s^{-1}$ for bare peat to 0.052 $m \cdot s^{-1}$ for hay meadows. The presence of moss, with its coarse structure, appears a contributing factor to lower overland flow velocities and higher roughness in both the peatland and low-density grazed pasture. This highlights the importance of vegetation structure in the first few centimetres (Pan et al., 2016) as this is the part that interacts with overland flow.

As overland flow velocity decreased (vegetation roughness increased) there was proportional increase in flow depth. Overland flow velocity is dependent on the 'roughness', which is related to the vegetation density of the first few cm of vegetation, where the overland flow is being intercepted by the vegetation. This is further highlighted through further comparison between the grass wood pasture and the grassland habits studied by Bond et al. (2020) (Table 3.6). The grass wood pasture investigated in our study had a lower grazing intensity (0.10 $LU \cdot ha^{-1}$) but higher overland velocity compared to the 'low-density grazing' (0.25 $LU \cdot ha^{-1}$) site in Bond et al. (2020). The generally higher flow depth exhibited by the habitats studied by Bond et al. (2020) suggest that more of the vegetation is intercepting overland flow down the hillslope. Bond et al., (2020) attributed this to the mossy understorey found at the 'low-density grazing' site. Furthermore, the rank grassland site, where grazing had been removed for 6 years, also exhibited lower overland flow velocities compared with sites investigated in our study. The rank grassland site looks visibly rougher, see Table 3.6, indicating the potential benefits of removing grazing. Comparing these grassland habitats offers an insight into the variability the management of land and species presence can have on overland flow.

Surface roughness and overland flow velocity depend on habitat management and vary seasonally due to changes in vegetation over the growing season (Bond et al., 2020). It would be expected that the habitats we investigated would also be impacted by seasonality, but we were unable to assess this since our study was restricted to measurements taken in October. The seasonal growth and dieback of bracken would likely influence roughness throughout the year dependent on management. Depending on how bracken grows and decays, its influence on roughness in the first few cm will vary and future work is needed to confirm its control on overland flow seasonally.

Table 3.5 Comparison of overland flow velocities at flow rates of 30, 15 and 3 l·m^{-1}·s^{-1} recorded in our study against previous work. Measurements from Bond et al. (2020) are from November, as the closest seasonal comparison to our study. U = Mean overland flow velocity (m·s^{-1}), n = Estimate of Manning's n coefficient (s·m$^{-\frac{1}{3}}$), f = Darcy-Weisbach Roughness 1/sqrt(f), d = mean flow depth (m).

Habitat	**Study**	**Flow rates**			**Mean overland flow velocity, *U***
		3 l·m^{-1}·s^{-1}	15 l·m^{-1}·s^{-1}	30 l·m^{-1}·s^{-1}	
Grass wood pasture	This	U= 0.011 n = 1.890 f = 0.026 d = 0.009	U = 0.031 n = 0.518 f = 0.066 d = 0.006	U = 0.047 n = 0.502 f = 0.100 d = 0.011	**0.033**
Hay meadows	Bond et al., 2020	U = 0.012 n = 0.917 f = 0.052 d = 0.005	U = 0.031 n = 0.545 f = 0.039 d = 0.009	U = 0.052 n = 0.341 f = 0.154 d = 0.010	**0.032**
Bare peat	Holden et al., 2008	-	-	U = 0.050	-
Established Semi-natural Woodland	This	U = 0.010 n = 2.410 f = 0.022 d = 0.011	U = 0.028 n = 0.76 f = 0.058 d = 0.009	U = 0.043 n = 0.592 f = 0.081 d = 0.012	**0.027**
Rushes	Bond et al., 2020	U = 0.007 n = 2.238 f = 0.023 d = 0.008	U = 0.026 n = 0.747 f = 0.071 d = 0.011	U = 0.039 n = 0.586 f = 0.093 d = 0.014	**0.024**
Bracken wood pasture	This	U = 0.008 n = 2.543 f = 0.019 d = 0.009	U = 0.023 n = 1.003 f = 0.052 d = 0.011	U = 0.038 n = 0.674 f = 0.075 d = 0.013	**0.023**
Peat grassland	Holden et al., 2008	-	-	U = 0.037	-
Rank Grassland	Bond et al., 2020	U = 0.004 n = 5.699 f = 0.010 d = 0.013	U = 0.019 n = 1.403 f = 0.039 d = 0.015	U = 0.030 n = 1.007 f = 0.056 d = 0.019	**0.018**
Low-density Grazing	Bond et al., 2020	U = 0.006 n = 3.255 f = 0.016 d = 0.009	U = 0.018 n = 1.392 f = 0.040 d = 0.016	U = 0.028 n = 1.053 f = 0.054 d = 0.020	**0.017**
Peat grassland and moss mix	Holden et al., 2008	-	-	U = 0.023	-
Peat moss	Holden et al., 2008	-	-	U = 0.023	-
Mean overland flow velocity, *U*		**0.008**	**0.025**	**0.040**	

Table 3.6 Comparison of grassland habitats with Bond et al. (2020).

Study	**Bond et al., 2020**	**Bond et al., 2020**	**This study**
Habitat	Low-density Grazing	Rank Grassland	Grazed wood pasture
Site description	Consisting of common grasses, underlain by moss throughout. Due to grazing, these species remain close to ground level. Grazing occurs in this area for short periods of time for treatments, shearing and separating lambs from ewes.	Typically species poor, rank grassland is dominated by tall, tussocky and coarse grass species and is produced in unmanaged, ungrazed grasslands.	Open pasture with scattered established trees. Grazed all year round, with intermittent high density grazing due to its preferential location next to farm buildings.
Grazing Intensity	0.25 $LU \cdot ha^{-1}$	No grazing for 6 years	0.10 $LU \cdot ha^{-1}$
Visual comparison			

3.5.4. *Impact of grazing*

The wood pasture environments investigated in this study were managed with a relatively low grazing intensity. Soil properties and vegetation roughness are strongly influenced by grazing regimes (Drewry et al., 2008) and we compared our findings with areas of grassland managed at different grazing intensities in the neighbouring valley, Swindale. The frequency of overland flow occurrence will determine the relative importance of surface roughness or permeability for flood management. If soils are shallow then overland flow may occur despite high Ksat, and therefore surface roughness is of greater importance. If compaction is preventing infiltration, then improving soil aeration and therefore Ksat, may be the best method. Grazing livestock can exert considerable pressure on the soil surface, causing compaction (Chandler et al., 2018; Wheeler et al., 2002), and reducing soil porosity (Clarke et al., 2008). If grazing density is too high, then decreasing stocking levels is likely to increase both roughness and Ksat. Future work is needed to investigate how different grazing intensity and livestock (e.g. cattle versus sheep) impacts on vegetation roughness and overland flow velocity. Some grazing systems may develop more heterogeneity in vegetation (Lunt et al., 2021) and roughness, with possible implications for downstream flood peaks.

3.5.5. Impact of tree canopy

This study has identified potential co-benefits and trade-offs between tree canopy cover, understorey vegetation and grazing. Chandler et al. (2018) found that both tree species and forest management (grazed versus ungrazed) have important effects on soil hydraulic properties with implications for surface runoff. Future research is needed to compare a wider range of tree species, woodland management and tree density.

Woodlands with closed canopy cover will typically have sparse understorey vegetation (Alder et al., 2018), resulting in lower surface roughness and greater overland flow velocity. Mature semi-natural woodlands with a varied age structure and canopy gaps, woody debris, shade-tolerant woodland flora communities and greater understorey are likely to result in greater surface roughness and reduced overland flow. Woodlands with a relatively open canopy may therefore combine the higher soil permeability typical of woodland soils in combination with the higher surface roughness associated with a denser understorey. In contrast, wood pasture has less permeable soils likely due to the lower density of trees but greater surface roughness where the ground vegetation below the open canopy is dominated by bracken. Wood pasture has a wide range of grazing regimes, understorey vegetation, tree density and canopy cover. Careful control of grazing levels to allow natural regeneration and increased tree density could increase soil permeability towards levels seen in woodlands whilst maintaining vegetation understorey and associated lower overland flow velocity.

3.5.6. Suitability of Manning's n roughness coefficient

We found that the Manning's *n* values calculated for the three habitats investigated in this study can be an order of magnitude higher than previous values reported by Chow (1959) and others (Arcement and Schneider, 1989). We compare our grass wood pasture site (n = 0.50 – 1.89) to the Chow (1959) floodplain habitat described as a short grass habitat (n = 0.030), the bracken wood pasture (n = 0.67 – 2.54) with floodplain covered with medium to dense brush (n = 0.100) and established woodland (n = 0.59 – 2.41) to the 'heavy stand of timber, a few down trees, little undergrowth, flood stage below branches' (n = 0.100). This supports the potential unsuitability of Manning's *n* roughness coefficients to represent hillslope vegetation roughness and shallow overland flows in hydrological modelling (Arcement and Schneider, 1989; Rose and Rosolova, 2015). Manning's *n* is not constant in these environments and varies with water depth across vegetated surfaces (Zhang et al., 2021). Different types of vegetation cover have more resistance to overland flow and are 'rougher' (Bond et al., 2020; Holden et al, 2008). Rougher land covers reduce the velocity of overland flow and can contribute to reducing flood risk further down the catchment (Gao, 2013). Therefore, it is important to represent these environments as accurately as possible when modelling for future flood

3.5.7. Woodland creation and management for NFM

Our results show the potential for woodland and wood pasture in the UK uplands to increase soil permeability and surface roughness and reduce overland flow. Results from our study and others (Murphy et al., 2020; Chandler et al., 2018) suggest that woodland type and management are important controls over soil permeability and overland flow velocity. We found bracken-dominated wood pasture to have the slowest overland flow, whilst the established woodland had the highest Ksat values. Bracken is often considered to be a weed that needs to be managed; our results demonstrate the benefits bracken can have on slowing overland flow. Trees play an important role in altering soil properties and permeability but tree density may not necessarily be the most important factor (Murphy et al., 2020). Future work is needed to assess the effects of tree stocking density on soil permeability and surface roughness to inform woodland creation grant schemes (e.g., England Woodland Creation Offer).

Many UK woodlands are grazed by sheep and deer with important impacts on tree regeneration and ground vegetation (Ford and Smith, 2016) that will alter soil properties including permeability, surface roughness and overland flow. Long-term changes in understorey are also occurring across UK woodlands due to changes in management and deer density (Amar et al., 2010) but the impacts on surface roughness and overland flow are not known. The impact of grazing intensity in wood pasture on tree regeneration, soil properties and surface roughness also needs further investigation.

In addition, modelling work is needed to understand the relative impacts of greater permeability in mature woodland compared to lower overland flow in bracken wood pasture on downstream flooding. Future work is needed to explore potential trade-offs between tree density, biomass carbon storage, soil permeability and overland flow and the implications for both climate mitigation and NFM.

3.6. Conclusion

In this study, we report saturated hydraulic conductivity (Ksat) and overland flow velocity in established semi-natural woodland and wood pasture dominated by either a bracken or grass understorey. We find that mature semi-natural woodland soils have the highest Ksat and storage of soil water, whilst the bracken wood pasture has the roughest surface, resulting in the lowest overland flow. However, it is important to note our study is carried out in a limited area to represent the three land covers investigated, further sites are needed to ensure these sites are consistent with other areas of similar land cover.

Combined, these habitats have the potential to reduce flood risk by temporarily retaining surface and subsurface storm water. During the initial stages of a storm, woodland soils have the potential to delay and reduce peak flow through storing storm water. However, once available soil storage is filled or compaction reduces infiltration, reductions in overland flow velocity are crucial. This is where areas of bracken wood pasture can play their part. Bracken, whilst often seen as a nuisance, can be beneficial on the slopes of a catchment to increase surface roughness, reducing overland flow velocity.

Future work is needed to understand how variations in canopy cover and understorey vegetation within woodland and wood pasture impact both soil permeability and surface roughness. Improved understanding may allow land management to be crafted to maximize the benefits of both woodlands and wood pasture for reduced downstream flooding.

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Chapter 4. The influence of land cover and catchment characteristics on active storage within UK river catchments

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4.1. Abstract

Land cover can have important impacts on hydrology but the effects at the catchment scale are still uncertain. We examined the form of hydrograph recession curves in 418 UK catchments and estimated the TOPMODEL storage parameter for all December periods with low rainfall and evapotranspiration. Available recessions broadly conformed to TOPMODEL assumptions, and so provide meaningful estimates of m for each catchment. These analyses provide a relevant measure of active storage capacity during rainstorms. The storage parameter varied significantly with dominant land cover in the catchment (grassland, 12.6 mm and woodland, 8.0 mm), arable (4.4 mm) and urban (1.6 mm) with weaker linkages to geology, elevation and rainfall. Catchment storage increased in grassland and woodland catchments and decreased in urban catchments over the period analysed (1985-2005 versus 2006-2017), perhaps reflecting widespread changes in land use, for example though reduced grazing pressures. These observed differences have implications for natural flood management.

4.2. Introduction

Catchment storage is one of the most important hydrological functions, regulating the generation of streamflow and buffering meteorological extremes (Staudinger et al., 2017). The role of catchment storage within the hydrological cycle has received increased attention in recent years (Spence, 2007; Soulsby et al., 2008; McNamara et al., 2011; Tetzlaff et al., 2011; Buttle, 2016; Fan, 2019; Buzacott and Vervoort, 2021) as a useful tool for comparing catchments (McNamara et al., 2011).

Catchment storage is distributed among snowpack, vegetation, surface water, soil moisture, and groundwater (McNamara et al., 2011). The responsiveness and dynamic range of all these storage elements determines the extent to which each catchment buffers storm precipitation, holding and delaying water and mitigating the severity of floods. The storage equation (4.1) is commonly used to summarise the water balance of a catchment.

$$\frac{dS}{dT} = P - E - Q \tag{4.1}$$

Where;

S = catchment storage and P, E, Q are respectively precipitation, evapotranspiration and runoff.

Measuring storage directly at catchment scale can be difficult, as current measurement techniques are expensive and are at scales that are too small or large (Ramillien et al., 2008; Creutzfeldt et al., 2014; Staudinger et al., 2017). However novel methods, such as recession analysis (Kirchner, 2009), tracer applications (Soulsby et al., 2008), and remote sensing methods at a larger scale (Ramillien et al., 2008) can provide approximate measures of catchment storage (Buzacott and Vervoort, 2021).

The most important component of catchment storage is active storage (McNamara et al., 2011; Tetzlaff et al., 2011) sometimes referred to as dynamic storage (Staudinger et al., 2017). Active storage refers to zones that fill and release water over the time span of a flood hydrograph (Staudinger et al., 2017). Kirchner (2009) assumed that if there was a single-valued relationship between catchment storage and discharge, then back-analysis of hydrograph recession curves for periods with negligible evapotranspiration and precipitation could generate the form of the storage-discharge function, equation (4.2), for some empirical function *f*:

$$-\frac{dQ}{dt} \propto f(Q) \tag{4.2}$$

Where;

Q = discharge and t = time.

Combining equations (4.1) and (4.2) provides the functional relationship between discharge and dynamic storage, *S.* This inverse method provides an estimate for the 'dynamic' storage, but is limited by difficulties in physically identifying this storage, as it doesn't account for antecedent conditions across a range of storm conditions. This method has been applied by a number of studies (Teuling et al., 2010; Birkel et al., 2011; Staudinger et al., 2017).

Here we use a parameterised estimate of active storage taken from the Beven and Kirkby (1979) TOPMODEL. The model uses three key parameters for catchment modelling: K the hydraulic conductivity of the soil; k_v the overland flow velocity parameter related to surface roughness and m the scaling parameter representing the active water storage in soil. The parameter m is defined as change in depth expressed in millimetres (mm) of water, through which the soil can dynamically absorb and transmit water. Importantly, for this study, the parameter m controls for antecedent conditions and may be calculated across multiple storms. In this study we estimate m for catchments across the UK and compare catchment characteristics. We further hypothesise that land cover will be a significant influence on m. As will be further seen below, this method is valid provided that equation (4.2) can be well approximated by the power law relationship in equation (4.3). Kirchner (2009) showed that this was approximately the case for the two UK Plynlimon catchments used to illustrate the method.

$$-\frac{dQ}{dt} \propto Q^2 \tag{4.3}$$

It is generally accepted that land cover influences hydrological processes such as interception and infiltration (Bronstert et al., 2002; Carroll et al., 2004; Chandler and Chappell, 2008; Marshall et al., 2014). Previous studies have also reported that a change in land cover can be associated with a change in catchment runoff. For example, it is well established that a conversion of agricultural or grassland into woodland can reduce average catchment runoff as a result of increased rainfall interception, transpiration and permeability (Brown et al., 2005; Rogger et al., 2017; Zhang et al., 2018). Yet the impact of land cover on catchment storage has been largely overlooked.

There have been a limited number of studies that have investigated the potential influence of physical characteristics on catchment storage. Greater catchment storage has been found to be strongly associated with large mean catchment elevation, range of elevation and slope (Staudinger et al., 2017; Buzacott and Vervoort, 2021). Rainfall intensity, along with the water production capacity of a catchment (Lazo et al., 2019) and bedrock geology (Pfister et al., 2017) is also thought to be a control for catchment storage. Further investigation into catchment characteristics is needed to help identify common controls on catchment storage (Buzacott and Vervoort, 2021).

Understanding the controls of catchment storage is becoming increasingly important as the frequency and severity of flood events is predicted to increase under future climate change (IPCC, 2018). The potential to alter the storage of a catchment and alleviate flood risk through land cover change offers a Natural Flood Management (NFM) solution. Evidence to support this is needed as the popularity of NFM strategies has grown in recent years (Dadson et al., 2017).

We estimate active storage as parameter m for 418 catchments across the UK, classifying catchments on the basis of dominant land use, in order to investigate trends between land cover and the storage parameter. Additionally, we calculate the change in m to identity if land cover has led to any changes in active storage. Furthermore we use generalized linear models (GLMs) to investigate the influence of land cover and other environmental variables, such as catchment size, elevation and rainfall on m.

4.3. Methods

We used river flow data (December data only, to assume negligible evapotranspiration) from across the UK to calculate active storage as parameter m and change in m for river catchments using flow recession analysis. We then explored relationships between m and environmental variables including land cover, catchment size and annual rainfall. We also calculated change in m between 1985-2005 and 2006-2017, to investigate the potential impact of land management on active storage. Finally, we examined trends between changes in land cover documented by the Centre for Ecology and Hydrology (CEH) between 1990 and 2015 and changes in m for the same duration.

4.3.1. Data sources

We used river flow data at 15-minute resolution from the UK National River Flow Archive (NRFA; https://nrfa.ceh.ac.uk/). We selected the 418 catchments classified by the NRFA as 'natural' (Figure 4.1), where there are no abstractions or discharges and gauged flow is considered to be within 10 % of the natural flow (Young *et al.*, 2003).

Catchment size and average catchment elevation was derived from the CEH Integrated Hydrological Digital Terrain Model (IHDTM). The natural catchments (n = 418) range in size from 0.9 - 9866 km^2 (median = 72 km^2) with average elevation ranging from 6 - 619 m Above Ordnance Datum (AOD) (median = 194 m AOD).

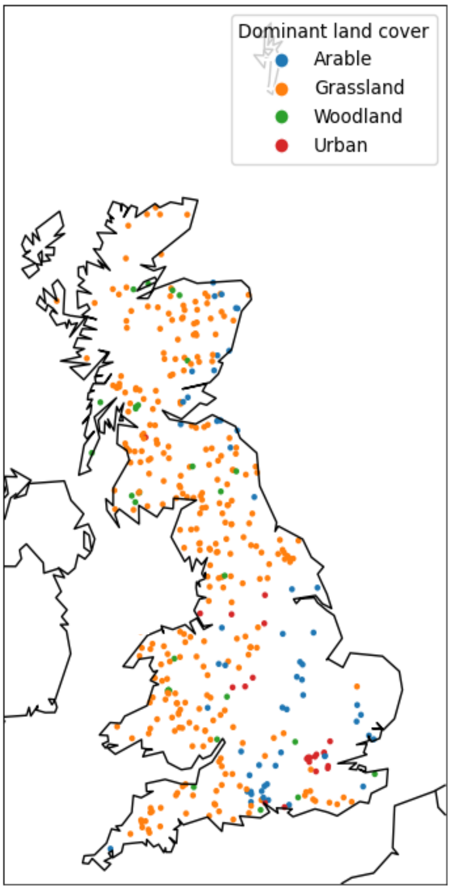

Figure 4.1 Location of natural catchments (n=418) in the NRFA database. Dots represent the location of the NRFA gauging station in each catchment coloured by the dominant land cover (≥ 50%); blue = arable, orange = grassland, green = woodland and red = urban.

Percentage land cover within a catchment was derived from the 2015 data documented in the CEH Land Cover Change 1990-2015 database (Rowland et al., 2020). We focused on four of the six land cover types categorised in the database: woodland, arable, grassland and urban (excluding categories 'water' and 'other'), the subcategories of which can be found in Appendix B.1. Average land cover across the natural catchments was grassland (61.2 %), woodland (17.3 %), arable (14.2 %) and urban (5.9 %). Water and 'other' coverage are 0.46 % and 1.0 % respectively (Figure 4.2a). Conifer woodland coverage was higher than broadleaf woodland (10.6 and 5.7 % respectively) (Figure 4.2b).

The statistics for categorizing the permeability of superficial deposits were derived from the classifications of the BGS (British Geological Survey) 1:625000 Superficial Deposits layer (Version 5), see Appendix B.2. Mean annual rainfall (1226.72 mm) over a catchment was taken from the daily catchment rainfall series held on the NRFA.

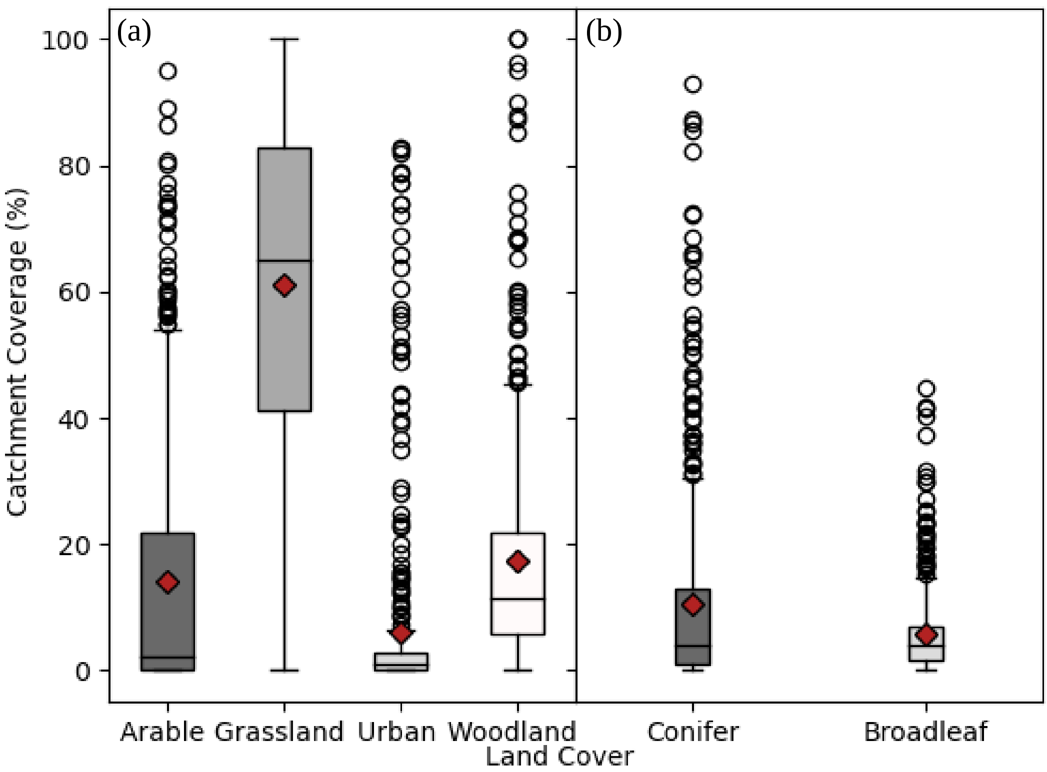

Figure 4.2 Percentage land cover within natural catchments (line: median, box: IQRs and whiskers: 5th and 95th percentiles, diamond: mean) for (a) major land cover types (b) woodland by type; conifer and broadleaf.

4.3.2. Active storage, m

Active storage, m was calculated for 418 natural catchments using river flow data from 1930s to 2018 (median length of flow data = 38 years). We calculated m from the recession in flow following a storm event, as used in the hydrological simulation model, TOPMODEL (Beven and Kirkby, 1979) using equation (4.4):

$$\frac{dj}{dt} = \frac{j(i-j)}{m} \tag{4.4}$$

Where j is the current runoff (discharge per unit area in mm·day^{-1}), and i is the net rainfall intensity (mm·day^{-1}).

Runoff, j, is calculated from discharge, q (m^3·s^{-1}) and drainage area, A (km^2) using equation (4.5):

$$j = \frac{q * 86400}{A * 1000} \tag{4.5}$$

We calculated m from equation (4.4) using December runoff data (when we can assume evapotranspiration is negligible) and for periods with no precipitation (Beven et al., 1984;

Gao et al., 2015). Under these conditions net rainfall (i) is zero and equation (4.4) can then be re-written in the form:

$$-\frac{1}{j}\frac{dj}{dt} = \frac{j}{m} \tag{4.6}$$

For each catchment the left-hand side of this expression can be plotted against the current runoff (*j*) to obtain a straight-line relationship with gradient 1/ m (Figure 4.3).

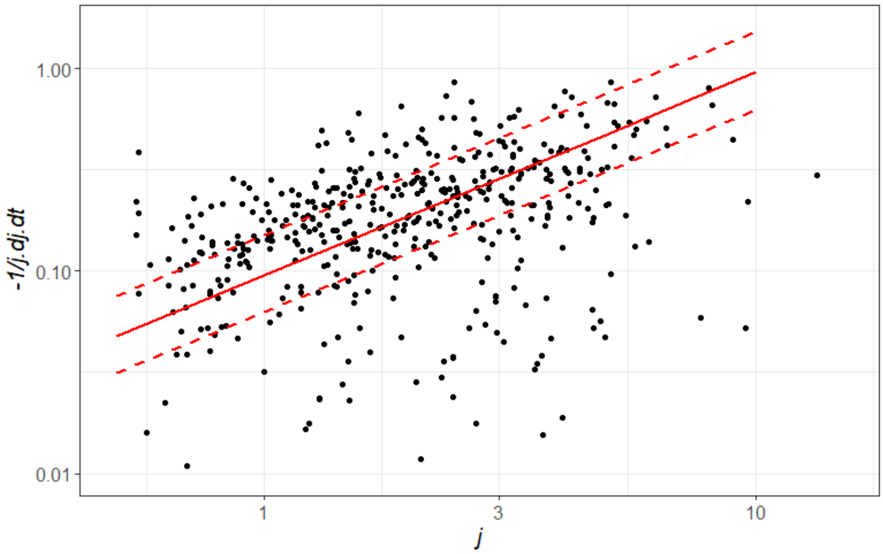

Figure 4.3 Example regression from NRFA Station 1001. Plotted points show all December recession curve data for 1995-2017. m-values are calculated for each data point. Solid line shows median value and dotted lines 25% and 75% percentiles.

The active storage available depends on storm rainfall and antecedent conditions. Applying TOPMODEL for an impulse storm of magnitude *R* (mm), when the antecedent runoff was j_0, and after a time delay or *T* (~ 24 hours, say), the fraction, p of storm rainfall retained as active storage is obtained by solving equation (4.6) to produce equation (4.7):

$$p = 1 - \frac{m}{R} \ln\left[1 + \frac{j_0 T}{m} exp\left(\frac{R}{m}\right)\right] \tag{4.7}$$

Where R = total storm rainfall (mm),

T is storm duration (hours) and

j_0 is initial pre-storm runoff (mm·hr^{-1})

The proportion of storm rainfall stored varies with m, for a range of sizes (Figure 4.4a) and antecedent runoff conditions (Figure 4.4b).

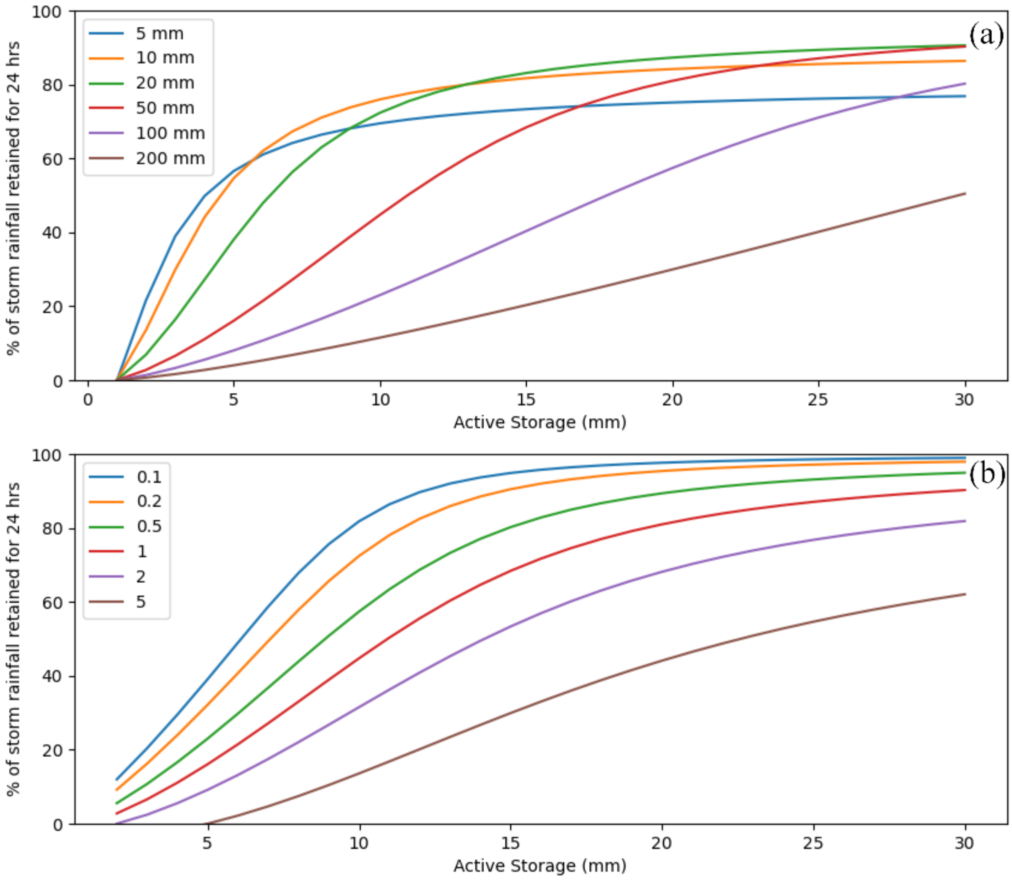

Figure 4.4 Storage of water in the soil as a function of active storage (a) after T=1 day as a percentage of storm size, expressed as a function of the active storage parameter (TOPMODEL m), for different storms sizes. Curves are drawn for pre-storm runoff of j_0=1 mm·day^{-1}.There is a consistent, but non linear increase in % storage with increasing m. b). Active storage percentage after T=1 day, as a function of active storage parameter (TOPMODEL m), for different values of antecedent runoff, j_0T, showing largest storage when antecedent runoff is least.

4.3.3. Change in active storage, m

We calculated the change in active storage, m for 354 of the 418 natural catchments with a flow data record between 1985 and 2017 (median (range) flow data = 32 years). m was calculated for December each year and a linear least-squares regression was then used to calculate the slope of change in active storage over the time series to establish the rate of change in active storage recorded (mm) per year (Equation 4.8; where f is the slope and b is the intercept):

$$y = fx + b \tag{4.8}$$

4.3.4. Data analysis

4.3.4.1. Impact of land cover on m and change in m

To isolate the impact of land cover change on m and change in m we selected catchments where one land cover covered ≥ 70 % of the catchment. All catchments dominated by woodland are all conifer woodlands (Figure 4.2b). We then calculated the catchment storage and change in catchment storage over time for each land cover type.

4.3.4.2. Statistical analysis

Shapiro–Wilks tests were used to identify the normality of active storage, *m* and change in active storage data sets. Non-parametric Kruskal–Wallis comparisons were used to determine a significant difference (significance determined at $p < 0.05$) between land covers. Further post-hoc tests were used to identify which land covers were significantly different to each other. Statistics were performed using SciPy (Virtanen et al., 2019) and scikit (Terpilowski, 2019) packages in Python.

4.3.4.3. Modelling m and change in m

We used generalized linear models (GLMs) fitted with a gamma distribution and a log-link function to examine which environmental variables (Table 4.1) have the greatest influence on determining m in natural catchments. Average rainfall, catchment size and all land cover variables were right skewed and were therefore log-transformed (rainfall and catchment size) and square-root transformed (land covers) prior to analysis. To examine whether woodland type had an influence on determining *m*, *m* was modelled again separating woodland coverage into conifer and broadleaf. We also used GLMs to examine whether land cover or percentage change in land cover between 1985 and 2015 determine change in *m* in natural catchments.

We tested for collinearity between predictor variables, using the Variance Inflation Factors (VIF) (Freckleton, 2011) and the R package 'performance' (Lüdecke et al., 2020). We ensured variables used to model active storage had VIFs < 5. Due to the large number of environmental variables, model simplification and selection was undertaken using Akaike's Information Criterion (AICc). Where there was no clear top-model (ΔAICc < 2; Burnham and Anderson (2002)) model averaged parameters and their relative importance were extracted from all models with ΔAICc < 2 using the R package 'MuMIn' (Barton, 2015). All analyses were undertaken in R (v 4.1.0.)

Table 4.1 Environmental variables used in GLMs to predict active storage and change in active storage in natural catchments.

Variable	Unit	Transformation for use in GLM
Woodland cover	%	√
Urban cover	%	√
Arable cover	%	√
Grassland cover	%	√
High permeability superficial deposits	%	-
Low permeability superficial deposits	%	-
Catchment Size	km^2	ln
Average annual rainfall	mm	ln
Average catchment elevation	m	-

4.3.4.4. Impact of grazing on m

To isolate the potential impact of grazing on *m* we selected grassland-dominated catchments (≥ 70% grassland), where no or little change in land cover had taken place (+/-2 %) between 1985 and 2017 (n = 62). Active storage, *m* was then calculated for 1985-2005, during high sheep numbers (20.7 million in 1990) and 2006-2017 low sheep numbers (14.3 million in 2011).

4.3.4.5. Trends in land cover change and change in active storage

We identified whether a land cover type had increased or decreased within the catchment by more than a threshold percentage between 1990 and 2015 using the CEH change in land cover dataset. We compare the changes in active storage ($mm \cdot year^{-1}$) for increasing and decreasing cover. This was done both for a zero threshold (i.e. any non-zero change) and for a 7 % threshold of change.

4.4. Results

4.4.1. Impact of land cover on active storage, m

Active catchment storage, *m* across natural catchments ranged between 0.10 – 57.49 mm (median = 11.13 mm) (Figure 4.5).

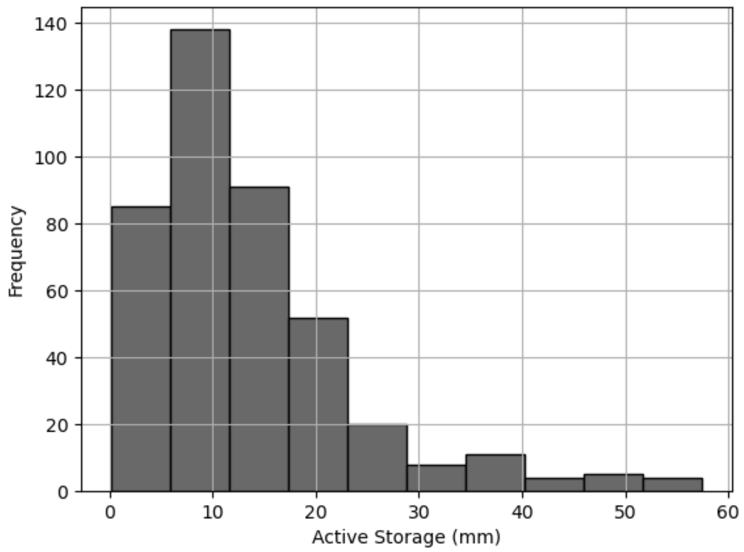

Figure 4.5 Active storage across natural catchments in the UK (n = 418).

In catchments with a dominant, ≥ 70 %, land cover (n = 223), catchments dominated by grassland (n = 190) are larger than catchments dominated by other land cover types (n = 33) (mean for grassland 89.7 km^2, mean for other land covers varied between 2.75 and 25.5 km^2) (Appendix B.3).

There was a significant difference in active storage between land cover types (Kruskal-Wallis, $p < 0.05$). Grassland catchments have significantly greater active storage than urban and arable catchments ($p < 0.05$) with no significant difference between grassland and woodland catchments (Figure 4.6). The majority (~90 %) of urban catchments have an active storage value less than 10 mm compared to grassland catchments, where approximately 90 % of catchments have an active storage value more than 10 mm (Appendix B.4).

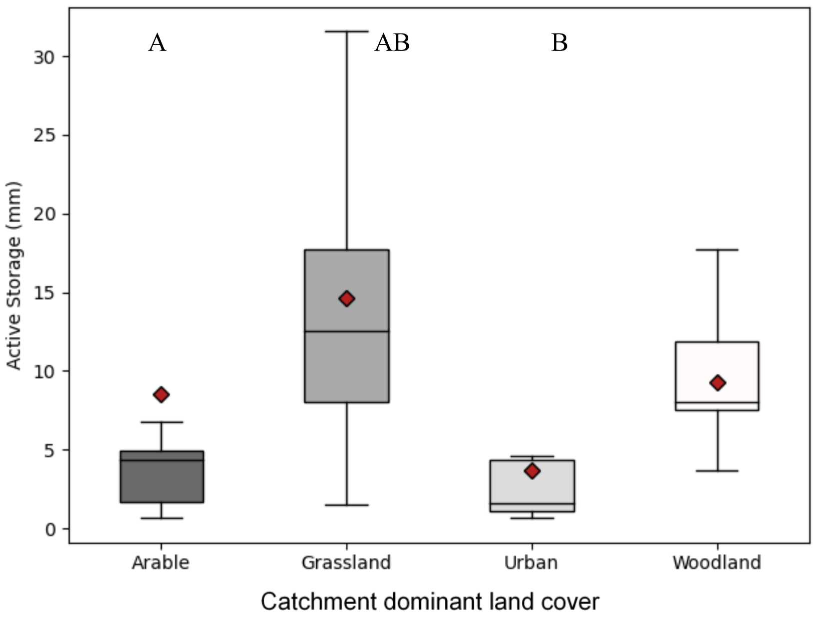

Figure 4.6 Active storage for catchments with ≥ 70 % of the catchment one land cover type (n = 223) (line: median, box: IQRs and whiskers: 5th and 95th percentiles, diamond: mean). Sites that are statistically different share a letter.

There was no clear top model (AICc < 2) to explain active catchment storage, but model averaging found catchment size, rainfall, arable land cover, grassland land cover and low superficial deposits to be the most important environmental variables (Table 4.2). Woodland cover either as conifer and broadleaf woodland combined or as separate land cover classes did not have a significant impact on active storage (Table 4.2, Appendix B.5 and B.6).

Table 4.2 Model-averaged estimates for factors affecting active storage (m) ; n = 4 models. GLM top model set can be found in Appendix B.7). Significant variables in bold text.

Variable	**Estimate**	**Std. error**	***p***	**Relative importance**
Woodland cover (sqrt)	2.27E-02	3.08E-02	0.37	0.66
Urban cover (sqrt)	**-6.92E-02**	**3.28E-02**	**<0.05**	**1**
Arable cover (sqrt)	**9.97E-02**	**2.80E-02**	**<0.001**	**1**
Grassland cover (sqrt)	**9.07E-02**	**3.98E-02**	**<0.01**	**1**
High permeability superficial deposits	-2.55E-04	2.08E-03	0.90	0.13
Low permeability superficial deposits	**-7.47E-03**	**2.60E-03**	**<0.001**	**1**
Catchment size (log)	**9.46E-02**	**3.86E-02**	**<0.05**	**1**
Average annual rainfall (log)	**8.44E-01**	**1.56E-01**	**<0.0001**	**1**
Average catchment elevation	-6.50E-04	5.21E-04	0.21	0.82

4.4.2. Impact of land cover on change in active storage, m

Between 1985 and 2017 the median change in active storage is 0.03 mm·year^{-1} (range between -2.49 - 3.96 mm·year^{-1}). Over this period active storage increased across 60 % (n = 213) of catchments and decreased across 40 % (n =141) catchments (Figure 4.7a).

A significant (test statistic = $p < 0.05$) change in active storage was identified in 35 catchments with a mean change of 0.01 mm·year^{-1}, whilst the median change is 0.09 mm·year^{-1} (minimum = -2.49 mm·year^{-1}, maximum = 1.32 mm·year^{-1}). Active storage increased across 57 % (n = 20) of these catchments and decreased across 43 % (n = 15) of catchments (Figure 4.7b).

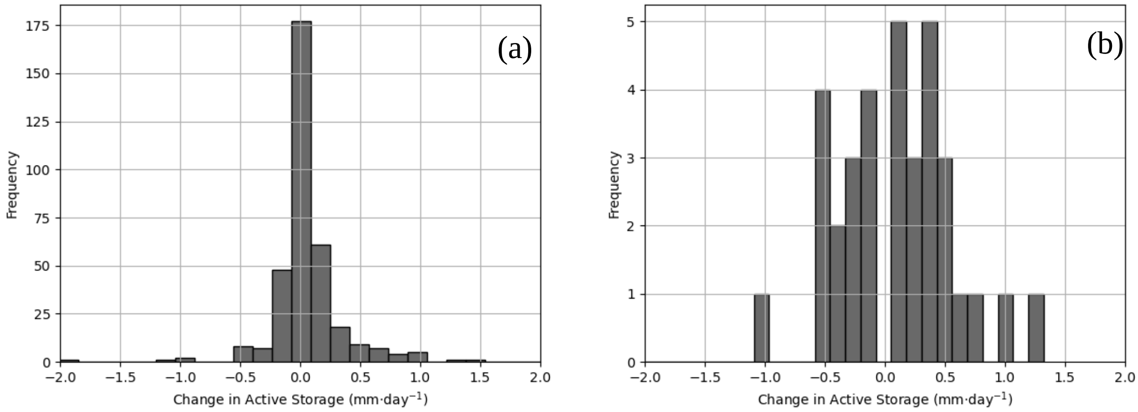

Figure 4.7 (a) Change in active storage over time (n=354) and (b) significant change in active storage over time (n=35).

Within catchments identified as having a dominant land cover type (≥ 70 %), the median change in active storage was positive in woodland (0.06 $mm \cdot yr^{-1}$) and grassland (0.03 $mm \cdot yr^{-1}$) catchments and negative on arable (-0.0002 $mm \cdot yr^{-1}$) and urban (-0.02 $mm \cdot yr^{-1}$) catchments (Table 4.3, Figure 4.8). However, there was no significant difference in the change in active storage between land cover types (Kruskal-Wallis, $p > 0.05$).

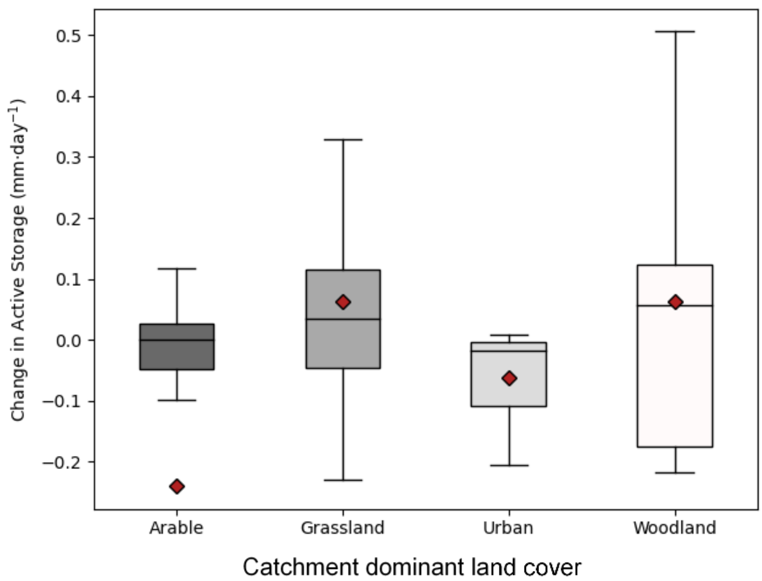

Figure 4.8 Change in active storage over time for catchments with ≥ 70% of the catchment one land cover type to the 5^{th} and 95^{th} percentiles (n = 186).

Table 4.3 Median change in storage over time (mm·year^{-1}) for catchments with ≥70 % coverage of one land cover type.

≥70% Catchment Land Cover Type	***n***	**Median Change in Active Storage parameter, *m* (mm·year^{-1})**	**Standard deviation (ddof =1)**	**Median catchment size (km^2)**	**Median Annual Rainfall (mm)**
Woodland	9	0.06	0.26	2.28	2000
Arable	10	-0.0002	0.79	39.75	645
Grassland	160	0.03	0.35	103.05	1300
Urban	7	-0.02	0.09	17.64	655

We found no relationship between change in active storage and either current land cover or change in land cover between 1990 and 2015 (see Appendix B.8), therefore we did not carry out GLMs for change in active storage.

4.4.3. Impact of sheep number/grazing on active storage

For grassland dominated catchments (≥ 70 % grassland) we compared active storage, *m* between 1985-2005 and 2005-2017. Median active storage for grassland dominated catchments was slightly higher for 2005-2017 than 1985-2005 (13.39 mm and 13.07 mm respectively). However, we found no significant difference (Kruskal-Wallis, $p > 0.05$) between active storage calculated for the different time periods (Figure 4.9).

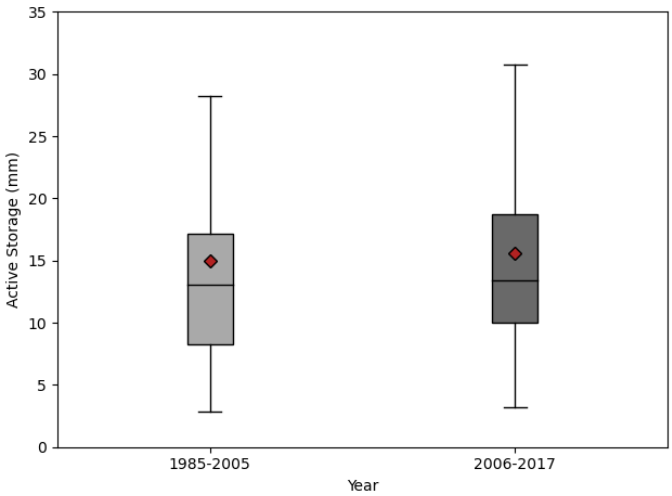

Figure 4.9 Active storage for grassland-dominated (≥ 70 %) catchments in 1985-2005 and 2006-2017 to the 5th and 95th percentiles (n = 62).

4.4.4. *Trends in land cover change and change in active storage*

Between 1990 and 2015, there was a small decline in median grassland cover (-2.95 percentage points) and small increase in median woodland (+1.5 percentage points) and urban (+0.33 percentage points) cover. At the individual catchment scale, change in land cover varied from -37 to +37 percentage points with the largest changes due to decline in grassland and an increase in woodland cover (Table 4.4).

Table 4.4 The change in land cover between 1990 -2015 (n = 418).

Land Cover	Percentage point change in land cover (%)		
	Maximum Loss	Maximum Gain	Median
Woodland	-3.55	36.65	1.50
Arable	-19.95	24.26	0.00
Grassland	-36.70	19.18	-2.95
Urban	-4.58	13.94	0.33

We investigated whether a change in land cover correlated with a change in active storage. The median change in active storage was slightly positive regardless of whether any type of land cover increased or decreased (Figure 4.10). An increase in arable land cover above 7 percentage points (%) correlated with a median increase in active storage, whilst an increase in grassland cover above 7 percentage points (%) correlated with a median decrease in active storage (Figure 4.10b). There are no catchments were woodland or urban cover decreased by more than 7 percentage points (%). There is no significant difference between change in active storage when individual land cover groupings increased or decreased e.g. no significant difference between change in storage of an increase in grassland and change in storage when there is a decrease in grassland.

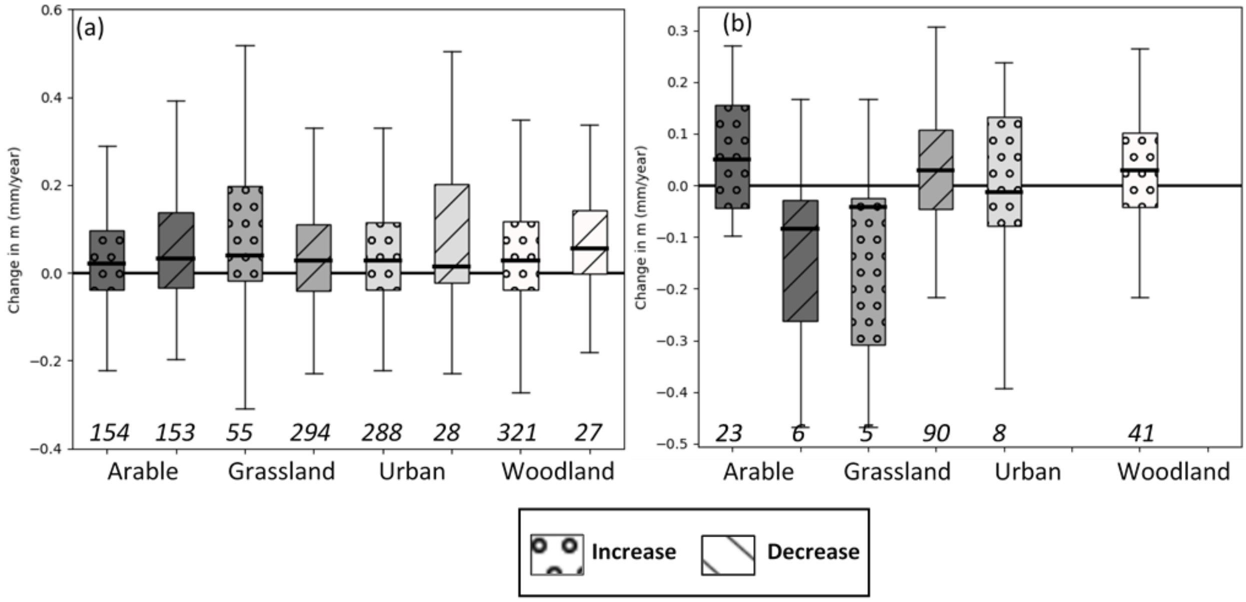

Figure 4.10 Change in active storage, mm/year (calculated from 1985 to 2017) in catchments which experienced land cover change between 1990 and 2015 a) threshold 0 percentage points and b) threshold 7 and -7 percentage points (line: median, box: IQRs and whiskers: 5th and 95th percentiles). Number of catchments (n) where land cover has increased or decreased labelled.

4.5. Discussion

4.5.1. Active Storage, m

We found that grassland and woodland-dominated catchments had the greatest active storage. Grassland-dominated catchments had significantly greater storage than arable and urban catchments. Analysing the impact of land cover on active storage was complicated by distribution of land cover across the UK. Natural catchments in the UK are dominated by grassland (mean coverage 61.2 %) with substantially lower coverage of woodland (17.3 %), arable (14.2 %) and urban (5.9 %) land covers. This means there was a larger sample size (n = 190) of grassland-dominated catchments compared with other land cover types. Grassland-dominated catchments were on average larger than catchments dominated by other catchments.

Estimates of active storage were calculated using December only runoff data in order to assume negligible evapotranspiration. In this study the parameter m controls for antecedent conditions and is calculated across multiple storms. However, during months of higher levels of evapotranspiration there may be changes to active storage. Furthermore, this investigation was limited to NRFA termed 'natural' catchments, this may have omitted other patterns or trends in the entirety of the UK runoff dataset.

Our GLM analysis suggests that the extent of grassland, arable and urban land covers have a significant impact on the active storage of a catchment. In particular, catchment storage declined with increased urban cover (Table 4.2). Woodland cover did not have a significant impact on storage. Average annual rainfall, catchment size, elevation and whether the superficial deposits have a low permeability were also significant. In the UK, grasslands are mainly located in the west where rainfall is greater, further complicating influences of land cover from other variables. Our analysis points to controls of geology on catchment storage which have been difficult to show empirically or understand mechanistically (Pfister et al., 2017).

Urban land covers have low infiltration due to the presence of impermeable surfaces (Bronstert et al., 2002; Wheater, 2006), which likely contributes to the low storage seen in our analysis. Livestock grazing can cause changes in vegetation structure and soil compaction leading to reductions in soil permeability (O'Connell et al., 2004; Meyles et al., 2006; Wheater and Evans, 2009; Alaoui et al., 2018). In contrast, we found that grassland catchments had high storage. Woodland catchments also have high active storage due to root water uptake by trees and more efficient macropores found in woodland soils (Alaoui et al., 2011).

Large impacts of land cover on peak flow are apparent at small spatial scales (Rogger et al., 2017; Monger et al., 2021) but this impact tends to become smaller as scale increases (Dadson et al., 2017; Peskett et al., 2021). The impact of land cover on flood mitigation is thought to decrease at larger scales due to non-managed environmental characteristics which vary across a catchment i.e. slope, mask the impact invariant controls (Beven, 2000; Soulsby et al., 2006). Furthermore, the spatial heterogeneity found within a catchment can dilute the impact of managed environmental characteristics, making the signal between differing land covers and storage noisy and less distinguishable at a catchment scale compared with hillslope or plot scale (Spence, 2007; Soulsby et al., 2008; Teuling et al., 2010; Tetzlaff et al., 2011). Our analysis suggests that land-cover still has an important effect on catchment storage, even at scales of ranging from 1 to 9866 km^2 with a median catchment size of 72 km^2 in our analysis.

We found that median active storage for grassland-dominated catchments was slightly higher for 2005-2017 than 1985-2005 (13.39 mm and 13.07 mm respectively). An increase in catchment storage could be related to reductions in sheep numbers that have occurred since the early 2000s (Milligan et al., 2016). Whilst relatively little is known about the effects of reducing stock grazing pressures, studies suggest it may take 48-62 years to see the benefits of reducing sheep grazing due to slow rates of recovery (Marrs et al., 2018; Marrs et al., 2020). Clearly the catchment-wide approaches we use here hide detail regarding changes to grazing management strategies in individual catchments.

Whilst the GLM model averaging used in this study identified some influencing factors for active storage, there is a large proportion of unexplained variance, suggesting a variable(s) not considered in this study may have a big impact on active storage. We tried to take into consideration the impact of location, using variables such as rainfall and elevation, as the UK can be generalised into predominately flatter and more arable land to the south east, whilst more undulating and often grazed grasslands. However, the slopes, elevation change within a catchment has not been considered.

4.5.2. Change in active storage

We found no significant change in catchment storage related to changes in land cover, perhaps due to the relatively short time frame investigated. Land cover changes in the UK over the period analysed were relatively modest with median changes at the catchment scale of less than 3 percentage points (Table 4.4). Previous studies have isolated specific UK catchments with larger changes in land cover and found increased runoff and high flows in urban catchments (Prosdocimi et al., 2015; Putro et al., 2016). Some modelling studies suggest larger changes in land cover can result in significant catchment-scale impacts (Reynard et al., 2001) and might be expected to cause more substantial changes in storage than seen here. However, Buechel et al. (2022) simulated widespread afforestation of large (500 – 10, 000 km^2) catchments and found no consistent change in extreme floods. Analysis of changes in catchment storage in regions where there have been more extensive recent changes in land cover is now needed. Furthermore, the percentage change in catchment active storage rather than actual change could be explored.

4.5.3. Land-use for flood management

There is continued interest in potential of using land cover as cost-effective method of NFM, however uncertainty surrounds whether land-use management can make an impact, particularly at the catchment scale (Burgess-Gamble et al., 2017; Lane, 2017). Our results suggest that land cover does have an impact on storage at the catchment scale. In particular woodland and grassland dominated catchments store significantly more water than urban and arable catchments. We found less evidence for changes in catchment storage that could be linked to changes in land use or land use management. We found small increases in active storage in woodland and grassland-dominated catchments possibly linked to woodland development and following reductions in grazing, respectively. Our study is one of the first to estimate and compare catchment storage for numerous catchments using the same method. Used in this way storage may provide useful metrics for catchment comparison, especially when developing a NFM strategy.

4.6. Conclusion

We calculated active storage using December runoff data for 418 'natural' river catchments across the UK and analysed the relationship between active storage and land cover. We found that grassland and woodland catchments had higher active storage compared to arable and urban catchments. Our analysis shows that land cover has detectable impacts on storage even at a larger scale up to nearly 10,000 km^2, despite the diluting effects of catchment heterogeneity. We analysed changes in active storage over 1985 to 2017 and estimate larger increases in storage in woodland and grassland catchments compared to urban catchments. Our analysis provides evidence that land cover and land cover management can provide flood mitigation benefits even at a larger catchment scale providing additional support for ongoing NFM interventions.

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Chapter 5. Investigating the impact of woodland placement and percentage cover on flood peaks in an upland catchment using spatially distributed TOPMODEL

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5.1. Abstract

Woodlands can reduce downstream flooding, but it is not well known how the extent and distribution of woodland affects reductions in peak flow. We used the spatially distributed TOPMODEL to simulate peak flow during a 1 in 50 year storm event for a range of broadleaf woodland scenarios across a 2.6 km^2 catchment in Northern England. Woodland reduced peak flow by 2.6-15.3 % depending on the extent and spatial distribution of woodland cover. Cross slope and riparian woodland resulted in larger reductions in peak flow, 3.9 % and 3.5 % for a 10-percentage point increase in woodland cover respectively, compared to a 2.1 % reduction for randomly located catchment woodland. Our results demonstrate that increased woodland cover can reduce peak flows during a large storm event and suggest that targeted placement of woodland can maximise the effectiveness of natural flood management interventions.

5.2. Introduction

Flooding is one of the most costly and widespread climate-related natural hazards (Jonkman, 2005), accounting for 44 % of all disaster events from 2000 to 2019 and affecting 1.6 billion people worldwide (UNDRR, 2020). Anthropogenic climate change is predicted to increase the frequency of extreme precipitation events, subsequently increasing the risk of flooding (Tabari, 2020). This has increased interest in natural flood management (NFM) as a means to mitigate flood risk. NFM approaches aim to mitigate against flood risk using natural processes (Cooper et al., 2021).

Woodland creation, is increasingly seen as an important way to deliver flood mitigation (Murphy et al., 2020). Forested catchments have a different hydrological response compared to un-forested catchments due to greater interception, evaporation, soil infiltration and available storage (Monger et al., 2021; Stratford, 2017; Nisbet, 2005; McCulloch and Robinson, 1993; Page et al., 2020). For the purposes of NFM, woodland can be divided into four types (Cooper et al., 2021):

- *Catchment woodland* - defined as the total area of all woodland within a catchment, comprising woodland cover of all types.
- *Cross-slope woodland* - the placement of smaller areas of woodland across hill slopes, broadly following the contours.
- *Floodplain woodland* – comprised of all woodland lying within the fluvial floodplain that is subject to a regular or natural flooding regime.
- *Riparian woodland* - woodland located within the riparian zone, defined as the land immediately adjoining a river channel and influenced by it (Burgess-Gamble et al., 2017; Cooper et al., 2021).

Cooper et al. (2021) reviewed the literature on these four woodland types as NFM strategies. The impacts of catchment woodland on the hydrological cycle has long been studied (Best et al., 2003; Law, 1956; Bosch and Hewlett, 1982). Catchment woodland can produce lower annual runoff compared to other land cover types, as demonstrated in numerous catchment-based studies including Stocks Reservoir (Law, 1956), Plynlimon (Hudson et al., 1997; Kirby et al., 1991); Coalburn (Birkinshaw et al., 2014; Robinson et al., 1998) and Balquhidder (Johnson, 1995). Peak flows are also found to be reduced in catchment woodlands (Monger et al., 2021), however this can be dependent on the size of the storms (Xiao et al., 2022; Dadson et al., 2017; Soulsby et al., 2008; Fahey and Payne, 2017; Archer, D., 2007).The potential for afforestation to reduce runoff peak flows during storm events is still not clear (Burgess-Gamble et al., 2017; Peskett et al., 2021; Stratford, 2017) particularly for larger catchments (Rogger et al., 2017). Cross-slope woodland has been investigated at the Pontbren experiment with the construction of shelter-belts, resulting in increased soil infiltration rates (up to 60 times greater under wooded areas than open grazed pasture) within 2–6 years (Carroll et al., 2004). Modelling studies of floodplain woodland regeneration (Connell, 2008; Thomas and Nisbet, 2007) report reductions in flow velocity and increases in water levels in the regenerated areas at small scales. Floodplain woodland restoration could also lead to delays in peak flow as well as desynchronizing the flood peaks from adjacent tributaries (Cooper et al., 2021). Modelling studies of riparian woodland often focussed on the impacts of in-channel wood and installation of “Large Woody Dams” within watercourses (Thomas and Nisbet, 2012; Burgess-Gamble et al., 2017). However, there is a dearth of numerical models of forest growth, with none applicable to the UK environment (Cooper et al., 2021; Dixon et al., 2019). More work is needed to understand how these different woodland management strategies may impact downstream flood peaks.

Hydrological modelling tools can be used to simulate the impacts of different land covers on peak flow (IHACRES, (Jakeman et al., 1990); TOPMODEL, (Beven and Kirkby, 1979); ReFH, (Kjeldsen, 2009)). More recently, models have been developed that allow investigation of different spatial land cover patterns (HBV, (Bergström, 1976); SD-TOPMODEL, (Gao et al., 2015); SWAT, (Arnold et al., 1998)). These models allow the potential outcomes of NFM interventions to be simulated before they are implemented offering insight to practitioners and policy makers (Gao et al., 2017). A modelling study by Gao et al. (2017) found that revegetating areas of bare peat with *Sphagnum* in riparian zones altered flow peaks up to 3 times as much as those same changes in headwater areas. This suggests that the location of land cover interventions is an important consideration when developing strategies to reduce flood risk (Peskett et al., 2021; Murphy et al., 2020). However, little is known about how the spatial distribution of woodland impacts flood risk.

In recent years the UK has also experienced notable flood events (Chatterton et al., 2016; Marsh et al., 2016; Schaller et al., 2016) causing significant economic and environmental damage (Priestley, 2017; Murphy et al., 2020) and increasing interest in woodland creation as NFM. The UK is one of the least densely wooded countries in Europe (Forestry Commission, 2021b). At present woodland covers 13.2 % (3.2 million ha) of the UK's land surface, up from 12 % cover in 1998 (Reid et al., 2021). In England, woodland covers 10 %, compared with 15 % in Wales, 19 % in Scotland and 9 % in Northern Ireland (Forestry Commission, 2021a). However the interest in and demand for woodland creation is growing, as woodlands are being increasingly viewed as key to simultaneously tackling both the climate and nature crises (Reid et al., 2021). DEFRA published the England Trees Action Plan 2021-24 in May 2021 (UK Government, 2021), where the UK Government pledged to increase tree planting rates across the UK to 30,000 hectares per year by May 2024 (UK Government, 2021). In addition, the Welsh government published 'Woodlands for Wales' (Welsh Government, 2021) and Scotland, a 'Forestry Strategy 2019–2029' (Reid, 2018). With plans for woodland cover to increase throughout the UK, it is important we understand what impact this could have on catchment hydrology and where to focus new plantings to provide the maximum benefits.

In this study, we used the spatially distributed TOPMODEL (SD-TOPMODEL) (Gao et al., 2016; Gao et al., 2015) to investigate how different spatial configurations of woodland cover within an upland catchment in northern England influences the flood peak. We provide the first consistent comparison of the impacts of catchment, riparian and cross-slope woodland on peak flow using the same modelling framework.

5.3. Study site

We selected the upland catchment, Naddle, in Cumbria, UK (54°31'50.9"N, 2°45'37.3"W) as the study site for this investigation (Figure 5.1). The catchment area covers 2.62 km^2 and is managed by the RSPB (The Royal Society for the Protection of Birds) on behalf of the landowners, United Utilities.

The catchment experiences mild winters and cool summers (Kenworthy, 2014), with mean monthly temperatures ranging from -0.3 °C to 18.3 °C and mean annual precipitation of 1779 mm, with monthly rainfall ranging from 88 to 231 mm (1981-2010 mean, Shap weather station at 255 m AoD, (Met Office, 2020).

The catchment consists of unimproved permanent pasture, grazed at a variety of densities, and woodland. The woodland is predominantly semi-natural ancient woodland designated as a site of special scientific interest (SSSI) fenced to exclude livestock and a small conifer plantation which has been recently felled to allow for natural regeneration. Soils in the study area are upland organo-mineral soils, predominately Malvern 611a (Chromic Endoleptic Umbrisol), a free draining acid loamy soil (Cranfield University, 2019).

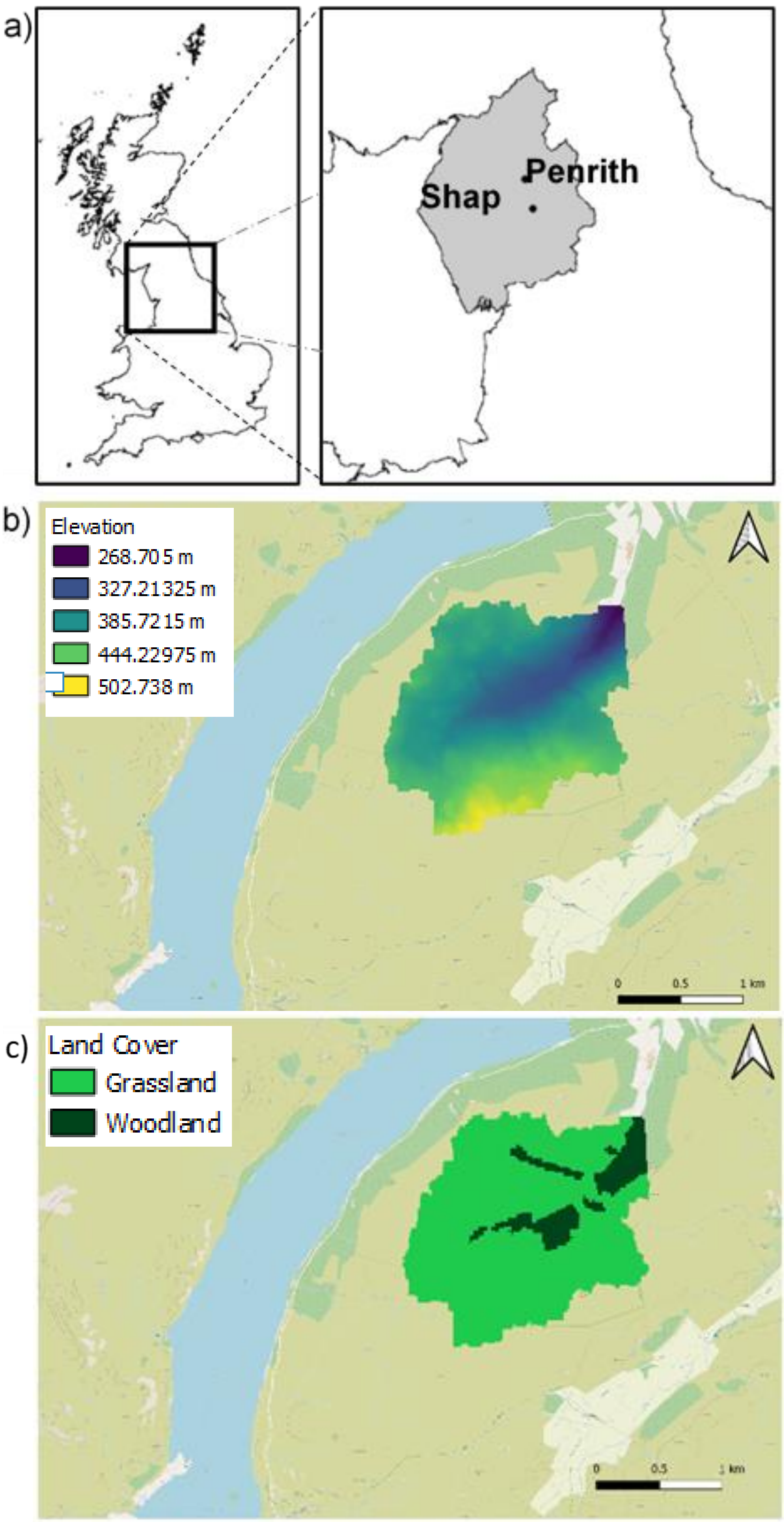

Figure 5.1 a) Map of the Naddle catchment location within the UK, b) Elevation (m) across the Naddle catchment and c) Current land cover in Naddle simplified into two classifications, 12.9 % woodland and 87.1 % grassland.

5.4. Methodology

5.4.1. SD-TOPMODEL

TOPMODEL is a rainfall-runoff model originally developed by Beven and Kirkby (1979) as the lumped or semi distributed TOPMODEL. We use the spatially distributed version of TOPMODEL (SD-TOPMODEL) (Gao et al., 2015). In this version, the hydrological behaviour of each cell in the Digital Elevation Model (DEM) data is calculated individually, with the original TOPMODEL hydrological equations downscaled from catchment scale to cell scale. In addition, subsurface flow and overland flow are treated separately making it a useful tool to examine the impact of land cover on flow (Gao et al., 2015). SD-TOPMODEL has been used to simulate the impact of land-cover change in peatlands (Gao et al., 2016), agricultural practices (Gao et al., 2017) and landscape features, such as stone walls and hedges (Willis and Klaar, 2021) on flood peaks in upland catchments.

Land cover in SD-TOPMODEL is represented using four key parameters; hydraulic conductivity of the soil *(K)*, interception (*i*), a scaling parameter representing the active water storage in soil *(m)* and an overland flow velocity parameter related to surface roughness, and equal to $1/n$ where n is Manning's roughness (K_v). Model parameters, excluding *m*, can vary spatially based on the land cover in the simulations, and the map of each parameter can be used to describe the heterogeneous properties of the catchment (Gao et al., 2017). *m* is calculated for each catchment and therefore used as a lumped parameter and unsuitable for spatial distribution.

5.4.2. Representing land cover in SD-TOPMODEL

Empirical data collected from the Naddle catchment informed spatially distributed *K* and *Kv* values within the model (Monger et al, 2021; Monger et al, 2022), see Table 5.1. For grassland, the median low-density grazing (pasture) Ksat (Monger et al., 2021) and median wood pasture dominated by grassland overland flow velocity (Monger et al., 2022) were used. The established semi-natural woodland median overland flow velocity (Monger et al., 2021) and median Ksat (Monger et al., 2022) values were used to inform the woodland in Naddle. The most conservative estimate for an Oak woodland in winter was used to represent interception (Dolman, 1987).

Table 5.1 Empirical land cover values

	Saturated hydraulic conductivity, Ksat ($m \cdot hr^{-1}$)	**Overland flow velocity ($m \cdot s^{-1}$)**	**Interception, i (%)**
Grassland	1.47×10^{-4} (Monger et al., 2021)	0.031 (Monger et al., 2022)	0
Woodland	2.94×10^{-3} (Monger et al., 2021)	0.028 (Monger et al., 2022)	13 (Dolman, 1987)

Saturated hydraulic conductivity, Ksat is a proxy for parameter *K* whilst overland flow velocity is a proxy for parameter *Kv*.

Relative values for all parameters, excluding *m*, are used when spatially distributing land cover in SD-TOPMODEL, see Table 5.2. For example woodland has a *K* value 20 times that of grassland.

Sensitivity analysis of spatial distribution of parameters shown in Appendix C.3.

Table 5.2 Relative land cover values used in SD-TOPMODEL

	K	***Kv***	***i***
Grassland	1	1	1
Woodland	20	0.9	0.87

5.4.3. Model calibration

Due to limited (less than 18 months) runoff and rainfall data available for the Naddle, a ReFH (Revitalised Flood Hydrograph) storm was used to fit a storm in SD-TOPMODEL. A 1 in 50 year event for a 24 hour time frame was produced. A 25 by 25 m DEM, delineated in QGIS is used to inform the model of the catchment boundaries and provide elevation data to the model (Figure 5.1b).

We used 50 test runs of the model to identify a good performing set of parameters (m = 6 mm, K = 6.9 $m \cdot hr^{-1}$, k_v = 40, see Appendix C.1). There was good correspondence between best fit simulated and observed flow in the calibration period (the Nash-Sutcliffe efficiency was 0.85) (Figure 5.2).

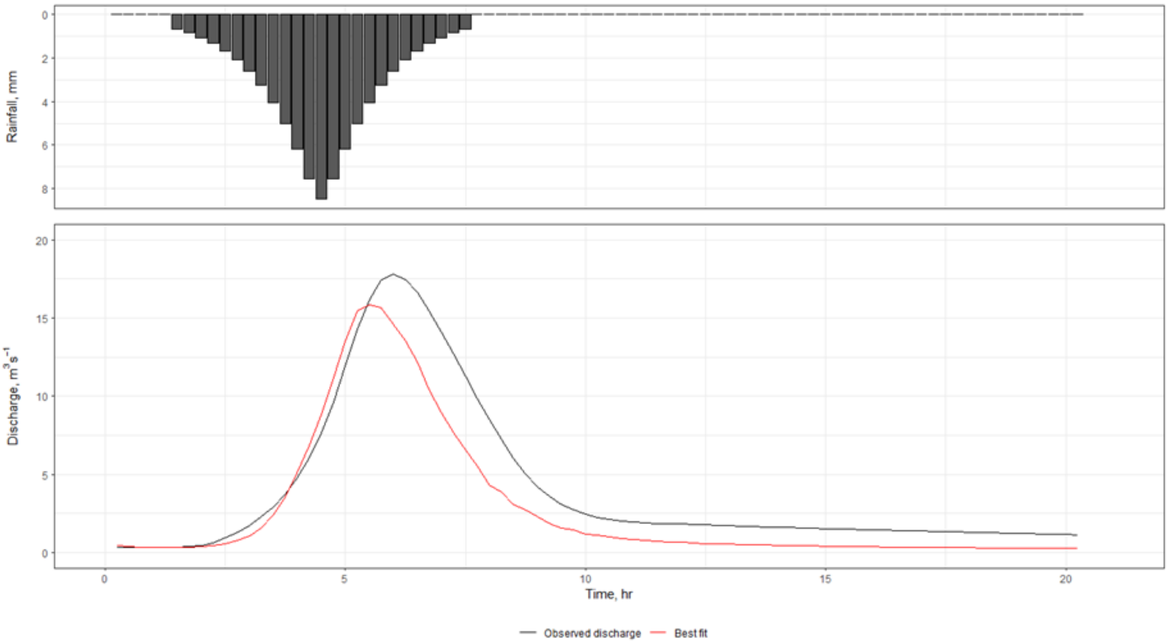

Figure 5.2 Hydrograph response to the 1 in 50-year ReFH storm, observed discharge and SD-TOPMODEL modelled discharge. A Nash-Sutcliffe efficiency (NSE) of 0.85, the best fit model.

5.4.4. Modelled woodland scenarios

We investigated a number of woodland scenarios described in Table 5.3 (scenario land cover maps included in Appendix C.2). Our woodland scenarios were intended to assess the potential of the different classes of woodland creation options in NFM interventions, including catchment, riparian and cross slope woodland.

Whilst we calibrated the model to fit the current Naddle land use, we compared the scenarios against a *no woodland* scenario (0 % woodland cover), where current woodland in the Naddle catchment was replaced with grassland. This allowed us to calculate the 10-percentage point increase in woodland cover for each NFM woodland type.

Table 5.3 Land cover scenarios modelled.

Scenario	Repeats	Woodland cover (%)	Description
No woodland	1	0	Naddle catchment if completely grassland
Catchment woodland	5	10	Randomised increased catchment-wide woodland cover
	5	20	
	5	30	
	5	40	
	5	50	
	5	60	
	5	70	
	5	80	
Riparian	1	10	Increased woodland cover along Naddle Beck
	1	20	
	1	30	
Cross-slope	1	10	Cross slope plots of woodland cover
	1	20	
	1	30	
Naddle	1	12.9	Current woodland cover in Naddle
Naddle + Catchment	5	20	Current woodland cover in Naddle plus randomised increased catchment-wide woodland cover
	5	25	
	5	35	
	5	45	
	5	55	
	5	65	
	5	75	
	5	85	
Naddle + Upland catchment	5	25	Current woodland cover in Naddle plus randomised increased woodland above 400 m
	5	35	
	5	45	
Naddle + RSPB	1	23	Current woodland cover in Naddle plus randomised planting based on soil properties

We explored three different sets of catchment woodland scenarios. In each set of catchment scenarios, areas of woodland cover were added randomly as 25 m × 25 m squares across the catchment, to replicate natural regeneration or random planting. For each woodland cover, we repeated 5 replicates with different random allocations of woodland cover and we report the mean and standard error. In the first set of catchment woodland scenarios we increased woodland cover across the catchment from 0 % to 80 % at 10 % intervals (Table 5.3, *Catchment*).

In the second set of catchment woodland scenarios we increased woodland cover from the current Naddle woodland (Table 5.3, *Naddle + Catchment*), which consists of 12.9 % woodland and 87.1 % grassland, see Figure 5.1c. Woodland cover in the *Naddle +*

Catchment scenarios increased from 25 % to 85 % at 10 % intervals. In the third set of scenarios we increased woodland cover above 400 m from 25 % to 45 % at 10 % intervals, including the current Naddle woodland (Table 5.3, *Naddle + Upland Catchment*).

To simulate riparian woodland, we created a set of scenarios where woodland planting was expanded out from the riparian zone along the Naddle beck, increasing woodland cover from 10 % to 30 % at 10 % intervals (Table 5.3, *Riparian*). To represent cross-slope woodland, we created scenarios where woodland was established across the slope from10 % to 30 % at 10 % intervals (Table 5.3, *Cross-slope*). Finally, we created a scenario based on the current plans for tree planting and expected natural regeneration in the Naddle catchment by the RSPB, increasing woodland from the current Naddle woodland to 23 % (Table 5.3, *RSPB)*.

5.4.5. Modelled scenario analysis

For each scenario the peak total discharge ($m^3 \cdot s^{-1}$) recorded during the storm was calculated and the relative change in peak total discharge (%) compared to no woodland scenario is reported. For each set of scenarios we calculate the gradient of the relative change in peak total discharge compared to the change in woodland cover using the least square method and report the reduction in peak flow for a 10-percentage point increase in woodland cover. The peak base flow ($m^3 \cdot s^{-1}$) and peak overland flow ($m^3 \cdot s^{-1}$) for each scenario are also recorded.

5.5. Results

5.5.1. Changes to peak total discharge

We found that the peak discharge during the modelled 1 in 50-year storm event was greatest for the un-forested catchment (Table 5.4, Figure 5.3). Woodlands reduced peak discharge by 2.6-15.3 % depending on the extent and spatial placement of woodland cover.

For all NFM interventions, increased woodland cover typically led to larger reductions in peak discharge. For *catchment* woodland scenarios, larger reductions in peak discharge occurred until woodland cover reached 60 %. When woodland cover increased beyond 60 % peak discharge started to increase (Figure 5.3). For woodland cover up to 60 %, peak discharge was reduced by 2.1 % for each 10-percentage point increase in catchment woodland (Figure 5.4).

The *riparian* and *cross slope* woodland scenarios resulted in larger decreases in peak discharge compared with catchment scenarios of the same woodland cover e.g. *riparian* scenario with 30 % woodland resulted in reduction in peak flow of 9.78 %, whilst the c*atchment* scenario with the same woodland cover reduced peak flow by 8.55 %. For woodland cover up to 30%, peak discharge is reduced by 3.45 % for each 10-percentage point increase in riparian woodland and by 3.93 % for each 10-percentage point increase in cross slope woodland (Figure 5.4).

The current woodland in the Naddle catchment (12.9 % woodland cover) results in a reduction of peak flow of 7.7 %. The potential *RSPB* management strategy scenario, increasing woodland cover to 23 %, reduced peak discharge by 9.55 %.

Table 5.4 Simulated peak discharge ($m^3 \cdot s^{-1}$) for woodland scenarios. All catchment woodland scenarios are repeated 5 times and mean (μ), standard deviation (σ) and standard error (SEM) are reported. Results for all model runs including repeats can be found in Appendix C.4.

Scenario	Woodland cover (%)	Peak discharge (m^3s^{-1})			Change in Peak discharge compared to no woodland scenario (%)		
		μ	σ	SEM	μ	σ	SEM
No woodland	0	17.17			0		
Catchment woodland	10	16.504	0.214	0.107	-3.88	1.248	0.624
	20	15.976	0.084	0.042	-6.95	0.488	0.244
	30	15.702	0.156	0.078	-8.55	0.909	0.455
	40	15.16	0.071	0.036	-11.71	0.415	0.208
	50	14.98	0.079	0.039	-12.75	0.459	0.229
	60	14.714	0.050	0.025	-14.30	0.291	0.146
	70	14.758	0.043	0.022	-14.05	0.251	0.125
	80	15.234	0.030	0.015	-11.28	0.175	0.088
Riparian	10	16.73			-2.56		
	20	15.76			-8.21		
	30	15.49			-9.78		
Cross-slope	10	15.98			-6.92		
	20	15.68			-8.68		
	30	15.41			-10.24		
Naddle	12.9	15.85			-7.70		
Naddle + catchment woodland	20	15.97	0.207	0.104	-7.01	1.207	0.603
	25	15.77	0.180	0.090	-8.17	1.049	0.524
	35	15.46	0.236	0.118	-9.96	1.375	0.688
	45	15.00	0.086	0.042	-12.64	0.500	0.250
	55	14.64	0.083	0.041	-14.74	0.483	0.242
	65	14.55	0.059	0.030	-15.27	0.346	0.173
	75	14.89	0.035	0.018	-13.27	0.206	0.103
	85	15.45	0.045	0.022	-10.04	0.262	0.131
Naddle + upland catchment	25	15.97	0.228	0.114	-6.98	1.326	0.663
	35	15.66	0.186	0.093	-8.82	1.083	0.542
	45	15.26	0.115	0.058	-11.11	0.672	0.336
Naddle + RSPB	23	15.53			-9.55		

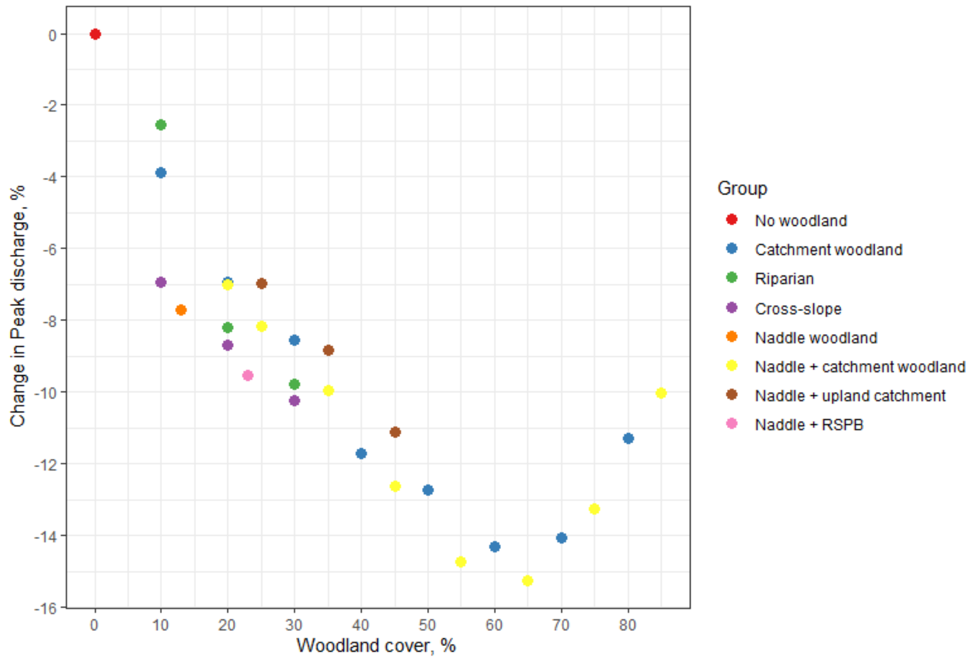

Figure 5.3 Change in peak discharge (%) for modelled scenarios.

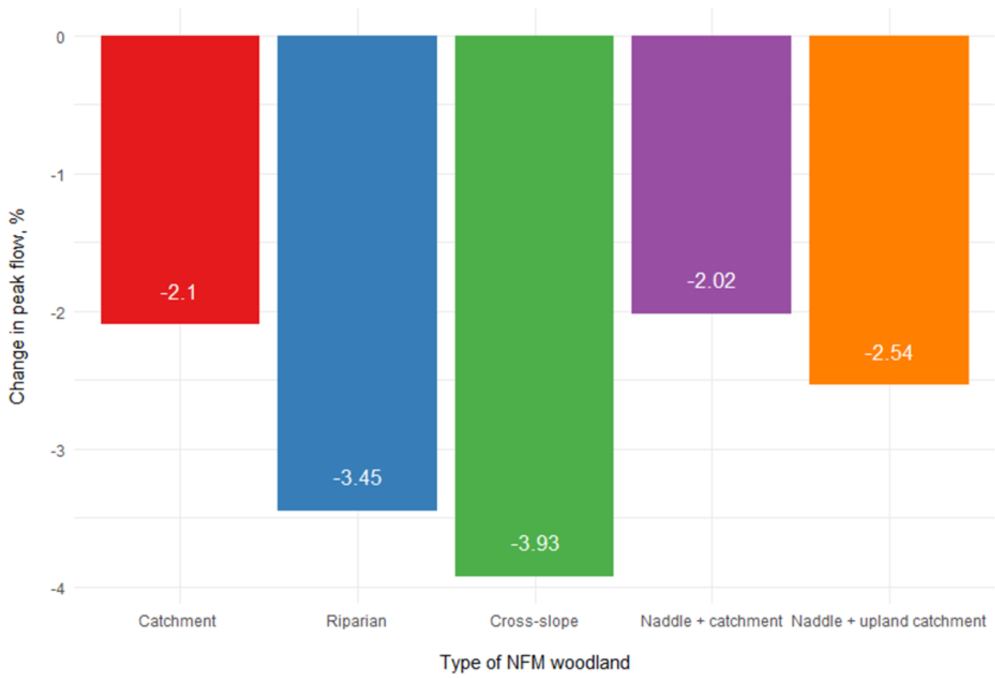

Figure 5.4 Change in peak flow for a 10-percentage point increase in woodland cover, grouped by type of NFM woodland.

5.5.2. Base flow and overland flow

As woodland cover increased, peak overland flow decreased whilst peak base flow increased (Figure 5.5). Overland flow is the primary runoff method for scenarios with woodland cover up to approximately 35 %, after which base flow becomes dominant.

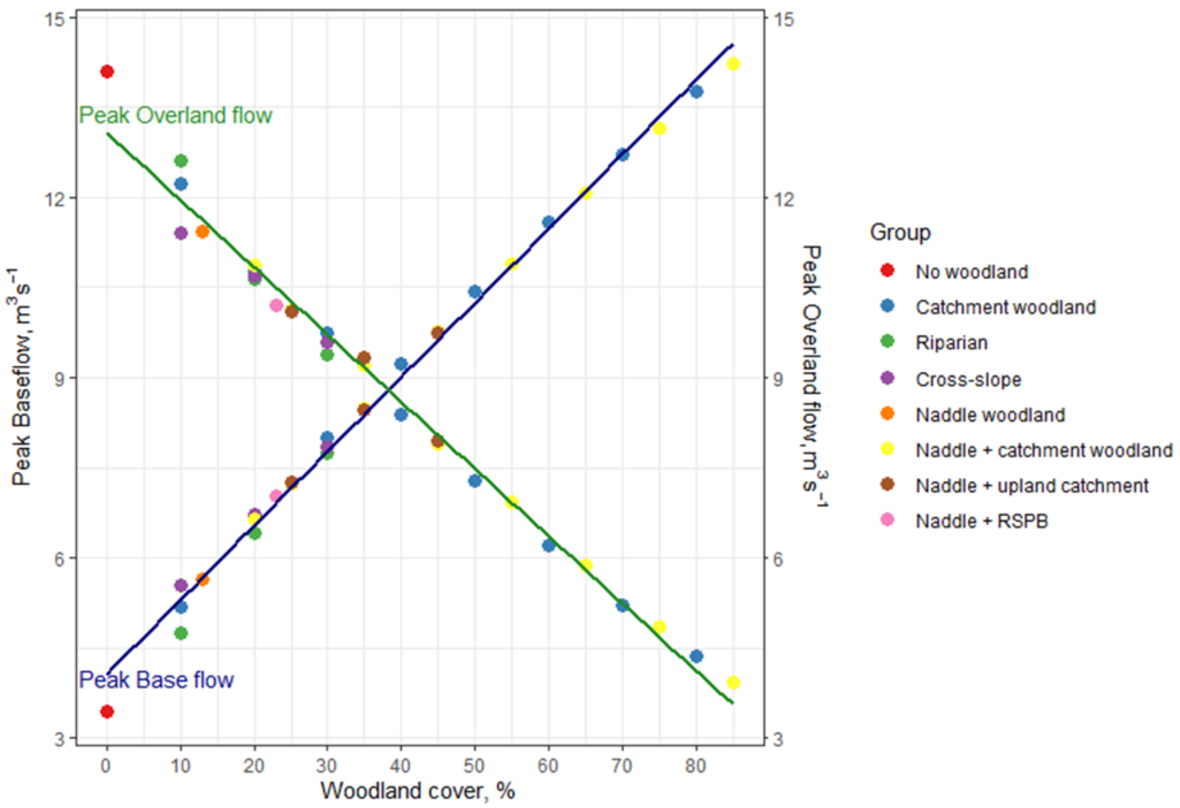

Figure 5.5 demonstrates peak overland flow is higher for a scenario with 25 % woodland cover (Naddle + catchment) compared to 75 % woodland cover. Peak base flow is higher in the 75 % woodland scenario.

Figure 5.6 demonstrates peak overland flow is higher for a scenario with 25 % woodland cover (*Naddle + catchment*) compared to 75 % woodland cover. Peak base flow is higher in the 75 % woodland scenario.

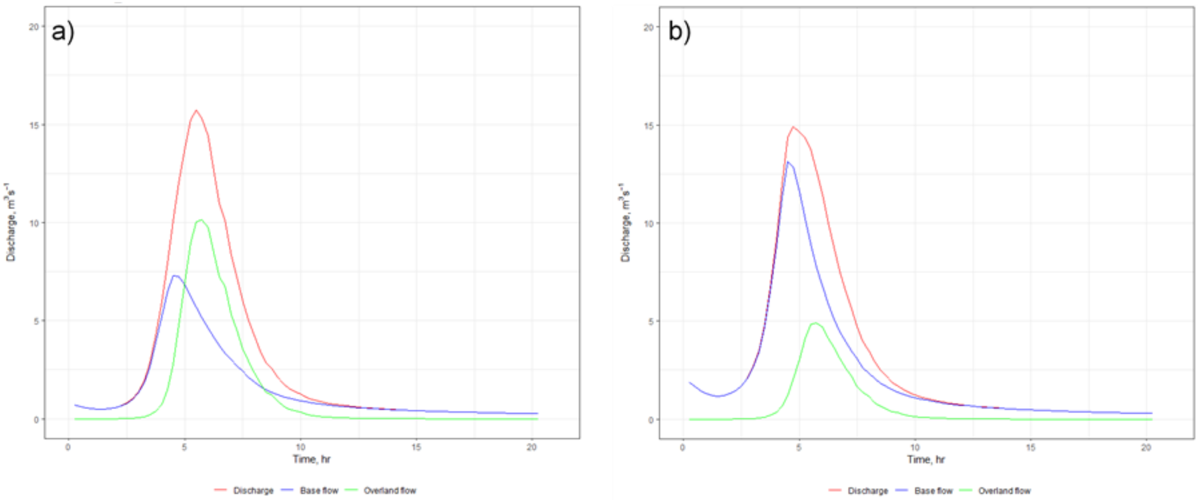

Figure 5.6 Total flow, base flow and overland flow ($m^3 \cdot s^{-1}$) for scenarios a) Naddle + catchment, 25% woodland and b) Naddle + catchment, 75 % woodland.

5.6. Discussion

This study used SD-TOPMODEL to investigate the impact of a range of woodland scenarios on peak discharge for a 1 in 50-year storm event. The modelled scenarios showed that increased woodland cover decreased peak total discharge when compared to the no woodland scenario. Increases in woodland was represented by increasing the soil permeability and decreasing overland flow velocity.

5.6.1. Impact of woodland on peak total discharge

We found that catchment woodland reduced peak flow by 4-15 % depending on woodland cover. Woodlands reduce flood peaks (Dadson et al., 2017; Kirby et al., 1991; Stratford, 2017) due to the increased permeability and water storage of woodland soils compared to other land covers (Archer, N.A.L. et al., 2013; Calder et al., 2008; Carroll et al., 2004; McCulloch and Robinson, 1993; Murphy et al., 2020).

At 80 % *catchment* woodland cover, peak discharge was reduced by 11.3 %. Wheater et al. (2008) modelled catchment wide afforestation in the Pontbren catchment (5.77 km^2) in Wales and found a reduction in peak flow during an extreme event of 10 % to 54 % (mean 36 %). Fraser et al. (2013) simulated conifer afforestation of mineral soils in the Hodder catchment (25.3 km^2) and simulated a 5 % to 7 % reduction in mean catchment peak flow.

We find that not only amount of woodland is important in reducing peak total discharge, but that location can play a vital role (Figure 5.4). Our *catchmen*t woodland simulations reduced peak flow by 2.1 % for a 10-percentage point increase in woodland cover. Similary, the *Naddle + catchment* woodland sceanrios and *Naddle + upland catchment* scenarios reduced peak flow by 2.0 % and 2.5 % respectively (Figure 5.4). In comparison, a 10-percentage point increase in *cross-slope* woodlands reduced peak flow by 3.9 % (Figure 5.4). This supports previous findings by Wheater et al. (2008), who found that cross slope woodland in the Pontbren catchment (5.77 km^2) could reduce peak flow during an extreme event by 5% (range 2-11 %).

Riparian woodland scenarios reduced peak total discharge by 3.45 % for a 10-percentage point increase in woodland cover. However, the 10 % *riparian* woodland cover scenario resulted in the smallest reduction in peak discharge. Whilst 20 % and 30 % *riparian* scenarios followed a similar pattern to *cross-slope scenarios,* exceeding catchment woodland reductions in peak flow. This suggests that there maybe an optimum distance from the river channel at which increased permeability and roughness is most influential. Our simulations do not include the impacts of in-channel wood which are likely to further increase the flood mitigation benefits of riparian woodland (Cooper et al., 2020)

5.6.2. Impact of woodland on base flow and overland flow

We found a catchment woodland cover of 65 % resulted the greatest reductions in peak flow. When woodland cover increased beyond 65 % reductions in peak total flow compared to our base scenario decreased. This response may be explained by the change in dominant flow, overland flow to base flow (Figure 5.5).

When the Naddle catchment is predominately grassland, the soils are less permeable and overland flow dominant. Increased woodland increases roughness, reducing overland flow and it takes longer for overland flow to reach the Naddle Beck (Figure 5.6). However, increasing woodland cover also increases the permeability of the soils. At 40 % woodland cover, base flow becomes dominant. Further increases in woodland cover result in reductions in peak total discharge until 65 % woodland cover, due to the greater roughness of woodland ‘slowing the flow’ of overland flow. Furthermore, a portion of the base flow will be actively stored in the more permeable woodland soils. For woodland cover greater than 65 %, peak total peak discharge is no longer reduced by the same extent. This may be explained by increasingly permeable soils which when soils storage is full results in further base flow converting to total flow. In addition, if less overland flow is being produced, the benefits of increase roughness cannot be utilised. We note that in this study we assume woodland and grassland have the same storage (Table 5.1). If woodland soils have greater storage than grassland soils, as suggested in some studies (Geris et al., 2015; Stratford, 2017), it is likely that woodlands would result in even larger reductions in peak flow than simulated here. However, it is worth considering whether this shift in dominant flow may be an artefact from the model set up. Future work, repeating this work for additional catchments, would identify whether the same patterns are replicated.

5.6.3. Future work

All our model scenarios are based on a 1 in 50-year storm event for a relatively small (2.6 km^2) upland catchment. Storm size, catchment size and antecedent conditions can all impact the extent to which woodlands impact flood response (Dadson et al., 2017; Bathurst et al., 2020). For larger UK catchments (500 – 10,000 km^2), Buechel et al. (2022) used the JULES model at 1 km^2 resolution to show that afforestation reduced median and low streamflow, but no consistent reduction in peak flow during large flood events. Future work is needed to use a consistent modelling framework to investigate how the impact of woodland varies for different storm and antecedent conditions and different catchment sizes.

We studied the impact of ungrazed and well-established, semi-natural broadleaf woodland compared to pasture grazed at relatively low grazing intensity based on empirical data collected in the Naddle catchment. Future work is needed to assess how the hydrological response varies with woodland age and woodland type (conifer, broadleaf, mixed) (Peskett et al., 2021) and with varying grazing intensity in pasture and woodland areas.

5.7. Conclusion

We used a rainfall-runoff model to investigate how broadleaf woodland cover altered peak flow in a 1 in 50-year storm event in a small upland catchment. Woodland cover reduced peak flow by as much as 16 %, with maximum reductions in peak flow for woodland cover of 65 %. For woodland cover above 65 %, lower reductions in peak flow were simulated due to increases in base flow. Reductions in peak flow depend on the placement of woodlands within the catchment. Larger reductions in peak flow were simulated for cross-slope and riparian woodland compared to catchment woodland with reductions in peak flow of 3.9 %, 3.5 % and 2.1 % respectively for a 10-percentage increase in woodland cover. Our results confirm that increased woodland cover can be an effective method of reducing peak flow even in large storm events.

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Chapter 6. Conclusion

6.1. Section outline

This chapter's purpose is to summarise the PhD project as a whole and discuss how the work in the preceding 4 chapters have contributed to answering the four overarching research questions set out in section 1.7:

A. What is the impact of semi-natural woodland on soil hydrological properties?
B. How do upland woodland habitats affect overland flow velocity?
C. Does woodland cover within a catchment impact streamflow?
D. Can trends and patterns be identified between land cover and catchment storage?

The findings in this project are discussed in response to these research questions and the wider implications of this work identified. In addition project limitations will be reflected upon and scope for future work considered.

6.2. Summary of key findings

Chapter 2 presents the findings from the establishment of a correlation catchment study consisting of 9 small (< 0.2 km^2) upland catchments. We compared soil hydrological properties and streamflow response for mature semi-natural broadleaf woodland where no stock grazing occurs to pasture with varied grazing intensity. We found that wooded catchments had a more muted streamflow response to the 28 storm events analysed, including a 1 in 10-year event, Storm Ciara. Woodland soils were 11-20 times more permeable than pasture soil.

Hillslope surface flow velocity measurements over different land covers are documented in Chapter 3. Wood pasture dominated by bracken had significantly lower overland flow velocity compared to wood pasture dominated by grassland and mature semi-natural broadleaf woodland. We found that established woodland soils exhibited 8- 80 times higher saturated hydraulic conductivity than wood pasture soils. Furthermore, we demonstrate that Manning's *n* is far from constant in these shallow overland flows.

In Chapter 4 catchment storage estimates for 418 catchments across the UK are presented. We find significant differences in the active storage, most strongly between grassland and woodland cover (grassland, 12.6 mm and woodland, 8.0 mm), arable (4.4 mm) and urban (1.6 mm) dominated catchments. Land cover significantly influences catchment storage alongside other environmental variables such as, rainfall and catchment size.

In Chapter 5 the impact of different woodland scenarios in a small (2.62 km^2) catchment on peak flow is modelled. In all scenarios, woodland cover reduced simulated peak flow for a 1 in 50-year storm event between 4-16 %. However as woodland cover reached 60 % to 65 %, the reduction in peak flow begins to decrease. This can be explained by the change in primary runoff method. Overland flow is the primary runoff method for scenarios with woodland cover up to approximately 35 %, after which base flow becomes dominant.

Figure 6.1 summarises key findings from this thesis.

6.3. What is the impact of semi-natural woodland on soil hydrological properties?

The results from this project provide additional empirical evidence supporting the consensus that established woodland soils are more permeable than other soils. The upland established semi-natural woodland topsoil (0-5 cm) investigated in Chapter 2 and 3 had significantly ($p < 0.05$) higher Ksat, 8-80 times higher, compared to wood pasture and pasture habitats. This is often attributed to the root networks of trees and shrubs, alongside the relatively undisturbed soil surface (Alaoui. et al., 2011). However, whilst previous studies also found higher subsoil permeability in woodland soils (Agnese et al., 2011; Mawdsley et al., 2017), Chapter 2 showed no significant ($p > 0.05$) difference in subsoil (> 15 cm) permeability between established semi-natural woodland soil and grazed pasture soils. This may be explained by the relatively shallow depth of the soils investigated in this project.

The soils investigated in this project are organo-mineral soils, predominately Malvern 611a (Chromic Endoleptic Umbrisol) and Bangor 311e (Dystric Epileptic Histosol) soils (Cranfield University, 2019). Organo-mineral soils are defined by the depth of surface organic material, generally < 40 cm deep, and consist of > 20 % organic content of that surface material (Holden et al., 2007; Bond et al., 2021; Bol et al., 2011). Chapter 3 presented results showing that the organic matter present in semi-natural woodland topsoil (0-5 cm) (24.0 %) was significantly ($p < 0.05$) higher than the bracken wood pasture (18.4 %) and grass wood pasture (16.6 %) soils. This supports statements by Bol et al. (2011) whereby relatively undisturbed soil systems such as woodlands tend to accumulate more organic matter in their soils. Greater amounts of soil organic matter are often associated with increased micropores and macropores, which increases the water holding capacity of the soil

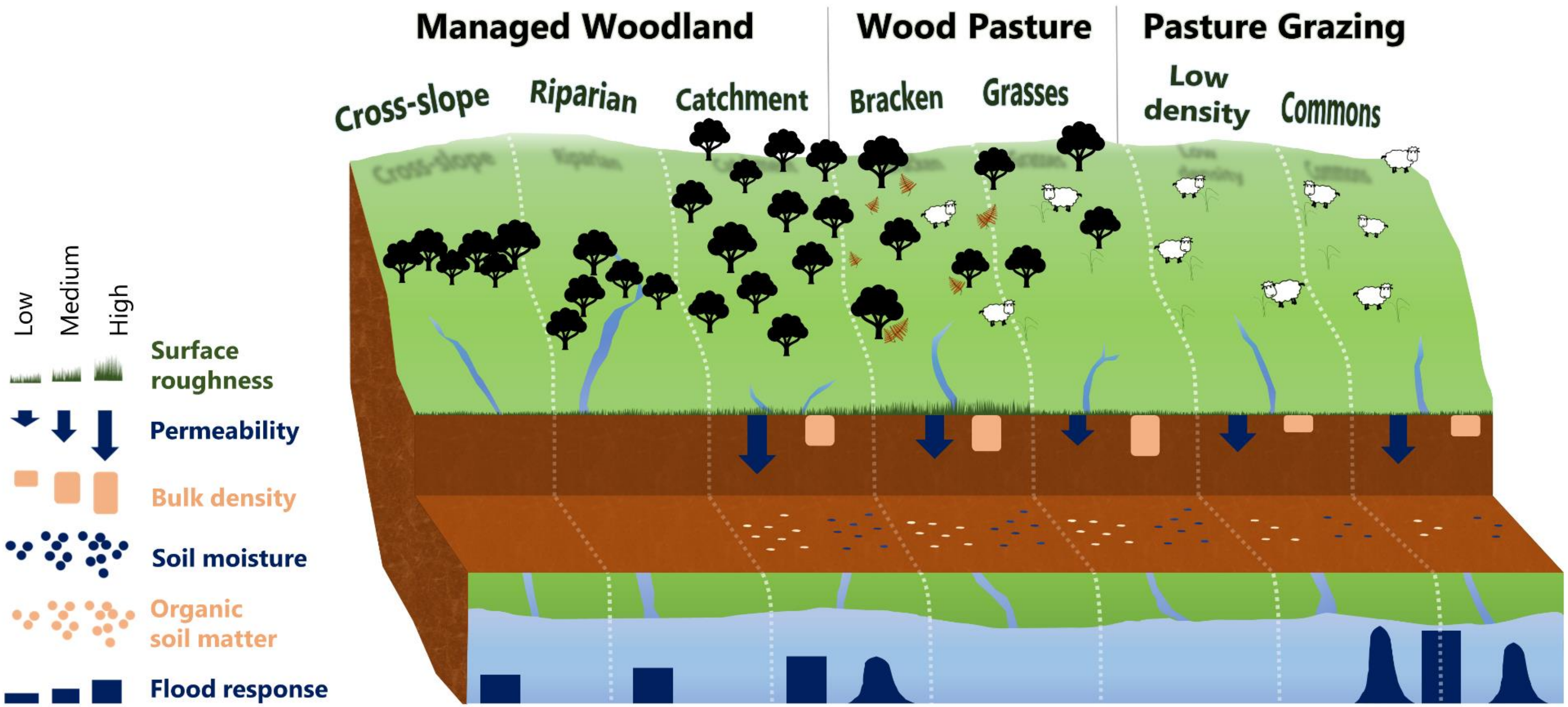

Figure 6.1 Summary of key findings by habitat: Managed Woodland including; Catchment, Riparian and Cross-slope, Wood pasture dominated by; Bracken or Grassland and Pasture Grazing consisting of Low-density Grazing (maximum grazing intensity of 0.10 $LU \cdot ha^{-1}$ May to October) and Commons Grazing (maximum intensity of 0.12 $LU \cdot ha^{-1}$ all year round). Relative values across habitats are indicated by; increased ground vegetation for higher reductions in overland flow velocity (Chapter 3), larger blue arrow for increased permeability (top < 5cm of soil, no significant difference in subsoil permeability between habitats) (Chapter 2 and 3), larger orange cylinder for higher bulk density (Chapter 2 and 3), increased circles density relates to higher soil moisture (blue dots) and organic matter (orange dots) (Chapter 2 and 3). Reduced peak flow is depicted by smaller hydrographs (field data) and smaller bar plots (modelled data). No field soil data (permeability, bulk density, soil moisture or organic matter) was collected for cross slope and riparian woodland.

(Bol et al., 2011; Holden et al., 2007). Certain types of soil organic matter can hold up to 20 times their weight in water (Reicosky, 2005). Compaction, often associated with grazing can contribute to a loss of soil organic matter as soil becomes degraded and loses its structural integrity (Bol et al., 2011).

It is generally accepted that soil moisture varies with vegetation type, especially with regards to the presence or absence of woodlands (Breshears and Barnes, 1999). This supports the findings in this project, as higher average soil moisture was recorded in established woodland soils, reported in both Chapter 2 and 3. In Chapter 2, results show that the average soil moisture for low-density grazing soils was significantly ($p < 0.05$) higher than the commons grazing soils. This confirms the consensus that higher levels of soil moisture are often attributed to lower levels of grazing (Xu et al., 2014). Antecedent conditions have a considerable impact on the collection of soil moisture readings. Average soil moisture of established woodland soils, whilst consistently higher, in Chapter 3 was 37.5 % higher than in Chapter 2. Soil moisture readings in Chapter 3 were collected over one 24 hour period, whilst soil moisture readings in Chapter 2 were collected at the nine sites on a monthly basis across the duration of a year.

This project found consistent average bulk density results for established woodland soils, (0.50 $g \cdot cm^{-3}$, Chapter 2 and 0.51 $g \cdot cm^{-3}$, Chapter 3). In Chapter 2 when compared with grazed catchments (commons grazing, 0.46 $g \cdot cm^{-3}$ and low-density grazing, 0.36 $g \cdot cm^{-3}$), the bulk density of woodland soils was 9-39 % higher. Whereas the bulk density of established woodland soil was significantly ($p < 0.05$) lower than both bracken wood pasture and grass wood pasture soils (respectively 21 % and 18 % lower) in Chapter 3. Increased bulk density is indicative of compaction and usually limited to the top 10 cm of soil (Greenwood and McKenzie, 2001). Compaction may result from stock grazing (trampling), however is dependent on a number of variables such as; stock density, animal weight, hoof size, soil type, plant type and field slope (Alaoui et al., 2018). Whilst all grazing intensities investigated in this project are relatively low-intensity, different grazing habits may have impacted the variability of bulk densities documented in this project. Wood pasture soils investigated in Chapter 3, with increased bulk density located near to the farm yard and regularly utilized whilst pasture soils in Chapter 2 are located further along the valley. Furthermore, changes to grazing management was introduced to the sites investigated in this project within the last 7 years and little is known about the effects of reducing stock grazing pressures. Marrs et al. (2018) and Marrs et al. (2020) suggest it may take 48–62 years to see the benefits of reduced grazing due to the long-term soil degradation. Whereas, Holden et al. (2007) found that the removal of grazing after just 5 years is enough to allow the system to recover towards that of a system that has had no grazing for over 40 years.

6.4. How do upland woodland habitats affect overland flow velocity?

Land cover can impact overland flow velocity by altering the roughness of the surface (Bond et al., 2020; Holden et al., 2008). Surface roughness plays a dominant role in delaying delivery of water to streams, extending the tail of the hydrograph and reducing flood risk. The role of roughness has been well studied regarding channel and bank flow (Medeiros et al., 2012), yet field investigations into the impact of vegetation on hillslope roughness have been limited (Pan et al., 2016).

This project presents the first in-field overland flow velocity measurements for woodland habitats. Investigations using a novel flume experiment in Chapter 3 found that wood pasture dominated by bracken had the slowest overland flow velocities in response to the three flow rates investigated. Overland flow velocity was 12-20 % lower for bracken wood pasture compared to established broadleaf woodland and 19-27 % lower than grass-dominated wood pasture. Wood pasture dominated by bracken sites contained large quantities of bracken leaf litter which may have created more resistance to the overland flow produced in the flume. Furthermore, grass wood pasture sites had short-cropped vegetation and the highest overland flow velocity. It is important to note that the overland flow velocity values found in Chapter 3 were restricted to the month of October and the impact seasonality was not investigated. It is probable that roughness would changes with season as found in Bond et al., (2020). The results in Chapter 3 support the idea that the vegetation structure in the first few centimetres is most important to reducing overland flow velocity (Pan et al., 2016). This is further supported when the overland flow velocity measured in our study's habitats is compared against those from upland peat (Holden et al., 2008) and grassland (Bond et al., 2020). The presence of moss, a typically coarse, dense structured plant is associated with the lowest overland flow velocities recorded.

Although established woodland was found to be less rough than wood pasture (Chapter 3), the woodland did have higher permeability (Chapter 2 & 3). Management approaches should consider whether roughness or permeability is the greater driver of runoff attenuation; for example, if overland flow is frequently produced, surface roughness may be the greater control. In a mosaiced landscape, different subsections of the catchment will have different propensities for overland flow generation therefore will require different management applications. This could come in the form of wood pasture with reduced stock grazing which is a dynamic system containing different successional stages between grassland and woodland.

6.5. Does woodland cover within a catchment impact streamflow?

Results presented in Chapter 2 and 5 include both empirical field evidence and results from rainfall-runoff model scenarios where the presence of woodlands reduces peak flow. This strengthens the argument for woodlands to be utilized as an NFM strategy.

In Chapter 2 streamflow response in 9 small (< 0.2 km^2) upland catchments wereas analysed for 28 storm events across a 13-month period. Our results found a more muted response in woodland catchments compared to pasture catchments. Semi-natural broadleaf woodlands reduced specific peak discharge by 23 %–60 % and peak runoff coefficients by 30%–60 % compared with pasture. Response to storm events took 14–50 % longer in woodland compared to pasture. The majority of research on the effectiveness of woodlands as an NFM strategy has found them significantly effective for small events only (Stratford, 2017; Burgess-Gamble et al., 2017). However, results in Chapter 2 found wooded catchments reduce flood response for both small (discharge peaks < 1 $mm \cdot hr^{-1}$) and large storms (discharge peaks > 1 $mm \cdot hr^{-1}$). Furthermore we found wooded catchments reduced volume runoff coefficient and peak runoff coefficient for storms with a return period greater than 1.5 years, which includes the met office named storm, Storm Ciara, a 1 in 10-year event.

In Chapter 5 we incorporated empirical results from on soil permeability (Chapter 2) and overland flow (Chapter 3) into the SD-TOPMODEL rainfall-runoff model to simulate the impacts of different woodland scenarios in a bigger (1 in 50-year) storm event and a bigger (2.62 km^2) catchment. We found that woodland cover reduced peak flow for this 1 in 50-year storm event by 4-16 % depending on the extent and spatial distribution of woodland cover. As catchment woodland cover increased from 0 % to 60/65 %, peak flow decreased. When catchment woodland cover exceeded 70 % the reduction in peak flow began to decrease (peak flow was still less than peak flow for 0 % woodland). Base flow becomes the dominant pathway, and surface roughness no longer can contribute to slowing the flow.

Dependent on their placement, smaller pockets of cross slope and riparian woodland resulted in larger reductions in peak flow, 3.9 % and 3.5 % for a 10-percentage point increase in woodland cover respectively, compared to a 2.1 % reduction for catchment-wide woodland. Similarly Gao et al. (2016) using SD-TOPMODEL found that land-cover change along narrow riparian buffer strips had the highest impact on river flow. This highlights the importance of NFM placement within a catchment and denotes woodland establishment can be targeted to the most effective parts of a catchment for greater flood risk reduction. Overall, the results demonstrate that riparian and cross slope woodland are particularly effective NFM measures.

Furthermore it is not often practical, cost-effective or desirable to convert a catchment fully into woodland. Catchments usually consist of a mosaic of land covers and management strategies, where targeted patches of land could be set aside for woodland creation.

6.6. Can trends and patterns be identified between land cover and catchment storage?

Catchment storage can be difficult to quantify directly, therefore in Chapter 4 recession analysis was used to approximate a parameterised catchment storage estimate. Parametrised active storage estimates were calculated for 418 UK catchments. Isolating catchments with a dominant (≥ 70 %) land cover revealed significant differences in median storage between grassland (12.6 mm) and woodland, (8.0 mm), and arable (4.4 mm) and urban (1.6mm) dominated catchments.

GLMs were used to further identify which environmental catchment characteristics significantly influenced catchment storage. The most important environmental variables for determining active catchment storage included catchment size, rainfall, arable land cover, grassland land cover and low superficial deposits. Woodland, both broadleaf and conifer woodland did not have a significant impact on active storage. This perhaps contradictory finding to current understanding that wooded catchments typically have higher water use e.g. Law (1956) may be due to the relatively low woodland cover in UK catchments (median woodland cover for natural catchments was only 11.44 %) and very few catchments were dominated by woodlands.

Modest increases in storage were found for grassland and woodland dominated catchments when comparing values over the available period of record (1985-2005 .v. 2005-2017).

Whilst there was slight evidence of reductions in storage for urban catchments. This perhaps reflects the widespread changes in land use experienced in the UK, for example through reduced grazing pressures. Additionally, an increase in active storage over time could also be indicative of tree growth for woodland dominated catchments. In terms of NFM, it's regularly highlighted that the benefits of woodland creation are not immediate and grow with time. Although improvements to infiltration have been shown to develop in a relatively short time-frame, 18 months after tree planting (Mawdsley et al., 2017). As trees continue to grow, root systems will continue to develop and further increases in infiltration will occur as the trees mature. Revell et al. (2021) found that reductions to peak flow could be modelled 15 years after tree planting, with expectations these reductions will continue to grow. However grassland coverage showed a greater reduction in peak flow than these young trees when compared to impermeable land cover. Grasslands make up 60 % of the UK land cover (DEFRA, 2016) and their management has been an integral part of UK history. Woodland creation for the purposes of NFM woodland creation will more often than not take place on these grasslands. Therefore the management of these areas whilst planted trees become established is crucial and important in the short-term for flood mitigation.

Scale is hugely influential in the understanding of results. Land cover impacts on peak flow are apparent at small spatial scales (Rogger et al., 2017) but this impact tends to become smaller as scale increases (Dadson et al., 2017; Peskett et al., 2021). This was true for field and modelling outcomes in this thesis. At the very small catchment scale (< 0.2 km^2, Chapter 2) woodlands reduced peak flow up to 60 % compared to pasture. Additionally, in catchments up to 2.62 km^2 (Chapter 5) modelled woodland creation was found to reduce peak flow up to 16 % compared to grassland. In comparison, in large catchments up to 10,000 km^2 (average 72 km^2) woodland was found to not be a significant impact on catchment storage (Chapter 4). This reflects previous studies that evidence supporting woodland as a flood mitigation method is greatest at plot and hillslope scale and for small catchments (< 10 km^2) (Burgess-Gamble et al., 2017; Rogger et al., 2017). There are multiple possible reasons for this. In larger catchments it's often difficult to separate background effects from land use cover. As physical catchment characteristics such as; geology or slope become more prominent, masking the impact of invariant controls i.e. land cover (Beven, 2000; Soulsby et al., 2006). In addition catchments often support a mixture of land cover (unlike the very small catchment scale 100 % woodland in Chapter 2) for which each cover type has its own controls on runoff and storage. The mosaic nature of catchments introduce the spatial heterogeneity which, in addition to physical catchment characteristics, dilute the impact of managed environmental characteristics. This makes the signal between differing land covers and storage noisy and less distinguishable at a catchment scale compared with hillslope or plot scale (Spence, 2007; Soulsby et al., 2008; Teuling et al., 2010; Tetzlaff et al., 2011). Often, hydrological models are suggested as a way to scale-up investigations for larger catchments, however this can be limited by the available processing power.

6.6.1. Wider implications

The work in this project focussed on investigating what impact semi-natural woodlands have on flooding in the UK. Empirical field evidence was collected from a catchment in England's uplands and used to inform parameters in a rainfall-runoff model. This project also explored additional environmental catchment characteristics alongside woodland cover. Some of these findings will have wider implications.

6.6.1.1. Considerations when creating NFM woodlands

Following increased interest in woodland creation as an NFM strategy, advisory documents (SEPA, 2016; Catchment Based Approach Partnerships, 2017) have been published to ensure future woodlands are strategically placed. Current NFM guidance suggests key woodland creation locations include across hillslopes, alongside watercourses and the development of shelterbelts. Results in Chapter 5 demonstrate the potential for such woodland locations; peak discharge was reduced by 3.45 % in riparian woodland and by 3.93 % in cross slope woodland for each 10-percentage point increase in woodland cover compared with 2.1 % for catchment woodland. Nonetheless, Chapter 2 found that catchment-wide woodland related to a more muted peak flow response compared with pasture catchments. However, it is important to note that for the purpose of developing an NFM strategy, the conversion of an entire catchment to woodland is unlikely. Here, a mosaiced landscape approach, host to range of land covers, offers an NFM solution. The development of areas with increased soil permeability (woodland soils, Chapter 2 & 3) and others with reduced overland flow velocity (wood pasture dominated by bracken, Chapter 3) could work together to provide flood mitigation alongside other ecosystem services. These may include improvements to soil health, water quality and biodiversity, which distinguish NFM strategies from traditional flood management. Furthermore, changes to UK legislation following Brexit, such as the proposed Environmental Land Management scheme (ELMS), will incentivise a more holistic land management approach for which land managers will be paid for improving and managing ecosystems services (Klaar et al., 2020).

6.6.1.2. Representation of surface roughness in catchment models

The Manning's *n* equation has been extensively used to calculate flow in open channels. However, the simplicity of the Manning's *n* equation has led its use to be extended to conditions other than those for which it was intended (Chiew and Tan, 1992). This includes its use to model the impact of land cover on flows, whereby the hydraulic roughness of different land covers are represented by Manning's *n* coefficients. Engman (1989) stated that the most severe "misuse" of the Manning equation was the calculation of shallow overland flow velocities. Results in Chapter 3 continue to support this, finding that Manning's *n* is far from constant in shallow overland flows. Furthermore, the Manning's *n* values calculated in Chapter 3 for the three habitats investigated in this study can be an order of magnitude higher than previous values reported by Chow (1959) and others Arcement and Schneider (1989). Future work is needed to develop a more extensive overland flow dataset, which can be used to improve the representation of overland flow for different land covers in flood rainfall-runoff modelling.

6.7. Limitations of the project and future work suggestions

6.7.1.1. Data collection

A key limitation associated with this project relates to the time constraints of a PhD timeline and lack of preceding data. In Chapter 2, the timeframe of streamflow data investigated was limited to 13 months, which captured one, 1 in 10-year storm event. Having data from before the PhD start date, or an extended timeframe, may have resulted in the opportunity to draw conclusions on larger storms and build further conclusions regarding the impact of woodlands on bigger storm events. The implementation and preservation of streamflow monitoring in future NFM projects is encouraged to develop the NFM evidence base against lager storms.

An additional limitation related to time constraints in this project was that data recorded reflects a single 'snap shot' in time. Because of this, property changes over time, such as any 'recovery' after management change (for example RSPB grazing reductions made in 2012) or seasonality, are not investigated. Data collection in Chapter 3 was limited one season (autumn) due to the Covid-19 pandemic. The opportunity to repeat flume experiments during different seasons would offer important insights into the influence of seasonality and growth cycles on overland flow velocity and therefore surface roughness. The role of land cover to mitigate against flooding needs to be understood for all seasons.

This project investigated pasture environments with relatively low grazing intensities. Future work should aim to corroborate the findings in this project with more intensely grazed environments. In addition, investigations into the grazing intensity which could support the natural regeneration of wood pasture is also important.

Furthermore, this project studied areas of established semi-natural woodland and wood pasture, however what tree density is needed to develop the soil properties and/or the impact on streamflow found in Chapter 2? Future work is needed to explore what tree density is essential for flood mitigation; this is especially crucial as the popularity of tree planting for NFM projects increases.

6.7.1.2. UK catchment storage

This project is the first to take the approach set out in Chapter 4 to estimate catchment storage for numerous catchments across the UK. Most, if not all of the UK's catchment have been modified by anthropogenic drivers, so the work in Chapter 4 tried to address this limitation by removing catchments from the analysis which had been classified by the NRFA as not 'natural'. The use of large multi-site datasets gave the opportunity to give a broad-scale comparison between dominant land covers and the use of GLMs identified significant variables on catchment storage. Notably, the relative importance of woodland for NFM was observed at the catchment scale (up to < 10,000 km^2); to date few existing studies have evidence of this (Burgess-Gamble et al., 2017; Dadson et al., 2017; Lane, 2017; Stratford, 2017). Future work is needed to further explore other environmental variables in more detail, such as rainfall and geology. In addition, other methodologies could be used to calculate storage for the same catchments and this analysis replicated.

6.8. Conclusions

Overall, the results of this project show that semi-natural woodlands contribute to flood mitigation in the UK. It has been shown that increased woodland coverage of a catchment can result in a more muted response in stream flow. It is understood that increased permeability and soil water storage contributes to these reductions in peak flows. Other catchment characteristics also play a significant role.

This work contributes to the growing evidence base supporting woodland creation, including catchment woodland, cross-slope woodland and riparian woodland, as a method of NFM. The work in this thesis provides a better understanding on how woodlands, wood pasture and grazing pasture can create a mosaiced landscape to ensure increased permeability, reduce overland flow velocity and reduced flood peaks. Since this thesis has demonstrated the importance of woodland location within a catchment and the benefits wood pasture can offer, it is hoped this work will influence future policy and aid decision making for better resilience against future flooding.

6.9. References

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Appendix A: Supporting Information for Chapter 2

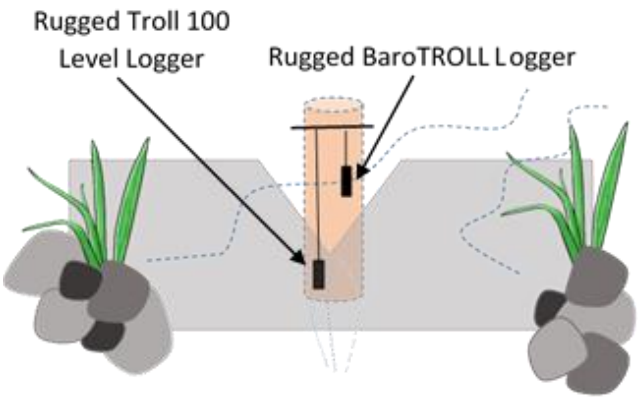
A.1: V-notch weir Construction

Details regarding the construction of V-notch weir:

The v-notch weir was installed at a point in the stream less than 1 m across. The weir was constructed by removing a small proportion of the stream banks before rebuilding them to offer support to a pre-cut 1.6 mm sheet of galvanised metal. A Rugged Troll Level Logger was placed within an adapted section of pipe casing (perforated with holes) which allowed for the free movement of water and positioned in the now-formed stilling pool.

The V-notch weir uses the basic principle that discharge is directly related to water depth above the bottom of the V and is referred to as the head (h) (International Organisation for Standardisation, 1980). The Rugged TROLL level logger records water level depth every 5 minutes which allows for the determination of head. A V-notch weir was chosen over other types of weir, such as the rectangular weir, as a large change in depth represents a small charge is discharge so produces far more accurate discharge measurements.

In addition, the use of a Rugged BaroTROLL logger, positioned out of the water allowed for correction of absolute level sensor data to eliminate barometric pressure effects from the measurements. The depth corrected values were used to calculate the effective head (h_1, m).).



A.2: Kindsvater-equation

Information on the use of the Kindsvater-equation used to calculate flow:

Flow (Q, $m^3 \cdot s^{-1}$) was calculated using the Kindsvater-Shen equation (*i*):

$$Q = C_e \, tan\left(\frac{\propto}{2}\right) \sqrt{g} \, (h_1 \, + k)^{2.5} \qquad (A.i)$$

Where:

C_e is discharge coefficient,
h_1 is the effective head (m),
k is the head correction factor (m),
g is acceleration due to gravity ($m \cdot s^{-2}$) and
α is angle of V-notch (°).

The v-notch weirs were constructed to meet the requirements of fully contracted 90-degree weirs, set out in Kulin and Compton (1975). Discharge coefficient, C_e and head correction factor, k values relating to fully contracted 90-degree v-notch weirs were used (Figure *i*).

Manual check of flow rate was measured at each v-notch weir. The time taken for a known volume to flow over the v-notch weir was recorded, and subsequently flow ($m^3 \cdot s^{-1}$) was calculated. This was compared against the streamflow calculated using the Kindsvater-Shen equation at the known time and date. Regular site visits were under taken to ensure data quality by identifying any equipment failures.

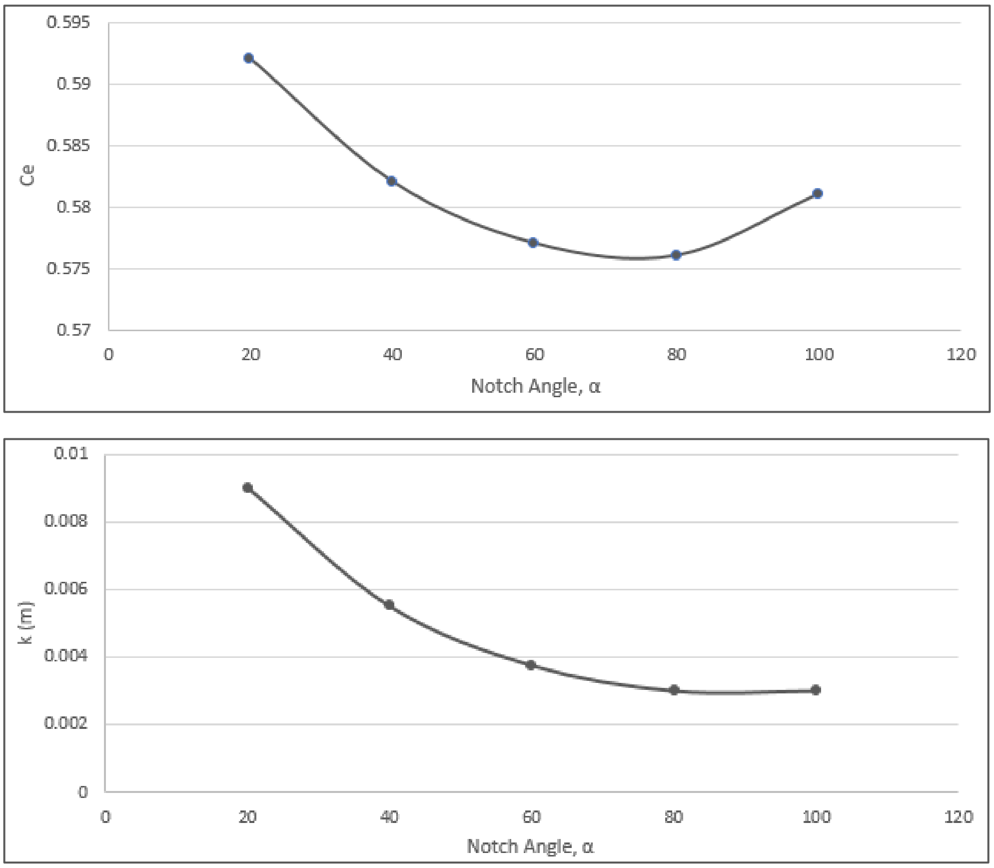


Figure A.i) a) Discharge coefficient, C_e, d b) Head corrector factor, k, taken from the Kindsvater-Shen relationship derived from Kulin and Compton (1975).

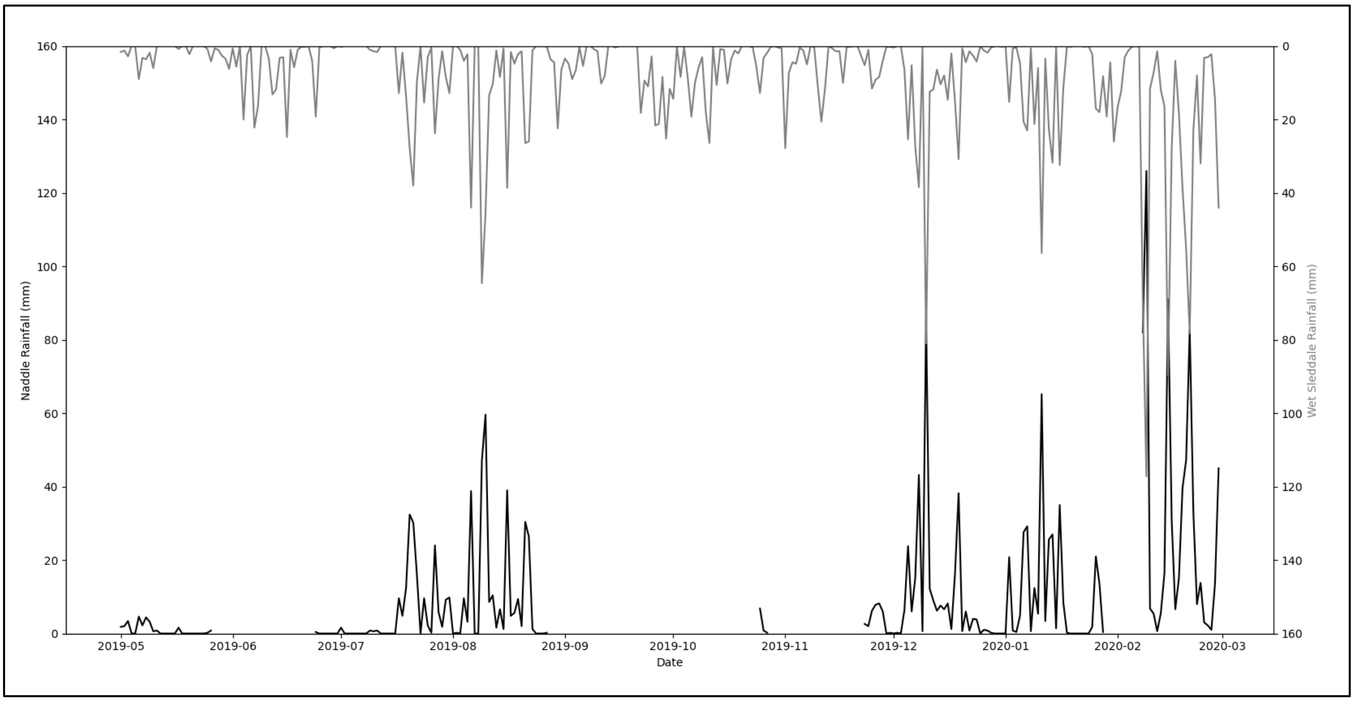
A.3: Summary of identified storm events.

Storm intensity is calculated as total rainfall divided by storm duration. The UK Met Office named two storm events: 26: Ciara, 27: Dennis.

Rainfall Event	Dates	Rainfall (mm)	Duration (hr)	Storm Intensity ($mm{\cdot}hr^{-1}$)	Return period (years)
1	25/01/2019 - 27/01/2019	22.20	26.50	0.84	<1.5
2	03/02/2019 - 04/02/2019	42.60	17.00	2.51	<1.5
3	07/02/2019 - 08/02/2019	81.20	15.75	5.16	1.5
4	09/02/2019 - 10/02/2019	41.40	21.25	1.95	<1.5
5	03/03/2019 - 04/03/2019	22.80	12.50	1.82	<1.5
6	06/03/2019 - 08/03/2019	25.40	41.00	0.62	<1.5
7	11/03/2019 - 13/03/2019	101.00	29.75	3.39	1.5
8	13/03/2019 - 14/03/2019	25.40	16.25	1.56	<1.5
9	16/03/2019 - 17/03/2019	75.20	15.75	4.77	1.5
10	19/07/2019 - 21/07/2019	44.80	25.00	1.79	<1.5
11	21/07/2019- 22/07/2019	46.20	17.50	2.64	<1.5
12	27/07/2019 - 28/07/2019	24.20	27.50	0.88	<1.5
13	06/08/2019 - 07/08/2019	38.80	14.00	2.77	<1.5
14	09/08/2019 - 11/08/2019	115.40	58.00	1.99	1.5
15	16/08/2019 - 17/08/2019	43.60	30.00	1.45	1.5
16	21/08/2019 - 23/08/2019	56.80	39.25	1.45	<1.5
17	05/12/2019 - 06/12/2019	29.80	27.00	1.10	<1.5

18	07/12/2019 - 09/12/2019	58.80	34.50	1.70	<1.5
19	10/12/2019 - 12/12/2019	98.90	31.75	3.11	<1.5
20	18/12/2019 - 20/12/2019	54.60	38.50	1.42	<1.5
21	06/01/2020 - 06/01/2020	27.60	9.50	2.91	<1.5
22	07/01/2020 - 09/01/2020	29.80	28.50	1.05	<1.5
23	10/01/2020 - 12/01/2020	72.00	35.00	2.06	<1.5
24	15/01/2020 - 18/01/2020	44.80	51.25	0.87	<1.5
25	25/01/2020 - 26/01/2019	21.80	13.75	1.59	<1.5
26	08/02/2020 - 10/02/2020	205.60	55.50	3.70	10
27	15/02/2020 - 16/02/2020	119.20	22.50	5.30	4
28	19/02/2020 - 23/02/2020	210.80	96.25	2.19	4

A.4: Comparison of rainfall recorded at Wet Sleddale and Naddle rain gauges.



A.5: Storm analysis equations.

Specific Peak Discharge (SPD) is calculated by:

$$SPD = \frac{Q}{A} \times 3600 \qquad (A.ii)$$

Where;

Q is peak rate of runoff ($m^3 \cdot s^{-1}$),

A is area (m^2).

Peak runoff coefficient (C) is calculated for each storm event using:

$$C = \frac{Q}{iA} \qquad (A.iii)$$

Where:

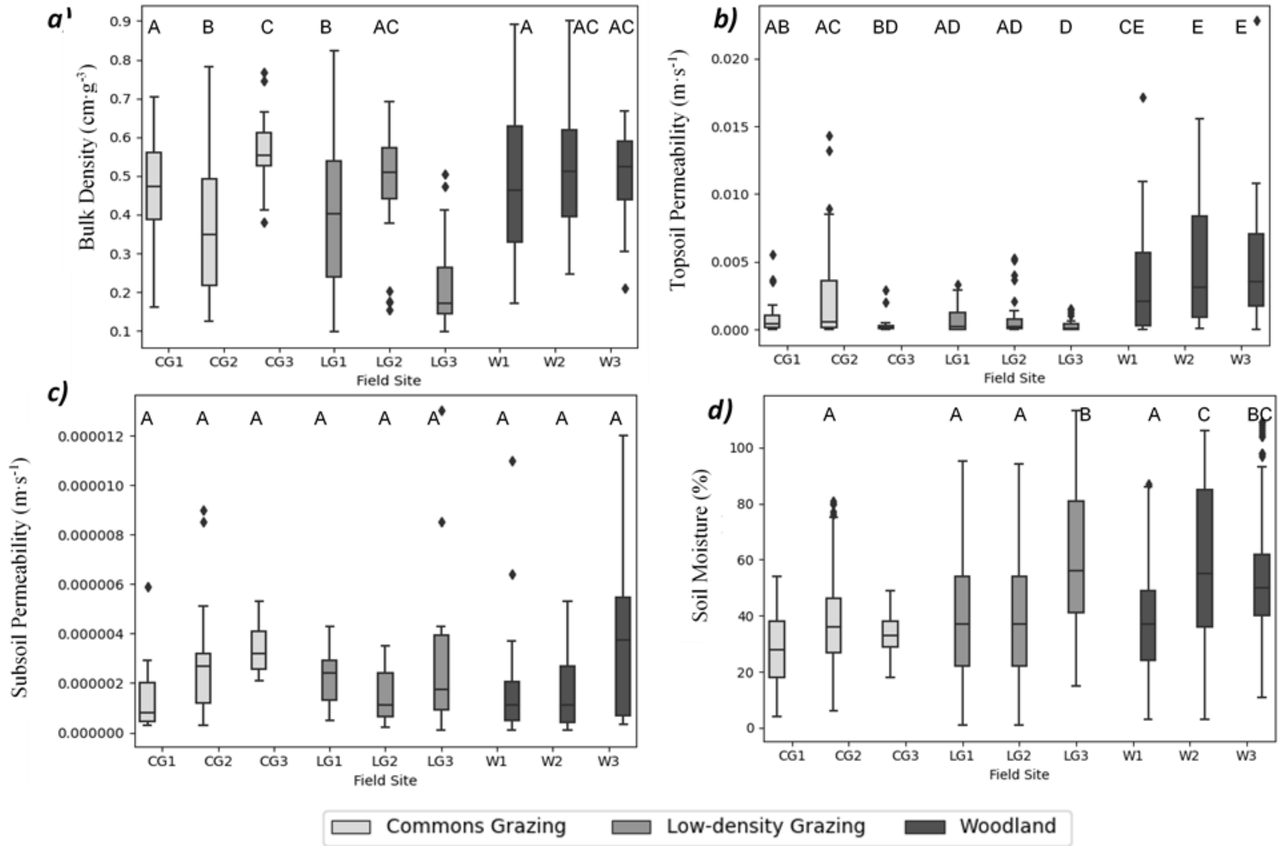
Q is peak rate of runoff ($m^3 \cdot s^{-1}$),

i is maximum rainfall intensity ($m \cdot s^{-1}$),

A is catchment area (m^2).

A.6: Soil properties, all sites (n=9)

Distribution of a) bulk density ($g \cdot cm^{-3}$), b) topsoil permeability ($m \cdot s^{-1}$), c) subsoil permeability ($m \cdot s^{-1}$) d) soil moisture (%) for the nine sites shown as median (line), 25th to 75th percentile (box), 5th to 95th percentile (whiskers).



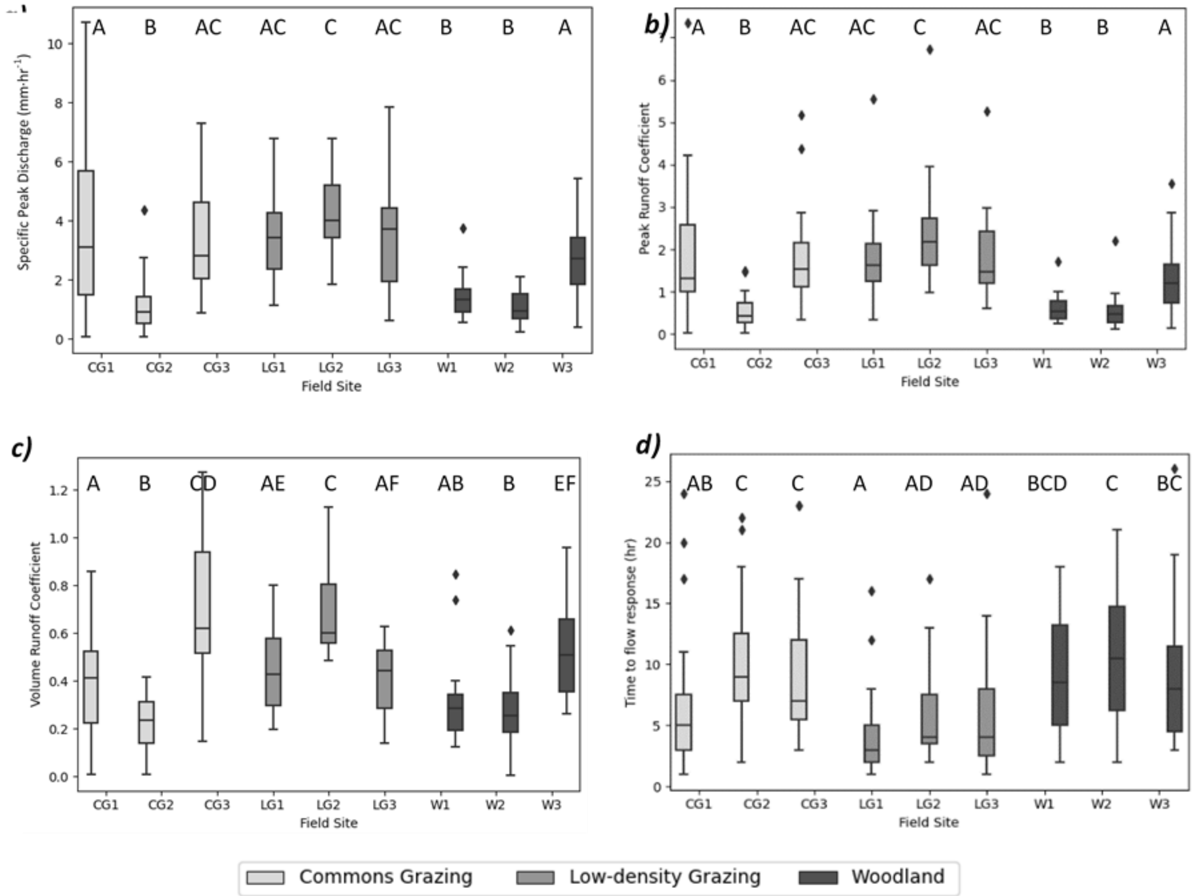
A.7: Soil properties tabulated data, all sites (n=9)

Median (η), mean (μ) and standard error (SEM) for bulk density, topsoil and subsoil permeability and soil moisture for the nine sites.

Site	Bulk Density ($g\cdot cm^{-3}$)			Topsoil permeability, kfs ($m\cdot s^{-1}$)			Subsoil permeability, kfs ($m\cdot s^{-1}$)			Soil Moisture (%)		
	η	**μ**	SEM	η	**μ**	SEM	η	**μ**	SEM	η	**μ**	SEM
Commons Grazing												
Average	0.49	0.46	0.02	2.78E-04	1.33E-03	3.01E-04	2.10E-06	2.47E-06	3.65E-07	32.3	32.7	0.5
CG1	0.48	0.48	0.02	4.44E-04	9.68E-04	2.85E-04	8.00E-07	1.57E-06	4.71E-07	28.2	28.0	0.8
CG2	0.35	0.35	0.03	6.61E-04	2.70E-03	7.88E-04	2.70E-06	2.84E-06	5.59E-07	36.3	37.1	1.0
CG3	0.55	0.55	0.01	1.96E-04	3.47E-04	1.13E-04	3.20E-06	3.45E-06	6.99E-07	33.1	33.4	0.5
Low density Grazing												
Average	0.39	0.36	0.02	1.47E-04	6.68E-04	1.29E-04	1.75E-06	2.25E-06	3.51E-07	47.0	45.6	1.0
LG1	0.40	0.39	0.03	2.44E-04	7.89E-04	2.19E-04	2.40E-06	2.11E-06	2.55E-07	37.1	40.3	1.6
LG2	0.51	0.51	0.02	2.05E-04	9.87E-04	3.19E-04	1.10E-06	1.59E-06	2.87E-07	37.1	40.0	1.5
LG3	0.17	0.20	0.02	7.31E-05	2.90E-04	8.16E-05	1.75E-06	3.49E-06	1.31E-06	56.2	60.7	1.9
Woodland												
Average	0.50	0.50	0.02	2.94E-03	4.58E-03	5.12E-04	1.10E-06	2.21E-06	3.74E-07	50.0	49.1	1.0
W1	0.46	0.49	0.04	2.29E-03	3.65E-03	8.47E-04	1.10E-06	1.91E-06	5.32E-07	36.8	38.2	1.2
W2	0.51	0.50	0.03	4.96E-03	4.97E-03	9.05E-04	1.1E-06	1.79E-06	4.09E-07	54.9	57.6	1.8
W3	0.54	0.52	0.02	3.21E-03	5.05E-03	9.06E-04	3.75E-06	2.36E-06	8.93E-07	50.2	53.3	1.6

A.8: Streamflow properties, all sites (n=9)

Distribution of a) specific peak discharge ($mm \cdot hr^{-1}$), b) peak runoff coefficient, c) volume runoff coefficient c) time to flow response (hr) for the nine sites, shown as median (line), 25th to 75th percentile (box), 5th to 95th percentile (whiskers). Sites which are not statistically different share a letter.



A.9: Soil properties tabulated data, all sites (n=9)

Median (η), mean (μ) and standard error (SEM) for Specific Peak Discharge (SPD), peak runoff coefficient, volume runoff coefficient and time to flow response for the nine sites.

Site	Specific peak flow (SPD) ($mm \cdot hr^{-1}$)			Peak runoff coefficient			Volume runoff coefficient			Time to flow response (hr)		
	η	**μ**	SEM	η	**μ**	SEM	η	**μ**	SEM	η	**μ**	SEM
Commons Grazing												
Average	1.96	2.71	0.26	1.03	1.27	0.12	0.43	0.46	0.04	7	8	0.5
CG1	2.31	3.27	0.55	1.31	1.67	0.24	0.41	0.41	0.04	5	5	0.7
CG2	0.83	1.05	0.15	0.41	0.45	0.06	0.24	0.22	0.03	9	9	0.8
CG3	2.71	3.45	0.38	1.52	1.65	0.17	0.62	0.71	0.06	7	8	0.8
Low density Grazing												
Average	3.76	3.72	0.21	1.82	1.83	0.10	0.52	0.51	0.03	4	5	0.5
LG1	3.44	3.65	0.37	1.58	1.62	0.15	0.43	0.44	0.04	3	4	0.7
LG2	3.99	4.08	0.32	2.17	2.20	0.20	0.60	0.70	0.05	4	5	0.7
LG3	3.59	3.42	0.39	1.47	1.71	0.16	0.44	0.41	0.03	4	5	0.9
Woodland												
Average	1.52	1.80	0.16	0.72	0.87	0.08	0.34	0.40	0.03	8	9	0.7
W1	1.27	1.30	0.18	0.50	0.56	0.08	0.29	0.34	0.07	9	9	1.4
W2	0.96	1.12	0.13	0.45	0.49	0.06	0.25	0.27	0.05	11	11	1.4
W3	2.30	2.48	0.25	1.19	1.25	0.13	0.51	0.54	0.05	8	8	0.9

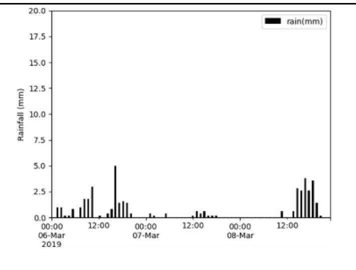
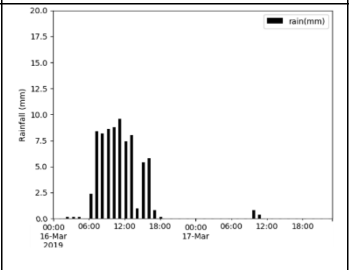
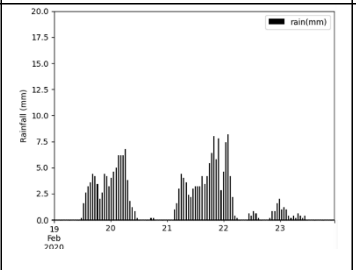
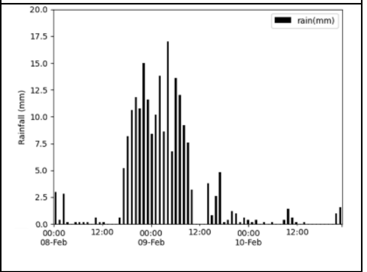
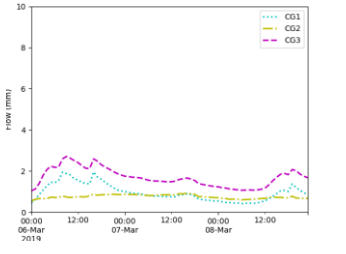
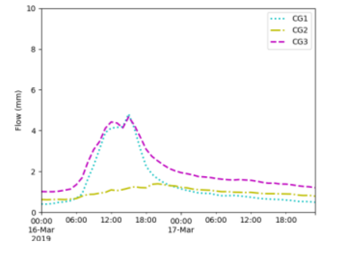
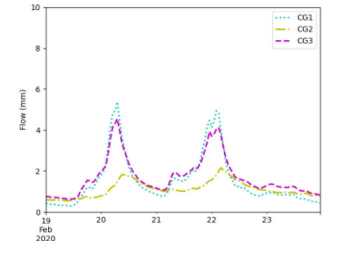
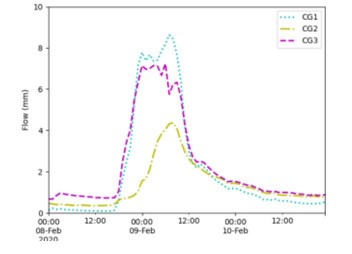
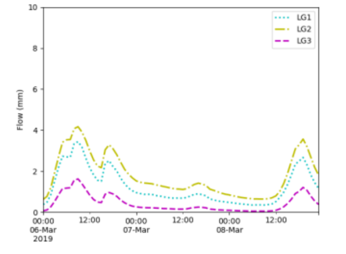
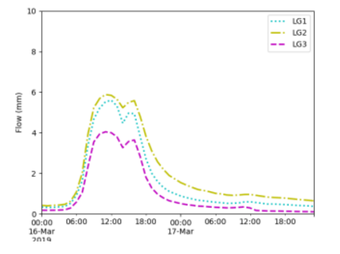
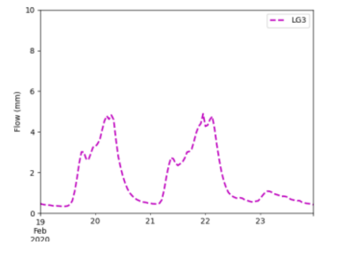
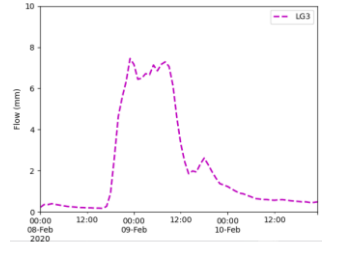
A.10: Streamflow properties divided by return period

Median (η) specific peak discharge (SPD), peak runoff coefficient and volume runoff coefficient for each land cover divided into storms with a return period more than 1.5 years (n = 8 storms), storms with a return period less than 1.5 years (n = 20 storms) with additional detail regarding Storm Ciara.

	Land cover								
	Commons Grazing			**Low-Density Grazing**			**Woodland**		
Return Period (years)	<1.5	>1.5	10 (Storm Ciara)	<1.5	>1.5	10 (Storm Ciara)	<1.5	>1.5	10 (Storm Ciara)
Specific Peak Discharge ($mm \cdot hr^{-1}$)	1.60	4.46	7.93	3.42	5.07	7.45	1.34	2.10	3.04
Peak runoff coefficient	1.09	1.20	1.96	2.07	1.48	2.01	0.76	0.62	1.01
Volume runoff coefficient	0.48	0.43	0.43	0.52	0.55	0.49	0.53	0.32	0.25

A.11: Example hydrograph response for return periods; <1.5 years, 1.5 years, 4 years and 10 years

Four rainfall events recorded at the Naddle rain gauge, with differing return periods and the relating hydrograph responses for the nine sites investigated. The rainfall event number relates to information in Appendix A.3.

Rainfall Event	6	9	28	26
Return Period (years)	< 1.5	1.5	4	10 (Storm Ciara)
Rainfall (mm)				
Commons Grazing				
Low-density Grazing				

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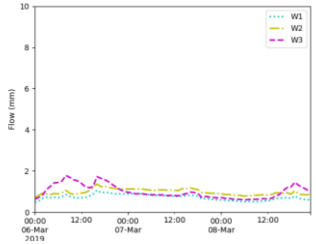
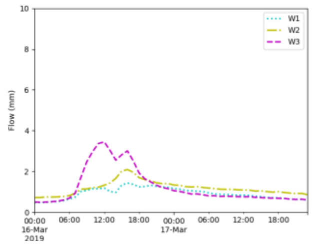
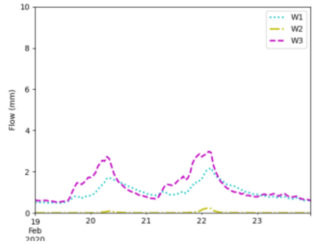
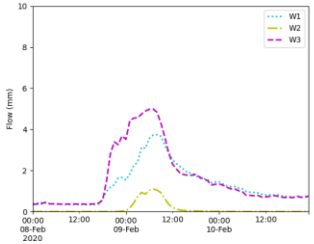
<td></td>
<td></td>
<td></td>
<td></td>
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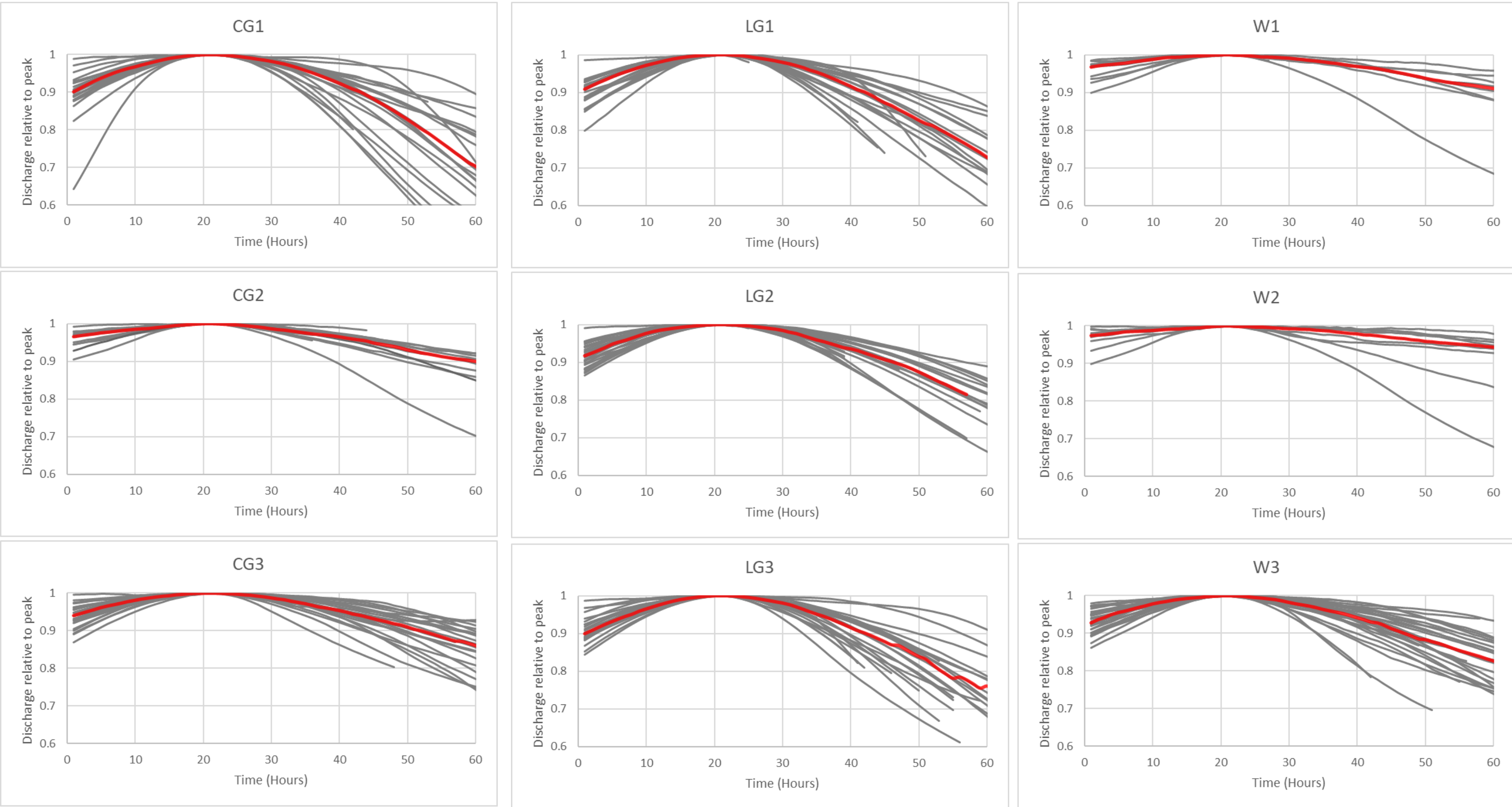
</table>

A.12: Forms of hydrograph peaks for all events exceeding the 97.5 % threshold.

Each figure shows all data for one catchment, with heavy line showing the medians, plotted together in Main figure 6. Data for each storm are normalised to 100 % for the peak flow. Time steps are in hours. Sites are close enough together to have received essentially the same rainfall series.



Appendix B: Supporting Information for Chapter 4

B.1: CEH NRFA Land cover categories

Class Number	Class Name	Equivalent LCM classes	LCM class numbers
1	Woodland (Forestland)	Broadleaved woodland Coniferous woodland	1,2
2	Arable (Cropland)	Arable and horticulture	3
3	Grassland	Improved Grassland Neutral grassland Calcareous grassland Acid grassland Fen, Marsh and Swamp Heather Heather grassland Bog Supra-littoral sediment Saltmarsh Littoral sediment	4,5,6,7,8,9,10,11,16,18,19
4	Water (Wetlands)	Freshwater	14
5	Built-up areas	Urban Suburban	20,21
6	Other	Inland rock Saltwater Supra-littoral rock Littoral rock	12,13,15,17

B.2: CEH NRFA Superficial Deposits categories

Superficial Deposits

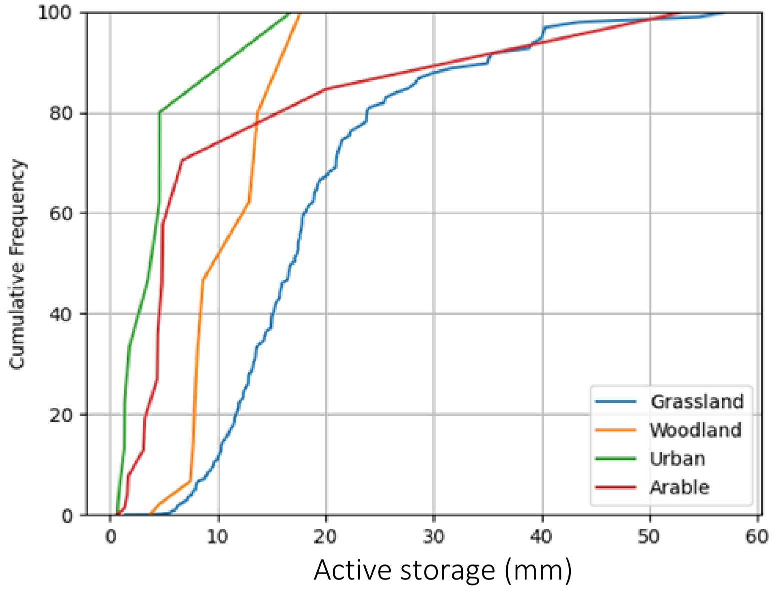
Proportions of the catchment covered by superficial deposits of generally high, generally low and mixed permeability based on classification of the BGS 1:625000 Superficial Deposits layer (version 5) as indicated in the table below. Superficial deposits generally have much more spatially variable permeability than bedrock. Superficial deposits vary greatly in their extent across the UK, some catchments having very extensive cover whilst others have negligible cover. Therefore, the percentages in each category do not sum to 100% in most catchments.

NRFA Superficial Deposits Permeability Class	BGS Lexicon Entry
Generally high permeability	BLOWN SAND
Generally high permeability	GLACIAL SAND AND GRAVEL
Generally high permeability	RAISED MARINE DEPOSITS (UNDIFFERENTIATED)
Generally high permeability	RIVER TERRACE DEPOSITS (UNDIFFERENTIATED)
Generally high permeability	SAND AND GRAVEL OF UNCERTAIN AGE AND ORIGIN
Generally low permeability	CLAY-WITH-FLINTS
Generally low permeability	LACUSTRINE DEPOSITS (UNDIFFERENTIATED)
Generally low permeability	PEAT
Mixed permeability	BRICKEARTH
Mixed permeability	TILL
Mixed permeability	ALLUVIUM
Mixed permeability	LANDSLIP

B.3: Active storage (*m*) tabulated data for catchments with ≥ 70 % coverage of one land cover type.

≥ 70% Catchment Land Cover Type	*n*	Median Effective Storage parameter, *m* (mm)	Standard deviation (ddof =1)	Mean catchment size (km^2)	Median Annual Rainfall (mm)
Woodland	10	8.00	4.07	2.75	1950
Arable	13	4.38	13.77	25.55	630
Grassland	190	12.56	9.10	89.74	1300
Urban	10	1.59	4.63	20.69	665

B.4: Cumulative percentage frequency distribution plot of active storage, *m* for natural catchments with ≥ 70 % one land cover (n = 233).



B.5: TOPMODEL sets (Conifer and Broadleaf GLM alternative)

TOPMODEL sets for factors explaining active storage, *m* which are included in model averaging (relating to B.6), where woodland cover is substituted for conifer and broadleaf cover. LogLik: log-likelihood; AICc: Akaike's information criterion corrected for small sample size.

		loglik	AICc	ΔAICc	AICc weight
1	Elevation + log (rainfall) + log (size) + low superficial deposits permeability + sqrt (arable cover) + sqrt (grassland cover) + sqrt (urban cover)	-1405.988	2830.4	0.00	0.271
2	Elevation + log (rainfall) + log (size) + low superficial deposits permeability + sqrt (arable cover) + sqrt (broadleaf cover) + sqrt (grassland cover) + sqrt (urban cover)	-1405.003	2830.5	0.13	0.254
3	log (rainfall) + log (size) + low superficial deposits permeability + sqrt (arable cover) + sqrt (broadleaf cover) + sqrt (grassland cover) + sqrt (urban cover)	-1406.472	2831.4	0.97	0.167
4	Elevation + log (rainfall) + log (size) + low superficial deposits permeability + sqrt (arable cover) + sqrt (conifer cover) + sqrt (grassland cover) + sqrt (urban cover)	-1405.474	2831.5	1.07	0.158
5	Elevation + log (rainfall) + log (size) + low superficial deposits permeability + sqrt (arable cover) + sqrt (broadleaf cover) + sqrt (conifer cover) + sqrt (grassland cover) + sqrt (urban cover)	-1404.476	2831.6	1.18	0.150

B.6: Model-averaged estimates (Conifer and Broadleaf GLM alternative)

Model-averaged estimates for factors affecting active storage (*m*); n = 4 models. Significant variables in bold text.

Variable	Estimate	Std. error	*p* value	Relative importance
Broadleaf cover (sqrt)	0.0292	0.0369	0.43	0.57
Conifer cover (sqrt)	0.0081	0.0199	0.69	0.32
Urban cover (sqrt)	**-0.0826**	**0.0287**	**<0.001**	**1**
Arable cover (sqrt)	**0.0900**	**0.0245**	**<0.0001**	**1**
Grassland cover (sqrt)	**0.0723**	**0.0323**	**<0.01**	**1**
High permeability superficial deposits	-	-	-	-
Low permeability superficial deposits	**-0.0075**	**0.0026**	**<0.001**	**1**
Catchment size (log)	**0.0992**	**0.0368**	**<0.001**	**1**
Average annual rainfall (log)	**0.8581**	**0.1543**	**<0.0001**	**1**
Average catchment elevation	-0.0007	0.0005	0.207	0.83

B.7: TOPMODEL sets

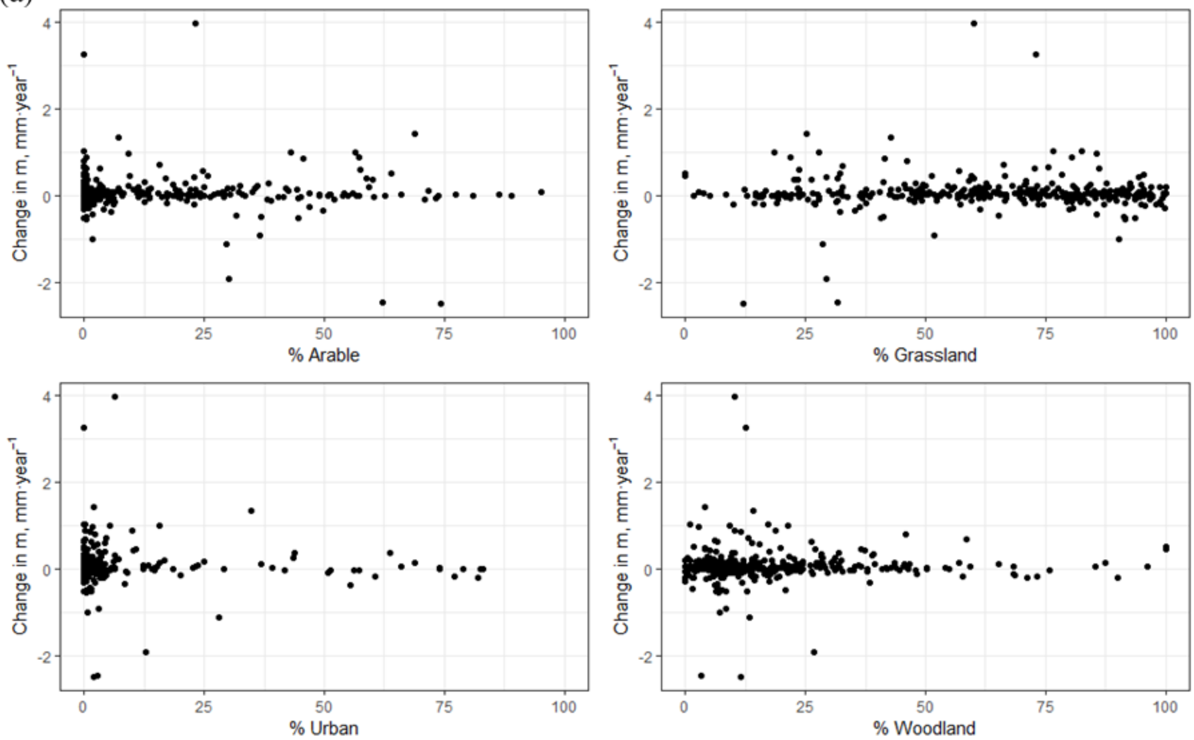
TOPMODEL sets for factors explaining active storage, *m* which are included in model averaging (relating to Table 4.2). LogLik: log-likelihood; AICc: Akaike's information criterion corrected for small sample size.

		loglik	AICc	ΔAICc	AICc weight
1	Elevation + log (rainfall) + log (size) + low superficial deposits permeability + sqrt (arable cover) + sqrt (grassland cover) + sqrt (urban cover) + sqrt (woodland cover)	-1404.92	2830.38	0.00	0.35
2	Elevation + log (rainfall) + log (size) + low superficial deposits permeability + sqrt (arable cover) + sqrt (grassland cover) + sqrt (urban cover)	-1405.99	2830.42	0.03	0.34
3	log (rainfall) + log (size) + low superficial deposits permeability + sqrt (arable cover) + sqrt (grassland cover) + sqrt (urban cover) + sqrt (woodland cover)	-1406.64	2831.72	1.33	0.18
4	Elevation + high bedrock permeability + log (rainfall) + log (size) + low superficial deposits permeability + sqrt (arable cover) + sqrt (grassland cover) + sqrt (urban cover)	-1404.83	2832.31	1.93	0.13

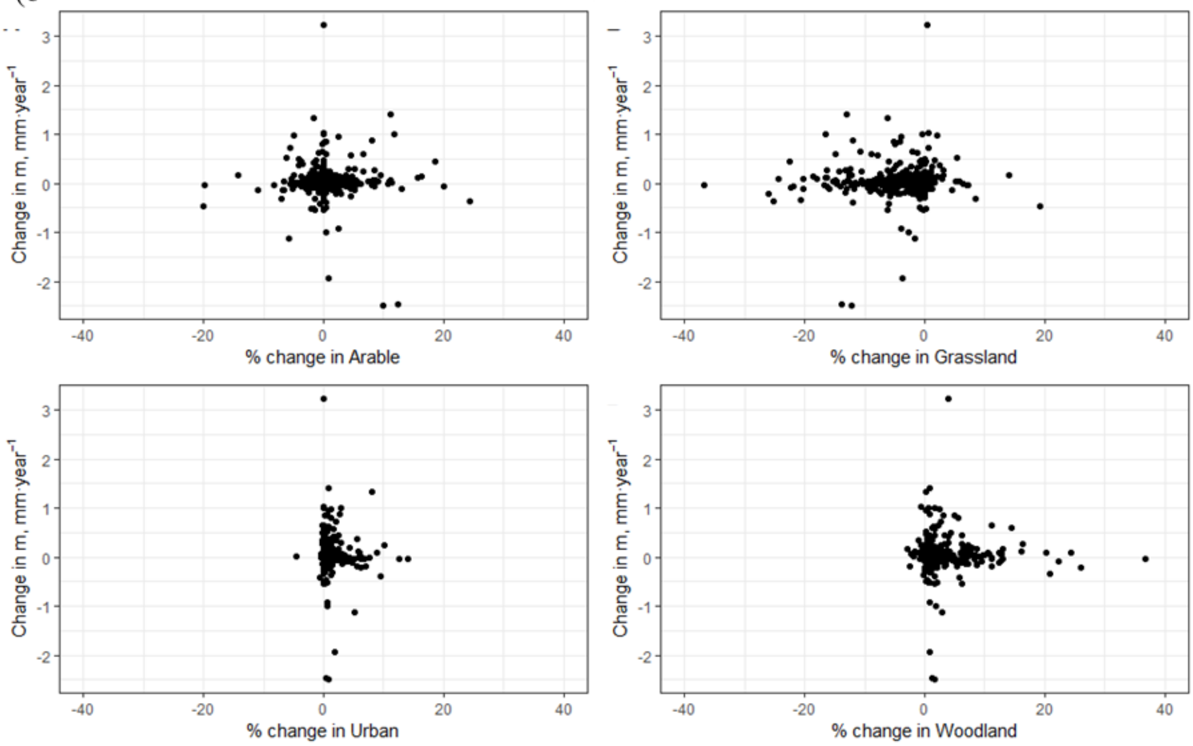
B.8: Land cover distribution plots

Change in active storage against a) current land cover or b) change in land cover between 1990 and 2015 for natural catchments (n=418).

(a)



(b



Appendix C: Supporting Information for Chapter 5

C.1: A sample of calibration runs used to identify the best fit parameters

Table c.i. Example range of parameters used in calibration runs.

	Parameters		
	m(m)	**K (m/hr)**	**Kv (-)**
Baseline	0.006	6.9	40
Calibration 1	0.006	6.7	40
Calibration 2	0.006	7.1	40
Calibration 3	0.008	6.9	40
Calibration 4	0.008	6.7	40
Calibration 5	0.008	7.1	40
Calibration 6	0.006	6.9	50
Calibration 7	0.006	6.7	50
Calibration 8	0.006	7.1	50
Calibration 9	0.008	6.9	50
Calibration 10	0.008	6.7	50
Calibration 11	0.008	7.1	50

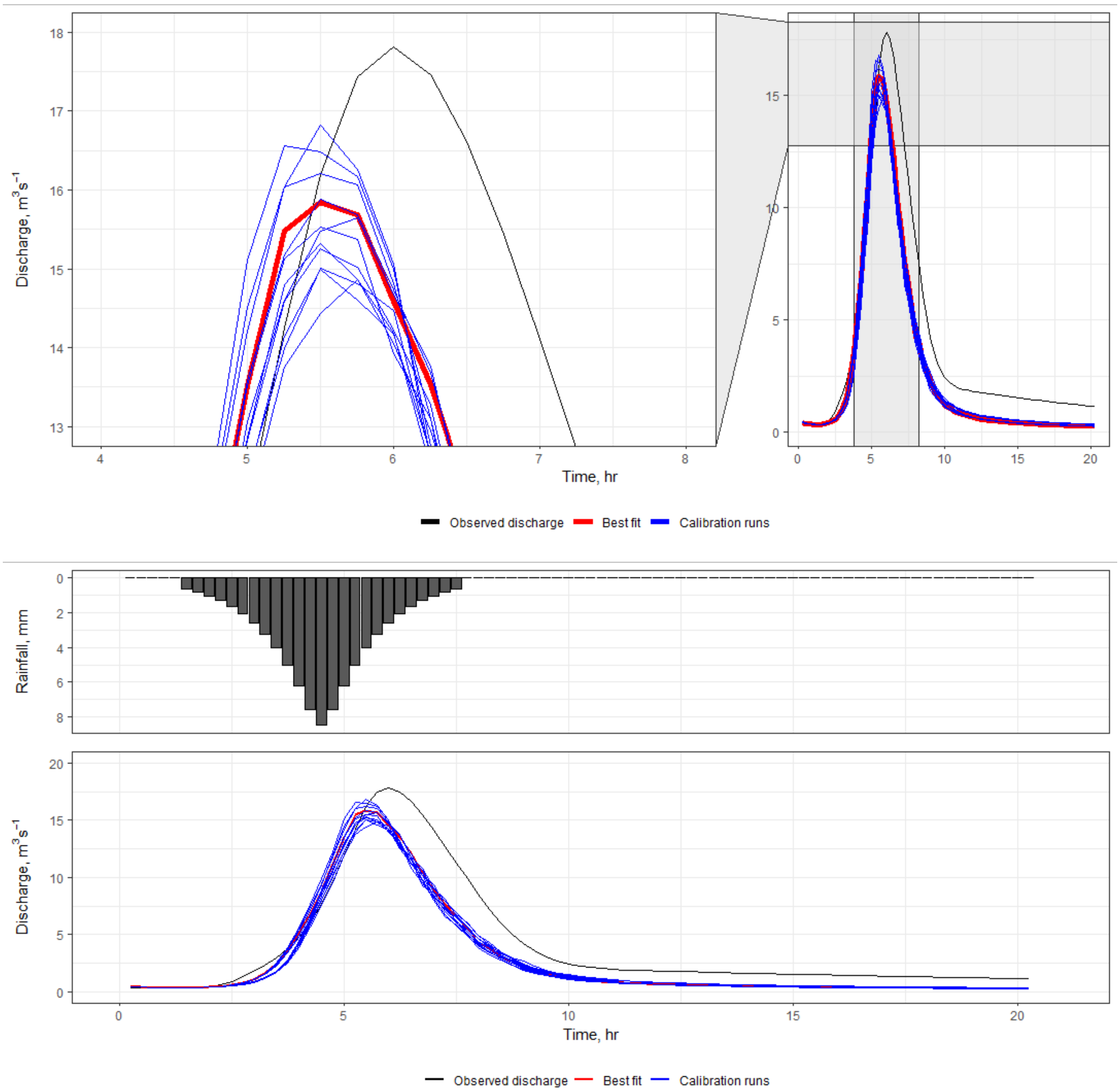
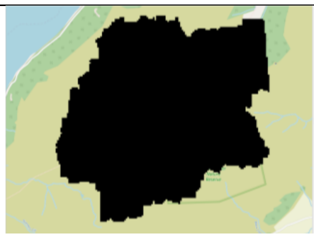
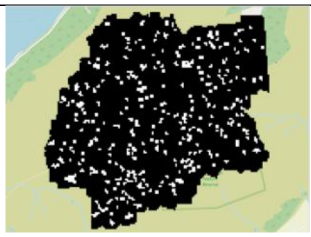
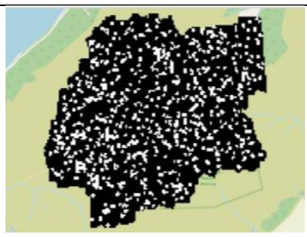
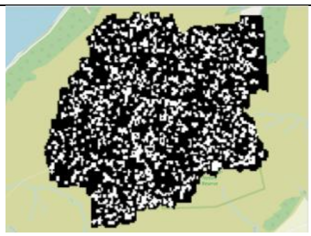
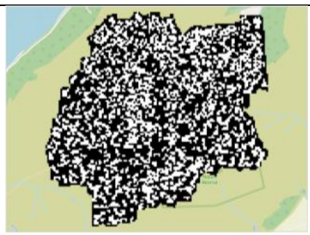
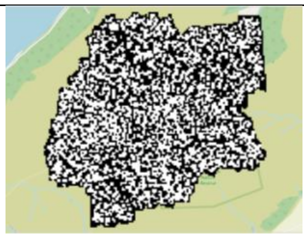
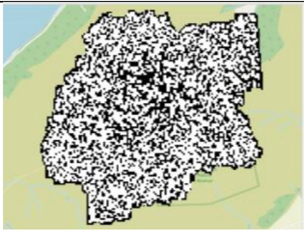
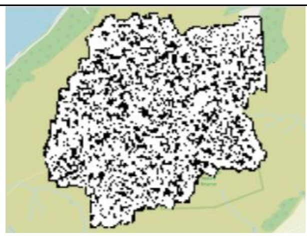
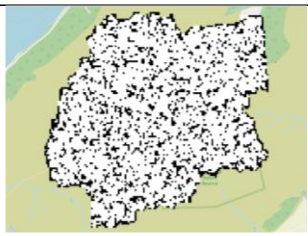
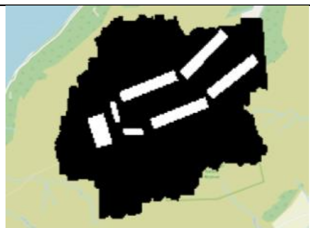
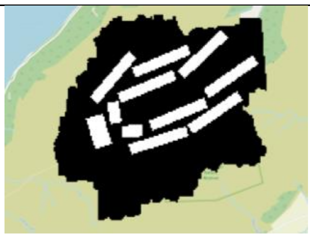
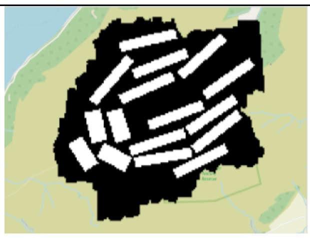
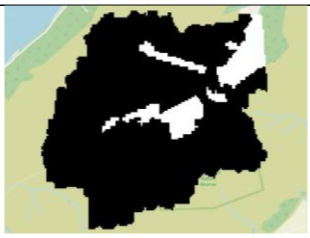
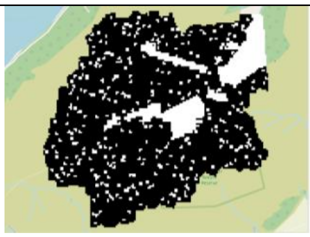
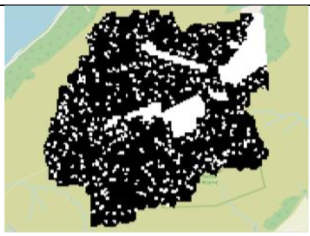
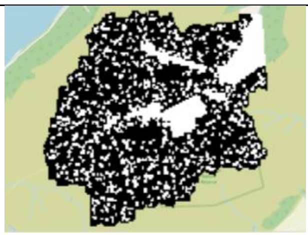
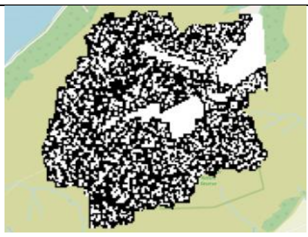


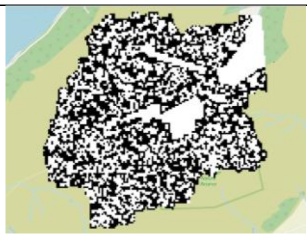
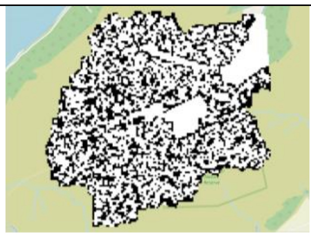
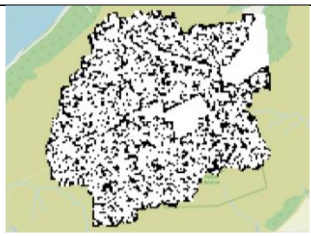
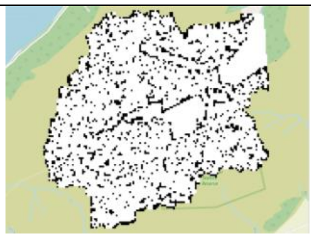
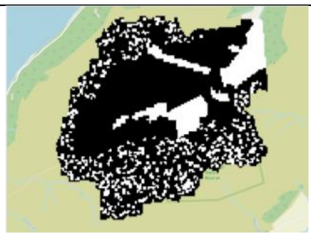
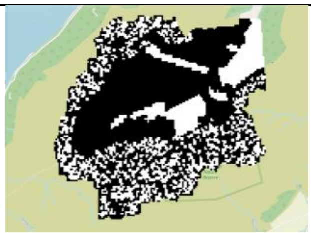
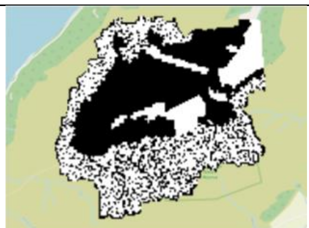
Figure c.i. Hydrograph response of calibrations in Table c.i, above, to the 1 in 50-year ReFH storm event.

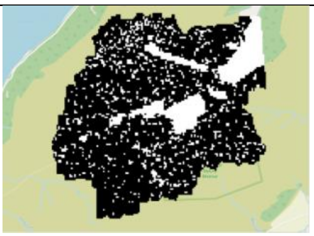
C.2: Scenario land cover maps

Scenarios	**Woodland cover (%)**	**No. of Repeats**	**Scenario land cover map (example if more than 1 repeat)**
No woodland	0	1	
Catchment	10	5	
	20	5	
	30	5	
	40	5	
	50	5	

	60	5	
	70	5	
	80	5	
Riparian	10	1	
	20	1	
	30	1	
Cross-slope	10	1	

	20	1	
	30	1	
Naddle (Baseline)	12.9	1	
Catchment + Baseline	20	5	
	25	5	
	35	5	
	45	5	

	55	5	
	65	5	
	75	5	
	85	5	
Randomised above 400m + Baseline	25	5	
	35	5	
	45	5	

RSPB + Baseline	22.9	1	

C.3: Sensitivity analysis of model parameters

We tested the sensitivity of parameters which are spatially distributed by land cover; interception, K (hydraulic conductivity) and Kv (surface roughness).

We changed values for wooded areas (grassland remained 1).

Model parameters remained the same [m = 6, K = 6.9, Kv = 40]

We tested the following ranges:
Interception: 0.8-0.9
K (hydraulic conductivity [KSAT and T0]): 5 to 80
Kv (Surface roughness – KLC): 0.95 to 0.7

Table c.ii. Spatially distributed parameter values for woodland

Scenario	Woodland spatially distributed values				Results	
	m(m)	K(T0) (m/hr) KSAT	Kv (-) KLC	Interception (%)	Peak flow (m^3/s)	% change
Current	1	20	0.9	0.87	15.85	
s_2	1	5	0.9	0.87	16.58	4.61
s_3	1	10	0.9	0.87	16.15	1.89
	1	20	0.9	0.87	15.85	
s_4	1	40	0.9	0.87	16.25	2.52
s_5	1	80	0.9	0.87	16.3	2.84
s_7	1	20	0.95	0.87	15.91	0.38
	1	20	0.9	0.87	15.85	
s_8	1	20	0.8	0.87	15.94	0.57
s_9	1	20	0.7	0.87	16.03	1.14
s_11	1	20	0.9	0.9	16.14	1.83
	1	20	0.9	0.87	15.85	
s_12	1	20	0.9	0.85	16.08	1.45
s_13	1	20	0.9	0.825	16.01	1.01
s_14	1	20	0.9	0.8	15.97	0.76

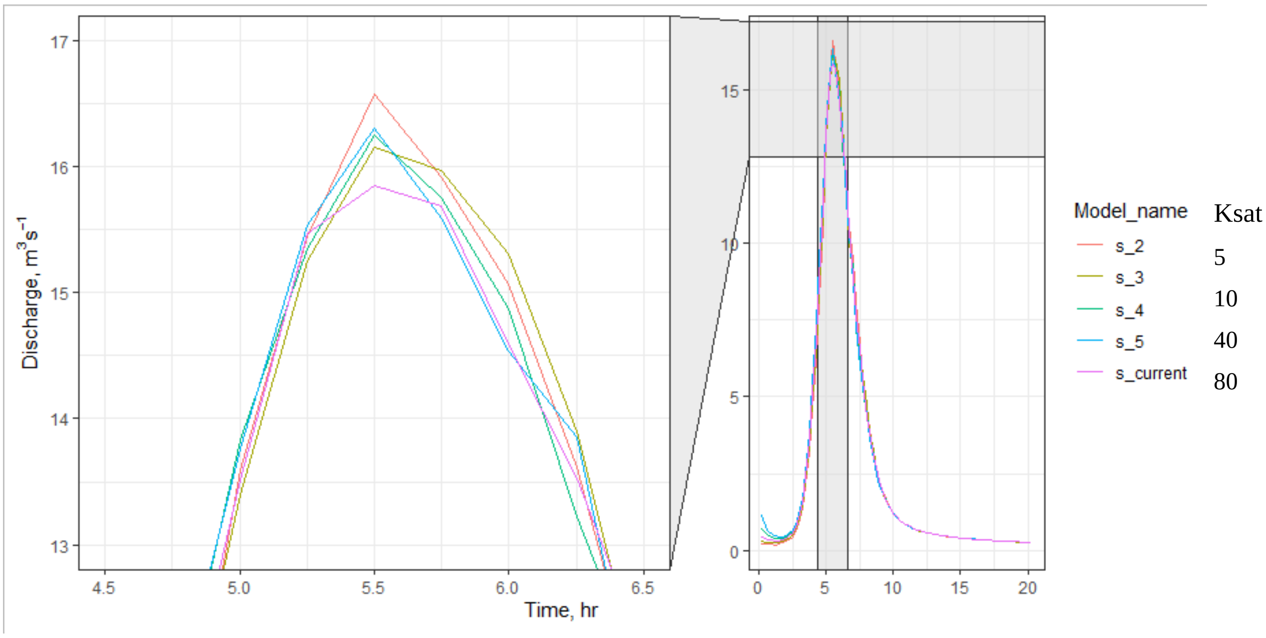


Figure c.ii: Changing K, hydraulic conductivity (roughness and interception the same)

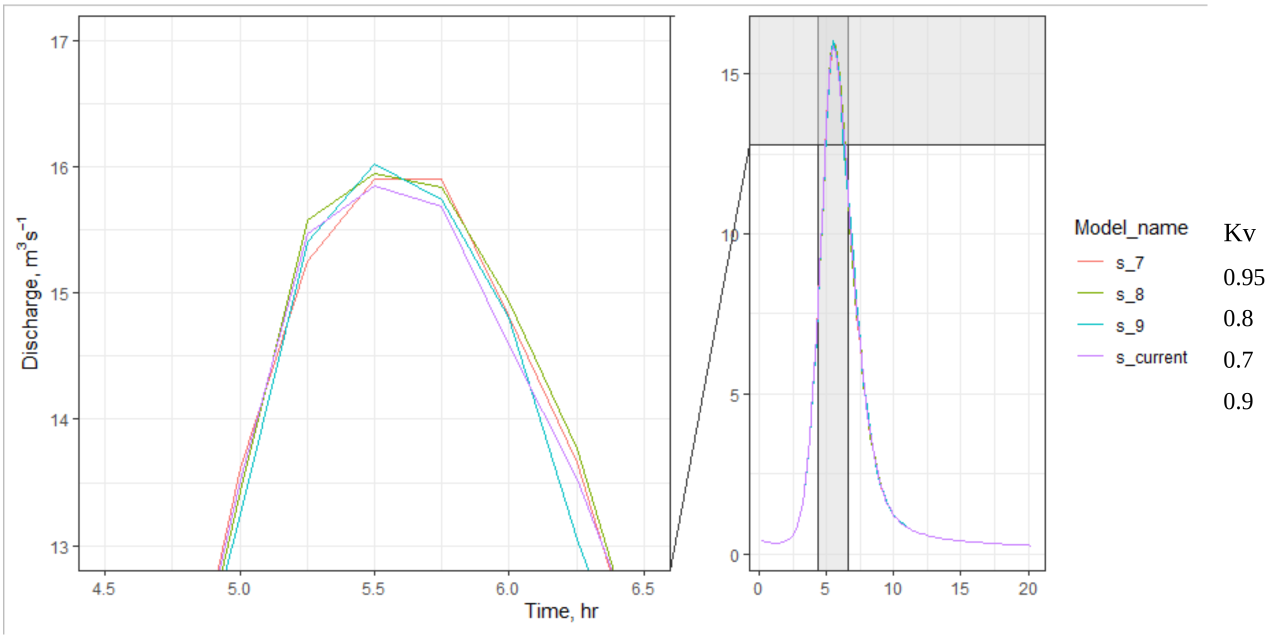
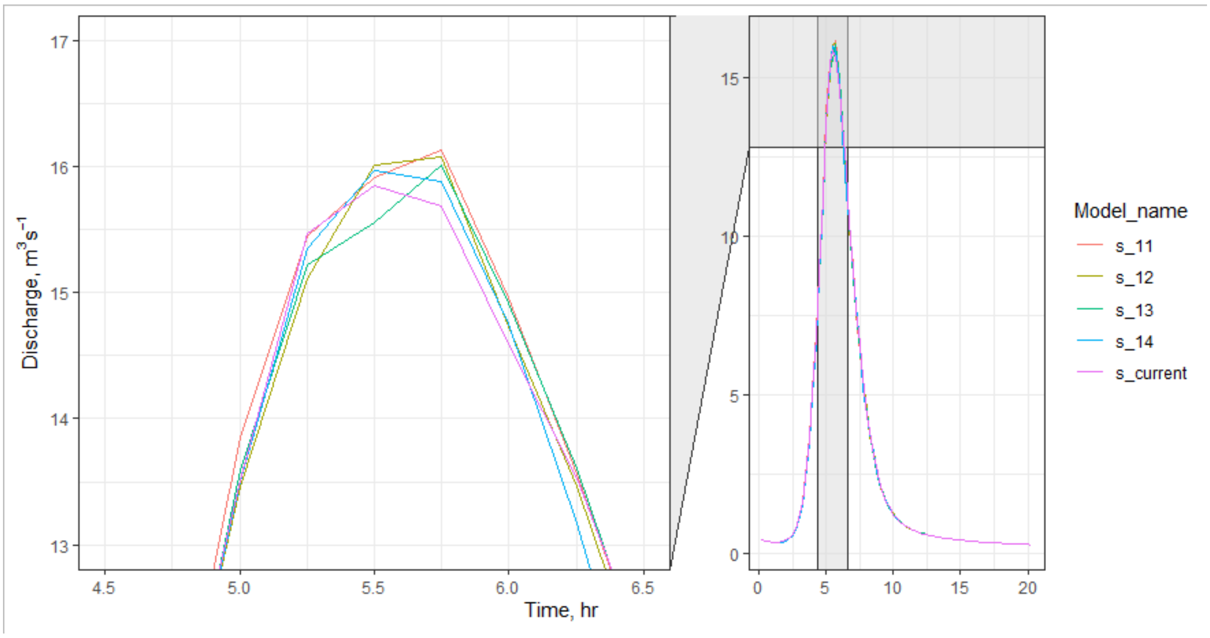


Figure c.iii. Changing Kv, surface roughness (permeability and interception the same)



Interception
0.9
0.85
0.825
0.8

Figure c.iv. Changing interception (permeability and roughness the same)

C.4: Results from all model runs

Scenario	Repeat No.	% woodland	% grassland	Peak discharge ($m^{-3}\cdot s$)	Peak change to un-forested (%)	Peak QB	Peak QSOF
No Woodland	1	0	100	17.17		3.44	14.09
Catchment woodland	1	10	90	16.79	-2.21	5.2	12.45
	2	10	90	16.58	-3.44	5.09	12.34
	3	10	90	16.18	-5.77	5.23	11.88
	4	10	90	16.35	-4.78	5.19	12.06
	5	10	90	16.62	-3.20	5.19	12.37
	1	20	80	16.05	-6.52	6.6	10.81
	2	20	80	15.88	-7.51	6.66	10.77
	3	20	80	15.97	-6.99	6.6	10.85
	4	20	80	16.09	-6.29	6.68	10.79
	5	20	80	15.89	-7.45	6.74	10.57
	1	30	70	15.53	-9.55	7.88	9.87
	2	30	70	15.75	-8.27	8.07	9.61
	3	30	70	15.72	-8.44	7.94	9.54
	4	30	70	15.96	-7.05	8	9.85
	5	30	70	15.55	-9.44	8.07	9.79
	1	40	60	15.06	-12.29	9.29	8.38
	2	40	60	15.28	-11.01	9.21	8.49
	3	40	60	15.13	-11.88	9.24	8.41

	4	40	60	15.16	-11.71	9.12	8.45
	5	40	60	15.17	-11.65	9.26	8.23
	1	50	50	14.87	-13.40	10.39	7.31
	2	50	50	14.98	-12.75	10.46	7.17
	3	50	50	15.00	-12.64	10.45	7.33
	4	50	50	15.11	-12.00	10.45	7.47
	5	50	50	14.94	-12.99	10.45	7.15
	1	60	40	14.76	-14.04	11.74	6.08
	2	60	40	14.66	-14.62	11.49	6.17
	3	60	40	14.77	-13.98	11.55	6.34
	4	60	40	14.73	-14.21	11.56	6.09
	5	60	40	14.65	-14.68	11.56	6.34
	1	70	30	14.76	-14.04	12.72	5.22
	2	70	30	14.82	-13.69	12.8	5.08
	3	70	30	14.74	-14.15	12.61	5.3
	4	70	30	14.69	-14.44	12.66	5.41
	5	70	30	14.78	-13.92	12.74	5.1
	1	80	20	15.18	-11.59	13.78	4.54
	2	80	20	15.25	-11.18	13.75	4.4
	3	80	20	15.23	-11.30	13.71	4.44
	4	80	20	15.24	-11.24	13.76	4.17
	5	80	20	15.27	-11.07	13.76	4.33
Riparian	1	10	90	16.73	-2.56	4.75	12.61
	1	20	80	15.76	-8.21	6.41	10.65

	1	30	70	15.49	-9.78	7.74	9.39
Cross-slope	1	10	90	15.98	-6.92	5.54	11.4
	1	20	80	15.68	-8.68	6.71	10.7
	1	30	70	15.41	-10.24	7.86	9.59
Naddle (Base)				15.85	-7.70	5.64	11.44
Naddle + Catchment woodland	1	20	80	16.25	-5.36	6.68	11.06
	2	20	80	15.69	-8.62	6.71	10.84
	3	20	80	15.97	-6.99	6.66	11.06
	4	20	80	16.13	-6.06	6.58	10.88
	5	20	80	15.79	-8.04	6.63	10.54
	1	25	75	16.02	-6.70	7.2	10.4
	2	25	75	15.52	-9.61	7.23	9.8
	3	25	75	15.66	-8.79	7.26	10.02
	4	25	75	15.92	-7.28	7.2	10.3
	5	25	75	15.72	-8.44	7.3	10.15
	1	35	65	15.29	-10.95	8.56	9.09
	2	35	65	15.22	-11.36	8.46	9.18
	3	35	65	15.89	-7.45	8.45	9.54
	4	35	65	15.51	-9.67	8.41	9.18
	5	35	65	15.39	-10.37	8.51	9.06
	1	45	55	14.86	-13.45	9.97	7.64
	2	45	55	15.09	-12.11	9.79	7.73

	3	45	55	14.99	-12.70	9.73	7.69
	4	45	55	14.97	-12.81	9.6	8.19
	5	45	55	15.09	-12.11	9.8	8.19
	1	55	45	14.64	-14.74	10.68	7.28
	2	55	45	14.62	-14.85	11.04	6.77
	3	55	45	14.78	-13.92	10.87	7.08
	4	55	45	14.64	-14.74	10.92	6.92
	5	55	45	14.52	-15.43	10.97	6.6
	1	65	35	14.45	-15.84	12.05	5.82
	2	65	35	14.53	-15.38	12.08	5.93
	3	65	35	14.55	-15.26	12.02	5.93
	4	65	35	14.63	-14.79	12.15	5.7
	5	65	35	14.58	-15.08	12.08	5.98
	1	75	25	14.83	-13.63	12.99	5.1
	2	75	25	14.94	-12.99	13.17	4.77
	3	75	25	14.90	-13.22	13.25	4.71
	4	75	25	14.89	-13.28	13.18	4.8
	5	75	25	14.90	-13.22	13.12	4.91
	1	85	15	15.37	-10.48	14.22	3.95
	2	85	15	15.45	-10.02	14.2	4.02
	3	85	15	15.46	-9.96	14.25	4.02
	4	85	15	15.44	-10.08	14.2	3.88
	5	85	15	15.51	-9.67	14.27	3.75
	1	25	75	15.86	-7.63	7.37	10.22

Naddle + Upland catchment	2	25	75	15.69	-8.62	7.23	9.8
	3	25	75	16.31	-5.01	7.26	10.02
	4	25	75	16.16	-5.88	7.2	10.3
	5	25	75	15.84	-7.75	7.3	10.15
	1	35	65	15.97	-6.99	8.53	9.69
	2	35	65	15.62	-9.03	8.46	9.18
	3	35	65	15.49	-9.78	8.45	9.54
	4	35	65	15.74	-8.33	8.41	9.18
	5	35	65	15.46	-9.96	8.51	9.06
	1	45	55	15.44	-10.08	9.86	7.94
	2	45	55	15.35	-10.60	9.79	7.73
	3	45	55	15.16	-11.71	9.72	7.69
	4	45	55	15.14	-11.82	9.6	8.19
	5	45	55	15.22	-11.36	9.8	8.19
Naddle + RSBP	1	23		15.53	-9.55	7.04	10.21