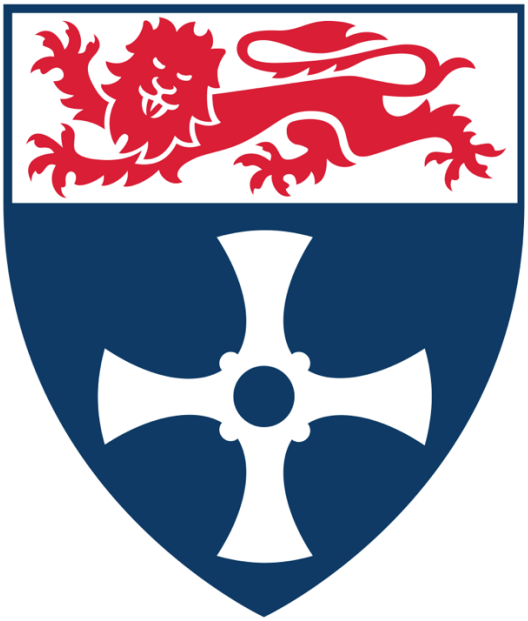
Predicting Hydromorphological Responses to Dam Removal

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A Thesis submitted to Newcastle University in partial fulfilment of the requirements for the degree of Doctor of Engineering (EngD)

January 2022

Declaration

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

Signed: 

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Abstract

The alteration of flow regimes through abstraction and impoundment represents a major threat to aquatic species and ecosystems. Dam and weir removal have become attractive restoration measures for mitigating flow regime alteration, habitat fragmentation and other ecological impacts. Yet, large heterogeneity between study sites, a lack of historical data and complex underlying processes, mean predicting the hydromorphological response remains challenging.

In this work, the novel application of Physically Based Spatially Distributed catchment modelling, Geomorphic Modelling and Level Pool Routing combined with extensive fieldwork is used to predict the hydromorphological consequences of weir removal at Ennerdale Water; a Special Area of Conservation in the Upper Ehen catchment, where impoundment and abstraction date back to the 1800s. The area is home to the last remaining stronghold of endangered freshwater pearl mussels in England and Wales and is subject to unique natural and managed conditions.

Using historic flow regime data, a national-scale assessment of freshwater pearl mussel rivers is conducted. It shows that healthier mussel populations inhabit rivers with less extreme fluctuations between low and high flows and catchments with lower magnitude high flow events. Healthy populations, however, are found in both regulated and unregulated rivers.

Abstraction has reduced mean flows in the Upper Ehen by 10% from 1995-2019. Without impounding infrastructure and flow regulation, minimum flows are predicted to fall by around 65% with no change to Q_{95} values. In the event of weir removal, mean lake levels are predicted to fall by 0.95m, resulting in a 5% reduction in the surface area of Ennerdale Water.

The novel application of water-borne ground penetrating radar has shown evidence of historic lake outlets and previous lateral channel movement at the head of Ennerdale Water. Lakebed analysis identified small pockets of fine sediment, totalling an estimated 825 m^3, directly upstream of the weir. Chemical analysis identified no high levels of contamination. Following weir removal, sediment transport rates were predicted to vary between 0.25 m^3 and 309m^3 per year, averaging around 30 m^3/year. Transport was highly correlated with extreme high flows.

Correctly managed weir removal represents an opportunity for United Utilities to align the interests of regulators, customers and wider stakeholders. The hydrological and geomorphic results presented, provide critical and unique evidence for achieving this outcome. Furthermore, this study presents a novel and thorough methodology for investigating weir and dam removal that can be applied to other sites requiring ecological restoration.

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Abbreviations

CAMS	Catchment Abstraction Management Strategy
CEH	Centre for Ecology and Hydrology
DEM	Digital Elevation Model
EA	Environment Agency
EU	European Union
FARL	Flood Attenuation by Reservoirs and Lakes
FDC	Flow Duration Curve
FPM	Freshwater Pearl Mussel
GEAR	Gridded Estimates of Areal Rainfall
GEP	Good Ecological Potential
GES	Good Ecological Status
GPR	Ground Penetrating Radar
HD	Habitats Directive
HES	Ecological Potential
HMWB	Heavily Modified Water Body
HRA	Habitats Regulations Assessment
IPENS	Improvement Programme for England's Natura 2000 sites
LPR	Level Pool Routing
MAF	Minimum Acceptable Flow
mAOD	Metres Above Ordnance datum
NE	Natural England
NRFA	National Rivers Flow Archive
NSE	Nash–Sutcliffe Efficiency

PBSD	Physically Based Spatially Distributed
PET	Potential Evapotranspiration
PSG	Project Steering Group
RAM	Resource Assessment and Management
RMSE	Root Mean Square Error
RoC	Review of Consents
SAC	Special Area of Conservation
SDC	Stage Duration Curve
SIP	Site Improvement Plan
SSSI	Site of Special Scientific Interest
TBR	Tipping Bucket Rain gauge
UU	United Utilities
WFD	Water Framework Directive
WRMP	Water Resource Management Plan
WRZ	Water Resource Zone

Chapter 1 - Introduction

1.1 Background and Rationale

The alteration of flow regimes through impoundment and abstraction is one the most serious threats to the sustainability of rivers, floodplains and wetlands (Bunn and Arthington, 2002). It is now widely accepted that the way society uses rivers in the future must account for the need to maintain healthy aquatic ecosystems (Horne *et al.*, 2017). There is also widespread recognition that natural river systems provide society with a range of ecosystem goods and services (Gilvear *et al.*, 2013). Yet, despite this, balancing the needs of society and the environment remains one of the major barriers to achieving truly sustainable water resource management (Alcamo *et al.*, 2008; Olden and Naiman, 2010; Randklev *et al.*, 2018).

Towards the end of the 20th century, the concept of 'environmental flows' (e-flows) rapidly spread in response to increased concern and awareness of the impacts of in-stream barriers and flow regulation on freshwater ecology (Arthington *et al.*, 2006). By the early 2000s a range of methodologies for estimating the environmental flow requirements of rivers and their floodplains existed and a 2003 global review of environmental flow methodologies identified the existence of more than 200 approaches in use across 44 countries around the world (Tharme, 2003).

Multiple definitions exist for the term environmental flows, however, the Brisbane Declaration (2007) provides the following widely used definition:

> *The quantities, quality and timing of water flows required to sustain freshwater and estuarine ecosystems and the services they provide to people are termed environmental flows.*

The primary aim of environmental flows is to alter the magnitude and timing of flow in systems modified by water infrastructure in an effort to restore flow regimes that benefit downstream reaches and riparian ecosystems (Poff and Matthews, 2013). Yet, whilst the general concept of environmental flows is universal, Horne *et al*, (2017) highlight that the precise definition and meaning of the term is context specific and may vary among different users and in different settings. Despite potential for different interpretations, the guiding principle of environmental flows is the flexible management of water resources in order to sustain ecosystems as well as people (Poff and Matthews, 2013).

A number of studies have attempted to quantify the level of flow regime alteration and river fragmentation internationally and globally, primarily focussing on the impact of large dams, reservoir operations, irrigation and inter-basin transfers (Dynesius and Nilsson, 1994; Nilsson *et al.*, 2005). Recent evidence suggests that around 65% of global river discharge, and the aquatic habitats this water supports are moderately to highly threatened and that the degradation of these habitats is increasing (Vörösmarty *et al.*, 2010).

Historical flow regulation and abstraction infrastructure is prevalent within the UK. According to the latest CIRIA (Construction Industry Research and Information Association) guide on weirs, the Environment Agency has records of over 13,000 weirs in England and Wales (Kitchen et al. 2016). In 2010, 16,725 weirs were identified in an Environment Agency report investigating hydropower opportunities in the UK (Environment Agency 2010.) In an island of approximately 200,000km of inland waterways (NRFA, 2019a) this represents one weir for every 11 km of river. Taking into account large impounding structures, which total around 2,700 (Environment Agency, 2019; SEPA, 2019), unregistered weirs or other instream-barriers and more than 36,000 km of flood control structures (Marsh *et al.*, 2000), the scale of the issue is further highlighted.

Large dams clearly disrupt natural hydrological variation due to their size and their huge storage capacity (Csiki and Rhoads, 2010). Additionally, the way in which low head dams and weirs impact upon the aquatic environment is also now better understood (Garcia De Leaniz, 2008). Research shows that even weirs less than 5m in height can severely impact upon sediment transport, flow, water temperature, stream habitat, biogeochemistry and the free movement of biota (Santucci *et al.*, 2005; Garcia De Leaniz, 2008). Furthermore, the sheer number and density of small and medium-sized structures leads to the severe fragmentation of river systems (Chin *et al.*, 2008).

As well as preventing migratory species reaching their spawning grounds, weirs can also alter feeding habits (Baumgartner, 2007) and prevent the recolonization of new river reaches by previously extirpated species (Beechie *et al.*, 2006). Weir removal has therefore become an attractive river restoration option to enhance the condition of rivers and streams by restoring the natural flow regime, removing barriers to migratory fish species and allowing the transfer of sediment (Gregory *et al.*, 2002; Hart *et al.*, 2002).

In recent years (2013 to 2017) the number of abstraction licenses issued by the Environment Agency in England has steadily fallen (DEFRA, 2019a). However, latest figures show that the volume of water abstracted from non-tidal sources in the England has steadily risen from an estimated 15,699 hm^3 to 16,302 hm^3 in the same period (DEFRA, 2019a). Over 80% of all

water abstracted from the freshwater environment comes from surface water sources (DEFRA, 2019a).

Domestic water supply is the major consumer accounting for just over half of all licensed non-tidal water abstractions (DEFRA, 2019a). Although domestic water abstraction has remained relatively static in recent years, uncertainty surrounding future climate variability and a growth in demand from other sectors is likely to place further strain on the management of water resources and the ecology they sustain.

A 2015 review of evidence for climate driven changes in UK rivers showed no strong decreases in low flows or summer flows, however, changes to winter flows, especially in the western uplands were evident (Hannaford, 2015). Evidence from 1969 – 2008 shows that winter Q_{95} flows in the north-west have decreased whilst overall run off has increased as a result of the greater range of flows experienced in winter months (Hannaford and Buys, 2012). A recent study of the Eden catchment (Cumbria) using UKCP18 projection data to model flows for 2040 – 2059 has also shown that Q_{95} flows are projected to decrease under all emission scenarios and Q_5 flows are predicted to increase under all scenarios (Kay *et al.*, 2020).

Over the last 50 years, the need to protect aquatic and riparian biodiversity from excessive abstractions and flow regime alteration has been increasingly recognised in environmental and conservation law. The UK Water Resources Act 1963 became one of the first pieces of environmental water legislation in the world requiring minimum acceptable flows (MAF) to be maintained in UK rivers in order to maintain fisheries and natural beauty (Acreman *et al.*, 2017). In 1996, the establishment of the Environment Agency led to the creation of Catchment Abstraction Management Strategies (CAMS) covering all of England and Wales (Acreman *et al.*, 2017). A Resource Assessment and Management (RAM) Framework tool was developed to aid the implementation of CAMS, which included the concept of hands-off flows (Acreman *et al.*, 2017). Under this system certain threshold values (river flows, water levels or salinity in aquifers) were assigned and this dictated when water abstraction must cease or become extremely limited (Petts *et al.*, 1999).

Since the year 2000, the European-wide Water Framework Direct has harmonized legislation for river basin management across the European Union (Acreman *et al.*, 2017). Under the WFD, water bodies were first assessed to evaluate and assign their ecological status as either High, Good, Moderate, Poor or Bad (Acreman and Ferguson, 2010). Based on this assessment, water bodies are then required to achieve High Ecological Status (HES), Good Ecological Status (GES) or in the case of the most heavily modified water bodies (HMWB),

Good Ecological Potential (GEP) (Acreman and Ferguson, 2010). In rivers were HES is to be achieved, flow regime is a primary element and required to be close to natural (Acreman and Ferguson, 2010). In order to achieve these objectives, targets are set through River Basin Management Plans (RBMPs) which contain programmes of measures for the 15 River Basin Districts (RBDs) within the UK (Robins *et al.*, 2017). Under the directive, river flow and form (hydromorphology) are recognised as important supporting elements for achieving GES (Acreman *et al.*, 2017). Member States are required to balance the need for water abstraction (domestic, industrial and agricultural) against the need to preserve river ecosystem health by either limiting abstractions and/or managing discharges from impoundments or other control structures (Acreman *et al.*, 2008).

Addressing the need to consider environmental flows for achieving GES the U.K. initiated two key projects, WFD 48 and WFD 82, making it the first European Country to do so (Acreman *et al.*, 2009). In WFD 48 the idea that reservoirs required 'active management' was introduced; the concept by which environmental protection is achieved through actively making releases from reservoirs (SNIFFER, 2006). As a result, WFD 82 was initiated which involved a larger research team to include experts in dam engineering and geomorphology. One of the main conclusions of the study found that it was not possible to apply a single generic rule to all U.K. rivers and that each water body downstream of a dam required individual analysis (Acreman *et al.*, 2009). The report also recommended that expansion or alteration of the existing hydroloigical monitoring network was required to define hydrological standards. It also called for further biological data and an assessment of the suitability of existing biological indices (SNIFFER, 2008).

In addition to the WFD, a further layer of legislation affects sites with wildlife designations. In the UK the main designation for wildlife conservation are Sites of Special Scientific Interest (SSSIs) (Mainstone *et al.*, 2012). In some cases, this is further strengthened through the designation of Special Areas of Conservation (SACs) under the European Union Habitats Directive (HD) (Mainstone *et al.*, 2012) or Habitats Regulations (HR) as translated into UK law. Unlike SSSIs, SACs are recognised as protected areas in the WFD (Mainstone *et al.*, 2012) and make up the European wide Natura 2000 network. Whilst both directives aim to protect aquatic ecosystems they do so in different ways. Species and habitats are protected under the HD and serve as indicators for the WFD (European Comission, 2011). In areas protected under both the WFD and the HD the 'most stringent requirement' of the environmental objectives in either directive must be applied (De Smedt and van Rijswick,

2014). Where protected species or habitats are at risk, the HD requires regulatory bodies to maintain or enhance the integrity of designated areas (European Comission, 2000).

The aim of the Habitats Directive is to conserve important habitats and species and whilst it makes no explicit reference to ecological flows, flow regime is a critical element controlling the conservation status of most aquatic ecosystems and their related protected habitats and species (European Commission, 2015). Therefore, both the HD and WFD are underpinned by supporting hydromorphological elements. Under the WFD, this includes 'quantity and dynamics of water flow' whilst under the HD relevant characteristics of the hydrological regime may form part of a site's conservation objectives (Schmedtje et al., 2011).

UK government policy currently states that where abstraction is at unsustainable levels for Natura 2000 sites, all necessary actions to rectify the situation will be taken regardless of financial implications (DEFRA, 2019b). Since 2008, the EA has prevented 230 billion litres of water per year being abstracted from unsustainable sources, with two thirds of this aimed at protecting Natura 2000 sites (DEFRA, 2019b).

Despite wide recognition from legislation, data availability (Hochkirch *et al.*, 2013) or a limited understanding of the complex links between freshwater ecology and hydrological processes often hinder the effective implementation of conservation management interventions (Richter *et al.*, 2003). Whilst dam removal may present an attractive ecological restoration option in many locations, legislative requirements, particularly in designated sites, add another dimension to the decision-making process. Furthermore, wider social, political and economic factors should all be taken into consideration to ensure a project's success; however, this can add greater complexity to an already challenging domain.

However, a number of recent initiatives have taken off with the aim addressing this challenge. The AMBER project (Adaptive Management of Barriers in European Rivers) funded by the European Union's Horizon 2020 research and innovation programme, is a European-wide initiative that "seeks to apply adaptive management to the operation of barriers in European rivers to achieve a more effective and efficient restoration of stream connectivity" (AMBER, 2021). The project has 22 active partnerships across, Government, Industry and academia working together to study the impacts of in-stream barriers and implement tools and solutions to address them (AMBER, 2021). Another related initiative is Dam Removal Europe, which has promoted or supported the removal of more than 100 in-stream barriers in Europed during 2020 alone (Dam Removal Europe, 2021).

In addition to direct flow regime alteration through abstraction, impoundment, controlled releases or flood control infrastructure, flow regimes have been anthropogenically altered in other ways. Some examples are mentioned here briefly but a detailed examination of these falls outside the scope of the present study, which instead is focused on impoundment and abstraction.

Land use change is one such example. This might occur as a result of agricultural activities or urbanization. Urbanisation can lead to an increase in impervious surfaces reducing baseflows and increasing the frequency and magnitude of storm flows (Walsh *et al.*, 2012). Additionally, urban development may be accompanied by the import and export of water through water supply and disposal, thus altering the catchment water balance and streamflow regimes (Stewardson *et al.*, 2017a).

The type and extent of vegetation cover can impact catchment water balance and flow regime. Deforestation may increase runoff (Andréassian, 2004), whilst different vegetation types are characterised by different rates of evapotranspiration and interception. The effect of vegetation change on flow regimes is varied but has been demonstrated through modelling studies (Brath *et al.*, 2006; Li *et al.*, 2009) and through paired catchment studies (Brown *et al.*, 2005).

Climate change has been shown to alter patterns of precipitation and impact evapotranspiration, leading to reduced surface water yields and groundwater recharge in subtropical regions (Stewardson et al., 2017a). Globally, streamflow variability and the threat of more frequent extreme floods continue to increase as a result of climate change (Arnell and Gosling, 2016). The Fifth Report from the Intergovernmental Panel On Climate Change (see Jiménez Cisneros *et al.*, 2015) provides a more extensive overview of observed hydrological changes due to climate change.

Case Study Description

The River Ehen Special Area of Conservation (SAC), which is the focus of this doctoral thesis, highlights the complexities surrounding abstraction and sustainable environmental water management. Located on the western limits of the Lake District National Park in North-West England (Figure 1), the River Ehen and Ennerdale Water have both been used for industrial and domestic water supply since the 1800s. The lake is known to have been impounded and regulated with engineered control structures since 1849 (Alvarez-Codesal and Sweeting, 2015).

The River Ehen, flowing from Ennerdale Water, received designated status under the HD in 2005. The primary factor for designation was the presence of freshwater pearl mussels (*Margaritifera margaritifera L.*). With an estimated population in excess of 500,000 individuals it is the largest population of freshwater mussels in England and Wales (Kileen and Moorkens, 2012). The Ehen is also considered to be *"one of the best areas in the United Kingdom"* (JNCC, 2016) for the species, making the site nationally and internationally significant. Atlantic salmon are also listed as an additional qualifying feature of the SAC. Mussels are one of the most threatened freshwater organisms and habitat degradation (Österling *et al.*, 2010), flow regime alteration (Gates *et al.*, 2015) and deteriorating substrate conditions (Geist and Auerswald, 2007) have all been cited as reasons for their decline.

Since its designation as an SAC, several attempts have been made to strike an appropriate balance between providing water for the local domestic supply network whilst maintaining sufficient river flows to avoid deterioration of the site.

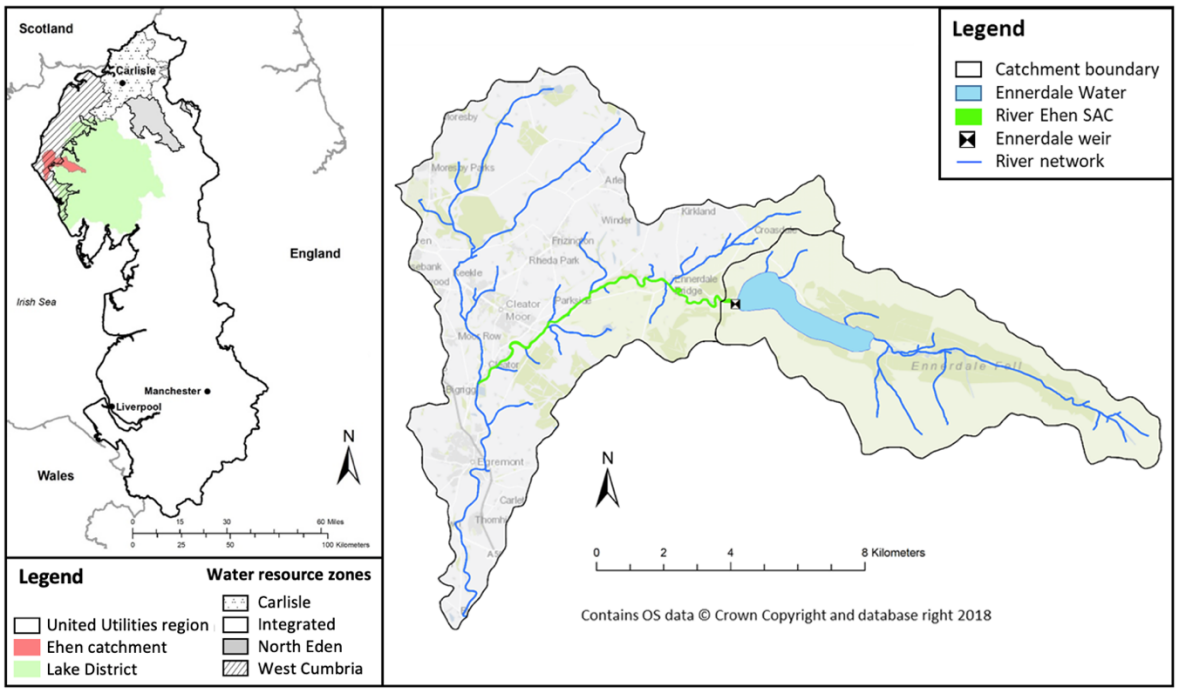


Figure 1. Maps showing United Utilities supply zone (left) and Ehen and Upper Ehen Catchments, Ennerdale Water and the River Ehen SAC (right).

United Utilities currently manage the abstraction, treatment and distribution of water from the lake to the domestic supply network. Ennerdale Water forms part of the West Cumbria Water Resource Zone (WRZ) (Figure 1) along with Crummock Water, Overwater and Chapel House reservoirs.

United Utilities' 2015 Water Resource Management Plan (WRMP), which is produced as a statutory obligation under Water Act, 2003, predicted that the West Cumbria WRZ would be

in deficit within the 25-year horizon of the plan (United Utilities, 2015). Subsequently, a number of options were then set out in the WRMP which aimed at tackling the shortfall in water supply.

Under the Habitats Directive, competent authorities must assess the potential impacts of any plans or programmes on the Natura 2000 network (Defra, 2012). Any 'likely significant effects' (LSE) are identified in order to categorise any adverse effects on a sites' integrity. These impacts are set against the conservation objectives of the designated site through a four stage Habitats Regulations Assessment (HRA).

Following this iterative process of WRMPs and stages of the HRA, in 2013, the decision was made that abstraction from Ennerdale Water should cease as 'soon as feasibly possible'. Alternative water supplies were identified, and the West Cumbria WRZ will now be incorporated into to the larger integrated WRZ through a transfer pipeline from Thirlmere reservoir (Figure 2).

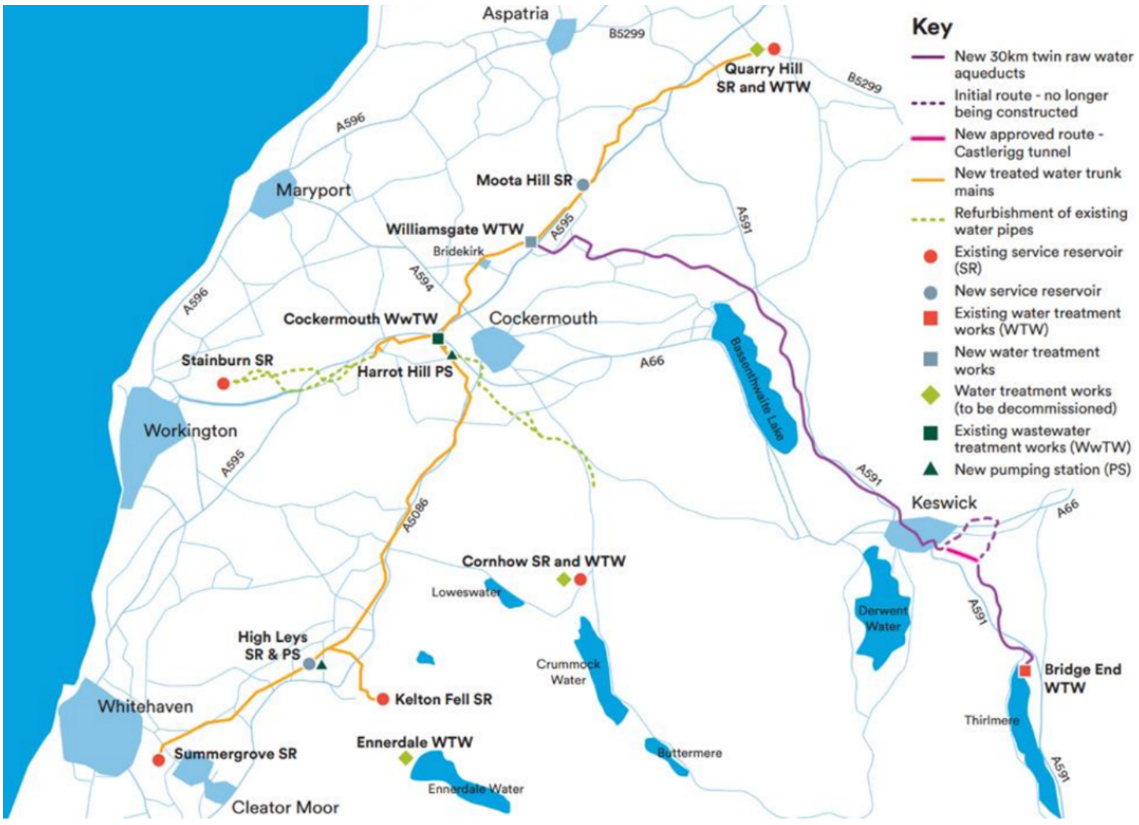


Figure 2. The Thirlmere pipeline and water supply network changes that will incorporate the West Cumbria Water Resource Zone into the Integrated Water Resource Zone (United Utilities, 2016).

As part of the Improvement Programme for England's Natura 2000 sites (IPENS), Natural England has produced Site Improvement Plans (SIPs) for each Natura 2000 site in England. The SIP for the River Ehen SAC (UK0030057) identified 11 priority issues and actions

including water abstraction, invasive species, agricultural practises, water pollution and siltation.

In response to these pressures, the same SIP lists a set of actions to be carried out to address them and enhance the integrity of the site. The first of these requires the restoration of the natural hydrology of Ennerdale Water and the River Ehen, through revoking the abstraction license, finding an alternative water source for public supply, reconnecting Ben Gill, a diverted tributary of the River Ehen and investigating the removal of the weir at the outflow of Ennerdale Water.

The first three actions are already underway or complete in the case of the reconnection of Ben Gill. The last of these actions (investigation into weir removal) also forms part of the set of compensatory measures established during the Review of Consents (RoC) process and is the principal driver for this doctoral thesis. Under the terms of the licensing agreement, United Utilities in part fund this research. The compensatory measures package is overseen by a project steering group (PSG) which is made up of United Utilities, Natural England, the Environment Agency, the National Trust and West Cumbria Rivers Trust.

Abstraction from Ennerdale Water will cease in 2022. However, decisions on the future of the impounding infrastructure must be based on a sound scientific understanding of the possible system responses and subsequent risk to the integrity of the SAC and freshwater mussel population.

Taking into account the factors contributing to the decline of *Margaritifera margaritifera L.,* and considering the legislative drivers*,* returning Ennerdale Water and the River Ehen to a more natural state appears to be a highly desirable management option. Despite this, due to the sensitivity of the species and its international significance, it is essential to assess how changes caused by the revocation of the abstraction licence and removal of the weir at Ennerdale might impact on the mussel population in the Ehen.

This research project represents a novel opportunity to investigate, in detail, the hydromorphological implications of weir removal at the site of the largest remaining freshwater mussel population in England. Wider research has shown that multiple benefits that weir removal can have on aquatic ecology (Garcia De Leaniz, 2008; Shaw *et al.*, 2010; Fjeldstad *et al.*, 2012; Birnie-Gauvin *et al.*, 2018). However, weir removal can also have unintended negative consequences, in particular the release of contaminants and fine sediments (Bednarek, 2001). Sediment deposition can clog substrates and reduce dissolved oxygen, with potentially fatal consequences for aquatic organisms including unhatched eggs

(Cheng and Granata, 2007). However, although these short terms risks are significant, they can be mitigated (Bednarek, 2001).

Despite the potential for ecological enhancement of rivers and their associated floodplains (Bednarek 2001; Doyle et al. 2005), not all schemes are successful in producing positive or desired ecological outcomes (Stanley and Doyle, 2003). Whilst removal is very effective at enhancing some aspects of riverine ecology it should be viewed as only one aspect of wider catchment-scale management (Stanford *et al.*, 1996). Truly effective weir removal projects will most likely need to be combined with additional conservation and restoration techniques (Hart *et al.*, 2002).

Ennerdale has been impounded and used for water abstraction for over 160 years so the shift from a regulated system to a naturalised flow regime will need to be carefully managed to ensure that the transition is as smooth as possible and that any potential undesirable impacts are mitigated. This thesis therefore focuses on understanding the natural hydrological regime of the Upper Ehen catchment and estimating the extent and timing of hydromorpholgical responses to revoking the abstraction license and removing the weir. Assessing the magnitude of these changes will help inform decisions on the future management of the site.



Figure 3. Photograph of the impounding weir and adjacent fish pass (left) at Ennerdale Water, taken in April 2018.

1.2 Research Problem

Despite becoming a more widely used river restoration approach in recent years, no standardised methodology for predicting hyrdromorphological responses to weir removal currently exists. Furthermore, weir removal projects in SACs are rare and thus guidance and detailed documentation of the procedures involved, and the hydro-ecological outcomes are scarce.

In many parts of the world, water resource managers are able to call upon historic data that provides insights into the hydrological conditions of a site prior to the construction of impounding infrastructure or before abstraction licenses were issued. At Ennerdale, as in much of the UK, the age of the infrastructure means that this data is not available. This study will therefore provide a methodology which can be replicated at similar sites where a lack of historic data hinders the decision-making process. Additionally, the characterisation of the UK's freshwater mussel rivers (Chapter 2.2) provides a previously unseen analysis of UK flow regime data which serves as a new reference point and basis for comparison.

Other approaches to a lack of historical or undisturbed discharge data rely on using proxy catchments or statistical analyses of hydrologically similar catchments that are free from anthropogenic influences. In the UK, this approach is limited due to the lack of near natural or undisturbed catchments. A methodology that is able to reconstruct the historic natural flow regime and predict the future conditions of the site in an otherwise unmanaged state is therefore required.

1.3 Novelty and Wider Implications

The flow regimes of the majority of the world's rivers have been altered. In light of abundant scientific evidence and increasing acceptance of the ecological consequences of this flow regime alteration, weir removal and abstraction reforms are increasingly common tools for the restoration of aquatic ecosystems. This work is unique in that it combines the disciplines of hydrology, geomorphology and ecology, domains which are often studied in isolation. It therefore provides a framework for evaluating flow regime alteration and understanding the hydromorphological implications of weir removal in ecologically sensitive sites and in sites where historic data is lacking.

The national-scale analysis of freshwater mussel rivers provides new insights into the physical and flow regime characteristics of the catchments sustaining the remaining UK populations of the species, serving as a reference point and comparison for ecologists and decision makers. Whilst only one of a number of key factors in the lifecycle of the species, flow regime is a driving force and interconnected factor within a number of other critical habitat conditions. This work therefore creates a pathway for more detailed investigations into areas such as near-bead hydraulic studies and sediment and nutrient transport.

As the site of the largest remaining freshwater mussel population in England and set within a strict regulatory framework, the research provides valuable insights into ecological restoration

practices in a unique setting. The additional site-specific data provides previously unseen evidence of the lake storage and discharge relationship allowing for greater certainty around predictions of future flow regime changes in an unmanaged system.

A new modelling tool is presented that allows the rapid assessment of a multiple natural lake outlet designs and responses to different inflow scenarios. This generates discharge, lake level and storage timeseries data for a given set of boundary conditions. Existing tools are computationally intensive, slow to run and require expert knowledge or training. This tool provides United Utilities with an easy to use means of rapidly testing multiple natural or engineered lake outlets and assessing their impacts on the future hydrology of the site.

Furthermore, the use of waterborne ground penetrating radar to survey lake sediments (**Chapter 5**) represents a UK first. This novel approach provides a non-intrusive method of gathering information on present day and historic sediment profiles within the lakebed which is necessary in an ecologically sensitive site. The methodology will thus serve as a reference point for the future deployment of the technique in similar settings and for a range of applications, such as assessing sediment infill rates in water supply reservoirs.

1.4 Aims, Objectives & Scope

1.4.1 *Aim*

The overall aim of this research is to estimate the natural hydrology of the Upper Ehen catchment and morphological responses to weir removal. This will support actions taken by United Utilities in fulfilling its legal obligations for the site.

1.4.2 *Objectives*

1) Characterise the flow regimes of UK freshwater pearl mussel rivers.
2) Estimate the natural flow regime of the Upper Ehen catchment and quantify the level of alteration across the full range of flows under the current managed hydrological regime.
3) Compare the current and future flow regime of the River Ehen with the flow regime characteristics of the UK's freshwater mussel rivers.
4) Characterise substrate properties of the outflow at Ennerdale Water, understand the natural geomorphology of the lake outlet and estimate geomorphic response to weir removal.

5) Quantify the sensitivity of lake storage and the downstream flow regime to changes in the morphology of the lake outflow.
6) Discuss the implications of the above findings for the future management of Ennerdale Water and the Upper Ehen catchment and in the wider context of ecological restoration of aquatic environments.

1.4.3 *Scope*

This multi-disciplinary Engineering Doctorate forms part of a suite of compensatory measures implemented under the requirements of the Habitats Directive. Its primary focus is on increasing understanding of the hydrological and geomorphological history of the site to complement a number of ongoing research projects and mitigation measures implemented at the site. These include a previous PhD (and subsequent publications) which examined sediment transport (Marteau, 2017), mussel surveys to assess early signs of how adult and juvenile populations are responding to physical and management changes at the site and water quality and benthic algal surveys.

This work does not intend to establish casual links between the hydrological and morphological regimes of the site and the current ecological status of the freshwater mussel population, although these are discussed. Rather, it aims to statistically analyse the past and present hydromorphological behaviours of the managed regime and predict and quantify changes to these under a set of future site conditions. This analysis will build on the existing knowledge base of this site and wider understanding of the optimum conditions for freshwater mussel ecology to assist the project steering group in determining the future management choices for Ennerdale Water and the River Ehen.

For the purposes of this study, the term ‘natural’ flow regime refers to a state by which flow has not been managed through impoundment, compensation flows or abstraction. Flow alterations that may have arisen as a result of land use changes or anthropogenic climate change, whilst discussed, are not the central focus of the study. The terms ‘natural flow regime’ and ‘unmanaged flow regime’ are thus used synonymously throughout the thesis.

1.5 Overall Methodology

Three numerical models form the basis of the methodology for this study. The selection criteria and rationale for their use are outlined in more detail in subsequent chapters (

Figure 5). These tools are supported by the identification of critical additional site-specific data and its subsequent acquisition through field experiments. How this additional data is employed and the connection between individual modelling tasks is visualised in Figure 4.

A national scale study of freshwater mussel rivers and their flow regimes characteristics sets this work in a wider context and forms a basis for comparison and discussion of the final results in Chapter 7.

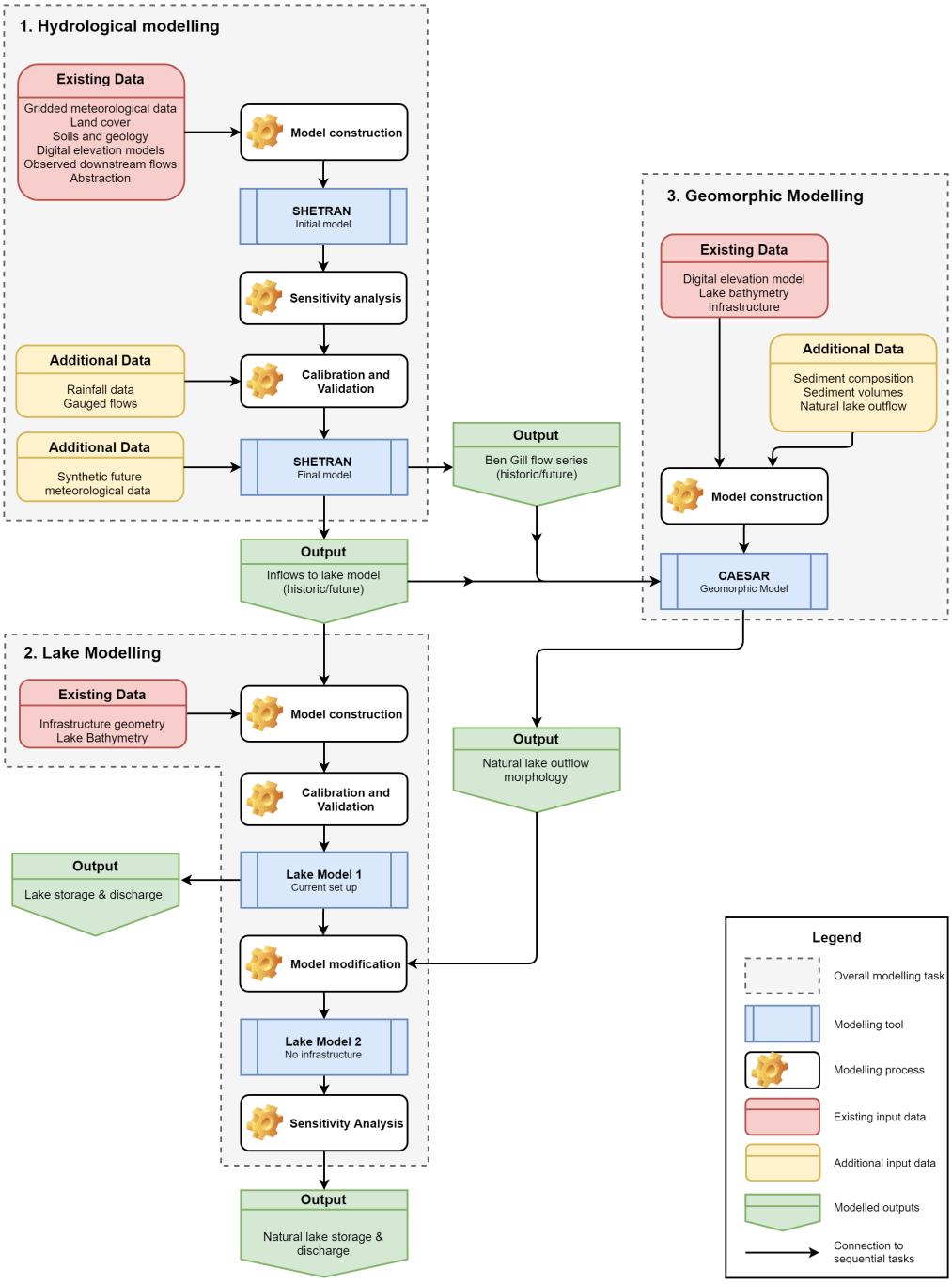


Figure 4. Overall workflow for modelling tasks and their connectivity. Hydrological modelling and lake model 1 are presented in Chapter 4. The geomorphic modelling is presented in chapter 5 before the outputs are brought together in the final modelling study presented in Chapter 6.

1.6 Thesis Structure

The overall thesis structure is outlined in Figure 5, however a brief description is also provided below to guide the reader through the document.

Chapter 2 sets the EngD thesis in context by providing a national-scale overview of freshwater mussel rivers and their flow regime characteristics. A literature review is carried out which highlights the causes and ecological impacts of flow regime alteration. The additional impacts of impounding structures are also reviewed. The ecology of the Freshwater mussel is described and linked to the ecological effects of flow regime alteration. The analysis in this chapter can be read as a stand-alone document but it also provides a basis for hydrological comparison in subsequent chapters.

Chapters, 3, 4 and 5 describe the data collection and individual modelling elements which form the building blocks for the final modelling chapter as presented in Figure 5. Literature reviews of the current modelling approaches for predicting natural flows and modelling geomorphic evolution are also contained within Chapter 4 and 5. Chapter 6 ties the three preceding chapters together by presenting a new lake modelling tool and detailed hydrological analysis. The implications of the findings and their context within the wider literature are then discussed in Chapter 7. A business case for the research is made with suggestions for how United Utilities can maximise the findings from this research and can continue to implement aspects of the work into their business strategy. Finally, the conclusions of the work are succinctly presented in Chapter 8 along with suggestions for further research.

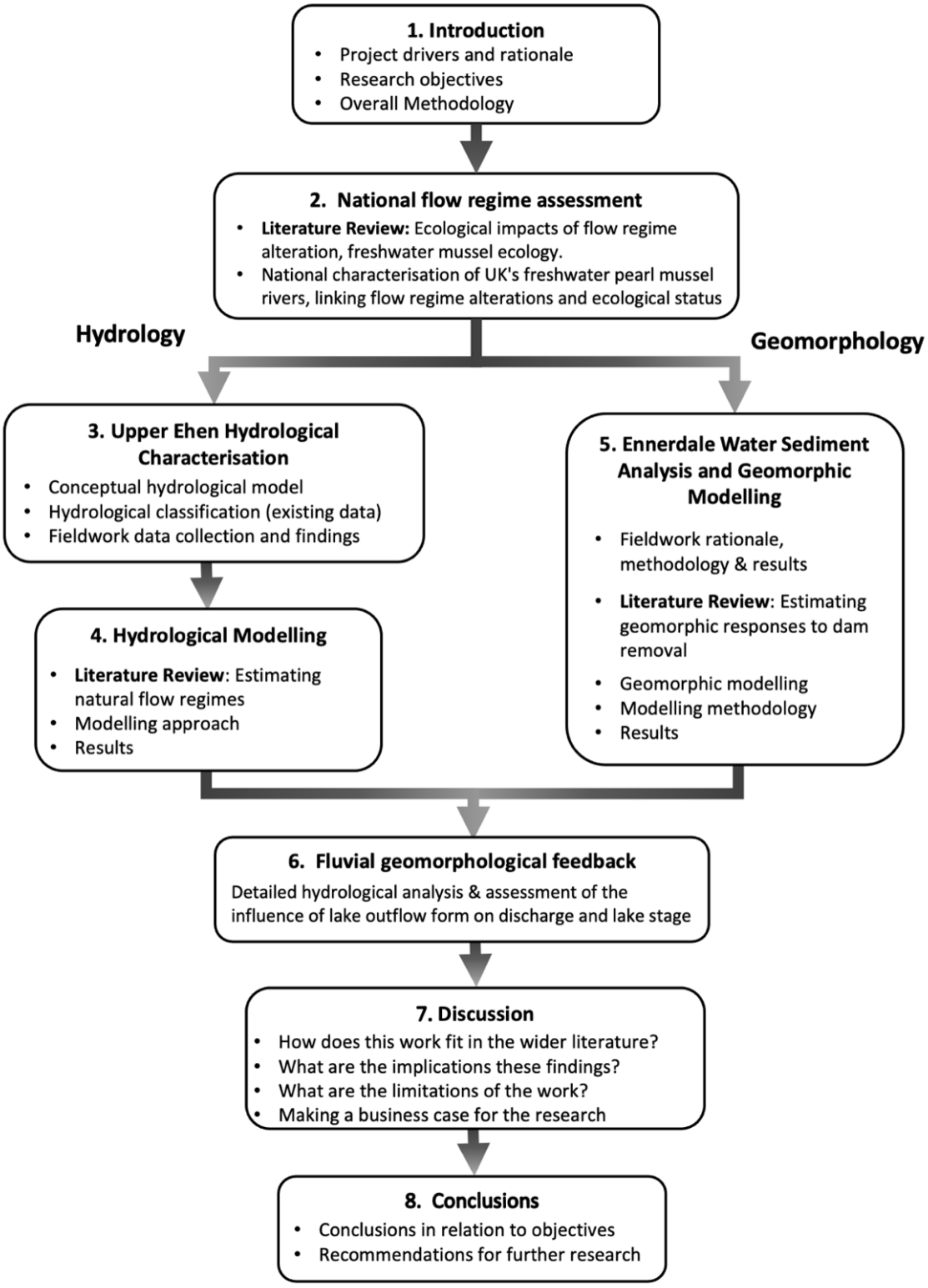


Figure 5. Thesis structure and overview of chapter contents.

Chapter 2 - Ecological Impacts of Flow Regime Alteration and Impounding Infrastructure

This chapter looks firstly at the broader ecological implications of anthropogenic alteration of flow regimes. This is then linked to the decline of freshwater mussels in the UK. An analysis of flow regime characteristics for the freshwater mussel rivers in the UK is then carried out. This chapter provides a basis for benchmarking the River Ehen in its current hydrological condition (Chapter 3) and under future flow regimes in an unmanaged system (Chapter 6).

2.1 Literature Review

2.1.1 *Ecological Impacts and Guiding Principles of Flow Regime Alteration*

On a broad scale, the principle defining controls of flow regime are rainfall, temperature and evaporation. At catchment scale, physical characteristics such as altitude, relief, geology and land cover exert further control on flows. River flows are also impacted directly and indirectly by human activities (Young *et al.*, 2003).

Freshwater habitats are characterised by environmental variability, much of which stems from the Earth's dynamic water cycle (Stewardson *et al.*, 2017b). This variability may be witnessed at a range of scales from localised storm events, to annual seasonal cycles or multiyear global climate events (Stewardson *et al.*, 2017b).

Freshwater environments cover approximately 0.8% of the Earth's surface but support a disproportionately high level of biodiversity (Dudgeon *et al.*, 2006) and are amongst the most productive and biologically diverse ecosystems in the world (WWF, 2020). However, the benefits that rivers provide require that they largely retain key characteristics and processes, such as connectivity and flow (WWF, 2020), yet anthropogenic flow modification is prevalent throughout the world (Nilsson *et al.*, 2005) and regarded by many as the most serious threat to freshwater ecology (Ward and Stanford, 1995; Bunn and Arthington, 2002; Richter *et al.*, 2003)

The latest Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) global assessment report on Biodiversity and Ecosystem Services analysed

more than 15,000 scientific publications. It's assessment of ecosystem extent found that inland waters and freshwater ecosystems show among the highest rates of decline and only 13 per cent of the wetland present in 1700 remained by 2000 (IPBES, 2019). Furthermore, recent estimates show that almost a third of freshwater species are threatened with extinction (WWF, 2020).

Natural hydrological variability is recognised as being essential for maintaining healthy freshwater ecosystems (Poff *et al.*, 1997). Furthermore, as the dominant abiotic control on aquatic environments, flow regime, has been described as a master variable for the ecological functioning of riverine ecosystems (Power *et al.*, 1995). Numerous mechanisms connect flow regime alteration and ecological degradation, often in a complex manner, meaning that our understanding of this field remains incomplete (Palmer and Ruhi, 2019).

A rivers' flow regime can be characterised by five key components (Figure 6); magnitude, timing, frequency, duration and rate of change (Richter *et al.*, 1996; Poff *et al.*, 1997). The magnitude of flow refers to the amount of water moving past a point at any given time (Poff *et al.*, 1997). In ecological terms, it can describe the availability or suitability of habitat as it controls attributes such as wetted area or water table conditions (Richter, 1996).

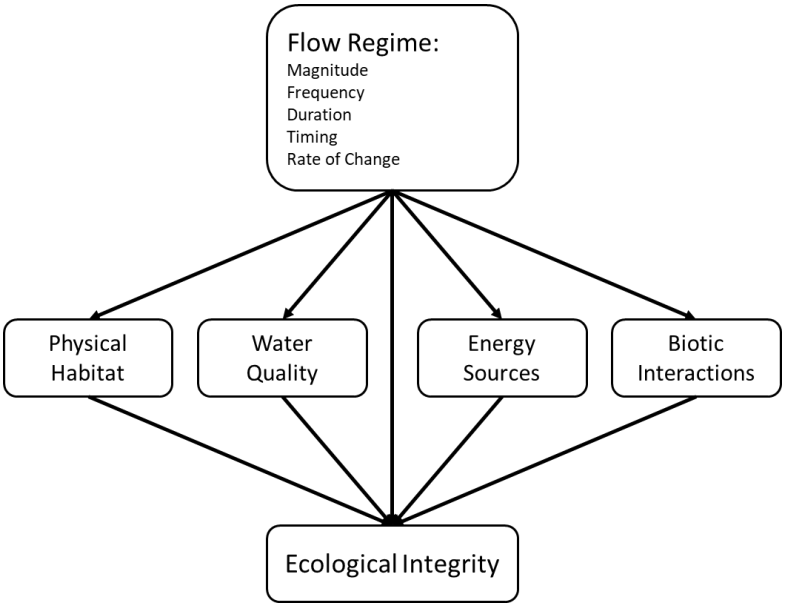


Figure 6.Direct and indirect influences of flow regime on ecological integrity (Poff *et al.*, 1997).

Frequency describes how often flow events above or below a given threshold occur (Poff *et al.*, 1997). The frequency of events such as floods or droughts can be linked to stressful or fatal ecological consequences for species that suffer from inundation or desiccation (Richter *et al.*, 1996).

Timing or predictability is a means of describing the regularity with which certain flow events occur (Poff *et al.*, 1997). This is linked to critical stages in the lifecycles of many aquatic organisms that require specific conditions at certain times and can influence stress and mortality rates (Richter *et al.*, 1996).

Duration refers to the length of time a certain flow condition is observed for. This can be defined for individual flow events or as a the number of days over a year in which a certain flow condition is observed (Poff *et al.*, 1997). This will determine the length of time aquatic organisms are exposed to conditions of stress (Richter *et al.*, 1996).

Finally, the rate of change describes how quickly flow conditions can change and may be referred to as flashiness or stability (Poff *et al.*, 1997). In flow regimes with high rates of change, organisms may become stranded by rapidly decreasing water levels or unable to colonise new areas due to constantly changing conditions (Richter, 1996).

Bunn and Arthington (2002), proposed four guiding principles that demonstrate how flow regime alteration impacts aquatic biodiversity. Firstly, flow has a major impact on the physical habitats of rivers and streams, which subsequently plays a key role in determining their biotic composition (Bunn and Arthington, 2002).

Physical changes to river channels occur naturally and are not always negative (Vietz and Finlayson, 2017). Rivers and streams are constantly adjusting their morphology according to the hydrological conditions and sediment loads that they are subjected to (Nicholas *et al.*, 1999). However the rate and effect of these physical changes if brought about by human induced flow regime alterations can be of ecological concern (Vietz and Finlayson, 2017). An example of this is bank erosion. Bank morphology and evolution is strongly related to streamflow and can have a major impact on river ecosystems (Vietz *et al.*, 2018). Subsequently this natural geomorphic process is often controlled or managed for conservation purposes. However, bank erosion aids the succession of riparian vegetation and provides dynamic habitats that are essential for a number of aquatic and riparian plants and animals (Florsheim *et al.*, 2008).

The second of the fundamental principles outlined by Bunn and Arthington (2002) states that aquatic species and their survival strategies have primarily evolved as direct responses to the natural flow regimes they experience. Through evolutionary processes, aquatic organisms develop traits that allow them to survive, exploit and possibly depend on the natural disturbances they experience in their environments (Lytle and Poff, 2004).

An example of this adaptation and response to flow regime variability can be observed in the lifecycles of numerous fish species, many of which have evolved reproductive cycles to coincide with optimum conditions for the survival and recruitment of their young (Beechie *et al.*, 2006). Low flow conditions provide stable river bed conditions for long periods and thus trigger spawning in some species (Humphries *et al.*, 1999) , whereas flood pulses can trigger upstream movement and spawning in others (Jager and Rose, 2003). Flooding can also provide access to important juvenile rearing habitat (Milton and Arthington, 1984) and aid the introduction of woody debris providing important micro-habitat (Crook and Robertson, 1999). Disruption to these natural disturbances and variation can thus negatively affect the lifecycles of a wide range of species.

The third principle described in Bunn and Arthington (2002) is the maintenance of longitudinal and lateral connectivity. This connectivity refers primarily to the exchange of water, resources and organisms between the river channel, its floodplain and aquifer (Ward and Stanford, 1995). Variation in these hydrological processes at spatial and temporal scales contributes to heterogeneity of riverine landscapes and thus greater biocomplexity (Amoros and Bornette, 2002).

Connectivity may by longitudinal, i.e. along the length of the river corridor, and the success of many aquatic species hinges heavily on having free movement through the stream or river network (Bunn and Arthington, 2002). Lateral connectivity refers to the connection between the river channel and the surrounding river corridor, floodplain, and wider catchment. This exchange of matter and organisms between the river and its floodplain is largely controlled by the pulsing of river discharge (Tockner *et al.*, 2000).

The large-scale impact of reduced connectivity and the subsequent fragmentation as a result of flow regime alteration has been shown to greatly impact on community structure and distribution of various species of fish (Perkin *et al.*, 2015). Other studies have demonstrated its influence on the distribution and abundance of riparian vegetation (Merritt and Cooper, 2000; Shafroth *et al.*, 2002). Nielsen *et al.,* (2013) also provided empirical evidence that hydrological stability reduced biotic diversity when studying the management of numerous wetlands in the Murrumbidgee River Floodplain, south-eastern Australia.

A third type of connectivity, which may be impacted on by anthropogenic alteration to flow regimes, is vertical connectivity, however this is less commonly considered. (Kondolf *et al.*, 2006). Water can flow into and out of stream beds depending on the hydraulic gradients present and this movement can provide important ecological functions such as supplying dissolved oxygen and nutrients to organisms in the hyporheic zone (Kondolf *et al.*, 2006).

The final principle set out by Bunn and Arthington (Bunn and Arthington, 2002) relates to the success and spread of invasive and non-native species. One such way this can manifest itself is through a reduction in the magnitude and frequency of floodplain inundation (Rolls and Bond, 2017) and by creating conditions that favour invasive species or reduce competition from native species (Catford *et al.*, 2011).

Mortensen *et al.,* (2012) showed how flow regime alteration allowed for the widespread colonisation of *Tamarix* which had previously been limited by scouring from large scale floods prior to the construction of the Glen Canyon Dam. Whilst Catford *et al.,* (2011) identified a link between hydrological alteration, declining native amphibious vegetation and increasing non-native terrestrial weed cover.

In the last 20-25 years since the seminal works of (Power *et al.*, 1995; Poff *et al.*, 1997; Ward, 1998; Bunn and Arthington, 2002), the number of papers describing the ecological responses to flow regime alterations has grown considerably.

In 2010, Poff and Zimmerman, carried out a review into the subject, examining over 165 papers published in the preceding four decades but with a focus on more recent studies (Poff and Zimmerman, 2010). The papers analysed, covered a wide range of species (fish, macroinvertebrates, riparian and aquatic vegetation, birds and amphibians) and geographical areas although this was heavily weighted toward North American and European studies with just 7 papers coming from South America and Africa (Poff and Zimmerman, 2010). The findings showed that in 92% of the studies, ecological metrics had decreased in response to flow alteration, although increased values were also reported in 13% of the papers (Poff and Zimmerman, 2010).

More recently, with increased data availability and sophisticated modelling tools, research is beginning to look at the mechanisms that cause flow regime alterations to affect biota and ecosystem processes and the interactions between them; the flow-biota-ecosystem processes nexus (Palmer and Ruhi, 2019). Rolls et al., (2012) proposed four underlying principles behind mechanistic principles of low flows and the processes and patterns of riverine ecosystems. They concluded that these principles do not act in isolation and that impact of low-flows can affect overlapping and simultaneous pathways, with the potential for synergistic and complex effects (Rolls *et al.*, 2012). Greater understanding of the complex interaction between process and pathways thus has significant implications for the management and restoration of riverine ecosystems and remains a frontier research topic (Palmer and Ruhi, 2019).

2.1.2 *Additional Ecological Impacts of Weirs*

In addition to altering flow regimes, weirs (often referred to as low-head dams in the literature) and other impounding or regulatory infrastructure can have a number of important additional ecological consequences. Some of these may affect riverine ecosystems in a similar way to flow regime alteration however, the mechanisms behind them are different.

Historically, scientific research has focused on the impact of large dams. However, weirs and low-head dams although smaller in size and storage capacity can still have a profound impact on freshwater ecosystems by degrading habitat and water quality and fragmenting river ecosystems (Santucci *et al.*, 2005).

The impacts of weirs and impounding infrastructure can be observed both upstream and downstream of their location. To demonstrate this, Mueller *et al.,* (2011) conducted a comprehensive study on the impact of weirs on both physical habitat and communities of periphyton, macrophytes, macroinvertebrates and fish in the upstream and downstream reaches of five German rivers.

A number of the principle ecological impacts of weirs are represented visually in Figure 7. These are then described in more detail with supporting literature focussing on weirs and low-head dams rather than large-scale structures, although many impacts are synonymous with both types of structures.

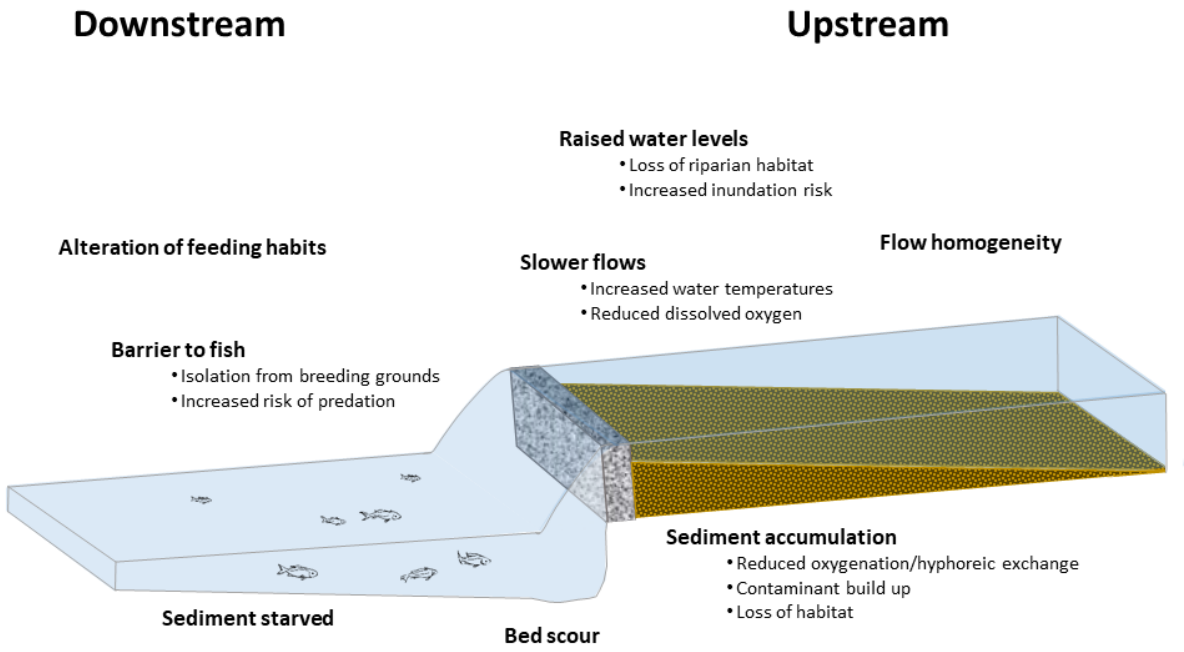


Figure 7. Illustrative examples of ecological and physical impacts of impounding structures in rivers and lakes.

Upstream of weirs, pools of deeper, slower flowing water are often created. This can result in changes to the distribution and abundance of aquatic organisms as the changing habitat provides favourable conditions for different species (Shaw *et al.*, 2010). Poulet (2007) demonstrated this by looking at fish communities in a US stream, showing that species richness increased downstream of the weir whilst there was a prevalence of non-native eurytopic or limnophilic species in the upstream study site.

Loot *et al.*, (2007) found that parasite diversity increased upstream of weirs and that the altered environmental conditions created by weirs could present an ecological risk. They also state that weirs may induce physiological stress in host species thus increasing their susceptibility to infection (Loot *et al.*, 2007).

In some cases, weirs and dams can cause permanent or increased periods of inundation in upstream riparian areas, leading to loss of habitat and shifts in riparian vegetation composition (Wollny *et al.*, 2019).

The slow flows and physical barrier of the weir itself can cause the accumulation of fine sediment and organic matter upstream of weirs (Shaw *et al.*, 2010). In gravel bed rivers this accumulation can limit oxygenation of the hyphoreic zone and deteriorate suitable habitat for the eggs of salmonid species (Soulsby *et al.*, 2001).

Conversely, as sediment is stored upstream of weirs, downstream reaches may be deprived of substrates and nutrients which can have substantial impacts on physical habitat and aquatic biota (Santucci *et al.*, 2005).

Longitudinal connectivity of river systems can be greatly impaired by the presence of weirs and low-head dams. These structures can act as a physical barrier preventing or reducing the probability of fish reaching their spawning grounds, with river and sea lamprey (Harvey *et al.*, 2010; Russon *et al.*, 2011), Atlantic Salmon and European eel (White and Knight, 1997) amongst those species heavily affected in British rivers. Additionally, the movement of fish upstream may be delayed and subsequently migratory species such as salmon may be more exposed to the risk of predation as they congregate below weirs (Garcia De Leaniz, 2008). Research has also shown that fish movement may be impaired in both up and downstream directions at different stages of their life history (O'Connor *et al.*, 2006; Gauld *et al.*, 2013). Furthermore, these barriers have also been shown to alter the diet and feeding habits of certain fish species and lead to reduced abundance downstream of these structures (Baumgartner, 2007).

In light of the ecological consequences outlined, weir removal is often viewed as an attractive river restoration option as it can alleviate a number of flow-related and physically degraded habitat issues. Feld et al. (2011) carried out a review of 31 restoration studies (and 5 general ecological papers) examining the effects of small dam and weir removal. Nearly all of the studies were carried out in North America with only one restoration study and two review papers originating from Europe (Feld et al., 2011).

The findings from this review showed that removing small weirs helps to restore a river's natural physical characteristics and can have positive effects on the instream fauna. Yet the authors also highlight the need for an integrated assessment of the impact of weir removal as the majority of the studies focused on abiotic factors or biological effects in isolation and rarely combined both (Feld et al., 2011).

2.1.3 *Freshwater Mussel Conservation Status, Lifecycle & Habitat Requirements*

Freshwater pearl mussels (*Margaritifera margaritifera L.*) are found in cool, oligotrophic rivers and streams of the Holarctic region (Geist, 2010; Lopes-Lima et al., 2017). Until the mid-19th Century the species was present in large numbers throughout its range (Geist and Kuehn, 2005) extending from Arctic regions of Scandinavia and western Russia to the north-eastern coast of North America (Skinner, Young and Hastie, 2003) and as far south as Spain and Portugal (Reis, 2003).

The decline of the species throughout Europe, has been well documented since the eighties (Young and Williams, 1983; Bauer, 1988). Complete extinction has also been observed in parts of Central Europe (Buddensiek, 1995). Populations of FPMs are estimated to have fallen by as much as 95% in Europe (Degerman *et al.*, 2009) and FPMs are now extinct from 80% of England and Wales (Kileen and Moorkens, 2012). In 2001 there were potentially as few as 100 viable populations globally (Cosgrove and Hastie, 2001) and extinction is a serious threat unless positive steps are taken to ensure their survival**.** Their status is currently listed as endangered on the IUCN Red List of Threatened Species (Moorkens *et al.*, 2018).

FPMs are of high conservation value and protected under the European-wide Habitats Directive. Their presence is an indicator of the health of an aquatic ecosystem. Freshwater mussels can simultaneously be seen as indictor, flagship, keystone and umbrella species making them important target species for the conservation of aquatic ecosystems (Geist, 2010). Scotland remains a stronghold for the species in the UK and whilst FPMs can be found

in a small selection of rivers in England and Wales, only the River Ehen holds a currently viable population (Kileen and Moorkens, 2012).

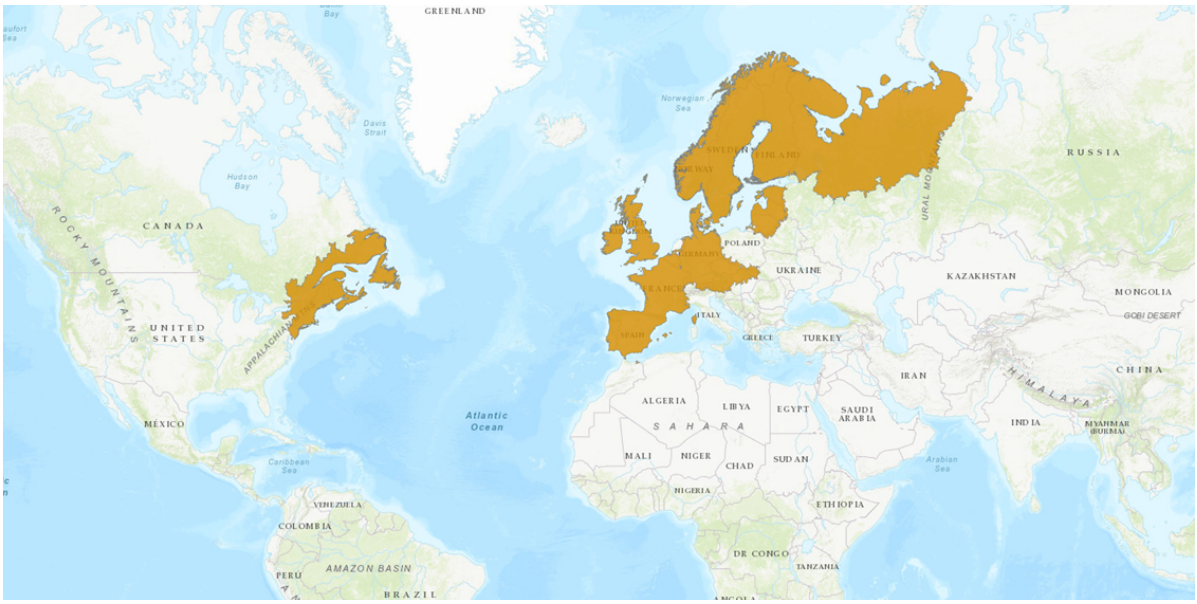


Figure 8. Global distribution of freshwater pearl mussels. Populations are limited to small geographical pockets in most European countries (Nature Serve *et al.*, 2019).

FPMs are long-lived, with individuals capable of surviving in excess of 100 years with individuals in populations in cooler Scandinavian climates living for over 200 years (Mutvei and Westermark, 2001). The species undergoes a complex lifecycle and requires different habitat characteristics at different stages of their development (Gumpinger *et al.*, 2015). They also have a parasitic stage which requires a salmonid host species; sea trout, brown trout or Atlantic Salmon in Europe (Skinner *et al.*, 2003).

Adults mature at 10-15 years, when they reach lengths typically in excess of 65mm. Sperm, released into the water by males in June and July, is inhaled by the females. Fertilised eggs are released from July to September as larvae (glochidia) measuring 0.6 - 0.7 mm (Skinner *et al.*, 2003). Glochidia are released by the females in a synchronised event, with each females capable of releasing between 1 and 4 million larvae (Hastie and Young, 2001).

Upon release, the glochidia have a short planktonic stage as they drift downstream where they must be passively inhaled or ingested by a host fish (Hastie and Young, 2001). During the subsequent parasitic stage, glochidia live on highly oxygenated gills of the host species (Cosgrove *et al.*, 2016). This stage lasts several months and does not appear to harm the host (Hastie and Cosgrove, 2001). However, more recent research on captive trout has shown that whilst this relationship has little or no impact on the haematological condition of the host, it can affect respiratory performance (Thomas *et al.*, 2013).

Glochidia drop off from the gills of host fish in May and early July and need to settle on clean gravelly or sandy substrate where they can begin to grow (Skinner *et al.*, 2003)**.** During this juvenile development, the mussels tend to bury themselves completely in the loose substrates of the river bed (Hastie *et al.*, 2000b). Well oxygenated and silt-free conditions are thus critical for this stage in the mussel life-cycle (Skinner *et al.*, 2003).

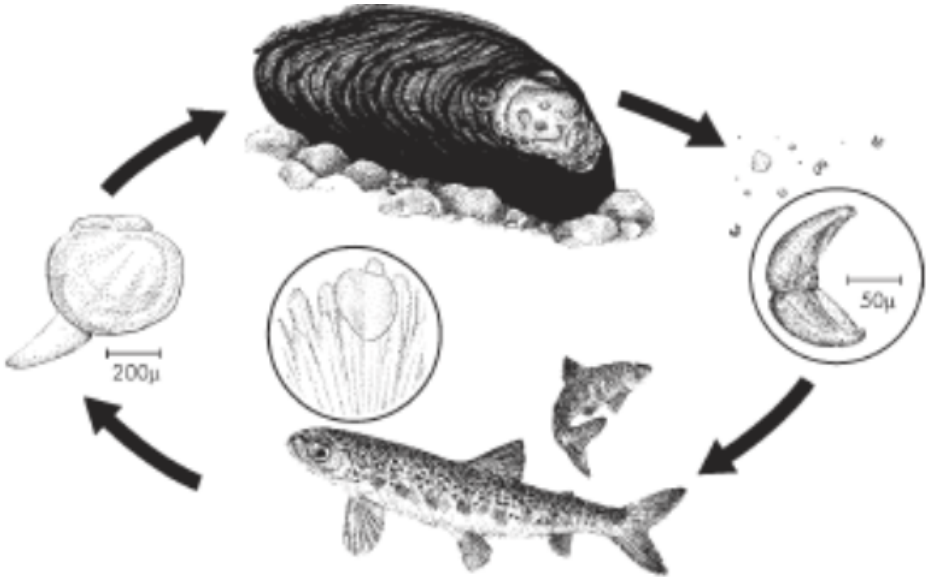


Figure 9. Main lifecycle stages of Margaritifera margaritifera L. (Skinner *et al.*, 2003)

FPMs are sensitive to changes in water chemistry, especially during the juvenile stage of their development (Skinner *et al.*, 2003). Much of the scientific literature on the water quality preferences of the species cites the work of Bauer, 1988. It is widely accepted that FPMs have a preference for nutrient poor rivers, with low conductivity and pH 7.5 or lower (Skinner *et al.*, 2003). More recently, Taskinen *et al.*, (2011) exposed juvenile mussels and glochidia to decreasing pH and increasing levels of iron (Fe) and aluminium (Al) to assess the tolerance of the species. In their test, 24 hours of exposure to pH 4.5 resulted in 100% mortality of glochidia (Taskinen *et al.*, 2011)**.** Other critical water quality parameter include biochemical oxygen demand (BOD), calcium and phosphate levels and nitrate concentrations (Skinner *et al.*, 2003).

Water depth and velocity are other habitat requirements that have been examined. However characteristics of preferential water depth and velocity remains an area for further research (Quinlan *et al.*, 2015a). Mussels frequently inhabit stretches of river with water depths between 0.3 – 0.4m (Hastie *et al.*, 2000a) however mussels have been recorded at depths between 3 – 6m in Scandinavia (Degerman *et al.*, 2009). Flow velocities between 0.25 – 0.75

m/s are believed to be optimum (Hastie *et al.*, 2000a) however this research did not distinguish between different mussel age groups (Quinlan *et al.*, 2015a)**.**

2.1.4 *Reasons for the Decline of the Species*

In the last 30 to 40 years, a variety of reasons for the decline of FPMs have been identified. A number of the reasons for the decline or the mechanisms behind them can be linked directly and indirectly to anthropogenic alterations of flow regimes and impoundment (Figure 10).

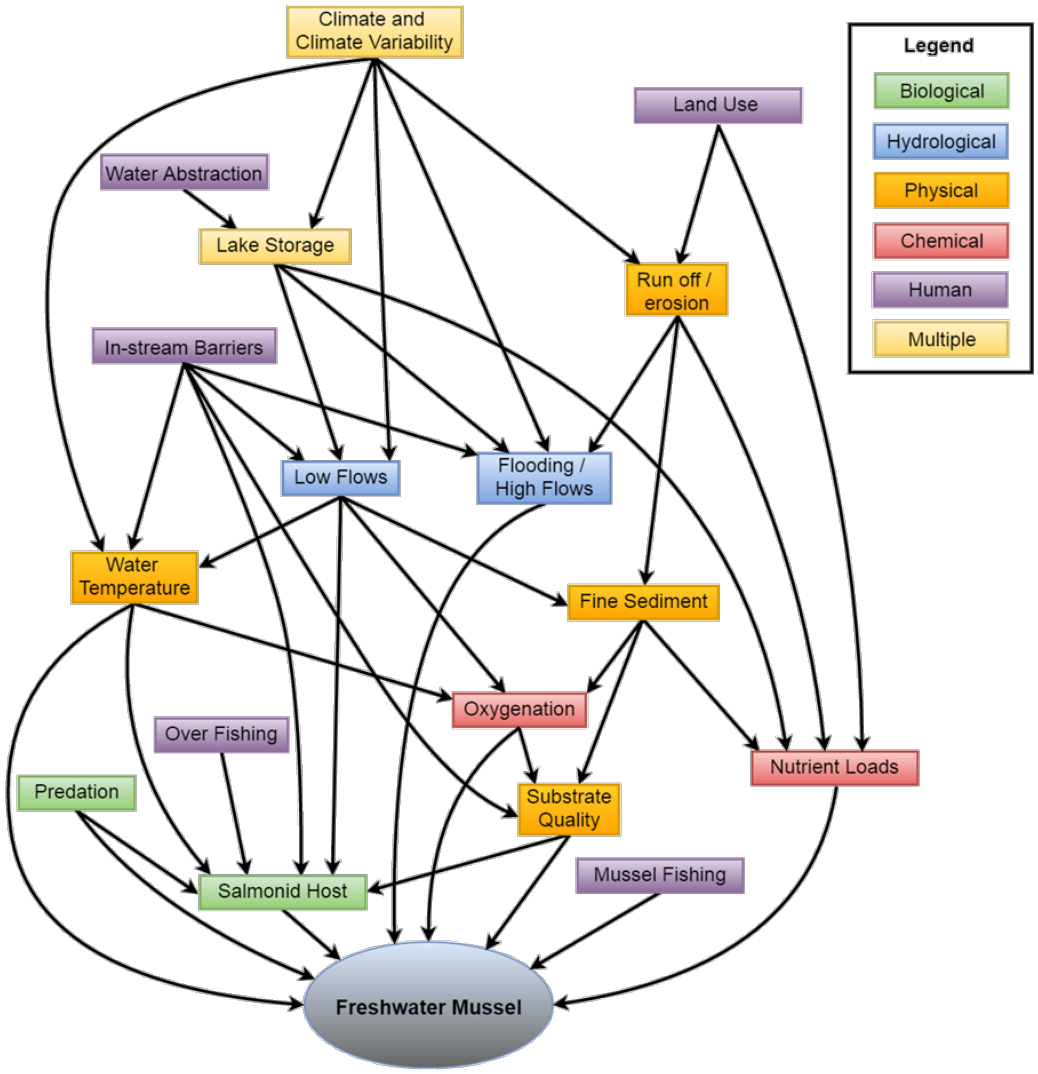


Figure 10. Relationship between factors affecting the lifecycle and abundance of Margaritifera margaritifera L. based on findings from peer-reviewed literature.

Streambed characteristics are critical factors for FPMs particularly in the juvenile stages of development. These conditions are largely controlled by a rivers' flow regime, which influences the rates of erosion, sediment mobilisation and deposition (Geist and Auerswald, 2007).

FPMs have specific habitat requirements and slight deviations from these conditions can dramatically reduce the viability of populations. In the last 15 – 20 years, a growing body of research has focused on the role of substrate quality and sedimentation on the decline of FPM. The 50th edition of the Journal Limnologica (Gumpinger *et al.*, 2015) was dedicated solely to FPM research. Five of the eleven articles draw special attention to the influence of fine sediment on freshwater mussel habitat degradation (Gumpinger *et al.*, 2015). Quinlan *et al*, (2015a) also highlight prior research into the importance of fine sediment, suspended sediment and substrate stability in the survival of the species in their review of physical habitat requirements for the species.

Research looking at 24 Swedish populations of the species found that juvenile mussel density was negatively correlated to sedimentation and turbidity and that these factors were responsible for recruitment failure in over half of the populations examined (Österling *et al.*, 2010). Fine sediment deposition is believed to limit oxygenation of the hyphoreic zone.

Sediment conditions are also known to affect other freshwater mussel species. Doyle *et al*. (2005) cite the total loss of the rare species, *Quadrula pustulosa*, in their Wisconsin case study, citing the transportation and deposition of sediment mobilised after weir removal, as having contributed to mussel mortality.

Research into other mussel species has also highlighted the impact of impoundments. Vaughn and Taylor (1999) identified a clear extinction gradient downstream of impoundments. With increasing distance from a main stem reservoir, species richness and abundance increased linearly and the rarest species of mussels, were only found in sites furthest from impoundments (Vaughn and Taylor, 1999).

Due to its dependence on a host species during glochidial development, the success of FPM populations may be affected by changes to salmonid stocks. Earlier research into the decline of the species suggests that changes to salmonid stocks has not played a major role in the overall decline of FPM in the last 50 to 100 years (Young and Williams, 1983; Young, 1991)**.** However, other studies show that, for some FPM populations, host fish density can be a limiting factor (Hastie and Cosgrove, 2001). As discussed in section 2.1.2, salmonid stocks are known to be negatively impacted upon by weirs and flow regime alteration, thus affecting the reproductive cycle of FPMs.

Several studies have been undertaken to identify possible causes for the decline of *Margaritifera margaritifera L.* and its salmonid hosts within the River Ehen SAC (Gibbins *et al.*, 2006; Killeen and Moorkens, 2012; Atkins and Gosselin, 2013; O'Leary and West

Cumbria Rivers Trust, 2013; Alvarez-Codesal and Sweeting, 2015; Quinlan *et al.*, 2015b). The weir along with the associated water abstraction and the compensation flow management have been highlighted as direct and indirect causes for the observed decline in juvenile recruitment and total mussel numbers.

Gibbins *et al*. (2006) suggest that the compensation flows put in place have resulted in the predominance of water velocities considered to be sub-optimum for freshwater mussels whilst Kileen and Moorkens (Killeen and Moorkens, 2012) suggest that there is substantial evidence showing that low flows are the principal cause of poor juvenile recruitment rates. In contrast, impoundment of the lake has reduced high flows and created a more stable environment (Gibbins *et al.*, 2006), however flow variability is important for substrate cleansing and oxygenation.

Additional research points towards the fact that populations of migratory salmon, Arctic charr and brown trout (potential host species) have been severely affected by the existence of the weir at Ennerdale and flow regulation structures within the catchment during a period of 160 years (Alvarez-Codesal and Sweeting, 2015). Despite the existence of a fish pass at the outflow of Ennerdale Water its inadequate design and the presence of multiple in stream barriers has impeded fish passage and the subsequent dispersal of mussel glochidia (Quinlan *et al.*, 2015b).

Although there are no data for sediment transport in the Ehen prior to construction of the weir at Ennerdale Water (Quinlan *et al.*, 2015b) it is possible that its presence has had a limiting role in the supply of substrate downstream for over 150 years. The recent reconnection of the ephemeral tributary Ben Gill[1] to the River Ehen has been carried out to restore the supply of sediment downstream and improve mussel habitat, and in particular provide a supply of gravels and cobbles, which are essential for mussels, but also provide salmon spawning habitat. To assess the impact of the recently reconnected (2014) Ben Gill, ongoing monitoring is being conducted at Aberdeen University. This is a continuation of studies carried out in the by Marteau (2017) that complement this EngD and together form part of the suite of compensatory measures for the River Ehen.

[1] Ben Gill is a tributary of the River Ehen that connects to the main river immediately downstream of the weir however in the 1970s it was diverted to flow directly into Ennerdale Water as a means of augmenting the water levels in the lake. In 2014 the diversion was removed and a new channel was created reconnecting Ben Gill to the River Ehen, just downstream of the outflow of the River Ehen from Ennerdale Water.

2.1.5 *Other causes for the decline*

One of the earliest known causes for the decline of the species is the direct impact of pearl fishing. This activity is documented to have taken place in Britain since Roman times (Young and Williams, 1983). Although pearl fishing has greatly diminished, this illegal activity continues despite protective conservation legislation and a ban on the activity in 1998 (Hastie, 2006). In Scotland, the activity is believed to have negatively impacted upon 99% of all known FPM populations (Cosgrove *et al.*, 2000).

Eutrophication caused by organic pollution has been shown to increase mussel mortality and inhibit the growth of juveniles leading to reduced mussel abundance (Bauer, 1988). Heavy metals, acidification, oxygen deficiency are also detrimental to the species (Ziuganov *et al.*, 1994).

Mussels are also a source of food for certain species. Although not documented as a major factor in contributing to the decline of the species in the UK, examples of boars scavenging on them during low flow events have been documented in Southern Europe (Sousa *et al.*, 2018).

Despite their requirement of relatively specific habitat conditions, FPMs also show resilient traits. They have been recorded in water temperatures ranging from 4°C to 23°C and tolerating temperature up to 28°C for very short periods (Ziuganov *et al.*, 1994). They have also been known to survive out of water for over 30 days albeit in shady conditions and in temperatures below 15°C (Ziuganov *et al.*, 1994).

2.2 Characterising Flow Regime Components of Viable Freshwater Mussel Populations

In line with the focus of this doctoral thesis, the following study aims to identify which flow regime components and catchment characteristics are associated freshwater mussel populations in UK rivers, to help inform future conservation strategies, particularly those related to catchment hydrology (e.g. flow regime management and abstraction reform).

Flow regime is known to both directly and indirectly influence mussel populations (Gibbins *et al.*, 2006). Under low summer flow conditions, algal growth can smother mussel beds. Additionally, interstitial-water mixing may be reduced (Skinner *et al.*, 2003). Water flow through the substrate is likely to be a critical factor for juvenile mussel recruitment, providing

food (Degerman *et al.*, 2013) and playing a critical role in cleansing and oxygenating substrate (Scheder *et al.*, 2015).

Autumn spates can remove algal growth and sediments that may have accumulated over the summer (Skinner *et al.*, 2003), however, extreme high flow events can also lead to mussel mortality (Hastie *et al.*, 2001) and habitat degradation. Heavily regulated rivers frequently switching between high and low flows are of particular concern in rivers supporting freshwater mussel populations (Skinner *et al.*, 2003). Thus, the large-scale impoundment of rivers has been a major factor in the decline of many freshwater mussel species (Vaughn and Taylor, 1999).

A relatively small body of research into the optimum flow conditions for a selection of freshwater mussel species has been carried out in North America. Gates et al., (2015) used biological traits of species guilds to make environmental flow recommendations for 10 native freshwater mussel species. Additionally, Ries et al., (2015) studied the relationship between river discharge and variation in recruitment of 18 species of mussels in a tributary of the Mississippi river. Whilst neither study included *Margaritifera margaritifera L.*, the approaches used offer valuable insights into the study of freshwater mussels and methodologies that could be transferable.

July maximum discharge and April minimum discharges were shown to account for some of the variation in three-ridge mussel (*A. plicatsa*) recruitment (Ries *et al.*, 2015). Recruitment in other species appeared to be influenced by the number of low flow pulses and mean duration of high-flow pulses between April and September (Ries *et al.*, 2015). In their research on the Kiamichi River, Gates *et al.,* (2015) identified the period between June and September as being critical for maintaining water flows and water temperatures.

Wilson et al, (2011), included some aspects of the flow regime in their study identifying areas of high conservation for *Margaritifera margaritifera (L.)* in rivers in Northern Ireland. Using a number of landscape and habitat parameters, in a species distribution modelling approach, they found a negative relationship between species presence and stretches of fast flowing water (Wilson *et al.*, 2011). However, the analysis of flow regime characteristics in this study was limited to four site specific or reach level flow conditions defined by the *"Presence or absence of (a) broken standing waves, (b) unbroken standing waves, (c) rippled flow or (d) smooth flow"* (Wilson *et al.*, 2011). These descriptions of flow regime are different to statistical flow regime metrics that characterise the hydrological conditions of entire catchments over multiple years.

Vaughn *et al.* (2016) also measured the impact of 'severe hydrological drought' on 12 mussel species. In their study, flows below the 10th percentile were defined as severe hydrological drought, which the IHA defines as extreme low flows (Richter, 1996). Their research showed that the mussel populations in their study area never recovered to pre-drought levels, identifying increased water temperatures and reduced oxygen levels as likely factors in mussel mortality (Vaughn *et al.*, 2016).

Besides some general recommendations for flow regime characteristics (Skinner *et al.*, 2003) and site specific studies, no national-scale studies have been conducted that classify and specifically evaluate the optimum flow-regime metrics for viable *Margaritifera margaritifera L*. populations in the UK. This is an important knowledge gap, and an area identified as a research priority by Quinlan et al., (2015a) with a specific focus required on high and low river discharges.

Previous examples of hydrological classifications for the UK, have looked at magnitude and timing of flow regime events for hydrological classification (Harris *et al.*, 2000; Bower *et al.*, 2004; Monk *et al.*, 2006). This work stands apart by using a wider set of ecologically relevant hydrological metrics across all five key flow regime descriptors (M, T, D, F & R) from daily to monthly timesteps. This newly gathered information is then set against data on mussel population health to characterise and identify commonalities in the flow regimes of successful and failing mussel populations.

The implications of this study are discussed, solutions to address the limitations of the approaches used are proposed, and areas for the further development of this and future research in this field are suggested.

2.3 Methods

This study uses an inductive approach to classification, following a number of steps (Figure 11) adapted from Olden *et al.,* (2012) which provides a framework for hydrologic classification. In addition to hydrological data, information on mussel population health was used to categorise the catchment subgroups in the subsequent analysis.

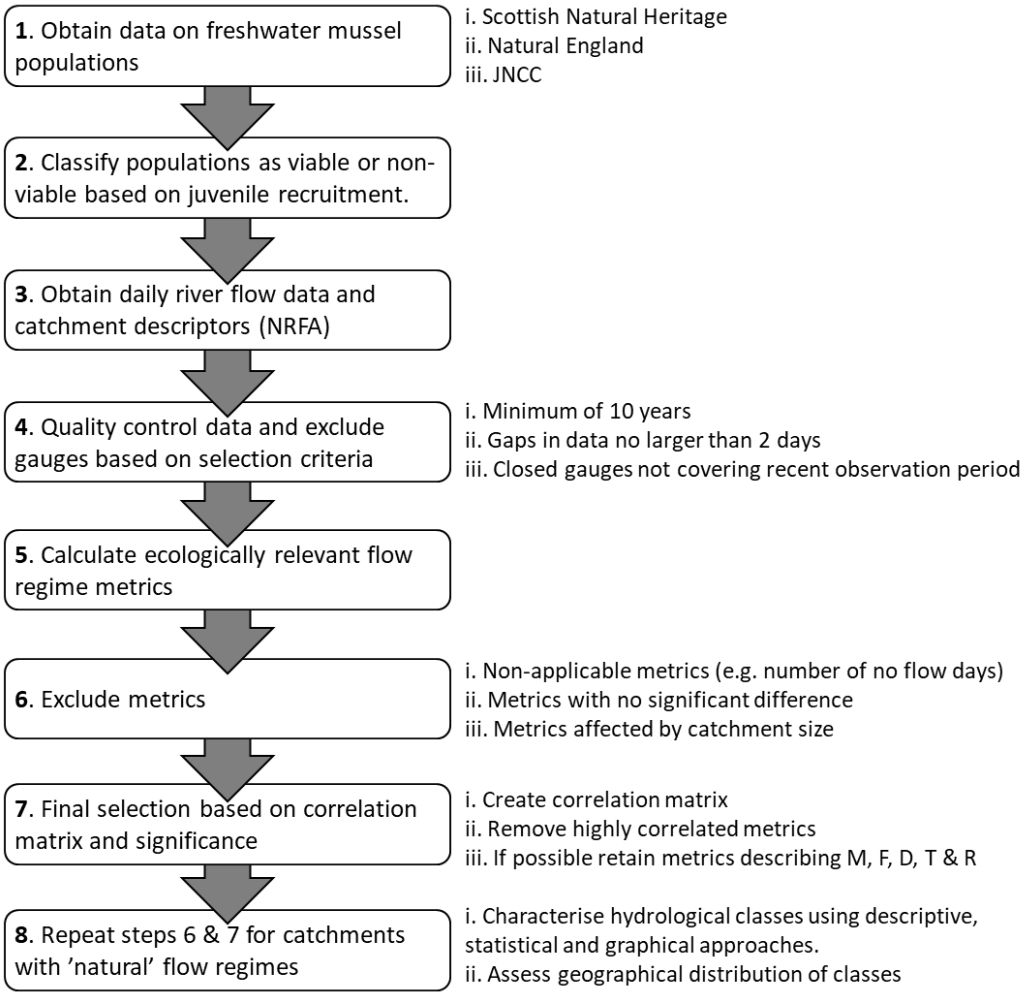


Figure 11. Outline methodology for the data collection, filtering and analysis in this study. Adapted from Olden et al., *(2012).*

2.3.1 *Mussel Data*

Information on freshwater mussel abundance and health was gathered primarily from Natural England and Scottish National Heritage. Where possible, the most recent mussel survey data was used to provide information on overall abundance and the percentage of juveniles within the population. Scottish Natural Heritage provided data for surveys carried out on 10 catchments and 16 water courses. Rivers were classified as sustaining either viable or non-viable populations based on the presence of juvenile mussels in the most recent surveys. This

classification was then verified by Scottish Natural Heritage. The data provided by Natural England had already been assessed internally. With the rivers, known to support mussels given a score and populations being assessed as viable or non-viable.

Data for additional sites was also obtained from the Joint Nature Conservation Committee website (JNCC). Each Special Area of Conversation (SAC) containing mussels as a qualifying feature has a site description and information about the health of the population with a grade from A – C assessing the overall health of the site. In the absence of additional, up to date information, this score was used to assess population viability. SACs with grades of A and B were classified as viable populations whilst those graded C were classified as non-viable.

2.3.2 *Flow Data*

Where available, daily flow data was gathered for each site from the National Rivers Flow Archive (NRFA). In the larger river systems, (e.g. Spey, Tay, Dee), flow data was gathered from multiple gauging stations along the main stems of the river and any subcatchments where mussel were present.

A number of sites, known to support populations of mussels were excluded from this study due to insufficient data in either the form of gauged flows, information on the mussel bed location or recent detailed population data. The remoteness of some watercourses known to support mussels meant that no gauged data was available at these sites. In some English and Welsh rivers, no recent survey data was available to confirm the presence of existing mussel populations despite historic evidence of their existence.

Where multiple gauges were present in the larger river systems, only the most relevant gauges were used. This selection was based on the proximity to mussel locations, the record length and the need to reduce redundancy in the data analysis. Under the terms of the data license agreement and protection of the Habitats Directive, detailed locations of the mussel beds are not shown in this research.

Additional data and catchment characteristics were compiled along with the discharge series from the National Rivers Flow Archive (NRFA). These included land cover statistics, information on elevation and topography, average rainfall data, factors affecting runoff (FAR), FARL and PROPWET as recorded on the NRFA website. The definitions and relevance of these catchment descriptors are outlined in the results and discussion sections.

As a means of visually identifying outliers or unusual hydrological behaviour, the mean daily discharge was plotted against the catchment area. The discharge time series data were then analysed using EflowStats R-package. This was run in python using the Rpy2 package. EflowStats (Thompson *et al.*, 2013), generates 171 ecologically relevant hydrological metrics as presented in the Hydrological Index Tool (Henriksen *et al.*, 2006). Initially the complete set of metrics (Table 1) was calculated in an effort to adequately describe all five components of the flow regime (M, T, D, F & R).

Table 1. Codes used for different flow regime metrics and the number of metrics describing each component of the flow regime in the HIT manual (Henriksen *et al.*, 2006).

Flow Component	*Code*	*Description*	*Quantity*
Magnitude	MA#	Magnitude, average flow event	45
	ML#	Magnitude, low flow event	22
	MH#	Magnitude, high flow event	27
Timing	TA#	Timing, average flow event	3
	TL#	Timing, low flow event	4
	TH#	Timing, high flow event	3
Duration	DL#	Duration, low flow event	20
	DH#	Duration, high flow event	24
Frequency	FL#	Frequency, low flow event	3
	FH#	Frequency, high flow event	11
Rate of Change	RA#	Rate of change, average event	9
		Total	**171**

Gaps in the streamflow record of up to 2 days were filled in using linear interpolation, however where gaps of three or more days were encountered the time series was considered to have ended. A minimum of record of 10 years is suggested for the generation of metrics using the Hydrological Index Tool, with 25 years being preferable (Henriksen *et al.*, 2006). However, using only gauges with at least 25 years of uninterrupted data created a vastly reduced pool of catchments for analysis so a compromise was made between meeting the minimum HIT recommendations and having a larger data set to analyse.

Metrics that were influenced by catchment size (mean flow, maximum discharge, etc.) were excluded from the classification. Standardizing these metrics by dividing them by catchment area or by the range of observed values was not considered due to the large number of initial metrics.

After further analysis, a small number of additional metrics were excluded, as they were not applicable to the study locations. Metrics DL18, DL19 & DL20 describe the number of zero flow days and zero flow months for a given discharge series. Within the analysed time series, no zero flow days were recorded.

The remaining metrics were then analysed to distinguish between normal and non-normal distribution using the Shapiro-Wilk test. To identify statistically significant differences between the metrics in the viable and non-viable catchments t-test (normally distributed data) and Wilcoxon-Mann-Whitney (data with non-normal distribution) scores were used. Metrics that did not show a significant difference between the two groups were then removed.

A correlation matrix was then constructed using Pearsons correlation scores. Highly correlated variables, with scores > 0.8 or < -0.8 were then removed to avoid redundancy. The decision of which metric to retain from the highly correlated subset was based on the results from the tests for statistical significance, with the metrics obtaining the lowest *P*-values retained.

The same process was then repeated using only catchments considered to have natural flow regimes or without any FAR listed in the NRFA database. The majority of natural gauged catchments in the NRFA network are generally small, upland catchments in rural areas of Wales, Scotland and South-west England (Bradford and Marsh, 2003). The number of natural catchments in this study was therefore disproportionately high compared to the national average. In some cases, the NRFA considers the flow regime of a catchment as natural even if abstractions or other modifying activities are present. However, these activities are usually small and consistent over time so as not to impact the analyses of natural changes and trends over time. In the present study the distinction is made between truly natural flow regimes and those with minimal alteration. Where flow altering activities are present, they are indicated and their impact on the flow regime is detailed.

2.4 Results

An initial set of 41 gauged daily flow time series was compiled for analysis. Gauged flow records varied considerably in length and continuity. For consistency, all metrics were generated from the same observational period. Data from several Scottish gauges was only available until 2016 as the stage discharge relationship at these sites has been impacted by the 2015/16 winter floods and work to update these records is ongoing.

After applying the quality control steps outlined in the methods section, a final set of 35 gauged records was obtained, each with a 12-year uninterrupted time series covering the period from 01/10/2004 to 30/09/2016. In the 35 retained catchments, 19 gauged records came from rivers or catchments supporting viable populations compared to 16 from non-viable populations. Of the six catchments excluded from the analysis, four were known to support viable mussel populations, with two containing non-viable populations (Figure 12).

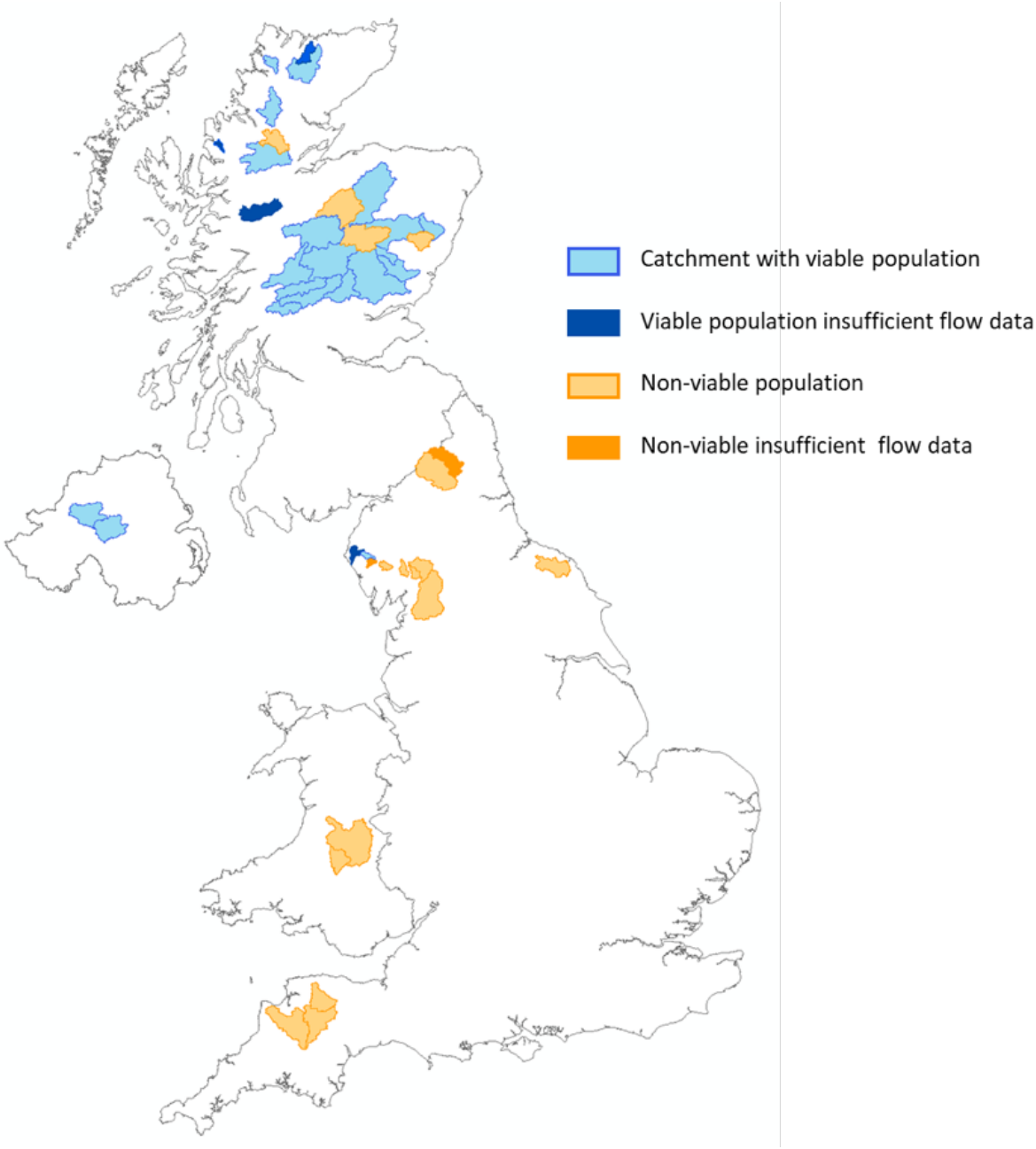


Figure 12. Map showing the geographical distribution of the initial pool of catchments from which flow data was obtained and then filtered to exclude short or fragmented time series.

2.4.1 *Catchment Characteristics*

The catchments analysed in the present study are characterised by their high altitude, having relatively steep river gradients, with minimal urban development and higher than average annual rainfall. This is clearly visible from the data in Table 2 and when observing the geographical distribution of the catchments in Figure 12.

*Table 2. Average values for the different catchment descriptors within the analysis and in context with UK catchments. Average values are calculated from data for 1,581 gauged UK catchments requested from the NRFA. Numbers in ( ) denote the averages of the final 35 catchments from which the hydrological metrics were calculated. * Denotes significant difference between viable and non-viable groups based on 35 catchments.*

Catchment Descriptor	UK	Analysed Catchments		Viable Populations		Non-viable Populations	
Altitude (mAOD)							
Minimum	66	65	(68)	57	(64)	74	(73)
10 Percentile	114	158	(163)	160	(166)	155	(159)
50 Percentile*	195	327	(337)	356	(380)	290	(285)
90 Percentile*	305	569	(578)	626	(653)	496	(488)
Maximum*	444	922	(933)	1041	(1069)	771	(771)
Land Use (%)							
Woodland	13	11	(12)	11	(11)	12	(12)
Arable/Horticultural	25	5	(5)	3	(3)	7	(8)
Grass*	41	40	(39)	29	(2)	54	(51)
Mountain Heath/Bog*	17	41	(41)	54	(54)	25	(26)
Urban*	8	1	(1)	0	(0)	1	(1)
Rainfall (mm/year)							
SAAR 41-70	1115	1609	(1557)	1673	(1619)	1527	(1483)
SAAR 61-90	1095	1570	(1523)	1640	(1590)	1481	(1443)
Other							
PROPWET (%)*	46	65	(64)	69	(68)	60	(60)
DPSBAR (m/km)	93	174	(171)	184	(187)	161	(51)
BFIHOST	0.51	0.42	(0.44)	0.42	(0.44)	0.42	(0.43)
FARL*	0.95	0.91	(0.92)	0.87	(0.89)	0.96	(0.97)
BFI	0.51	0.46	(0.45)	0.50	(0.49)	0.40	(0.4)

The average value for DPSBAR, the catchment descriptor for overall catchment steepness (Bayliss, 2008), is 165 m/km within the selected catchments compared to a national average of around 93 m/km. The DPSBAR value in viable catchments is lower than those with non-viable populations. Average rainfall is around 40% higher than the national average of NRFA gauged catchments. There is little difference in rainfall between catchments supporting viable and non-viable mussel populations. The PROPWET value, which is a measure of soil moisture, for the analysed catchments was 65%. This is far higher than the national average of 46%. Between the two groups, catchments supporting viable populations of mussels had slightly higher values for PROPWET than the non-viable group.

Mountain heath and grassland were the two most notable differences between the different groups of catchments in terms of landcover. Nationally, grassland covers on average 41% of

gauged catchments which is similar to the 40% for the analysed pool of catchments. The major difference lies between the viable and non-viable catchment subgroups, which have grass coverage of 29 %, and 51% respectively. Catchments with viable populations averaged more than 50% coverage of mountain heath compared to 26% for the non-viable catchments.

FARL is a measure of flood attenuation by reservoirs and lakes that was developed for the Flood Estimation Handbook (NRFA, 2020). A score of one implies no attenuation whereas a value below 0.8 implies that lakes and reservoirs have a considerable impact on the flood response (NRFA, 2020). The analysed catchments had an overall FARL value of 0.91 compared to a national average of 0.95. For viable catchments, FARL averaged 0.87 in comparison to 0.96 for non-viable catchments.

The catchment represented by station number 203012 was the Balinderry River in Northern Ireland, which had noticeably different land cover and elevation characteristics to the other catchments in the viable group. Catchments 12003 and 8010 were part of larger river basins, the Dee and Spey in Scotland where viable populations of mussels are present and thus have many physical characteristics in common with the viable group of catchments.

Black Water (4007) is a subcatchment of the River Conon in Scotland and has a number of lochs, thus reducing the FARL value and distinguishing it from the remaining catchment in that group. Catchment 13008, The South Esk, is one the most south-easterly Scottish catchments in the analysis and has a much lower than average annual rainfall. The soil moisture indicator PROPWET is therefore outside the normal range of values for the viable catchments.

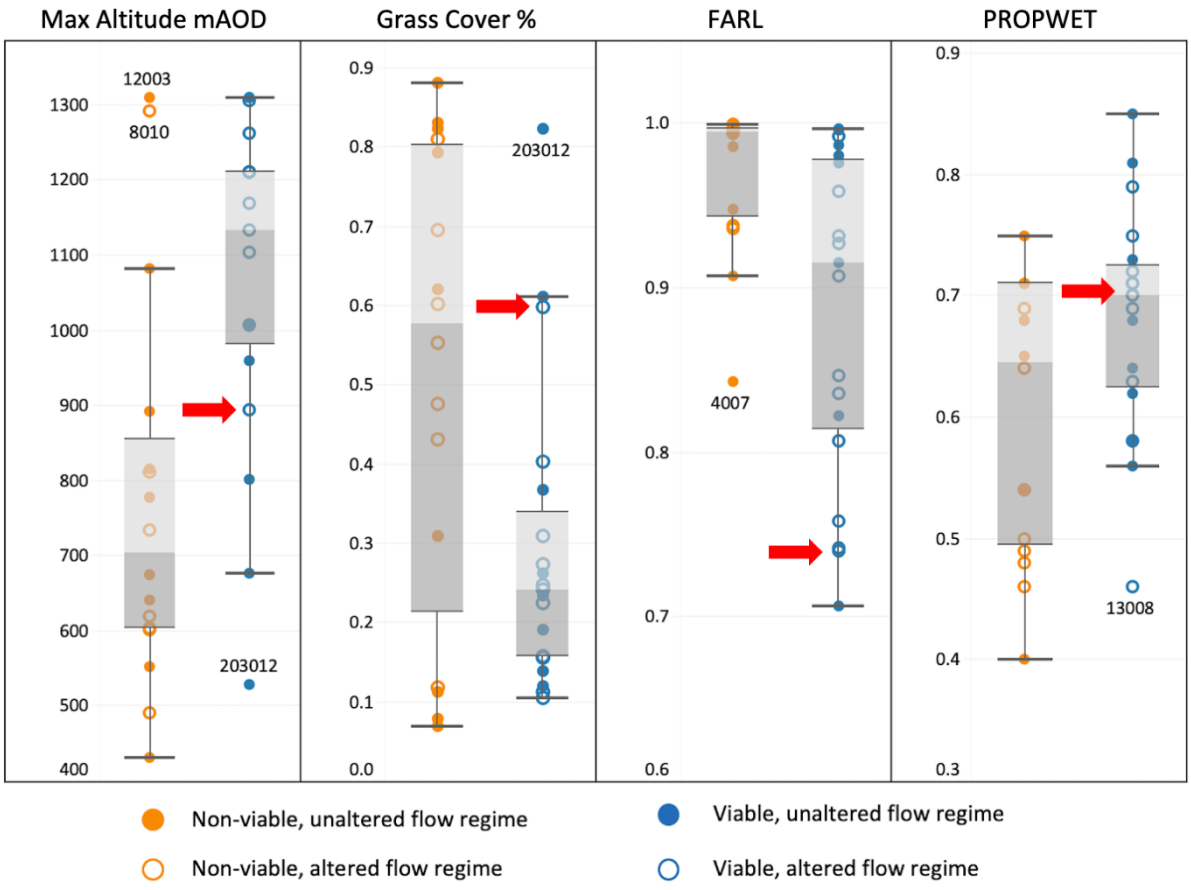


Figure 13. Box and whisker plots showing catchments descriptors obtained from the NRFA website for non-viable (orange) and viable (blue) catchments. Red arrows indicate the Upper Ehen catchment. Outliers are labelled with the NRFA code for the gauging station.

2.4.2 *Flow Regime Characteristics*

In the final pool, catchment sizes ranged from 45 km^2 to 4587 km^2 with mean daily flows ranging from 2.91 m^3/s to 172 m^3/s over the 12-year period of analysis. Average monthly flow as a percentage of total annual discharge was obtained for each catchment, with the mean flows for both subgroups, viable and non-viable populations, also calculated (

Figure 14).

Both groups of catchments showed peak discharge in January and minimum discharge in June. However, catchments with viable populations tended to have less extreme seasonal variation showing lower peaks in January and slightly higher minimum summer flows. Additionally, a number of catchments belonging to the viable populations group showed a small increase in discharge from March to April compared to a continual decline in discharge from winter to summer as seen in the majority of other catchments.

This seasonal variation in the annual hydrographs between the viable and non-viable groups reflects the average FARL values described in the previous section. FARL scores are in essence an inverse of flashiness in that they described the attenuation of peak flows.

Catchments with greater flow attenuation from lakes and reservoirs (FARL) show lower seasonal extremes between average winter flows and average summer flows. Whilst this may also be due to climate factors and other variable such as vegetation, the attenuation of flows by lakes will reduces the variation in extreme flows at both ends of the hydrograph.

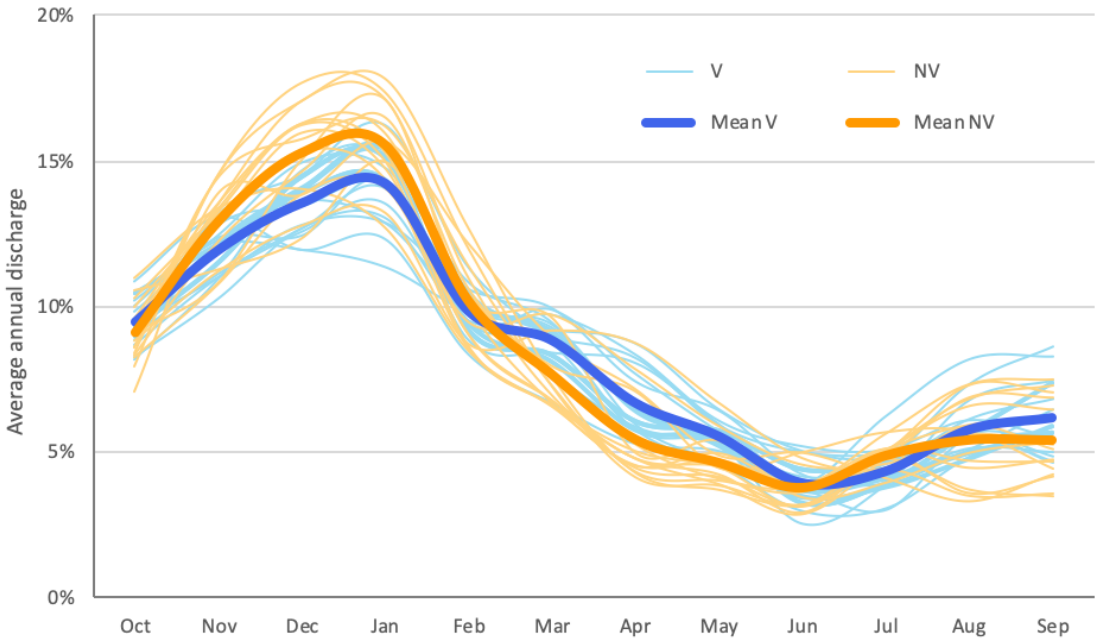


Figure 14. Average annual flow patterns for catchments with viable (blue) and non-viable (orange) mussel populations from 2004 - 2016. Dark colours represent the mean values from each group.

A number of the analysed catchments are classified as 'natural' under the NRFA system for estimating the factors affecting runoff (FAR) within a catchment. A further subset form part of the UK Benchmark Network of Reference Gauges (Harrigan *et al.*, 2017) which are selected for their long-term data series and 'near natural' flow regime. These sites allow useful comparison within the present study to highlight any differences between the two groups of mussel rivers.

Within the non-viable subgroup, nine of the 16 analysed catchments were described as having natural flow regimes or having no FAR, under the NRFA catchment descriptions. Within the viable subgroup, only eight of the 19 analysed flow regimes were considered natural.

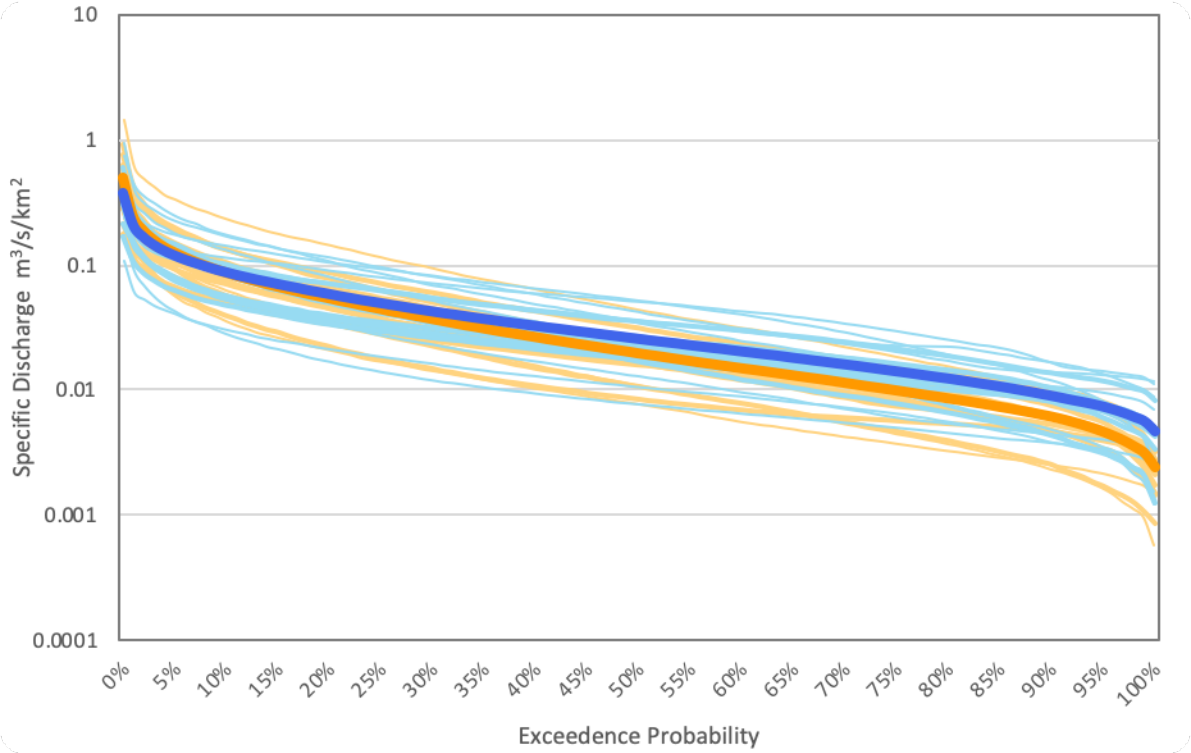


Figure 15. Flow duration curves normalised by area for the analysed catchments. Light blue lines show FDCs for catchments with viable populations, dark blue represents the mean. Orange lines represent rivers with non-viable populations, dark orange represents the mean.

Eight of the rivers supporting viable mussel populations were affected by hydroelectric schemes and three were affected by reservoirs or public water supplies. In the non-viable group, only one catchment was affected by a hydroelectric scheme but another seven were affected by either flow regulation, reservoir storage or public, industrial & agricultural water abstraction. The mean daily discharges of all catchments were plotted against the drainage area (Figure 16) to visually assess the impact of flow regime alterations and compare modified and natural flow regimes as advocated in Olden *et al.*, (2012).



Figure 16. Mean daily discharge vs catchment area. Lower left corner is enlarged for clarity. Red arrow indicates the Upper Ehen catchment.

Of the initial set of 171 metrics included in the HIT, only 157 were calculated due to problems running the code in Python and a lack of peak flow data for every site. This was then further reduced to a set of 59 final metrics (Table 3) after excluding metrics affected by catchment size, the non-applicable metrics and metrics showing no statistically significant difference between the two groups of catchments.

Table 3. Hydrological metrics excluded during the catchment classification process and reasons for their exclusion.

Reason for Exclusion	*Flow Regime Metrics*				
	Magnitude	***Frequency***	***Duration***	***Timing***	***Rate of Change***
No peak flow data		FH11	DH22 - 24	TA3, TH3, TL3-4,	
Error in Code	MH24-27				RA8 - 9
Not Applicable			DL18 - 20		
Catchment Size	MA1, MA2, MA12-23, ML1-12, MH1-12		DL1 - 5, DH1 - 5		RA1, RA3,
No significant difference	MA6, MA27, MA33, MA41, MA43 - 45, ML14, ML16, MH18-20	FL1 - 2, FH1 – 2, FH5-6, FH8-10,	DL8, DL11 - 17, DH6 - 10, DH15 - 18, DH20 - 21	TL1 – 2, TH1-2	RA7

Of the 59 remaining metrics, 40 described magnitude, four described frequency, nine duration, two timing and four the rate of change of flow events. Many of the remaining metrics showed either a strong positive or negative correlation (Figure 17). Removing these highly correlated metrics , based on the their statistical significance (see methods), left a final set of eleven metrics for analysis and discussion.

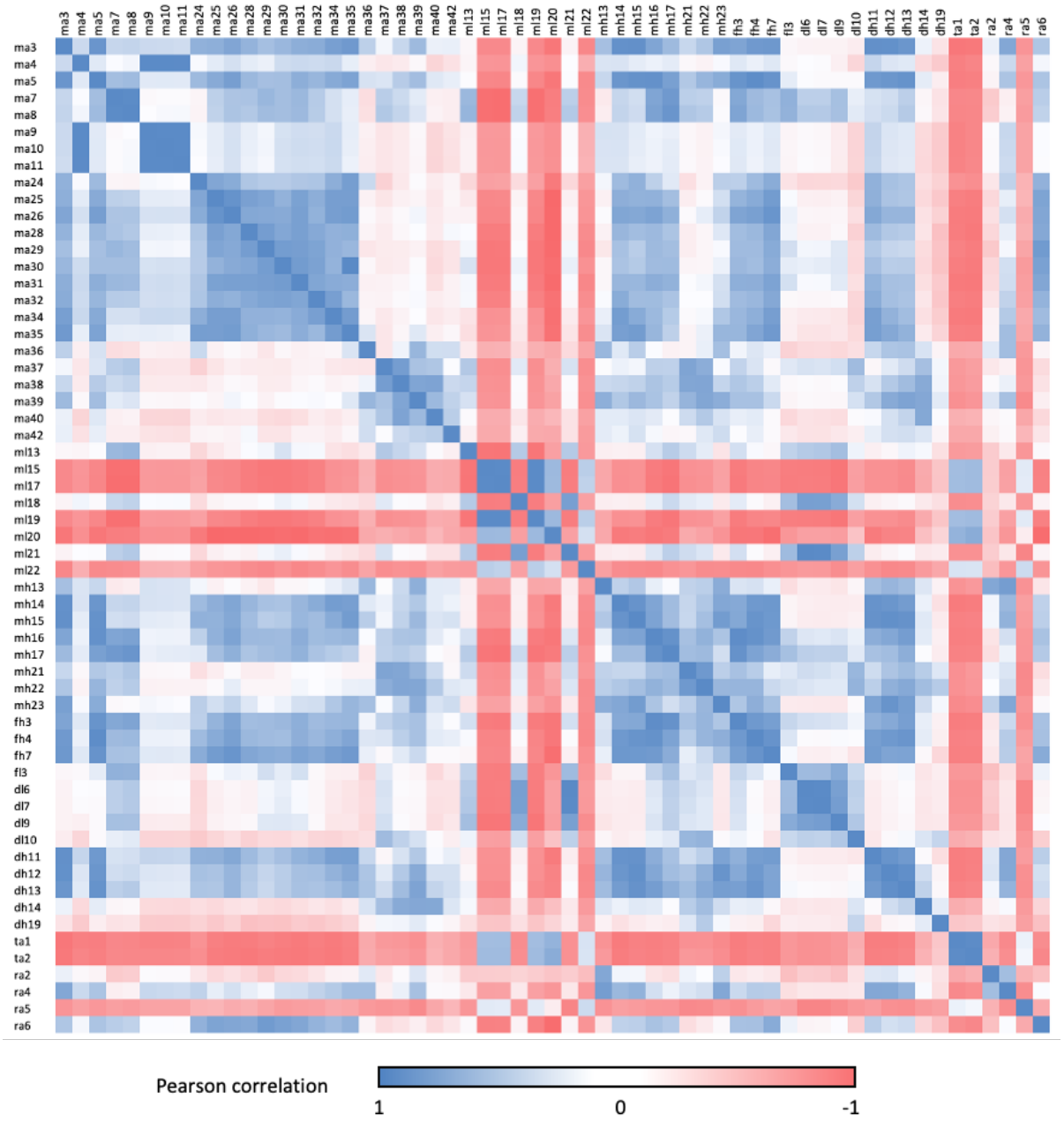


Figure 17. Correlation matrix of final 59 hydrological metrics. Darker blue represents highly positively correlated metrics and darker red highly negatively correlated metrics. Lighter areas denote lack of correlation.

Of the five different components of flow regime only three were represented in the final selection of 11 metrics, magnitude, duration and rate of change. Seven of the metrics describe the magnitude of flow events, with three describing average conditions and an additional two describing low and high flow events respectively. Two metrics described the duration of high flow events and one described low flow events. Finally, a single metric described the rate of change of average conditions (Table 4).

Individual values for each metric, the interquartile range are shown using box and whisker plots, with any outliers labelled using the NRFA gauging station number (Figure 18).

Table 4. Flow metrics showing greatest variation between catchments with non-viable and viable mussel populations. NV-N = non-viable mussel population in a natural flow regime. V-N = viable populations in a natural flow regime. Descriptions summarised from (Henriksen *et al.*, 2006).

Flow Component	**Metric**	***Mean Values***				***Description***
		NV	***V***	***NV - N***	***V - N***	
Duration	DH13	5.50	3.90	5.70	4.02	Annual maximum of 30-day moving average flows divided by the median (dimensionless – temporal).
	DH19	1.75	1.51	1.57	1.40	The average duration of high flow events seven times the median flow value (days – temporal).
	DL9	38.3	28.3	34.6	35.1	Variability of annual minimum of 30-day moving average flow (percent - spatial).
Magnitude	MA11	1.11	0.59	1.49	0.66	Spread in daily flows. (dimensionless – spatial).
	MA36	5.05	3.84	5.24	3.67	Variability across monthly flows (dimensionless – spatial).
	MA42	0.570	0.479	0.568	0.482	Variability across annual flows (dimensionless – spatial).
	MH16	4.36	3.40	4.37	3.67	High flow discharge index. The 10 percent exceedance value divided by the median flow (dimensionless – spatial).
	MH22	10.58	5.77	9.58	5.79	High flow volume. The average volume for flow events three times the median flow divided by the median flow (days - temporal).
	ML13	71.8	56.1	68.4	60.6	Variability (coefficient of variation) across minimum monthly flow values (percent – spatial).
	ML22	0.004	0.007	0.004	0.005	Specific mean annual minimum flow (m3/s/km2 – temporal).
Rate of Change	RA5	0.32	0.36	0.32	0.35	Number of day rises (dimensionless – spatial).

Average flow conditions were described by metrics MA11, MA36, MA42 and RA5. MA11 describes the spread in daily flows which was 80% larger in the non-viable group. MA36 and MA42 describe variability of monthly and annual flows respectively. In both cases greater variability was observed in the non-viable catchments.

Metrics describing the high flow characteristics of the two groups were DH13, DH19, MH16 and MH22. Both the duration and magnitude of high flow events were larger in the non-viable catchments

Finally, low flow events were described by metrics DL9, ML13 and ML22. The variability in duration and magnitude of minimum monthly flows was greatest in the non-viable group. Specific discharge (ML22) was also lower for these catchments.

When comparing the results from the entire pool of catchments with only those possessing unaltered flow regimes the differences between the two groups decreased in 8 out of the 11 metrics. The three metrics that showed a greater difference were DH13, MA11 and MA36.

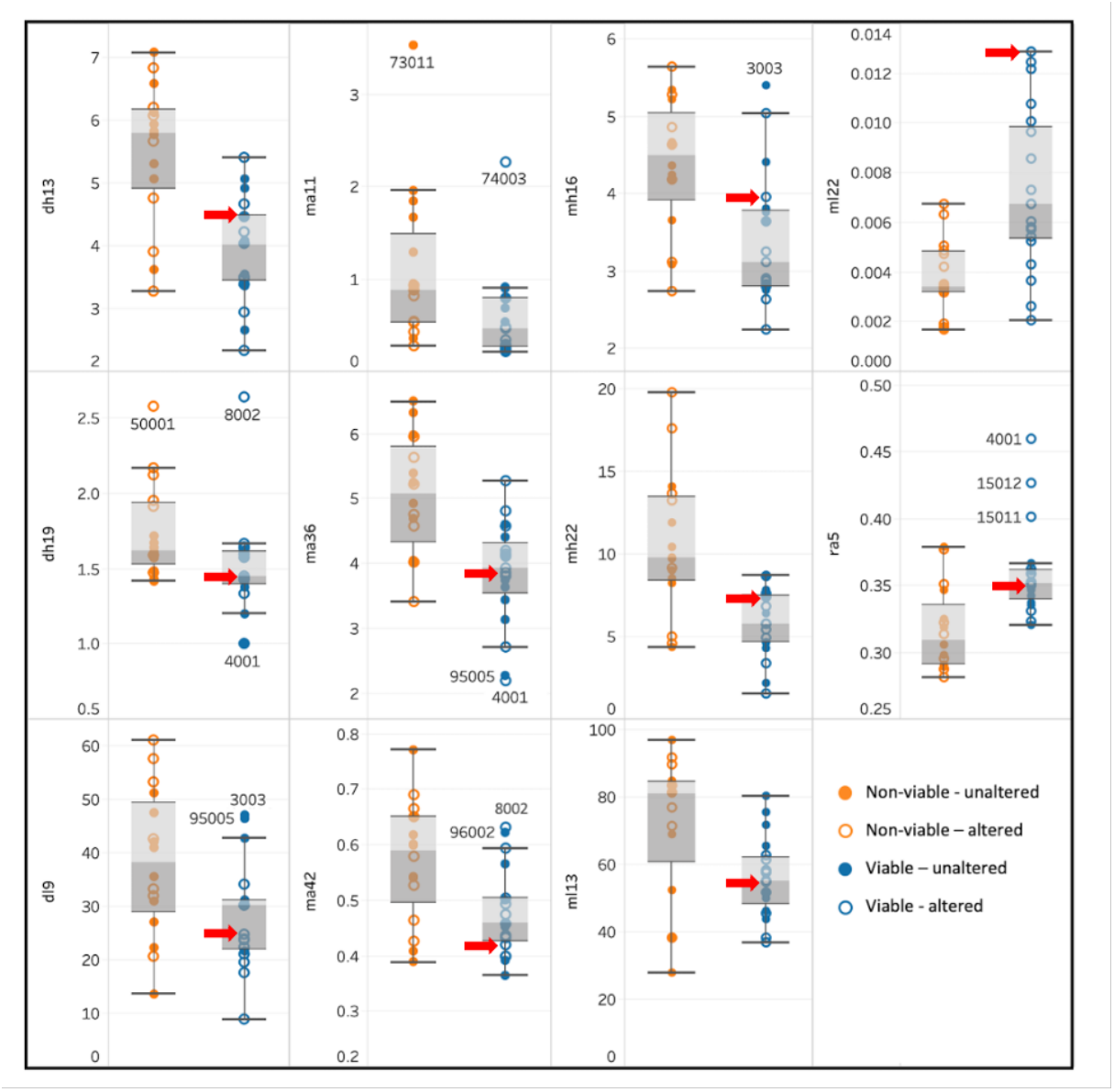


Figure 18. Box and whisker plots for indices for the non-viable and viable catchments with modified (ring) and unaltered flow regimes (solid dot). Shaded areas represent the interquartile range. Red arrows indicate the Upper Ehen catchment. Outliers are labelled with the NRFA code for the gauging station.

For comparison, the entire procedure for metric exclusion and selection was repeated using only the 17 catchments with natural or unaltered flow regimes. This produced a reduced set of 28 metrics, which was then further reduced to seven metrics using the correlation matrix. Four of these metrics where repetitions from the initial analysis however metrics MA38, MH18 and

TA1 were new. MA38 described variability across monthly flows using the 25th and 75th percentiles. MH18 describes the variability of annual maximum flows and TA1 is constancy.

The average values for MH18 when comparing all 35 catchments were 7.94 (non-viable) and 5.95 (viable) with no significant difference between the two groups. When looking solely at natural catchments the value within the non-viable group changed to 9.65 whilst the for the viable group the value only increased slightly to 5.99, showing a significant difference in the variability of maximum annual flows between the two groups.

Constancy values (TA1) averaged 0.23 (non-viable) and 0.37 (viable) when looking at all 35 catchments. These values changed to 0.17 (non-viable) and 0.28 (viable) when looking at natural catchments. In both cases the difference between the two groups was shown to be statistically significant.

Average values for MA38 were 2.13 (non-viable) and 1.65 (viable) for the 35 catchments and showed very little change when looking only at natural catchments with average values of 2.03 (non-viable) and 1.65 (viable) respectively.

2.5 Discussion

2.5.1 *Implications for Conservation*

Flow Regime Metrics

The results from this exploratory analysis of the UK's freshwater pearl mussel rivers provide a useful initial quantification of the flow regime characteristics required to sustain healthy populations. The findings support previous evidence in the literature that flow regulation (Gibbins *et al.*, 2006; Addy *et al.*, 2012; Quinlan *et al.*, 2015b) and rapid fluctuations between high and low flows (Skinner *et al.*, 2003) can have a detrimental impact on the species and hinder juvenile recruitment. Evidence from both altered and natural flow regimes supports these ideas.

Flow regimes in a number of the analysed catchments within the viable group (e.g. 74003), are managed to ensure minimum flows are maintained at certain times of the year for conservation of freshwater mussels and other aquatic organisms. This may also imply that in the non-viable catchments, altered flow regimes may not be managed for conservation purposes or are not being managed appropriately. When looking at the normalised mean FDCs (Figure 19) for the two groups as a whole compared to natural catchments the lower end of the curve diverges more in the entire pool. Low flows in the non-viable group are

therefore more extreme as a result of anthropogenic alterations to the flow regime. In the viable group low flows are less extreme, possibly as a result of flow regime management for conservation.

Extended periods of low flows can have a negative impact on a wide range of aquatic organisms, yet for many catchments they are part of the natural variation caused by seasonally changing climatic conditions. Low flows may become more problematic however, when their duration, magnitude, frequency, timing or rate of change become less predictable.

In addition to the FDC, the metrics describing low flows show that minimum monthly flows and the minimum 30-day moving average vary more greatly in the non-viable group. Specific discharge (ML22) is also considerably lower.

High flows events within the non-viable group were larger in magnitude and duration. High flows are believed to be particularly critical in the juvenile stage of the mussel lifecycle. During this period mussels can remain buried in river substrates for up to 5 years and the intensity and frequency of flood events can be a critically limiting factor for their survival (Geist, 2010).

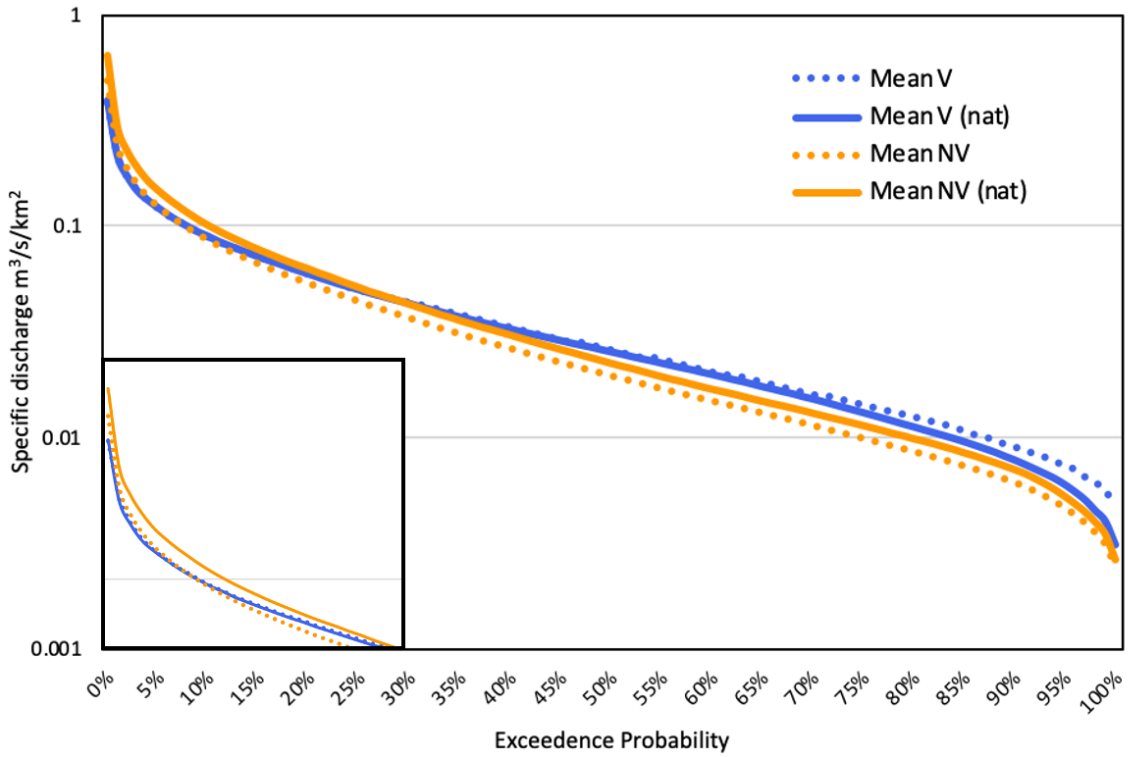


Figure 19. Flow duration curves for all catchments (NV & V) compared to natural flow regimes (NV &V).

Average flow conditions also varied more greatly within the non-viable group. The spread of daily flows was wider, and the variability of monthly and annual flows was greater. RA5, the number of days that the flow rate rises, was smaller for the non-viable group. This is a

reflection of the climatic conditions, suggesting that rainfall events occur less frequently and as such recessional limbs of the hydrograph are longer.

Of the eleven remaining flow regime metrics, four described average condition, four described high flow conditions and three described low flows. In all but three cases the differences between the two groups diminished when looking solely at natural catchments.

Metrics describing the frequency and timing of flow regime events were absent from the final initial selection however metric TA1 was present when analysing the smaller pool of natural catchments. The timing of flow regime events is significant for aquatic species as they can determine the success of different stages of their lifecycles and may act as physiochemical cues for activities such as spawning or migration.

TA1 is the metric for constancy. It is the converse of uncertainty with values ranging from 0 – 1. Flows are divided into 11 categories and the number of times a flow falls into each category is recorded. An equal distribution across all flow categories would achieve a score of 1 and imply a single state across the hydrological year (Gan *et al.*, 1991).

Scores for constancy were higher in both groups of catchments, viable and non-viable, when all 35 catchments were included, implying that the modification of flow regimes is reducing the annual variation in flows. This would therefore limit the range of extreme low flows at the lower end of the flow duration curve (Figure 19). This is evident in the case of the catchments with viable populations however for non-viable catchments lower flows are more extreme. This provides further evidence that the management of low flows in the non-viable group is either inadequate or non-existent.

Previous studies of freshwater mussel species have shown a clear extinction gradient downstream of impoundments (Vaughn and Taylor, 1999), whilst further site specific studies have used descriptive (Hastie *et al.*, 2003b) and quantifiable flow characteristics (Gosselin *et al.*, 2014) for predicting habitat suitability and use in freshwater pearl mussels. Although useful in the context of this study they rely on the availability of fine resolution data and detailed site-specific surveys.

With access to finer resolution data and uniformity of data across multiple sites, it may be possible to replicate methodologies from studies into different mussels species such as those of Ries et al., (2015) and Gates *et al.,* (2015), who identified key flow conditions for mussels and the success of the species. The present study provides a national scale overview which could be used to identify specific locations in which to conduct further detailed studies into the flow conditions and characteristics of the mussel populations at these sites.

Low flows have been cited as a limiting factor in the lifecycle of freshwater mussels, particularly during juvenile development. However, the present study did not identify a prevalence of low flow metrics when distinguishing between viable and non-viable catchment. Analysis of the catchment descriptor FARL, which is also describes the flashiness of a catchment, suggests that the attenuation of peak flow events by lakes and reservoirs may positively impact freshwater mussel populations.

Extreme flood events have been shown to directly reduce mussel populations by scouring river beds and physically removing them (Hastie *et al.*, 2001). However, extreme high flow events may also directly impact on the host species of the mussel by washing out the redds of salmonids and reducing their overall abundance in a river. Physical degradation of habitat may also impact upon the wider health of aquatic ecosystems and if these events occur regularly it may limit the opportunity for species and habitats to recover.

Catchment Descriptors

Within the analysed catchments, the lowest FARL values were observed in regions of the UK with high density of lakes and lochs, such as Upland Scotland and The Lake District. The lowest observed FARL values were within the viable subgroup. The River Borgie (96006), the River Laxford (95005) and River Kerry (94003) have FARL values of 0.66, 0.71, and 0.71 respectively. Neither the Kerry nor the Borgie were included in the generation of flow regime metrics as their time series didn't meet the quality control criteria, however as rivers supporting viable populations, their catchment characteristics are still of interest.

Both the River Borgie and the River Laxford have no recorded FARs on the NRFA website. The attenuation of flood peaks within these catchments is therefore based on natural physical characteristics i.e the presence of lochs and lakes. The River Kerry, however, is heavily regulated and compensation flows have been established to aid the conservation of mussels downstream of hydropower schemes.

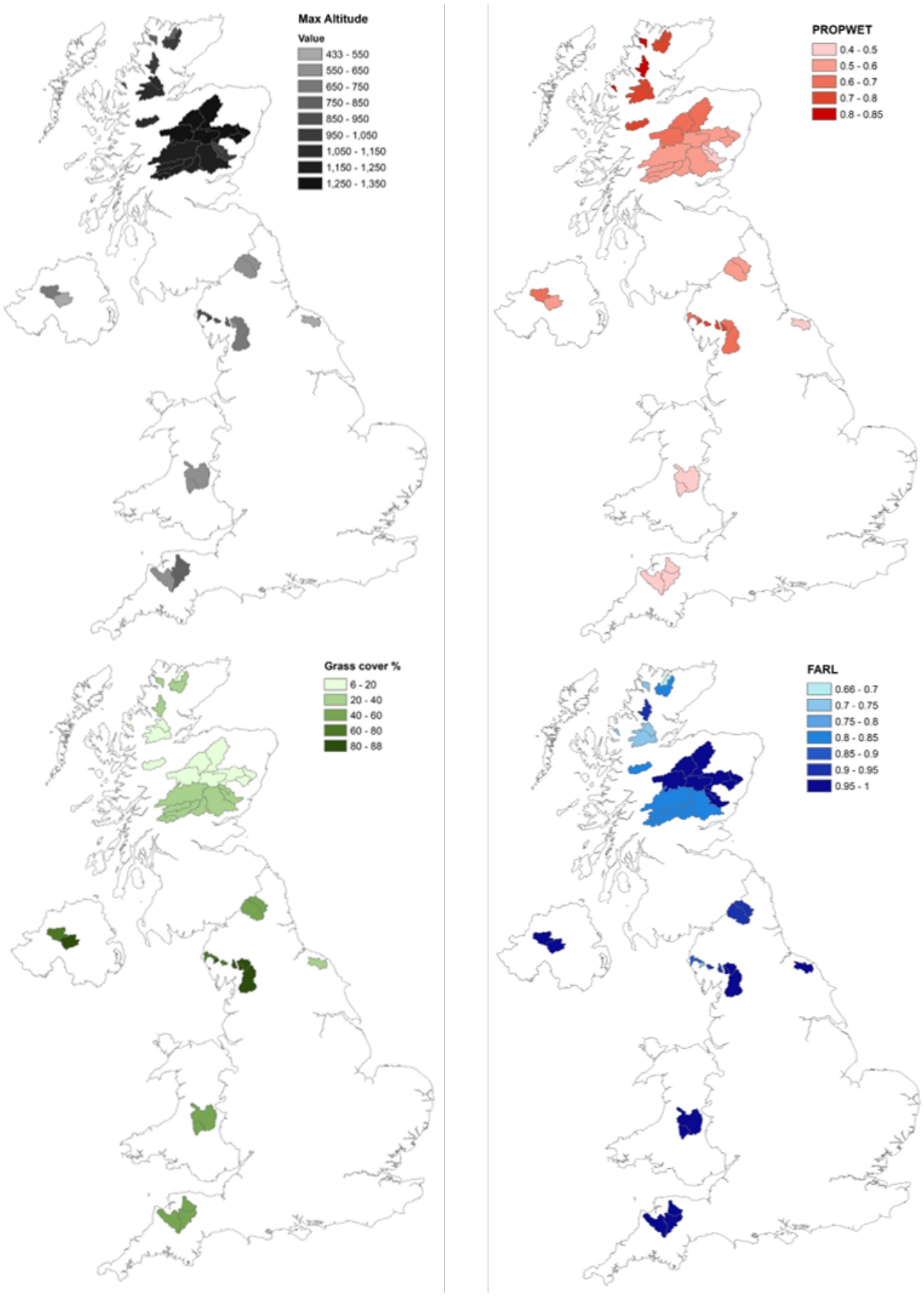


Figure 20. Geographical distribution of PROPWET and FARL values for the initial 41 analysed catchments.

The next lowest FARL values were in the River Ehen and River Conon, both with FARL values of 0.74. The River Ehen has some abstraction and flow regulation, which may

influence the FARL value. However, Ennerdale Water is a natural lake within the catchment and would thus have attenuating properties in the absence of abstraction and impoundment.

Several hydroelectric dams influence the hydrological regime of the River Conon. However, unregulated lochs are present upstream and in some nested subcatchments. The FARL score is likely to be impacted by the production of hydroelectricity however, natural attenuation is also present.

Based on the findings from this analysis, the attenuation of flood peaks, natural or otherwise, appears to have a positive influence on the viability of freshwater mussel populations. This may be because it helps limit the amount of physical habitat degradation that can occur in extreme flood events. Previous research has shown the impact large flood events can have on freshwater mussels, with one study analysing the impact of a 1-100 year flood on the River Dee, estimating mussel mortality in excess of 50,000 individuals (Hastie *et al.*, 2001). Additionally, flood-attenuating properties help to dampen the annual flow discharge regime, subjecting mussels to less extreme flow variation throughout the year and potentially subjecting them to less stress. The results from the final metrics (DH13, DH19, MH16 & MH22) add weight to the importance of high flows as a factor in the population status of freshwater mussels.

From a conservation perspective, this raises some interesting possibilities for alternative approaches to catchment and flow regime management. Flood attenuation by natural means (often referred to as natural flood management NFM) is a growing area of research and has gained popularity in some quarters as a more ecologically beneficial, sustainable and cost-effective alternative to hard engineering solutions. There may potentially be scope to employ these methods for conservation or in some cases for flood protection and conservation purposes simultaneously.

The benefits of NFM techniques are still difficult to quantify and most evidence of their effectiveness has been witnessed on smaller catchments (Dadson *et al.*, 2017). However, in heavily modified flow regimes or areas of high conservation value these types of techniques could potentially add to wider catchment-scale restoration efforts. Research has also shown the NFM techniques such as afforestation can have additional catchment benefits beyond flood control, highlighting improvements to water quality and climate regulation as examples (Dittrich *et al.*, 2019).

More recently, the term catchment systems engineering (CSE) has been introduced to cover a range of management options. CSE describes an interventionist approach to sustainably

managing water quality and water quantity by manipulating hydrological pathways modifying catchment scale runoff regimes and nutrient dynamics (Quinn *et al.*, 2010). The approach often relies on runoff attenuation features (RAFs) that slow, store and filter water and allow for greater infiltration thus providing multiple benefits (Quinn *et al.*, 2010). In the case of freshwater mussels, as well as altering flow regimes, these techniques could help to reduce sediment and nutrient loads reaching water courses thus further aiding the conservation of the species.

Flow regime alteration, impoundment and the associated mechanisms that alter or degrade physical habitat have been highlighted as major concerns for the conservation of freshwater mussels. Addy *et al.*, (2012) studied the impacts of flow regulation on freshwater pearl mussels in the River Moriston in Northwest Scotland. This catchment was included in the initial study however as the gauging station closed in 2011 the flow data did not meet the criteria for HM calculation. Their work suggests that flow regulation has resulted in habitat degradation in the form of bed compaction which may be limiting juvenile recruitment (Addy *et al.*, 2012)

Landcover was another distinguishing catchment characteristic. The percentage of grass cover ranged considerably within both groups of catchments. However, non-viable populations were mostly associated with grass cover of more than 40% with several exceeding 80%. In the viable catchments, mountain heath was generally the predominant land cover.

A high percentage of grassland in a catchment may influence a river's flow regime as runoff responses and evaporative losses will differ when compared to other landcover types. Furthermore, the high percentage of grassland may be associated with agricultural practises that exert an influence on water quality parameters.

As a conservation measure, reducing the level of grassland across an entire catchment is impractical and challenging for a number of reasons. However, targeting specific areas of the catchment for land use change may be a viable solution in some cases.

Freshwater mussels are known to favour shady stretches of river, as this helps minimise fluctuations in water temperature. Tree planting along riverbanks has thus been employed as a conservation strategy to help increase levels of shade and also aid bank stabilisation. Despite being a natural and important part of river ecosystem dynamics (Florsheim *et al.*, 2008), limiting bank erosion is seen as a valuable tool in reducing fine sediment loads in rivers. However, riparian planting of trees may need to be extended beyond the riverbanks and include a wider buffer of woodland to be fully effective.

Implementing this type of conservation method is time consuming and requires collaboration with landowners. Grassland for grazing or other agricultural practises can be a valuable commodity and persuading farmers to give over a proportion of this to riparian planting is not always straightforward. However, several schemes have had some success in this area. Horton *et al.*, (2015), studied the effectiveness of these types of measure for the reintroduction of captive bred mussels to the Balinderry River catchment (Station No. 203012), Northern Ireland. Their study highlighted the need for stakeholder engagement and education to better involve the local community and riparian landowners and successfully implement these measures (Horton *et al.*, 2013).

Due to the length of time, most trees require to establish themselves and the long complex lifecycle of freshwater mussels; it can be difficult to quantify the ecological impacts these measures have on mussel populations. Furthermore, it is extremely difficult to attribute any observed improvements in the health of the species to one specific conservation measure.

The catchment descriptor PROPWET, describes the proportion of the time the soils within are catchment are wet. Values typically range from around 80% in the wetter parts of the country to 20% in the driest parts (NRFA, 2020). This measure is used by the FEH as an indicator for flooding, as saturated soils are often antecedent to large floods (NRFA, 2020).

However, this understanding of the PROPWET descriptor may be misleading for the analysed catchments. The low FARL scores observed in the viable catchments would attenuate the impact of large floods potentially bought about in response to highly saturated soils. Instead, the continually high levels of soil moisture may be indicative of a slower release of water from catchment stores thus helping to delay hydrograph response and minimise the frequency or severity of low flow events.

PROPWET is influenced by a number of factors and has been shown to be highly correlated with altitude and arable land cover in a previous UK catchment study conducted by (Chiverton *et al.*, 2015). In the present study, PROPWET was highly correlated with altitude and highest PROPWET values were observed at more northerly latitudes and more westerly regions of the UK.

2.5.2 *Limitations*

The methodology applied in this study was designed to provide a rapid assessment and characterisation of the flow regime types experienced in UK freshwater mussel rivers. As such there are areas where the approaches could be expanded and improved upon with the availability of more data, time and expertise from other disciplines.

Hydroecological studies are often constrained by a lack of long-term ecological data. By its very nature as an endangered and protected species, collecting sufficient information on mussel population health and flow regimes is challenging. The small sample constrains the type of statistical analyses that can be carried out on the generated data. In other hydroecological studies, techniques such as principle component analysis (e.g. Olden and Poff, 2003; Monk *et al.*, 2006; Tetzlaff *et al.*, 2008) or other variable reduction techniques have been used for classification and to identify the key distinguishing features between groups. However, these types of approaches are better suited to larger sample sizes.

The number of rivers and watercourses supporting mussels in the UK is greater than that analysed in this study. The majority of designated sites (SACs) were included as they are of high conservation value and extensively monitored. However, a small number of them lack long-term gauged records and detailed flow regime data. Additionally, the way in which monitoring is conducted and data is collected can vary from site to site and between national regulatory agencies. For large-scale comparative studies there is therefore a need for a more standardised approach to monitoring to allow for consistent and more robust analyses to take place.

This need was recently recognised by the European Committee for Standardization (CEN) and addressed through the development and publication of a monitoring protocol for *Margaritifera margaritifera* (Boon *et al.*, 2019). The standard monitoring approach was devised by experts in pearl mussel ecology from 11 European countries and represents the first CEN standard dedicated solely to one species (Boon *et al.*, 2019). However, it will be several years until this is applied to all sites and a large uniform data set is obtained.

In the present study, ecological data were gathered from multiple sources. As such, they were available with varying levels of information, acquired using different assessment methods which were obtained from surveys with variable start and end dates. This made it difficult to create different subgroups for classifying the health status of each sites. Consequently, only two groups were created, viable and non-viable. With more consistent and detailed information across all the sites it may have possible to create additional groups to help further analysis and potentially identify different trends or commonalities between flow regime metrics and mussel health.

Inferring causal relationships between flow regime alterations and ecological responses can be very challenging (Webb *et al.*, 2012). In the case of freshwater mussels, detecting causal relationships is further impeded by the complexity of their lifecycle, their long lifespan and the potential for catchment scale (or beyond) factors to influence flow regime changes

(Degerman *et al.*, 2013). Juvenile mussels take around 10 – 15 years to reach reproductive maturity (Skinner *et al.*, 2003)**.** There is likely to be some latency between the effects of flow regime alterations or within catchment changes (water quality, water temperature, etc.) and the evidence of their impact on the species. Therefore, any impacts may not be fully understood or observed for many years, further highlighting the need for continual long-term ecological monitoring.

Catchment scale issues may have a direct impact on flow-regime and be evident in hydroecological studies of this type. However, a number of other critical factors are known to impact on the species and thus factors such as water quality, temperature and sediment also need to be considered. In catchments with apparently suitable flow regime conditions but non-viable or extinct mussel populations, attention should be focused on these catchment scale issues or other limiting factors. Without this additional information it is not possible to determine whether flow regimes are or have previously been optimal for a species and whether or not other causes are responsible for its decline.

Degerman *et al.,* (2013) used a combination of a GIS data on riparian land-cover and use, water chemistry data, data describing the abundance host fish species, brown trout, obtained from electrofishing in 56 streams. Their study found that mussel population viability was best predicted by total phosphorus was the best predictor for mussel viability. Whilst this is informative it poses the same issue as solely relying on flow regime characteristics to identify optimum habitat conditions for mussels. The authors of the study note that this information can be used for screening waters potentially suitable for mussels however analysis of wider factors should also be included. The same applies to this study in that flow regime characteristics can be used to indicate or screening likely suitable flow conditions for mussels. However, wider factors (e.g. water chemistry) are also at play and an assessment of how these have changed over time may be required to fully understand their impact on mussel health.

Finally, the discharge time series used to generate the flow regime metrics was only 12 years long. A minimum of 10 years is required however 25 is preferable. Twelve years were used to increase the number of catchments available for analysis, however, this will have impacted the flow regime metric values. To demonstrate this, metrics were calculated for catchment 12001, using time series data extending from 2016 to 1985. For metrics DH13, MA11 and RA5 values changed from 3.37, 0.286 and 0.342 in the shortened time series, to 3.30, 0.298 and 0.344 with the longer time series. Whilst these changes may appear small it is important to acknowledge the impact of the length of the time series on the robustness of the final metrics.

2.5.3 *Further Research*

The focus of this research has been on identifying similarities and differences between the flow regime characteristics of rivers supporting freshwater mussels in the UK. A major development of the present study would be to integrate the flow regime data and information on the physical catchment characteristics with additional data on water quality. Including data on factors such as turbidity, temperature and water chemistry could further help identify causes for the decline of the species and also help to distinguish between possible causes for the non-viability of populations in catchments with seemingly suitable flow regime conditions.

A number of the analysed catchments, particularly the SACs, are known to have introduced a wide range of monitoring under the terms of the Habitats Directive. Collecting, processing and interpreting all of these data sets across a large number of catchments would be a challenging but potentially enlightening exercise which could further inform best conservation practices for the species. Current conservation approaches tend to be focussed on a site-by-site basis however making use of large data sets may provide new evidence of wider trends in the decline of the species.

A complimentary further investigation to this study would be to expand the pool of analysed flow regimes. Using hydrological models is one such approach for extending time series in rivers with short data records. This approach can also be used to generate new time series in ungauged catchments or to estimate flows at locations closer to known mussel populations. The use of models to generate new flow time series is also widely used for assessing the level of flow regime alteration in catchments lacking pre-disturbance data enabling the degree and type of alteration to be quantified (e.g. Irving *et al.*, 2018).

Expanding the pool of analysed catchments would allow for more robust statistical analyses of the flow regime similarities and differences between the two groups of catchments. Furthermore, generating longer time series would provide more information on the flow regime characteristics of each site, potentially allowing for the detection of changes over time and for the generation of more robust flow regime metrics.

Additionally, extending the discharge time series into the future could allow for the impacts of future climatic conditions to be analysed under a range of climate change scenarios. These types of studies have been conducted elsewhere for a number of species including freshwater mussels.

Research looking at Portuguese populations of *Margaritifera margaritifera* suggest that summer low flows and water depths could fall below critical levels (Santos *et al.*, 2015). However in the UK, more intense rainfall and increased flood frequency may present a different threat (Hastie *et al.*, 2003a). Quantifying these changes with the use ecologically relevant flow regime metrics would provide important information at national level and help guide future conservation strategies and target areas of highest concern or with greatest restoration potential.

Linking the current set of flow regime metrics to specific aspects within the reproductive and survival strategies of is an important next step. Expert knowledge of the species and the biological implications of different flow regime events are needed to further understand the complex processes and relationships. Furthermore, it may be possible to identify or establish new hydrological metrics that quantify aspects of the flow regime that are specific to the species. For example, spates in autumn are believed to be beneficial to the species as they help remove algae from the substrate and serve to improve physical streambed conditions. It may therefore be possible to quantify this in the form of a new hydrological metrics and confirm this link through analysis of the flow regimes in the successful and failing mussel populations.

2.5.4 *Summary*

Flow regime alteration is a major threat to aquatic biodiversity globally and a number of species, including freshwater pearl mussels, face the threat of extinction. Categorising flow regimes and quantifying different components is an important first step in understanding the complex links between flow and ecological responses.

Despite the small pool of analysed catchments and other limitations of the study, a number of distinguishing features between the viable and non-viable subgroups can be observed in the calculated flow regime metrics. The results from the metrics are not designed to be used as prescriptive values for the management of flow regimes. However, they provide a quantitative measure of different aspects of the flow regime and help identify areas that may require further investigation or management interventions for the conservation of freshwater mussels in struggling or failing populations.

Although understanding and managing flow regimes is likely to play a central role in the conservation of freshwater mussels it should always be looked at within a multidisciplinary setting and in conjunction with wider water quality and catchment scale issues.

Acknowledgements

Special thanks to Iain Sime of Scottish Natural Heritage for providing recent mussel survey data, reports and providing consultation on Scottish rivers and mussel status. Thanks also to Gavin Measures of Natural England for providing information on mussel population health in English and Welsh rivers along with additional reports.

Chapter 3 - Upper Ehen Hydrological Characterisation

This chapter describes the unmanaged and modified hydrological features of the catchment. The hydrological regime is characterised using existing historic discharge time series to generate ecologically relevant metrics which serve as a basis for comparison with the findings from Chapter 2. The hydrology of the unmanaged region of the catchment is captured using data from extensive field monitoring which serves as a basis for model calibration in Chapter 4.

3.1 Conceptual Hydrological Model

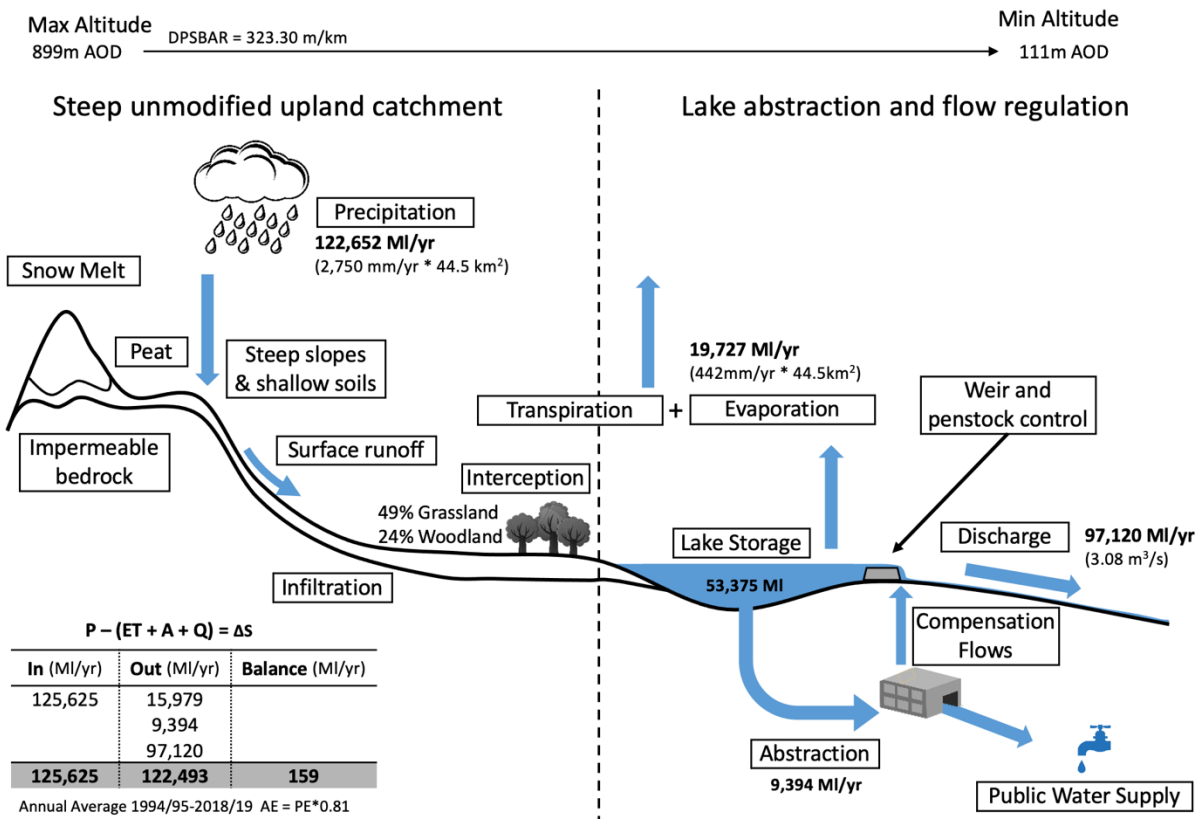


Figure 21. Simplified hydrological processes in the Upper Ehen catchment and average annual water balance where P = precipitation, PET = potential evapotranspiration, AE = estimated actual evapotranspiration (approximately 80% of PET), A = abstraction & Q = discharge

The hydrology of the Upper Ehen catchment was defined using data from the hydrological year 1995/1996 to the present day (hydrological year 2018/2019). Flow data was available from the 1970's, however, the period of analysis was chosen as the last major physical change

to the outlet of the Ennerdale Water was made in 1995 with the construction of a new impounding weir. This therefore represents a key change in the time series and a logical starting point for hydrological analysis. Furthermore, it provides a long enough data series to generate statistically robust hydrological indices.

Using the available hydrological, abstraction and climate data for this period, a simplified estimate of the average annual catchment water balance was made (Figure 21). This conceptual representation of the principle hydrological process in the catchment provides a useful starting point for characterising the catchment and informing the monitoring and modelling approaches used in subsequent chapters.

The inflows into Ennerdale Water are unregulated and the least modified flows of the entire system. High altitude and steep slopes provoke the highest annual rainfall totals and most rapid runoff responses. At the lower end of the valley and the north-western corner of Ennerdale water, altitude decreases, and more gently sloping farmland can be found, however, the southern edge of the lake remains flanked by steep slopes for almost its entirety.

3.2 Catchment Characteristics – Natural and Managed

The following section describes the key climatic and physical characteristics that control the hydrology of the entire catchment. Additionally, the influence of abstraction infrastructure and its operation on the hydrological behaviour of the site are described.

3.2.1 *Climate*

The North-west of England is home to some of the coldest and wettest parts of the country (UK Met Office, 2012). Prevailing Atlantic winds and abrupt changes in topography exert a strong influence on local weather patterns.

The average annual rainfall within the upper Ehen Catchment during the period between 1975 and 2015 exceeds 2600 mm. In the lower regions of the catchment, annual rainfall is around 1,800 mm whilst at the highest altitudes, annual average totals approach 4,000 mm. October to December is the wettest period of the year with monthly average rainfall exceeding 300 mm (Figure 22).

The driest hydrological year during the observation period occurred in 1995/96 when an estimated total rainfall of 1,857 mm was recorded. The wettest year was 2008/09 with an estimated 3,396 mm of rainfall falling across the catchment.

The highest monthly rainfall total within the observation period was 667 mm occurring in October 2008. The single highest 24-hour rainfall total occurred on the 19th of November 2009, when an estimated 144 mm of rain fell over the catchment. The ten highest rainfall totals observed in the catchment have all occurred since 2005 and of the 20 largest events only one occurred prior to this date, occurring in 1997.

July is the warmest month of the year, with temperatures averaging highs of around 17 °C and lows of 9 °C (Figure 22). The annual average temperature in the catchment is slightly above 7 °C. For every 100m increase in altitude average temperatures decrease by around 0.5 °C (UK Met Office, 2012). Based on values from 1995 to 2019 potential evapotranspiration averages around 442 mm per year.

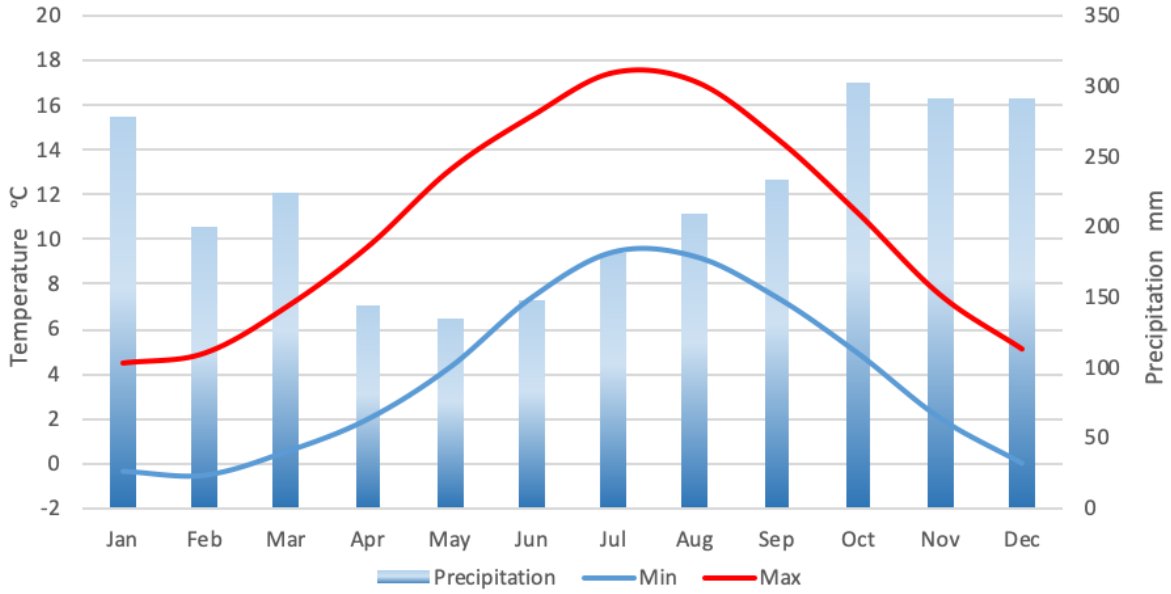


Figure 22. Area weighted climate statistics for the upper Ehen catchment. Source data taken from and CHESS (temperature) and CEH GEAR (rainfall) data sets (Robinson *et al.*, 2017; Tanguy *et al.*, 2019).

3.2.2 *Geology and Geological Formation*

The valley that forms the Upper Ehen catchment is an archetypal over-deepened glacial trough, which has been carved from Ordovician volcanic rocks of the Borrowdale Volcanic Group. It is typical of the radial pattern of glacial troughs, which characterise the central Lake District (Graham and Hambrey, 2009). The upper valley is formed of Caledonian granitic rocks of the Ennerdale Granophyre whilst mudstone and sandstone (Ordovician Skiddaw Group) flank Ennerdale Water. The landscape surrounding Ennerdale water shows a number of landforms attributed to the recession of the last (Devensian) British Ice Sheet that occupied the region (Graham and Hambrey, 2009).

Glacial moraines are prevalent in upper limits of the catchment (Figure 23) and extend for approximately three kilometres from the head of the valley (Graham and Hambrey, 2009). These have been described as the finest examples of glacial moraines in the Lake District after previous mapping exercises of the upper valley features (Ward, 1873).



Figure 23. Upper limits of Ennerdale Valley and the headwaters of the River Liza showing shallow soils and evidence of glacial features. Taken November, 2019.

3.2.3 *Elevation and Topography*

The Upper Ehen catchment is characterised by steep and rapidly varying topography (Figure 24). Within a UK context, the catchment has a very high average elevation and a range of almost 800m between the highest and lowest points. The maximum elevation in the Upper Ehen can be found at the Eastern end of the catchment at the summit of Great Gable, which has an elevation of approximately 899 mAOD (Figure 24) and extremely steep slopes. The peaks of Pillar (892m) and Kirk Fell (802m) in the southwest of the catchment and Brandreth (715m), Haystacks (582m) and High Crag (704m) encircle the upper valley (Graham and Hambrey 2007). Ennerdale Water dominates the lower part of valley, occupying approximately 3 km^2, with the outflow into the River Ehen found at an elevation of approximately 112m AOD.

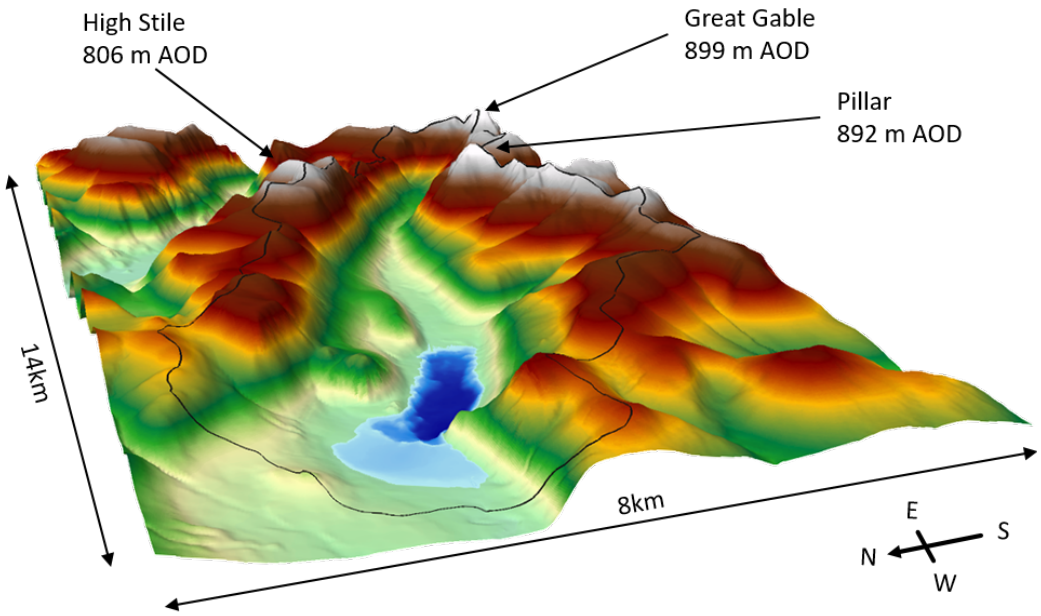


Figure 24. Digital terrain model showing a view of the Upper Ehen catchment (approximately west to east) with relative elevation change, lake depth and principal landmark summits.

The catchment has a DPSBAR value of 323 m/km (NRFA, 2019b). This value is calculated from the inter-nodal values of the catchment DTM and describes the overall catchment steepness. Values greater than 300 are typical of mountainous terrain with the lowest lying areas of the UK having values below 25 (NRFA, 2020). This ranks the catchment as the 10th steepest of approximately 1,500 catchments held within the NRFA database.

3.2.4 *Land cover and vegetation*



Figure 25. View of Upper Ehen catchment from Pillar (892 mAOD) showing land cover variation in the Liza Valley looking West. Photograph taken in November 2019.

A mixture of coniferous and deciduous woodland flanks the lower sides of Ennerdale Valley, upstream of Ennerdale Water (Figure 25). From around 300 – 350 mAOD, woodland gives way to acid grassland. On the steepest slopes, bare rock and scree slopes can be observed. The lower end of the River Liza floodplain has a small area of improved grassland for grazing pasture as do the western and north-western areas of Ennerdale Water.

Grassland in various forms is the dominant type of vegetation in the catchment accounting for over 60% of the surface area (Figure 26). Acid grassland accounts for around half of the vegetation covering the upper catchment, whilst heather/heather grassland and improved grassland account for an additional 15% and 7% respectively (Figure 26). Woodland accounts for around a quarter of the vegetation cover, however this is predominantly managed coniferous woodland that undergoes a regime of planting and felling.

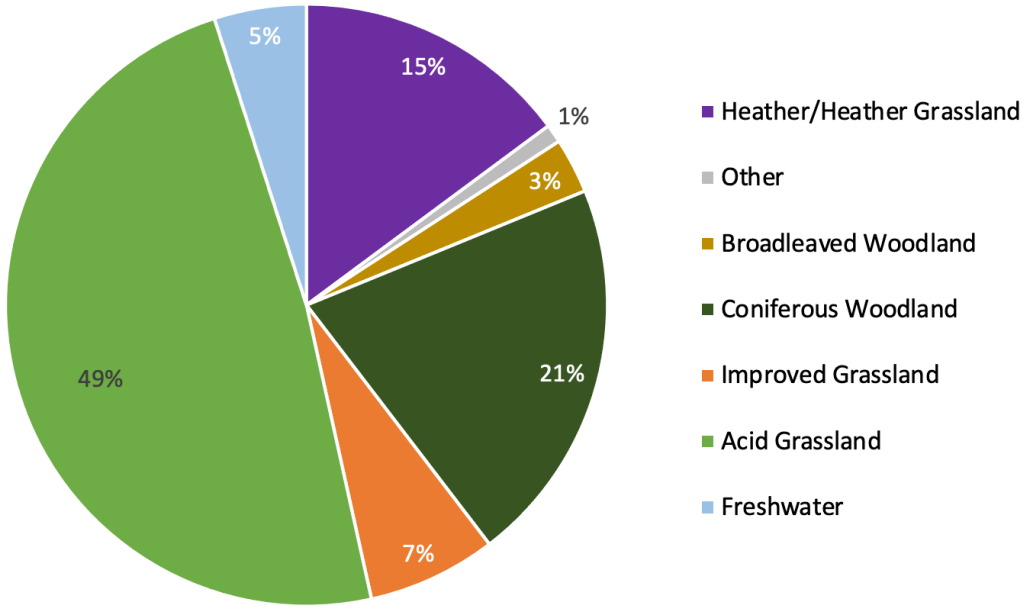


Figure 26. Land cover statistics for the upper catchment from 2015 (Rowland *et al.*, 2017)

3.2.5 *Ennerdale Water – Bathymetry, Abstraction & Infrastructure*

Ennerdale Water has a surface area of approximately 3 km^2 and a maximum depth of around 44 m. Steep sides and a bathtub-like form, typical of the glacially formed lakes of the region, characterize the eastern two-thirds of the lake. The western end of the lake fans out and gently shallows towards the outlet and surrounding lake shoreline.

At weir-crest water level (112.31 mAOD) the lake holds approximately 53,342 Ml of water. Under the current hydrological regime, this equates to a residence time of approximately 228 days (total storage/average annual discharge).

A DEM of the lake was created by combining bathymetric data from survey in 2000 conducted by CEH Wallingford and a more recent higher resolution Bathymetric Lidar Aerial survey of the lake margins and shallowest areas (). Firstly, a shapefile of point data was created from the 2000 survey and then a TIN raster was created in ArcMAP. This was then converted to a .tiff file and resampled from approximately 14m to 1m resolution using cubic resampling. This was then merged with the 1m resolution Lidar data to create a new complete DEM.

This technique means that some data may have been lost in the deeper parts of the lake (from approximately 7m below weir crest down) but the 1m LIDAR data adds greater detail to the upper levels which is the area of interest for modelling. Furthermore, at depths beyond 7m, natural discharge from the lake cannot occur and abstraction is not permitted. Detailed information of the volume of water in this part of the lake is therefore less critical for the aims of this study.

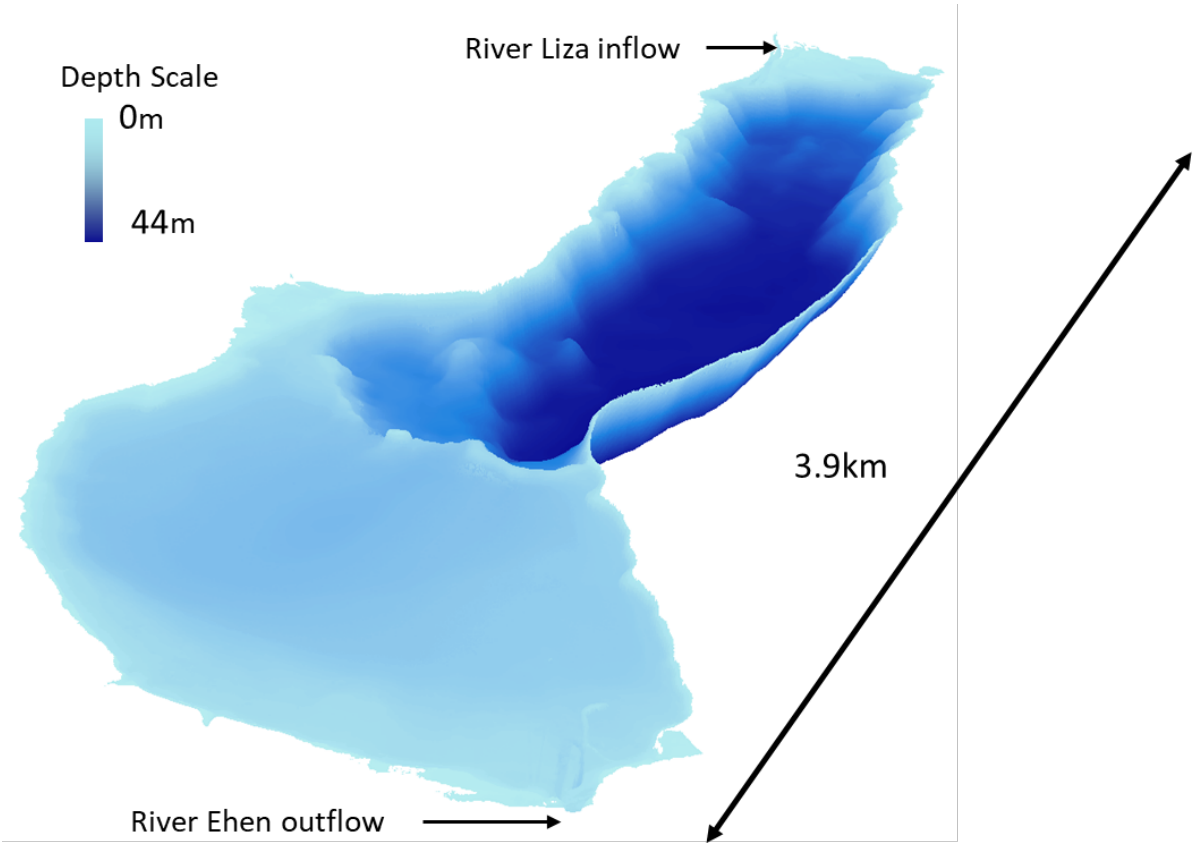


Figure 27. Digital elevation model (DEM) of Ennerdale Water bathymetry made from lidar data and historic survey data. The DEM shows lake bathymetry up to an elevation of 114 mAOD. Darker shades of blue represent deeper areas of the lake. Weir crest level is 112.31 mAOD.

3.2.6 *Catchment Descriptors Summary*

The Upper Ehen has unique catchment properties in both a UK context and when compared to other gauged catchments supporting freshwater mussel populations (Table 5). The average

river channel slope (DPSBAR) is the 10th steepest of all gauged catchments on the NRFA database. Annual rainfall is over double the UK average and around 1000mm more than the average for freshwater mussel catchments.

These high rainfall values are also reflected in the PROPWET value which is an indicator of soil wetness and dependent on climate conditions as well as soil properties. This descriptor indicates that soil is wet (less than 6mm of soil moisture deficit) 71% of the time. The flashiness of the hydrological regime that might be expected in such a steep catchment is somewhat offset by the flood attenuating effect of Ennerdale Water. The catchment has a FARL value of 0.74 compared to a national average of 0.95 (lower values = greater flood attenuation). Land used for arable or horticultural practices is non-existent within the catchment however grass cover is around 50% more than both the UK average and average for freshwater mussel catchments.

Table 5. Catchment descriptors for the Upper Ehen as outlined on the NRFA website compared to averages for gauged freshwater mussel catchment and the UK as a whole.

Catchment Descriptor	Upper Ehen	UK Freshwater Mussel Catchments	UK
Altitude (mAOD)			
Minimum	111	68	66
10 Percentile	123	163	114
50 Percentile	369	337	195
90 Percentile	634	557	305
Maximum	894	933	444
Land Use (%)			
Woodland	14	12	13
Arable/Horticultural	0	5	25
Grass	60	39	41
Mountain Heath/Bog	18	41	17
Urban	0	1	8
Rainfall (mm/year)			
SAAR 41-70	2567	1557	1115
SAAR 61-90	2545	1523	1095
Other			
PROPWET (%)	71	64	46
DPSBAR (m/km)	323	171	93
BFIHOST	0.42	0.44	0.51
FARL	0.74	0.93	0.95

3.3 Catchment Hydrology – Managed Regime

In addition to the natural physical and climatic features of the catchment, the hydrology of the River Ehen is influenced by the abstraction regime and associated infrastructure.

Ennerdale Water has been impounded to provide water supply for the local community since the 1800's. The current weir was constructed in 1995 but the surrounding infrastructure has undergone several modifications to improve its operational flexibility and to limit or reverse any adverse hydrological and geomorphological impacts on the aquatic ecology. The pumping and fish pass control systems and reconnection of the ephemeral tributary Ben Gill are all such measures (Figure 28).

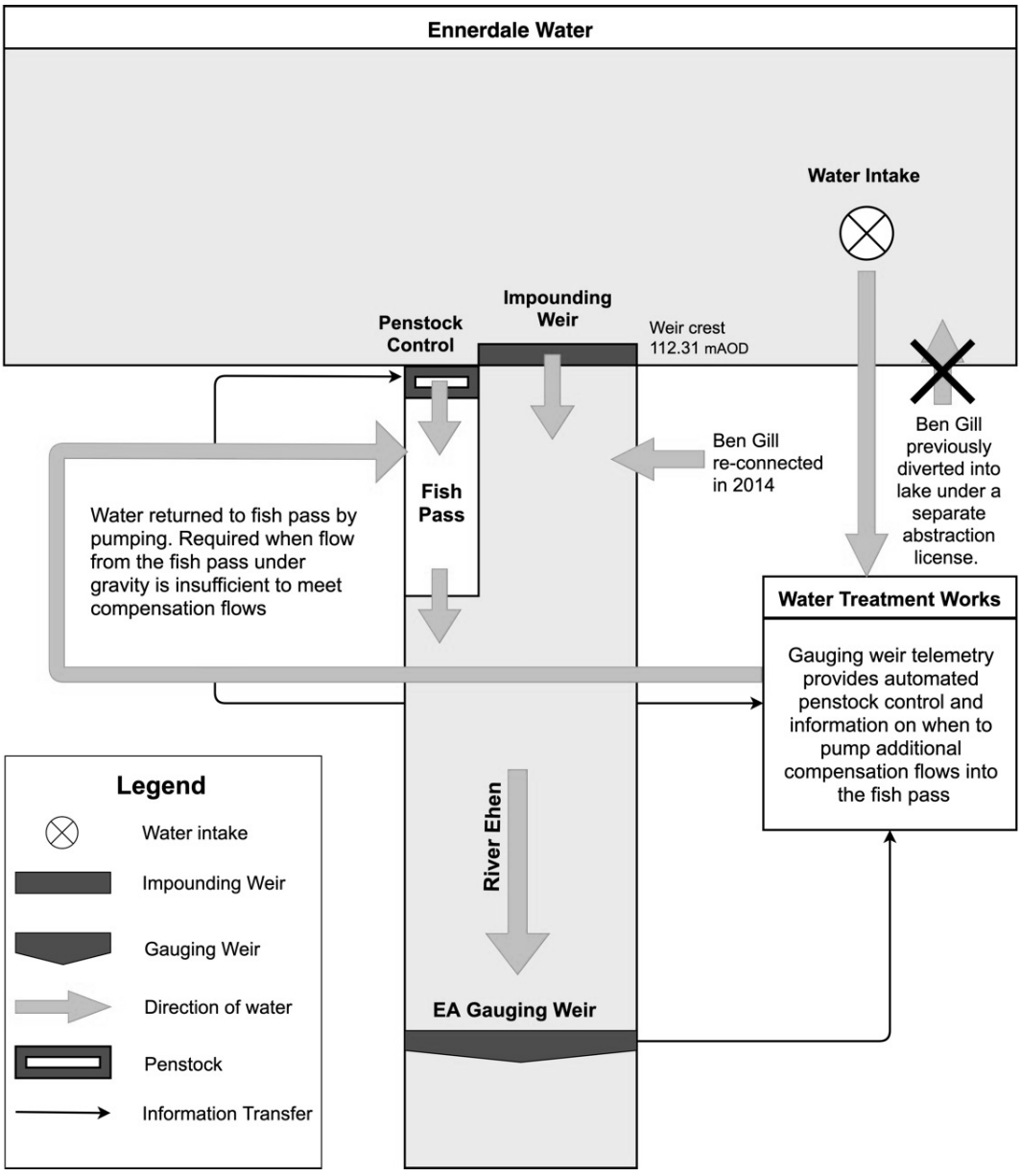


Figure 28. Abstraction, impounding and flow regulating infrastructure currently in operation at the outflow of Ennerdale Water and the upper River Ehen

Water abstraction takes place from the lake however, in severe low flows water can be returned to the fish pass running adjacent to the weir. In times when lake water levels are too low to allow the minimum compensations flows to reach the River Ehen under gravity, additional water can be pumped back from the water treatment works to the fish pass.

The current system of infrastructure allows compensation flows to be controlled more accurately. The previous arrangement involved the manual opening and closing of the fish pass penstock in response to flow readings from the downstream gauging weir. This resulted in lag times between adjustments and prevented the fine tuning of compensation flow releases. To prevent non-compliance with the impounding and abstraction license agreements, compensation flows often exceeded the minimum requirements and thus excess water was released unnecessarily from Ennerdale Water. Computer software currently controls automatic regulation of the penstock meaning that continual adjustments are made as information on downstream flows is fed back to the treatment works operating system. An example of the operation of this system being required occurred in the summer of 2018 (Figure 29). Extended periods of dry weather saw inflows and the lake level continually drop throughout May. By the end of the month additional water was required through the pumping system, further drawing the lake down. Rainfall in June resulted in a small rise in lake levels, however inflows were insufficient to return the lake to weir crest levels or to a height sufficient enough to cease pumped compensation flows.

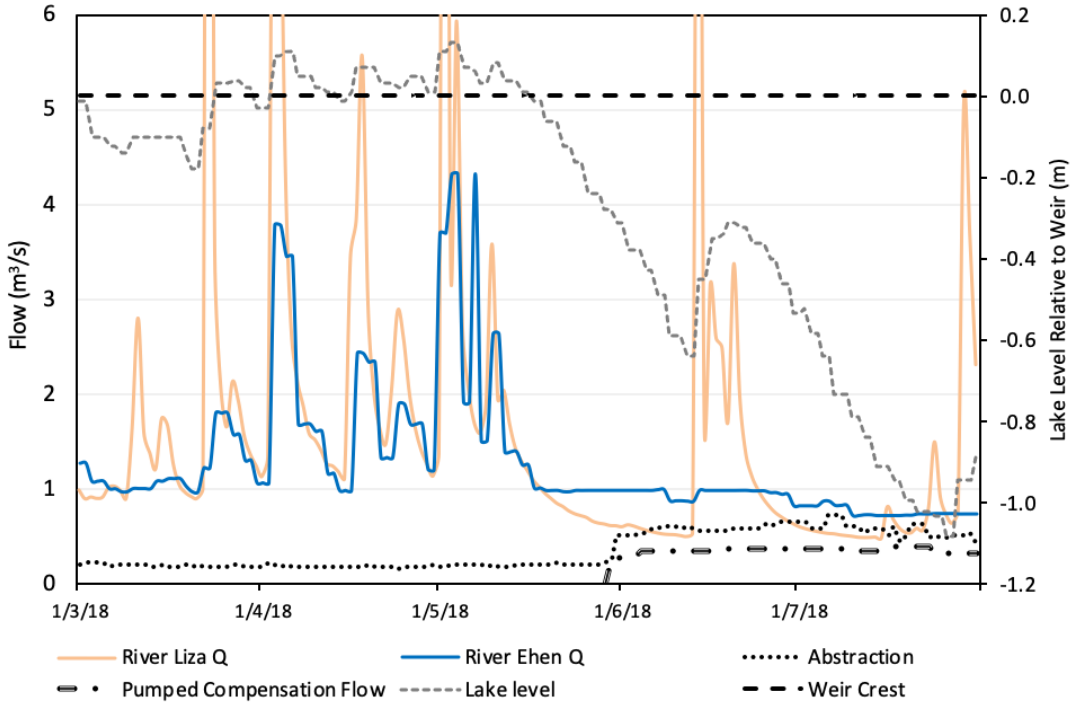


Figure 29. Estimated lake inflows from the River Liza and observed lake level, abstraction, pumped compensation flows and River Ehen discharge at daily timestep over a five-month period from March to August 2018.

Current maximum permitted abstraction rates are set at 3.31 Ml/h which is equivalent to 47.42 Ml/day or 11,338 Ml/year. No abstraction is allowed to take place if the lake level is below 110.61m AOD (1.7m below weir crest). Current compensation flow regulations are set according to the lake level in relation to weir crest. At water levels up to 0.5m below weir crest, a minimum compensation flow of 80 Ml/d must be maintained with a minimum instantaneous flow of 0.93 m^3/s (70 Ml/d). When the lake level is between 0.5 and 0.75m below weir crest compensation flows are set at 70 Ml/d with a permitted minimum instantaneous flow of 0.69 m^3/s (60Ml/d). For all lake levels at 0.75 m below weir crest or lower, daily compensation flows are set at 60 Ml/d with a minimum instantaneous flow of 0.64 m^3/s (55 Ml/d).

During the period from 1995 to 2019 the lake level was at or above weir crest 65% of the time. During the same period, the most frequently observed lake level was between 112.31 – 112.34 mAOD (Figure 30). Regulation of the fish pass and its dimensions mean that flow through this outlet is limited which helps maintain lake levels at or above weir crest most of the time.

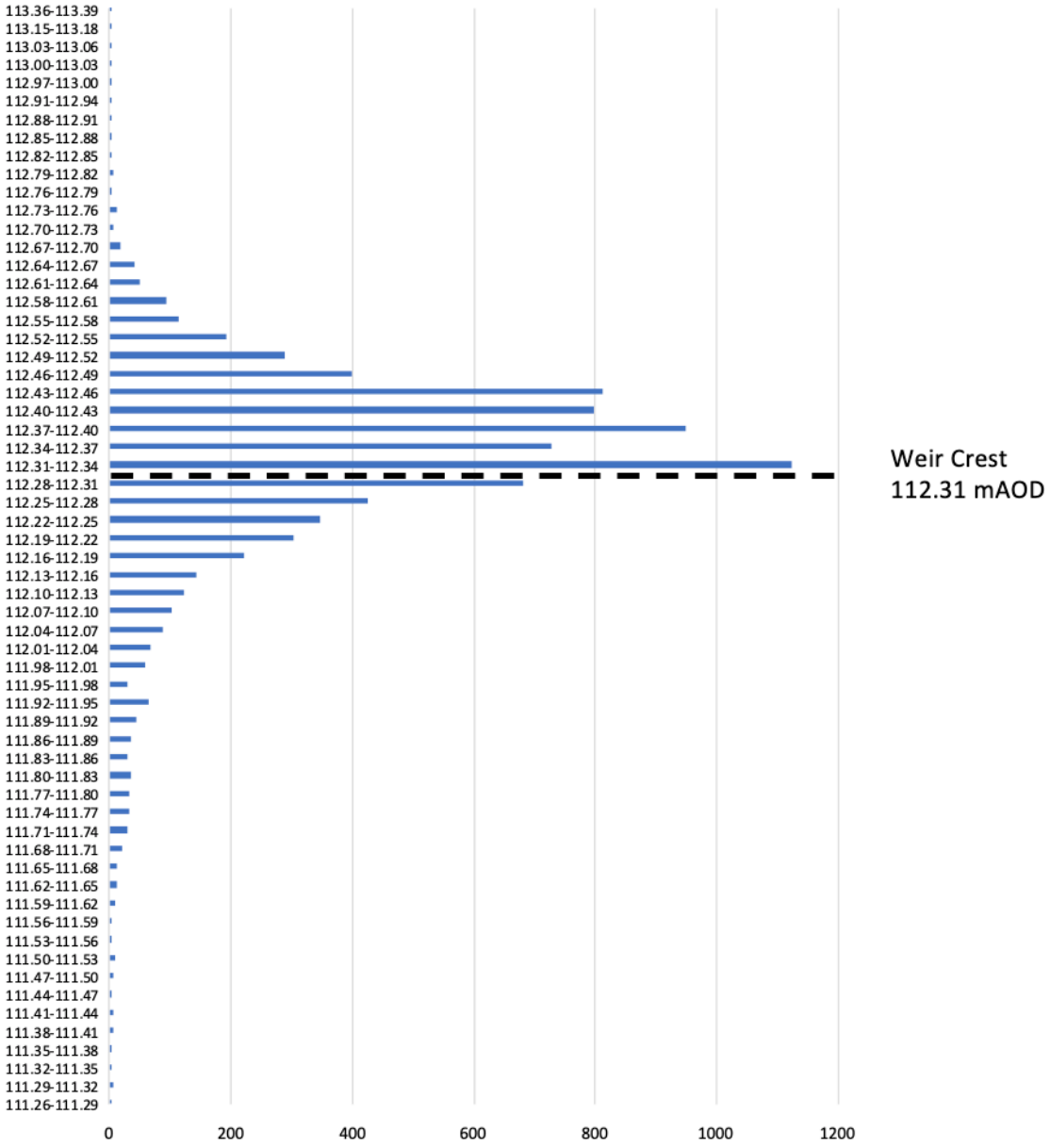


Figure 30. Histogram showing the number of days lake levels were observed at a given stage range for the 1995-2019 period. Lake stage increases in increments of 0.03m (3cm) on the y axis.

Abstraction has steadily decreased during the analysed period (Figure 31) from a daily average of 0.52 m^3/s (45.2 Ml/d) in 1995/96 to around 0.27 m^3/s (23.3 Ml/d) in 2018/19. Mean daily discharge values have gradually increased during the observation period, corresponding with decreasing abstraction and the introduction of compensation flows. The largest observed flow in the discharge time series occurred on 20th November 2009. The instantaneous peak extrapolated from the stage data was estimated to have been around 102 m^3/s on this date (NRFA, 2019b). Rainfall records show that 144 mm fell over the catchment in 24 hours during this event. The daily mean discharge for this date was estimated to be 80.2 m^3/s (Figure 31).

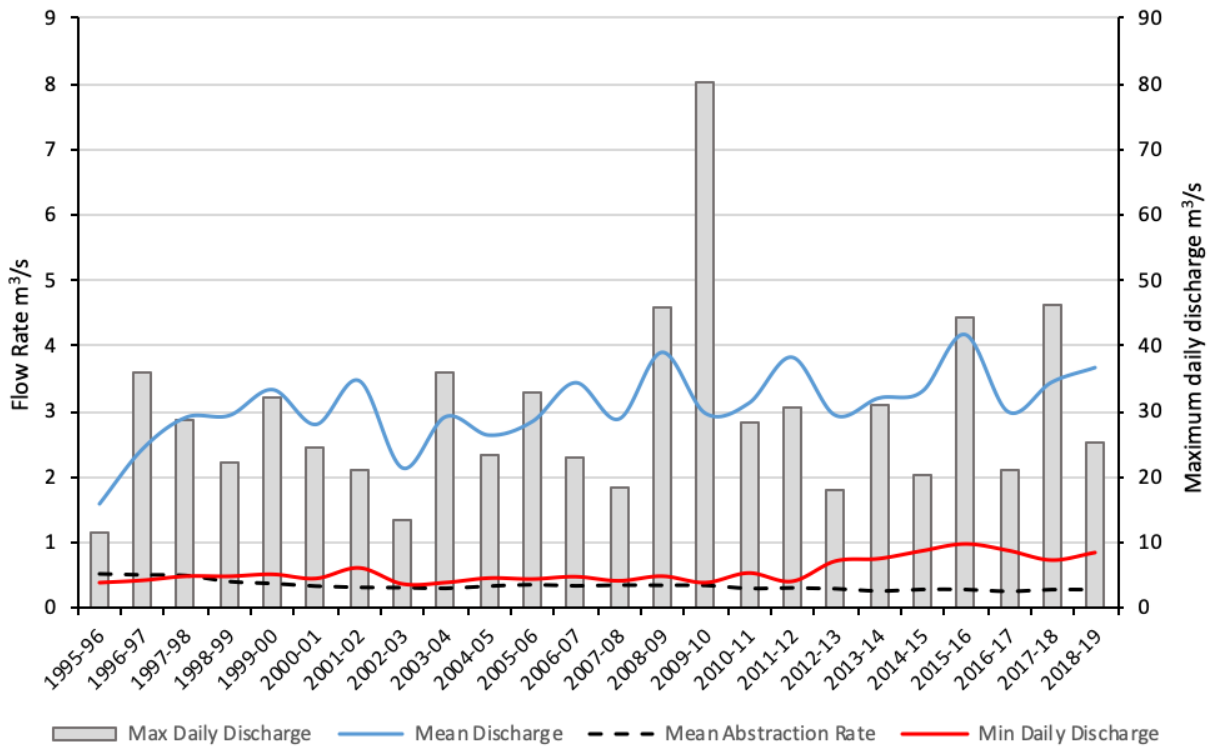


Figure 31. Discharge and abstraction statistics by hydrological year from 1995/96 – 2018/19. Maximum flow values are shown on the secondary axis (values are 10x primary axis).

Based on rainfall data from 1995/96 to 2018/19 annual abstraction accounts for around 10% of the total water balance. Abstraction remains relatively constant throughout the year, meaning that its impact on lake levels and river discharge is most significant during summer as rainfall reduces and PET increases (Figure 32). Mean daily discharge during the period of analysis was 3.1 m^3/s. Highest average daily flows occurred in November and lowest average daily flows occurred in June with discharges in these months averaging 4.5 m^3/s and 1.6 m^3/s respectively. Seasonal analysis of the discharge in the river Ehen using flow duration curves (Figure 33) highlights the impact of flow regulation and management. This is shown by the flattening of the curves when flows in the river approach $1m^3/s$. This flattening is not observed when looking at winter months in isolation as extended dry periods and subsequent flow management are less likely to occur during these periods.

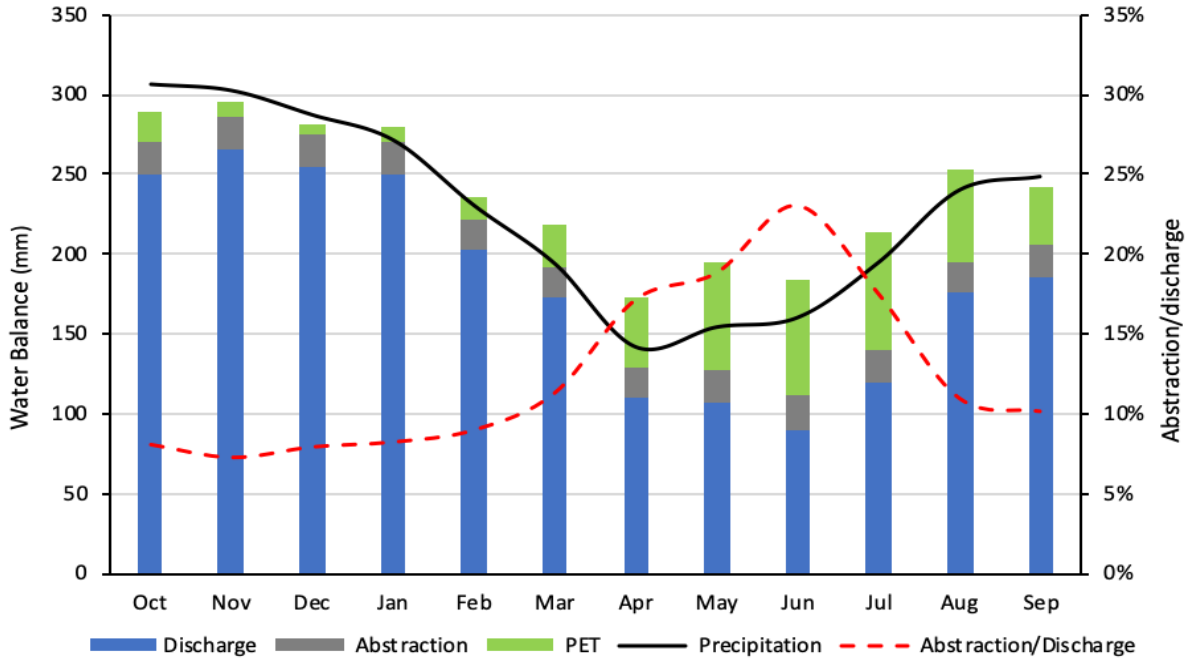


Figure 32. Average monthly values in mm for discharge, abstraction, precipitation and potential evapotranspiration for hydrological years 1995/96 – 2018/19. Monthly values for abstraction also shown as a percentage of discharge on secondary axis. PET data is up to 2015 only.

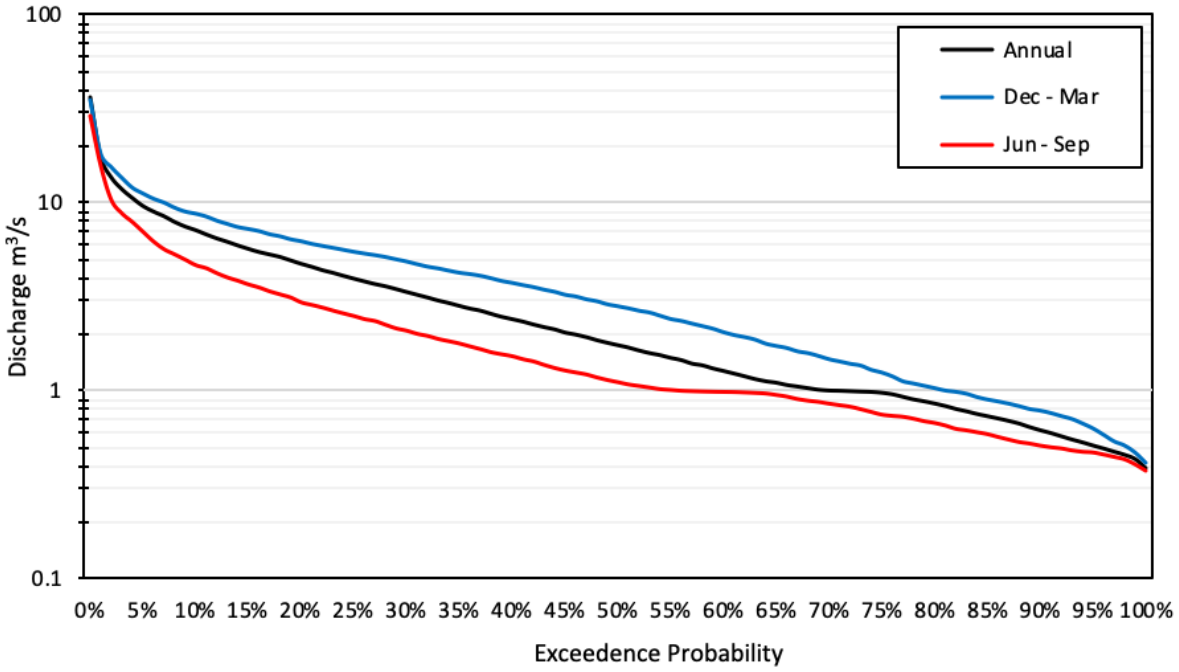


Figure 33. Annual and seasonal flow duration curves for Bleach Green gauged flows from 1st October 1995 to 30th September 2019.

Comparing the hydrology of the Upper Ehen with a hydrologically unimpacted site provides further illustrative evidence of the influence that abstraction and flow regulation exert on its hydrological characteristics. The River Irt drains a neighbouring catchment with very similar physical characteristics to the Upper Ehen. The Irt has a small amount of abstraction for private consumption but no flow regulation structures on its lake outlet and thus provides a

useful example for hydrological comparison. Both catchments drain an area of just over 44 km^2. Maximum elevation and annual rainfall are both slightly higher in the Irt at 937 mAOD and 2,629 mm respectively compared to 893 mAOD and 2,542 mm in the Upper Catchment. Mean flows for the period from 1/10/95 to 30/9/19 are 3.69 m^3/s in the River Irt and 3.1 m^3/s in the Ehen.

The Upper Ehen has a flashier hydrological response which is shown by the higher flood peaks and steeper, lower recessional limbs on the hydrograph (Figure 34). The flow duration curve for the River Ehen is also steeper, however it levels off at around 1 m^3/s which is when the lake level falls below weir crest and flow is only possible through the fish pass. The lower end of the FDC is also artificially elevated by the compensation flow releases.

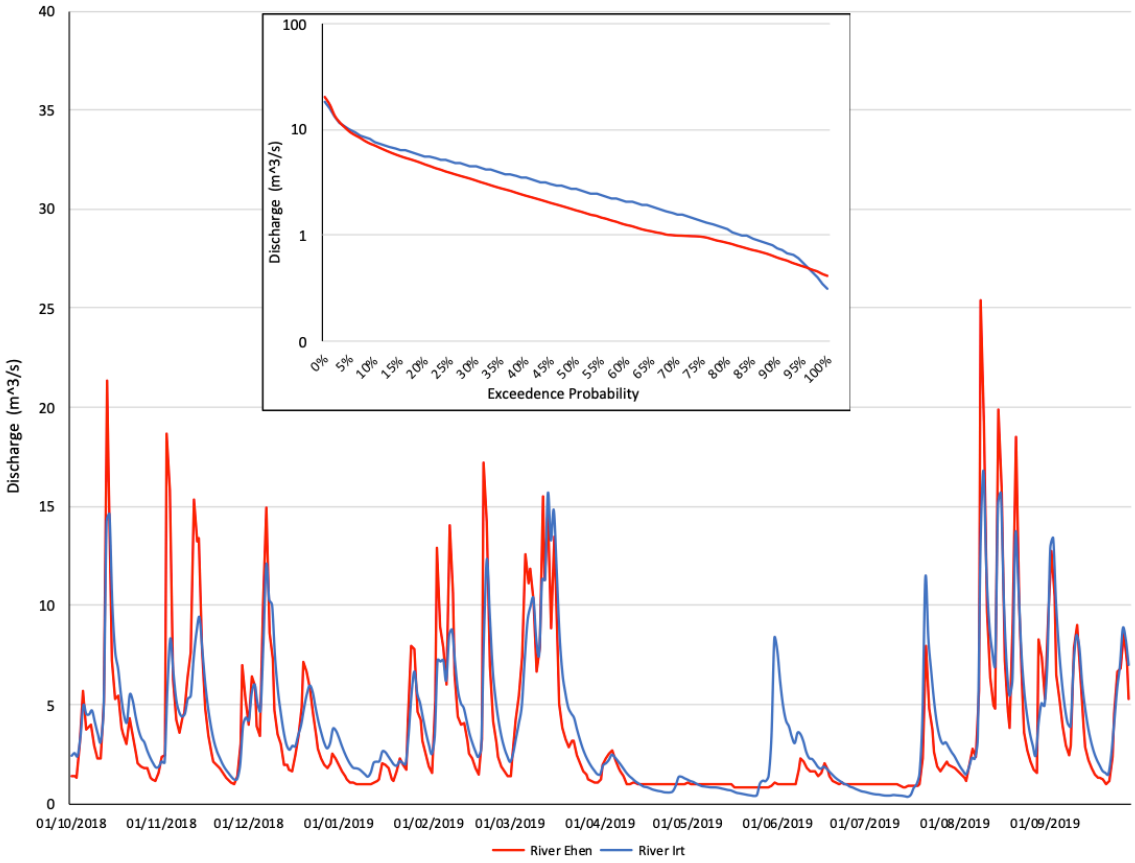


Figure 34. Example hydrograph for the hydrological year 2018-19 for River Irt (blue) and the River Ehen (red). Inset: Flow duration curves for River Irt (Blue) and the River Ehen using data from 1/10/95 to 30/9/2019.

The discharge time series from 1st October 1995 to 30th September 2019 was used to generate ecologically relevant streamflow metrics. These provide a basis for comparison between other freshwater mussel rivers and a means of quantifying any projected changes under unmanaged flow regime scenarios. The Hydrological Index Tool outlined in Chapter 2 was used to generate metrics for the 24-year period. The key metrics that were identified in this national-scale study are shown here (Table 6) whilst the full list of calculated metrics and

definitions is given in Appendix A. These metrics are also used as a means of testing the modelled discharges in Chapter 4. In the initial pooled analysis only 12-years of data were used to generate flow regime metrics. Although sufficient, this provides less robust information about the hydrological character of a catchment.

Table 6. A comparison of metrics identified in the national-scale study of flow regime using 12-year and 24-year discharge timeseries.

Indicie	Flow Regime Component	Flow Condition	12 years	24 years	Numerical Change	% Change
ma11	Magnitude	Average	2.28	2.6	0.325	14%
ma36	Magnitude	Average	3.86	4.2	0.343	9%
ma42	Magnitude	Average	0.42	0.9	0.481	115%
ml13	Magnitude	Low	54.74	55.6	0.856	2%
ml22	Magnitude	Low	0.01	0.01	-0.006	-46%
mh16	Magnitude	High	3.96	4.1	0.142	4%
mh22	Magnitude	High	7.44	8.2	0.762	10%
fh4	Frequency	High	9.00	9.7	0.700	8%
dl9	Duration	Low	24.88	26.3	1.419	6%
dh13	Duration	High	4.47	4.5	0.031	1%
dh19	Duration	High	1.43	1.5	0.065	5%
ta1	Timing	Average	0.12	0.10	-0.022	-18%
ra5	Rate of Change	Average	0.35	0.4	0.050	14%

Using a longer time series for generating the hydrological metrics had most impact on two hydrological metrics that measure magnitude, ma42 and ml22. These changed by 115% and -46% respectively. A longer time series is more likely to capture a greater range of flows and changes to these flow metrics can thus be attributed to the inclusion of this extra data. Additionally, the management of compensation flows and regulation of the fish pass have changed over time which will also have impacted the metrics. Changes to the values of the remaining metrics did not exceed ± 18%.

3.4 Upper Catchment Hydrological Monitoring

Due to the longstanding presence of water infrastructure at Ennerdale Water little is known about the natural hydrology of the upper catchment. Monitoring was conducted to capture and constrain the unmodified (or least altered) hydrological behaviour of the upper catchment. The additional hydrological data, as well as providing a better understanding of catchment processes, is used for calibrating and improving the performance of hydrological models. Without the additional data the model can only be calibrated against data at the catchment

outflow leaving multiple possible scenarios as to what is happening in the rest of the upper catchment. By collecting data from multiple locations, the uncertainty of the model performance can be greatly reduced thus giving greater confidence in the model outputs.

3.4.1 *Monitoring Network*

A preliminary desktop survey of the area was conducted to select a number of potential areas and sites for locating the monitoring equipment. This was based primarily on the hydrological characteristics of the catchment but also on practical and logistical criteria such as accessibility, ease of installation and risk of damage or theft.

A scoping visit carried out at the end of August 2017 allowed for potential sites to be visually inspected and to verify their suitability for installing monitoring devices and returning at regular intervals to maintain equipment and collect any recorded data.

The River Liza was identified as a key monitoring location due it being the main inflow to Ennerdale Water. However, several characteristics of this river made finding a suitable monitoring location within the main channel challenging. Firstly, the size of the river meant that it would have be difficult and dangerous to carry out flow gauging in the channel during high flows. Secondly, in several locations the bed appeared to be highly mobile and as such obtaining a constant rating curve during the course of the hydrological year could have been difficult.

The two locations selected within the River Liza were chosen to capture the hydrological behaviour in the valley headwaters and the entire valley as close to Ennerdale Water as feasibly possible. Monitoring too close to Ennerdale Water could have meant that, in times of extremely high lake levels, the flow and stage within the river channel could have been impacted by the backing up of water. The lower of the two sites was thus set sufficiently upstream from the lake as well as meeting the criteria of accessibility, safety and channel stability.

In addition to the River Liza, three sub catchments were identified as suitable locations to carry out monitoring. The rationale behind their inclusion was to have some built in redundancy (Woundell Beck & Upper Liza) within the monitoring programme and to capture the behaviour of different areas of the catchment (Smithy Beck & Gill Beck).

Carrying out monitoring in multiple sub catchments was a desirable option that despite being more costly and time consuming had several benefits over solely monitoring the flow of the main Liza channel near its inflow into Ennerdale water. Firstly, multiple monitoring sites

spread the risk of equipment loss or failure meaning that the full period of monitoring was more likely be covered in at least some if not all parts of the upper catchment. Secondly, it allowed for a representative spread of different sub catchments in the upper valley to help gain a better understanding of the hydrology of the catchment. Thirdly, multiple data points allow for better calibration of the SHETRAN model used in Chapter 4. After an initial period of river and rainfall monitoring, additional shallow groundwater monitoring sites were added to the network.

3.5 Monitoring Techniques

The following subsection provides details of the instrumentation used for the collection of meteorological and hydrological data. The justification for their use and limitations of each are also outlined.

3.5.1 *Rainfall Data*

Rainfall distribution, intensity and amounts are known to vary greatly in heavily undulating catchments (MALBY *et al.*, 2007). Orthographic uplift can intensify and amplify rainfall amounts in upland areas (MALBY *et al.*, 2007), whilst areas on the leeward side of upland regions can experience reduced amounts of rainfall . Collecting additional rainfall data from strategically sited rain gauges can help provide further information on the effects of topography and climatic characteristics.

Three rain gauges were installed in the catchment to monitor rainfall during the study period and to extend existing rainfall data sets. Aerodynamic tipping bucket rain gauges designed and manufactured by EML Ltd to reduce the effect of under catch were used. In exposed upland sites, rainfall amounts can be underestimated by up to 23% on average (Pollock *et al.*, 2018). Rain gauge tipping buckets are designed to record 0.200mm of rainfall on each tip. In the rain gauges provided, the buckets had been pre-calibrated, and the final values ranged from 0.197mm to 0.199mm per tip. Limpet Logger data recorders, set to log individual tips with a date time stamp, were installed in all gauges. The advantage of this method over recording cumulative tips at 15 minute or hourly intervals is that rainfall intensity can be observed at a finer detail and data can later be aggregated to the desired time steps. Level loggers may have limited storage space however and recording each tip may cause the memory to be used up more quickly than recording cumulative totals.

A known limitation of tipping bucket rain gauges lies in their ability to accurately record intense rainfall events. In extreme rainfall intensities, the bucket may overfill slightly on each tip, meaning that more rainfall has fallen than shown by the recorded tip. However, quality control checks can be implemented in the data processing stages to identify data collection issues. At the most remote site an additional monthly manual rain gauge was installed (Figure 35) as a means of validating the cumulative totals over extended periods. This gauge was also telemetered to allow for easier surveillance of data collection and potential equipment failure. A thermometer was also installed at this site as a means of differentiating tips generated by rainfall and snowmelt.

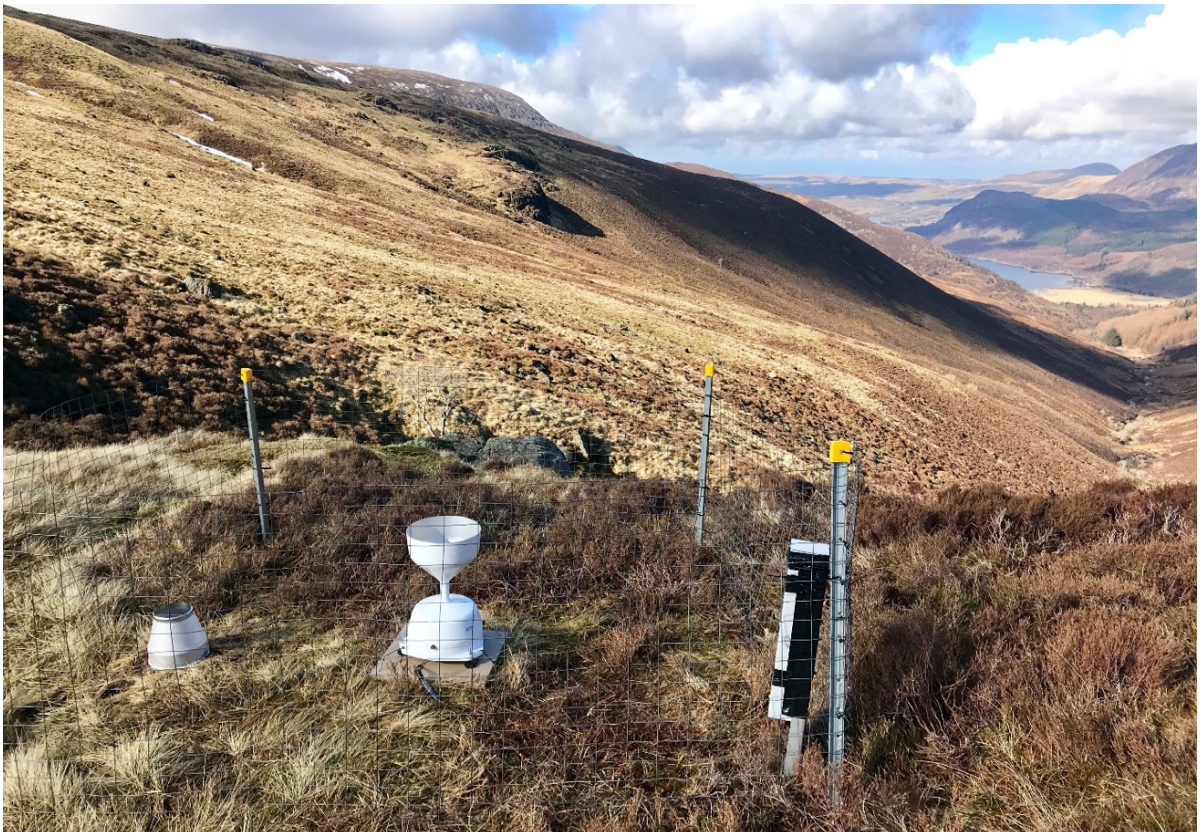


Figure 35. Rain gauge setup at Great Cove Sheepfold upland site - 500m AOD. A manual monthly gauge (left) was used to minimise the risk of data loss prior to the installation of a telemetered system (right).

In addition to the potential for undercatch in exposed upland sites, tipping buckets may also underestimate rainfall in extremely intense rainfall events. Water entering the gauge and filling the bucket may be lost during each tip in extreme cases.

3.5.2 *River Stage*

River stage was used as a proxy for measuring river discharge. The remoteness of the sites, challenging and changing stream conditions, the need to minimise physical and visual impact

on the landscape and prohibitive costs of alternative measures meant that direct continual discharge measurements were not possible.

Solinst Levellogger® Junior Edge Model 3001 (M10) pressure transducers were used at all sites. Able to record water depths of up to 10m, they have an accuracy of ± 5mm (Solinst Canada Ltd., 2015). The loggers have a storage capacity of 40,000 readings and were set up to take measurements every 15 minutes meaning over a year of data could be stored if required. However, data was downloaded more frequently to reduce the risk of large data losses.

Loggers were fixed in position using metal fencing stakes, PVC tubing cable ties, Kevlar string and padlocks in the more exposed sites (Figure 36). The PVC tubing was fixed to the fencing stake in a position slightly off the riverbed to allow water and sediment to leave the stilling well created using the tubing and to prevent any blockages. Additional holes were drilled into the tubing to allow the water level to rise and fall freely and always correspond to the river level. Cable ties positioned across the bottom of the PVC tubing ensured that the level logger was always returned to exactly the same position after removal for data downloading. The gap between the riverbed and PVC tubing allowed the correct siting of the pressure logger to be verified by hand.

Loggers were located in sites that were relatively accessible but discrete to limit the possibility of tampering. They were also positioned away from the main flow of the river channel to prevent damage from objects mobilised in high flows but located in a part of the river channel that would most likely be submerged even in low flows. As the full range of flow conditions had not been observed prior to installation this required some judgement based on in-field observations.

In addition to the submerged pressure transducers, barometric pressure transducers were used to compensate the recorded stage data. These were again set up to record measurements every 15 minutes to correspond with the level loggers. The Solinst Levlelogger Edge 3001 transducers came with a lifetime factory calibration and a full-scale accuracy of 0.01% (Solinst Canada Ltd., 2019). A single pressure transducer was used to compensate the data however a second transducer was also installed for redundancy in case there were any technical issues with recording the data and as a means of assessing the consistency of measurements.

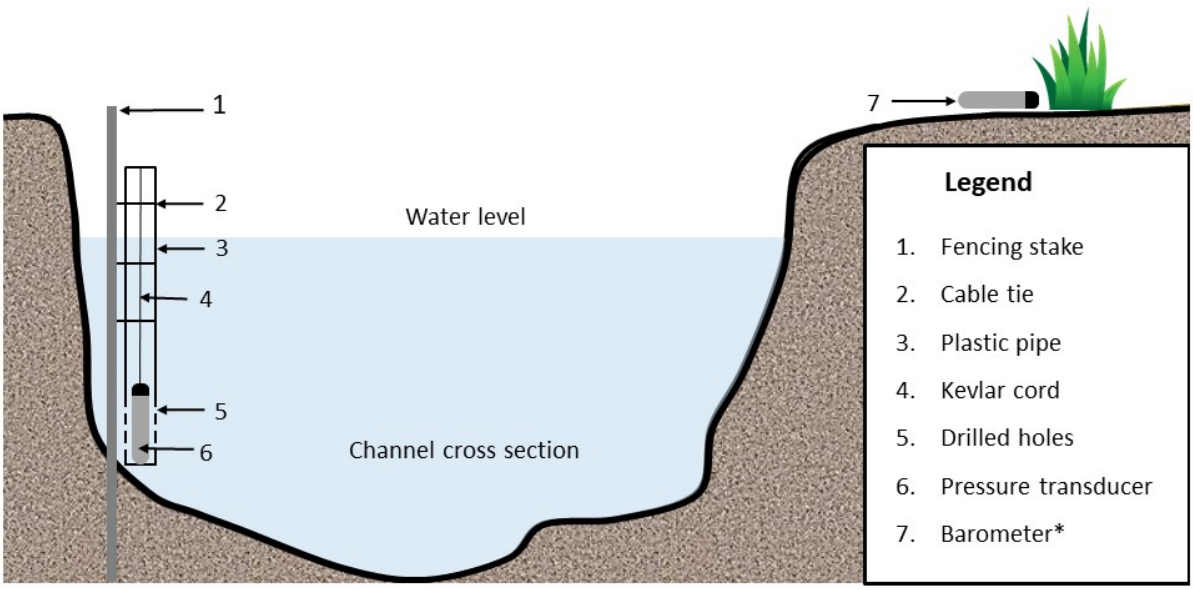


*Figure 36. Standard level logger setup at each sub catchment site (*two barometers covered the entire study area).*

Stage data was gathered from five sub catchments covering a total area of 30.7 km^2 (Table 7). This accounts for approximately 75% of the total land surface of the Upper Ehen catchment (69% of total catchment area). Although predominantly acid grassland, the varying size, topography and locations of the catchments allowed for a range of different hydrological characteristics and processes to be observed.

Table 7. Areas and physical characteristics of the sub catchments where river stage and flow were monitored in the upper catchment.

Catchment Name	***Area*** (km2)	***Max Elevation*** (mAOD)	***Land Cover*** (Type)	(%)	***Channel Width*** (m)
Gill Beck	1.2	440	Acid grassland	83	2
Smithy Beck	3.4	633	Coniferous Woodland	40	4
Lower Liza	11.1	806	Acid grassland	48	14
Woundell Beck	5.3	797	Acid grassland	59	7
Upper Liza	9.7	893	Acid grassland	77	10

3.5.3 *Stage Discharge Rating Curve*

To obtain a discharge series from the recorded stage a rating curve was established for each of the five sites. Discharge measurements were taken at each site under a range of flow conditions. To accurately measure the flow, salt dilution was chosen as the preferred method. Tracer methods such as salt dilution are often the only accurate method of estimating flow in rocky shallow streams (Herschy, 1995).

Salt dilution gauging requires the preparation of a salt solution of known concentration. This is then added to the stream or river to be gauged. The subsequent change in specific

conductance is then recorded over time until levels return to the background readings observed prior to adding the salt solution (Figure 37).

Guidelines on the concentrations and quantities of salt required vary depending on the flow conditions and the characteristics of the site. In this instance 1kg of salt was used for every 7 litres of river water. For the highest recorded flows around 10kg of salt were used. The maximum gauged flows are therefore limited by the physical ability to mix sufficient quantities of salt solution and inject them into the river or stream in a safe manner.

To ensure the technique works, sufficient distance between the point of adding the solution and the conductivity probe is essential. A recommended distance is 25 times the width of the channel (Day and Day, 1977) in the sampling reach. This distance may vary depending on the morphology of the site and the flow conditions at the time of gauging (Day, 1976).

Two 500ml bottles of river water were taken prior to injecting the salt solution into the main flow of the channel. A further two 50ml bottle of the salt solution were obtained for later laboratory calibration of the salt dilution process.

The conductivity probe was set up to take a reading every two seconds and a note was also made of the background reading to ensure that the recording was only stopped once conductivity had returned to background levels.

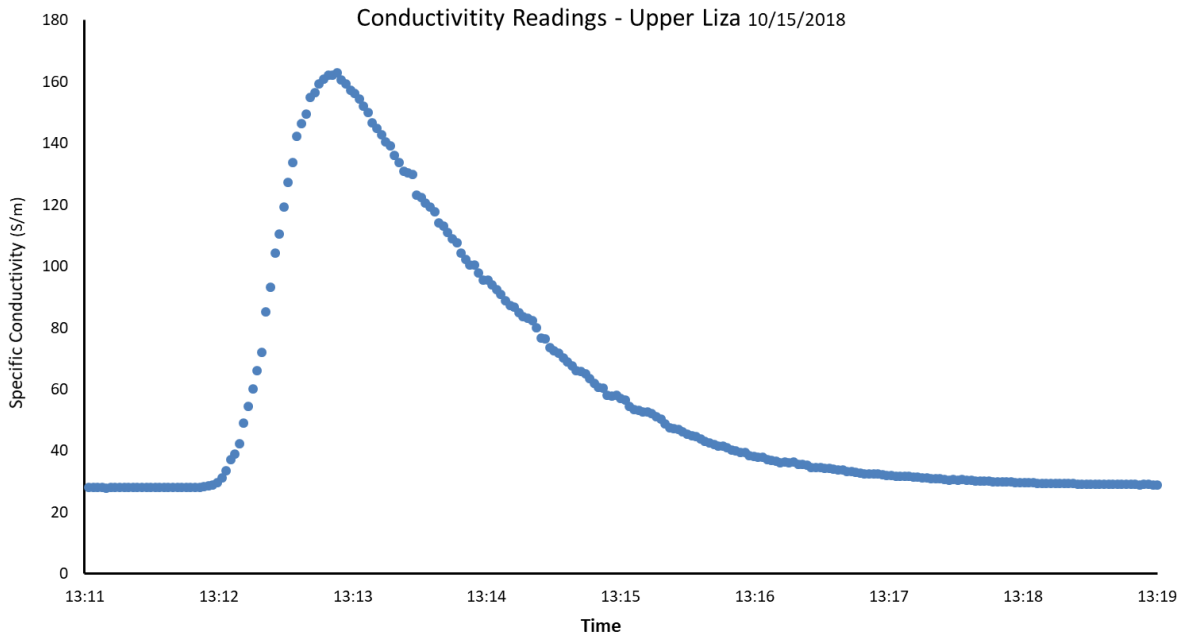


Figure 37. Example of the conductivity spike observed in salt dilution gauging from which river discharge is calculated.

Due to ecological considerations of the site, spot gauging was restricted to the months of May to October. The Upper Ehen Catchment is home to Arctic Char (*Salvelinus alpinus*) which

spend their lives in Ennerdale Water and swim upstream into the River Liza to spawn during November and December (McCubbing *et al.*, 1998). After this time access to the river was avoided to prevent possible disturbances to the redds and the development of the eggs. This is believed to be the last remaining UK population of Arctic Charr that has retained the trait of swimming upstream to spawn and is one of the rarest freshwater species in the UK.

Research into the impacts of salt dilution gauging on benthic macroinvertebrates has concluded that long-term detrimental impacts are unlikely (Wood and Dykes, 2002). However, the impact of the technique on the unhatched eggs and fry of the Arctic Char is unknown. The decision not to carry out gauging during the spawning period was thus a precautionary one. This also avoided the possibility of spawning grounds being disturbed when entering the river.

On each flow gauging date, samples of both the river water and salt solution added for gauging were taken. These were then used to perform a site-specific calibration of the gauged flows. The calibration procedure involves continually adding a known volume of salt solution to a given volume of river water and recording the step changes in conductivity (Figure 38). A 'K' factor is then obtained from the gradient of the slope brought about by the change in conductivity. This then used to adjust the final flow measurement.

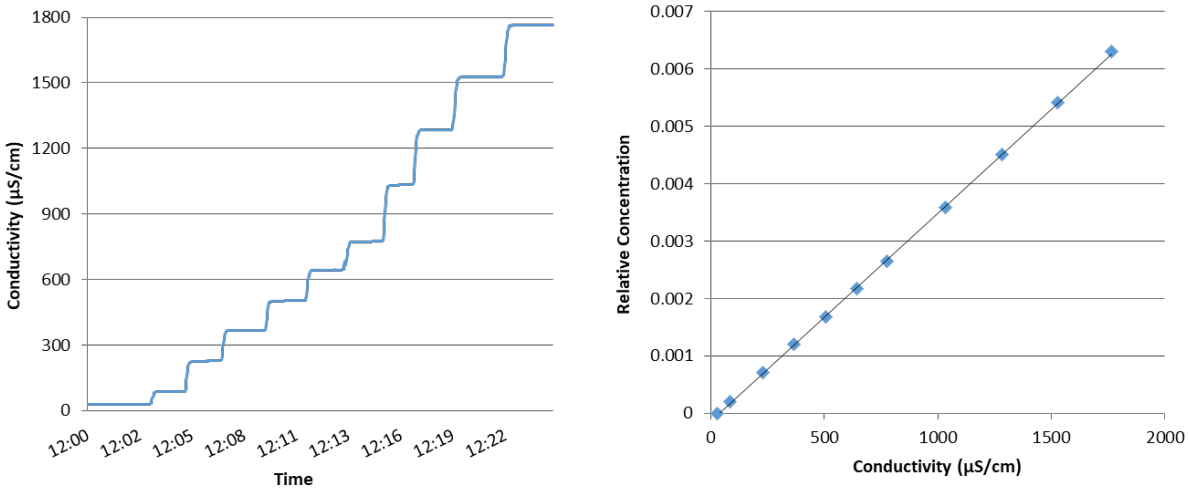


Figure 38. Results of laboratory calibration process and K factor results for Upper Liza – 15/10/2018

Rating Curve construction

Due to time and logistical constraints, the number of successful spot gauging surveys at each site was limited. The channel geometry and stability at Gill Beck was impacted upon by a heavy rainfall event on the 11th of September. Later spot gauging measurements at the site highlighted inconsistencies in the stage discharge relationship after this date (Figure 39).

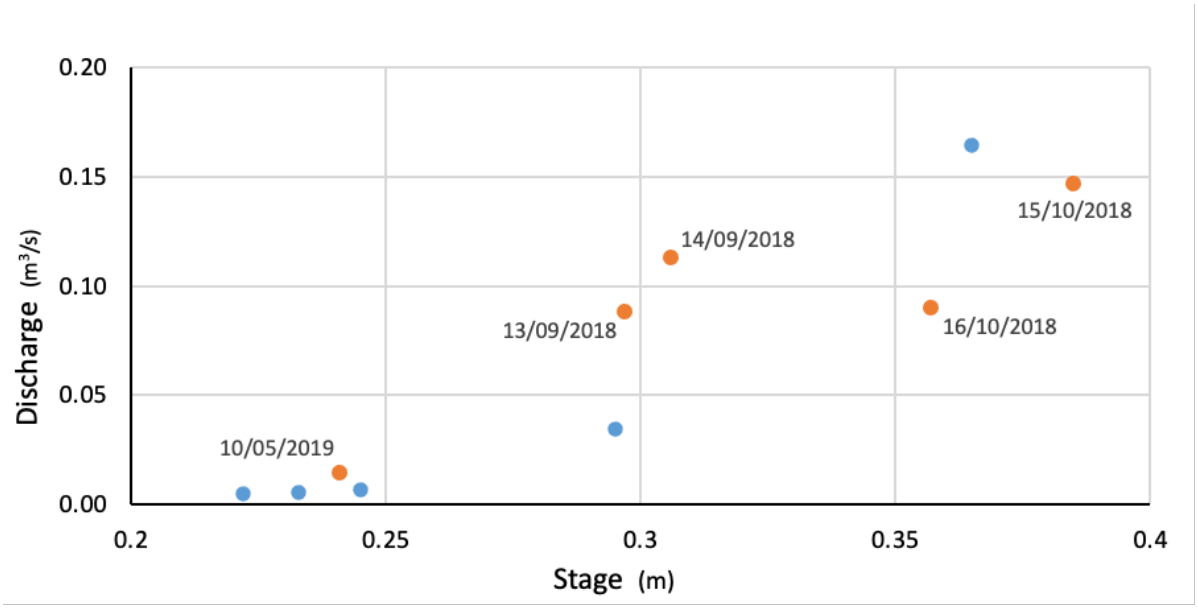


Figure 39. Results of salt dilution gauging from Gill Beck. Points shown in orange highlight changes to the stage discharge relationship after the September 2018 high flow event.

Stage measurements continued to be taken at the site for the remaining monitoring period. The stage readings were compared (Figure 40) for the pre- and post-high flow event to assess the level of change experienced in the channel. On average, stage readings were 3.5 cm lower (2.6 – 4.4 cm range) for the remaining period of observation, however, it was not possible to ascertain the length of time taken for the channel to stabilise after this event.

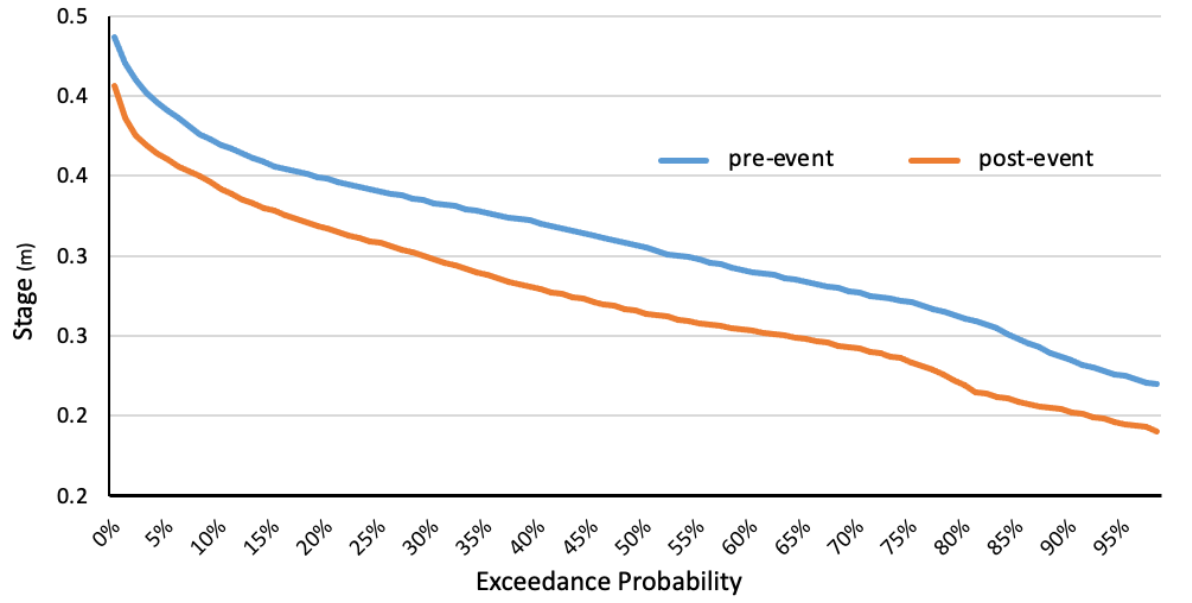


Figure 40. Stage duration curves for the Gill Beck site with data collected prior to and after the channel disturbance showing the shift in stage caused by channel adjustments after a high-flow event.

Attempts to gauge higher flows at the Lower Liza site also proved difficult. On later analysis, the cause of the unsuccessful gauging attempts was attributed to insufficient distance between the conductivity probe and the injection of the salt dilution. This stretch of the Liza has

relatively uniform geometry and in high flows, the water moves in a less turbulent manner, reducing the mixing of the salt solution. Despite these setbacks, a range of flows were still captured within the spot gauging campaign (Table 8).

The channel at each gauging location was surveyed to obtain a channel cross section and the average slope of the river. Surveys were conducted using a Leica Jogger 20 Automatic Level, tripod and levelling staff. A reference height for the level logger in relation to the streambed was also obtained for each site. This allowed for true river stage adjustment to be made and the complete stage time series to be obtained at each site. Information on channel geometry and slope enabled estimates of channel roughness (Manning's n) to be made for each gauged flow depth.

Table 8. Recorded flows and corresponding stages for each site. Numbers in red indicate unused data due to errors in the field procedures or changes to the site and channel geomoetry.

Date	Gill Beck		Smithy Beck		Lower Liza		Woundell Beck		Upper Liza	
	Stage	Flow	Stage	Flow	Stage	Flow	Stage	Flow	Stage	Flow
31/05/2018	0.21	6.8	0.15	27.6	0.18	177.5	0.11	32.6	0.14	102.1
29/06/2018	0.20	5.5	0.14	21.8	0.19	238.5	0.11	41.4	0.17	134.3
16/07/2018	0.18	5.0	0.14	20.0	0.23	592.2	0.14	80.9	0.20	212.0
15/08/2018					0.51	2941.4	0.32	863.2	0.49	1381.9
16/08/2018	0.33	164.7	0.29	414.0	0.58	NR	0.36	1306.8	0.53	1889.2
29/08/2018					0.33	1022.8				
30/08/2018	0.26	34.3	0.21	105.0			0.18	156.0	0.26	377.3
13/09/2018	0.26	87.9	0.27	246.0	0.40	2725.4	0.25	344.1	0.36	898.1
14/09/2018	0.27	112.9	0.27	269.9	0.56	1669.5	0.32	642.1	0.51	1031.0
15/10/2018	0.35	146.7			0.46	2146.0			0.39	920.9
16/10/2018	0.32	89.7	0.27	249.4			0.24	314.8		

3.5.4 *Shallow Groundwater*

Two piezometers were also installed part way through the monitoring campaign. This was done to support an MSc project and also gain additional information on the physical properties and hydrological behaviour of the catchment. After an initial desk-based study to identify suitable locations for their installation, piezometers were constructed on a trial-and-error basis as access to land, equipment and appropriate conditions for installation were limited.

Crude piezometers were made using PVC plastic tubing with holes drilled in the bottom 30cm. this was then protected using cotton fabric to allow water to enter the piezometer but prevent infilling and clogging from sediment. A Cobra percussion corer (Figure 41a) was used to drill holes for the piezometers. In each location the maximum depth of the piezometer was constrained by the composition of the substrate and the water level at the time. Upon reaching saturated substrate the hole would collapse upon itself preventing excavation to greater depths.

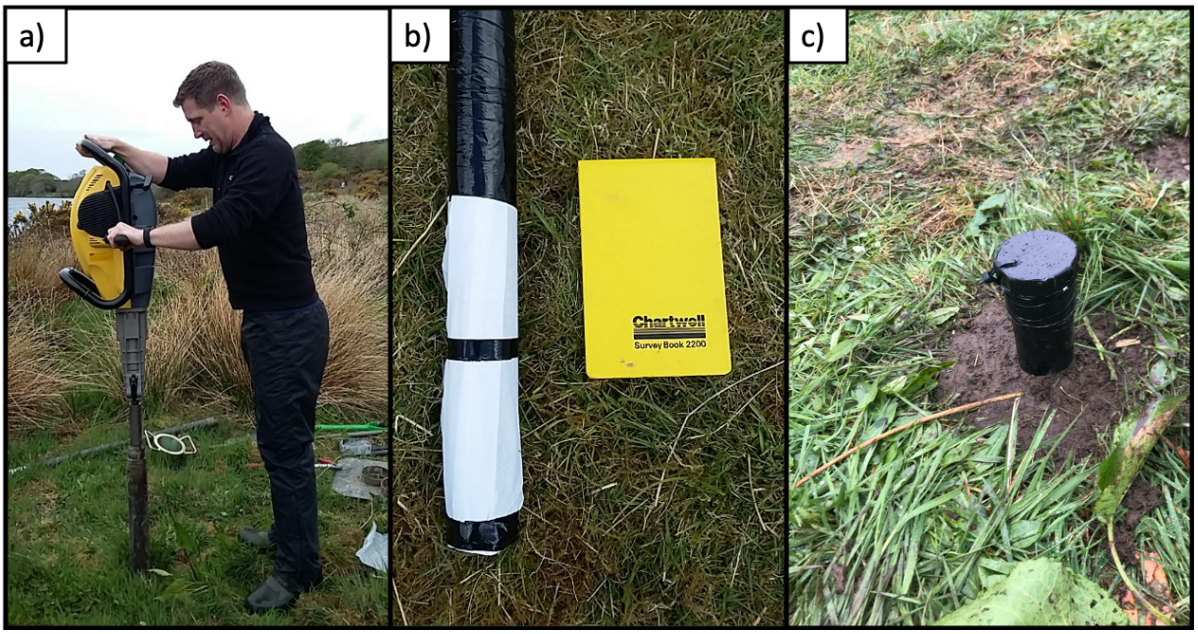


Figure 41. Equipment and materials used for piezometer installation (a & b) and final piezometer setup at the lake shore site.

Two piezometers were successfully installed. At site one, approximately 5 metres from the edge of Ennerdale water and site two at the lower end of the Liza Valley (Figure 43). Piezometers were installed to depths of 1.54m and 0.94m below ground level at sites one and two respectively (Figure 42).

Solinst® Levelogger Junior Edge pressure transducers were again used and setup to record readings at 15-minute intervals in sync with the other loggers in the catchment. These

readings were then compensated using the barometric readings in the same way as the river stage measurements.

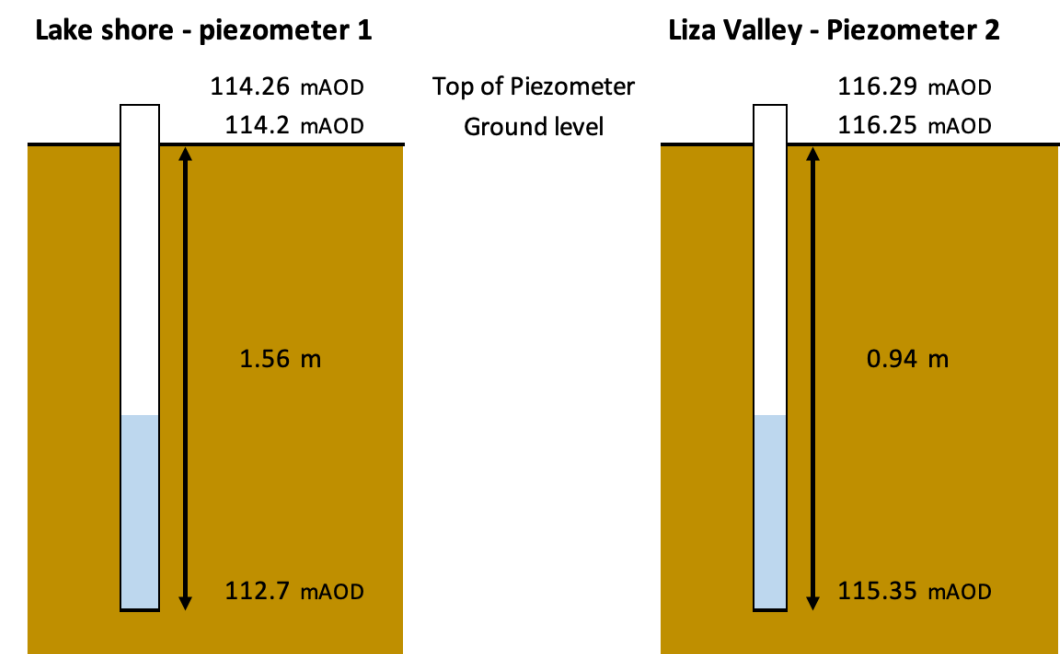


Figure 42. Diagram showing piezometer dimensions and elevations for the two monitoring locations.

3.6 Results

The following sections outline the findings from the individual elements of the hydrological monitoring campaign. Monitoring was carried out over two hydrological years however, gaps in the data were incurred at several sites. The timeline of obtained data and along with any gaps in the data are shown in Figure 52.

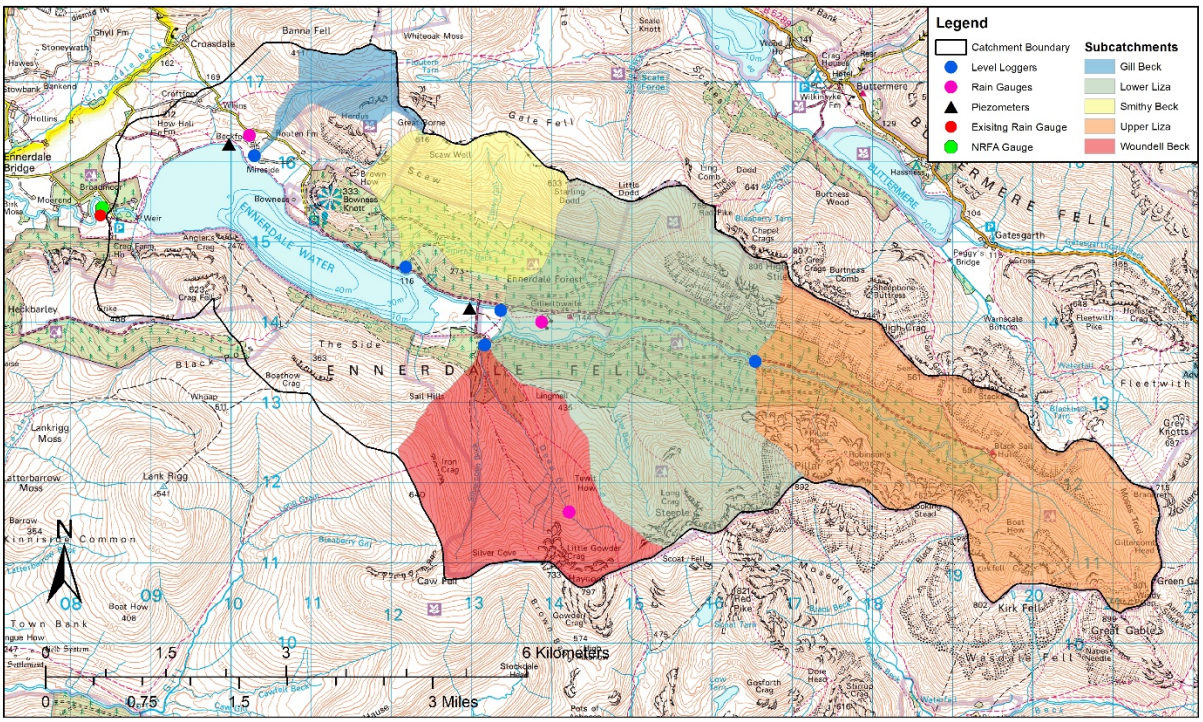


Figure 43. Final monitoring network for the upper Ehen catchment.

3.6.1 *Rainfall Data*

Technical issues impacted the collection of rainfall data at the Great Cove Sheep Fold site. The cause of these faults was not identified but was either due to a failure in the logging equipment or as a result of human error in the installation process. Due to the remoteness of this site, a second telemetered logger was installed and became operational on 1 December 2017. No data logging issues were reported after this date.

The two months of missing data at this site did not impact the final modelling input data as an existing national rainfall data set covering the entire catchment was available until the end of the 2017 calendar year. The monthly totals for each rain gauge are plotted alongside the monthly average of the three gauges and the long-term monthly means in Figure 44. This data set is described in more detail in Chapter 4.

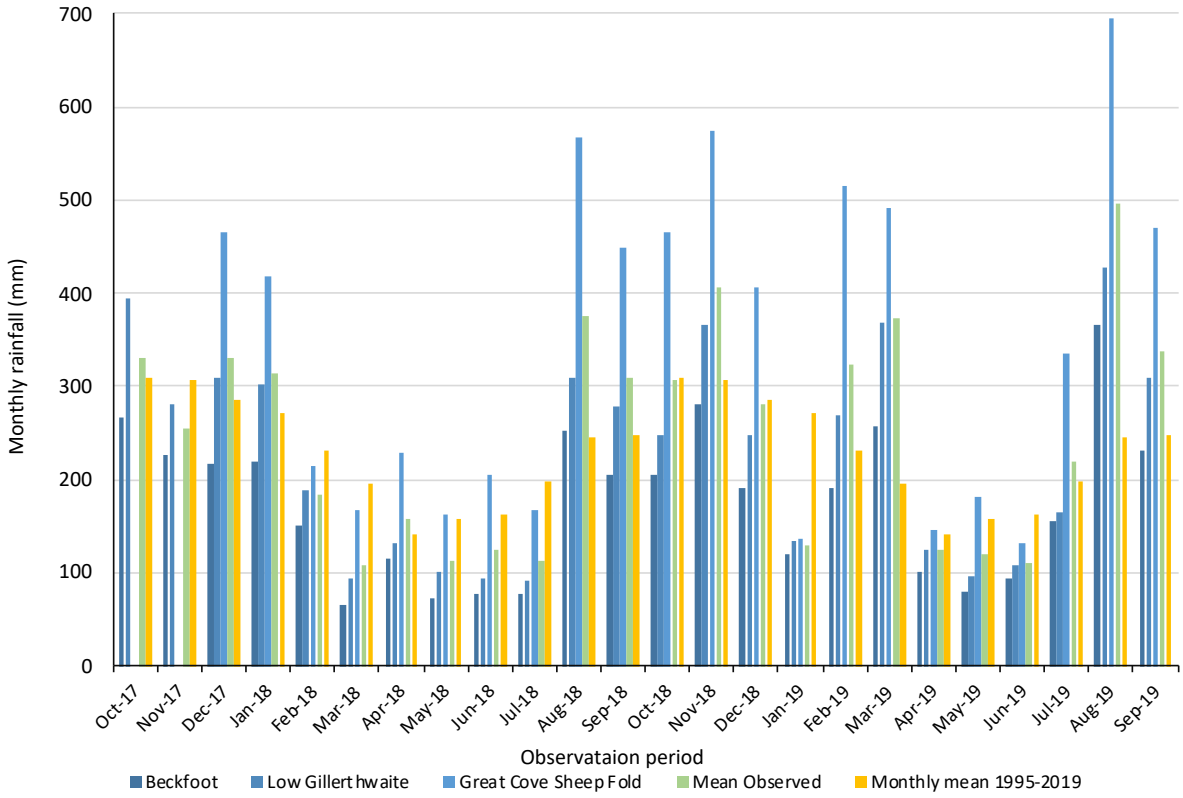


Figure 44. Observed monthly rainfall totals for the three gauges and comparison of the observed monthly mean totals with long-term monthly means.

As expected, based on historic rainfall data for the catchment, rainfall totals consistently varied between the three locations. Beckfoot, the furthest west and lowest elevation site received the least rainfall whilst Great Cove Sheep Fold, at 500m elevation and further into the valley consistently recorded the highest rainfall totals.

Based on the longer-term averages October and November are the wettest months in the catchment. However, during the monitoring period the highest monthly rainfall total was observed in August 2019 across all three sites. Great Cove Sheep Fold recorded 695 mm of rain during this month, whilst the Beckfoot and Low Gillerthwaite rain gauges recorded 366 mm and 428 mm respectively. This is also reflected in the discharge data which shows peak flows in the same month across all sites with the exception of Gill Beck where data was not available.

Minimum monthly rainfall totals occurred during March 2018, when 65 mm was recorded at Beckfoot and a monthly average of 109 mm across all three sites. This compares to a monthly average of 192 mm for January based on the 1995 -2019 period. However, in percentage terms the largest rainfall deficit was observed in January 2019 when the average across the three sites was 129 mm compared to the longer-term average of 271 mm.

3.6.2 *Discharge Data*

In three of the five locations two complete hydrological years of data were gathered. In sites 1 and 4 (Gill Beck and Woundell Beck) data collection was hampered due to the initial poor siting of the river stage recorders. For each site the range of stages that lie within the upper and lower gauged flows were quantified (Figure 45). The monitoring period covered an unusually dry period and as such, the range of low flows was well covered.

The remoteness of the sites and some procedural difficulties in the field meant that capturing the highest flows for each site proved difficult. Despite this, fewer than 15% of the highest observed stages fell outside the gauged range of flows. The number of occurrences of out of bank flow events varied greatly between sites. In 3 sites, with more constrained and steeper sided channels, the observed stages remained in-bank for the entire monitoring period.

In Smithy Beck, 3 out of bank flows were observed and 4 in the Upper Liza site. The Lower Liza site has relatively low-sided banks in relation to its drainage area. The surrounding land is also the natural flood plain for the river. As such, 22 out of bank flows were estimated to have occurred during the two-year hydrological period.

Low flows are of particular concern for Freshwater mussel conservation and are the main focus of this study. Low flows were well represented during the observation period. The least well represented sub catchment was Woundell Beck with 9% of low flows falling below the minimum gauged stage. In all other sites the number of recorded stages below the minimum gauged stage was 6% or less.

Using the range of gauged flows rating curves were established by plotting a line of best fit through the plotted stage discharge points. Additional points that covered the lowest observed stage and highest observed in-bank stage where also plotted. Estimates of the flow at these stages were made using the Manning's formula along with the dimensions and slope of the channel at each location. The Manning's roughness values for these points were based on the back calculated roughness for the gauged stages based on the discharges and channel geometry. Whilst there is some uncertainty around these values it helps to constrain the peak discharges at each site by avoiding the use of exponential curves and physically unrealistic discharge values.

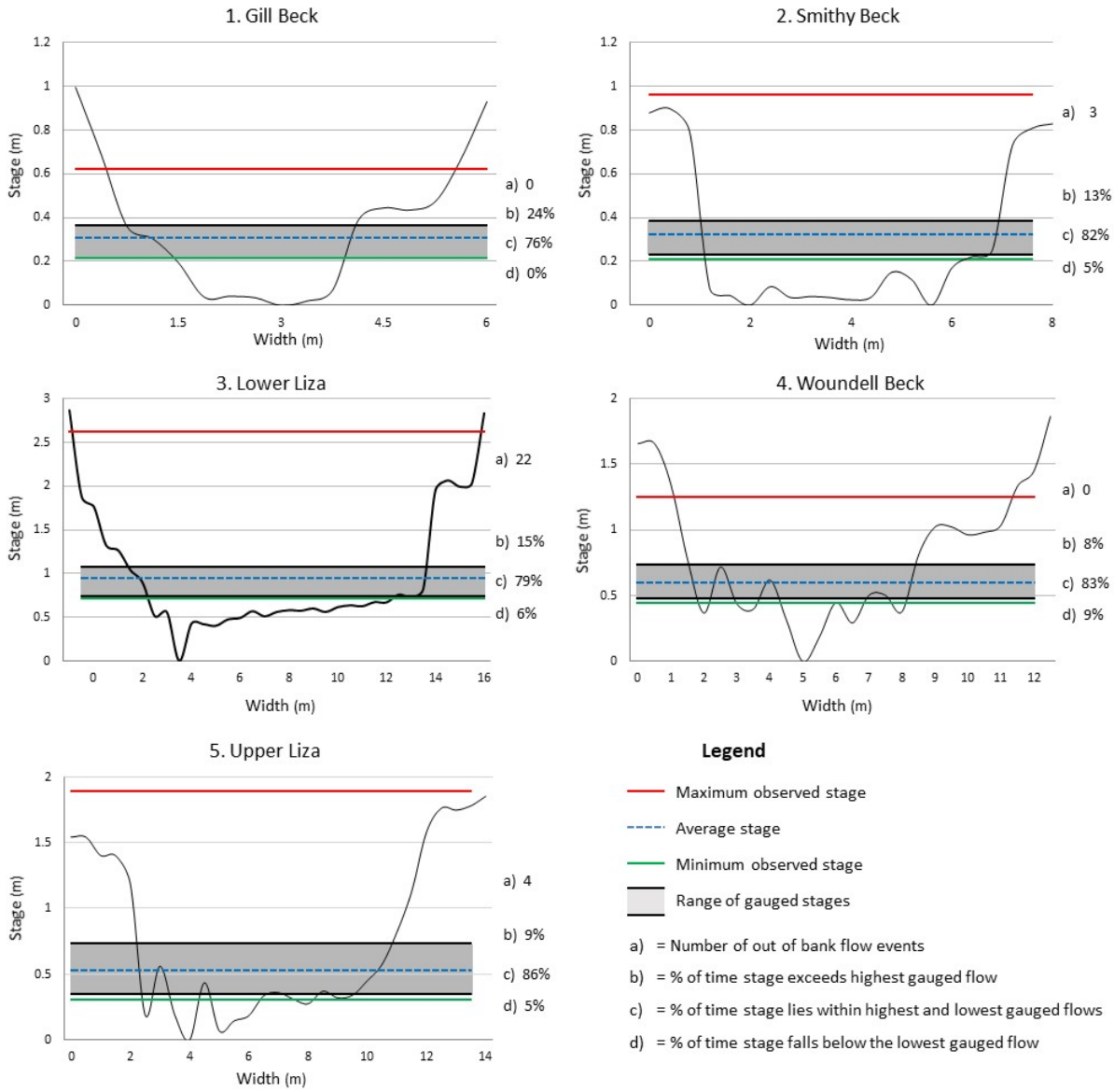


Figure 45. Visual representation of the range of stages captured by the salt dilution flow gauging campaign.

The evidence from the gauged flows suggested that Manning's roughness n changes at each site as stage increases or decreases. Generally speaking, as stage and hydraulic radius increase, the Manning's value decreases. In the five study sites Manning's roughness decreases as stage increases. This is supported by a number of studies that have shown how roughness increases as stage decreases particularly in mountains streams (Reid and Hickin, 2008) or step pool streams (Lee and Ferguson, 2002).

In three of the five sites, the line of best fit through the observed values used an exponential curve (Figure 46). However, for values above the highest gauge stage, the curves were constrained by fitting a polynomial curve to the highest recorded in-bank stage. This required the use of a two-step formula for different parts of the stage discharge curve. In Smithy Beck

and the Upper Liza sites a single polynomial curve was able to capture both the observed and estimated peak and minimum flows.

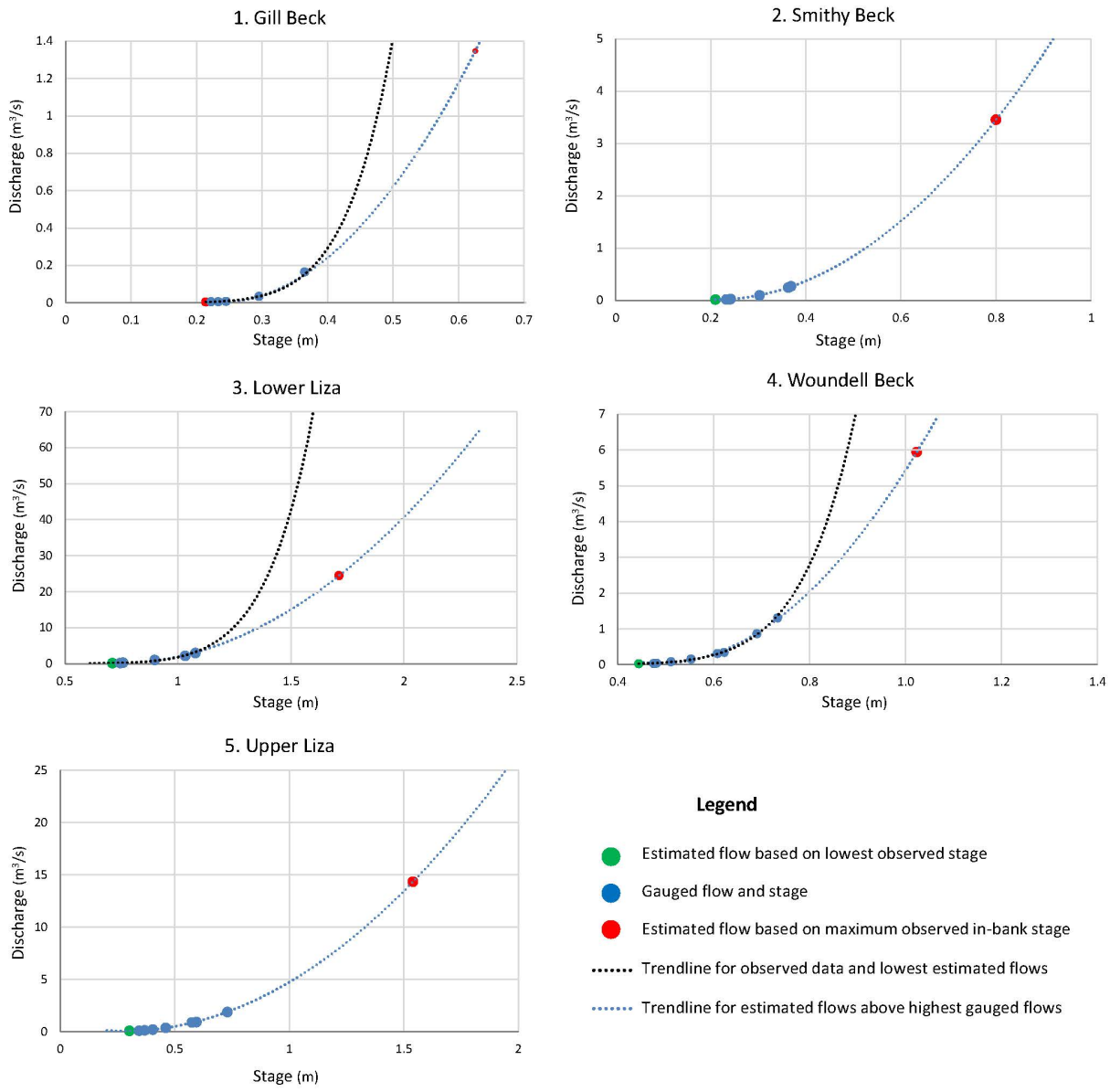


Figure 46. Rating curves for stage discharge relationships at each site.

As a further constraint on the peak flows, rainfall estimates for large events were compared with the estimated discharges to ensure that volumes of water flowing downstream didn't exceed total rainfall volumes over the sub catchment. In some cases, when occurring on already saturated soils and during winter months, runoff coefficients were extremely high and close to 100%. Normalised flow duration curves for each of the five sites are shown in Figure 47 and a selection of hydrological statistics including interquartile flows are shown in Table 9.

Table 9. Selected hyrdological statistics for the sub catchments monitored from Sept 2017 to Oct 2019.

	Gill Beck	Smithy Beck	Lower Liza	Woundell Beck	Upper Liza
Gauged Area (km^2)	1.2	3.4	26.1	5.3	9.7
Start Date	08/09/2017	08/09/2017	08/09/2017	22/05/2018	08/09/2017
End Date	10/09/2018	12/10/2019	12/10/2019	12/10/2019	12/10/2019
Total Days	367	764	764	508	764
Mean Daily Q (m^3/s)	0.05	0.19	2.26	0.5	0.95
Max Daily Q (m^3/s)	0.31	2.33	40.11	5.58	15.14
Min Daily Q (m^3/s)	0.005	0.02	0.15	0.03	0.06
Q95 (m^3/s)	0	0.02	0.19	0.03	0.1
Q70 (m^3/s)	0.02	0.08	0.55	0.11	0.31
Q50 (m^3/s)	0.05	0.13	0.92	0.22	0.51
Q10 (m^3/s)	0.17	0.39	5.26	1.27	2.02
Q5 (m^3/s)	0.23	0.55	9.07	2.11	3.23
Specific Q ($m^3/s/km^2$)	0.04	0.06	0.09	0.09	0.1

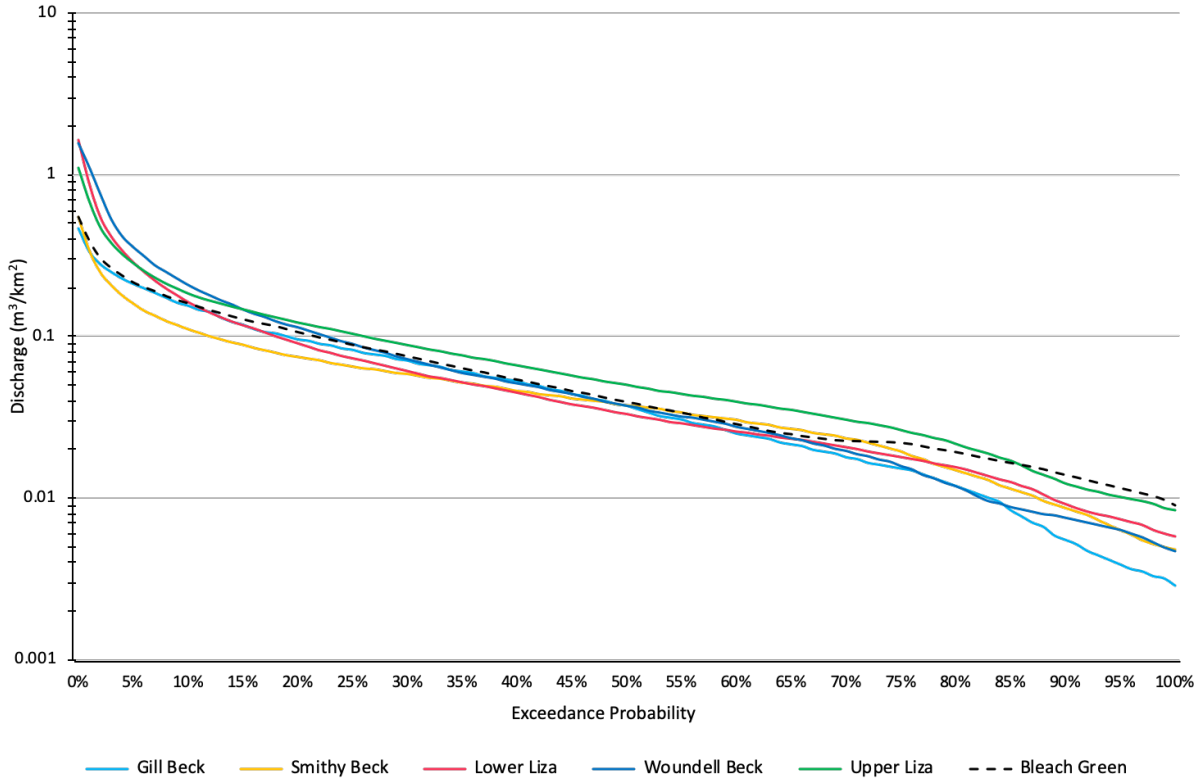


Figure 47. Normalised flow duration curves for each catchment based on dicharge estiamtes at 15-minute intervals for the entire observation period at each site.

The peak flow attenuating properties of Ennerdale Water were also captured during the monitoring campaign. By recording high resolution data to complement existing 15-minute

lake level and downstream discharge data, precise measurements of and timings of hydrological responses were obtained.

Figure 48 shows the runoff responses of the upper and lower River Liza for a series of rainfall events over 11 days following an extended dry period. In the three events preceding a major rainfall event on 18th – 19th of August peak flows in the River Liza followed around 6 hours after the peak rainfall. In the largest event the response to peak flow was reduced to around 3 hours. The reduction in runoff response times may be a direct result of the increased rainfall intensity but also due to the effect of soil moisture contents being higher than the previous events which took place after an extended dry period.

In the lower Liza the maximum stage recorded for the largest event was approximately 2.4m. This implies that out of bank flows took place and there is considerable uncertainty with the peak flow estimates. However, the average flow in the Lower Liza during the full 11-day period was 5.5 m^3/s compared to 6.7m^3/s for the River Ehen at Bleach Green.

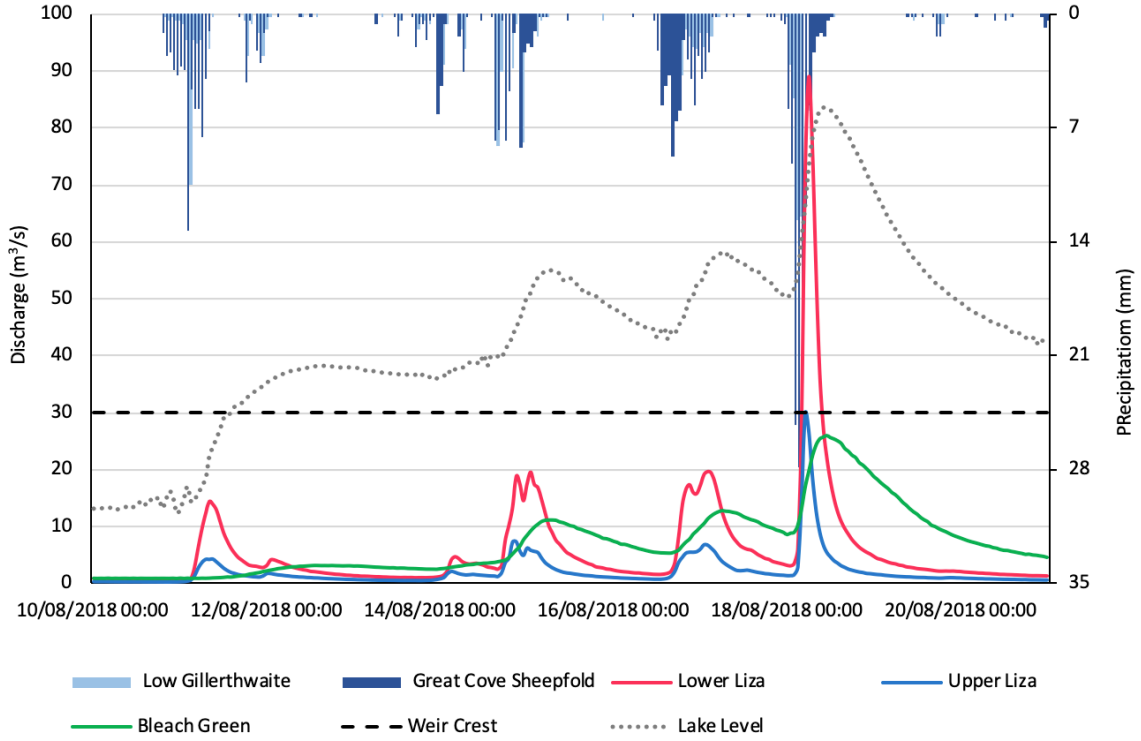


Figure 48. Example of runoff responses of the River Liza, lake level changes and discharges into the River Ehen at hourly timesteps for a 10-day period in August. Lake level changes are relative to weir crest and correspond in cm to the m^3/s discharge where 30 is = to 0cm above weir crest, 20 = 10cm below weir crest.

The shape of River Ehen discharge hydrograph (Figure 48) shows how Ennerdale Water dampens the flood peak and creates a more gradual recessional limb. The rising limb is also more gradual with the only sharp rise occurring in the major rainfall event at the later part of

August. Due to intense summer storms the total rainfall for the month of August 2018 was uncharacteristically high when compared to the longer-term average (Figure 52 & Figure 53).

During extreme rainfall events, instantaneous peak flows in the River Liza were frequently observed in excess of double the values in the river Ehen. In the largest flow event, peak flows in the River Ehen occurred 4 hours after the River Liza. The maximum estimated hourly discharge in the River Liza was 89.52 m^3/s compared to 46 m^3/s in the River Ehen at Bleach Green on 24-25th of December 2017 (Figure 49). The peak flow observed in the Liza on the 24/12/201 was only slightly larger than the high flow event in August 2018 (Figure 48) however the duration of the rainfall event lasted 24 hours resulting in an initial peak closely followed by a second slightly larger flood wave.

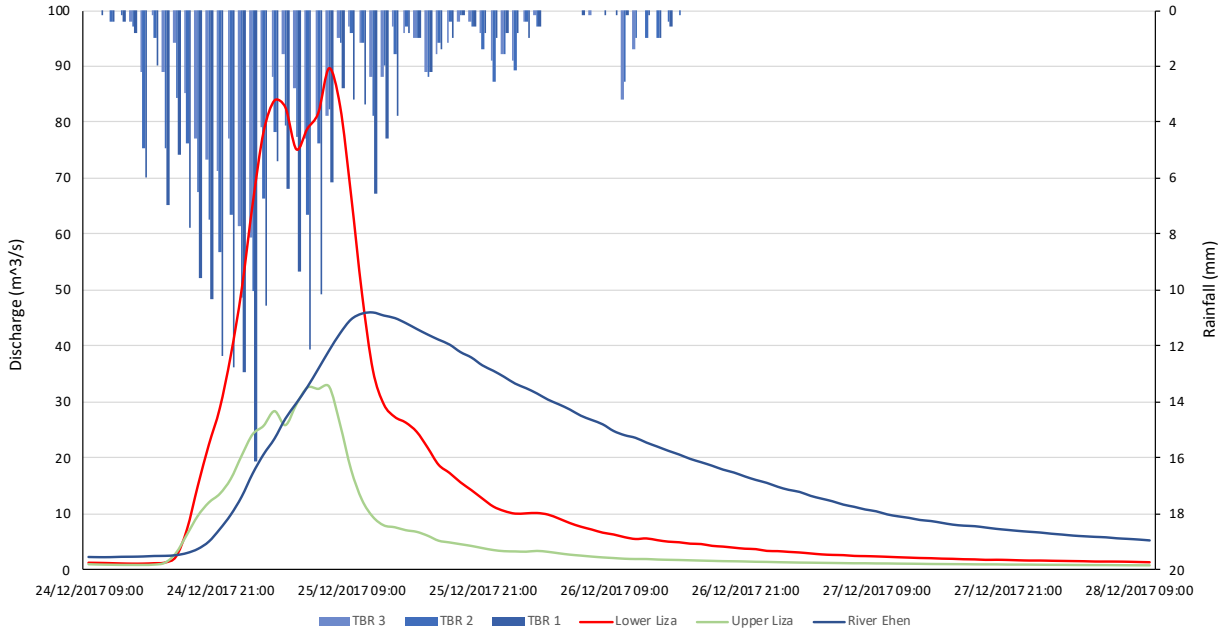


Figure 49. Storm hydrographs for rainfall event on 24th -25th December 2017.

3.6.3 *Shallow Groundwater Levels*

Shallow groundwater levels were recorded at two sites for a period only partially covering the two-year monitoring campaign (Table 10). The data provides useful insights into the response times of the subsurface water flow and the transmissivity and conductivity of the substrate immediately proximal to the two locations.

At piezometer 1, water depths always remained within the maximum observable range of the piezometer (Figure 50). At site 2, water levels frequently dropped below the maximum observable range however spot checks during these periods showed that moisture was present at the bottom of the piezometer.

Table 10. Shallow groundwater data obtained from the field monitoring capaign.

Site	Mean Depth (m)	Max Depth (m)	Min Depth (m)	Start Date	End Date	Dur (d
Lake Shore (P1)	0.08	1.45	0.04	10/5/2018	30/9/2019	5
Liza Valley (P2)	0.19	0.88	0.00	22/05/2018	13/07/2019	4

The variations of observed behaviour in the two piezometers can be attributed to the location and physical properties of the subsurface. Piezometer 1 was located at the bottom of a slope and within close proximity to Ennerdale Water. In general water levels rose rapidly in response to rainfall events and receded gradually due to the shallow hydraulic gradient caused by the lake level. This is shown more clearly in Figure 51. The base of the piezometer was estimated to be approximately 112.7 mAOD which is 0.41 m above the crest of the weir at Ennerdale Water.

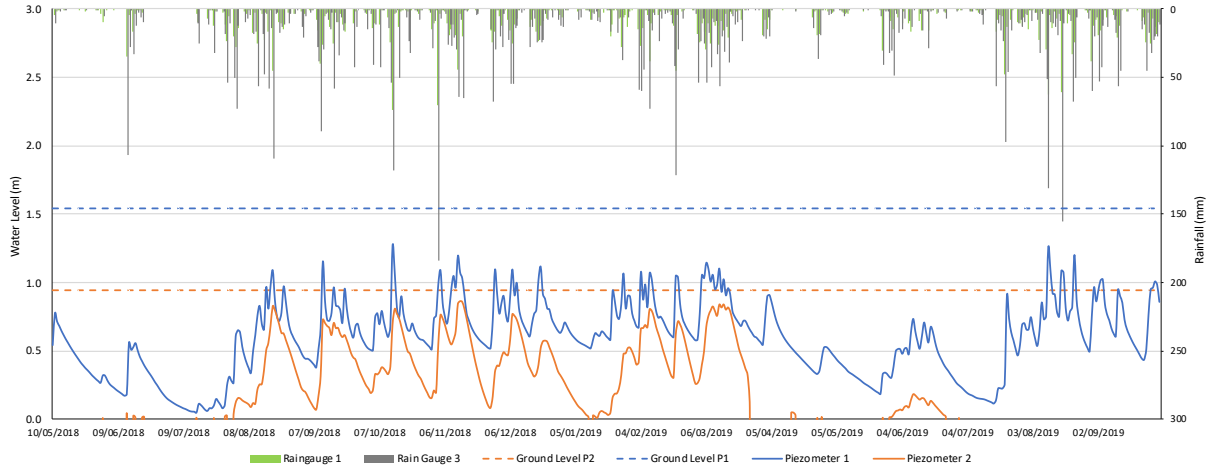


Figure 50. Recorded water depths in piezometers relative to ground surface and maximum piezometer depth.

Changes to the subsurface water levels vary considerably depending on both the duration and intensity of rainfall events and the antecedent conditions. From the onset of the first rainfall event (Figure 51), after a 2-day period with no rain, water levels begin to rise 9 hours after the onset of rain. However, during closely followed rainfall events, water levels in the piezometers respond within 2-4 hours. In contrast, the receding limb of the stage hydrograph for Piezometer 2, shows no response to a number of smaller rainfall events. This is particularly evident in the two rainfall events during April 2019 (Figure 51).

Piezometer 2 was located further away from the lake and on an area of flat alluvial deposits. It was not possible to construct a piezometer any deeper than 0.94m so the full range of

subsurface water levels could not be captured. However, based on the lack of observable response in water level to rainfall events it is possible to hypothesise that the water level was considerably lower than the base of the piezometer. Water levels in piezometer 2 receded at a much faster rate than at piezometer 1 due to the lack of lake influences and the more porous nature of the substrate.

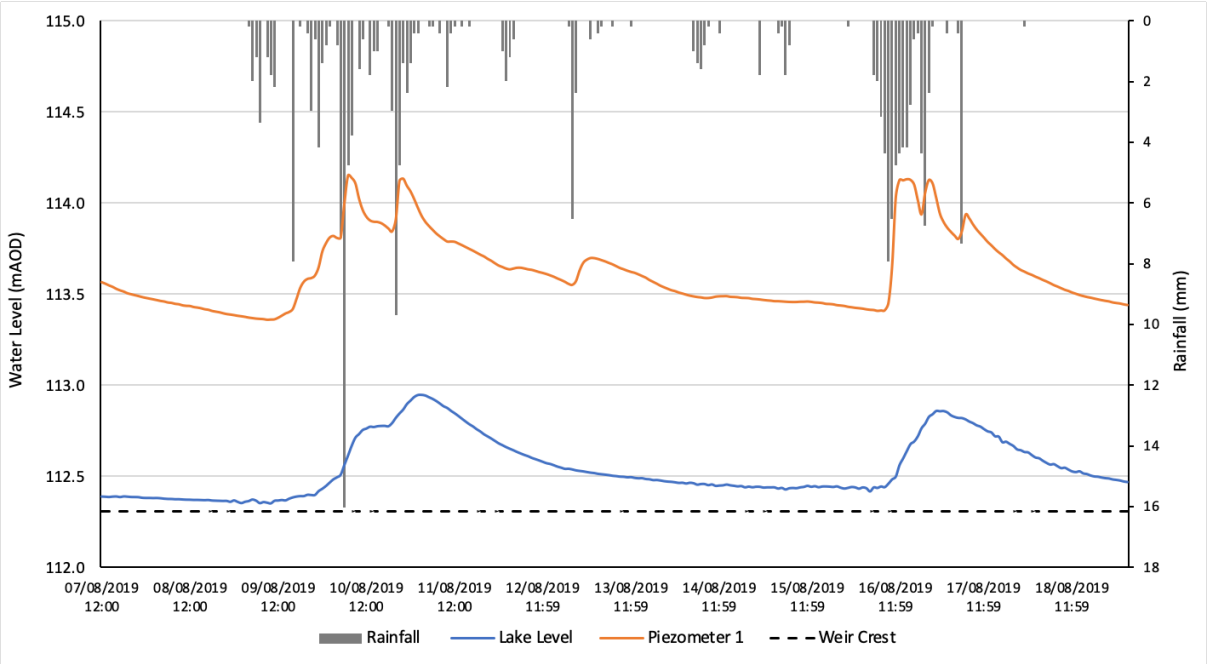


Figure 51. Subsurface water and lake level responses to rainfall shown at hourly timesteps

3.7 Synthesis

A large quantity of data at high temporal resolution was gathered during the two-year monitoring campaign. Two stream gauging locations suffered from extreme high flow events resulting in damage to the logging equipment or major adjustments to the stream channel.

Rainfall data collection suffered minor interruptions due to undiagnosable logger failure however the spatial distribution of the gauges was able to capture the high variability of rainfall events within the catchment (Figure 52).

Annual rainfall totals were above average when compared to long term average from 1995/6 to 2016/17 however this was distributed unevenly in both of the years covered. An extended dry period lasting from May to July 2018 was followed by a wetter than average August. The following hydrological year again saw a wetter than average year with 2944 mm falling over the catchment. A dry January was followed by a wet February and March. Late spring and early summer were then drier than average whilst August totals were around double the average for the preceding 21 years.

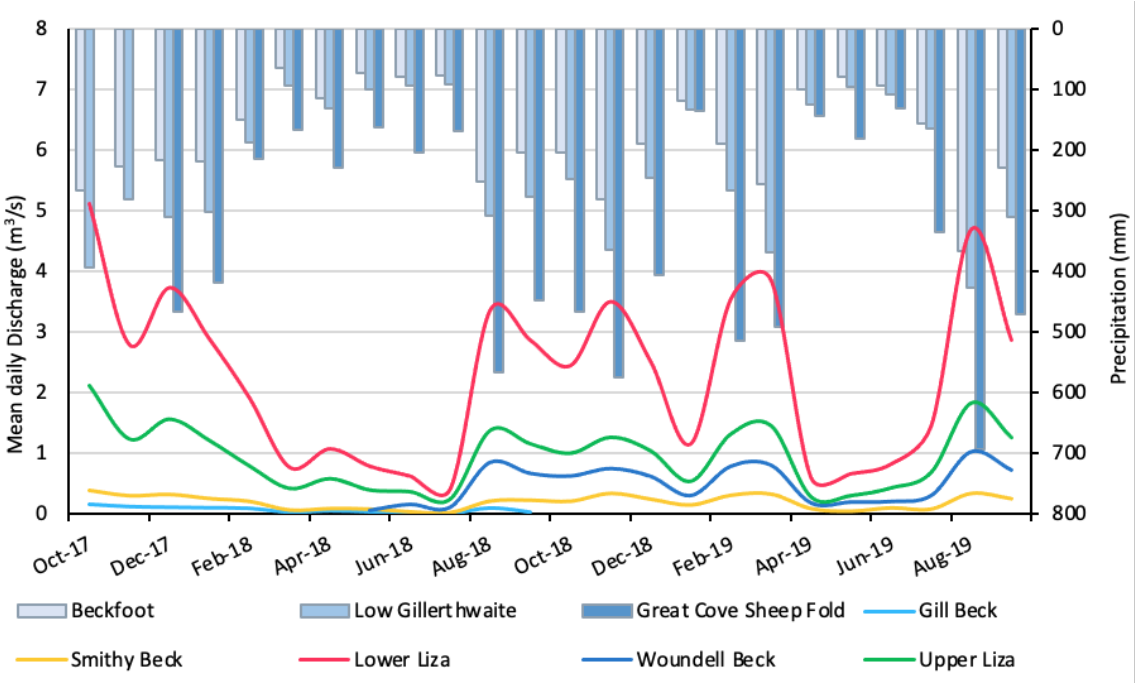


Figure 52. Monthly flow and rainfall data for the five gauged sites and three rain gauges for the 2017-2019 monitoring period.

The range of observed flows at the outlet of the catchment and the recorded rainfall for the monitoring period are compared with the averages for the 1995/96 to 2018/19 period (Figure 53). Extended low flow periods in the 2017/18 hydrological year enabled good coverage of extreme low flows to be recorded and runoff responses to large rainfall events under a range of antecedent conditions were also captured during the full two-year campaign. The findings from this monitoring provide previously unseen evidence of the hydrological behaviour of large parts of the unmodified upper catchment and a better understanding of the storage and flow attenuating properties of Ennerdale Water in its current physical form.

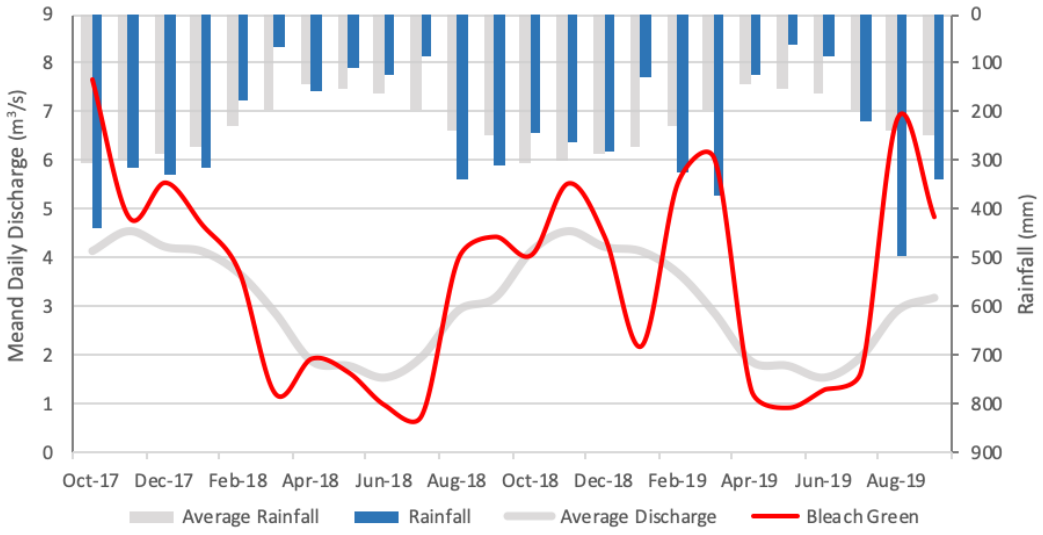


Figure 53. Discharge at Bleach Green gauging weir and rainfall observations for the two-year monitoring period compared with long-term averages shown in grey using data from 1995/96 – 2018/19 (NRFA, 2019a; Tanguy *et al.*, 2019)

Despite being a relatively small dam, the weir at Ennerdale Water and the management of compensation flows has a considerable impact on the hydrological regime of the upper catchment. Understanding the hydrological responses of different areas of the Upper Catchment and the main inflows into the lake has many important implications for water resource management and for land use and conservation management of the Upper Catchment. Forestry and low intensity agriculture are important activities for the local economy and cultural identity of the UNESCO designated Lake District National Park. The way in which they are managed in the possible absence of flow regulation structures at the outlet of Ennerdale Water may have important implications for the future hydrology of the catchment.

3.8 Discussion and Limitations

Streamflow Data

Collecting streamflow data in remote mountainous areas presents a number of challenges. On a limited budget a compromise must be made between the extent of monitoring and the technology used to carry out the monitoring. An increasing availability of lower cost technologies with more sophisticated recording sensors and higher data storage capacities is helping to overcome monitoring constraints. However, ultimately a judgement must be made as to how much data is required (extant in space and time) over the quality of data (accuracy, uninterrupted time series, etc.) needed to adequately answer the research question. As a project progresses this is likely to become clearer and monitoring equipment may be added or removed accordingly. However, even with unlimited financial resources, there would likely come a point when excessive volumes of data either stopped providing useful additional information or became a hindrance to a project due to the requirements of collecting, storing and processing (e.g. quality controlling) data. For the purposes of this project a wider coverage of monitoring locations was given preference as it allowed for redundancy to be built into the network. This meant that should failure occur at one site data would likely be available from another site. It also allowed for the effects of large variations in the physical characteristics of the catchment and its impact on the hydrology to be captured.

Choosing the correct location for positioning monitoring equipment can also be a complex process. Prior knowledge of both the catchment and the equipment being used play an important role in helping to avoid data loss however, unforeseen events, technological failure or human interference can impact the best thought out and planned monitoring network

designs. Data loss is therefore an inevitable part of field work and monitoring in hydrological studies. This is an important consideration when assessing the quantity and quality of data collected during the 2017-2019.

Using stage measurements as a proxy for river discharge is a well-established method. Its success however depends largely on the use of a well-constructed rating curve. The rating curves for this study were built using salt-dilution spot gauging. This technique is known to perform well in mountainous streams however a drawback of the technique is the need for large amounts of bulky and heavy equipment. In remote streams with limited accessibility this becomes an increasing problem. For this reason, it was necessary to carry out spot gauging with at least three team members. However, planning these trips was time consuming and needed to be done well in advance. This did not prevent low flow events being captured during the monitoring campaign as there were several low flow periods during the two-year monitoring programme however it was not possible to respond at short notice to high flow events and assemble a team to conduct monitoring and capture the higher end of the rating curves in each channel. Furthermore, access to the river channels in the winter months was restricted due to Arctic Charr spawning and egg development previously mentioned.

The rating curves established for all sites are therefore, limited by the number of gauged flows and the range of gauged flows. However, whilst this is a limitation, the high temporal resolution of the data means that the hydrological responses and behaviours of the upper catchments can be well characterised despite uncertainty around the discharges at higher flows.

Hydrological data for the River Ehen and Ennerdale Water, have been collected for over 40 years. Yet, despite the use of Ennerdale as an important part of the local domestic water supply network, no previous continuous flow measurements of the River Liza or other significant inflows into the lake have previously been undertaken. This newly collected data therefore adds new insight into the hydrological behaviour of the Upper Ehen catchment and the storage discharge behaviour of Ennerdale Water.

Rainfall Data

Despite appearing a relatively simple task, rain gauge measurements are subject to a range of uncertainties which affect even the most sophisticated measurement devices.

Rain gauges are subject to wind induced undercatch, which becomes more significant in exposed upland sites. To negate this effect, tipping bucket rain gauges with aerodynamic design were used. However, the extent of undercatch and variation between sites remains

uncertain. Using the same rain gauges at each site, applying the same criteria for site selection and the same criteria for setting up the rain gauges, reduces inconsistencies between data collection.

A final limitation of the field data overall is the relatively short period of observed data compared to the overall study period of 1/10/95 to 30/9/19. The range of climatic behaviours and hydrological responses can be limited by probability however this is a constraint on any research project with a finite time period.

A more detailed comparison between the collected rainfall data and the long-term data set used for the modelling is made in Chapter 4. However, the final rainfall data set collected during the two-year period was able to represent the overall behaviour of the catchment reasonably well. This is validated by comparing the modelled water balance to the observed water balance in Chapter 4 and also through the direct comparison of the field data with existing datasets for an overlapping period.

3.9 Chapter Summary

This chapter has characterised the modified hydrological regime of the Upper Ehen catchment by analysing existing hydrological, climatological and geographical data. Comparing the Upper Ehen with a physically similar neighbouring catchment, the Irt, has illustrated the extent of hydrological alteration, particularly at the lower end of the flow duration curve. Furthermore, the newly generated hydrological metrics provide a quantitative baseline for a more detailed assessment of the levels of hydrological alteration in Chapter 4 and Chapter 6.

The field monitoring campaign has provided extensive flow data for 5 previously ungauged catchments and includes two of the main inflows into Ennerdale Water. This empirical data has captured the attenuating properties of Ennerdale Water and provides greater certainty for modelling the future behaviour of Ennerdale Water and the River Ehen under future management scenarios.

The additional rainfall data provides further empirical evidence to support data showing the longer-term rainfall distribution within the catchment and allows the modelling period to be extended beyond the range of currently available data. Exceptional rainfall events during the monitoring period, provide further insights into the range of hydrological responses of the upper catchment previously not captured, whilst extended dry periods during both years of the monitoring campaign provide further additional data for characterising extreme low flows, a critical area of concern for freshwater mussel conservation.

Chapter 4 - Estimating the Natural Flow Regime

This chapter reviews some of the main techniques for generating natural flow regimes in the absence of unmodified time series (Section 4.1). Using the data collected in Chapter 3, the novel application of SHETRAN and loosely coupled lake models are presented and validated. New hydrological metrics are generated to compare the modelled discharge and lake level timeseries and provide evidence to support the chosen modelling approach. An example application of modelling a natural lake outlet profile is also shown.

4.1 Literature Review

4.1.1 *Techniques for estimating the natural flow regime*

Recognition of the ecological impacts of anthropogenic flow regime has led to the creation of a number of tools, metrics, frameworks and methodologies for estimating a river's natural flow regime and assessing the levels of alteration. Understanding the level of alterations to natural flow regimes is important in order to assess the ecological health of rivers and streams (Falcone *et al.*, 2010) and to inform river management and restoration efforts such as dam operations or infrastructure removal projects. However to estimate the level of alteration it is necessary to define the natural or unaltered regime for purposes of comparison and this can be a major challenge (Homa *et al.*, 2013).

When flow regimes are altered or diminished, the ensuing ecological degradation results in the loss off the significant societal benefits that healthy functioning ecosystems provide (Mathews and Richter, 2007). Recognition of the value of these ecosystem services has seen a growth in the demand for river scientists to provide accurate estimations on the amount and timing of flows needed to sustain healthy riverine ecosystems (Mathews and Richter, 2007).

The late 80s saw the advent of research into statistical approaches for classifying flow regimes and assessing hydrological alteration (Poff *et al.*, 2017). The following decade saw the development of a number of flow indices for ecological studies and by the early 2000s hundreds of indices existed (Olden and Poff, 2003). By this stage choosing which indices were most appropriate to represent the five main components of the flow regime (magnitude,

frequency, duration, timing and rate of change) had become increasingly challenging for river ecologists (Clausen and Biggs, 2000).

During this period, one of the first widely available tools for assessing the anthropogenic alteration of flow regimes was developed by The Nature Conservancy in the 1990s and described by Richter *et al.,* (1996). The Indicators of Hydrological Alteration (IHA) method involves the calculation of 32 ecologically relevant hydrological parameters across five main components of flow regime (magnitude, frequency, duration, timing & rate of change). These are calculated for pre-defined periods of flow records covering pre-disturbance and altered flow regime periods (Richter, 1996). Another example is the Hydrological Index Tool (HIT), a software developed by the US Geological Survey. This provides the user with 171 metrics across the five components of flow regime as demonstrated in Chapter 2 and 3.

In some cases, gauged flow records covering both pre- and post-disturbance periods may be available allowing for a direct comparison between the two periods. However, in many cases pre-disturbance flow data is unavailable due to the existence of long-standing impoundments, abstractions or other anthropogenic flow altering factors. To overcome this lack of data a number of techniques are available. In many cases the techniques for estimating natural or unmanaged flow regimes are the same as predicting flows in ungauged basins. Two of the most commonly used techniques are through rainfall runoff modelling or statistical regional regression models (Hailegeorgis and Alfredsen, 2017). These two techniques are outlined in the following sections along with examples and an assessment of their strengths and limitations in estimating unaltered flow regimes and generating ecologically relevant streamflow metrics. Other techniques for estimating unmanaged flow regimes are also briefly covered along with a justification for the chosen approach to generating natural flows in the Upper Ehen catchment.

4.1.2 *Hydrological Modelling*

Probably the most common technique for the reconstruction of natural flow conditions is the use of hydrological models to simulate discharge time series for the catchment under unaltered or pre-disturbance conditions (Wen, 2009; Peñas *et al.*, 2016). Rainfall-runoff models can be used to generate discharge time-series at specific locations and under different scenarios (Smakhtin, 1999) allowing the impacts of changes such as vegetation cover or climatic conditions to be estimated (Pechlivanidis and Jackson, 2011). They can also be successfully employed when a natural or unaltered discharge time-series needs to be generated (Smakhtin, 1999).

The ecological limits of hydrologic alteration (ELOHA) framework (Poff *et al.*, 2010), advocates the use of hydrological models for providing ecologically relevant and accurate estimates of streamflow. Models provide the opportunity to evaluate levels of anthropogenic flow regime alteration and anticipate the impact of proposed human activities. They also offer the possibility to extract flow data from ungauged sites or specific sections within gauged catchments (Poff *et al.*, 2010).

A number of studies have used rainfall-runoff or physically based catchment models to simulate river discharges under natural conditions in order to assess levels of flow regulation (Dunn and Ferrier, 1999; Frazier *et al.*, 2005; Povilaitis and Querner, 2008). To generate natural discharge time-series the model can initially be constructed to represent the altered conditions in the target catchment and calibrated against existing time-series data. This ensures that the pattern of flow variability is correctly reproduced and representative parameter values are obtained (Smakhtin, 1999).

Several different types of hydrological models are available for studying and predicting hydrological behaviour but all provide simplifications of real world processes (Refsgaard, 1996) and each has advantages and disadvantages. A detailed review of the different types of hydrological models available and their strength and limitations is beyond the scope of this chapter, however, there are a number of recent reviews within the literature and numerous books dedicated to the subject. Moradkhani and Sorooshian, (2008) published a general overview of rainfall runoff modelling which provides information on model calibration, data assimilation and model uncertainty. Whilst Devia *et al.*, (2015) reviewed five commonly used hydrological models; variable infiltration capacity model (VIC), TOPMODEL, HBV, MIKESHE and the soil and water assessment tool (SWAT) to assess the modelling tasks and situations to which they are best suited.

Hydrological models are powerful tools for a range of applications however a limitation often associated with their use in natural flow regime estimation is their ability to accurately predict ecologically relevant streamflow metrics (ERSFMs) or statistics (Shrestha *et al.*, 2014). This is often borne of the fact these statistics are not explicitly considered in the model optimization process. The use of objective functions as indicators of model performance can lead models to perform well in one aspect of the hydrograph at the expense of another (Wagener *et al.*, 2001). One of the most commonly used performance indicators for hydrograph fitting is the Nash-Sutcliffe efficiency (NSE), however due to the quadratic nature of the formula (Hallouin *et al.*, 2019), greater importance is given to fitting high flow peaks. Consequently, models calibrated solely with these type of criterion may be unable to

accurately simulate low flow conditions (Shrestha *et al.*, 2014). The type of objective function used will ultimately lead to differences in the ecologically relevant metrics produced by the model (Shrestha *et al.*, 2014).

Recent research has looked into different methods for overcoming this limitation of hydrological models for predicting ecologically relevant flow characteristics. Vis *et al.*, (2015), used combined objective functions to assess model performance when focussing on 12 ecological flow characteristics. Their research suggested that this technique could increase model accuracy and have the potential to aid the decision making process (Vis *et al.*, 2015)

Similarly, Hernandez-Suarez *et al.,* (2018), evaluated the model calibration approaches to assess the predictability of hydrological indices using the SWAT model. Using multi-objective model calibration approach they were able to better represent a wider range of flow regime components when compared to single-objective approaches (Hernandez-Suarez *et al.*, 2018).

4.1.3 *Statistical Models*

Regional or statistical regression is a commonly applied approach used to estimate streamflow characteristics in the absence of gauged data, which can be viewed as analogous to lacking unaltered flow data. Regression techniques have been used for estimating general hydrological signatures (e.g. Flow Duration Curves, Base Flow Index) in ungauged catchments for several decades (Murphy *et al.*, 2013) and techniques have developed further since the promotion of the PUB (Predictions in Ungauged Basins) initiative by the International Association of Hydrological Sciences (Sivapalan, 2003).

Statistical models such as linear regression, partial least square regression and principal component regression can be used to forecast long-term hydrological behaviour usually at annual or seasonal timescales (Masselot *et al.*, 2016). However, the information produced thus lacks sufficient detail about the hydrograph shape and timings of peak flow events (Masselot *et al.*, 2016).

In contrast, other models such as artificial neural networks, wavelet regressions and support vector regressions, can be used to predict or model only a few days or data points at once (Masselot *et al.*, 2016). This may be useful for examining future extreme flow events but not for long-term forecasting or identifying overall trends in stream flow (Masselot *et al.*, 2016).

Regionalisation techniques use data from neighbouring catchments by selecting model parameters from basins in close spatial proximity or with similar physical characteristics.

Oudin *et al.*, (2008), compared the effectiveness of these two approaches against a regression based approach using one or more donor catchments to generate model parameters for ungauged catchments. On both models, spatial proximity was more important than physical similarities for model efficiency. The study concluded that for their region of study, spatial proximity provided the best regionalization solution for predicting ungauged flows and regression provided the worst (Oudin *et al.*, 2008)

Statistical models have recently started being used for estimating ecological streamflow metrics in ungauged basins or in the absence of unaltered streamflow time series. Sanborn & Bledsoe (2006) used multiple regression models for predicting 84 ERFMs across Colorado, Washington and Oregon. They were able to accurately predict streamflow characteristics describing magnitude, timing and rate of change across a large heterogeneous region (Sanborn and Bledsoe, 2006)

Knight *et al.,* (2012) used stepbackward regression to predict 19 ERSFMs in Tennessee and Cumberland river basins. Whilst, Murphy *et al.,* (2013) later applied the same approach to six gauged streams in Kentucky to compare modelled results with observed data.

The approach has recently been taken further and used to predict sub-daily ERSFMs in regulated rivers and ungauged basins. Hailegeorgis and Alfredsen (2017), assessed a regional regression model against a regionally calibrated rainfall-runoff model on Norwegian catchments. Best overall model performance (NSE) was achieved from the regression model, whilst both performed similarly in the estimation of a small selection of streamflow metrics (Hailegeorgis and Alfredsen, 2017).

4.1.4 *Hydrological vs Statistical Models*

A number of studies have directly compared the use of statistical models and rainfall runoff models for estimating ERSFMs. Murphy et al., (2013) compared a stepbackward regression technique (see Knight, Gain and Wolfe, 2012) with the use of hydrological models on six gauged sites in Kentunky. The research found that the regression technique produced median values closer to the observed values for almost all characteristics. However, modelled results showed a range of variability closer to that of the observed natural variability (Murphy *et al.*, 2013).

Caldwell *et al.*, (2015), conducted a model comparison to quantify the differences in streamflow predictions from a range of models used for water resource management decision makers. Four fine scale models and two regional-scale models were used and compared using model fit statistics and bias in ERSFMs (Caldwell *et al.*, 2015). The work showed that no

specific model performed better than others when compared across all sites and all measures of performance. The work provided evidence that the calibration procedure is as likely to affect model performance as model structure (Caldwell *et al.*, 2015).

Vigiak et al., (2018), carried out an uncertainty analysis on the use of modelled streamflow for ecological assessment in Southern Europe. The study compared a number of different catchment models including mHM (mesoscale Hydrologic Model), Lisflood, and SWAT set up for Spanish catchments. Models were primarily assessed on their ability to replicate mean monthly flows using RMSE, RSR, PBIAS and NSE as performance indicators. Models were then assessed on their ability to reproduce flow regime indicators by comparing RMSE, RSR and R2 between modelled and observed data across all sites. The study found that with the exception of medium to high magnitude indicators, uncertainty was high. However several limitations of the study were highlighted by the authors, including insufficient record length for observed flow times series and the use of poorly performing hydrological models for comparison (Vigiak et al., 2018).

4.1.5 *Streamflow Transposition*

Where unaltered streamflow records are unavailable, another approach for generating this data is to use records from nearby, unimpacted catchments. This data can be transposed using a transposition factor to estimate unimpacted flows at the site of interest (Stewardson *et al.*, 2017b). A limitation of this technique is the availability of hydrologically similar catchments free from anthropogenic flow regime disturbances. In the UK particularly and other parts of densely populated Europe this can present a challenge as the majority of catchments have some flow regulation.

Lowe and Nathan (2006), proposed a methodology and criteria for transposing gauged streamflow to ungauged locations using evidence from a study in Victoria, Australia. Their research found that mean annual flow (MAF), base flow index (BFI) and median summer flow (MSF) were useful indicators of hydrological similarity for general water resources and environmental flow studies however they were less useful for the transposition of flood related indices (Lowe and Nathan, 2006). The methodology presented is useful for selecting the most appropriate basin to use for transposition, however, it assumes multiple unaltered basins are available to choose from.

In larger basins it may be possible to use data from unaltered upstream gauges to generate unaltered discharge time series further downstream. This technique was demonstrated by

Blythe and Schmidt (2018) who used a mass balance approach to measure daily flows and levels of flow regime alteration in the Northern Branch of the Rio Grande.

4.1.6 *Synthesis of Natural Flow Regime Estimation Techniques*

After reviewing the literature, assessing the aims of the project and the requirements of the sponsor and wider stakeholders, a hydrological modelling approach complemented by the collection of additional field data was chosen as the most appropriate technique. It was felt that this would give greater flexibility over the range of outputs and provide a tool for further investigations of future management options that may come to light at a later date. Furthermore, time was also available to carry out a thorough modelling study and collect additional data including new gauged flows in the unaltered part of the catchment.

A review of hydrological model types (e.g. lumped, distributed, semi-distributed) was conducted to assess their suitability against the objectives of the thesis. This included reviewing individual cases studies where specific models were applied and literature where a variety of model types were compared or reviewed (see Refsgaard, 1996; Dunn and Ferrier, 1999; Frazier *et al.*, 2005; Moradkhani and Sorooshian, 2008; Povilaitis and Querner, 2008; Devia *et al.*, 2015).

SHETRAN, a physical based spatially distributed model was selected, partly due to its range of functionality and partly due to evidence of its suitability and performance when being applied to physically similar catchments. SHETRAN is a further development of the Système Hydrologique Européen (SHE) which allows for the modelling of surface and subsurface flow in river basins (Ewen and Parkin, 2000). Other examples of developments of SHE include the hydrological models MIKESHE and SHESHED.

SHETRAN has been used for a variety of applications including environmental impact studies of land erosion, pollution, land use and climate change scenarios and analysis of the interactions between surface-water and groundwater for water resource management (Ewen and Parkin, 2000), studies assessing the impact of groundwater abstraction on river flows (Parkin *et al.*, 2007) and investigations into the impacts of deforestation on sediment transport and peak flows (Birkinshaw *et al.*, 2011). A strength of SHETRAN is that it allows surface and groundwater interactions to be analysed, which is fundamental for conservation of the environment as well as for understanding water levels and flows (Winter *et al.*, 1998). PBSD models are most appropriate for smaller domains due to their high parametrisation and fine resolutions which makes them computationally demanding at larger scales (Caldwell *et al.*,

2015). A review by Refsgaard *et al.*, (2010) provides further information on the development of the SHE and its contribution to hydrological science.

Other PBSD models have functionality equal to or greater than SHETRAN and could easily be applied to this or similar projects. However, an advantage of SHETRAN over some commercially developed variants of SHE or other commercial PBSD models is that is freely available meaning that the methodologies used in this project can be easily replicated or scrutinised by other researchers without the need to purchase licensed software. Furthermore, much of the developmental and user experience is based at Newcastle University, which provided scope for additional training and the possibility of developing bespoke model adjustments for the project. Licensed software often has limited capacity for making bespoke user adjustments as the source code is hidden. Instead, they often consist of a finite list of user settings. Furthermore, access to technical help or training services may be limited which can lead to delays in a project.

Additionally, physically based models are considered by many hydrologists to provide a better representation of the heterogeneity of hydrological processes within a catchment when compared to lumped models (Dukić and Radić, 2016). SHETRAN is able to generate detailed temporal and spatial information within a catchment, which can be visualised using SHETRAN viewer, which can help to communicate the model results to stakeholders more effectively.

The requirements of the project steering group and future model users was also an important consideration. SHETRAN, like other spatially distributed modelling software has the ability to model land cover or land use changes. This may be a useful option when considering the future management of the catchment. It is also a valuable tool in assessing the impact of historic land use changes on catchment hydrology. Although, this was not a central consideration in the scope of this study, providing a modelling tool with this option was an important consideration for the project. Land cover is discussed further in section 4.3.5.

Despite the advantages offered by SHETRAN and other PBSD models, a common criticism is their over complexity and parameterisation of hydrological processes. If a model user lacks experience, good hydrological knowledge or understanding of their catchment and its processes, attempts may be made to calibrate the model based on curve fitting rather than sound hydrological principles. Another issue with the higher complexity of PBSD models is the possibility for equifinality. This refers to a situation in which there are multiple combinations of parameter settings that will achieve the same result or good model performance. If the model user is not familiar with their catchment or its processes, unrealistic

values may be assigned to certain parameters for the sake of achieving good model performance.

A large number of techniques are available for estimating natural flow regimes. The chosen approach for a specific study may often be selected based on a range of criteria e.g., availability of data, purpose of the study, user experience or preference and project timeframes. However, all methodologies present their own range of practical advantages and disadvantages as well strengths and limitations in the results they produce. With this in mind, it is therefore essential to effectively communicate the performance of hydrological models. Harmel *et al.*, (2014) provide a review and recommendations for evaluating, interpreting and communicating the performance of hydrological models. A number of these recommendations (e.g., assessing calibration data uncertainty) are taken into consideration when assessing the model outputs in this chapter and discussed in more detail in Chapter 7.

4.2 Upper Ehen Hydrological Modelling

This section contains information about the functioning of SHETRAN as a physically based spatially distributed model and describes the benefits and limitations of using this particular model for answering the project-specific research questions. Two alternative lake models are proposed and run using inputs from SHETRAN. Initial results using lake outflows from previous studies are presented as a means validating the modelling methodology. Final results and simulations are run in Chapter 6 using the outputs of the geomorphic model runs in Chapter 5.

Figure 54 outlines the main steps in the modelling process which are then each described in more detail in the following subsections of the current chapter. The methodology has been devised with the intention of creating a modelling procedure that can be easily repeated to allow the outputs to be verified and to allow the same methodology to be readily applied to different catchments by other modellers. With this in mind, the majority of data was obtained from freely available data sets. Where this is not the case, alternative data sets are recommended. Additional field data was gathered (Chapter 3) to provide empirical lake inflow data which was not available. In catchments with water supply reservoirs historical inflow data may be readily available which could potentially negate the need for additional hydrological monitoring.

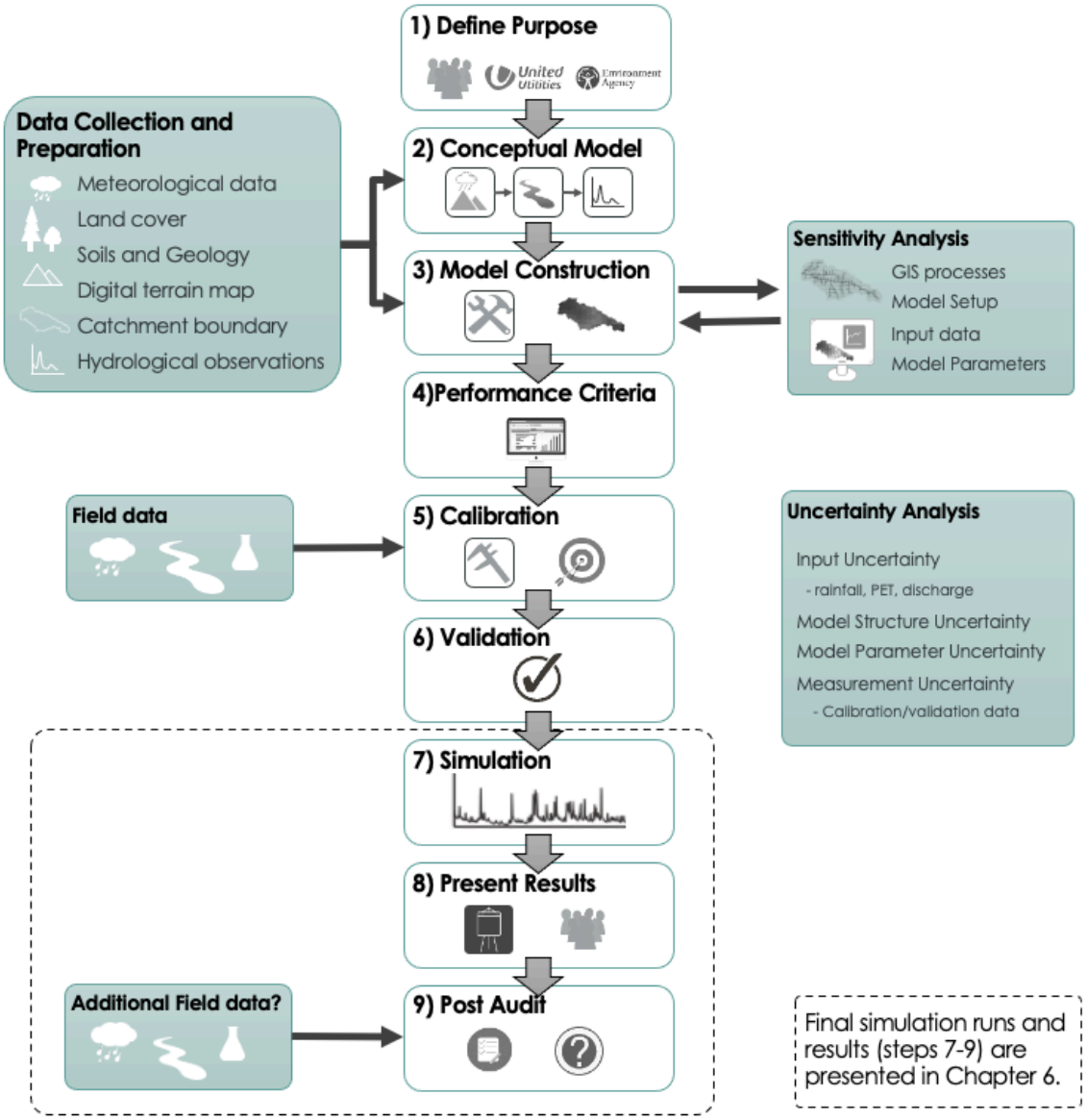


Figure 54. Modelling steps described in the following section of this chapter. Adapted from (Refsgaard, 1996)

4.2.1 *Define Purpose*

The primary purpose of the modelling task is to better understand the natural hydrological behaviour of the Upper Ehen catchment and estimate hydrological changes under a range of weir removal scenarios. To achieve this is it necessary to demonstrate that the model can accurately replicate the current hydrological behaviour of the catchment. This will then allow the natural flow regime for the same period to be modelled under a range of future scenarios. Finally, the modelling will allow changes to the storage and lake level to be analysed.

4.2.2 *Conceptual Model*

The upper Ehen catchment can be characterised by three distinct hydrological zones, inflows into Ennerdale Water, Ennerdale Water and discharge from Ennerdale Water and Ben Gill into the River Ehen. To represent this, a catchment model was used to provide inflows into Ennerdale Water, whilst a separate lake model was used to account for the flow attenuation of Ennerdale Water and flow regime management features at the lake outlet. The different elements of the upper Ehen are shown in the conceptual model (Figure 55) and described in more detail below.

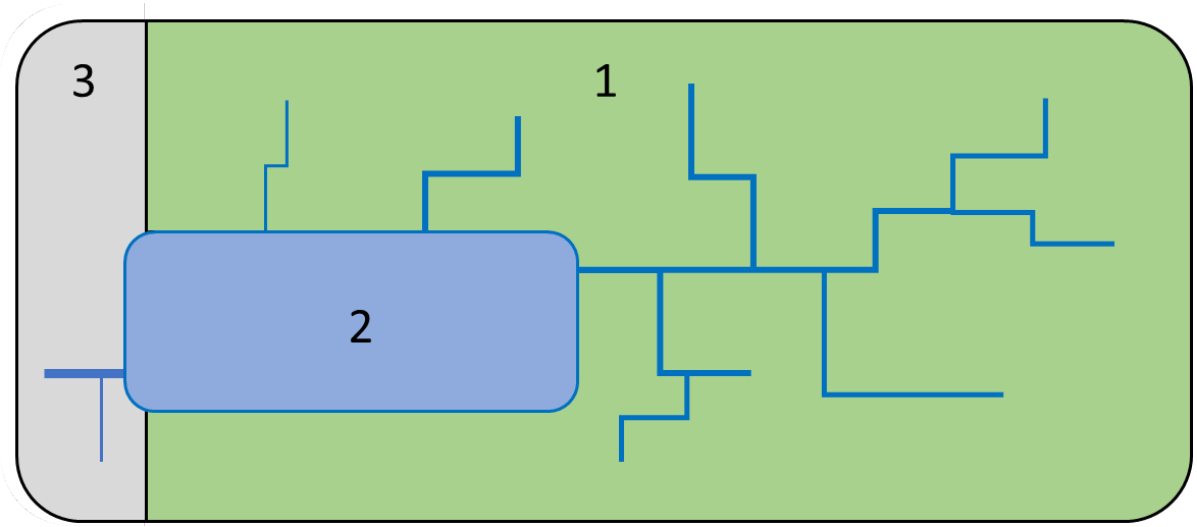


Figure 55. Conceptual model of the Upper Ehen catchment and the three components to be modelled: 1) Upper catchment and inflows into Ennerdale Water; 2) Ennerdale Water; 3) downstream discharges into the River Ehen

Component 1 - The upper catchment containing all the inflows into Ennerdale Water

The upper Ehen is the area of the catchment least impacted upon by human activities. No surface-water abstraction takes place and there no flow regulation structures in the main river and its tributaries at present. Prior to this study, no continuous discharge data was available for this part of the catchment

Component 2 - Ennerdale Water

Lakes naturally serve to dampen the hydrological regime of the observed inflows (Chadwick *et al.*, 2013; Chin, 2013). Ennerdale water has been impounded for flow regulation and abstraction since the 1800s. The water level, volume of water stored and rate of discharge from the lake are thus all influenced by the physical dimensions of the impounding infrastructure in addition to the abstraction and flow regulation.

Component 3 - The outflow from Ennerdale Water and Ben Gill

The area immediately downstream of the impounding weir consists of a small stretch of the main River Ehen and the ephemeral tributary stream, Ben Gill. Flows from Ben Gill thus need to be incorporated into the final discharge timeseries as the observed data is taken from a gauging station 800m downstream of Ennerdale water.

4.3 SHETRAN Model Construction

SHETRAN is a physically based spatially distributed (PBSD) hydrological model which can be run at wide range of spatial and temporal scales however domains with high spatial resolution are computationally demanding and can negatively impact model stability and model run times. Spatial resolution of the model is decided by the user but is likely to depend on the resolution of available data, the purpose of the modelling study, the size of the modelling domain and the quantity and duration of model runs required.

The following sections outline the data requirements of SHETRAN and the range of model set up options available. The final model setup is presented along with a justification for the choices of data and resolutions used.

4.3.1 *SHETRAN Data Requirements*

SHETRAN is available in a number of versions that have different capabilities however all variations require a minimum quantity of 5 input files: a catchment mask, an average Digital Elevation Model (DEM), a minimum DEM, rainfall data, potential evapotranspiration and land cover (Figure 56). These are input into the model as text files and can be generated using GIS applications or software such as excel. For this study a combination of both were used to format and quality control the data for modelling. A lake mask may be needed if the catchment has large water bodies and temperature data can also be included if snow melt is a significant element of the catchment hydrology.

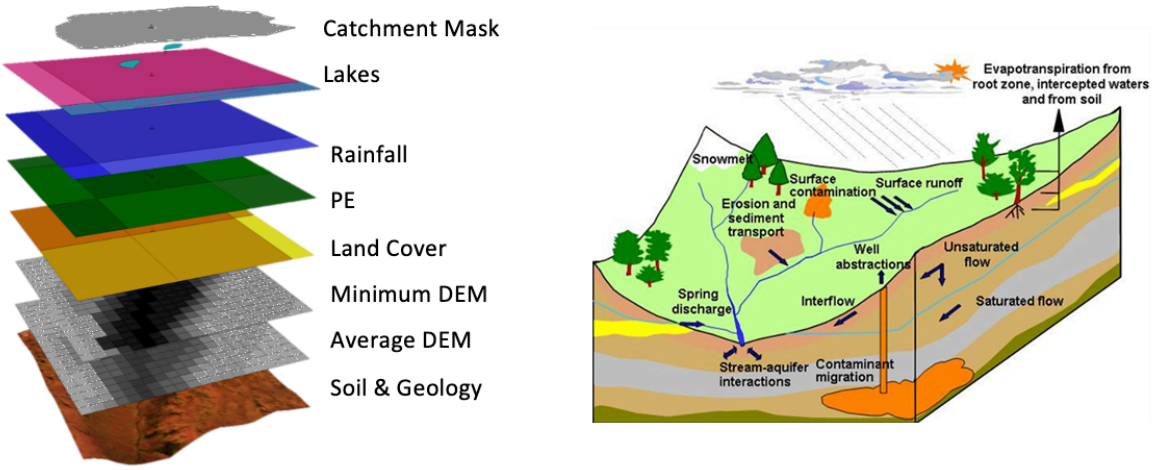


Figure 56 Visual representation of the input data requirements (Lewis *et al.*, 2018) *and processes modelled by SHETRAN* (Newcastle University, 2018)

4.3.2 *Modelling Domain*

The catchment covers an area of approximately 44.5km^2 however to facilitate data processing and ensure consistent reproducibility of modelling runs and results a fixed modelling domain was used. An 8 by 14km grid was created in ArcMAP to define the modelling area and aid the visual inspection of model input data (Figure 57). A resolution of 200m was chosen as it offered the best compromise between the varying resolutions of available data (5m – 1km) whilst taking into account computer processing requirements and speed of completing multiple modelling runs.

Models were run at 500m 250m, 200m and 100m resolutions to assess the model sensitivity at each scale. The results of this assessment are outlined in section 4.3.10.

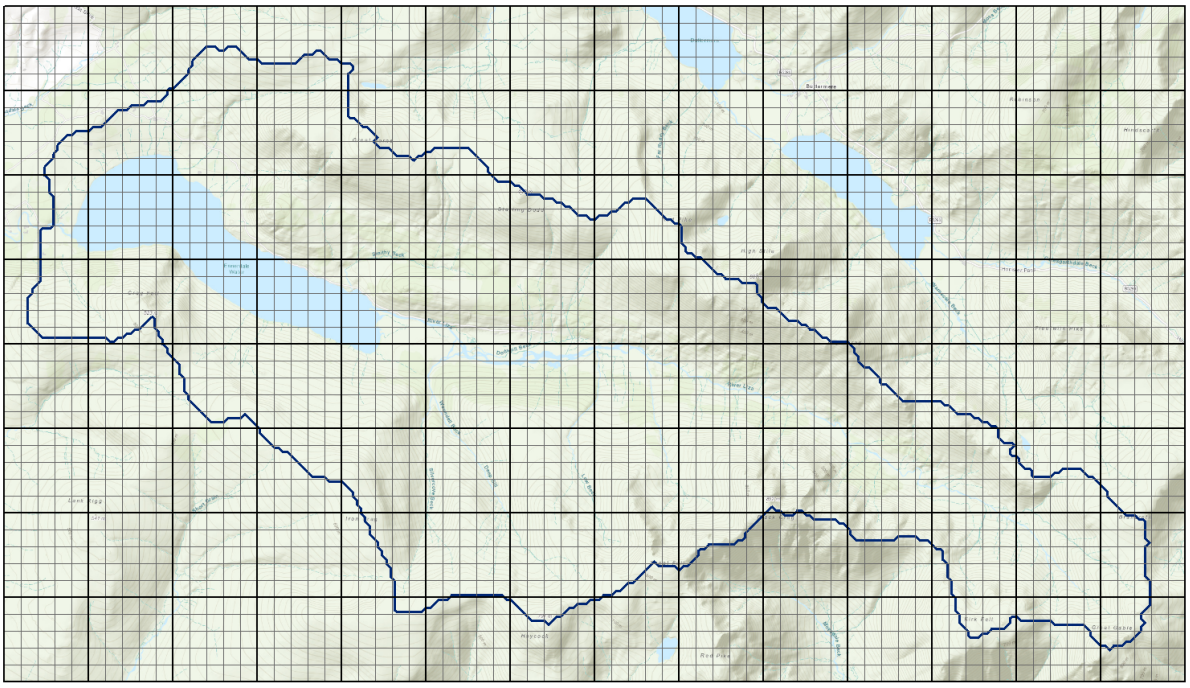


Figure 57. Upper Ehen catchment contained within the 8km by 14km modelling domain. The grid occupying the modelling domain facilitated the visual verification of model input data.

Bounding Coordinates = Left 308,000 – Right 322,000 – Top 518,000 – Bottom 510,000

Area = 112km^2 (8km by 14km).

Total cells = 2,800 (40 x 70) at 200m resolution.

4.3.3 *Catchment Mask*

Two catchment masks were produced to assess the sensitivity of the model to different catchment dimensions. The first was created using the catchment boundary shapefile downloaded from the NRFA website. This polygon was converted to a raster (text file) of 200m-by-200m cell size within ArcGIS. For reference, an 8km-by-14km grid of 200m resolution was also created using the fishnet function in ArcGIS. Of the 2,800 cells in the initial modelling domain, the catchment mask covers 1,128 cells with a total area of 45.12 km^2.

The second catchment mask was created by obtaining a catchment boundary from the DEM used in step 4.3.4. This was carried out to ensure consistency within the data processing and identify any significant differences in the hydrological performance of the final model. Once a catchment boundary had been created using the watershed tool in ArcMap a second mask was created following the same steps outlined above. This produced a smaller catchment mask of 44.64 km^2 (1,116 cells) and with slightly different dimensions (Figure 58). The hydrological implications of these variations are outlined in Section 4.3.10.

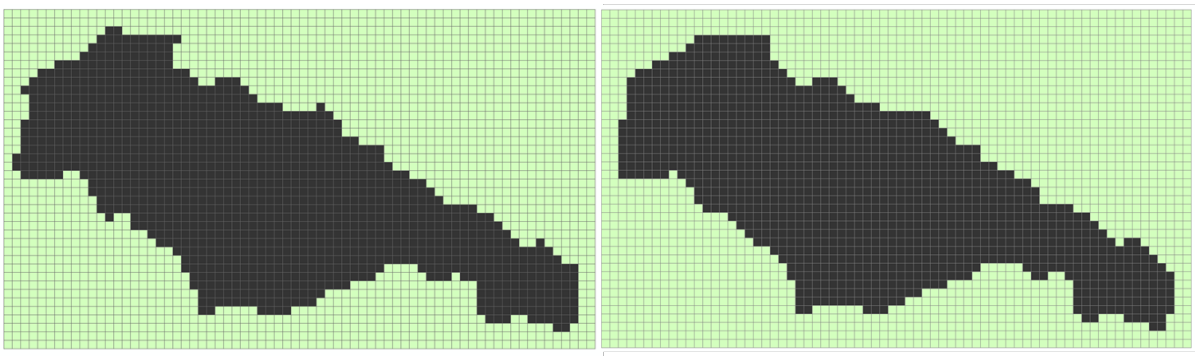


Figure 58. Catchment masks for the Upper Ehen catchment obtained using existing NRFA catchment boundary shapefile (left) and ArcMAP watershed tool (right).

4.3.4 *Digital Elevation Model*

A 5m resolution digital elevation model (DEM) was downloaded from data.gov and then clipped to the modelling domain in ArcGIS using the clip tool (Data Management -> raster -> clip). The grid created in Section 4.3.2 was used as the clipping template. This 5m resolution DTM was subsequently converted to a 200m-resolution raster (Figure 59) using the resample tool in ArcGIS (Data Management -> raster -> resample). Resampling was performed using the bilinear method as this is most appropriate for continuous data.

A second DEM was generated in ArcGIS using the aggregate tool (Spatial Analyst -> Generalization -> aggregate). Applying this to the original 5m raster file, selecting the mean value option and applying a cell factor of 40, obtained a final raster resolution of 200m. The impact of these geoprocessing techniques on the model performance was then assessed (see section 4.3.10).

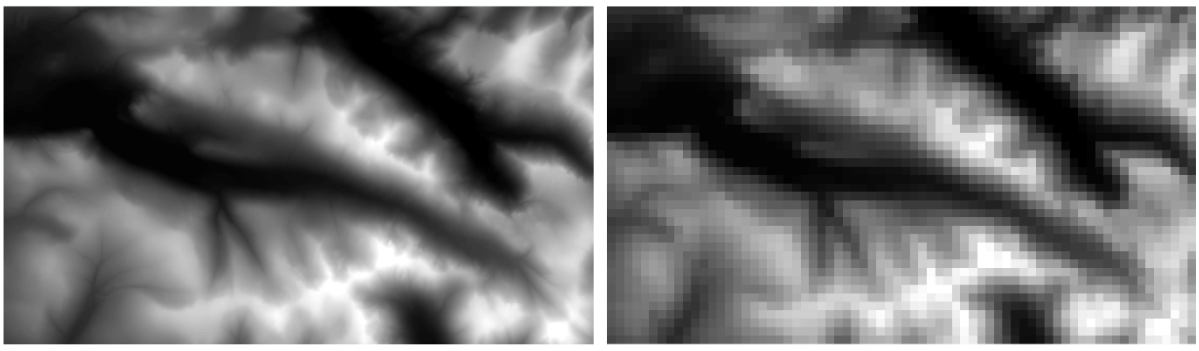


Figure 59. The original 5m resolution DEM of the catchment and surrounding area (left) and the 200m DEM obtained from the resampling process (right). Darker colours represent the lowest elevation in the catchment.

Finally, a DEM was generated in ArcGIS to obtain minimum elevations of the individual grid cells. This was achieved by applying the aggregate tool (Spatial Analyst -> Generalization -> aggregate) to the original 5m raster file and selecting the minimum value option and applying

a cell factor of 40 to obtain a final raster resolution of 200m. No alternative approach for obtaining this DEM was available.

4.3.5 *Land Cover*

Land cover information was obtained from the Land Cover Map 2015 (Rowland *et al.*, 2017) produced by CEH. In its raw form, the data is available at 25m resolution and contains 21 classes of land cover (Figure 60a). The land cover classes were then reclassified using the reclassify tool (Spatial Analyst -> Reclass -> Reclassify) in ArcMap to produce a subset of seven land cover types for SHETRAN. The final step in the process was to resample the raster from 25m to 200m resolution (Figure 60c).

SHETRAN is able to incorporate any number of land use classifications however this should be set against the available data and knowledge of how different vegetation types may influence the hydrological behaviour of the catchment. Furthermore, using a larger number of land use classifications only adds to the model complexity and increases the permutations for model calibration without necessarily resulting in improved model performance.

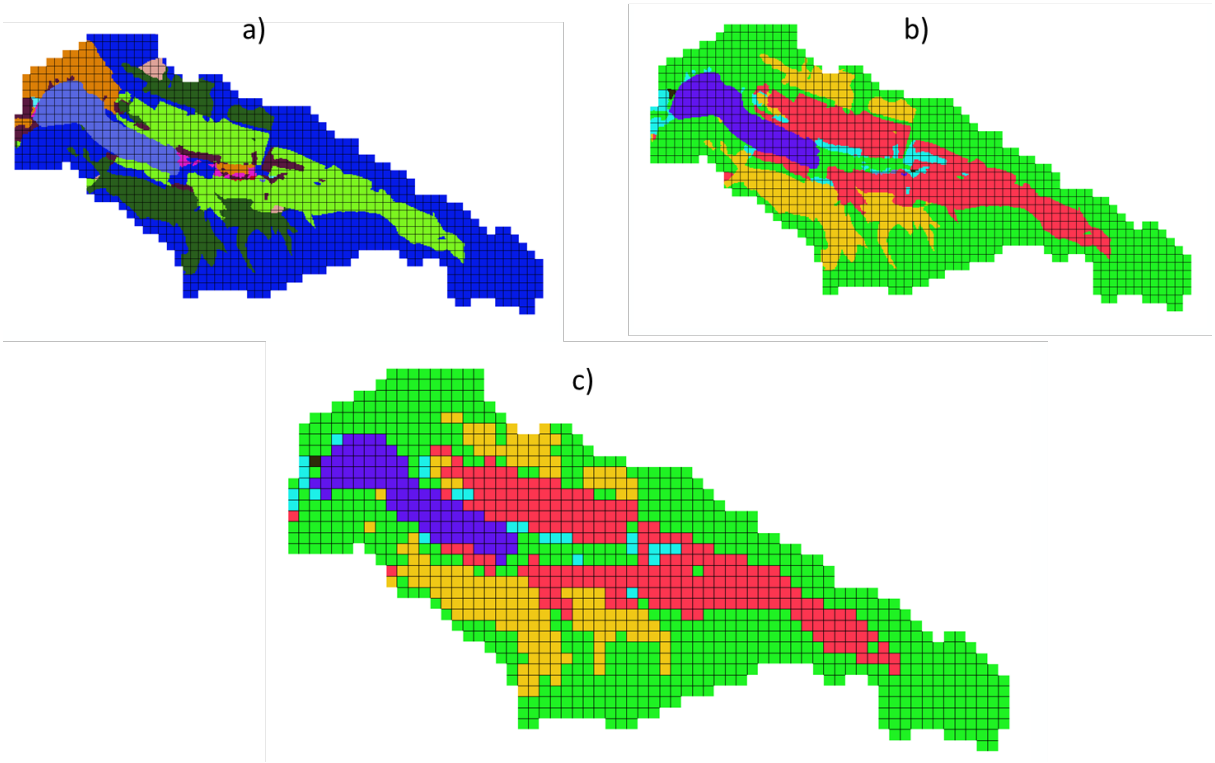


Figure 60. Original land cover data (21a) and processed data for use in SHETRAN (Rowland *et al.*, 2017).

The land cover map produced by CEH in 2007 (Morton R.D.;Rowland, 2014) was also considered for use in the modelling process. Comparisons were made between the two data sets in both their original forms, their reclassified forms and at differing resolutions (Table 11 and Figure 61).

Table 11. Comparison between CEH 2007 and CEH 2015 land cover classsifications for the Upper Ehen catchment showing area in km² and as percentage of catchment.

Classification	SHETRAN LC2007		SHETRAN LC2015	
	Area km²	%	Area km²	%
Arable	0.28	1%	0.04	0%
Bare Ground	6.00	13%	3.12	7%
Grass	13.64	31%	25.12	56%
Deciduous Forest	2.64	6%	1.00	2%
Evergreen Forest	7.4	17%	9	20%
Shrub	14.40	32%	6.36	14%
Urban	0.28	1%	0	0%
Total	**44.64**	**100%**	**44.64**	**100%**

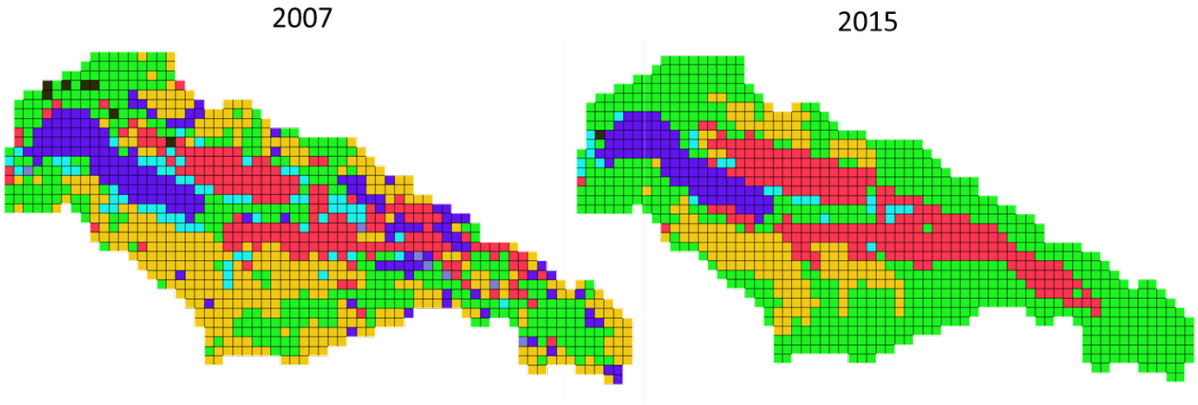


Figure 61. Comparison of SHETRAN land cover classifications using the 2007 Land Cover Map (left) and LCM 2015 Land Cover Map as source data (Morton R.D.;Rowland, 2014; Rowland *et al.*, 2017).

Land cover data from an earlier period was also considered as change to vegetation cover may have influenced the catchment hydrology. However, additional data was not required. A map of land cover change for the UK covering the period 1990 – 2015 (Rowland *et al.*, 2020) was downloaded from the CEH data portal and examined. This showed no land cover changes for the modelling domain over the 25-year period. Landcover prior to 1990 was not analysed as the hydrological modelling period only dates back as far as 1995, however this may be a consideration for any longer-term studies.

4.3.6 *Soils*

A nation-wide SHETRAN modelling study of UK catchments by Lewis *et al.*, 2018 used a European 1km resolution soils map from the European Commission Joint Research Centre (Liedekerke *et al.*, 2006) to develop a system to automatically set up SHETRAN for UK

catchments. This data is freely available and was shown to perform well at catchment-scale for a range of catchment sizes and types. This was used in the initial model set up as a starting point for model construction and to test model performance (Figure 62a).

An alternative soils classification is the NATMAP – National Soils Map (Figure 62b) and forms part of the Land Information System operated by Cranfield University (LandIS, 2020). Soil data is available under license in a variety of formats and for a variety of uses however the data must be paid for.

The NATMAP HOST (Hydrology of Soil Types) data set is derived from the National Soil Map. It was developed by CEH, The James Hutton Institute and the National Soil Research Institute. The data set is designed to be a concise and easily interpreted description of the hydrological responses of soils typically found across England and Wales (LandIS, 2020).

Data for this study was obtained under license without a fee as the STREAM doctoral program is run by Cranfield University. However, as previously stated, the freely available European Soils data has been shown to perform well across a range of UK catchments and serves as an adequate substitute.

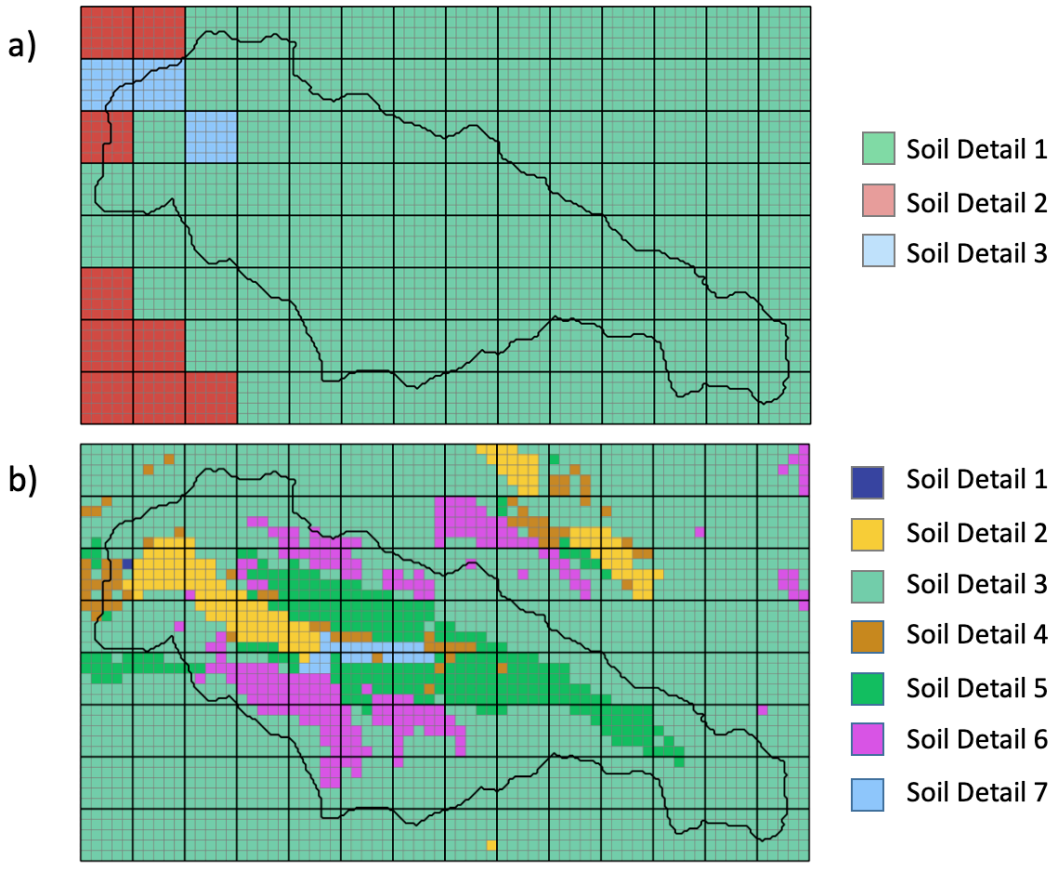


Figure 62. Soil maps showing the distribution and resolution of (a) the European Soil Map (Liedekerke et al.*, 2006) and (b) the NATMAP – National Soils Map (© Cranfield University) considered for the final SHETRAN model.*

4.3.7 *Lakes*

The lakes mask was obtained using the same input data as the Land Cover map. By using the reclassify feature in ArcGIS it was possible to classify all land cover types as no data, leaving only the freshwater category visible. A check was then carried out to compare the surface area of the lake generated in this manner against the known surface area of the lake.

The lake cells were then edited manually as the surface area of the lake cells generated from the GIS processing was greater than expected. The reason for this may be due to reclassifying the land cover map from a 25m resolution to a 200m resolution. During this process, areas around the lake margins may be incorrectly classified as water.

The lake mask is used by the SHETRAN model to identify which river links require different Strickler values. The Strickler value is a roughness coefficient used to replicate the attenuating effect of a lake or area of surface water by slowing the flow of water through the specified river links in the model.

4.3.8 *Meteorological Data*

SHETRAN requires meteorological data in the form of rainfall, potential evapotranspiration and minimum and maximum temperatures. Temperature data is only necessary if snow melt is considered a significant factor.

Rainfall Data

Rainfall data was available for the majority of the modelling period as 1km gridded daily data. This was downloaded from the CEH GEAR repository (Tanguy et al., 2019) and extracted using a python script to obtain the gridded data for the modelling domain. The latest revision of this dataset goes up to the end of 2017 (Figure 63), so the rainfall data collected from the field campaign (Chapter 3) was used to extend the timeseries and cover the entire observation period. Several methods for combining the two data sets were trialled in order to provide a consistent single modelling run and to allow future model users the option to update and replace the existing data with the CEH GEAR rainfall as and when it becomes available.

A comparison was made between the field data and the CEH GEAR data for a 45-period when all three rain gauges were recording data (Figure 65). The average of the three rain gauges during this period closely matched the average for the CEH GEAR data over the modelling domain.

When comparing the values for the specific 1km^2 cells for each gauge however there was some variation between the field data and the CEH GEAR estimations (Figure 64). At site one

observed data overlapped with the CEH GEAR dataset for over three months. During the period from 20/09/2017 to 31/12/2017 the rain gauge at Beckfoot recorded 789 mm of rainfall, this compared to an estimated 785 mm from the CEH GEAR data set. This represents a difference of less than 1%. At site two, Low Gillerthwaite, the tipping bucket rain gauge recorded 1,085mm of rain during the same period. Compared to an estimated 1,041 mm for the corresponding CEH GEAR grid, this is a difference of 4%. At the third rain gauge, Great Cove Sheepfold, the observation period only overlapped with the CEH GEAR data set for 45 days. During this period the tipping bucket rain gauge recorded 753 mm of rainfall compared to an estimated total of 598 mm from the CEH GEAR dataset.

1702	1779	1860	2000	2205	2288	2198	2315	2372	2165	2317	2968	3230	3167
1712	1796	**1817**	1966	2153	2481	2480	2782	2866	2609	2526	2817	3283	3417
1878	1936	1996	2012	1967	2187	2474	2947	3198	3249	2934	2932	3212	3608
2045	2220	2345	2302	2146	**2269**	2395	2593	2831	3279	3341	3023	3107	3854
2105	2263	2544	2536	2373	2474	2642	2870	2931	3233	3634	3315	3161	3975
1984	2209	2419	2467	2688	2830	2822	3176	3619	3421	3645	3820	3782	3945
2006	2266	2311	2467	2910	2955	**2995**	3444	3694	3366	3358	3709	4089	4050
1878	2066	2129	2304	2700	2854	3077	3216	3304	3004	2701	2784	3411	3573

Figure 63. Gridded daily rainfall data set showing historic annual average gridded rainfall in mm from 1994 – 2017. Data Source: (Tanguy *et al.*, 2019)*. Text in bold and double underlined represents cells where rain gauge observations were made for 2017 -2019.*

Three approaches were used to make the field data compatible with the existing long-term data (Figure 66 a-c) and prevent the need for breaks in the modelling run and inconsistencies in the model set-up. Firstly, (Figure 66a) the average values for the three rain gauges were distributed across the modelling domain using the equivalent percentage distribution as the historic CEH GEAR data. Secondly, simple Theisen polygons were constructed with the observed values for each rain gauge being applied to the corresponding grid squares. Finally, the values for each range gauged where distributed within the Theisen polygon according to the distribution of the CEH GEAR data. A single average value distributed uniformly over the entire catchment was not considered as it was not consistent with the format of a longer-term dataset and didn't reflect the spatial variability of rainfall of the site.

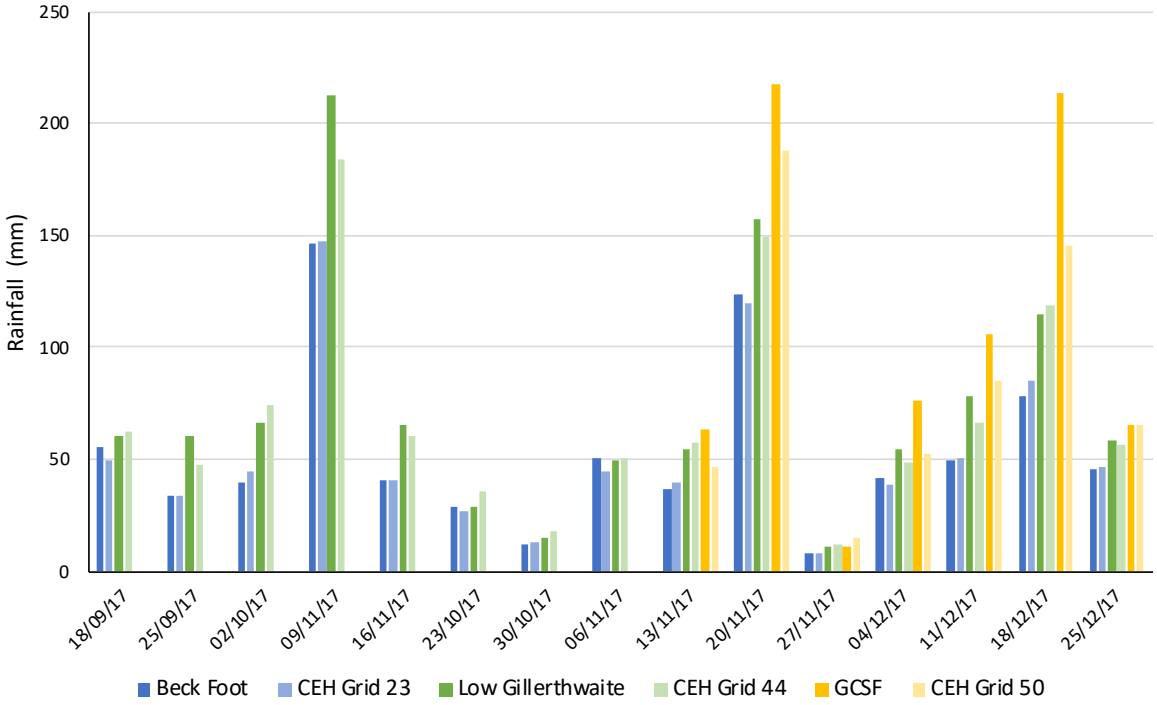


Figure 64. Weekly rainfall totals for each rain gauge and estimates from the CEH GEAR data set for the corresponding location. Darker colours represent observed values.

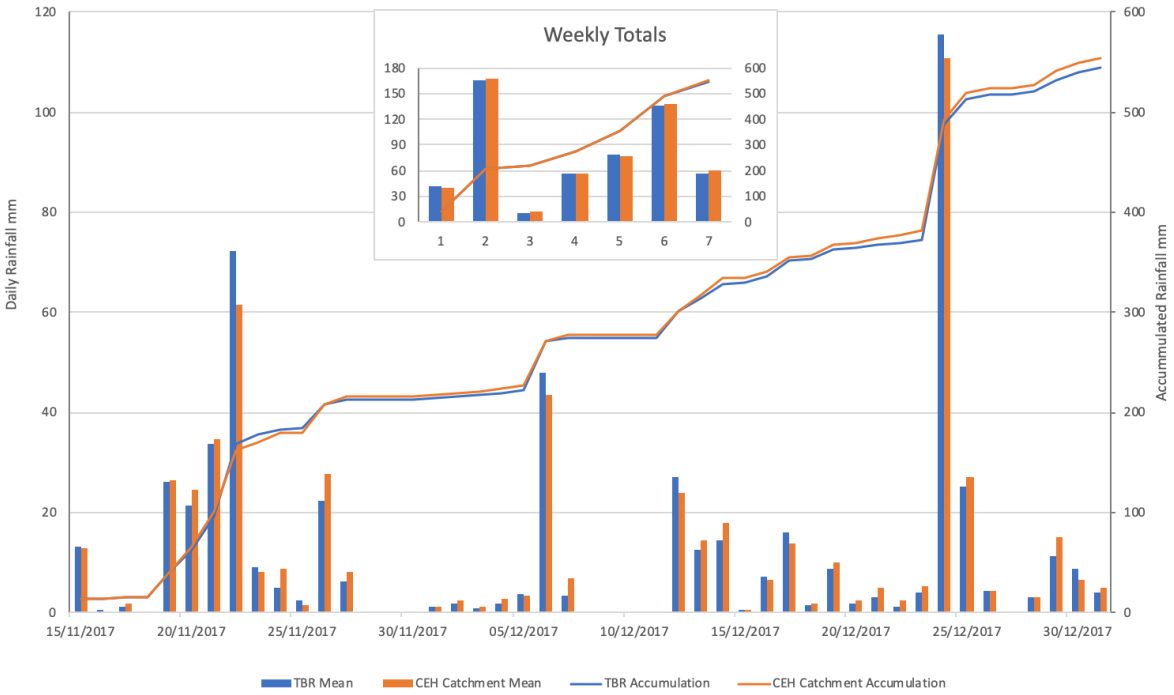


Figure 65. Average rainfall data across the three monitoring locations compared with the CEH GEAR data set for the 45-day overlapping period. Average daily rainfall values are shown in main graph with average weekly totals in centre graph.

To avoid multiplying the calibration steps unnecessarily, a final rainfall configuration was selected before the calibration process. An identical uncalibrated model was run three times using each of the data sets. All rainfall sets performed reasonably well, however, option 3 overestimated the total water balance of the catchment whilst option two underestimated it.

The first option (Figure 66a) produced a water balance accurate to within 2% of the observed data for the Upper Ehen catchment for the 2017 – 2019 period.

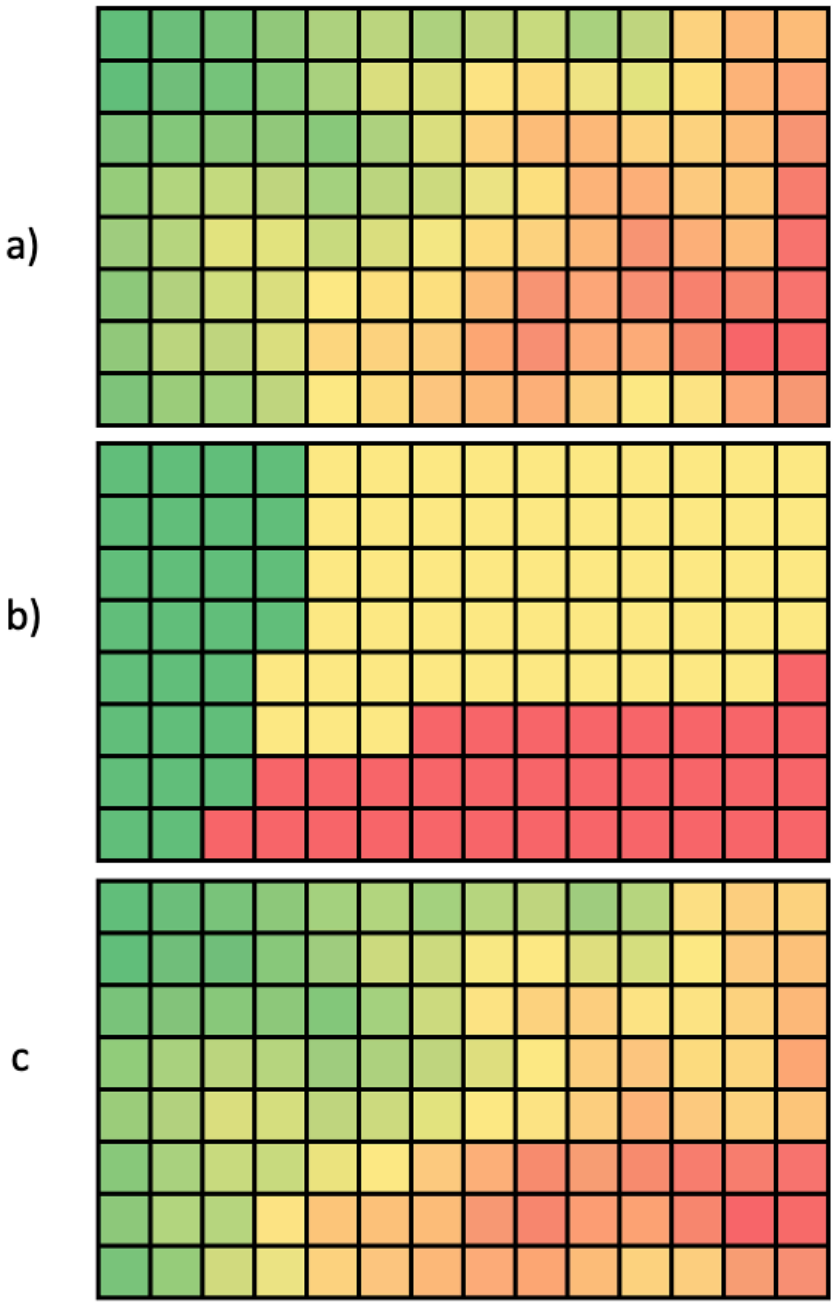


Figure 66. Rainfall data configurations tested using observed rainfall data with the same gridded format as CEH Gear data using different spatial distributions.

Potential Evapotranspiration

Potential Evapotranspiration (PET) values were also obtained in a 1km gridded format at daily timesteps. This was available from the CEH as part of the Climate, Hydrology and Ecology research Support System (CHESS) data set (Robinson *et al.*, 2017). This was however only available until 2015 (Figure 67) and needed to be extended to cover the remaining period to 30/09/2019. Two options were considered for this. Firstly, average daily

values were used for the preceding 20 years and repeated for the following 4 years. Secondly, data was obtained as a single daily PET value for the entire catchment for the missing period from the MOSES data service. This was initially used as a single value and then distributed across the catchment using percentage distribution of the observed data for the preceding period, in the same way as the rainfall data.

The range of daily PET values was measured and the sensitivity of the model to the three different data formats was assessed. The different data and configuration had limited impact on the overall model performance and water balance (Table 13) however the observed single PET value using the CHESS distribution was selected in the final model setup. This retained a consistent approach for the model set up and used estimations based on observed meteorological data rather than historic average values.

424	393	359	425	459	444	356	376	394	399	300	423	403	383
394	394	385	346	402	419	422	378	351	398	386	350	444	374
409	411	411	432	417	428	437	393	372	365	415	399	422	429
398	363	397	392	380	399	400	386	423	431	352	322	296	325
454	405	406	365	354	341	343	351	339	455	455	316	342	453
397	402	456	409	394	324	364	363	349	377	375	381	367	405
410	452	402	415	448	395	389	368	451	457	477	379	398	392
425	433	374	403	433	455	448	456	377	409	389	397	449	450

Figure 67. Average annual potential evapotranspiration values for 1995-2015. Source:(Robinson *et al.*, 2017)

Average Annual PET values from 1995 to 2019 ranged from 296 mm to 477 mm per year over the modelling domain. Areas with higher average annual PET values correspond with the higher elevations in the catchment which may be due to increased wind speeds in the more exposed upper areas of the catchment. Controlled laboratory experiments have shown that windspeed increases first stage evaporative processes in soil water (Davarzani *et al.*, 2014).

4.3.9 *Additional Setup Options*

SHETRAN allows the user to make a number of additional choices prior to running the model. This includes adjusting the density over the river network, adjusting the minimum

elevation change between consecutive river links and controlling the depth of the river channels in relation to the surrounding grids (Figure 68).

The need to adjust the river network density depends on the modelling objectives. In this instance, a network that more accurately represented the upper catchment was required to ensure the discharge data collected during the monitoring campaign could be used to calibrate the model.

The river network density has an impact on the model stability and the computational demands of different model runs. A denser network requires more calculations and is thus more computationally demanding and slower to run. The impact on model performance was also assessed using a range of grid densities (Section 4.3.10).

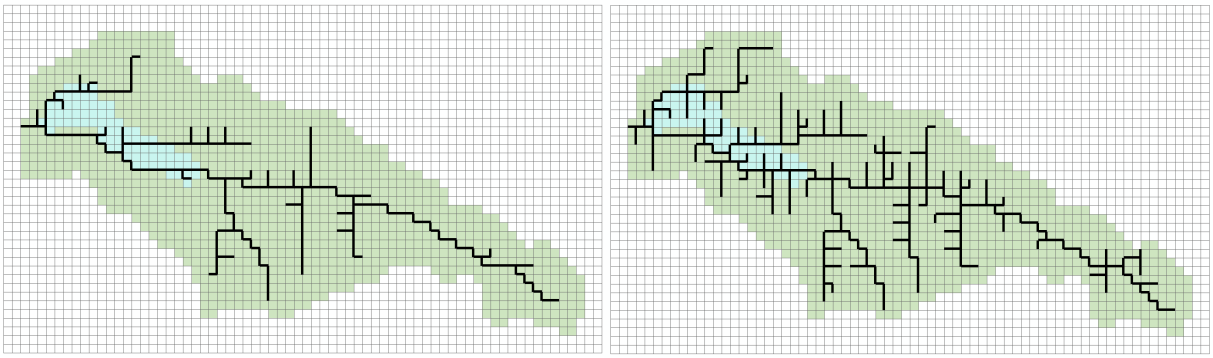


Figure 68. Comparison of different river network densities assigned prior to each model run. The model accumulates flow from 10 cells (left) before creating a river channel compared to four cells (right).

Abstraction

SHETRAN has the option to include abstraction however this is more commonly used for groundwater abstractions. Abstraction from Ennerdale Water comes directly from the lake. Model runs were trialled using abstraction data from river links within the model however this had limited impact on the downstream discharge. It was therefore decided that abstraction should be subtracted from the discharge timeseries when assessing the overall water balance and model performance.

4.3.10 *Sensitivity Analysis*

The impact of the following setup variations and input data formats are quantified in the following tables (Table 12 and Table 13). Based on this information the final setup details are outlined. Due to the model complexity, the sensitivity of individual elements of the model setup were assessed in isolation to avoid excessive multiplication of model runs caused by a growing number of possible setup combinations. By simply accounting for the setup and input data options outlined below, 3072 different model runs would have been possible.

During this stage of model evaluation, the main assessment criteria was the impact on the overall water balance and Q_{95}. The NSE was also calculated however as the model was not yet set up to replicate the managed flow regime this was a secondary consideration at this preliminary stage of the modelling process.

Model Construction

The water balance of the model was assessed by comparing the modelled discharge against observed discharge and abstraction as observed discharges were lower than modelled discharges due to abstraction. In general, the aspects relating to the physical model structure had minimal impact on the overall water balance. The catchment mask had some impact as the different shapes and surface areas captured different rainfall intensities in different areas of the catchment. This also had a small influence on the percentage of land use cover. The final model setup options are highlighted in blue along with comments and justifications for their choices. Although variations in model outputs were limited, this was an important part of the modelling process as it eliminated any doubts about the final model setup and demonstrated a rigorous approach to testing the model.

Table 12. Model setup options tested during the model construction phase and the individual impact of each on the overall model performance

Setup Element	Options Tested	Run Period	Water Balance	Q95	Q95 (obs)	Comments
Model Resolution	500m	1975-2015	1.00	0.64	0.94	Input data was available at spatial resolutions from 5m - 1km. A trade-off was made between loss of data, model compexity/run-time and accurate representation of the river network. Multiple attempts were made to run the model with at 100m resolution. Due to instability no runs were succesfully competed.
	250m	1975-2015	1.00	0.81	0.94	
	200m	1975-2015	1.01	0.92	0.94	
	100m	1975-2015	NA	NA	0.94	
Cathment Boundary	ArcMAP derived from 5m DEM	1975-2015	1.01	0.92	0.94	Minimal differnece between masks and impact on model performance. Option 1 was chosen for the final mask as its area (44.6km2) is closer to the recorded area on the NRFA website (44.5 km2) and produces a better water balance.
	CEH cathment Boundary	1975-2015	1.02	0.93	0.94	
GIS processing of 5m DEM	Aggregate	1975-2015	1.00	0.86	0.94	Resample approach was chosen although overall impact on model perfoamce was minimal
	Resample	1975-2015	1.00	0.93	0.94	
River Grid Squares	10	1975-2015	1.00	0.78	0.94	Model run speed and stability was impacted upon by a denser river network. To account for instability, river chanel depths were changed from 2m to 2.5m.
	8	1975-2015	1.00	0.81	0.94	
	6	1975-2015	1.01	0.86	0.94	
	4	1975-2015	1.01	0.92	0.94	

Input Data

The largest difference to overall model performance and water balance were caused by changes to input data. These were tested over a range of timescales. The most significant factor in terms of impact on the water balance was the final spatial distribution of the rainfall data collected from the field monitoring. Using the average of the three gauges and applying the percentage distribution obtained from the CEH GEAR data set provided the closest match to the overall water balance. These initial tests further highlight the suitability of the chosen data sets and the SHETRAN model for the aims of the modelling task

Table 13. Different data inputs considered for the final model setup and their individual impact on the overall model performance

Input Data	Options Tested	Run Period	Water Balance	Q95	Q95 (obs)	Comments
Land Cover	LCM 2007	1975-2015	1.01	0.94	0.94	Using the different landcover maps had very little ipact on the overall model perfomance. It was therefore decided to use the most recent map.
	LCM 2015	1975-2015	1.01	0.92	0.94	
Soil Map	1km European	1975-2015	1.01	0.92	0.94	Despite producing a lower Q95 the NATMAP HOST soild dat was chosen primaril as it offered greater flexibility to fine tune dfiferent ares of the model druing the calibration process.
	25m HOST	1975-2015	1	0.48	0.94	
Rainfall Distribution	Avergae P using CEH % Distribution	2017-2019	1.02	0.69	0.85	To cover the missing CEH GEAR data from 1/1/2018 to 30/9/2019 option 3 was used. This had the advantage of using a spatial rainfall distribution based on observed and historic data and still retain the core information from the field data by distributing the value spatial within each polygon
	Single value Thiessen Polygons	2017-2019	1.03	0.69	0.85	
	Thiessen polygons distributed on CEH values	2017-2019	1.01	0.69	0.85	
	CEH distribution using 3 rainfall values	2017-2019	0.96	0.66	0.85	
Potential Evapotranspiration data format	Historic average single value	2017-2019	1.00	0.64	0.85	For consitency with the data format the avergae PE value for the entire catchment was redsitributed using the same % distribution as the historic Chess PE values. Although repeating the historic values achieved the best model performance this wasn't an accurate representation of the observed data
	Observed single value	2017-2019	0.99	0.60	0.85	
	Average PE distributed using Chess %	2017-2019	1.00	0.61	0.85	

4.4 Lake Model and Level Pool Routing

A separate lake model was necessary in order to carry out a detailed analysis of the changes to lake level and the lake storage discharge relationship as this is not a key feature of the SHETRAN model. SHETRAN has an option to include a weir equation at different grid elements in the model to replicate the discharge relationship observed at Ennerdale. However, this alters the topography of the catchment and doesn't model the lake level and volumes of stored water as effectively. It was therefore decided that a 1D model would be used to model the storage and discharge in the lake based on the physical infrastructure (Figure 69 & Figure 70), abstraction regime and compensation flow requirements and the operation of the fish pass penstock. SHETRAN was used to provide the inflows into the lake model.

The fish pass has also changed from a manually operated system to a fully automated system controlled by telemetry software meaning that a simple weir equation placed into the SHETRAN model may have been cumbersome, more time consuming and required expert knowledge of the model to allow for future modifications.

The relationship between lake stage and river discharge was obtained from observed data. However, if this data is unavailable, the model can also use fundamental hydraulic principles to estimate the discharge through the fish pass and over the impounding weir (Figure 71). This offers great flexibility as a number of outflow designs can easily be substituted in place of the existing weir dimensions to model multiple weir removal scenarios including retaining the existing infrastructure with no abstraction or flow management and complete removal with a natural outflow.

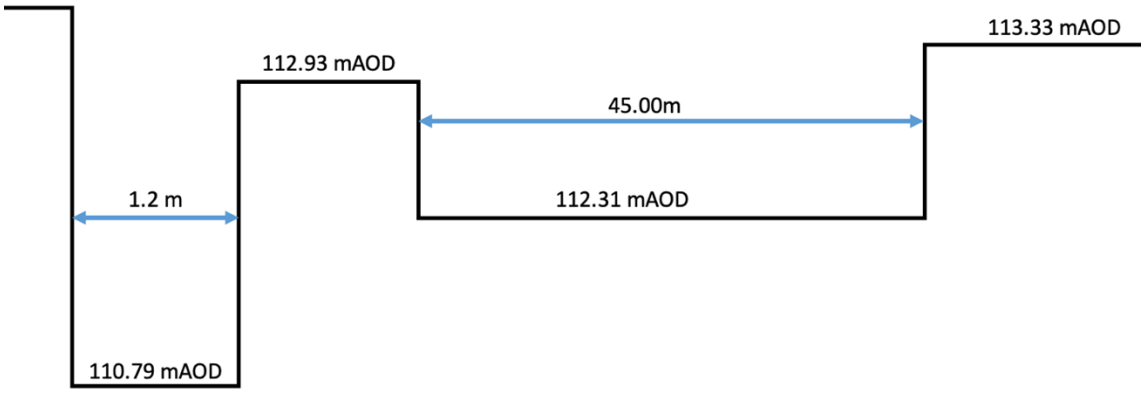


Figure 69. Cross sectional representation of the outflow profile facing upstream towards the weir and adjacent fish pass at the outlet of Ennerdale Water.

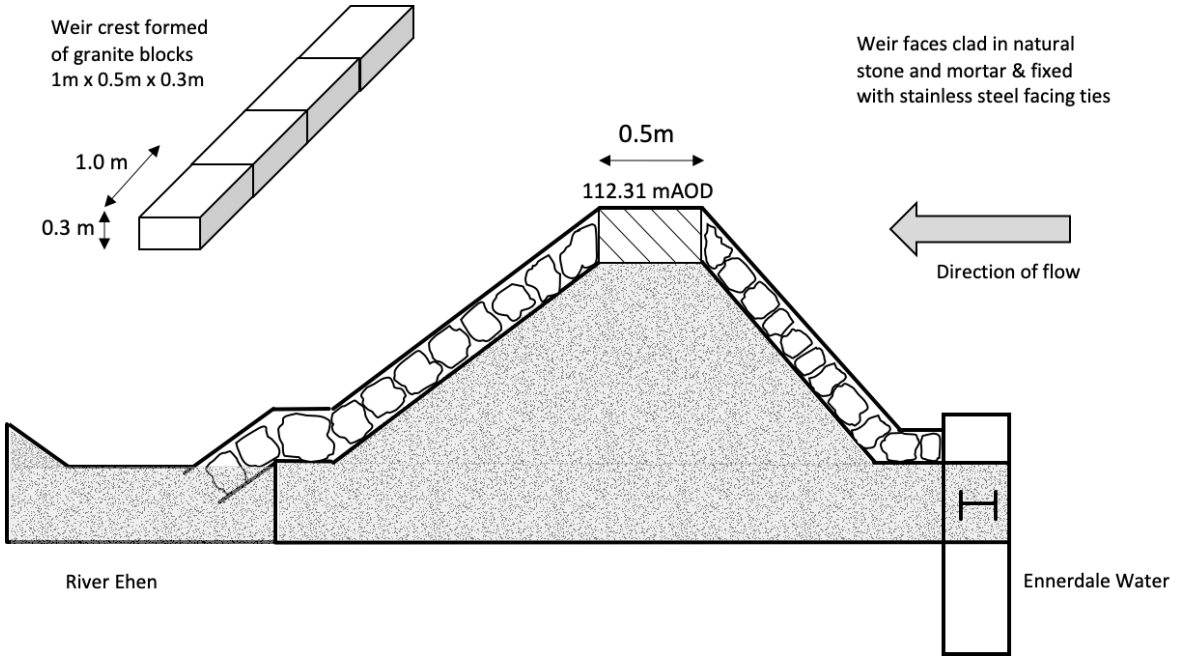


Figure 70. Simplified cross section of weir masonry and construction details based on original drawings by Betchel Water Technology Ltd 1994

The equation for calculating the discharge over the weir is based on the physical properties of the weir, the water level of the lake (head) and a discharge coefficient for broad crested weirs.

$$Q_w = C\ L\ h^{\frac{3}{2}}$$

Where: C = broad crested weir coefficient of 1.6

L = Length of weir crest (45m)

h = head of water over the weir (variable based on lake stage)

The weir equation is applied to all stages above 112.31 mAOD. When the lake level drops below this height flow downstream is only possible through the fish pass. This can occur in two ways. Firstly, under gravity with a penstock control to regulate flows and ensure that compensation flows are met. Excessive discharges are regulated to prevent lake levels falling too rapidly. The second method is by returning abstracted lake water from the treatment works to the fish pass through a pumping system. This option is only activated when lake levels are insufficiently high to allow compensation flows through the fish pass to be met under gravity.

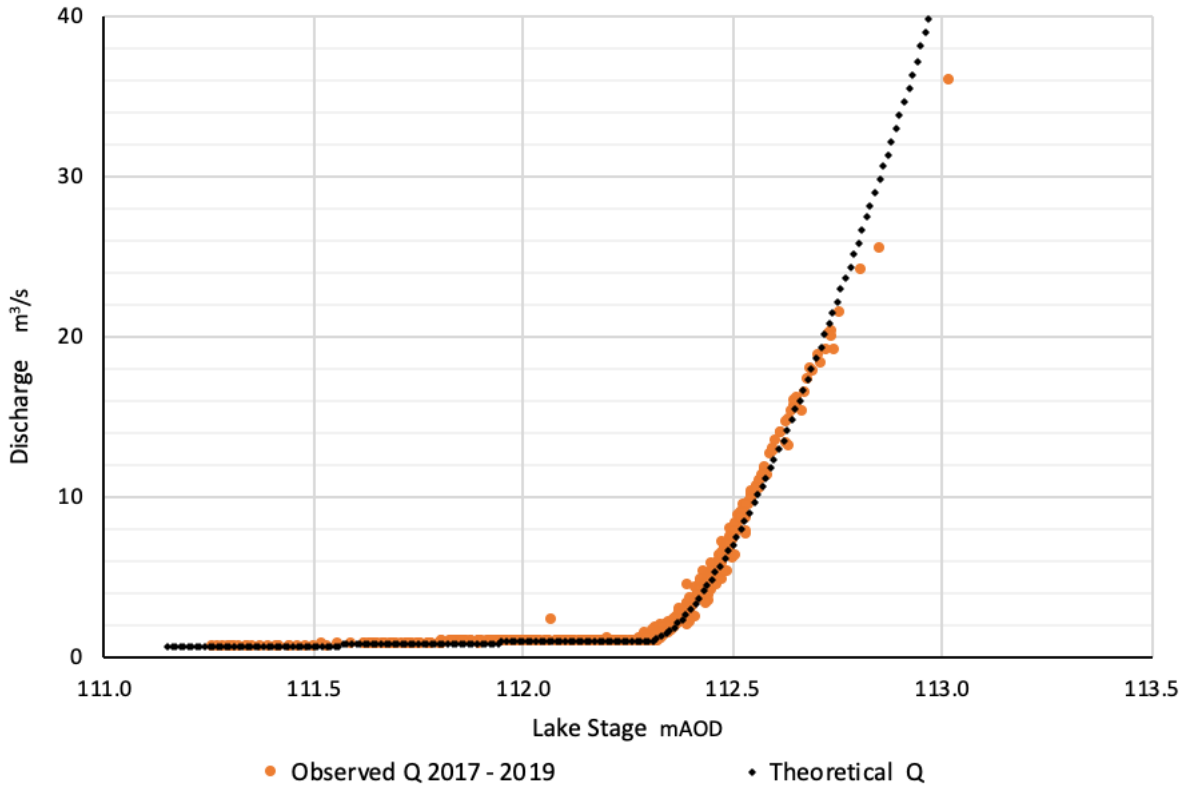


Figure 71. Observed stage and discharge relationships from 1/10/2019 – 30/9/19 compared to theoretical discharges for a given lake stage based on infrastructure dimensions and compensation flow rules.

The modelled discharge through the fish pass is therefore controlled by one hydraulic open channel flow equation based on the slope and geometry of the fish pass and lake stage until the lake reaches 111.81 mAOD (0.5m below weir crest). Below this point the compensation flows rules are implemented and fixed rules corresponding to the license agreement are applied. For lake levels between 0.5 and 0.7m below weir crest a fixed flow of 0.81 m^3/s was used. This is higher than the permitted minimum instantaneous flow but ensures that the minimum daily volume of 70 Ml/d is released downstream. At levels more than 0.75m the model applies a fixed flow of 0.69 m^3/s which ensures a minimum daily release of 60 ML.

The flow through the fish pass (Q_{fp}) for lake levels above the compensation flow rules, was calculated using Manning's open channel flow equation. This requires information on the dimensions, slope and construction materials of the channel. This information was obtained from the original engineering drawings.

Discharge was calculated using velocity area method.

$$\boldsymbol{Q = V\,A}$$

Velocity was estimated using the below manning's equation.

$$\boldsymbol{V = \frac{1}{n} \times R_h^{2/3} \times S^{1/2}}$$

Where: $\boldsymbol{n}$ = manning's roughness coefficient

$\boldsymbol{R_h}$ = Hydraulic radius ($\boldsymbol{P_w}$ x $\boldsymbol{A}$)

$\boldsymbol{S}$= Slope

Cross-sectional area was calculated by multiplying the height of water in the channel (obtained from lake level) by the channel width. The fish pass has a width of 1.2m.

Total discharge from the lake is therefore the sum of the two calculated discharges. Abstraction is subtracted from the lake volume but not included in the Q total unless it is returned to the fish pass to maintain minimum compensation flows.

$$\textbf{Total Q} = \mathbf{Q_w + Q_{fp}}$$

Lake Stage for the current timestep is obtained by subtracting the total volume leaving the lake from the total volume entering the lake over the previous timestep. The stage is then calculated from the linear stage volume relationship (Figure 74a). This was established using the lake bathymetry data and GIS software to calculate the volume of water stored in the lake for given depths.

The above approach was initially tested using estimated hourly inflows for the 2017 – 2019 observation period. Inflow data prior to this period was unavailable. These estimates were obtained for the two-year period by summing the discharges for Smithy Beck and the Lower Liza. This value was then divided by the combined area of the two catchments (29.5 km^2) and multiplied by the approximate area of the catchment up to Ennerdale Weir (43 km^2).

A later model was then run using the same governing equations but with daily inflow time series generated from the calibrated SHETRAN model.

At hourly time steps the model performed well and was able to replicate both discharge and lake level with high levels of accuracy (Section 4.8). However, at daily timesteps the model became very unstable when representing large fluctuations in flows.

For the longer model runs, input data was not available at hourly time steps. It was therefore decided that using an hourly model for the 1995-2019 period was not an option. Instead, level pool routing was applied to the lake model in an attempt to solve the instability. The steps involved in this procedure are outlined in the following section.

Level Pool Routing

Routing describes the process of determining the spatial and temporal distribution of flow rates (Chin, 2013). Reservoirs and lakes have flow attenuating properties which vary based on the physical characteristics of the water body and outlet form.

A simplified visual representation of this process is shown in Figure 72. This shows that the flood peak is reduced or attenuated by the lake and also delayed or translated resulting in a shallower downstream hydrograph. As inflow is equal to outflow, the total volume of water or area under the two hydrographs is the same over the duration of the flow event. The routing process established a transfer function based on the inflow hydrograph and known properties of the storage body to produce the downstream hydrograph.

The attenuating effect of Ennerdale Water is demonstrated in Chapter 3, Figure 48 (Section 3.6.2), however, this only shows the inflow hydrograph from the River Liza which represents around 60 - 70% of total lake inflows. To obtain the total inflow volume into Ennerdale Water, the calibrated SHETRAN model was used.

Various forms of routing models exist. For reservoir storage, hydrologic routing is normally used unless the reservoir is narrow in relation to its length, in which cases hydraulic routing methods are used (Chin, 2013).

Level pool routing can be applied when downstream discharge, lake volume and lake stage are known or if the physical dimensions of the reservoir and its outlet are known and no flow regulation or control occurs. In the case of Ennerdale Water the dimensions of the outlet structures are known but flow is regulated so normal hydraulic equations could not be easily applied. Observed stage and discharge along with known lake storage for a given stage were used for the model.

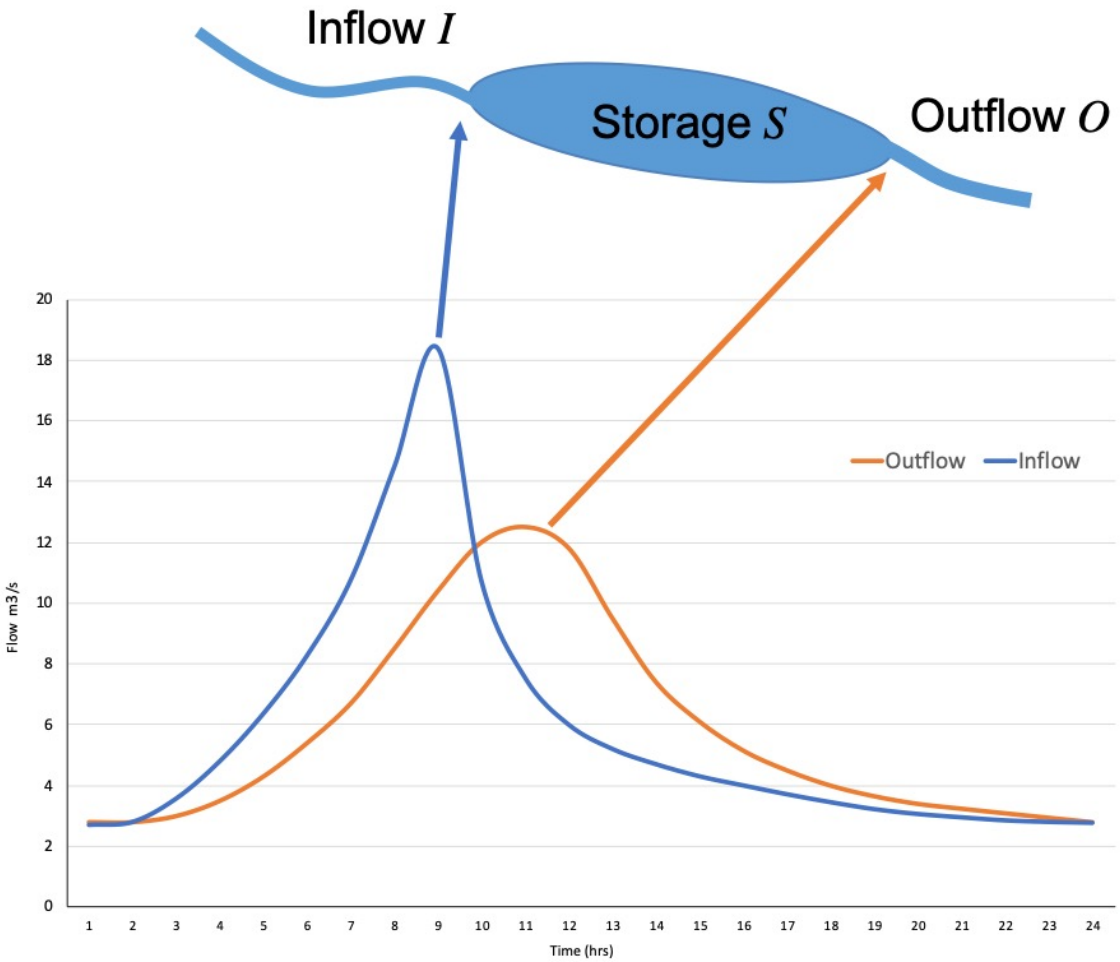


Figure 72. Simplified representation of level pool routing and the attenuation affect created by lakes and reservoirs (adapted from (Ionescu and Gogoase Nistoran, 2019)

Hydrologic routing using the Modified Puls was applied to the lake model.

This is based on the continuity equation expressed as:

$$\frac{dS}{dt} = I(t) - O(t)$$

Where: S= the storage between upstream and downstream sections

t= time

I= inflow into the reservoir (from field observations and SHETRAN model)

O= the outflow rate at the downstream section

Using the relationship between lake stage and storage and the relationship between lake stage and discharge a combined characteristic is represented visually in Figure 73:

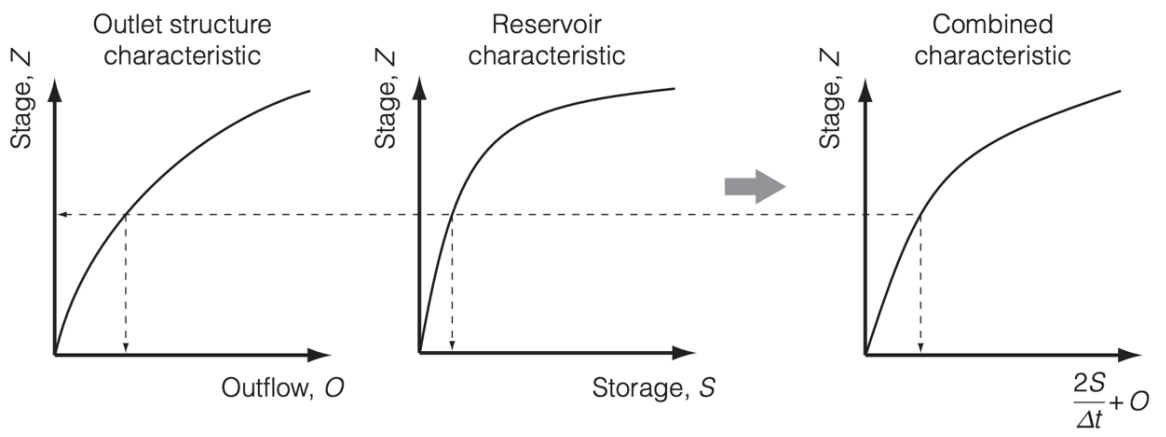


Figure 73. Characteristics of lake and lake outlet used to create a combined characteristic for the level pool routing transfer function (Chin, 2013)

The input data obtained from Ennerdale and data used for the level pool routing are shown in Figure 74. The relationship between lake stage and discharge was obtained from observed data from 2017 to 2019, however, using theoretical discharges based on the outlet dimensions produced similar results (Figure 71).

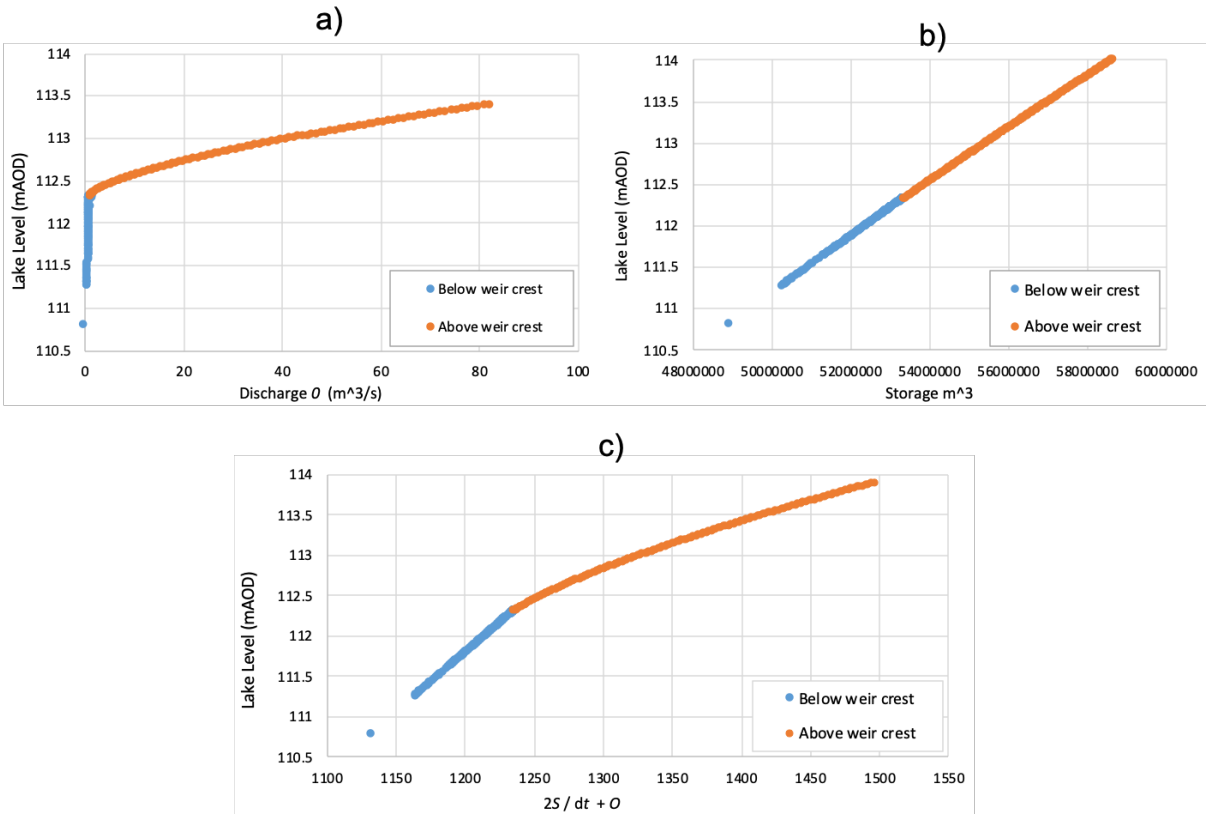


Figure 74. Relationships between lake level, storage and discharge used to obtain the transfer function for the level pool routing model.

Analysis of the observed data showed that the relationship between lake stage and discharge was not constant over time. This is due to a number of factors. Firstly, the management of discharges through the fish pass has changed over time in response to the requirements of the

abstraction and impounding license agreement. Management options include flow regulation through pen stock control and pumping water from the water treatment works. These both impacted the data. A second cause of the changes is due to physical changes to the downstream channel directly below the weir and changes to the integrity of the weir structure.

Physical changes to the channel below the weir are the result of increased sediment deposition as a consequence of reconnecting Ben Gill to the River Ehen. This has impacted the hydraulic conditions and efficient functioning of the weir. Degradation of the weir structure over time is also likely to have some, albeit less significant, impact on the stage discharge relationship. Figure 75 highlights these changes by showing selected hydrological years of data in different colours. The period prior to 2013 shows lower minimum flows which reflect the terms of the previous abstraction and impoundment license agreement. After 2013 minimum flows are increased however the stage discharge relationship becomes unstable above weir crest (112.31 mAOD) which reflects the shifting physical conditions downstream of the weir after the reconnection of Ben Gill in 2014. The data shows that by 2017 the downstream conditions have begun to stabilise, and a more regular stage discharge relationship is restored.

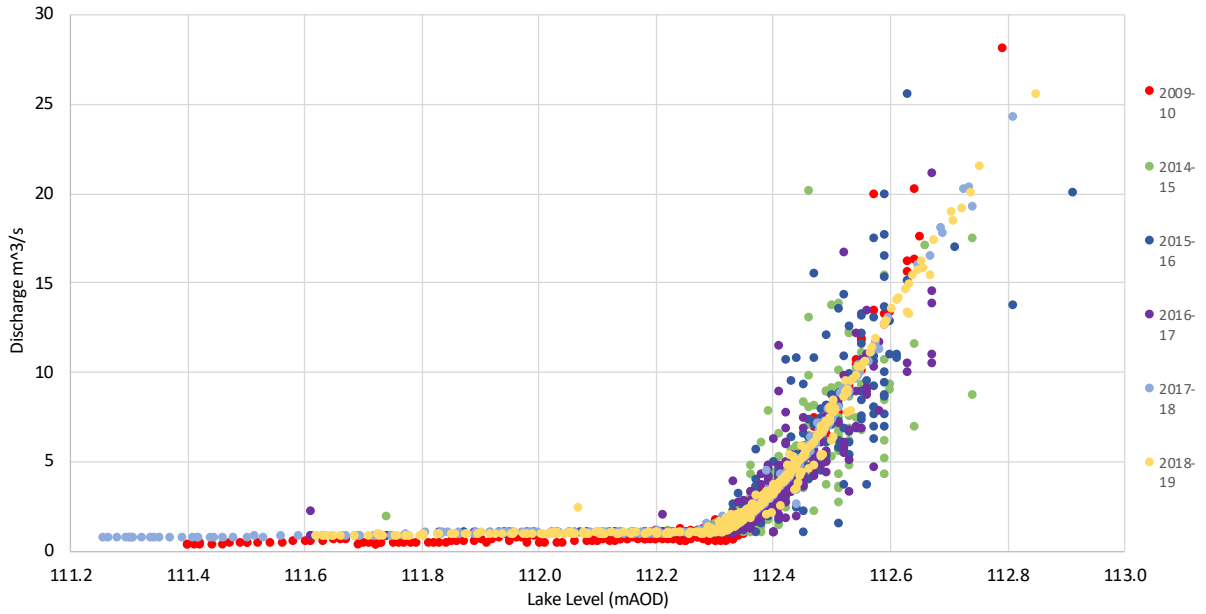


Figure 75. Observed lake stage and discharge relationships for selected hydrological years.

To overcome the impact of these fluctuations on the model performance, using a unique stage discharge relationship for each hydrological year was considered. However, this would have required the creation of multiple independent files for each year of the 24-year model run negating the time saving gains and flexibility intended of a computational modelling approach. A comparison of model performance using both observed data and theoretical data is shown in the results section of this chapter.

Natural Outflow

The level pool routing method can be applied to natural lake outlets if observed discharges are available or for theoretical discharges based on the channel dimensions and estimations of physical properties such as slope and channel roughness.

In this first phase of the study, natural outlet profiles are used from a previous consultancy report commissioned by the Environment Agency and that were subsequently used in and MSc project carried out at Newcastle University and supervised as part of this EngD. Further natural outflows are investigated in **Chapter 6** which are provided from the geomorphic modelling outputs in **Chapter 5.**

In the previous two studies, four outlet profiles were analysed. These consisted of two designs which were then each duplicated at an increased elevation of 0.2m along the width of the river channel (Figure 76).

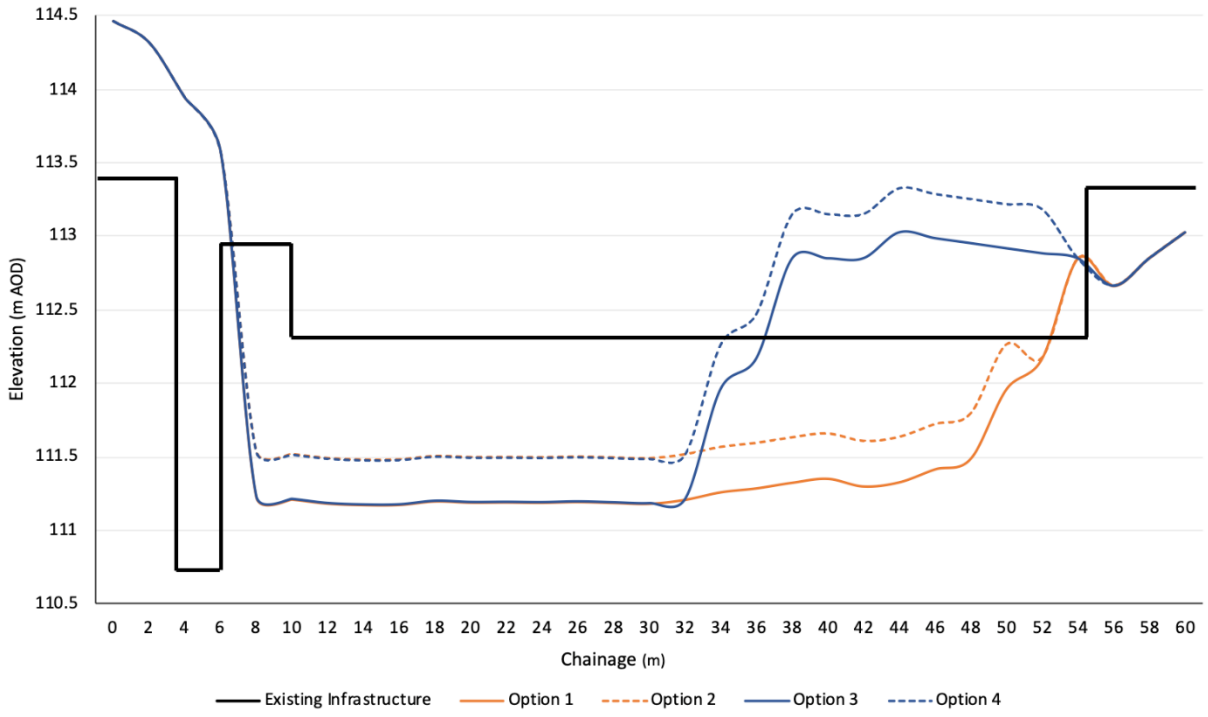


Figure 76. Previously analysed 'natural' outlet designs compared to current elevations for weir and fish pass

Rating curves were produced for the four outflow designs, using Manning's open channel flow equations. In all four cases a single Manning's roughness coefficient of 0.07 was used. A slope of 0.008 was used based on the current slope of the River Ehen channel directly downstream from the weir. This was obtained using the profile tool in QGIS and creating a profile extending 100m from the weir and using the average slope.

4.5 Performance Criteria

4.5.1 *SHETRAN*

The first criteria for the coupled catchment lake models was that SHETRAN could accurately reproduce the inflows into Ennerdale Water for the period between 1995 and 2019. This ability depends largely on the quality of the gathered field data and the effective calibration of SHETRAN. Three principal indicators were used to assess improvements to the model during the calibration period and overall performance after calibration. These were, Q_{95} for the main inflow into the lake, the Nash Sutcliffe Efficiency of the model for this part of the river network and the overall water balance of the catchment.

Low flows are of particular concern to the conservation of freshwater mussels and as such emphasis was placed on accurately modelling this part of the hydrograph. The Q_{95} shows the flow that is met or exceeded 95% of the time. Olsen *et al.*, (2013), assessed the use of low flow metrics including Q_{95} for model calibration in a study using a coupled groundwater surface water model for low flow evaluation. This highlighted the importance of using appropriate metrics for model calibration that suit the aims of the modelling study.

4.5.2 *SHETRAN + Lake Model*

The second set of performance criteria apply to the coupled SHETRAN + Lake models. The ability to accurately reproduce the discharge from Ennerdale Water and changes to lake levels in Ennerdale Water under the current managed regime was assessed using a range of metrics.

NSE efficiency was used to evaluate the overall model performance along with a number of other hydrological metrics. Percentage exceedance values for both discharge and lake level were compared at Q_{95}, Q_{70}, Q_{50}, Q_{10} and Q_5 intervals (S_{95}, S_{70}, S_{50} etc. for stage values). Minimum and maximum values were compared to ensure upper and lower limits of the model were within the range of expected values and accuracy of the overall water balance was assessed using mean flows.

In addition to NSE, model performance indicators used were Root Mean Square Error (RMSE), Percentage Bias (PBIAS), R squared (R^2) and Ratio of Standard deviation to RMSE (RSR). Model performance indicators provide a quantitative means of assessing model performance against a reference dataset. In this study, simulated and observed values for discharge at the downstream outflow of Ennerdale Water are compared using the following model performance indicators. These indicators are frequently applied in hydrological studies

(Henriksen *et al.*, 2003; Ritter and Muñoz-Carpena, 2013; Golmohammadi *et al.*, 2014) and are summarised below.

Root Mean Square Error (RMSE) aggregates the magnitude of error between simulated and observed values and serves to indicate the accuracy of a simulated output. RMSE is always non-negative and an RMSE of 0 indicates a perfect fit between simulated and observed values, therefore a small value of RMSE is preferable.

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n}(S_i - O_i)^2}{n}}$$

Nash-Sutcliffe Efficiency (NSE) quantifies how well observed vs simulated values fit a 1:1 relationship. An NSE of 1 implies a perfect fit of simulated to observed values. An NSE of 0 implies that the model performs no better than using the average of the observed values. For this study a NSE of above 0.6 was considered adequate.

$$NSE = 1 - \left[\frac{\sum_{i=1}^{n}(O_i - S_i)^2}{\sum_{i=1}^{n}(O_i - \bar{O})^2}\right]$$

Coefficient of determination (R^2) is a measure of how well changes in simulated values can be predicted by changes in observed values. R^2 is calculated in the range 0 to 1 and an R^2 closer to 1 indicate a better model fit.

$$R^2 = \left[\frac{\sum_{i=1}^{n}(O_i - \bar{O})(S_i - \bar{S})}{\sqrt{\sum_{i=1}^{n}(O_i - \bar{O})^2}\sqrt{\sum_{i=1}^{n}(S_i - \bar{S})^2}}\right]$$

The percentage bias (PBIAS) between simulated and observed results measures the average tendency for predictions to be consistently higher or consistently lower than observations. PBIAS is calculated in the range -100% < PBIAS <100%, where a PBIAS value tending to 0 indicates an unbiased model.

$$PBIAS = 100\ \frac{\sum_{i=1}^{n}(S_i - O_i)}{\sum_{i=1}^{n} O_i}$$

RSR: This the ratio of the RMSE and the standard deviation of the observed values.

$$RSR = \left[\frac{\sum_{i=1}^{n}(O_i - S_i)^2}{\sqrt{\sum_{i=1}^{n}(O_i - \bar{O})^2}}\right]$$

Where n is the total number of intervals in the period under consideration, O_i is the observed value at interval i, S_i is the simulated value at interval i , $\bar{O}$ is the mean observed value and $\bar{S}$ is the mean simulated value.

A final criterion of the model was the ability to reasonably replicate ecologically relevant flow metrics under the current regime. This was assessed by measuring the percentage change between the values obtained for the metrics using the observed data and those obtained using the timeseries from the model outputs.

4.6 Calibration Steps

Calibration was a two-stage process. SHETRAN was initially calibrated against field observations to provide inflows to the lake model. The lake model was then calibrated for the same period using observed discharge at the downstream NRFA gauging station (74003) and by adjusting the parameters controlling the discharge equations for the weir and fish pass.

4.6.1 *SHETRAN*

SHETRAN was calibrated using the flow data from the gauged sites during the monitoring period. Primary consideration was given to the Lower Liza site as this is the main inflow into Ennerdale Water and captures around 58% of the catchment surface area (Figure 77).

The model was calibrated to replicate low flows as well as possible as this was the least uncertain range of flows recorded during the observation period. It is also the component of the flow regime of most concern for the conservation of freshwater mussels and the future management of the site.

Minimum flows and Q_{95} were used as target calibration metrics. Nash-Sutcliffe Efficiency (NSE) was also considered for assessing overall model performance. As a final check the model discharge was compared to the observed discharge at the NRFA gauging station with mean flows and mass balance being the reference criteria.

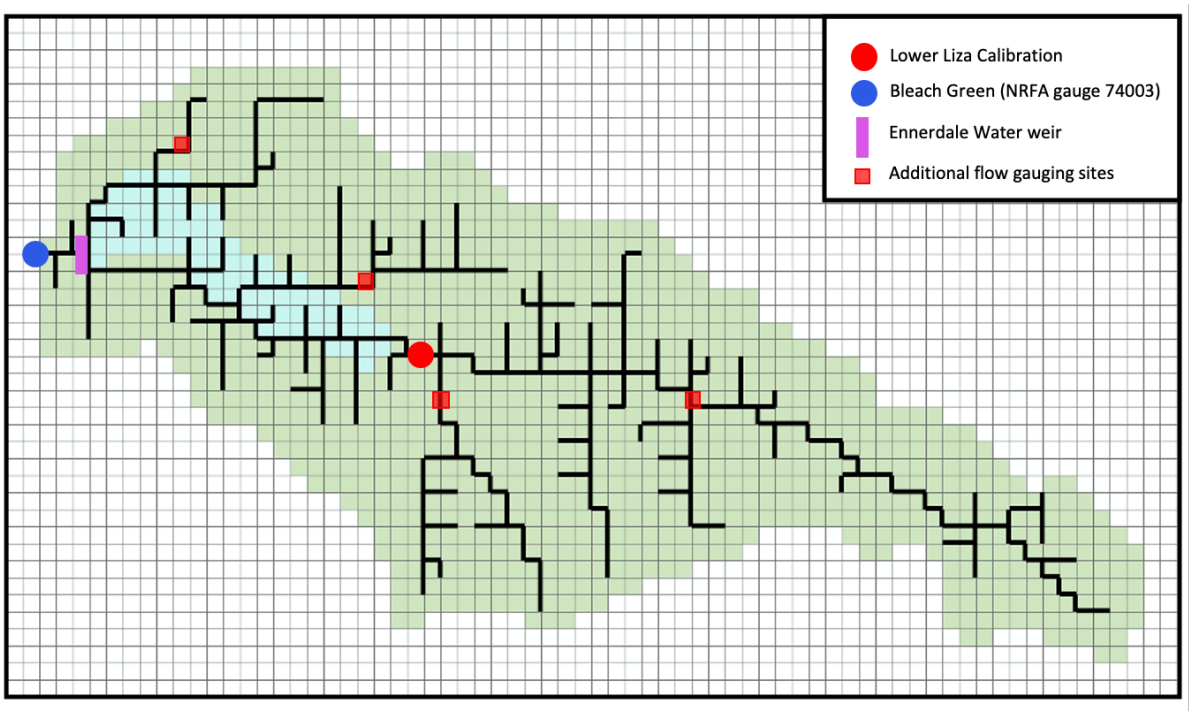


Figure 77. Map of the SHETRAN-generated river network and catchment mask showing the locations of the, NRFA gauge and calibration and reference gauging points for the SHETRAN model.

The main steps in the calibration process were:

1 Check overall mass balance for the catchment. Adjust PE/AE if required
2 Calibrate the model to observed Lower Liza sub catchment
3 Changes soil type parameters. Depth and storage.
4 Adjust hydraulic conductivity.
5 Adjust Strickler values to increase or decrease response times.

Calibrating the model to the Lower Liza site was challenging due to the physical properties of the river floodplain and the presence of small offshoot channels within the proximity of the main stem of the river. This is highlighted in Figure 78 which shows the model representation of the river network.

Due to the model resolution, the confluence of the River Liza, Woundell Beck and a smaller flood plain channel are represented as one. However, in reality, the smaller channel enters the River Liza downstream of this confluence, as shown in the map (Figure 78).

The observed field data and gauging didn't account for this additional water and modelled flows are slightly elevated as a result. This only partially accounts for the difference between observed low flows and modelled low flows for this stretch of the river. However, it is an important consideration for assessing the model performance and understanding the physical behaviours of the site.

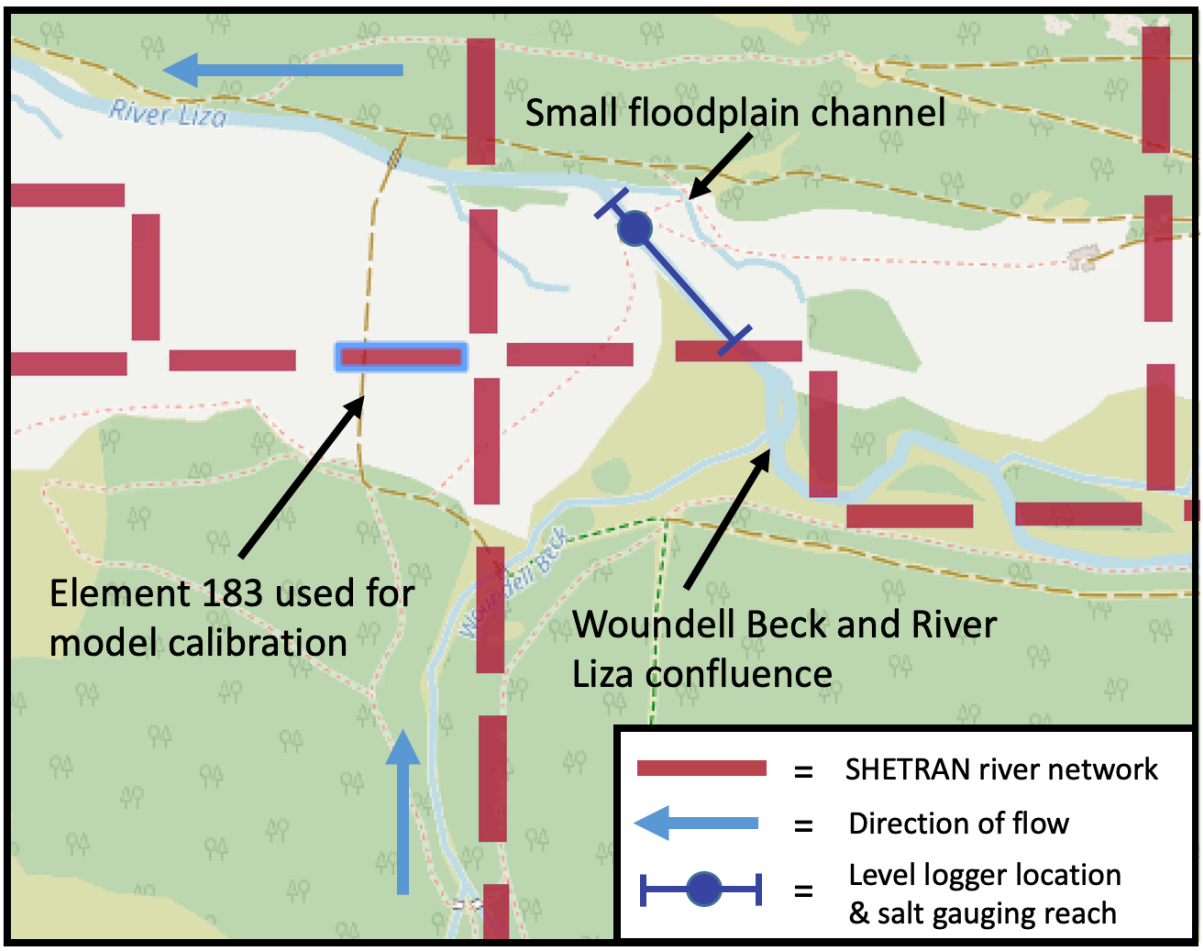


Figure 78. Magnified section of the River Liza and tributary river network as represented by SHETRAN overlayed on a map showing actual river locations.

Additional discrepancies between the overall water balance in this part of the catchment may also be due to the subterranean movement of water in the alluvial deposits of the flood plain. The floodplain has many active channel offshoots which are clearly visible and transporting water. However, there are also several relic channels across the floodplain which, although not always visible, have the potential to transport substantial volumes of water and play an important role in the hydrological connectivity between the river, lake and floodplain as studies from other sites have shown (Amoros and Bornette, 2002; Owen and Dahlin, 2010; Heeren *et al.*, 2014). The mouths of these channels are visible closer to the lower end of the River Liza and the shores of Ennerdale Water. Even at extremely low flows these can be seen to transport water (Figure 79). It is therefore possible that there is a discrepancy between the conveyance of this water in the field and how SHETRAN tries to represent this physically complex hydrological system.

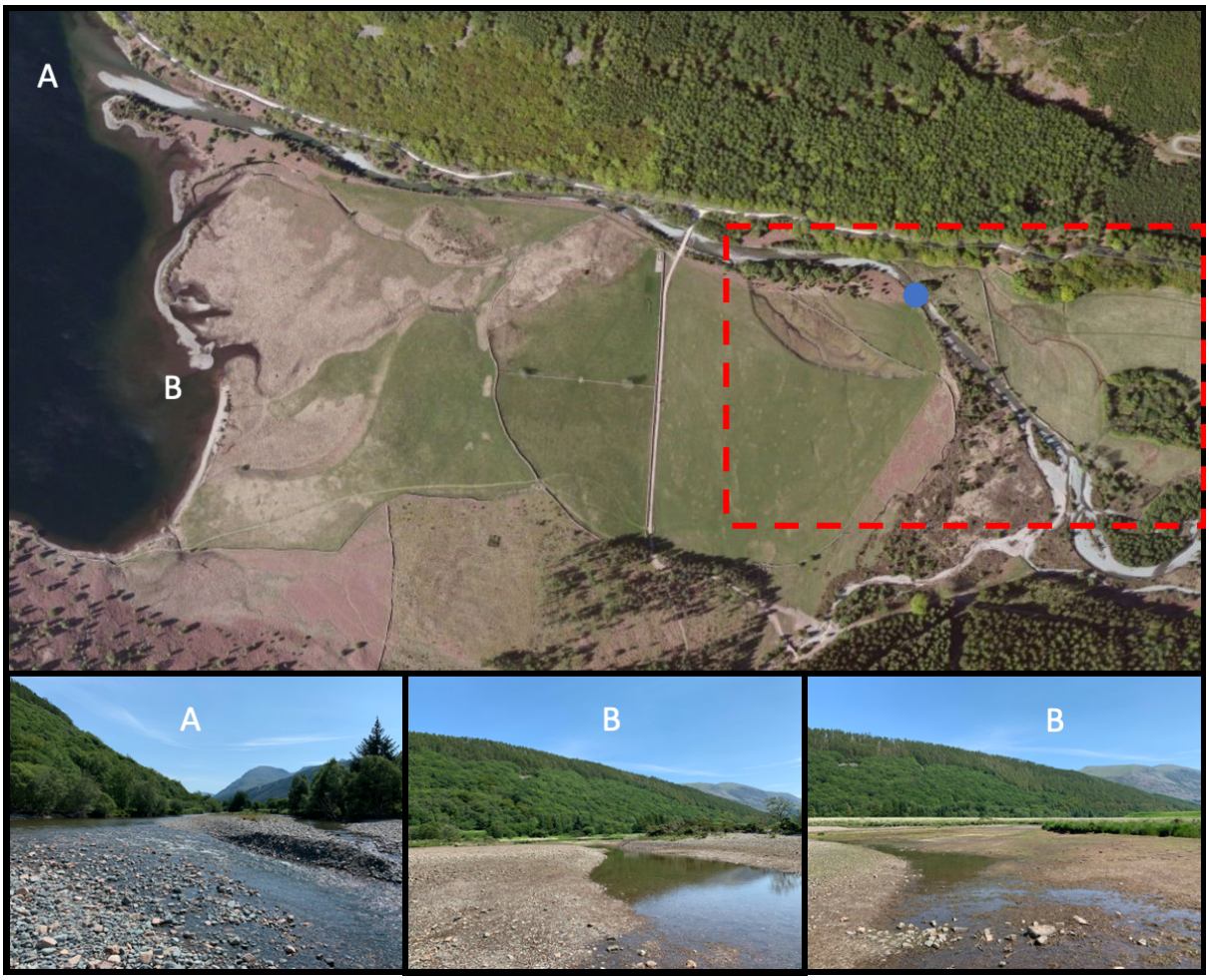


Figure 79. Evidence of relic channels conveying water during extreme low flows (25/6/2020). Red box shows evidence of active and paleo channels on the floodplain proximal to the Lower Liza gauging location.

The overall model performance was improved for the second hydrological year of observations achieving an NSE of 0.82. Calibrating the model to the observed Q_{95} was challenging however for the two-year period a modelled Q_{95} of 0.19 m^3/s was achieved, matching the estimations obtained from the field data.

Matching the FDC to the observed values was not possible, in part, due to the reasons previously outlined. However, the FDC matches reasonably well at the upper and lower limits for the period of observation. The model performance is presented in more detail in the results section of this chapter.

4.6.2 *Lake Model*

The lake model was calibrated against observed stage discharge relationships for above and below weir crest level.

Two stage discharge relationships were initially established however this failed to capture the true behaviour of the discharge at the site. In reality, the cut-off point between water spilling

over the weir or not is not a simple binary relationship. In some cases, lake levels, may be at or around weir crest but wave action allows water to spill over the weir. Furthermore, the structure allows some water to spill even when the water level is slightly below weir crest. Accuracy of lake level recordings may also be another area which creates discrepancy between the stage discharge relationship. To account for this, a third stage discharge relationship was established to soften the transition between the two clearly defined stage discharge relationships. By slightly adjusting the extent of this transition it was possible to fine tune the model to more accurately represent the observed discharges downstream.

4.7 Validation and Preliminary Simulations

The models were validated against available observed data. A further set of model runs were performed using natural outlets to provide evidence of the applicability of the modelling approach (Table 14). Further model run scenarios are analysed in Chapter 6 to show the evolution of the hydrological regime in response to a changing lake outlets.

Table 14. Model runs conducted with the final daily timestep level pool routing model.

Model Runs	Timestep	Start Date	End Date	Duration (years)	Rationale
Calibration Period	Hourly	01/10/2019	30/09/2019	2	Compares modelled discharge to observed Q at Bleach Green Weir for calibration period
Calibration Period	Daily	01/10/2019	30/09/2019	2	Compares modelled discharge to observed Q at Bleach Green Weir for calibration period
Validation	Hourly	01/10/2003	30/09/2019	16	Comapres available observed hourly data to SHETRAN + Hydraulic model
Validation	Daily	01/10/1995	30/09/2019	24	Compares modelled discharge to observed Q at Bleach Green Weir
Baseline + natural outlets	Daily	01/10/1995	30/09/2019	24	Analyses the impact of no flow regime management and the proposed natural outlet designs as analysed by MWH.

4.8 Results

4.8.1 *SHETRAN Model Calibration*

SHETRAN was calibrated against the estimated discharges at the Lower Liza site from 1/10/17 to 30/9/19. Model performance was slightly better for the second year of observation data. An overall NSE of 0.77 was achieved for the two-year period with scores of 0.73 and 0.83 for the first and second hydrological years respectively (Table 15).

The model was able to replicate the upper and lower ends of the FDC. A modelled Q_{95} of 0.19 m^3/s was obtained for the two-year period, matching the estimated observed Q_{95} for the field data for the Lower Liza Site.

Table 15. Model performance indicators for Lower Liza internal catchment calibration and overall catchment water balance for the 2017-2019 observation period.

	Two-year Period		2017-2018		2018-2019	
	Observed	Modelled	Observed	Modelled	Observed	Modelled
Mean Q	2.26	2.68	2.20	2.55	2.33	2.81
Max	40.11	34.83	40.11	34.83	27.97	33.40
Min	0.15	0.10	0.15	0.10	0.17	0.16
Q95	0.19	**0.19**	0.18	0.17	0.21	0.23
Q70	0.55	0.92	0.55	0.89	0.55	0.96
Q50	0.92	1.45	0.98	1.43	0.90	1.50
Q10	5.26	5.86	4.67	5.34	5.68	6.42
Q5	9.07	9.16	7.96	8.92	10.40	10.14
WB		1.19		1.16		1.21
NSE		**0.77**		0.73		0.82

4.8.2 *SHETRAN and Lake Model Combined Performance*

The development of the lake model was an iterative process that allowed two different methods to be tested for their suitability in accurately modelling the level of Ennerdale Water and discharge into the River Ehen.

SHETRAN + Hydraulic Model

The initial hydraulic model used a weir equation and open channel flow principles with compensation flow conditional statements. This performed very well at hourly timesteps (Figure 80). The inflow data was estimated by scaling up the field discharge observations to the area of the catchment so there is some uncertainty around the inflow timeseries. This is highlighted by the maximum discharge values generated by the model (Table 16).

The second model used inflows generated by the SHETRAN catchment model which incorporates all river links up to the weir. This produced a better overall hydrological performance for the downstream discharge however the lake levels were less accurately replicated. This is shown most clearly when the lake level dops below weir crest suggesting that the model inflows may slightly underestimate the actual inflows at times of lower-than-average flow (Figure 81). However, despite this, the average flow over the two-year period was 3.49 m^3/s when using inflows generated by SHETRAN compared to 3.36 m^3/s with estimated inflows. Both models slightly underestimated the observed mean flow of 3.58 $m^{3/}s$. This may in part be due to the fact that compensation flow releases were frequently slightly higher than the terms of the impoundment license. This margin for error in the management of the system ensures that sufficient water is always flowing downstream, and the terms of the license agreement are not breached. Furthermore, pumped returns to the fish pass help maintain flows above levels that would not be achieved under gravity thus making it more difficult for the model to replicate the observed behaviour at extremely low lake levels.

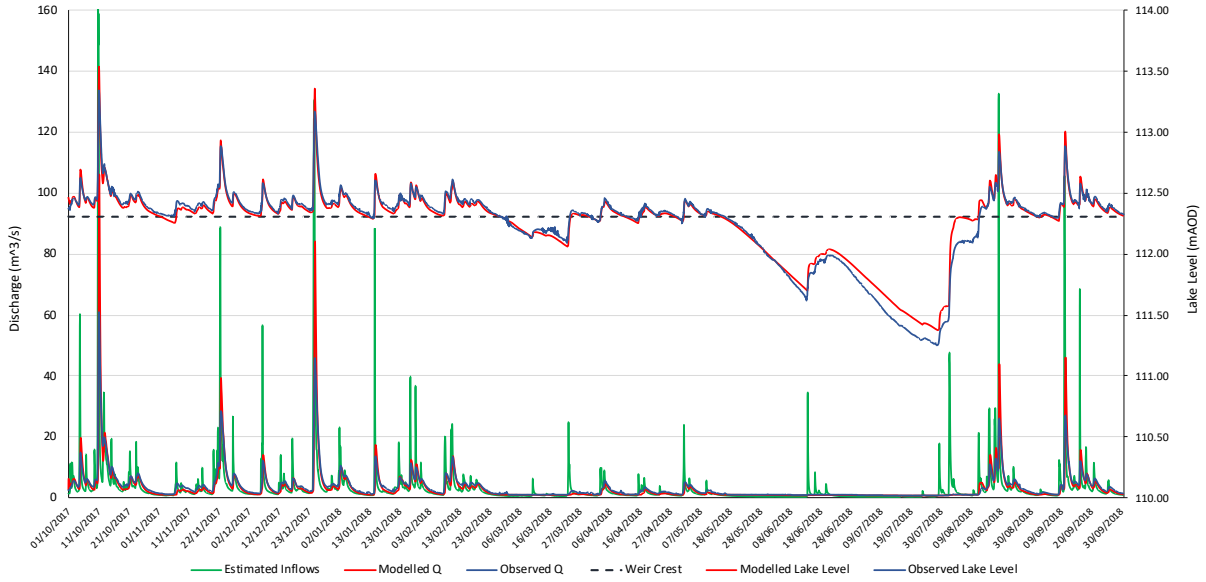


Figure 80. Results for the hydrological year 2017-18 lake model using weir, open channel flow equations and compensation flow rules at hourly timesteps with estimated inflows.

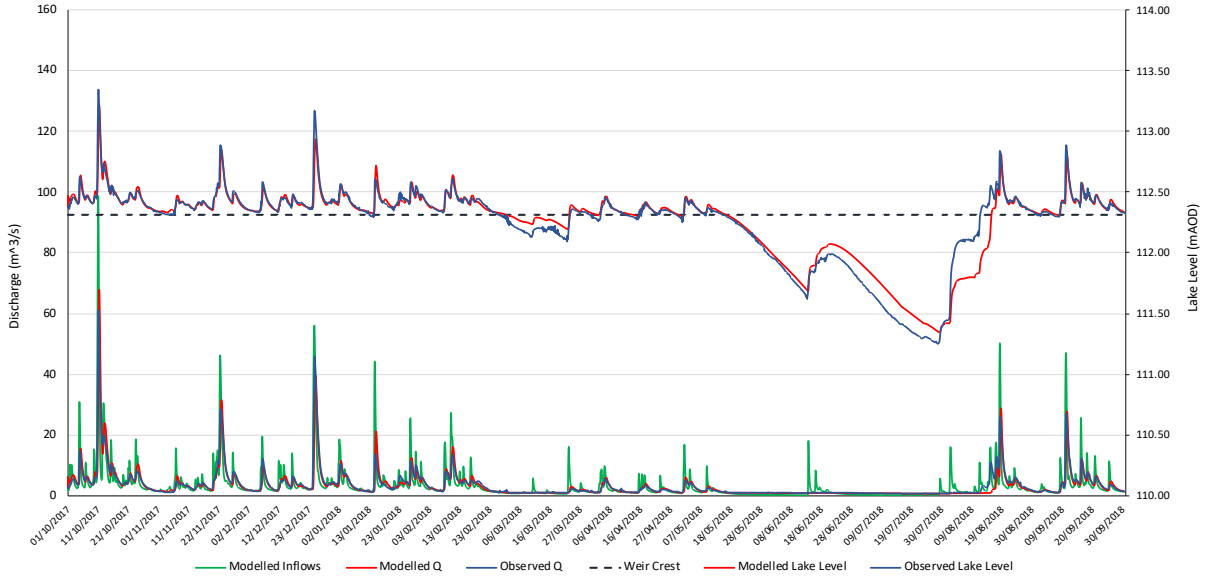


Figure 81. Results for the hydrological year 2017-18 lake model using weir, open channel flow equations and compensation flow rules at hourly timesteps with modelled inflows.

When using inflows generated by the calibrated hourly SHETRAN model, the lake model achieved a NSE of 0.92 for discharge and 0.94 for lake levels for the calibration period. A comparison of model performance when using estimated inflows based on field observations and modelled SHETRAN inflows for the 2017 – 2019 is shown in Table 16. Modelled inflows were used in the final model set up.

Table 16. Hydrological metrics and model performance indicators for discharge and lake levels.

Flow Metric		Observed Values	Estimated inflows	Modelled Inflows
Mean	m^3/s	3.58	3.36	3.49
Max	m^3/s	60.92	103.16	66.56
Min	m^3/s	0.66	0.69	0.69
Q95	m^3/s	0.85	0.81	0.81
Q70	m^3/s	1.10	1.13	1.16
Q50	m^3/s	1.97	1.33	1.88
Q10	m^3/s	7.73	7.20	7.48
Q5	m^3/s	11.75	12.26	10.90
Model Performance				
		NSE	**0.85**	**0.92**
		RSR	**0.39**	**0.29**
		R2	**0.94**	**0.92**
		RMSE	**1.73**	**1.27**

Stage Metric		Observed	Estimated inflows	Modelled Inflows
Mean	mAOD	112.28	112.25	112.28
Max	mAOD	113.34	113.56	113.24
Min	mAOD	111.25	111.28	111.27
S95	mAOD	111.67	111.64	111.72
S70	mAOD	112.27	112.25	112.29
S50	mAOD	112.34	112.34	112.35
S10	mAOD	112.51	112.50	112.50
S5	mAOD	112.58	112.59	112.57
Model Performance				
		NSE	**0.96**	**0.94**
		RSR	**0.19**	**0.24**
		R2	**0.98**	**0.94**
		RMSE	**0.05**	**0.07**

At daily timesteps the hydraulic model performed poorly. High flows over 24 hours caused excessive volumes of water to leave the lake, creating unrealistic discharge values. This resulted in extreme fluctuations between high and low flows and an unstable model (Figure 82). This option was not developed any further as it was unable to meet the performance criteria set out in Section 4.5.

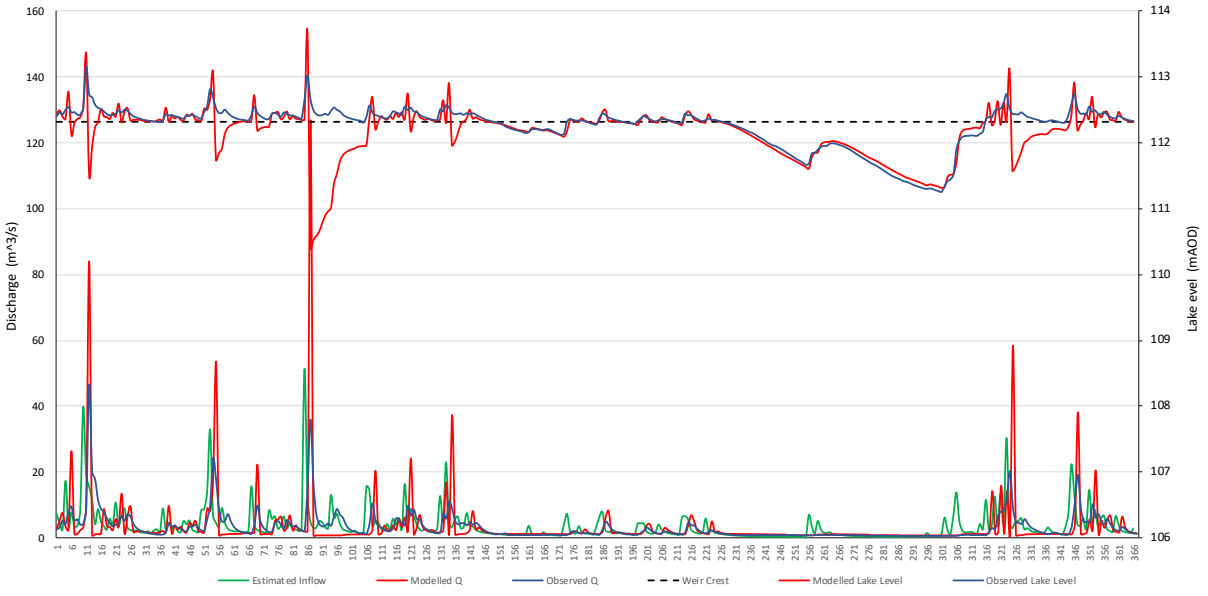


Figure 82. Results from the lake model using a weir equation and open channel flow equation with compensation flow rules at a daily time step.

4.8.3 *SHETRAN + Level Pool Routing Model*

At daily times steps the SHETRAN + level pool routing model performed reasonably well. As expected, performance indicators were consistently higher in all cases for the calibration runs.

For the calibration period 2017 -2019 an NSE of 0.88 was obtained for discharge and 0.77 for lake levels. This fell to 0.8 for the simulated discharge over the full model run but remained at 0.77 for lake levels over the same period. The modelled discharges and lake levels for the calibration period using both observed downstream conditions and theoretical conditions are plotted against the observed values in Figure 83.

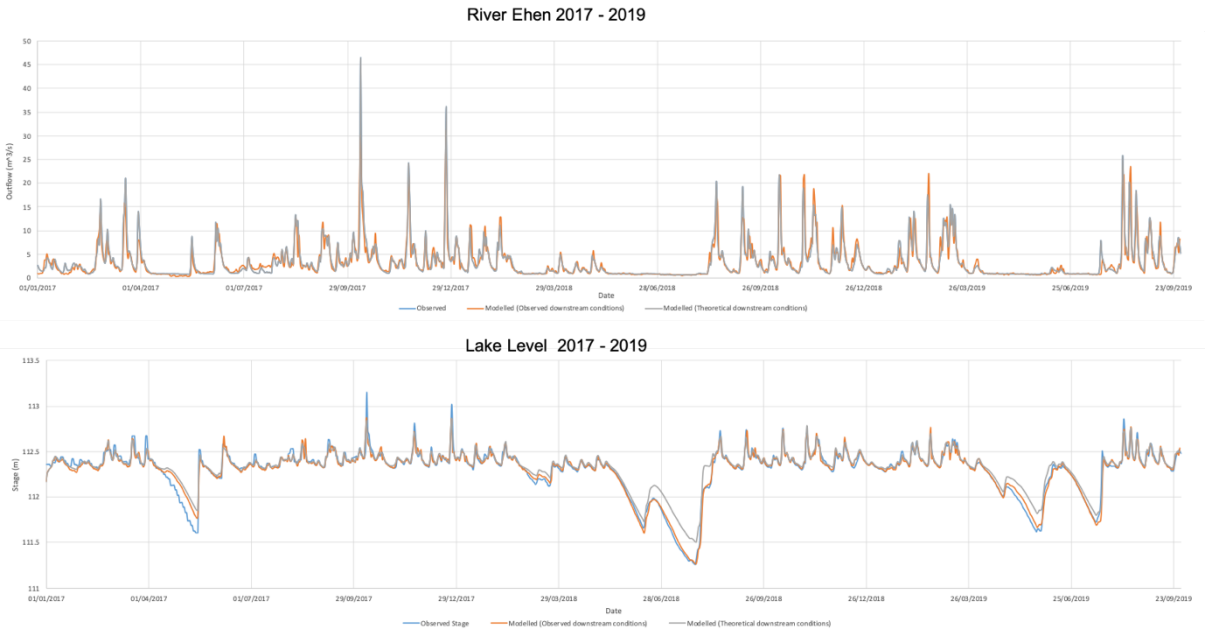


Figure 83. Level pool routing model outputs comparing the use of theoretical and observed stage discharge relationships. Modelled inflows are shown in blue.

During the calibration period the model showed a slight tendency to underestimate discharge and overestimate lake levels. However, for longer model runs discharge was slightly overestimated by between 1.63 – 1.73 % (Table 17) compared to the long-term mean of 3.08 m^3/s. Lake levels on the longer runs showed almost no bias to under or overestimation.

A RMSE of 1.63 was achieved for the simulated discharge for the full modelling period compared to 1.22 for lake level.

Table 17. Model performance metrics for discharge and lake levels at daily timesteps comparing calibration, validation and overall model performance.

	Discharge		
Performance Indicator	***Calibration 20017-2019***	***Validation 1995-2017***	***Combined 1995-2019***
RSR	0.35	0.43	0.44
PBIAS	-0.17	1.63	1.76
R2	0.86	0.82	0.81
RMSE	1.50	1.56	1.63
NSE	0.88	0.81	0.80

	Lake Level		
Performance Indicator	***Calibration 20017-2019***	***Validation 1995-2017***	***Combined 1995-2019***
RSR	0.48	0.49	0.48
PBIAS	0.04	-0.01	0.00
R2	0.98	0.78	0.78
RMSE	0.13	1.22	1.22
NSE	0.77	0.76	0.77

Hydrological Metrics

The ability for the model to replicate the hydrological metrics identified in Chapter 2 (Table 18) was an important criterion for the SHETRAN + Level Pool Routing Model. The overall performance was assessed by evaluating the percentage change of metrics.

The largest percentage change was shown in the timing metric ta1 however this metric has values ranging between 0 and 1. The value change was only 0.03. All other metrics were replicated to within 22%.

Table 18. Selected hydrological metrics from chapters 2 and 3 comparing those generated from observed discharge data with modelled discharge data.

Indicie	Flow Regime Component	Flow Condition	Observed	Modelled	% Change
ma11	Magnitude	Average	2.6	2.5	-5%
ma36	Magnitude	Average	4.2	3.9	-8%
ma42	Magnitude	Average	0.9	0.8	-16%
ml13	Magnitude	Low	55.6	54.5	-2%
ml22	Magnitude	Low	0.01	0.01	12%
mh16	Magnitude	High	4.1	4.1	0%
mh22	Magnitude	High	8.2	7.5	-10%
fh4	Frequency	High	9.7	11.7	17%
dl9	Duration	Low	26.3	23.3	-13%
dh13	Duration	High	4.5	4.5	0%
dh19	Duration	High	1.5	1.6	8%
ta1	Timing	Average	0.10	0.13	22%
ra5	Rate	Average	0.40	0.46	13%

Natural Outflows

Four natural outflows were used to demonstrate the application of the SHETRAN + Level Pool Routing model (Figure 76, page 129) and validate the methodology. These outflow channel profiles were used in a previous consultancy study assessing the sensitivity of lake levels to the height of the outlet channel and how discharge into the River Ehen would be affected by the width of the lake outlet.

Using the SHETRAN inflows from 1995-2019 discharge time series were generated for all four outlets. Options 1 and 3 both have minimum elevations of 111.176 mAOD, and widths of approximately 40 m and 26 m respectively.

Outlet elevation had minimal impact on the discharge time series, with width being the main factor. The flow duration curves (Figure 84) for the natural channel outlets show how the attenuating effect of the lake continues to dampen the peak flows however the lack of flow regulation causes minimum flow values to fall below the current observed flows under the

managed regime. The below example provides evidence of the model's ability to substitute different lake outlet profiles and an initial point of comparison however a more detailed analysis is presented in Chapter 6. This includes an assessment of the impact of the width and elevation of different outlet profiles on the downstream discharge. Where required, statistical tests are used to describe the significance of any differences.

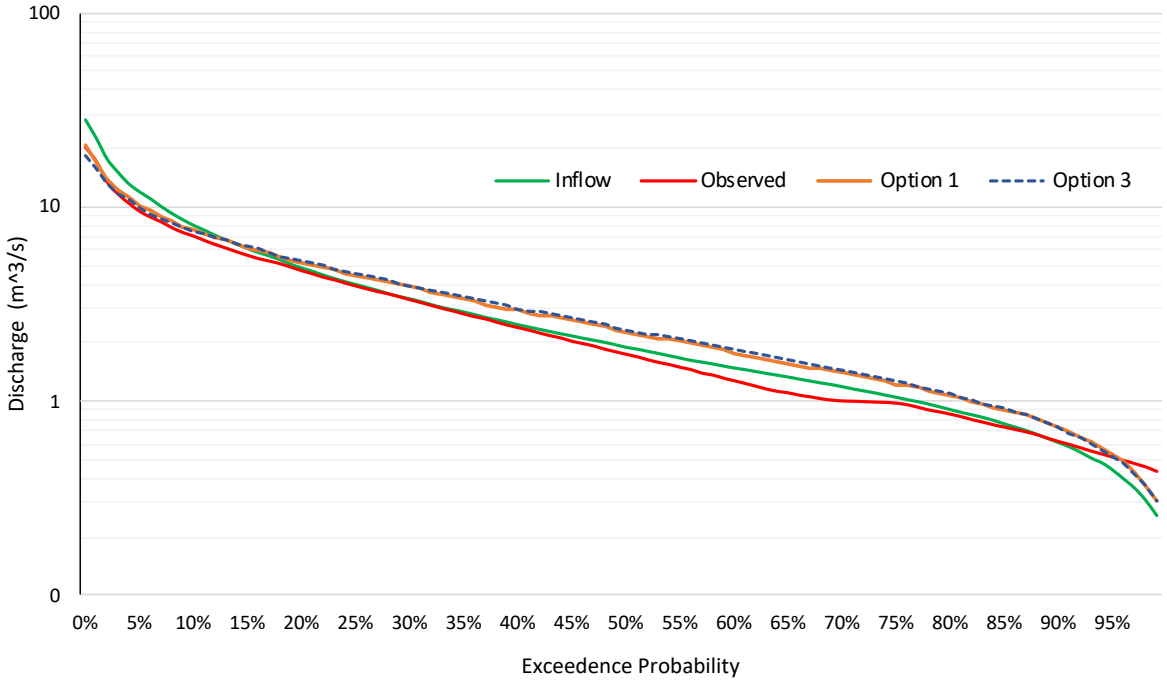


Figure 84. Upper Ehen flow duration curves for natural channel outlets 1 and 3 with no flow regulation compared to observed discharges and modelled inflows.

The example hydrograph for 2017-2018 (Figure 85) shows how minimum flows might be altered under an unmanaged regime. Minimum flow values are predicted to be lower than those presently observed however, low flow periods are interrupted with small flood pulses which would otherwise be absent due to the impact of abstraction and flow regulation.

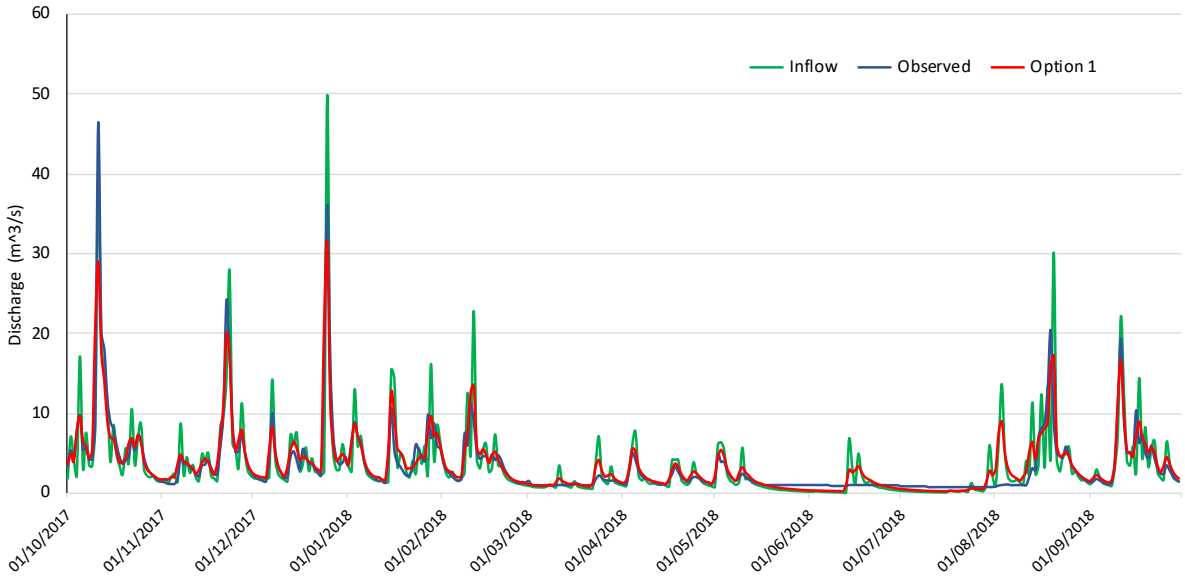


Figure 85. Example of a modelled natural hydrograph for the Upper Ehen using an outlet profile with no regulating structures or pumping returns compared to observed discharges for 2017 – 2018 and lake inflows.

Further analysis of natural flow regimes is carried out in Chapter 6 using the lake outlet profiles generated in the CAESAR -Lisflood model described in the following chapter.

An assessment of the model outputs against previous modelling studies and existing water resources models used by United Utilities was not made at this stage. These are presented in Chapter 6.

4.9 Chapter Summary

This chapter has shown that two different lake models used in combination with a calibrated PBSD hydrological model (SHETRAN) can accurately reproduce observed downstream flow conditions and lake levels in a heavily regulated site.

At hourly time steps, the hydraulic model using modelled SHETRAN values for the calibration period performed extremely well. An NSE of 0.94 was achieved for both modelled discharges and lake level. A lack of available data at hourly times steps both for model inputs and observed values to validate the model meant that it was not possible to use the model for replicating the observed flows between 1995 and 2019. However, the evidence from the hourly model suggests that with sufficient data it performs well with modelled inflows generated by SHETRAN.

Simulating a heavily managed system that has undergone multiple physical and regulatory changes is a challenging task, however the SHETRAN + Level Pool Routing (LPR) model was able to replicate observed flows reasonably well.

The SHETRAN + LPR model was also able to replicate a selection of hydrological metrics with reasonable accuracy. This is an important performance criterion as the hydrological metrics generated in the final modelling chapter will form the basis for assessing the levels of hydrological alteration and describing possible future flow regimes.

The ability to closely model the current managed flow regime was an important assessment criterion. The hourly model using the weir and hydraulic model outperformed the LPR model in this area however, a lack of data meant that it was not possible to fully assess the performance of this model and test its application for modelling discharge and lake stage with a natural outlet profile. Discharge data at daily timesteps is required to generate the hydrological metrics for flow regime assessment and is generally more readily available. It therefore makes the LPR modelling approach more suitable for this study and for application at other sites.

Chapter 5 - Ennerdale Water Sediment Analysis and Geomorphic Modelling

Chapter 5 presents a detailed modelling study of the geomorphic evolution of the lake outlet after weir removal. This is supported by the collection of additional field data through the novel application of Ground Penetrating Radar (GPR) and complimentary sediment coring techniques. A literature review of modelling approaches for predicting response to weir removal is presented along with a review of different applications of GPR for sedimentary analysis. The results of the modelling study quantify sediment transport and channel evolution, including the rate of upstream migration and changes to the and outflow form (cross-section and gradient). Additionally, the model-predicted channel outlet profiles, presented in this chapter, provide critical input data for the final hydrological modelling assessment presented in Chapter 6.

5.1 Aims and Objectives

Chapter 5 aims to predict the geomorphic responses of the Upper Ehen and Ennerdale water to weir removal. In order to achieve this aim, the following objectives are addressed in the forthcoming chapters:

- Review techniques for analysing lakebed sediments and evidence of paleo outlet channels.
- Review approaches to predicting geomorphic responses to weir removal
- Map and characterise the physical and chemical properties of the lakebed sediment upstream of the weir.
- Quantify the temporal and spatial geomorphic evolution of the lake outlet in response to weir removal.
- Generate a series of predicted natural lake outlet profiles, which will later be used in to analyse the impact of geomorphic changes on Ennerdale Water and downstream hydrology (Chapter 6).

5.2 Literature Review

5.2.1 *Predicting geomorphic responses to barrier removal*

In the following section, the current understanding of the complex geomorphic processes and responses to weir and small dam removal are outlined. Conceptual and numerical models are examined along with examples of their application in the literature and the associated limitations of these tools.

For the reasons outlined in section 2.1.2 of this thesis, weir removal for ecological restoration has become increasingly popular. However, the removal of instream barriers represents a large-scale disturbance event to riverine ecosystems. Dam owners, environmental regulatory agencies and engineers therefore require detailed predictions of the physical changes to the river channel and its connectivity, induced by the removal of these structures (Gartner *et al.*, 2015)

The scale of disturbance caused by removal may depend on a number of factors such as the size and age of the infrastructure, along with the local natural and manmade boundary conditions (Poeppl *et al.*, 2019). It is therefore also necessary to understand how the presence of these structures has influenced the movement of sediment and river morphology whilst in situ (Csiki and Rhoads, 2010).

Whilst there is abundant evidence within in the literature highlighting the geomorphic benefits of dam removal (Bednarek, 2001; Magilligan *et al.*, 2016), other studies highlight the unintended negative ecological consequences of this restoration approach (Ashley *et al.*, 2006; Chang, 2008; Orr *et al.*, 2008). Furthermore, there are several examples of negative impacts on freshwater mussel species, often caused as a result of sediment releases from poorly managed dam removal projects (Sethi *et al.*, 2004; Doyle *et al.*, 2005). Other studies have cited the loss of freshwater mussel species upstream of dams as a result of lost lentic habitat (John E. Cooper, 2011; Hogya *et al.*, 2017). However, lentic species of freshwater mussels are not present in Ennerdale Water and the focus of restoration efforts are *Margaritifera margaritifera L.* a lotic species of mussel found downstream of the lake.

Predictions for geomorphic responses to dam removal are generally based on four main approaches: conceptual models, physical models, numerical models and observations from previous removal projects. Conceptual models and numerical models are the most commonly used approaches and thus the focus of the following sections. However, the utility of observations from dam removals and physical modelling techniques is also reviewed.

5.2.2 *Conceptual Models*

Conceptual models are useful tools that provide qualitative descriptions and predictions of geomorphic evolution (Van de Wiel *et al.*, 2016). Their simplicity is both their main strength and weakness, meaning that they can provide informative insights into geomorphic processes but cannot provided quantitative predictions (Van de Wiel *et al.*, 2016). However, they are often an important first step in the modelling process and afford greater understanding of the system to be developed quickly prior to the application of more complex tools (Van de Wiel *et al.*, 2016).

Simon and Hupp (1986) developed a conceptual model describing and illustrating channel response to dam removal. This has since been used as a basis for numerous dam removal studies and further developed or adapted in many others (Doyle *et al.*, 2002; Pizzuto, 2002; M. and Crowe, 2012) including this present thesis (Figure 86). The basis for this conceptual model stems from the upstream migration of a knickpoint created by a drop in river base level (Womack and Schumm, 1977; Gardner, 1983), in this case brought about by dam or weir removal. Although the processes outlined may vary from site to site depending on sediment grain size or depth (Pizzuto, 2002), this conceptualisation provides an important starting point for further, more detailed, quantitative predictions.

The first stage in the conceptual model describes the channel with the weir or impoundment structure in place, prior to any disturbance. The second stage shows the water level response to the removal of the structure but still prior to any geomorphic adjustments. Stages c to f show how the channel might be expected to adjust over time before finally reaching a more stable state in which vegetation has had time to fully establish.

An important consideration or limitation of this conceptual model is the one-dimensional nature of its description. Whilst longitudinal profile disturbances may be the primary driver of geomorphic responses to dam removal, lateral movement of the channel is also likely to occur (Poeppl *et al.*, 2019). This lateral movement is not just in the narrowing and widening of the channel as shown in Figure 86, but also the lateral movement of the channel itself in relation to the surrounding land as observed in meandering or wandering rivers.

A further limitation of existing conceptual models is that they tend to show weir or dam removals where the lake or reservoirs is completely lost once the impounding structure has been removed. In the case of Ennerdale Water, the River Ehen will likely migrate upstream into the lake and both the lake bathymetry and lake water level will impact the rates and form of channel evolution. These examples are not well-documented within the literature, however

increasing the storage of otherwise natural lakes is relatively commonplace in the UK so the findings form this chapter will provide useful insights for other sites where barrier removal is being considered.

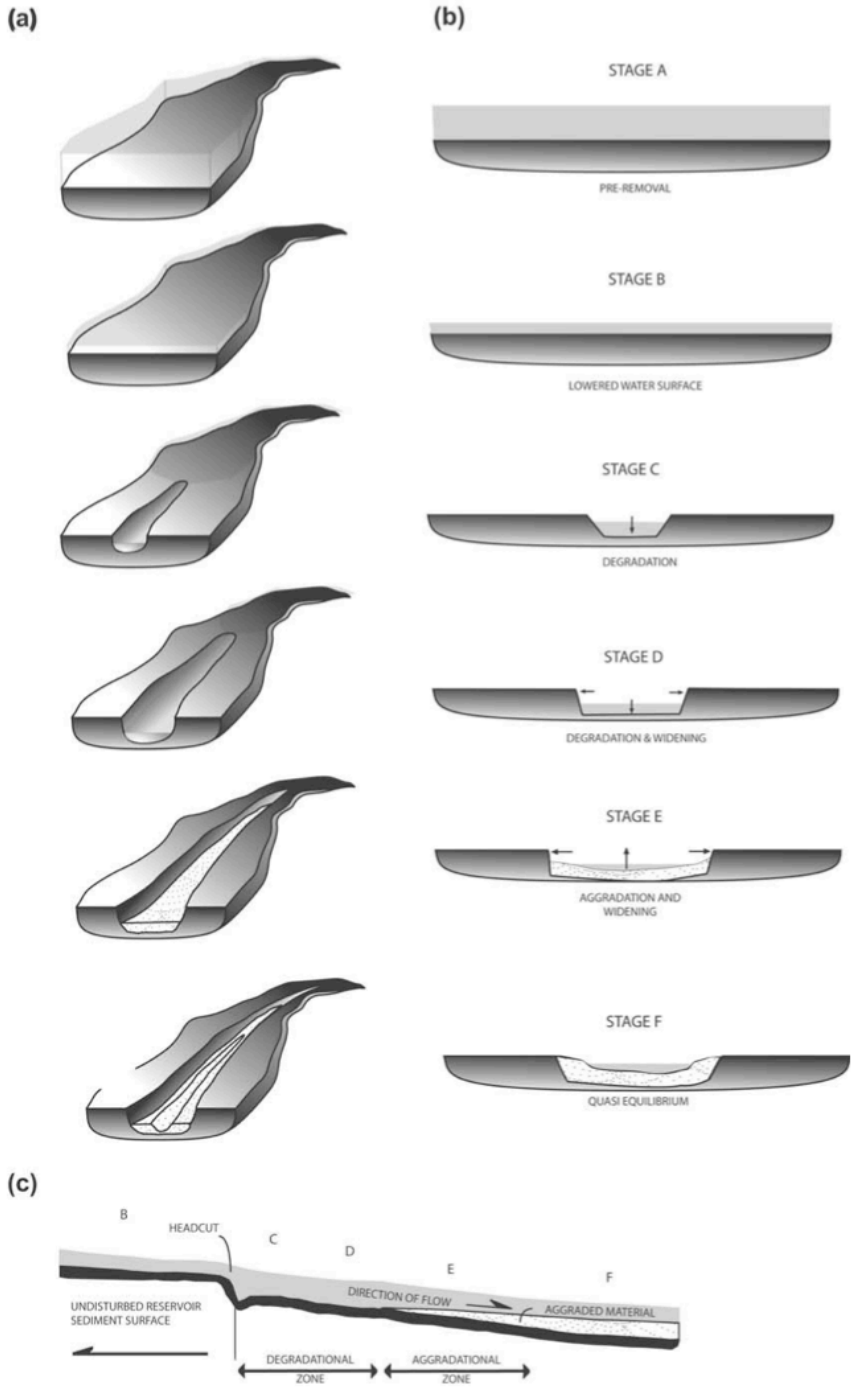


Figure 86. Conceptual model showing channel cross sectional evolution and upstream migrations following dam removal (Doyle *et al.*, 2003b).

In addition to the quantity and physical properties of sediment and substrate, hydrology and vegetation are additional factors exerting some level of control on the rate of geomorphic responses to dam removal. Cannatelli and Curran (2012) studied channel evolution after dam

removal and observed the rate of erosion and different stages of the conceptual model were heavily influenced by the local hydrology and establishment of riparian vegetation. However, Foley *et al.*, (2017a) conducted a review and synthesis of dam removal studies in the United States and found that hydrology is unlikely to effect the total amount of sediment eroded and substantial erosion can take place in the absence of high flows. Low flows may be sufficient to erode sandy sediments or cohesive silt and clay however gravels and coarser material are only likely to be moved in high flow events (Egan *et al.*, 2001)

The incision processes likely to occur after dam removal are described by Pizzuto (Pizzuto, 2002) and are likely to depend on the height of the sediment in relation to the downstream channel and the grain size. If cohesive silt and clay sediments are present then it is likely that a vertical headcut will migrate upstream through the fill (Pizzuto, 2002). Having data on the physical properties of the sediment upstream of the weir at Ennerdale Water will therefore provide useful information to inform predictions of channel migration post weir removal.

5.2.3 *Numerical Models*

Geomorphic modelling is a rapidly growing field in river restoration (Nelson *et al.*, 2016), and despite the high complexity and non-linearity of fluvial geomorphology, advances in computational power and computing methods are assisting this development (Coulthard and Van De Wiel, 2013). Modelling tools, can be used for a wide range of applications in fluvial geomorphology (Van de Wiel *et al.*, 2016) and, due to their increasing numbers, selecting the most appropriate model to address a specific research problem is often the most challenging step for inexperienced modellers (Nelson *et al.*, 2016).

Despite the observed advances in model development, relatively few numerical modelling studies examining the geomorphic responses to dam removal have been published in the last decade (Poeppl *et al.*, 2019). Those published have also been limited to 1-D models, that lack the ability to incorporate multi-dimensional processes, such as lateral movement (Poeppl *et al.*, 2019), and bank erosion (Foley *et al.*, 2017a) in their simulations.

An early example of the use of numerical models for dam removal studies was presented by Conly & Martz (1998). This study used the mobile boundary flow model, MOBED, to show the impact on longitudinal river profile following weir removal. Their results were able to estimate the depth of degradation of the river channel over a length of 300m and aggradation extending 2000m downstream (Conly and Martz, 1998).

Despite the limitations of 1-D models as described by Poeppl *et al.*, (2019), studies using these tools can produce useful information, generally with preferable computation time than

their 2D counterparts. Gartner *et al.*, (2015), were able to predict the location of sediment erosion and deposition using the HEC-RAS model and information on the pre-removal channel geometry.

More recently, 2D modelling has been employed to assess physical responses to weir removal. Howard, Coulthard and Knight (2017), used the hydromorphic model CAESAR-Lisflood to simulate erosion and deposition following weir removal in the River Derwent, Derbyshire, UK. The modelling was able to demonstrate how the presence of weirs had altered channel dynamics and identify erosion hotspots directly upstream of the structures after their removal (Howard *et al.*, 2017). Poeppl *et al.*, (2019), used the same tool to model channel evolution and sediment delivery following dam removal. The model simulated incision through reservoir deposits and knickpoint retreat in a similar way to studies and observations in the literature. However, in the absence of sufficient field measurements to set up and validate the model, only a qualitative assessment of the model performance was made (Poeppl *et al.*, 2019). As highlighted by *Poeppl et al.,* (2019), whilst numerical models are undoubtedly useful for studies in fluvial geomorphology, they should complement field or laboratory experiments rather than replacing them entirely (Coulthard and Van De Wiel, 2013).

5.2.4 *Physical models*

Physical models in fluvial geomorphology are usually scaled representations of real-world environments set up in a flume or laboratory (Van de Wiel *et al.*, 2016). Physical models much like numerical models are useful as they offer experimental control (Van de Wiel *et al.*, 2016) and repeatability. However, a major drawback is the non-linear scaling of physical governing law and equations (Paola *et al.*, 2009); a problem which is further emphasised when considering the influence of vegetation or animals on a physical environment (Rice *et al.*, 2010). An important role that physical models can play is in providing data with which to calibrate and validate numerical models. Laboratory environments can provide the conditions within which to make detailed observations and measurements and thus inform numerical models (Van de Wiel *et al.*, 2016). Within the literature there are a number of detailed studies that assess the physical modelling tools in geomorphology. Peakall *et al.*, (1996) present a critique of the principles determining an model's ability to accurately replicate both the dynamics and form of natural alluvial systems. Their study contains examples from a range of models and analyses both their strengths and limitations whilst highlighting their contribution as a complimentary tool to the field of fluvial geomorphology (Peakall *et al.*, 1996). A more

recent overview of the different types of physical models and their applications is provided by Green (2014). Within this book chapter a summary of the main advantages and disadvantages of physical models is also provided. An example of a limiting factor for their widespread application provide by Green (2014) is that physical models often require large amounts of space and specialist equipment which may not always be available. Compared to the flexibility offered by computational modelling techniques this represents a major limitation, in addition to time and cost constraints.

5.2.5 *Observations from Dam and Weir Removals*

Monitoring and measuring physical responses to dam removal provides further evidence of the types and rates of responses that can occur after dam removal (Doyle *et al.*, 2002). For example, depending on the physical properties of the sediment upstream of the dam or impounding structure, a clearly marked wave of sediment transportation may occur or material may be gradually eroded away (Doyle *et al.*, 2002).

Observations and monitoring studies of dam and weir removal projects are undoubtedly useful for gathering more information and understanding of the geomorphic responses to this management strategy. However, their utility in predicting how a specific river or catchment may respond to dam removal is questionable due to the large heterogeneity between sites. Factors such as the height of the dam, the fill sediment, the impoundment size and the age of the structure can all influence geomorphic response however there are a host of other factors (Pizzuto, 2002). Amongst these are the dam removal strategies themselves which can also lead to a wide range of channel responses (Major et al., 2017). Despite this, a coupled-system response of erosion and downstream sediment transport and deposition is common to most sites although the rates and processes by which this occurs can vary between sites (Major et al., 2017).

Doyle *et al.*, (2005) proposed that geomorphic responses to dam removal should be most evident in the areas proximal to the removal site and decrease exponentially with both distance time. They found however, that there were few studies from which to base these qualitative predictions (Doyle *et al.*, 2005). Pizutto, (2002) highlighted that fluvial processes are complex, occurring over long time periods. Geomorphic responses to dam removal are myriad, and with a lack of direct observational data, predictions are inherently uncertain. However, the processes associated with dam removal such as incision, channel development and increased sediment supply all occur naturally elsewhere and have been well studied by geomorphologist and can thus serve as analogies for dam removal predictions (Pizzuto, 2002).

In the last two decades the number of dam removal studies has increased. Tonitto and Riha, (2016) reviewed the literature and studies of small dam removals in the Eastern United States. The research found that despite the large number of dam removals, observations of the removal process were limited making it difficult to form general conclusions about the impacts. The research also suggested there is a need to base dam removal planning on observations but that these observations have limited replicability (Tonitto and Riha, 2016).

Unlike models, observing and monitoring changes from dam removals provides and opportunity to understand physical process at scale. However, a lack of systematic Before-After Control-Impact (BACI) studies has limited their application. A synthesis of observations from 20 dam removal studies in the United States by Major et al., (2017) concluded that "consistent and thorough documentation of physical and ecological responses to dam removals is needed to improve our understanding of environmental benefits and detriments of dam removal". The authors of the study also called for thorough documentation on the pre-removal and where possible, pre-impoundment conditions, in addition to monitoring sediment changes over the long-term after dam removal, to increase our understanding of the factors that influence different response to dam removals (Major et al., 2017).

Combining observations with or comparing them to model predictions provides an opportunity to test and potentially enhance the performance of numerical models. Cui *et al.*, (2019) used post-removal monitoring to validate pre-removal sediment modelling forecasts made using the DREAM 1 model. The study found that reach-scale predictions were generally accurate however, at specific locations discrepancies between the predicted and observed magnitudes of sediment deposition occurred (Cui *et al.*, 2019). The authors also note that, rerunning the model with improved input data did not substantially improve model performance or impact general conclusions, and advocate *"a simplified data collection approach that enables timely predictions for decision making and minimizes study costs"* (Cui *et al.*, 2019).

5.2.6 *Ground Penetrating Radar*

Non-intrusive methods for analysing lake sediment are limited. Water-borne Ground Penetrating Radar (GPR) provides a novel means of characterising the sediment up-stream of the impounding-weir without the immediate need to disturb any substrate. This method, therefore, reduces the risk of harming downstream fauna in the riverine SAC and also reduces

the likelihood of causing harm or physical damage to species or habitats within Ennerdale Water SSSI.

GPR has become a valuable tool for analysing sediment properties in a range of environments. Recent studies have shown its application in glacial, fluvial and coastal, and peat environments (Robinson *et al.*, 2013). Other varied applications include forensic surveys for criminal investigations, detecting buried utilities (Daniels, 2005) and monitoring groundwater fluctuations (Lu and Sato, 2007). It has also been used extensively for historical reconstruction of depositional environments and sedimentary processes (Neal, 2004).

A case study of wandering channel bars in a wandering gravel-bed river (Wooldridge and Hickin, 2005) used a combination of GPR and bathymetry and aerial photography to map 50 years of channel evolution in British Columbia. Another well-documented use of GPR is to locate and map paleo-channels and other river landscape features. (Nimnate *et al.*). Cross-bedding, also known as cross-stratification, refers to areas of sub-stratum that intersect horizontal layers at an angle. GPR can detect evidence of cross-bedding, or the presence of a thalweg in the sub-stratum which can provide evidence of paleo-channels (Wyatt and Temples, 1996). In some areas, surface evidence of paleo-channels may be lacking (Buynevich, 2019) or obscured by water, as may be the case behind dams and weirs, so the use of GPR can provide a method of overcoming this.

GPR uses electromagnetic pulses to detect changes in the electrical properties of materials below the ground (Neal, 2004). The system typically uses a transmitting antenna and receiving antennae. The electromagnetic pulses emitted by the transmitting antennae travel into the subsurface and reflect of interfaces or scatter from point sources (Baker and Jol, 2007). Sudden changes to the dielectric constant of subsurface materials causes some of the reflected energy to be picked up by the receiving antenna of the GPR system (Daniels, 2005). The radar frequency typically range from 10-1000 MHz (Neal, 2004). The choice of radar frequency depends on the purpose of the survey and the physical characteristics of the study site. Higher frequency radar tends to produce higher resolution data and imagery whereas lower frequency radar can penetrate to greater depths but at the expense of detailed data.

GPR is most commonly used for conducting land-based surveys and within the literature there are a comparatively small number of examples of its application for water-borne surveying of lake sediments and currently no cited previous application of this technique in the UK. Other examples of its application in both natural and man-made water bodies include, Santaniello *et al.*, (2010) Sambuelli & Bava (2012) and Mihu-Pintilie *et al.*, (2016). Hunter *et al* (2003) concluded that GPR can be a quick and cost-effective method for assessing sediment infill of

reservoirs during their study of Ship Creek Reservoir, Alaska. However, unlike the aforementioned examples, the investigation in Alaska (Hunter *et al.*, 2003) took advantage of a frozen lake surface from which to conduct the survey which negated the need for a boat and was therefore akin to conducting a land-based survey. However, in addition to other examples of GPR surveys on lakes, it provides further evidence of the suitability of the application of the technique at Ennerdale Water albeit from a boat rather than on ice.

5.2.7 *Literature Review Summary*

A number of methods are available for predicting the hydromorphological responses to dam removal. The literature highlights the importance of gaining a detailed understanding of the catchment morphology and sediment characteristics to support numerical modelling. Given time and resources, the most suitable approach would be to use a suite of predictive tools that provide complimentary and overlapping sets of predictions with the aim of increasing understanding of the process and confidence in the predictions (Van de Wiel *et al.*, 2016). For the upper Ehen a numerical modelling study complimented with additional field data was carried out. However conceptual models and observations from previous studies were also used to inform the study and in the interpretation of the modelling results. This approach offered greatest control over controlled scenarios and repeatability of model runs. Secondly, the model outputs provided a sound evidence base for further detailed hydrological studies on the impact of weir removal on the lake level and the downstream hydrology, a key aim of this thesis.

5.3 Sediment Surveys and Modelling Approach

5.3.1 *Fieldwork Rationale*

Sediment surveys were conducted to assess the impact of engineering works upstream of the weir, characterise the lakebed sediments and provide information on substrate grain size and bedrock depth critical inputs into the geomorphic model.

For most studies in fluvial geomorphology, the only way to understand the long-term variability of river channels is through the collection and analysis of historic data (Grabowski and Gurnell, 2016). The lake outlet at Ennerdale Water has undergone several dramatic and abrupt physical changes over the last 150 years. Most recent of these was the reconnection of Ben Gill, a high gradient ephemeral tributary, that now joins the River Ehen, directly

downstream of Ennerdale Water (Figure 87). In 1995, the current weir was installed, replacing the previous longer structure positioned slightly further upstream at wider point of the lake outlet. In the 1970s, channels were excavated upstream of the weir. During drought conditions, this allowed water to reach a gravity fed intake to the water treatment works, which is no-longer in use. In the same decade Ben Gill was diverted into the lake via a culvert in an attempt to provide additional inflows into Ennerdale Water. Gathering evidence on both the current and historic geomorphology of this region of the lake is therefore necessary to provide further data for the model construction and evidence to help interpret the result geomorphic modelling study.

Fine sediment deposition and poor substrate quality have been shown to have a detrimental impact on freshwater mussel populations (Geist and Auerswald, 2007). Identifying potential sources of fine sediment and characterising the physical and chemical attributes of these is therefore essential as these may be mobilised in the event of weir removal, posing a threat to the downstream riverine ecology. Quantifying any risk posed by sediment currently stored upstream of the weir infrastructure will help inform the need for mitigation measures and ultimately influence decisions over the future of the impounding weir. Understanding the geomorphic characteristics of the site is critical to understanding future evolution of the channel. The additional data generated through field surveys will underpin the modelling process as it will facilitate model calibration based on physical conditions. Such initial and boundary conditions influenced by the physical properties of the domain: spatial distribution bed morphology, water depth, sediment grain sizes and roughness coefficients (Van De Wiel *et al.*, 2011). Properties of sediment texture and particle size distribution also influence erosion rates in specific landscape evolution models (Hancock *et al.*, 2011).

The results from the field work are presented in section 5.5, prior to the geomorphic modelling study, presented in 5.6. Section 5.6 also describes the influence of field results on model inputs including field justifications of model parameters.



Figure 87. Aerial imagery of the weir at Ennerdale Water and upper River Ehen. The top image is from 2008 prior to the reconnection of Ben Gill. The bottom image shows the changes caused by Ben Gill in 2018, approximately 26 months after reconnection. Evidence of the old weir structure upstream of the current weir and excavated channels from periods of drought is also shown. Source: Google Earth, 2020

5.3.2 *Modelled scenarios*

A range of preliminary modelling scenarios were presented to the primary stakeholders overseeing the management of the River Ehen Compensatory Measures. This allowed for the discussion of, and amendment to, the proposals. After an initial consultation period, it was decided that a revised set of scenarios should be prepared taking on board the following issues and suggestions raised.

- Possibility of modelling multiple weir removals
- Need to model a zero or no change scenario
- Weir removal only scenario should not be considered

The outflow of Ennerdale Water is a dynamic system with complex natural and human interactions to consider. The following proposals were designed to address all of these issues without creating an extensive range of modelling scenarios that could potentially put excessive resource and time constraints on the project.

Scenario 1 - No infrastructure removal (Model Spin-up)

Scenario 1 offers a baseline for comparison by assuming no change to the existing infrastructure. Additionally, this scenario will validate the ability of the geomorphic model to accurately replicate existing conditions. Under this scenario no physical changes to the existing infrastructure are implemented within the model, however the input flow regime will be altered to account for the cessation of abstraction from Ennerdale Water. The effects of this abstraction change and the interaction with Ben Gill in its current reconnected state can therefore be analysed.

Scenario 2 – Infrastructure Removal

Under this scenario the weir and the associated infrastructure are be removed. This includes the fish pass, disused water intake, the historic weir upstream of the current weir, the walls that line the upstream banks of the River Ehen and the lake outlet. The only remaining infrastructure is the walls at the confluence of Ben Gill and the River. The riverbed and area immediately upstream of the weir are regraded in order to negate the effect of relic channels excavated in drought periods. If the man-made channels were left in the DEM, water from the lake would simply follow these channels, eroded any exposed sediments. These are not natural features of the lakebed and would therefore create an artificial influence on the rates and type of channel evolution. An aim of the thesis is to understand the natural geomorphology of the lake outlet and modelling the geomorphic response of the lake in a more natural conditions will thus help achieve this aim.

Scenario 3 – Additional barrier removal

The final scenario also involves the complete renaturalisation as proposed in scenario 2 however, it extends further downstream to incorporate two additional instream barriers: an old

bleaching weir approximately 450m downstream of Ennerdale Water and Bleach Green gauging weir (NRFA station No. 74003) approximately 800m downstream of the weir. These barriers are highlighted in Figure 98 (Section 5.6.2).

5.4 Ennerdale Water Sediment Survey Methodology

Field work was carried out in two phases using two techniques. First, a widespread exploratory survey of the lakebed morphology was conducted using Ground Penetrating Radar (GPR). Secondly a more focused sediment sampling approach was used to obtain information on the physical and chemical composition of sediments in specific points of the lake.

5.4.1 *Ennerdale Outlet GPR Survey and Equipment Set-up*

The GPR survey was conducted between the 19th and 25th of July 2017. Surveys were conducted by small fieldwork team consisting of; a boat operator; a GPR operator and navigator; additional team members on the lake shore contactable for logistical back-up and, if needed, emergency assistance by walkie-talkie. Prior to arrival at the field site for the survey work, trial runs were carried out to ensure that all the equipment was working and to decide upon the most appropriate equipment configuration within the boat. Due to space restrictions and additional hazards associated with working on water, substantial care and attention were paid in setting up the GPR in the zodiac rigid Inflatable boat (RIB). A system was implemented which minimised clutter in the boat and reduced the risk of damaging the equipment and the potential for accidents. The final layout of the equipment is shown in Figure 88.

A plastic crate (Figure 88) with lattice sides was located centrally in the boat and used as a central hub for storing and protecting the bulk of the fibre optic cables, whilst at the same time allowing them to be fed through at various points close to their corresponding connections. Wooden braces fixed to the deck of the zodiac were used to hold the crate and contents firmly in place and so negate any risk of damage to equipment during surveying. The crate was covered with a plastic lid to provide protection from rain or spray from waves. This was secured in place using a bungee cable and had the additional benefit of offering a stable platform for additional equipment (ruggedized laptop, field survey book, handheld GPS etc.). The control panel was also held firmly in place on the seat of the zodiac by means of a bungee cable to prevent it moving or falling.

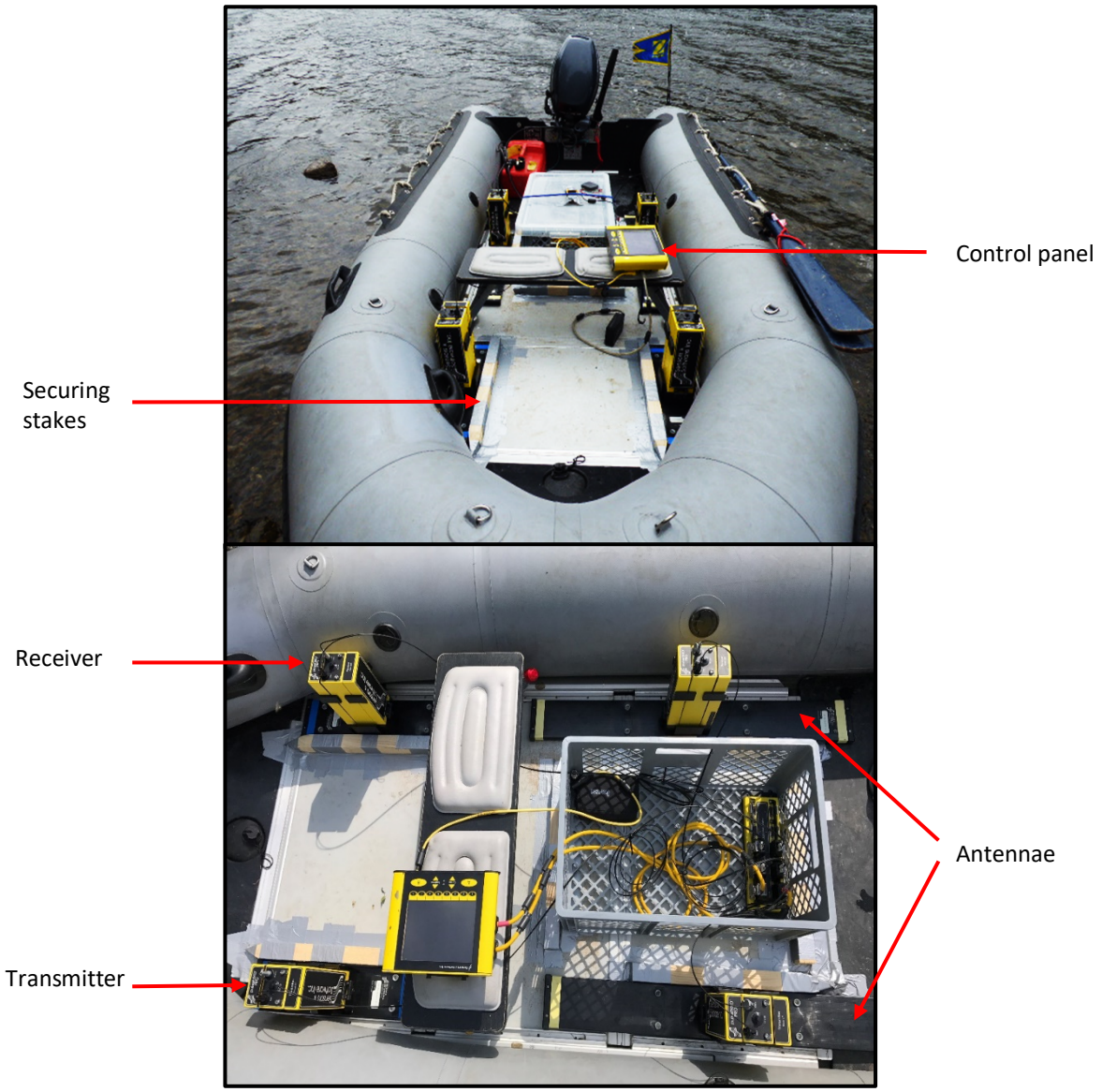


Figure 88. Final setup and configuration of GPR equipment for Ennerdale Water sediment survey.

Whilst the optimum spacing for antennas should be 1m for the 100 Mhz system and 0.5m for the 200 MHz system, the limitations of the dimensions of the boat and space required for working on board meant that a compromise had to be made. The GPR antennas were positioned as close to the side of the zodiac as possible with a spacing of 75cm. The position of the antennas was marked out on the deck of the zodiac and wooden braces (Figure 88) were fixed in place. These served to prevent the antennas moving and add stability to the transmitters and receivers whilst in transit.

A network of gridded lines at 50m intervals was drawn using GIS software which covered the western end of the lake and surrounding area. This was then clipped using a shapefile of the lake to leave a series of gridded lines which provided a set of initial and end points to use as navigational guides when piloting the zodiac across the lake. Navigation of the boat between the start and end coordinates was achieved using a handheld Garmin GPS system with

instructions on which heading to follow being passed from the navigator to the boat operator. Minor limitations of this technique were the delay between the GPS signal informing the boat operator to adjust course, the ability of the boat operator to adjust the course accordingly and maintaining a steady course given the effects of wind.

5.4.2 *Sediment Coring*

A second field study (17th June 2019) was undertaken to collect samples of sediment from the lakebed. The objectives of this were twofold. Firstly, a physical characterisation of the lakebed sediment would assist in defining the final modelling parameters (which are set out in part 2 of this chapter). Secondly, identifying areas of fine sediment, estimating the volume and characterising the chemical composition enabled an initial assessment of the potential risk to the downstream mussel population. This highlighted the need for any future remedial engineering works that might need to be carried out in addition to weir removal, should the decision be taken to proceed with this management option.

Two types of corer were used for the fieldwork. The first method for collecting sediment samples was with the use of a UWITEC gravity corer (Figure 94). This was operated from the side of a small Zodiac inflatable boat. The corer works by penetrating the sediment under its own weight. Once the corer has reached the lakebed and been allowed to settle a cord is pulled abruptly to release the mechanism holding the cap off the corer. The cap then creates a vacuum allowing both the corer and sediment to be slowly retrieved to the surface. At the surface a bung is paced under the corer whilst still submerged to prevent loss of sediment. This then creates a sealed tube, from which the sediment core can be extracted either in the boat or onshore. In this instance all samples were extruded onshore due to the inclement weather conditions.

To obtain a longer sediment core and penetrate more resistant substrates, a Rod Corer or Livingstone Square-rod piston corer was used. This was operated from a floating platform anchored firmly in place above the chosen site. The Livingstone corer consists of a core tube which provides a guide for the square rod and piston cable to run through. Casing tubes were used to keep sediment package in-tact and minimise disturbance of the core tube contents. The casing also ensures that the corer penetrates the same location which each additional drive of the corer. Additional drives are achieved by attaching additional lengths of piston rods. The operation of this method of coring requires at least three people and preferably 5. Setting up and operating this system on a floating platform is both time consuming and complicated.

A series of locations from which to take sediment samples was established and plotted on the map using ArcGIS. This was designed to allow for a systematic coverage of the area behind the weir, providing both cross-sectional and longitudinal profiles of the lakebed sediment.

The area behind the weir has been impacted upon by engineering works during the construction and removal of different weirs, remedial actions during drought conditions and changes to surrounding infrastructure. The survey therefore aimed to analyse areas of both disturbed and intact sediment packages.



Figure 89. Location of sediment samples extracted during field monitoring. Red dots show locations of short surface sediment cores. Blue dot shows location of the longer Livingstone corer sample.

5.5 Sediment Survey Results

The results from the field studies are presented prior to the detailed description of the geomorphic modelling as the results of these surveys helped inform the final model setup.

5.5.1 *Ground Penetrating Radar*

A large quantity (>38kmof GPR profiles) of high-quality data was obtained over the two full days of surveying. GPR data processing was carried out using ReflexW software (Sandmeier Geophysical Research). Once a suitable set of processing steps was identified, processing flows were saved and then reapplied simultaneously to numerous lines of GPR data. However, due to the large distances covered by the GPR and the varying depths of the lakebed and sediment profiles the majority of the lines required slightly different values for many of the steps, making data processing more time consuming.

Five key processing steps were applied to the GPR data in order to improve the clarity of the data and allow for better interpretation and mapping of the sediment profiles (Table 19).

Table 19. The five processing steps applied to the raw GPR data to increase clarity and aid interpretation. Information on the function of each step is taken from the ReflexW built-in help guide and user experience.

Processing Step	***Function***
Move Start time	A filter that acts on each trace independently and facilitates a static correction in time. The value to adjust the start time can be obtained using the wiggle window function.
Subtract Mean (dewow)	Used for eliminating a possible low frequency part of the trace. A running mean value is calculated for each trace and subtracted from the central point.
Bandpass frequency	Used to supress noise. A Butterworth bandpass frequency was used and values for the lower and upper cut-off assigned.
Manual Gain	Used to emphasize y-ranges where structures are present.
Background Removal	A filter to subtract an averaged trace over a chosen time/range of the selected profile.

Of the two radar frequencies, the 100MHz GPR consistently produced the highest quality imagery, which is consistent with findings from previous water-borne GPR surveys (Hunter *et al.*, 2003;; Sambuelli and Bava, 2012; Mihu-Pintilie *et al.*, 2016). An example of an unprocessed 100MHz GPR profile compared with an unprocessed 200MHz profile is shown in Figure 90, along with the location the data was obtained from. An example of raw GPR data and the final processed data is shown in Figure 91. Horizontal banding in the water column was difficult to remove, however its presence doesn't hinder the interpretation of the lake sediment layers.

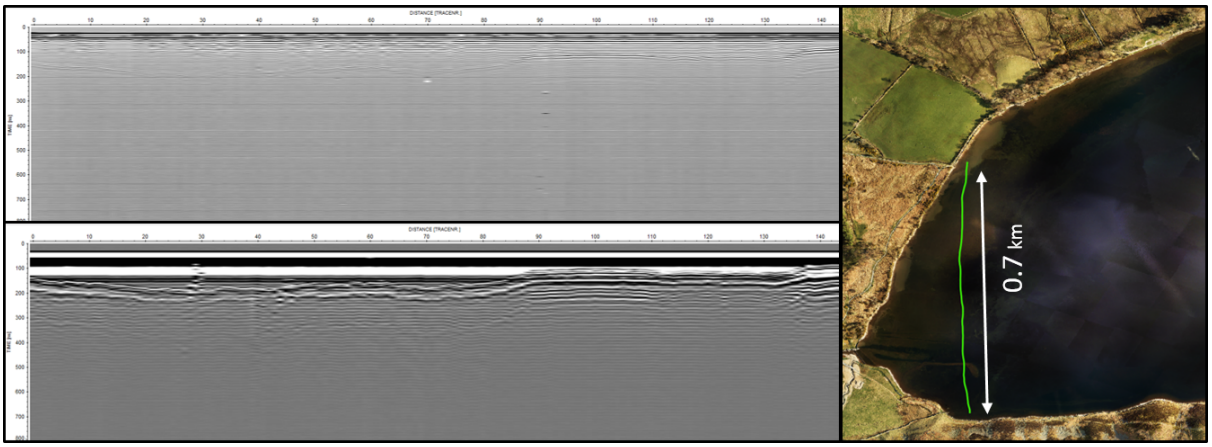


Figure 90. Comparison of raw 200 MHz (above) and 100 MHz (below) GPR data. The profile was taken North (right hand side of profile) to South across Ennerdale Water

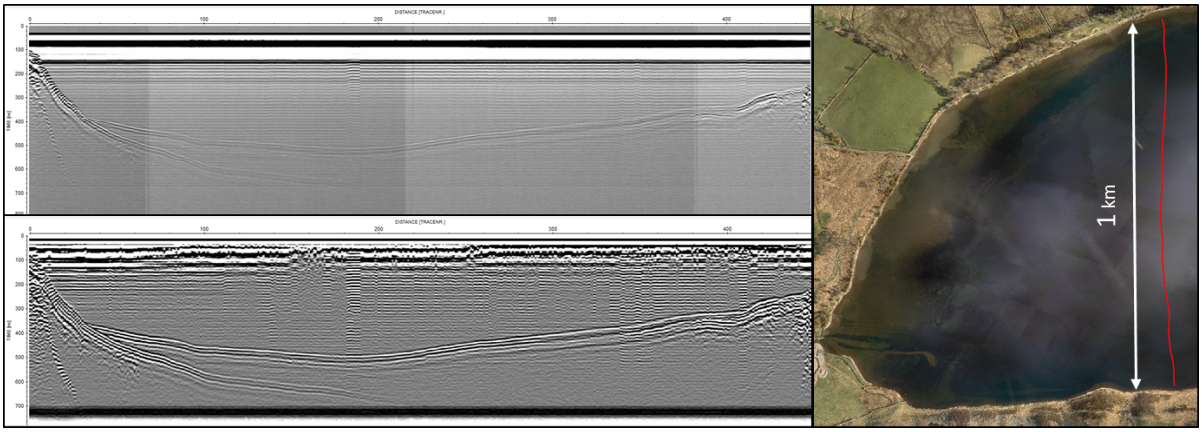


Figure 91. Images showing raw GPR data (top) in the ReflexW software viewing window; processed data (bottom) using the processing steps outline in table 1 and the location (right) where the data was obtained. GPR lines run from South on the left of the image to North on the right of the image.

One of the objectives of the survey was to find evidence of paleochannels in the area directly upstream of the weir. The collection of GPR data in this area was limited due to the shallow water depths, which made it difficult to manoeuvre the propeller driven boat. The closest survey line to the weir does show some evidence of cross-bedding in the area approximately 20 metres north of the current intake into the fish pass (Figure 92). Evidence of channel migration is also evident in the downstream section of the Upper Ehen as shown by the cut off meanders in the imagery in Figure 93.

The results of the GPR data and processed images also provide evidence of the lack of near-surface bedrock in the area immediately upstream of the weir. This is an important input for the subsequent geomorphic modelling. The presence of bedrock in the upper surface could limit the rates and extent of both upstream and lateral channel migration after weir removal. However, the GPR evidence suggests that any Bedrock is well below the influence of fluvial erosion.

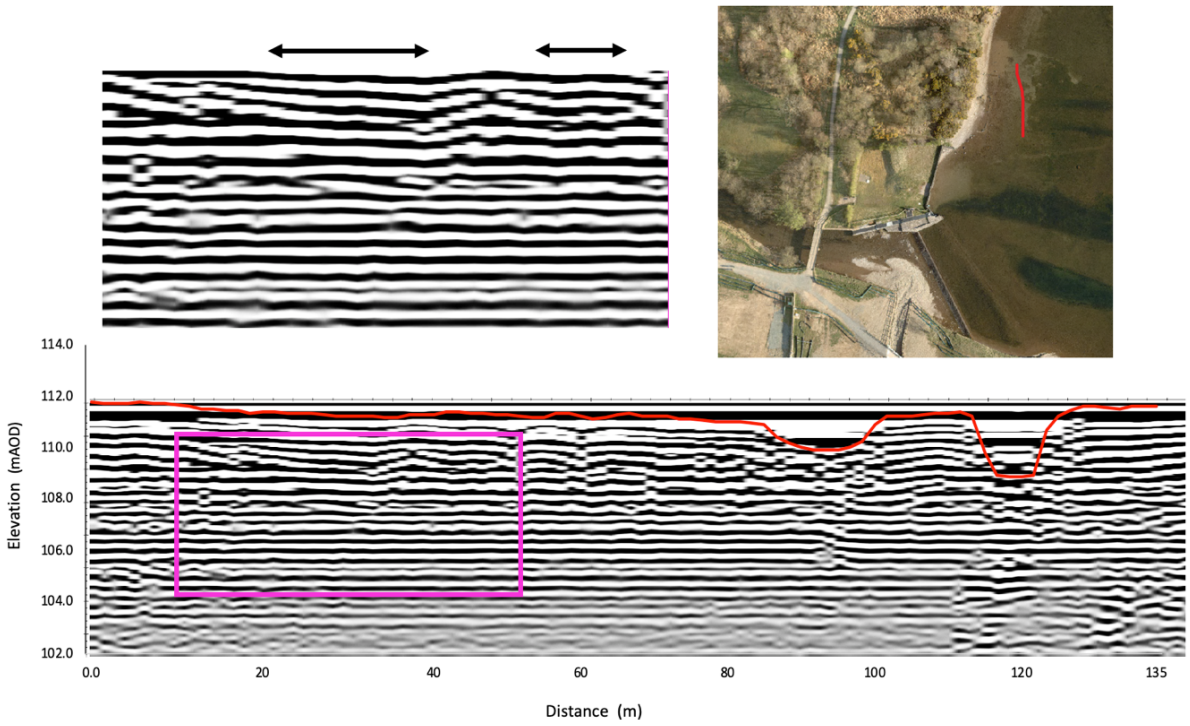


Figure 92. Evidence of cross-bedding to the north of current outlet may suggest previous fluvial activity and lateral channel movement. Ride line in aerial phot shows approximate location of subsurface features

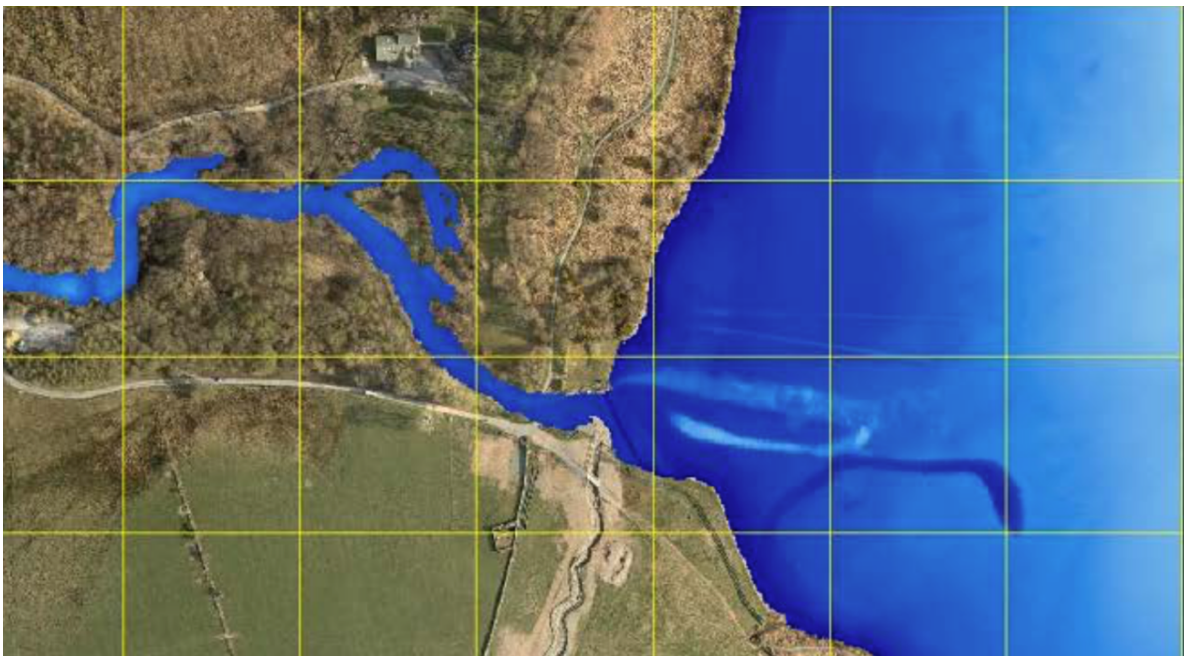


Figure 93. Supporting evidence of channel migration shown by relic meanders downstream of Ennerdale water. Grids are 100m by 100m.

5.5.2 *Sediment Analysis*

An initial attempt to extract a longer core at the proposed site was unsuccessful. This was due to the impenetrable nature of the lakebed which was found to be largely comprised of large gravel and angular cobbles. Further attempts were made at progressively deeper areas of the

lake until a suitable site was eventually found (Figure 94). This process was hampered by strong winds and rain which made relocating and securing the coring platform difficult and time consuming. In the time available during the field visit, a single sediment core measuring approximately 3m was extracted using this technique.

Multiple attempts were also made to sample the lake sediment at the chosen sites using the gravity corer. These again proved unsuccessful for the same reason inhibiting the Livingstone Corer. It was therefore decided that it would be of more use to collect multiple samples from the man-made trenches in close proximity to the weir rather than attempting to collect lake sediments from further into the lake in areas unlikely to be mobilised in the event of weir removal.

Sediment samples were taken from both trenches. These were extracted from the corer at 5cm increments and stored in zip lock bags. Due to the unconsolidated nature and high water content of the sediment it was not possible to keep the cores intact. The samples were kept in refrigeration to limit bio-chemical degradation occurring until further analysis could be carried out. In addition to these samples an exploratory investigation of the sediment in the trenches was conducted using a gouge corer. This was used to attempt to penetrate the lakebed below the layer of fine organic sediments and retrieve a sample of the underlying substrate.

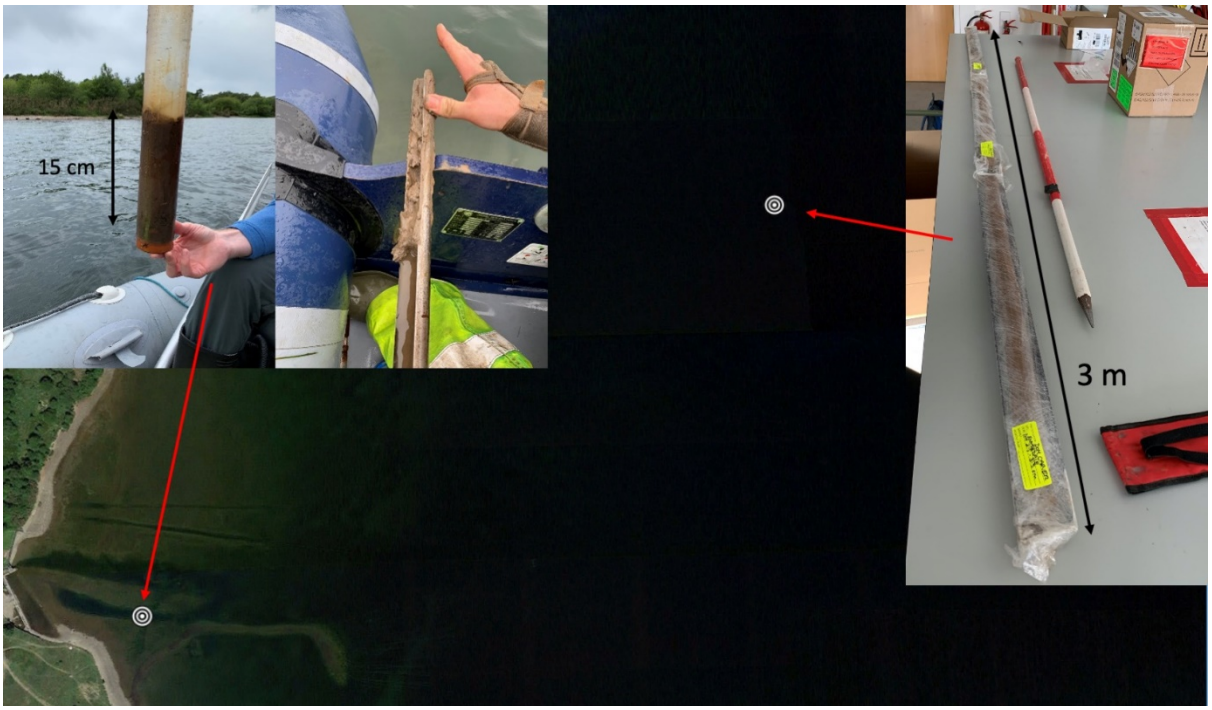


Figure 94. Locations of sediment samples from field surveys. Aerial imagery credit Google Earth

Both coring methods were unable to extract sediment in the area directly upstream of the weir. Multiple attempts were made however the Livingstone corer was only able to penetrate the lakebed substrate once the platform had been positioned in water approximately 5metres deep.

The area close to the weir is influenced by two important factors that mean the sediment is largely comprised of coarser gravels and large cobbles. The first is the proximity to the alluvial fan or debris cone created by Ben Gill. This has deposited coarse material both upstream and downstream of the weir over centuries of fluvial erosion and deposition.

A second important factor is wave action across the lake. The area behind the weir is shallow and this part of the lake has a large fetch meaning the wave action keeps any finer material in suspension and thus limits the accumulation of fine sediments. Despite the lack of samples in close proximity to the weir, the additional evidence collected from the surveys shows that fine sediments are not a major component in the surface layers of substrate close to the weir.

Sediment was however obtained from the man-made channels close to the weir. Two samples were analysed in the laboratory by technicians from the Environmental Engineering research group in the School of Engineering, Newcastle University. These samples were analysed using Ion Chromatography to identify anions and cations (Table 20) and using inductively coupled plasma mass spectrometry to identify metals (Table 21).

Ion chromatography has been used for determining a range of anions and cations in freshwater sediments since the late eighties (Bak *et al.*, 1991). Anions were determined with chromatography (Dionex Integrion High Pressure Ion Chromatography System, Thermo Fisher Scientific, US. Metal analysis of the wastewater was conducted by ICP-OES (VISTA-MPX, Varian, US)

Table 20. Results from the Ion Chromatography laboratory analysis. Values shown in ppm.

Chemical	**Symbol**	**Sample 1 - Depth**			**Sample 2 - Depth**		
		0-5 cm	**5-10 cm**	**10-15 cm**	**0-5 cm**	**5-10 cm**	**10-15 cm**
***Fluoride** (ppm)*	*F*	0.036	0.045	0.043	0.016	0.015	0.018
***Chloride** (ppm)*	*Cl*	2.800	2.300	5.460	11.500	2.930	3.440
***Nitrite** (ppm)*	NU_2	-	0.316	0.188	-	0.079	-
***Bromide** (ppm)*	*Br*	0.293	0.167	-	0.140	0.071	0.067
***Sulphate** (ppm)*	SO_4^{2-}	2.224	13.943	126.557	2.640	2.720	3.170
***Nitrate** (ppm)*	NO_3^-	20.928	11.640	6.145	21.015	4.750	1.910
***Phosphate** (ppm)*	PO_4^{3-}	1.970	1.600	0.647	1.029	0.960	0.540

Table 21. Results from the inductively coupled plasma mass spectrometry in ppm

Metal (ppm)	Symbol	Sample 1 (Depth)			Sample 2 (Depth)		
		0-5 cm	5-10 cm	10-15 cm	0-5 cm	5-10 cm	10-15 cm
Calcium	*Ca*	3.505	5.157	39.496	2.450	0.728	0.550
Magnesium	*Mg*	0.953	0.855	1.637	0.957	0.301	0.220
Sodium	*Na*	1.466	1.219	3.752	4.620	1.470	1.509
Potassium	*K*	0.327	0.236	2.743	2.740	0.380	0.278
Zinc	*Zn*	0.016	0.025	0.004	0.013	0.004	0.005
Lead	*Pb*	0.004	-	-	-	0.003	-
Nickel	*Ni*	0.004	0.004	-	0.003	-	0.003
Aluminium	*Al*	0.015	0.032	0.020	0.011	0.007	0.007
Iron	*Fe*	0.013	0.060	0.019	0.002	0.008	0.013
Manganese	*Mn*	0.006	0.006	0.001	0.006	0.002	0.001
Silicon	*Si*	2.052	2.139	1.382	2.327	5.110	7.290
Chromium	*Cr*	0.001	-	-	0.901	0.661	0.802
Sulfur	*S*	0.729	3.408	29.070	0.009	0.003	0.002
Strontium	*Sr*	0.012	0.015	0.116	0.231	0.005	0.003
Barium	*Ba*	0.035	0.062	0.040	-	-	0.029

In 2007, the European Committee for Standardisation (CEN) published a guidance document which established clearly defined methods for monitoring freshwater pearl mussel populations and the environmental requirements for sustaining healthy populations (CEN, 2017). Within the guidance, Phosphorus, Nitrate and Calcium are listed in the checklist of recommended water quality parameters to monitor. The guidance does not say what the acceptable levels of these parameters are, however, the information provided in the sediment samples highlights additional considerations that need to be taken into account prior to weir removal.

The results provide critical additional information for use by the regulatory agencies and the team of expert freshwater mussel ecologists that will use the data to inform management decisions and decide upon possible mitigation measures. The implications of the results of the sediment analysis are discussed in more detail in Chapter 7.

An estimate was made of the quantity of fine sediments that have accumulated in the man-made trenches since the end of the 1970s (). This was done by drawing a polygon around the trenches in ArcGIS to obtain the area. This was then multiplied by an average thickness of 15cm. Sediment thickness was likely to be greater than this as the coring method is not 100% efficient however it is not a compact sediment package and 15cm was deemed an appropriate estimate. Based on this assumption a total volume of 825 m^3 was obtained. It is important to note that this sediment package is very loosely consolidated and has a very high water content. The total volume of solids is likely to be lower, however based on results of the field

surveys this zone is still likely to be the major source of fine sediment if mitigation measures are not implemented post weir removal.



Figure 95. Estimated volume of fine organic sediment with potential for mobilisation based on an average thickness of 15cm across the bottom of the channels.

5.6 Modelling the Geomorphic Response to Weir Removal

This section contains general information about how CAESAR-Lisflood functions as a modelling tool and describe the benefits and limitations of using this particular model for answering the project-specific research questions.

5.6.1 *Justification of model choice and modelling approach*

A major requirement of the EngD thesis is to be able model lake levels and how these may change in response to weir removal scenarios and different outlet forms. This will provide important information on the future appearance of the site and for managing the transition from the current conditions to a more natural system.

Whilst, several geomorphological models are available for showing landscape and channel evolution in response to dam removal, an in-depth review and comparative testing of different model performance and functionality was beyond the scope of this study. However, after reviewing the application of a number of different models in the academic literature (e.g. Goodell and Bradley, 2005; Cui and Wilcox, 2008; Greimann and Andrews, 2010; Van De Wiel *et al.*, 2011; Poeppl *et al.*, 2019), CAESER-Lisflood was chosen for this study as it has a set of characteristics which make it a more

appropriate and practical tool compared to alternative models. For example, several studies from the United States have documented the use of the Dam Removal Express Assessment Models, DREAM 1 and 2, and supported their use with field observations (Cui *et al.*, 2019). However, DREAM 1 requires data on long-term average sediment supply rates from the upstream end of the study reach and tributaries as model inputs (Cui *et al.*, 2006). This data was not available for the Upper Ehen catchment and collection of this additional data was beyond the scope of the study.

CAESAR-Lisflood, has been applied to a range of similar scenarios in the UK (Howard *et al.*, 2017) and Europe (Poeppl *et al.*, 2019) to model the impact of dam and weir removal, which provides evidence of its suitability for this application. It is freely available meaning the methodologies applied in this study can replicated or scrutinised without the need to purchase software. It has the option to modify or expand on its functionality using computer coding but also has an existing set of additional in-built features such as visualisation of model outputs to Google Earth, which facilitates the communication of model outputs, an important consideration of the project and the end users of the modelling tool.

CAESAR-Lisflood modelling tool is able to generate lake level data as the sum of water depth and bed elevation for each cell within a gridded domain. Lake level is therefore inferred given the bed elevation for a particular cell and requires knowledge of this elevation for every cell within the domain at every timestep. Simulations using a daily timestep would therefore require the model to generate a DEM of the entire domain for ever day of the simulation duration. This would be computationally expensive in terms of both processing power to complete the simulation and memory intensive due to the large numbers of files needed to be generated and stored. Simulations of more than a year in duration at this scale could take days/week to complete. This is impractical for testing multiple engineering options, such as the original design of the lake outlet, or landscaping of the area around the water after its removal. Running CAESAR-Lisflood at catchment or lake scale would also need to be done at lower resolution to avoid excessively long model runs. At coarser resolutions, (>5m) detail of the lake outlet profile would be lost, making it harder to analyse the impact of outlet form on the future discharge and lake storage relationship.

Using SHETRAN to provide the inflows into the CAESAR-Lisflood model meant that a more thorough model calibration based on observed data and hydrological behaviours could take place. The use of SHETRAN and Level Pool Routing also allows for a range of future management scenarios to be rapidly modelled allowing for a quick assessment of their impact on the lake level and downstream hydrology when compared to running the CEASAR-Lisflood multiple times. It was therefore decided that using different models best suited for the individual aims of the project would produce better outputs and greater flexibility.

Furthermore, during the project, different tasks could be run concurrently to optimise efficient use of time.

5.6.2 *CAESAR Model Construction and Input Data*

The model was run at 2m resolution for two domain sizes. A 1m resolution DEM was available and tested however this resulted in extremely slow model runs, in some cases taking over three weeks to complete 140 days of modelled data.

Digital Elevation Models and Infrastructure Representation

The DEM was taken from a 1m resolution bathymetric lidar survey. This was carried out in 2007 for United utilities. This survey was conducted prior to the reconnection of Ben Gill. Ben Gill is an ephemeral channel which was reinstated after more than 30 years of diversion via a culvert. A DEM of Ben Gill produced in January 2018 was provided by Joel Blackburn (Aberdeen University) and combined with the main DEM using ArcGIS. The original Ben Gill DEM (Figure 96a) was resampled to 2m (Figure 96b) and then manually edited to remove the bridge near the confluence with the River Ehen (Figure 96c). This was then merged with the full 2m DEM.

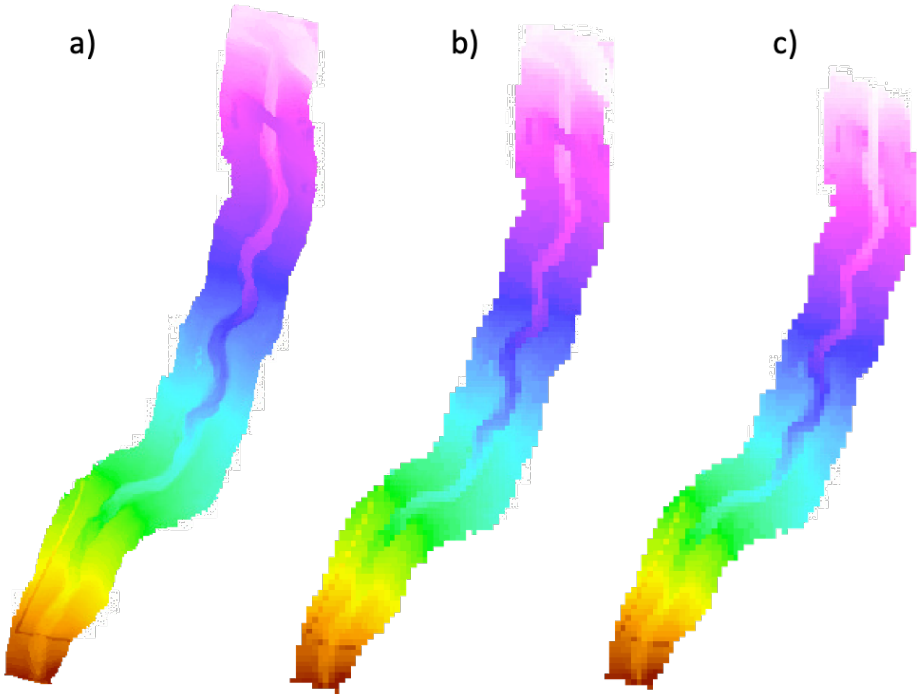


Figure 96. Image showing the results of the processing steps applied to the DEM of Ben Gill prior to merging this channel with the main modelling DEM.

Processing both the original 1m DEM and Ben Gill DEM to alter the resolution to 2m and later merge the two files resulted in some loss of detail. It was therefore necessary to manually edit some aspects of the final DEM to better replicate the physical conditions of the site. The

cells representing the weir, were identified and changed to 112.31m AOD. The Ben Gill channel was deepened to prevent frequent out of bank flows which occurred on initial runs and to better reflect the real depth of the channel in relation to the surrounding terrain at the date the DEM was created.

The lakebed was edited to remove the manmade channels to prevent their influence on channel evolution under different weir removal scenarios (Figure 97). The channels were infilled following the natural gradient of the lakebed by interpolating the data from either side of the channels. Infrastructure was represented by editing the existing DEM to contain the weir, fish pass and walls at the lake outlet.

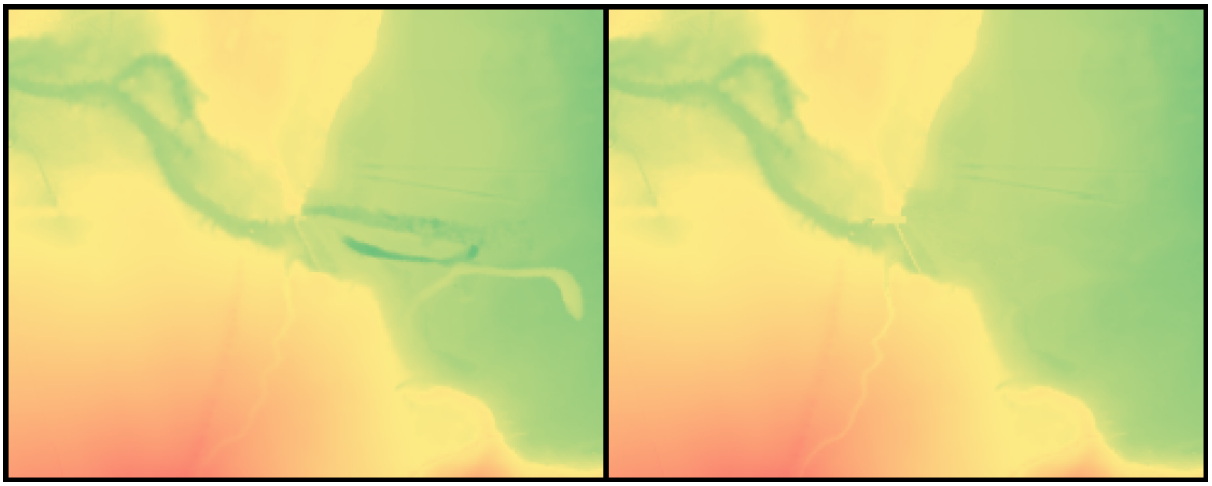


Figure 97. Image showing the 2m DEM including Ben Gill and current lakebed (left) and the edited DEM with smoothed lakebed and edited Ben Gill and Infrastructure (right).

The weir, fish pass and walls around the River Ehen and Ben Gill confluence were represented using a bedrock layer in the CEASER model. The bedrock parameter uses data in the form of a DEM to select areas of the modelling domain where erosion cannot take place (Figure 98). This was achieved by manually editing a DEM to remove all data except for key areas where infrastructure existed. In other areas of the modelling domain, the bedrock layer was set to a depth below the ground surface corresponding to the estimated depth of bedrock based on field observations.

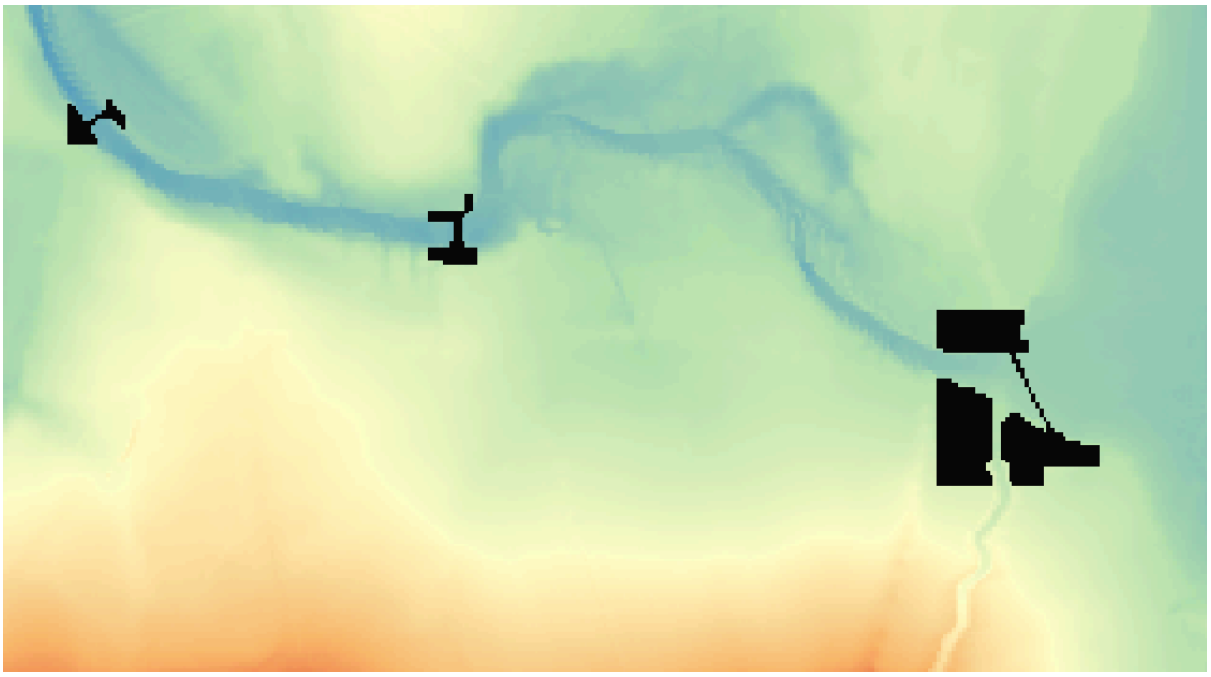


Figure 98. Image showing infrastructure locations at the lake outlet and further downstream. Areas in black represent infrastructure. CAESAR-Lisflood reads these areas as bedrock that cannot be eroded.

Based on the evidence from the GPR and observations from field visits, the bedrock layer was applied only to the southern part of the modelling domain. This was set a 0.1 m below the surface for the steeper part of Ben Gill (Figure 99a) and then at 0.2m below surface elevation for the upper plateau and source of Ben Gill (Figure 99b). This was consistent with the soil depths used in the SHETRAN hydrological model. For the lower alluvial fan and main river corridor no bedrock depth was set. The raised border (Figure 100) around the eastern and northern edge of the lake was also set as bedrock layer at the same elevation as the original DEM (116mAOD) to prevent any erosion of these cells.



Figure 99. Evidence from Ben Gill channel and upper plateau used to inform bedrock layer in CAESAR-Lisflood. Photos taken by Joel Blackburn Aberdeen University

Modelling Domain

A modelling domain of 1,050 m by 1,300 m used for the final modelling scenarios. This allowed sufficient area of the lake and lengths of Ben Gill and the River Ehen to be incorporated. Despites its small drainage area and ephemeral nature, Ben Gill is major source of sediment to the upper River Ehen and it was therefore necessary to include its full length in the modelling domain (Figure 100).

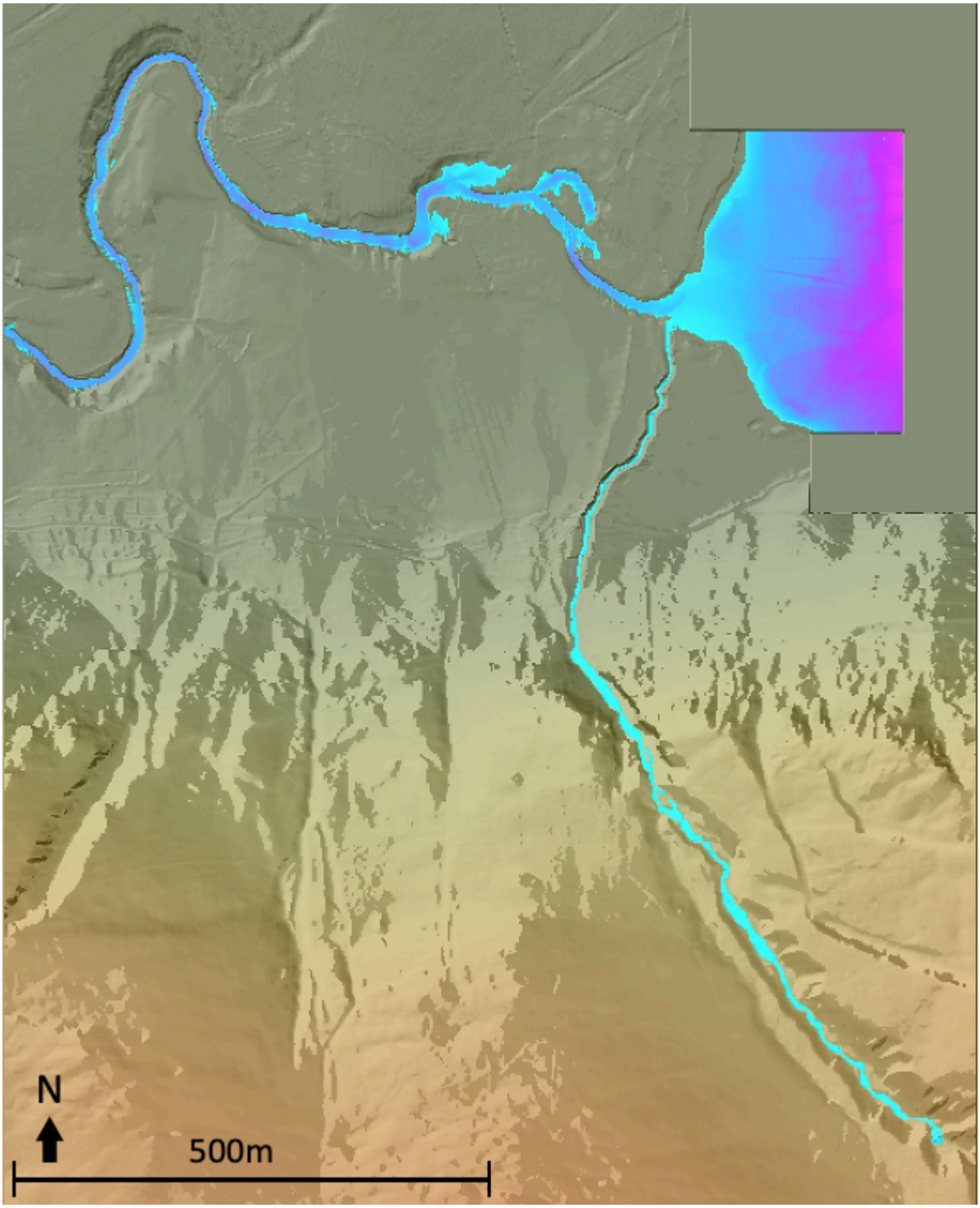


Figure 100. Full extent of the modelling domain used in the final model runs. Dimensions 1,050m by 1,300m. Bottom left corner coordinates x = 308150 and y = 514300. Lake border raised to allow model to correctly drain from right to left and to speed up model runs.

To increase model speed, only the western end of the lake was included in the modelling domain as waiting for the lake to fill for each model run proved to be time consuming in the initial model runs. Furthermore, evidence form the initial model runs showed that the hydrological process in this area of the lake had no impact on the geomorphic evolution of the lake outlet. The edge of the modelling domain to the East and North needed to be raised to prevent water from the lake draining out of the DEM instead of filling the lake. The DEM was edited so that a raised border set to 116mAOD enclosed the edges of the lake and the width was extended to reduce the number of cells containing water to further increase the mode run speed as no hydraulic calculations would need to be performed in these model cells.

Sediment Data

The results from the sediment coring field data did no provide direct model inputs due to the problems described in section 5.3. However, the exploratory surveys did show that the lakebed upstream of the weir was predominantly comprised of coarse sediment with the exception of the pockets of fine sediment within the manmade channels. Information on sediment grainsize was therefore obtained from a combination of recent and ongoing complementary studies of the area upstream and downstream of Ennerdale Weir (Table 22). These form part of the compensatory measures package described Chapter 1, that are designed to ensure no further deterioration to the site whilst impoundment and abstraction continue. The recently reconnected (2014) Ben Gill continues to be monitored to assess the type and rate of sediment supply entering the Upper Ehen. This work is being conducted at Aberdeen University and is a continuation of studies carried out in the by Marteau (2017) as part of the compensatory measure package for the River Ehen. The latest data was provided by Joel Blackburn (pers comm.) of Aberdeen University and is the average of five Wolman Pebble counts conducted in the Upper Ehen between March 2018 and October 2019.

Whilst the samples from the Wolman pebble counts are limited in quantity and taken over a small timeframe, the ongoing nature of the study means that additional data that continues to be collected can be used to update the sediment information in the model if required. The approach also provides a standardised methodology that could be replicated in other areas of the site if required allowing for comparable data to be used in different locations. This could then be used for further refinement of the model input data.

However, although the current location of Ben Gill is downstream of the weir, the debris cone or alluvial fan it has created extends both to areas both upstream and downstream of the weir. In addition to the evidence from the sediment surveys, the data gathered from the ongoing

studies of Ben Gill provided representative information of the sediment grain sizes in the area directly upstream of the weir.

The CAESER-Lisflood model is capable of handling up to nine grain size classes however data on finer grains were not available from the pebble counts. To account for the fraction of finer sediment a further two grain size classes were added to allow for comparative model runs to be made. These were 2mm (0.002m) and 0.5mm (0.0005m) to represent sand and silts respectively. The values for these were based on research carried out by Marteau *et al*, (2018) which found that fine sediments represent a relatively small proportion of total sediment inputs into the River Ehen. The two new sediment classes represented 10% of the total grainsize-distribution. This was calculated as 67% sand and 33% finer material (Marteau *et al.*, 2018).

Table 22. Average grain size classes and distribution from five Wolman pebble counts between March 2018 – October 2019. Data provided by Joel Blackburn, Aberdeen University.

Clast Size	Proportion	
	Option 1	Option 2
Fine 0-0.5mm	0	0.033
Sand 0.5-2mm	0	0.067
Fine gravel 2-8mm	0.016	0.014
Medium gravel 8-16mm	0.077	0.069
Coarse gravel 16-32mm	0.241	0.217
Very coars gravel 32-64mm	0.387	0.348
Small cobble 64-90mm	0.174	0.157
Medium cobble 90-128mm	0.084	0.076
Large cobble 128-180mm	0.02	0.018
Total	**1**	**1**

Initial model runs were carried out using both sets of sediment data however for the final runs only the second set of data was used. This was to prevent excessive multiplication of model runs. Initial comparison runs showed that the inclusion of all 9 grain sizes had little impact on model run speed, rates of erosion or volumes of sediment transported from the modelling domain.

Inflow Data

The model was set up to replicate the system in its current state as accurately as possible. To achieve this, flow into the River Ehen was entered from three sources within the modelling

domain; 1) over spilling the weir when the lake is sufficiently high 2) through the fish pass downstream of the weir, and 3) ephemeral flows from Ben Gill. The existence of the fish pass means that the Weir at Ennerdale Water doesn't always overtop

An initial attempt was made to edit the DEM to replicate the fish pass as a physical outlet next to the weir. This proved difficult as the fish pass has a width of 1.2 m and the model was run at 2m resolution. To overcome this, an additional discharge point was created downstream of the weir. This was positioned on an area of the bedrock DEM layer so that scour and erosion did not occur at the discharge point but instead was able to occur in the river channel in an attempt to better replicate the current situation.

Flow data was obtained from observed NRFA data from the Bleach Green gauging weir (Station No. 74003). This was then divided into two parts, lake inflows and fish pass flows.

A fixed flow of 1 m^3/s was given for the fish pass as this was the approximate minimum observed flow during the initial observation period. Flows through the fish pass do vary and reach values in excess of 1 m^3/s and as low as 0.69 m^3/s however for the purposes of this modelling study a fixed value was deemed adequate as limited sediment movement is taking place at low flows. The flow through the fish pass was subtracted from the modelled flows from the lake meaning that periods of no flow over the weir were obtained.

Flow for Ben Gill was obtained by downloading the discharge time series from the corresponding river Element on the SHETRAN river network (Figure 101). Ben Gill is an ephemeral stream with a complex combination of surface and subterranean flows. Initial observations from ongoing research of the site were used to help inform the input data for this part of the model.

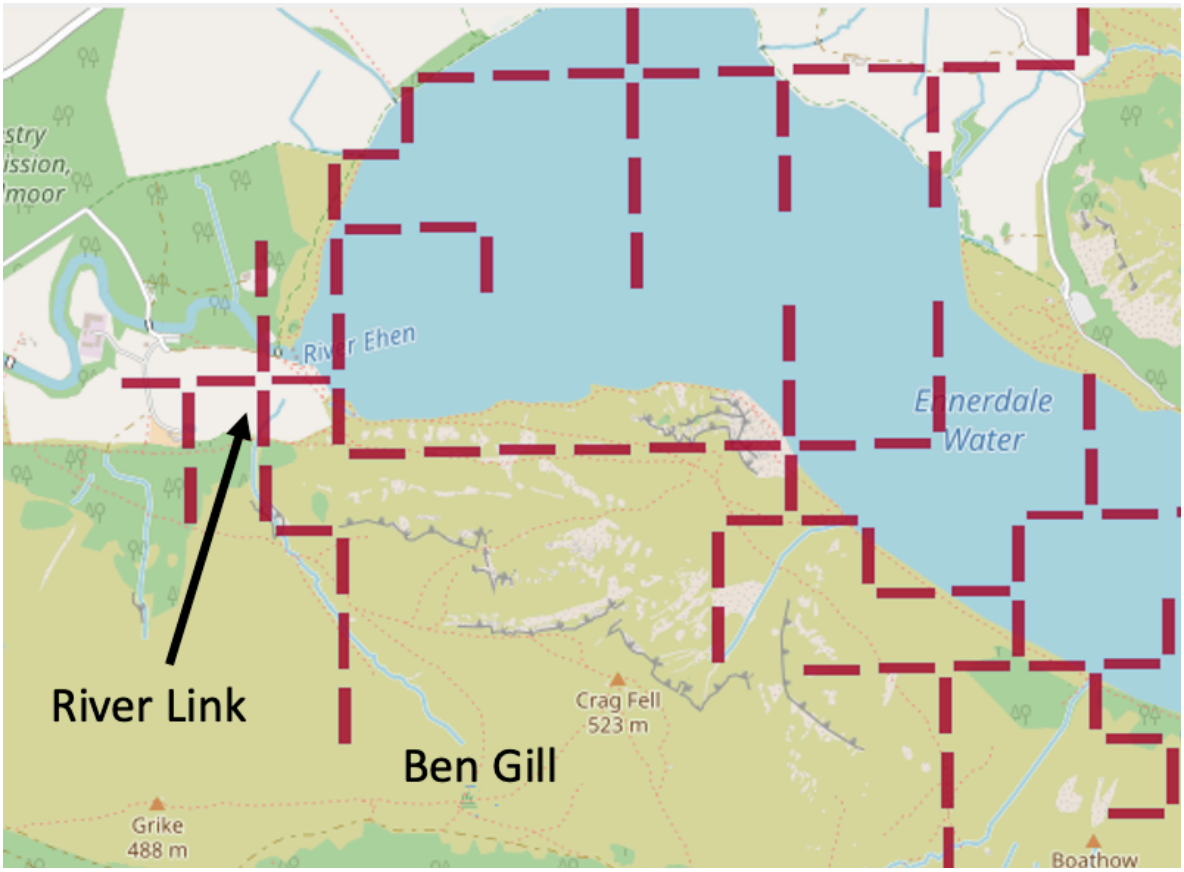


Figure 101. SHETRAN representation of the Upper Ehen river network showing location of Ben Gill and river link used for obtaining the discharge time series for Ben Gill.

In its upper reaches Ben Gill flows consistently throughout the year however due to the size of the sub catchment and underlying physical properties of the alluvial fan, flows are rarely sufficient for surface water to reach the River Ehen. On average Ben Gill flows into the River around 20% of the time (pers comm.), equivalent to 60-70 days a year. The final SHETRAN model of the entire Upper Ehen catchment was however unable to accurately represent this localised behaviour instead producing a continuous time series with extremely low flows. Calibrating or separately modelling this sub catchment was also beyond the scope of this study.

To represent the ephemeral nature of Ben Gill, a minimum threshold flow was established to ensure that flow into the River Ehen only occurred around 20 % of the time. Based on field observations a minimum flow threshold of 0.05m^3/s was established. All flows produced by the SHETRAN model below this threshold were removed. This created an intermittent flow time series for Ben Gill with an annual average of 62 flow events per year (Figure 102).

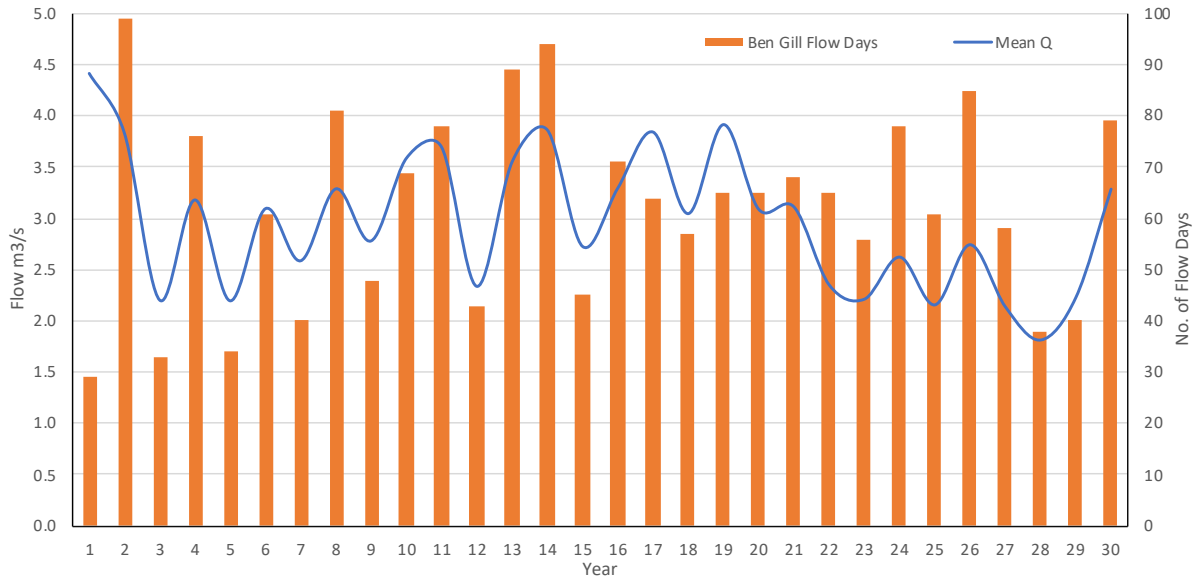


Figure 102. Times series used for the 30-year modelling period showing mean annual flow for the River Ehen and number of days per year flow occurred in Ben Gill.

Sensitivity Analysis

Sediment testing

Runs were initially carried out on a reduced domain size using a modified DEM to replicate a no weir scenario. The model was also run in an un-primed state, meaning that no sediment distribution within the modelling domain had occurred. When a model run is initiated the DEM essentially serves as a blank canvas and the model requires a certain period to distribute the sediments throughout the river network. The length of model run required to achieve this depends on a number of factors including hydrology and physical characteristics of the site. Sediment transport rates are higher than would normally be excepted whilst this part of the modelling procedure. Models were also run without vegetation activated during this analysis. As a result of these settings, erosion rates were extremely high and not representative of observed or expected values for the Upper Ehen. The information did however provide a direct comparison of the influence of flow and chosen sediment transport methods.

Runs were set for 200-day period with an initial inflow of 15 m^3/s on the first day to allow for the lake to fill more rapidly. This was then followed by 200 days at a fixed discharge. Runs were carried out using 1, 2, 3, 4, 5, 10, and 15 m^3/s. A flow of 0.5m^3/s was applied to Ben Gill every 20 days which increased to 0.8 m^3/s on days 100 and 200 of the runs. The sediment model was set to the Wilcock and Crowe method for each run. The same procedure was then duplicated using the Einstein sediment model.

Sediment data for the site was unavailable at the initial stage of model testing so values were used from a case study site provided in the model download. Lateral erosion was not turned on at this stage in the model runs. A further test of the different sediment transport calculation method was conducted on a fully primed model. In this analysis, three identical runs were compared using the same observed hydrological input data and the same initial conditions. The only change between the three runs was the use of either Wilcox-Crowe, Einstein and Meyer. After running the models for 5 years, the results of the sediment transport were compared.

Final Model Setup

The model was run in reach mode, not catchment mode. The reason for this was to allow for a more detailed study of the lake outlet and downstream areas. SHETRAN had also provided data on catchment hydrology so this was not needed. Another key consideration was model run time which would have greatly increased in catchment mode and limited the repeatability of modelling runs.

An initial period of 20 years to model the geomorphic changes of the lake outflow was used. However this was then increased to 30 years based on the methodology outlined in Howard *et al.*, (2017) in which the CAESER-Lisflood model is used to estimate geomorphic responses to multiple weir removals in the Derwent Valley. CAESAR-Lisflood has a very large number of applications and vast array of parameters and settings. Configuring the model for the final runs was an iterative process that required a systematic approach to parameter adjustment. Consultations were also carried out with the model developer to receive additional expert advice on the choice of setting and model setup. The final model runs were carried out using the Wilcock-Crowe method. This is one of the most widely used equations for estimating the rates of bed load transport in gravel bed rivers (Gaeuman *et al.*, 2009). The method has also been used more frequently in previous studies analysing channel response to weir removal with CEASER-Lisflood.

Initial Outline Profile Designs

A selection of twelve outlet profiles were chosen based on the existing topography of the lakebed and surrounding land. The influence of the recently reconnected Ben Gill channel was evident from field observations and the initial model runs. This also helped to inform the initial profile design and where to locate the lowest point of the outlet channel in relation to the current weir. Channel outlets of three different widths were proposed to assess the impact of rates of channel incision and lateral migration. Each design was the replicated at

increments of 0.1m elevation from an initial elevation of 111.4 mAOD. This was to assess the impact of starting elevation and channel slope on channel evolution and sediment transport rates. This resulted in 12 outlet designs and 12 model runs of 30 years.

Table 23. Outline profile dimensions for final model runs (denotes replicated model runs with the removal of two additional downstream barriers)*

Dimension	Outlet Profile Design											
	1	1a	1b*	1c	2	2a	2b*	2c	3	3a	3b*	3c
Width (m)	14	14	14	14	18	18	18	18	26	26	26	26
Elevation (mAOD)	111.4	111.5	111.6	111.7	111.4	111.5	111.6	111.7	111.4	111.5	111.6	111.7

The cells that were modified in the different outlet DEMs are shown in Figure 103 and the cross-section profiles of the different initial outlets are shown in Figure 104.

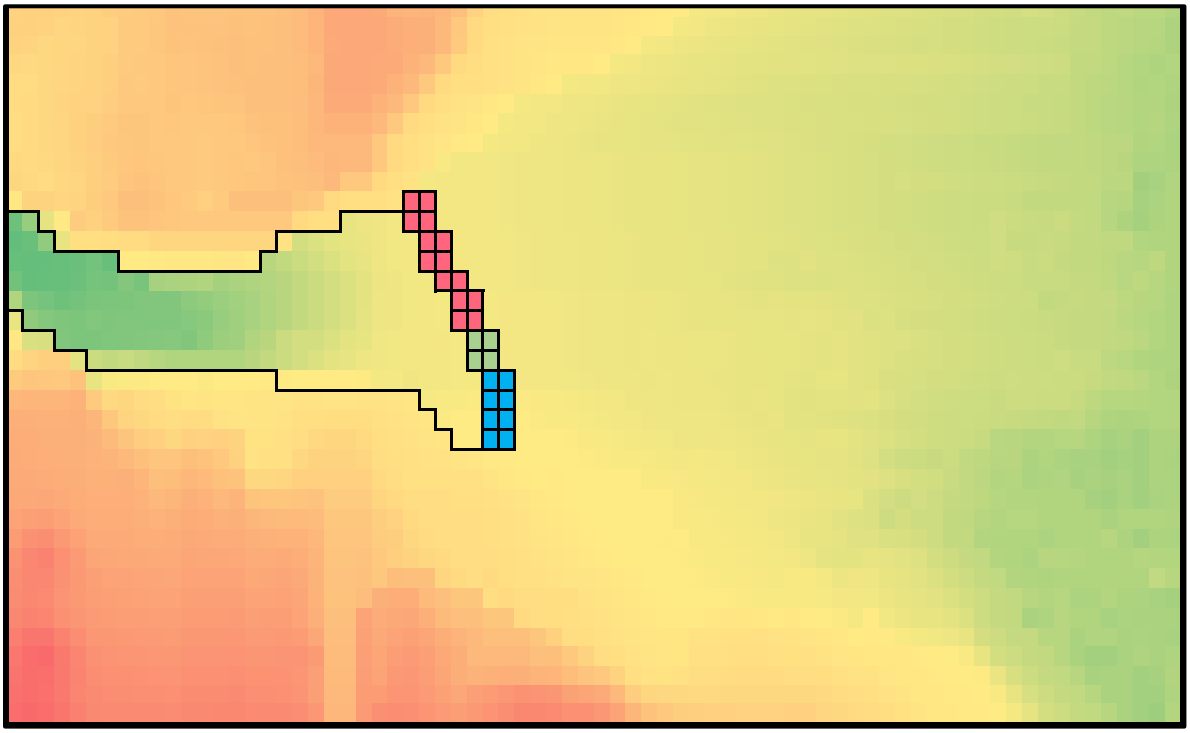


Figure 103. Lake Outlet DEM with cells that were modified for different outlet profiles shown in red (profiles 1-1c), blue (profiles 2-2c) and green (profiles 3-3c).

A further three models from the initial twelve were selected to assess the impact of removing two additional in-steam barriers further down the main river Ehen Channel. The first of these occurs approximately 450 metres downstream and is an old bleaching weir believed to have been present since the 1870s (Alvarez-Codesal and Sweeting, 2015) and no longer in use but never fully removed. The second is the NRFA flow gauging weir (Station No. 74003) which is located 800m downstream of Ennerdale Water. Alternative approaches to flow gauging are currently being considered and no decision has been taken as to whether this weir will be removed but the inclusion of these two weirs was considered an important element of the

study during the stakeholder consultations. This resulted in a final set of 15 model runs which are analysed in section 5.9.

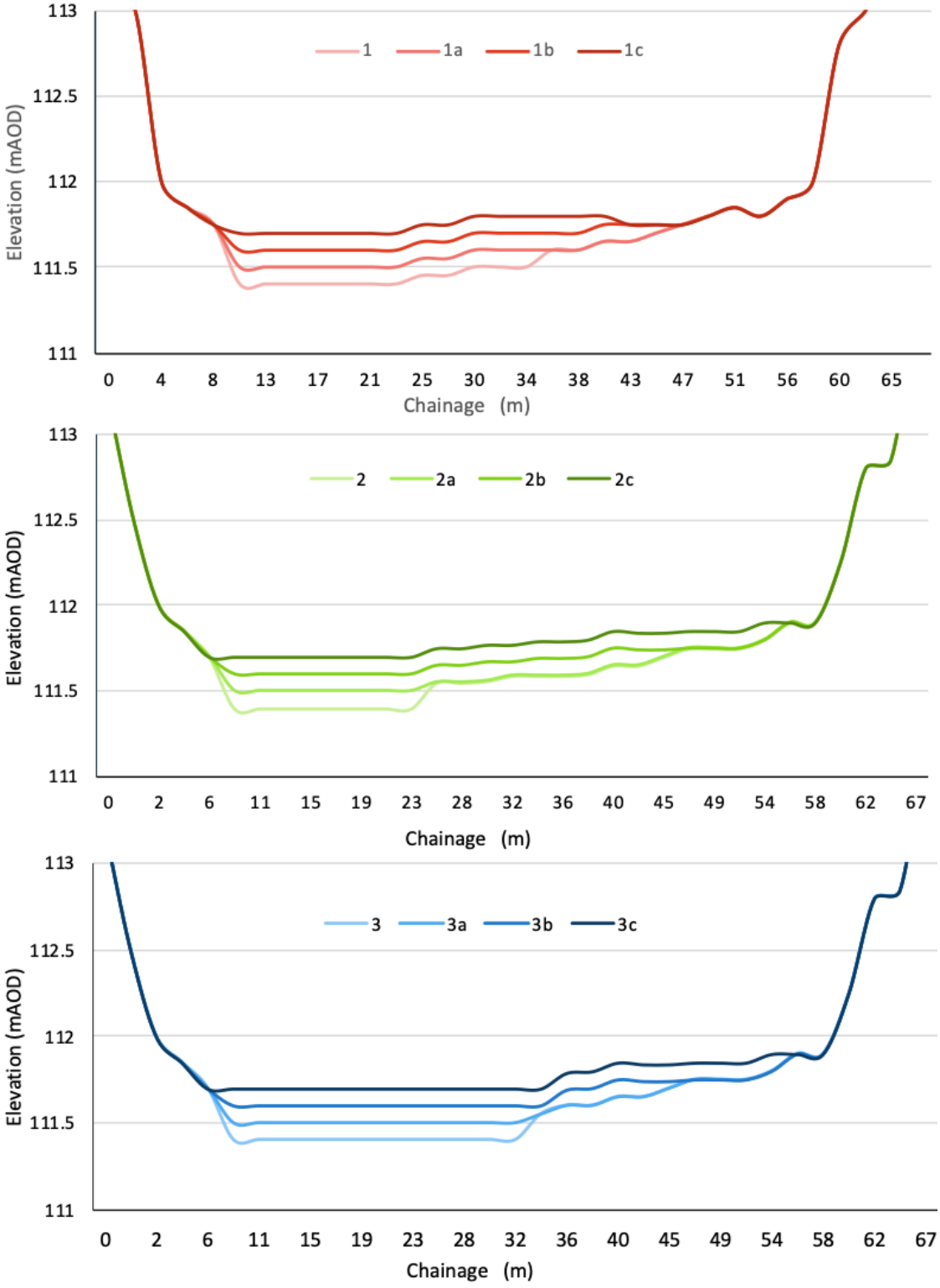


Figure 104. Initial lake outlet cross-sections for 30-year model runs. Designs were based on previous hydrological modelling of the lake outlet and elevations of upstream and downstream regions.

5.7 Geomorphic Modelling Results

A total of 15 models were run, each for a duration of 30 years. The first part of the results section looks at the 12 model runs in which only the upper weir was removed (Figure 105). The second part compares the results from the three model runs in which additional downstream barriers were removed with the corresponding upper weir removal runs i.e. run 1b with 1b*.

The first analysis was to quantify the total volume of sediment leaving the modelling domain during the 30-year period. The average volume across all 12 model runs was 708 m^3 of sediment. This varied between a minimum value of 659 m^3 in Profile 3 to a maximum of 751 m^3 in Profile 2a.

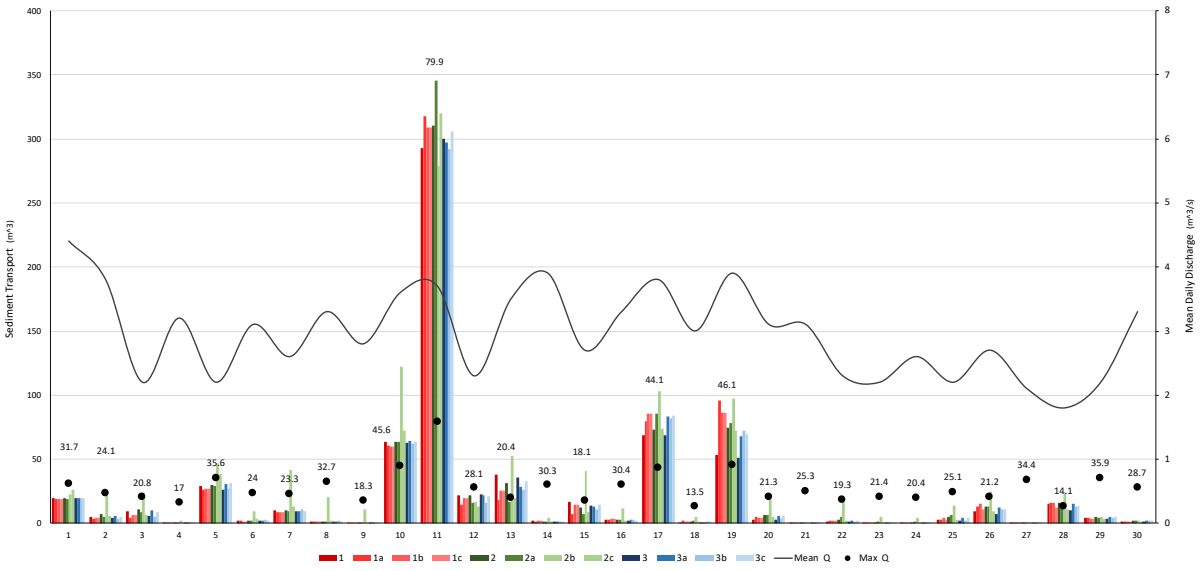


Figure 105. Annual sediment loads under different outlet designs. Number above columns indicates maximum observed discharge for that year in m^3/s.

The average annual sediment load across all model runs was 24 m^3 however there was a wide range in annual totals. During the 30-year period the largest annual amount of sediment moved through the modelling domain was 345 m^3 from model run 2a in year 11. The smallest annual total volume was 0.01 m^3 in model run 2b.

The average annual sediment loads were compared to the corresponding mean annual and annual maximum discharges. The results showed that maximum flow values play a more significant role on sediment transport than mean discharge. An r^2 value of 0.75 was observed when comparing sediment transport and maxim discharge in contrast to an r^2 value of just 0.16 for mean daily discharge versus sediment transport (Figure 106).

The maximum observed annual sediment load was observed in year 11 of the model run. This data was observed flow data from 2009-2010 when severe winter storms impacted large parts of the UK. The average sediment yield for this year across 12 model runs was 309 m^3. The minimum recorded sediment load occurred in year 27 of the model runs. An average of 0.25 m^3 of sediment was transported during this hydrological year which had a mean daily flow of just 2.1 m^3/s and a max flow of 34 m^3/s.

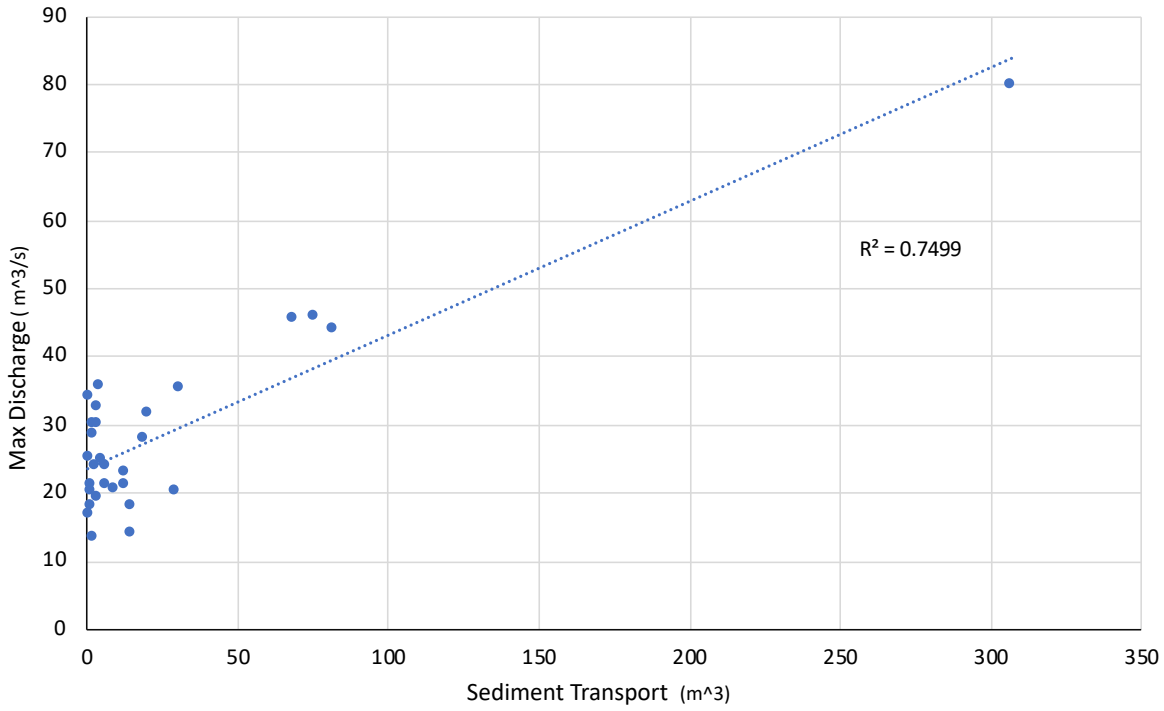


Figure 106. Mean annual sediment transport and maximum discharges for 30-year model runs

The rate and extent of channel evolution at the lake outlet was also compared between the 12 model runs. The longitudinal profile was measured to assess how far upstream the vertical headcut had migrated and to assess changes to channel width and elevation (Figure 107).

The average headcut migration was approximately 40m over the thirty-year period, ranging from the lowest distance of approximately 34m to 45m at the upper limit. Channel migration was not uniform, with the largest upstream migration in the first 6 months of the modelling period with channels migrating a distance of over 10m upstream.

In addition to upstream migration a vertical incision, the channel widths also reduced during the initial stages of the modelling runs. This was later followed by a widening at the upper most section of the channel in some cases when the channel had fully adjusted the longitudinal slope and reached the limits of its downward incision. This widening is shown in Figure 108.

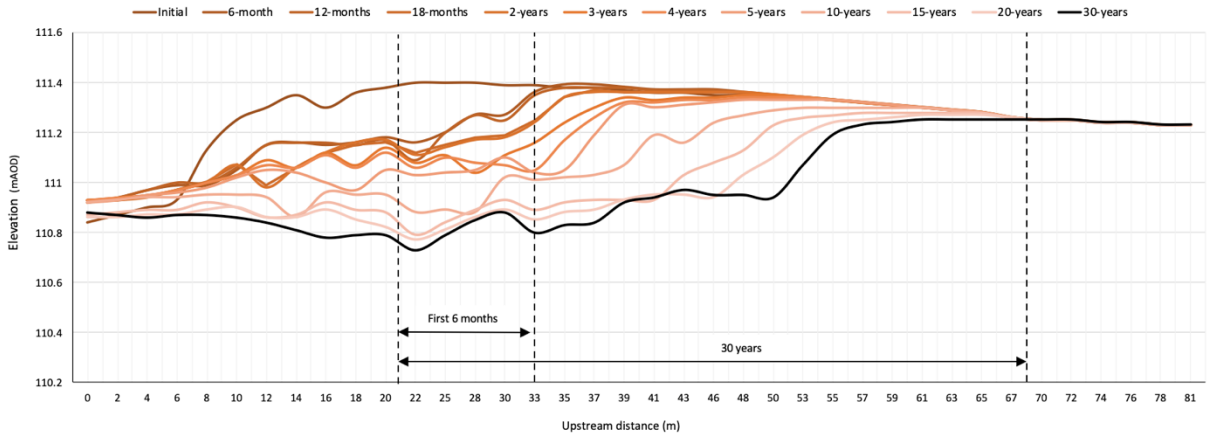


Figure 107. Results from model run 1 showing longitudinal profile changes over time showing distance of upstream migration and elevation adjustments.

The minimum vertical incision was observed in the outlet profiles which had the lowest initial elevations. In these cases, the elevation change from the initial headcut to the final headcut was 0.15m. In contrast the profile with initial elevations of 111.7mAOD showed vertical incision into the lakebed of up to 0.35m. The lowest that the lakebed elevation incised was 111.25mAOD.

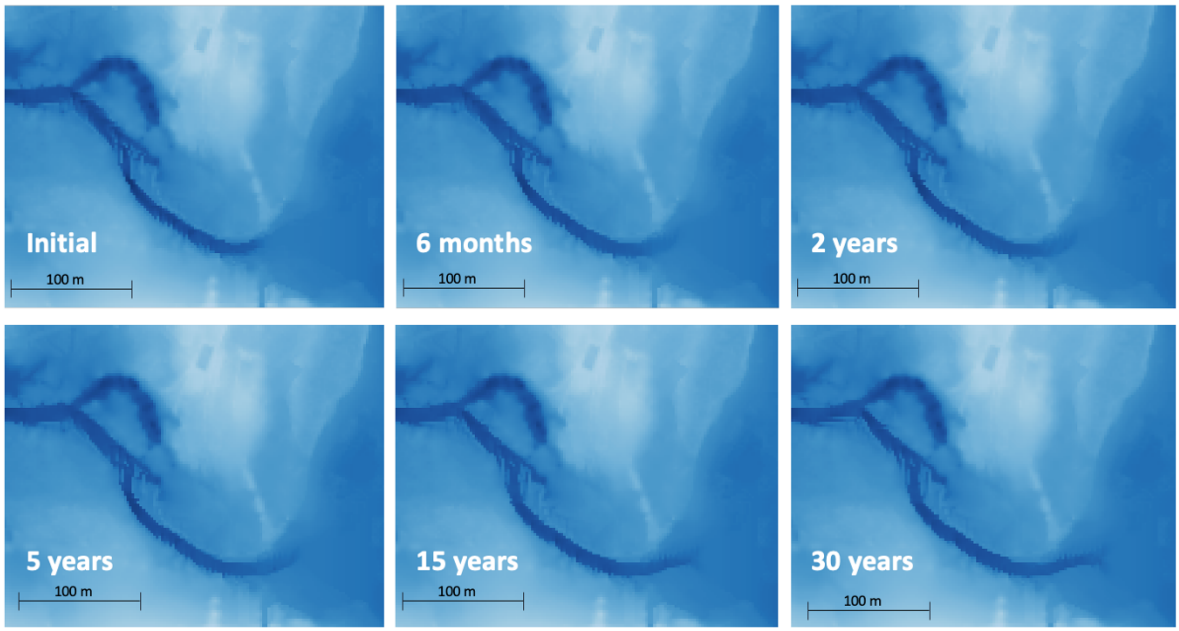


Figure 108. Results from model run 1 showing changes to lake outlet at selected timesteps over the 30-year model run. Darker colours represent lower elevations. Ennerdale Water is to the right in each image. Flow is moving downstream from right to left.

The final DEMs generated from all 30-year model runs are shown in Figure 109. This shows the range of variation in the extent of upstream erosion and lateral movement of the lake outlet. Additionally, there is also considerable variation on the width of the main channel that forms upstream of where the weir was previously located as shown by comparing Profile 1c

and 2c. A number of the profiles appear to show that the channel divides into two at the most upstream point where the lake drains into the River Ehen.

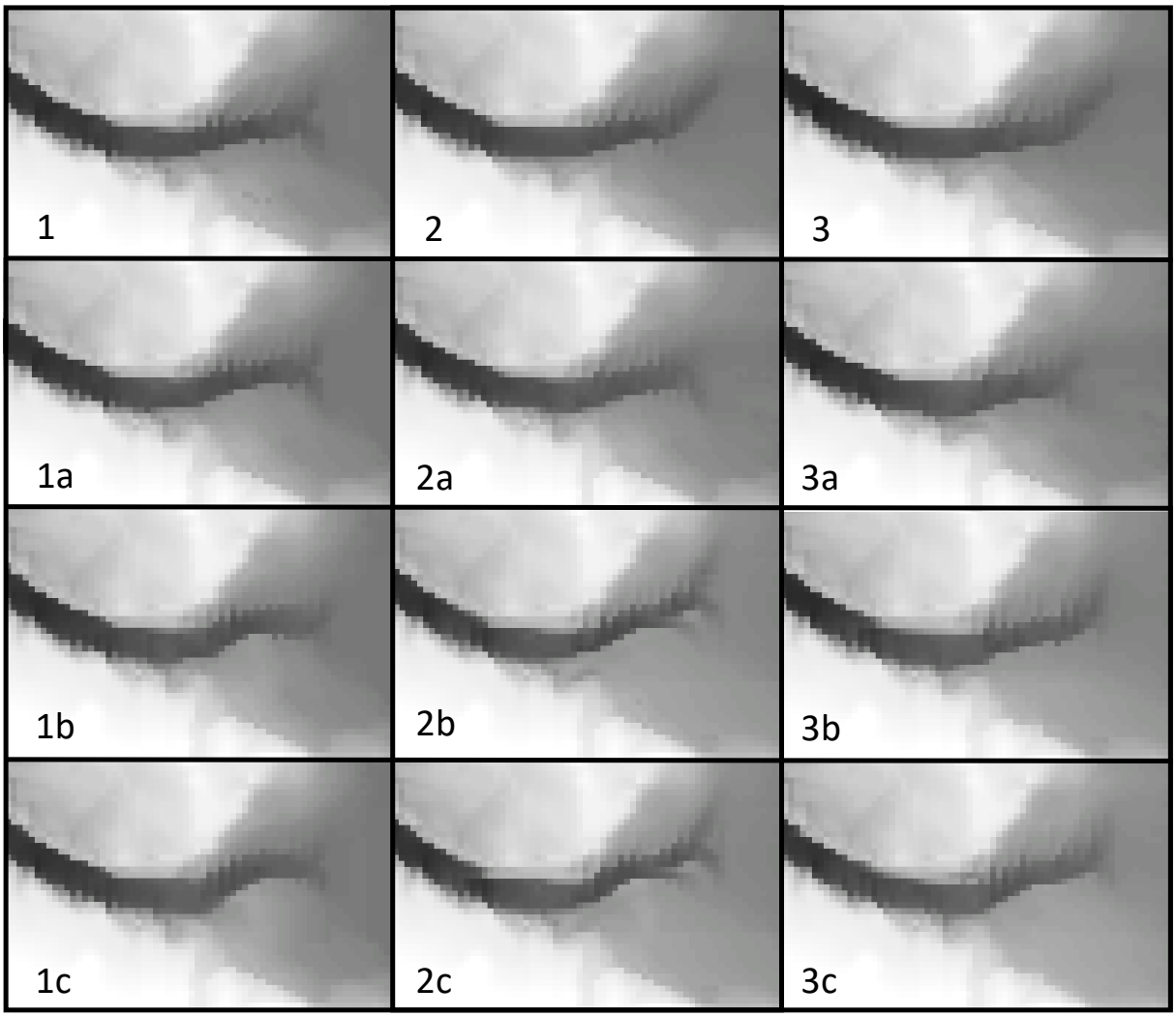


Figure 109. DEMs for all model runs after 30 years. Dimensions of individual frames are 130m x 85m. Darker colours represent lower elevations. Ennerdale water is shown to the right and flow downstream is from right to left in each DEM.

A further model run was carried out using the DEM generated from model run 2c after 30 years. This was then repeated for an additional 30 years using the same hydrological data from the original run. The purpose of this was to assess whether the original runs had captured the full extent of the upstream and lateral channel migration. Model run 2c was on the DEMs that showed the formation of two channels at the lake head and it was therefore selected to show how this formation might evolve over a longer timeframe.

The results from the DEM (Figure 110) show that limited additional upstream migration occurs after the first 30 years. For the single model run the channel advanced approximately 6m further into the lake. The main changes that occurred were a widening of the channel at the lake head and further lateral erosion and deposition on the bend in the channel

downstream of the current weir location. The total sediment transport volume for the second 30-year run was 727m^3 compared to 719m^3 in the first 30 years. This suggests that bed material will continue to be transported downstream, provided sufficient sediment is available, and this is not solely restricted to the initial period of adjustment after weir removal.

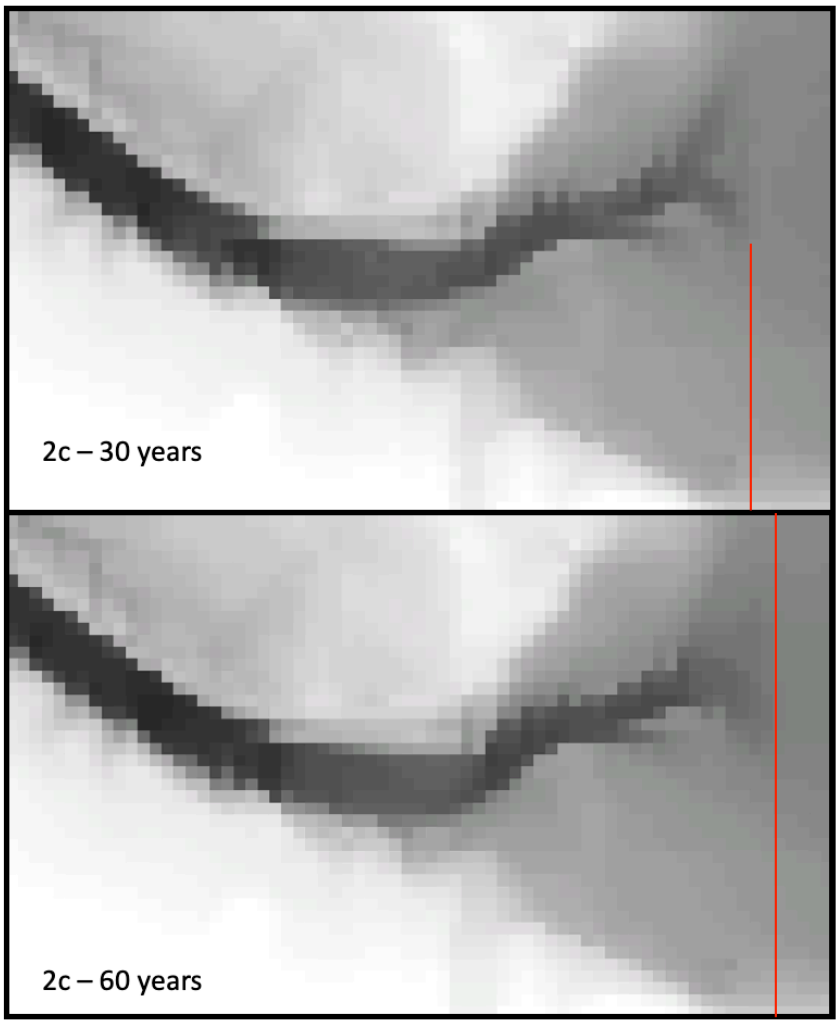


Figure 110. Comparison of lake outlet using initial profile 2c after 30 years (top) and 60 years (bottom). Red lines indicate the extent of upstream channel migration. Pixels represent the 2m x 2m model resolution.

The DEM suggests that the formation of the two smaller side channels would eventually be replaced by a wider single channel that has flatter profile and a more funnel like shape at the lake outlet. Another feature of the CEASER-Lisflood model is the ability to export images directly during the model run or at given timesteps specified in the model run setup. These images are georeferenced and can then be displayed in Google Earth, which can enhance the communication of the modelling results to wider stake holders. Figure 111 shows the water depth for Profile 3c in Ennerdale Water and the River Ehen after a 30-years of evolution. The corresponding inflow rate into the lake was 4.1 m^3/s and 0.3 m^3/s from Ben Gill. At this flow rate the lake stage of Ennerdale Water is estimated to be around 111.48mAOD.

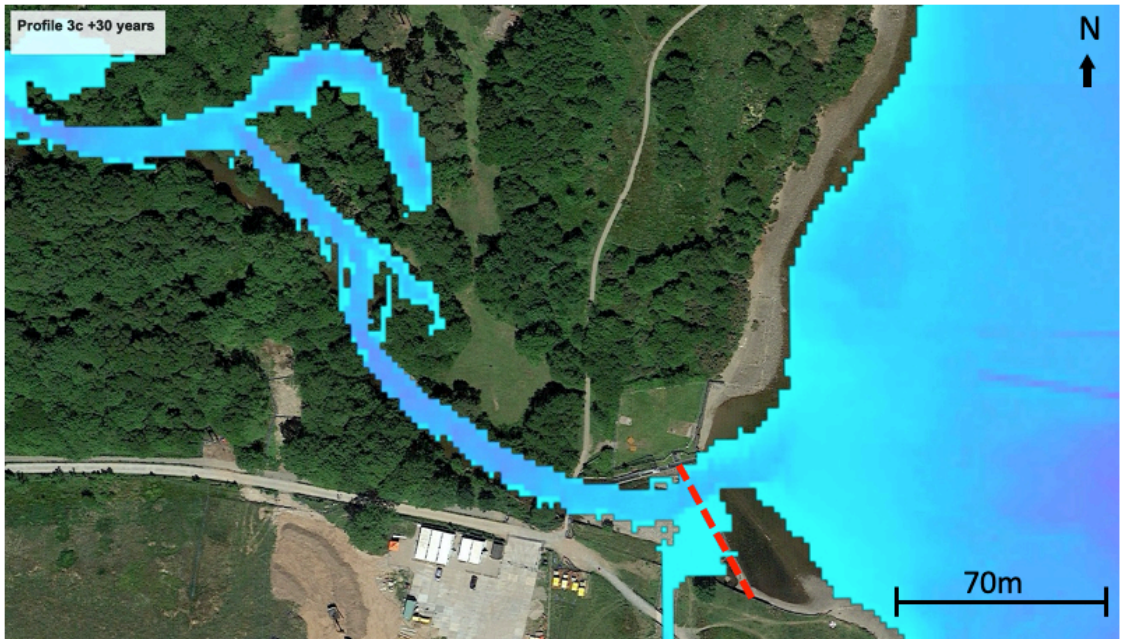


Figure 111. Water depth of lake and river Ehen at Ennerdale Water outlet superimposed onto Google Earth image from 2018. Lighter blue shades represent shallower water. Dashed red line shows current weir location.

A comparison was made between the volumes of sediment leaving the modelling domain with the removal of a single weir and the removal of two additional downstream weirs (Figure 112).

The removal of two additional in-stream barriers resulted in the mobilisation and transport of larger volumes of sediment. The greatest change was observed when comparing the two runs using outlet profile 2b. The total sediment transport with single weir removal was 696 m^3 over the 30-year period. This increased to 1,314 m^3 when the additional downstream weirs were removed. The average across the three model runs (1b, 2b and 3c) was 699 m^3 with a single weir removed compared to 250 m^3 with the removal of two additional downstream barriers.

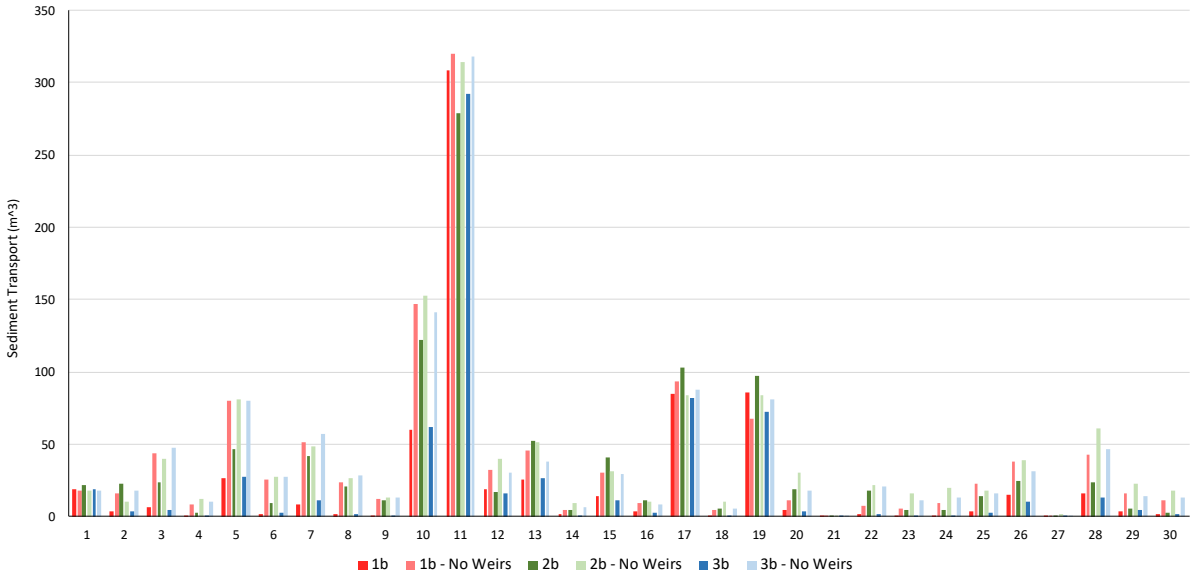


Figure 112. Comparison of annual sediment transport between single weir removal and multiple weir removal (labelled no weirs) model runs.

5.8 Chapter Summary

The aim of this chapter was to characterise the lake sediments proximal to the weir and identify any evidence of historic outlet channels. The results provide useful additional information for further understanding the site and provide critical information on the initial conditions for the geomorphic modelling. A further aim of this chapter was to estimate rates of channel evolution and estimate volumes of sediment transport in response to weir removal. In addition to this, lake outlet profiles were obtained to allow detailed hydrological analysis in Chapter 6.

Multiple engineering works have altered the morphology of the lake outlet which has impacted the availability of historic data on the location and form of previous natural lake outlets. However, evidence of paleochannels gathered from the GPR data suggests that the lake outlet has migrated, and the current location may be subject to change if left unconstrained by infrastructure. The modelling results suggest that lateral movement of the lake outflow in the first 30 years after removal may be as much as 26m.

Additional information was gathered on the physical properties of the majority of the sediment behind the weir and the chemical composition of sediment in the manmade channels from the 1976 drought. The GPR radar data has shown that near surface bedrock did not need to be included in the geomorphic model in the main areas of fluvial activity in the model. Furthermore, the evidence from the sediment collection surveys has shown fine sediment accumulation is limited in the shallow areas behind the weir. The exception being the two

man-made channels that have deposits of fine sediment and organic matter in depths of around 15-20cm.

Whilst no finalised design or methodology has been proposed for the lake outlet in the event of weir removal, some form of engineering works will be necessary to prevent erosion occurring in the existing channels. Failure to carry out this work is likely to result in an initial large pulse of fine organic sediments downstream and preferential flow pathway that would otherwise not exist.

The exploratory analysis of the lakebed sediments has also identified pockets of fine sediments but also highlighted that the area immediately upstream of the weir is predominantly composed of coarse substrates. This has provided corroboratory evidence to support existing research that shows sediment in the upper reaches of the River Ehen is predominantly larger material with a very small percentage of sand or finer material.

Time and resource constraints limited the quantity of sediment data collected for this study. Prior to weir removal, any further studies into the substrate conditions upstream of the weir, would benefit from additional sampling locations to increase the spatial coverage of sediment analysis. Additionally, if financial and time are available deeper core samples using mechanical rather than manual tools would facilitate a more rapid collection of samples and should provide a clearer picture of the subsurface conditions than currently exists. This is discussed further in section 7.4.1.

The modelling study has shown how the lake outlet might be expected to change over time in response to a range of initial physical conditions. However, as with any modelling study it is necessary to communicate any limitations. Geomorphic systems are inherently complex and modelling them is thus filled with uncertainty. A major challenge in the present study was the interaction between Ben Gill and its associated alluvial fan and the outlet of Ennerdale Water and the Upper Ehen. These elements all have different physical characteristics and trying to model them collectively with a single set of model parameters means compromises must be made. For this study, the behaviour of Ben Gill was constrained and whether it will retain its current position over the next thirty years is uncertain. Since its reconnection in 2014 the channel has shown signs of stabilisation in both physical appearance and monitoring of downstream sediment deposition. Although only a small ephemeral stream its influence on the Upper Ehen is now and in the event of weir removal likely to be substantial as it is a major source of sediment supply.

The upstream migration of the channel is consistent with conceptual models and observations from within the literature. Due to the relatively low level of the weir at Ennerdale and the coarse substrate, rates of upstream migration are relatively moderate compared to examples of reservoirs infilled with fine unconsolidated sediments (Doyle *et al.*, 2003b; MacBroom, 2005) . The results provide essential information for the future design of a 'natural' outflow form that may need to be engineered once the weir has been removed. This could potentially prevent the transport of large volumes of sediment being washed downstream immediately after weir removal and allow for a more gradual supply of sediment and less abrupt reconnection.

Chapter 6 - Fluvial Geomorphological Feedback and Lake Outlet Profiles

The final modelling chapter builds on the previous work to examine how the outflow form influences the downstream hydrology. The findings from Chapters 3,4 and 5 are all tied together to produce an easy-to-use modelling tool that allows the comparison of hydrological regimes under a range of physical outflows. The application of this modelling tool is demonstrated, and the results are set in the context of flow regime characteristics from the UK-wide study set out in Chapter 2. Additional emphasis is given to low flow metrics as these are of ecological concern to the freshwater ecology of the Upper Ehen and for the future management of the site.

6.1 Background

Previous studies by the consultancy MWH have been conducted using the Flood Modeller Pro (formally ISIS 1D). These studies were based on a limited number of outflow designs and assumed a static lake outlet form. According to the authors of the original study errors in the lake level and storage relationship were estimated to be in the region of 5-10% (MWH 2013).

The MWH study was also carried out prior to the reconnection of Ben Gill which, based on continuing hydrological and physical studies of the lake outflow, has already had a measurable influence on the hydromorphology of the system. However, these studies lacked any empirical data for lake inflows. The two years of newly collected inflow data (Chapter 3) have provided greater certainty over the modelled inflows produced by the SHETRAN model in Chapter 4.

Inflows into the original ISIS model were provided to the consultant by the Environment Agency. A comparison of these inflows and those generated by the SHETRAN model for the over lapping time period is provided in the results section of this chapter.

The previous modelling study also only analysed the impacts on hydrology based on one outlet profile. Lake levels were only briefly studied, and the flood attenuating properties of Ennerdale Water were not assessed. This chapter therefore provides a more robust analysis and examines a wider range of hydrological behaviours of the lake and downstream hydrology.

6.2 Aim of Chapter 6

There are three key aims in this modelling chapter. Firstly, a detailed comparison of the levels of hydrological alteration is provided by comparing observed historic data from 1995-2019 with a modelled natural flow regime. This modelled flow regime is based on the 12 outlet cross-sections generated from the geomorphic modelling. This assumes channel stabilisation will have started to occur after thirty years and that changes to the outlet profile beyond this period will be minimal. This is based on the rates of change observed from the modelling study that showed incrementally decreasing rates of change over time.

The second aim of the modelling in this section is to assess how the hydrology of the Upper Ehen catchment and Ennerdale Water may evolve over time in response to a geomorphologically adjusting lake outlet. This is set in the context of both historic meteorological data and future climate projections.

The third, and final, aim is to demonstrate the application of a modelling a tool, developed specifically for this study, that allows the end user to input numerous lake outflow designs and assess their impact on the lake levels and the storage discharge relationship.

6.3 Methodology

A simple modelling tool was developed that allows multiple level pool routing model runs to be quickly carried out under a range of physical outlet scenarios. The application demonstrated in this chapter uses modelled cross-sectional profiles however the approach used in this modelling tool is applicable to engineered channels or control structures as demonstrated in Chapter 4. The model, written in Python follows a two-stage process. Firstly, a cross sectional profile is obtained (modelled or otherwise). This is input to the model as a .csv file with two columns, chainage in m in column 1 and elevation in mAOD in column 2. This is then read by the program which plots the cross-section for visual verification of the data by the user (Figure 113). The user is prompted to enter a slope for the channel and a Manning's roughness coefficient. Using this data, the model calculates the cross-sectional area of the channel for a given stage based on trapezoidal areas across the channel. The user is able to specify the number of stage readings (i.e., the data resolution) from which the model will generate a stage discharge curve using the Manning's open channel flow equation. This is

then combined with the lake level and storage data to produce the downstream conditions or transfer function file.

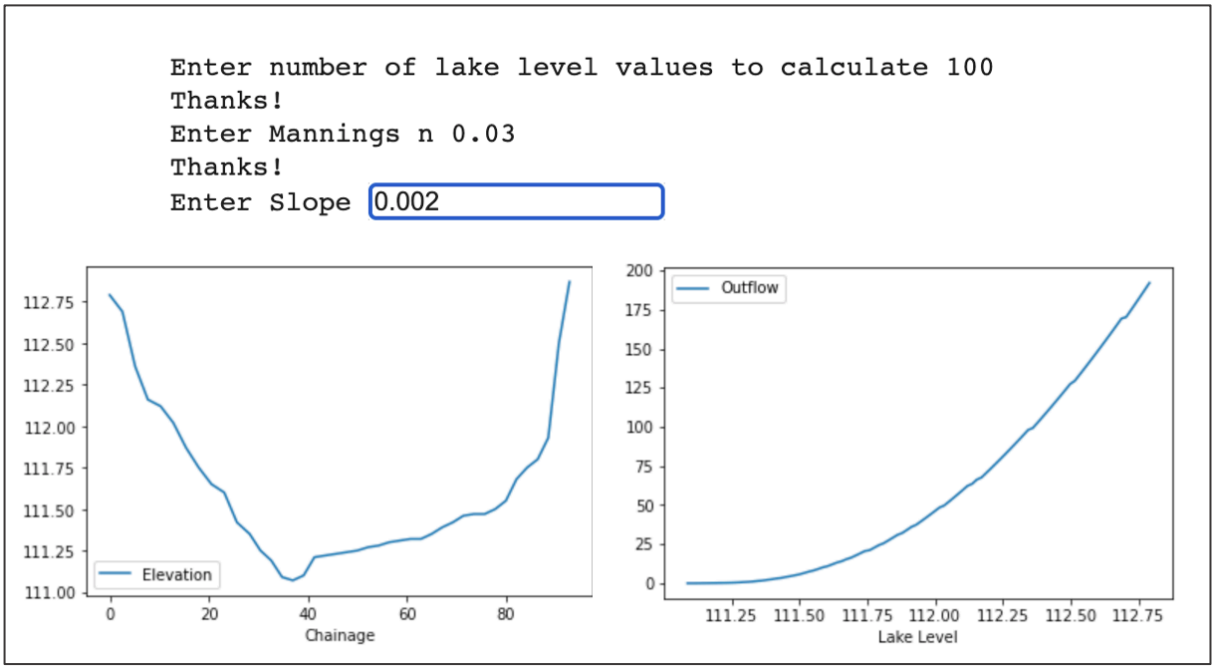


Figure 113. Example of the first stage of the model process which produces a stage discharge curve based on a cross-section profile, and information on the slope and channel roughness.

The second stage of the modelling process uses the downstream conditions (transfer function) file and an inflow time series. Inflow data is provided as a two-column .csv file with a datetime header in column 1 and discharge in column 2. The model user then inputs the file names (and file path if using inbuilt python program) for both the downstream condition file and the inflow file. Upon running the code, the user is then prompted to enter the timestep, initial lake storge, initial discharge and lake level. Finally, the user is asked if they wish to include abstraction. If abstraction is to be include a separate .csv file is required in the same format as the inflow file but using abstraction as the second column header. The model produces an output of discharge time series and lake stage time series corresponding to the length of the inflow data (Figure 114). The modelling tool also features the option to plot observed data on the same time series if this is imported as a .csv file by the user, otherwise observed data is ignored.

```
Enter timestep (hours)24
Thanks!
Enter initial storage (m^3)50433049
Thanks!
Enter initial outflow (m^3/s)1.133
Thanks!
Enter initial lake level (m)111.31
Thanks!
Importing Inflow.csv...
Inflow file successfully imported!
Importing Downstream.csv...
Downstream file successfully imported!
Calculating downstream conditions...
Do you want to include Abstraction?no
Ok
Checking abstraction data...
Calculating outflow for  366  day time series....
No observed_data found...Observed_data.csv missing!
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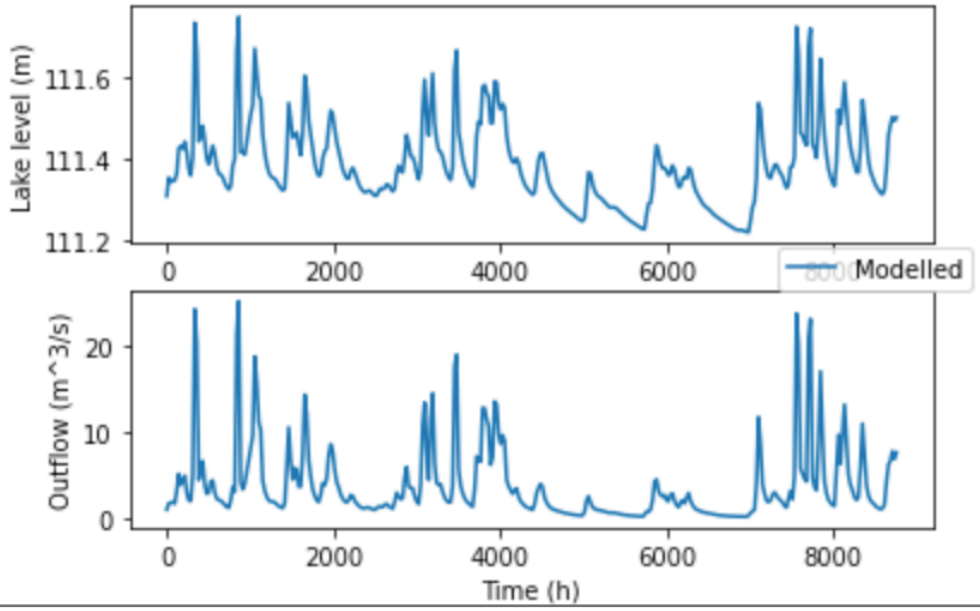
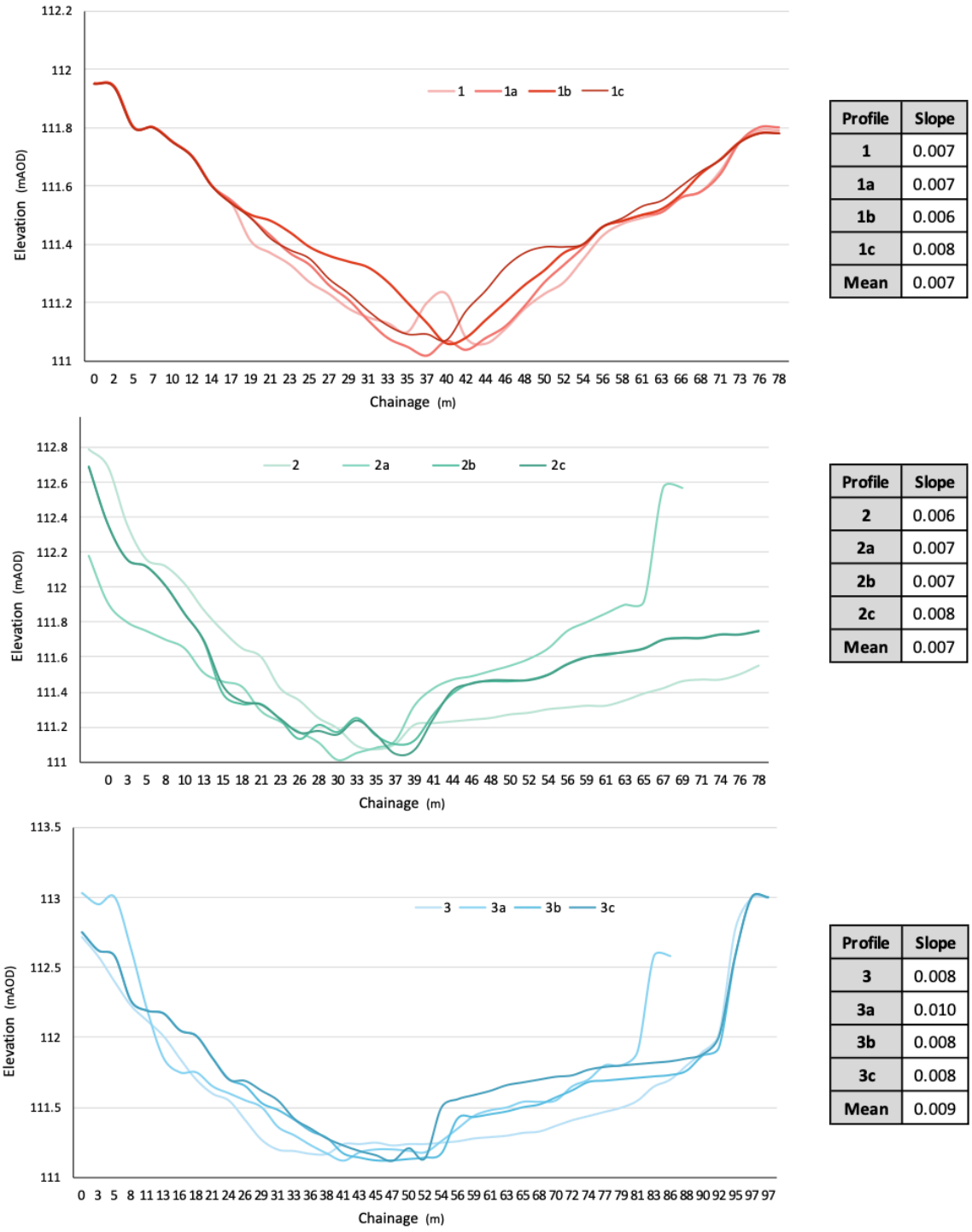


Figure 114. Stages and confirmation messages shown when running the automated level pool routing model.

Outlet Profiles

Outlet profiles were extracted from the modelling outputs produced in Chapter 5. This was done using the profile add-in tool in Q-GIS. A DEM was obtained for each year of each model run and cross-section profiles were extracted. The profile add-in tool allows the chainage and elevation to be copied and output either in graphical form as an image file or in numerical form as a .csv file. Channel slope was also obtained using the same tool by extending a profile line downstream for 100m to obtain an average slope. Twelve outlet

profiles (Figure 115) were generated from the 30-year geomorphic model runs and used to assess the influence of the outflow form on both the downstream hydrology and lake level.



Profile	Slope
1	0.007
1a	0.007
1b	0.006
1c	0.008
Mean	0.007

Profile	Slope
2	0.006
2a	0.007
2b	0.007
2c	0.008
Mean	0.007

Profile	Slope
3	0.008
3a	0.010
3b	0.008
3c	0.008
Mean	0.009

Figure 115. Outlet profiles generated after 30-years of model simulation used in the hydrological analysis.

Inflow data

Inflow data was obtained from three different sources and for different time periods. Historic inflow data was obtained using the calibrated SHETRAN model from 1995-2019. Used in conjunction with the level pool routing model in Chapter 4, this was shown to replicate the

observed managed regime with a NSE of 0.8 from 1995-2019. The inflow data generated by SHETRAN was compared to existing inflow data generated by the EA for the period from 1975-2012. Inflow data were also available from United Utilities Water Resource Management Plan, upon which the decision to proceed with the Thirlmere Pipeline project was partly based. This data set contained 200 x 87 years of inflow data generated from a CATCHMOD rainfall runoff model using stochastic rainfall and Potential Evapotranspiration data. This data was used to produce a 1,143-year inflow timeseries (12 x 87 years) to generate hydrological statistics from two natural outlet profiles. The choice of natural outlet profiles was based on the low flow metrics produced from the 1995-2019 modelled period.

A further climate change scenario was modelled using UKCP18 rainfall projections. Potential Evapotranspiration was calculated using adjusted Thornthwaites formula from projections of temperature for the period 2025 -2049 from the (CEDA) data portal. Rainfall and PET distributions were applied using the same methodology as the 1995-2019 model runs described in Chapter 4. The calibrated SHETRAN model was then used to generate an inflow time series based on the UKCP18 projects. No land change scenarios were considered.

Model Runs and Data Analysis

1. The first analysis to be conducted was a comparison of the newly generated lake inflows from the calibrated SHETRAN model and the existing inflow data used in previous studies by the Environment Agency. The existing inflow data has been used in a number of previous natural flow studies for the Upper Ehen (MWH and Environment Agency, 2013). This comparison therefore provides an opportunity to highlight any major differences between the two data sets and demonstrate where the SHETRAN model has overcome limitations of the previous data set.
2. An initial model comparison was made using the EA inflow data and the new level pool routing (LPR) model. This was done to provide evidence that the LPR model can replicate the existing ISIS model if given the same inflow data and boundary conditions (cross-section profile, slope etc.). This provided a means of further validating the suitability and functionality of the LPR model for the purposes of this study.
3. The LPR model was run using the existing infrastructure dimensions to demonstrate the impact of abstraction on the hydrological regime. The model was firstly run with no abstraction but with compensation flow regulation in place and then for a second time with neither abstraction nor compensation flow regulation in place. The purpose of this was to demonstrate the hydrological behaviour of the Upper Ehen in a period when abstraction has ceased and prior to the removal of any weir infrastructure.

4. The LPR model was then run for the period 1995-2019 using the inflow data from the SHETRAN model and the 12 initial cross section profiles used in Chapter 5. The results were then compared to the observed data for the same period. The aims of this comparison were two-fold. Firstly, by comparing the 12 model outlets with one another it is possible to demonstrate how the width and elevation of an engineered channel outlet may influence the downstream hydrology and the lake level responses. Secondly, by using hydrological metrics to quantify change, it enabled an initial assessment to be made of the level of hydrological alteration from the observed flow to a more natural regime.
5. In a similar way to the comparison in step 3, the LPR model was used to generate a flow time series using the inflow data from 1995-2019 and the outlet cross-section from the end of the 30-year geomorphic modelling runs. The main aim of this was to assess how the hydrology and behaviour of Ennerdale lake would change after 30 years. A comparison was again also made between these modelled hydrological outputs and the observed data to assess the levels of hydrological alteration between the managed and hypothetical natural regimes.
6. Stochastic inflow data obtained from the United Utilities Water Resource Management Plan was used to generate a robust series of hydrological statistics for the final outlet channel profiles. Two profiles were chosen that represented the upper and lower extremes of the low flow metrics generated from the model runs in step 5.
7. Finally, model runs were conducted using future climate scenarios. Inflow time series were generated using SHETRAN, as set up as in Chapter 4 and using climatic projections from the UKCP18 emissions scenario 8.5. The rationale for these runs was to demonstrate possible future hydrological behaviour under high-emission climate change scenarios.

6.4 Results and Analysis

The results section examines different model runs, and scenarios set out as described in the previous section, with a focus on the changes to the downstream discharge. A detailed analysis of the impact of different outlets forms and inflow data on lake level is assessed in a separate final section.

6.4.1 *Evaluation and Comparison of Inflow Data*

Inflows generated by the SHETRAN model were compared to inflows generated from a previous study carried out by the Environment Agency, Cumbria office and used in the MWH study. The timeseries was compared for the overlapping period 1995-2012 (Figure 116). Both

inflow timeseries showed very similar mean flow values of 3.33 m^3/s for the EA data and 3.38 m^3/s for the SHETRAN-generated inflows. Average monthly flows varied more during winter months with the SHETRAN-generated inflows being greater in all winter months. During the summer months inflows generated by the SHETRAN model were slightly lower than the EA inflow time series.

Maximum monthly flow estimates were also largest across all months in the SHETRAN-generated timeseries. The largest variations were observed for October and June. Maximum monthly flow estimates for September showed the least variation between the two data sets.

An unpaired t-test was used to analyse the monthly means for both time series over the 17-year period. Across all months the differences between the two data sets were shown to be not statistically significant, with P values ranging between 0.998 and 0.686. Under conventional criteria a P value below 0.05 (typically ≤ 0.05) is statistically significant for this test.

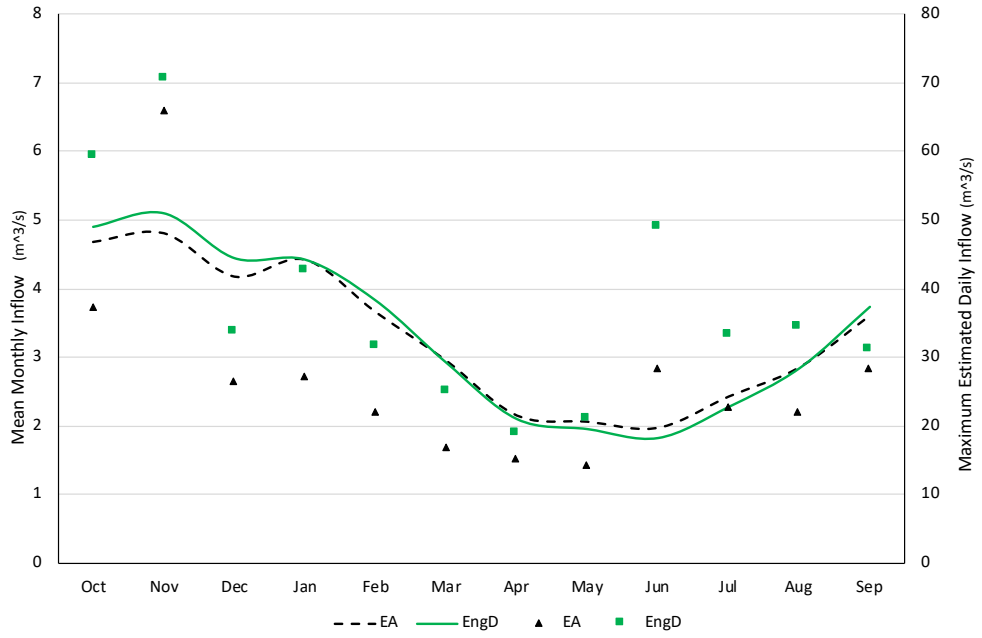


Figure 116. Comparison of inflow data showing mean monthly and maximum daily inflow estimates for 1995-2012. Black lines and symbols represent values for the inflows generated by the Environment Agency, green colours labelled EngD represent the inflows generated by this modelling in the thesis.

A time series was also generated using the UKCP18 data. This was taken from the 8.5 emissions scenario. The rainfall data used to generate the inflow time series suggest shifting seasonal distribution of rainfall with decreasing rainfall over time in winter months and increasing rainfall during the later summer months (Figure 117).

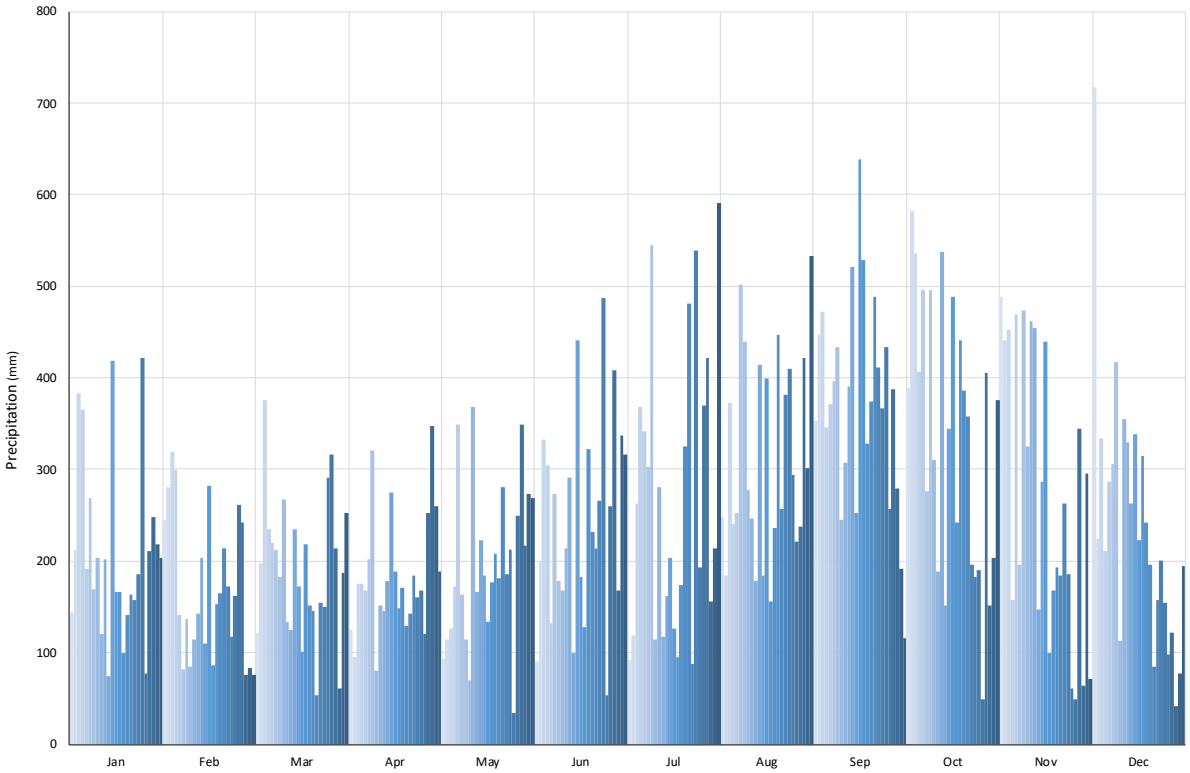


Figure 117.Changing monthly rainfall totals under a high emissions climate change scenario. Darker colours in each month represent years further into the future. Data period is 2025-2049.

Peak flows under a high emissions scenario are predicted to occur in September. Lowest average monthly flows are observed in April (Figure 118). Maximum daily flow events occur in September, August and July.

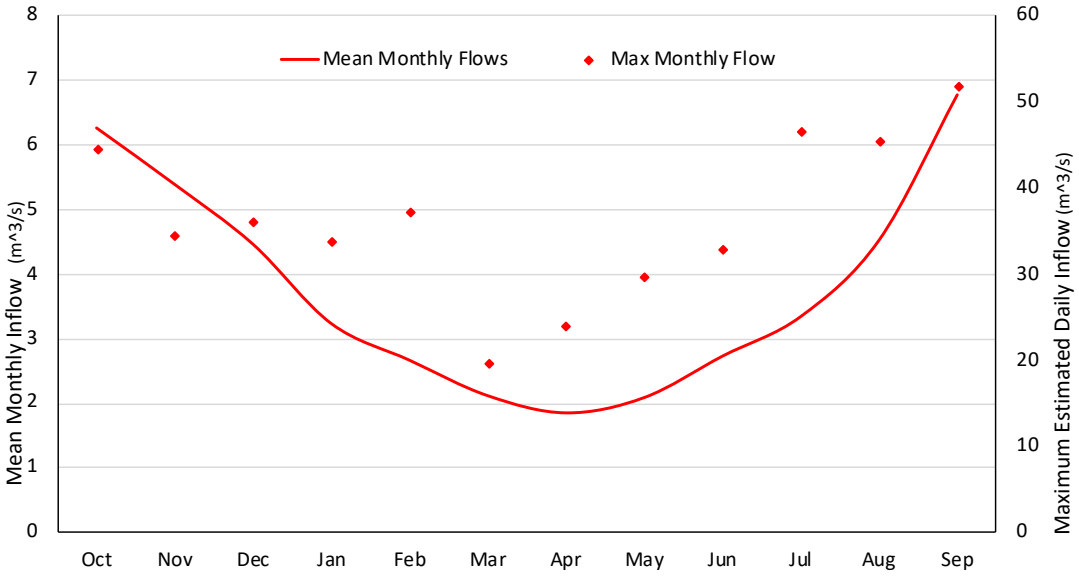


Figure 118. Mean monthly flow projections under a high-emissions scenario from 2025 - 2049

The statistical distribution of all three inflow time series was analysed in order to produce flow duration curves. Whilst the UKCP18 inflow data shows a shifting seasonal pattern the

overall statistical distribution of flows is similar to the inflow time series for earlier periods (Figure 119). The main difference can be observed at the upper range of flows. The average flow in the future climate projection scenario is raised from 3.4 m^3/s to 3.8 m^3/s.

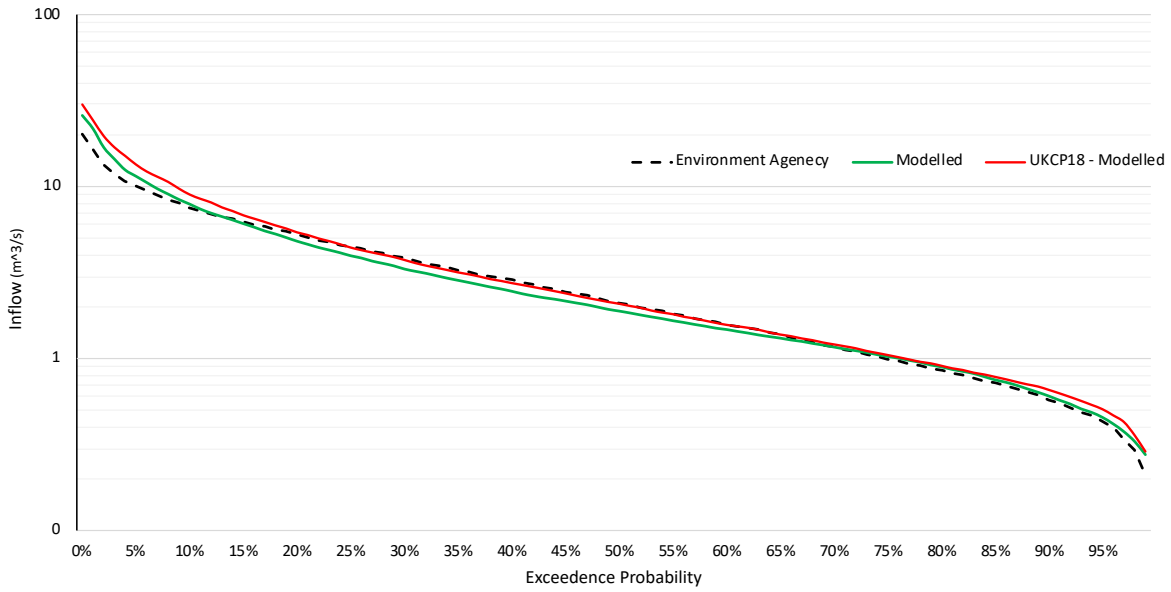


Figure 119. Flow duration curves for estimated inflows for period 1995-2012 and UKCP18 2025-2049. Black dashed line shows previous Environment Agency estimates. Green shows modelled SHETRAN inflows from 1995-2019. Red line shows SHETRAN modelled flows for 2025-2049 using UKCP18 data.

6.4.2 *ISIS and LPR Model Comparison*

The LPR model was set up using the proposed outlet profile from a previous natural flow study commissioned by the Environment Agency and carried out by MWH. This model used an initial slope of 0.001637 and Environment Agency inflow data. No data for streambed roughness was provided from the previous study so an initial Manning's roughness value of 0.04 was used. Additional runs were conducted using Manning's roughness values of 0.05 and 0.03. These represent the typical range of values for gravel bed rivers (Chin, 2000; Chadwick *et al.*, 2013).

The LPR model was able to replicate the ISIS model outputs with a NSE of 0.97 when using a Manning's roughness coefficient of 0.05 and the same slope and cross-section. This model also produced the best overall performance when looking at RMSE, R^2 and RSR indicators. However, the LPR model with a 0.03 Manning's roughness coefficient produced more similar values to the ISIS model when comparing Q_{95}, Q_{70} and Q_5.

The results also show how channel roughness can influence the flow regime. Lower roughness values allow larger volumes of water to be conveyed more rapidly. This results in higher peak flows and reduced low flows as shown in the modelled Q_{95} and Q_5 values (Table

24). The results from this comparison provide further validation of the LPR model and show that it is capable of reproducing natural discharges when given specific boundary conditions and inflow data.

Table 24. Hydrological outputs and model performance indicators for LPR model. Numbers below LPR model name indicate Manning's roughness coefficients used.

	ISIS Model	LPR Model (0.05)	LPR Model (0.04)	LPR Model (0.03)
Mean Q (m^3/s)	3.18	3.18	3.18	3.18
Max Q (m^3/s)	44.03	54.02	54.91	57.85
Min Q (m^3/s)	0.04	0.06	0.05	0.04
Q95 (m^3/s)	0.38	0.42	0.39	0.38
Q70 (m^3/s)	1.23	1.25	1.21	1.22
Q50 (m^3/s)	2.15	2.16	2.17	2.11
Q10 (m^3/s)	7.49	7.09	7.10	7.07
Q5 (m^3/s)	9.30	9.15	9.18	9.20
Model Performance				
R^2		0.98	0.97	0.95
RSR		0.16	0.18	0.22
RMSE (m^3/s)		0.48	0.56	0.68
NSE		0.97	0.96	0.95

6.4.3 *Impact of Abstraction*

The impact of abstraction on the hydrological regime has been demonstrated in terms of its contribution to the overall water balance of the upper catchment (Chapter 3). However, demonstrating how the hydrology of the Upper Ehen might change if abstraction ceases and the infrastructure remains in place is important for the management of the site in the interim period prior to weir removal.

Figure 120 shows a comparison of the flow duration curve using observed flows for the 1995 – 2019 period with modelled flows for the same period in a scenario with no abstraction and a scenario with no abstraction and no flow regime management. In both scenarios the impact of no abstraction and thus a greater abundance of water in the catchment is evident in the elevated flows shown in the flow duration curve. When compensation flows are removed, low flows (95% -99% exceedance) reduce slightly when comparing the two modelled scenarios however in both scenarios they remain above observed values.

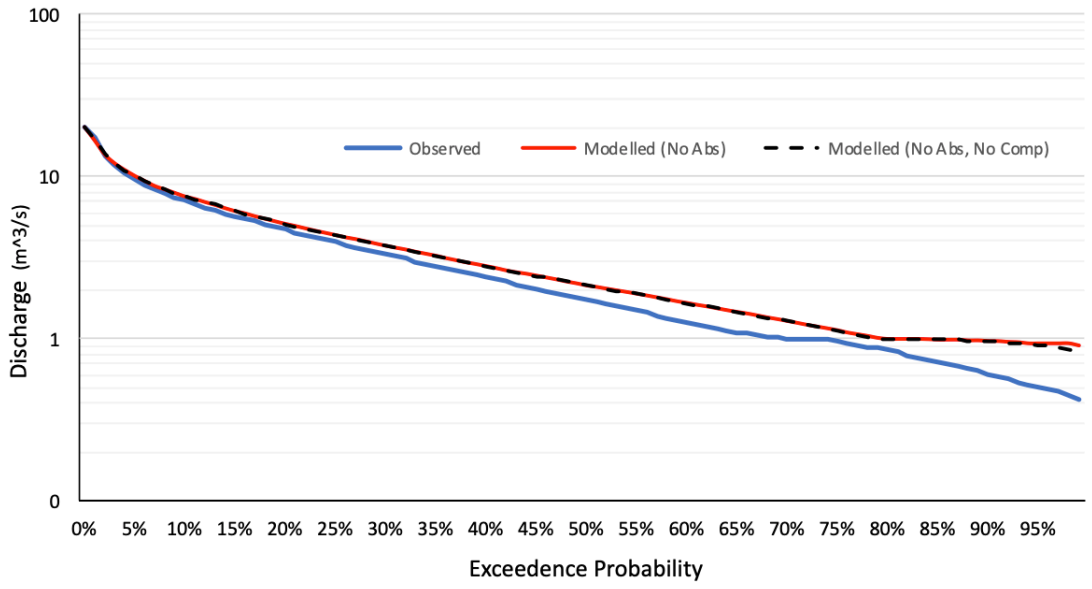


Figure 120. Flow duration curves for observed hydrology and modelled scenarios with existing infrastructure but varying levels of flow regime management

Removing abstraction from the model extends the middle section of the stage duration curve (Figure 121). The lower limits are also lessened under a management regime with no abstraction and lessened further when compensations flows are removed. This is since the lake level is not drawn down as rapidly without abstraction. At extreme low flows additional drawdown occurs as pumped compensation flows are currently required to meet the terms of the United Utilities' impoundment licence.

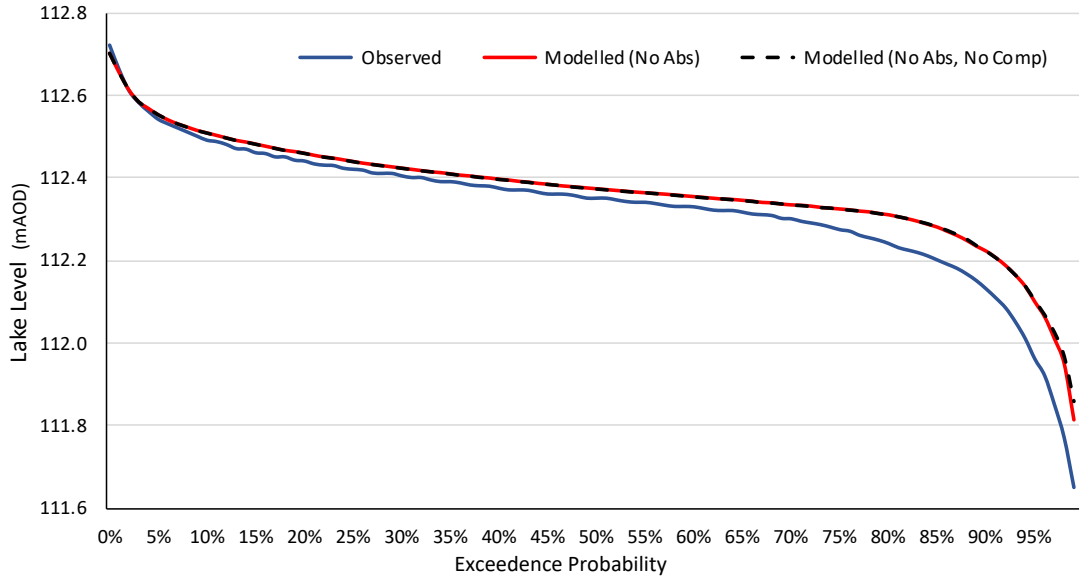


Figure 121. Lake stage duration curves for existing hydrological regime and a regime with no abstraction and no abstraction or compensation flow regulation. All scenarios with current infrastructure in place.

6.4.4 *Initial Engineered Outlet Cross-sections*

Flow regimes were generated using inflow data from 1995-2019 and 12 initial outlet cross section designs from the geomorphic modelling study (Chapter 5). The results show how removing abstraction and infrastructure from the outlet of the lake is predicted to influence the flow regime.

Figure 122 shows a comparison of the mean monthly flows for the entire observed time series against the average of the modelled mean flows obtained using the engineered outlets. This is shown by subtracting the average of the modelled monthly mean flows from the observed monthly mean. (For example in the first cell, October 1995, the observed monthly flow was 5.1 m^3/s however the model predicts a mean monthly flow of 6.1 m^3/s, a change of 1.0)

Each column in the grid shows a hydrological year with each cell representing a month in that year. Cells in red highlight where model predictions show lower monthly means than the observed time series. Green cells show a modelled increase in mean flows, whilst white cell indicate no variation between modelled and observed values.

The overall trend from the 12 model runs shows an increase in average monthly and annual flows. However, in 49 of the 288 months during the period from 1995 to 2019, average modelled flows were lower than the observed flows under the current management regime, equivalent to 17% of the time (Figure 122).

Of the months where monthly average modelled flows were lower than those in the observed historic timeseries, the majority occurred in April, where the modelled predicted lower flows than the observed values in 6 of the 25 years.

	95-96	96-97	97-98	98-99	99-00	00-01	01-02	02-03	03-04	04-05	05-06	06-07	07-08	08-09	09-10	10-11	11-12	12-13	13-14	14-15	15-16	16-17	17-18	18-19
Oct	1	1.5	0.5	1.1	0.8	1	0.5	1.1	0.3	0.5	0.7	-0.2	0.7	-0.8	0.8	0.7	1	0.8	0.2	0.5	0.5	0.3	-0.6	0.5
Nov	0.9	0.9	1	1	0.5	1	0.5	1.3	0.8	0.5	0.3	0.9	0.8	0.6	-0.6	0.4	0.6	0.5	0.2	0.0	-0.7	0.8	0.2	1.3
Dec	0.4	0.6	1.1	0.6	0.8	0.5	0.3	0.4	0.3	0.6	0.7	0.3	0.2	0.4	1.1	0.6	0.7	1.2	0.8	0.4	-1	0.1	0.0	0.4
Jan	0.9	0.2	0.6	0.3	0.4	0.7	0.6	0.4	0.9	-0.1	-0.1	0.2	0.7	0.2	0.1	0.3	0.6	0.6	0.6	-0.2	0.4	0.2	0.4	-0.1
Feb	1.2	1	0.0	0.5	0.4	0.5	0.6	0.2	0.1	0.4	0.4	0.7	0.2	0.3	0.5	0.4	0.1	0.2	1	-0.1	0.3	0.1	0.2	0.3
Mar	0.5	0.0	0.5	0.3	0.1	0.1	0.5	0.3	0.4	0.5	0.6	0.2	0.5	0.2	0.3	0.5	0.0	0.3	-0.2	-0.1	0.2	-0.7	0.3	0.0
Apr	0.2	0.3	0.6	0.0	0.6	0.2	0.0	0.2	0.3	0.3	0.1	-0.1	0.2	0.3	-0.1	-0.2	0.6	0.2	-0.3	0.0	-0.1	-0.2	0.3	0.2
May	0.4	0.2	0.1	0.3	0.4	0.0	-0.1	0.2	0.1	0.1	0.2	0.3	-0.2	0.0	0.0	0.7	0.0	0.0	0.2	0.2	-0.1	0.1	-0.1	0.0
Jun	0.0	0.1	0.4	0.4	0.3	0.2	0.2	0.2	0.3	-0.3	0.0	0.8	0.7	0.3	-0.1	0.3	0.3	0.1	-0.3	-0.2	0.4	0.2	0.1	0.5
Jul	0.3	0.6	-0.1	0.0	0.2	0.3	0.0	0.3	0.2	-0.1	0.0	0.1	0.2	0.9	0.7	0.3	0.3	0.2	0.3	0.1	-0.1	0.1	-0.2	0.2
Aug	0.3	0.4	-0.2	0.4	0.3	0.3	0.2	0.0	0.9	0.9	1.1	0.1	0.4	0.1	0.1	0.5	0.5	-0.3	0.4	0.1	0.5	-0.5	1.1	-0.1
Sep	0.8	0.1	0.5	1.3	1	0.7	0.3	0.8	0.0	0.7	0.7	0.2	0.2	0.0	0.8	0.5	1	0.3	-0.3	0.4	0.1	0.2	0.2	0.1

Figure 122. Comparison of modelled mean monthly flows with observed values by hydrological year. Values are shown in m^3/s. Green cells indicate that modelled natural flows are higher than observed flows. Red cells show

where modelled flows are lower than observed flows. White cells show no variation between modelled and observed values.

Two reasons are proposed for these predicted changes. Firstly, an under estimation of lake inflows caused by a combination of the model set up and input data (e.g. under estimates of rainfall) may explain some of the variation. The second cause is due to the management of compensation flows. In extended periods without rainfall, lake outflows are artificially elevated through the controlled release of water. This has the effect of producing flattened hydrographs (Figure 123) and maintaining elevated discharges which would otherwise not be observed in an unmanaged system. The largest increases between the observed data and modelled natural flows were present in the monthly means for October, November and December. Modelled discharges during the winter months, consistently showed higher values than the observed flows under the current system of flow control.

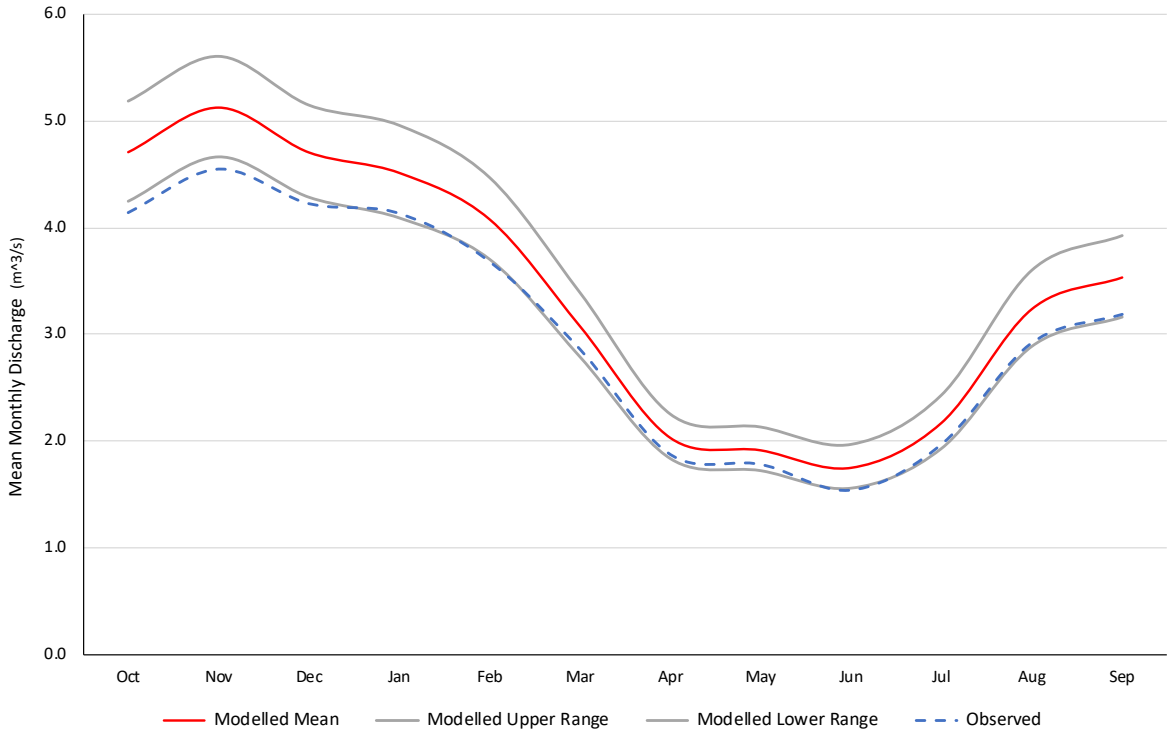


Figure 123. Range of monthly mean modelled flows using initial outlet cross-sections compared to observed values for 1995-2019

A one sample t-test was used to assess the difference between the observed monthly means and the modelled monthly means. In all cases the test returned P values below 0.0001 indicating that the difference between the modelled and observed means is extremely statistically significant.

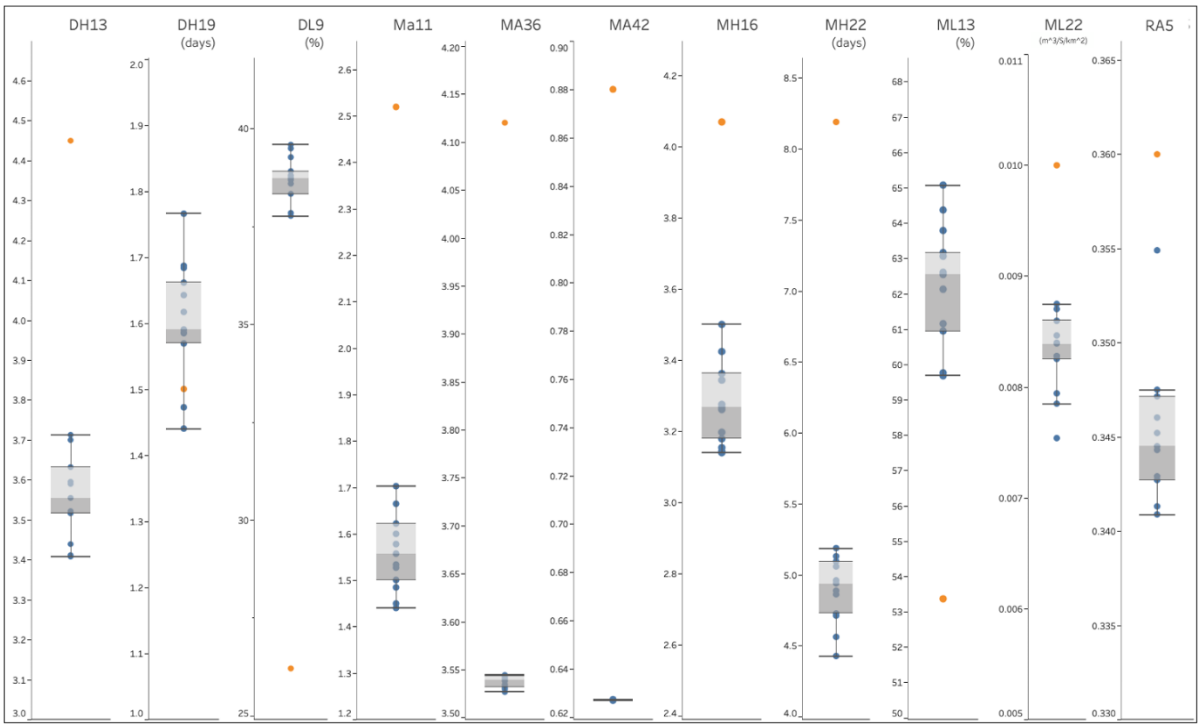
6.4.5 *Cross-sections after 30 years (natural)*

The 11 hydrological metrics identified in the national scale study were generated using the newly modelled time series and examined visually using box and whisker plots (Figure 124). With the exception of metric DH19 the observed values fall well outside the range of modelled values. These metrics were then further evaluated to test for statistical differences between the observed and modelled data.

A one sample t-test was used to assess the difference between the metrics obtained from the observed values and the mean of the metrics from the modelled values. A one sample t-test assumes a normal distribution. The data were therefore first assessed for normal distribution using the Shapiro-Wilk test to confirm this assumption.

The results from the t-test produced P values below 0.0001 for all but two of the 11 metrics. Metrics DH19 and RA5 produced p values of 0.0016 and 0.0008 respectively however in all cases the score indicate extremely significant differences between the metrics obtained with modelled natural flow regime data and observed data

Figure 124. Box and whisker plots showing 11 key hydrological metrics for 12 outlet profiles after 30 years (blue) compared to metrics under observed conditions (yellow). Metrics are dimensionless unless stated below metric label.



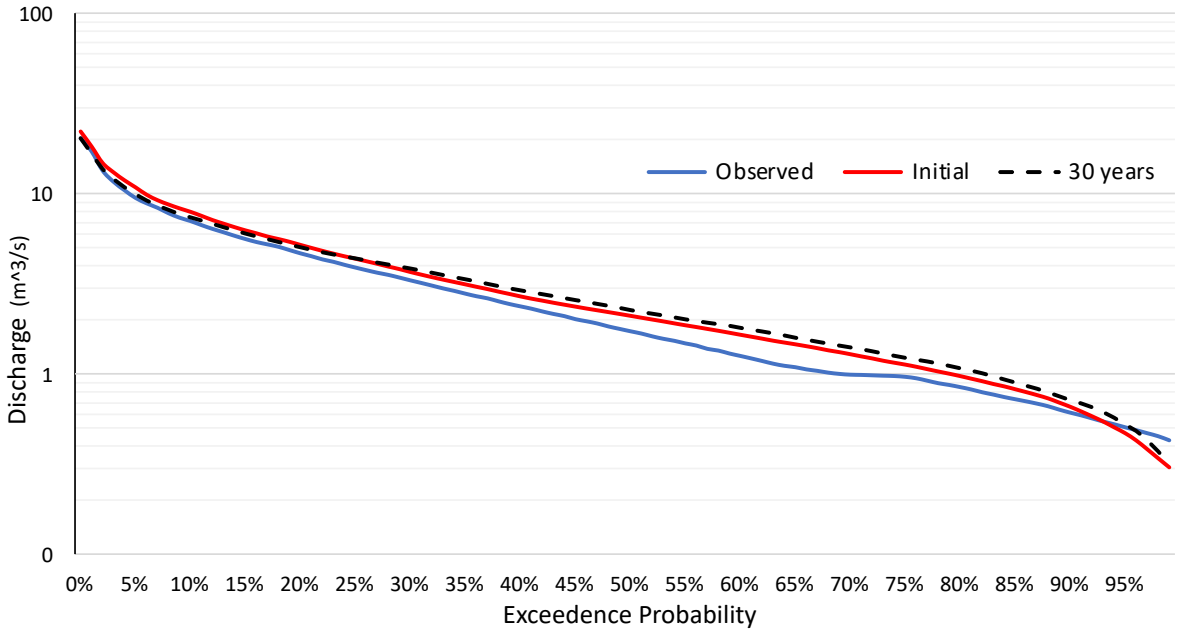


Figure 125. Average flow duration curve for 12 outlet profiles at different stages of evolution compared to flow duration curve for observed data1995-2019

As expected, in the absence of compensation flow releases, the model predicted that absolute low flow values would be lower in an unmanaged regime given the same inflow volumes (Table 25). Modelled mean monthly minimum flows were only lower in May, June and July and higher than observed monthly flows for the remainder of the year. Short duration low flow events showed a decrease in minimum discharge volumes. The mean annual minimum flows, 3-day and 7-day moving average minimum flows (metrics dl 1-3) for the modelled natural regime were 0.37 m^3/s, 0.38 m^3/s and 0.42 m^3/s. This compared to 0.55 m^3/s, 0.57 m^3/s and 0.59 m^3/s for the corresponding metrics calculated from observed historic discharge timeseries. The differences between the modelled means and the observed vales were compared using a one sample t-test. In all cases p values below 0.0001 were returned, suggesting that the differences were extremely statistically significant.

Metrics dl4 and dl5, the 30-day and 90-day minimum moving averages, showed an increased under modelled natural conditions. These increased from 0.71 m^3/s and 1.2 m^3/s respectively based on the historic discharge data to mean values of 0.74 m^3/s and 1.42 m^3/s under a natural outlet and regime without abstraction or flow regulation. The t-test result returned p values below 0.0001 indicating that the differences between the observed values and modelled means were extremely statistically significant.

Table 25. Low flow metrics calculated for each lake outlet at end of 30-year geomorphic model run compared to metrics for observed flow regime using data from 1995-2019. Green numbers indicate highest number and red indicate lowest number for each metric. Metrics with no units are dimensionless.

Indices		Units	Observed 1995 - 2019	Modelled Low Flow Metrics 1995 - 2019												
				Mean Modelled	1	1a	1b	1c	2	2a	2b	2c	3	3a	3b	3c
ml1	*January Minimum*	*m^3/s*	1.11	1.48	1.45	1.50	1.52	1.51	1.42	1.56	1.50	1.49	1.37	1.45	1.52	1.54
ml2	*Febuary Minimum*	*m^3/s*	1.09	1.38	1.37	1.41	1.42	1.39	1.34	1.43	1.38	1.38	1.28	1.35	1.42	1.41
ml3	*March Minimum*	*m^3/s*	0.83	1.04	1.01	1.05	1.09	1.06	1.01	1.09	1.04	1.04	0.96	1.00	1.06	1.06
ml4	*April Minimum*	*m^3/s*	0.71	0.72	0.71	0.72	0.76	0.74	0.70	0.76	0.72	0.72	0.66	0.69	0.72	0.73
ml5	*May Minimum*	*m^3/s*	0.72	0.64	0.63	0.65	0.68	0.66	0.63	0.67	0.64	0.65	0.58	0.61	0.64	0.65
ml6	*June Minimum*	*m^3/s*	0.69	0.54	0.53	0.54	0.57	0.55	0.54	0.56	0.54	0.54	0.49	0.51	0.53	0.54
ml7	*July Minimum*	*m^3/s*	0.73	0.71	0.69	0.73	0.75	0.73	0.70	0.75	0.71	0.72	0.65	0.68	0.72	0.72
ml8	*August Minimum*	*m^3/s*	0.93	1.03	1.00	1.04	1.06	1.04	0.99	1.08	1.03	1.05	0.96	1.01	1.04	1.06
ml9	*September Minimum*	*m^3/s*	0.86	1.00	0.98	1.02	1.04	1.02	0.97	1.06	1.00	1.01	0.92	0.96	1.01	1.02
ml10	*October Minimum*	*m^3/s*	1.02	1.43	1.39	1.46	1.45	1.45	1.36	1.51	1.43	1.46	1.30	1.38	1.49	1.47
ml11	*November Minimum*	*m^3/s*	1.14	1.68	1.64	1.71	1.71	1.69	1.61	1.78	1.69	1.70	1.56	1.63	1.73	1.73
ml12	*December Minimum*	*m^3/s*	1.13	1.60	1.56	1.62	1.64	1.60	1.55	1.70	1.60	1.62	1.49	1.56	1.65	1.65
ml13	*Coefficient of variation*	*%*	53.37	62.35	62.12	63.16	59.76	60.93	59.68	63.78	62.55	63.06	61.15	62.61	65.06	64.37
ml14	*Mean of annual minimum annual flows*	*m^3/s*	0.33	0.16	0.17	0.16	0.18	0.17	0.18	0.17	0.16	0.17	0.16	0.16	0.15	0.16
ml15	*Low flow index*	-	0.18	0.11	0.11	0.11	0.11	0.11	0.11	0.11	0.11	0.11	0.10	0.10	0.10	0.11
ml16	*Median of annual minimum flows*	*m^3/s*	0.3	0.16	0.16	0.16	0.17	0.16	0.17	0.17	0.16	0.16	0.15	0.15	0.15	0.16
ml17	*Base flow*	-	0.19	0.12	0.12	0.12	0.13	0.12	0.13	0.13	0.12	0.12	0.11	0.12	0.12	0.12
ml18	*Variability in base flow index 1.*	*%*	26.78	31.42	30.78	32.24	31.03	32.17	27.54	31.45	30.21	31.74	32.73	32.20	33.19	31.70
ml19	*Base flow*	-	18.02	10.85	10.86	10.80	11.45	11.08	11.44	11.26	11.00	10.99	9.86	10.27	10.39	10.84
ml20	*Base flow.*	-	0.39	0.44	0.43	0.44	0.44	0.44	0.42	0.46	0.44	0.45	0.40	0.42	0.45	0.45
ml21	*Variability across annual minimum flows*	*%*	34.71	34.55	34.16	34.25	34.37	36.51	28.41	33.53	33.46	35.73	36.52	36.03	36.54	35.16
ml22	*Specific mean annual minimum flow*	*m^3/s/km^2*	0.010	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
fl1	*Low flood pulse count*	*Events/year*	8.83	9.42	9.08	9.38	8.96	8.92	9.50	9.13	9.46	9.54	10.58	9.71	9.33	9.42
fl2	*Variability in low pulse count*	*%*	54.31	26.73	26.75	28.28	26.43	27.45	24.24	27.67	28.06	28.15	22.79	25.32	27.48	28.17
fl3	*Frequency of low pulse spells*	*Events/year*	0	0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.00	1.00	1.00	0.00
dl1	*Annual minimum daily flow*	*m^3/s*	0.55	0.37	0.37	0.37	0.39	0.38	0.39	0.38	0.37	0.37	0.34	0.35	0.35	0.37
dl2	*Annual minimum 3-day flow*	*m^3/s*	0.57	0.38	0.38	0.38	0.40	0.39	0.40	0.40	0.39	0.39	0.35	0.36	0.37	0.38
dl3	*Annual minimum 7-day flow*	*m^3/s*	0.59	0.41	0.41	0.41	0.43	0.42	0.42	0.43	0.41	0.42	0.38	0.39	0.40	0.41
dl4	*Annual minimum 30-day flow*	*m^3/s*	0.71	0.74	0.74	0.75	0.76	0.75	0.74	0.76	0.75	0.75	0.72	0.73	0.74	0.75
dl5	*Annual minimum 90-day flow*	*m^3/s*	1.2	1.42	1.42	1.43	1.43	1.43	1.42	1.43	1.43	1.43	1.42	1.42	1.43	1.43
dl6	*Variability of annual min daily average flow*	*%*	34.71	34.55	34.16	34.25	34.37	36.51	28.41	33.53	33.46	35.73	36.52	36.03	36.54	35.16
dl7	*Variability of annual min of 3-day moving average flow*	*%*	34.47	34.33	34.06	34.41	34.19	36.57	28.73	33.33	33.09	35.47	35.64	35.52	36.08	34.82
dl8	*Variability of annual min of 7-day moving average flow*	*%*	32.32	33.64	33.01	34.43	33.56	34.93	28.99	33.48	32.32	34.10	35.09	34.57	35.32	33.94
dl9	*Variability of annual min of 30-day moving average flow*	*%*	26.21	38.73	38.68	38.81	37.76	38.59	37.85	38.32	38.75	38.73	39.59	39.49	39.26	38.90
dl10	*Variability of annual min of 90-day moving average flow*	*%*	32.89	28.57	28.60	28.55	28.50	28.60	28.59	28.49	28.57	28.54	28.71	28.66	28.56	28.52
dl11	*Annual min daily flow/median for entire record*	-	0.32	0.16	0.16	0.16	0.18	0.16	0.18	0.17	0.16	0.17	0.15	0.16	0.15	0.16
dl12	*Annual min 7-day moving average/median entire record*	-	0.34	0.18	0.18	0.18	0.20	0.18	0.19	0.19	0.18	0.19	0.17	0.17	0.17	0.18
dl13	*Annual min 30-day moving average/median entire record*	-	0.41	0.33	0.32	0.33	0.34	0.32	0.34	0.33	0.32	0.33	0.33	0.33	0.33	0.32
dl14	*Low exceedence flows (75% Exceedence)*	-	0.56	0.54	0.52	0.54	0.57	0.53	0.54	0.57	0.53	0.55	0.54	0.54	0.54	0.54
dl15	*Low exceedence flows (90% Exceedence)*	-	0.35	0.32	0.30	0.33	0.33	0.30	0.33	0.34	0.30	0.32	0.31	0.31	0.32	0.31
dl16	*Low flow pulse duration (events below 25th percentile)*	*#days*	9.25	9.35	9.42	9.84	9.75	9.31	9.30	9.55	9.24	9.18	8.09	9.17	9.89	9.48
dl17	*Variability in low pulse duration*	*m^3/s*	64.54	37.55	37.73	38.39	40.38	37.48	34.97	41.03	38.21	38.42	31.41	36.29	37.71	38.63
tl1	*Julian date of annual minimum*	-	197	175	175	175	175	175	175	175	175	175	176	174	175	175
tl2	*Variability in date of annual minima*	*m^3/s*	58.9	39.1	39.0	39.1	38.8	38.8	39.0	39.0	39.0	39.0	39.5	39.2	39.2	39.1

6.4.6 *Stochastic Inflows*

Based on the findings from the previous section, outlet Profiles 2a and 3 were used to generate hydrological statistics from the stochastic inflow data. These outlet profiles were chosen as they showed the more extreme values for the low flow metrics and thus capture the upper and lower limits of expected low flow behaviours from the natural lake outlets. Mean inflow for the 1,143-year period was 3.05 m^3/s. This is lower than the modelled inflow average of 3.4 m^3/s for the 1995 - 2019 period. This may be in part due to the longer time period, which captured more droughts and drier years. It may also be due to the set up and structure of the CATCHMOD model. Maximum inflows for the stochastic timeseries were 30.5 m^3/s compared to a maximum inflow of 70 m^3/s for the modelled period.

As expected, based on the flow regime metrics in the previous section, Profile 2a (Figure 126) showed higher minimum flow values than Profile 3. The minimum flows generated were 0.036 m^3/s and 0.025 m^3/s for Profiles 2a and 3 respectively. The corresponding Q_{95} values were 0.31 m^3/s and 0.25 m^3/s respectively.

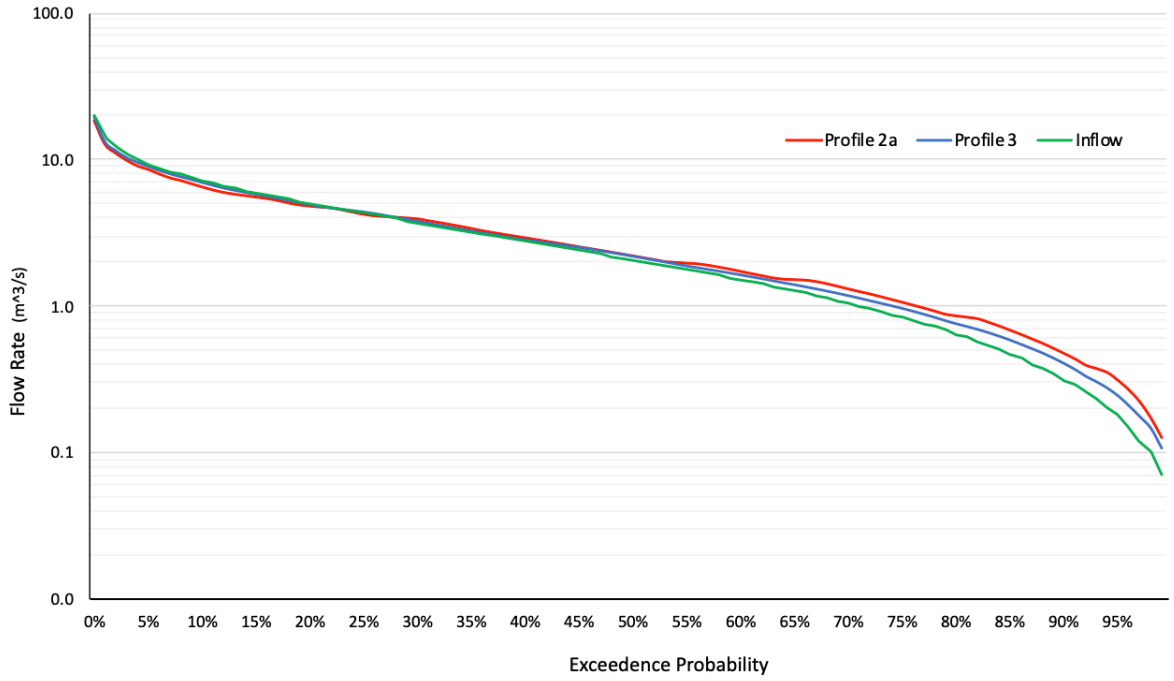


Figure 126. Flow duration curves generated from stochastic inflow data for outlet profiles 2s (red) and 3 (blue). Inflow timeseries characteristics are shown in green.

6.4.7 *Future Climate Scenarios*

Two principal variations between the modelled natural flow regime under historic flow conditions and the future climate scenario are evident in the data (Table 26). Firstly, the seasonal shift in the average monthly flows is clearly marked. Secondly, absolute minimum values are lower in the high emission future climate scenario. The mean annual flow increases to 3.8 m^3/s however additionally the minimum flow values are also lowered. In metrics such as the annual minimum, 3-day and 7-day moving average these decreases are minimal. In the 30-day and 90-day moving averages these differences are more marked, however.

The overall trend for the high-emissions future climate scenario is greater variation between the mean monthly flows with increased seasonality albeit a shift from the currently observed seasonal patterns.

Table 26. Comparison of low flow metrics under natural flow regime and future high-emissions natural flow regime scenario.

Indices		Units	Observed 1995 - 2019	Modelled 95-19			Modelled UKCP18		
				Mean	Min	Max	Mean	Min	Max
ml1	January Minimum	m^3/s	1.11	1.48	1.37	1.56	0.72	0.66	0.76
ml2	Febuary Minimum	m^3/s	1.09	1.38	1.28	1.43	0.69	0.63	0.73
ml3	March Minimum	m^3/s	0.83	1.04	0.96	1.09	0.65	0.59	0.68
ml4	April Minimum	m^3/s	0.71	0.72	0.66	0.76	1.06	0.97	1.12
ml5	May Minimum	m^3/s	0.72	0.64	0.58	0.68	1.30	1.17	1.40
ml6	June Minimum	m^3/s	0.69	0.54	0.49	0.57	1.61	1.45	1.70
ml7	July Minimum	m^3/s	0.73	0.71	0.65	0.75	1.50	1.40	1.58
ml8	August Minimum	m^3/s	0.93	1.03	0.96	1.08	1.63	1.48	1.76
ml9	September Minimum	m^3/s	0.86	1.00	0.92	1.06	1.74	1.57	1.86
ml10	October Minimum	m^3/s	1.02	1.43	1.30	1.51	1.38	1.25	1.45
ml11	November Minimum	m^3/s	1.14	1.68	1.56	1.78	1.00	0.91	1.05
ml12	December Minimum	m^3/s	1.13	1.60	1.49	1.70	0.79	0.73	0.83
ml13	Coefficient of variation	%	53.37	62.35	59.68	65.06	63.92	61.33	66.81
ml14	Mean of annual minimum annual flows	m^3/s	0.33	0.16	0.15	0.18	0.15	0.14	0.16
ml15	Low flow index	-	0.18	0.11	0.10	0.11	0.10	0.09	0.11
ml16	Median of annual minimum flows	m^3/s	0.3	0.16	0.15	0.17	0.14	0.13	0.15
ml17	Base flow	-	0.19	0.12	0.11	0.13	0.11	0.10	0.12
ml18	Variability in base flow index 1.	%	26.78	31.42	27.54	33.19	40.44	36.50	41.89
ml19	Base flow	-	18.02	10.85	9.86	11.45	9.94	9.13	10.54
ml20	Base flow.	-	0.39	0.44	0.40	0.46	0.42	0.38	0.44
ml21	Variability across annual minimum flows	%	34.71	34.55	28.41	36.54	30.24	25.68	32.04
ml22	Specific mean annual minimum flow	m^3/s/km^2	0.01	0.01	0.01	0.01	0.01	0.01	0.01
fl1	Low flood pulse count	Events/year	8.83	9.42	8.92	10.58	8.00	7.08	9.58
fl2	Variability in low pulse count	%	54.31	26.73	22.79	28.28	31.06	28.75	33.35
fl3	Frequency of low pulse spells	Events/year	0	0.25	0.00	1.00	0.00	0.00	0.00
dl1	Annual minimum daily flow	m^3/s	0.55	0.37	0.34	0.39	0.36	0.33	0.38
dl2	Annual minimum 3-day flow	m^3/s	0.57	0.38	0.35	0.40	0.37	0.34	0.40
dl3	Annual minimum 7-day flow	m^3/s	0.59	0.41	0.38	0.43	0.41	0.38	0.43
dl4	Annual minimum 30-day flow	m^3/s	0.71	0.74	0.72	0.76	0.63	0.62	0.65
dl5	Annual minimum 90-day flow	m^3/s	1.2	1.42	1.42	1.43	1.26	1.25	1.26
dl6	Variability of annual min daily average flow	%	34.71	34.55	28.41	36.54	30.24	25.68	32.04
dl7	Variability of annual min of 3-day moving average flow	%	34.47	34.33	28.73	36.57	30.78	26.20	32.40
dl8	Variability of annual min of 7-day moving average flow	%	32.32	33.64	28.99	35.32	32.85	28.78	34.24
dl9	Variability of annual min of 30-day moving average flow	%	26.21	38.73	37.76	39.59	33.41	32.55	34.11
dl10	Variability of annual min of 90-day moving average flow	%	32.89	28.57	28.49	28.71	31.29	31.13	31.49
dl11	Annual min daily flow/median for entire record	-	0.32	0.16	0.15	0.18	0.15	0.14	0.16
dl12	Annual min 7-day moving average/median entire record	-	0.34	0.18	0.17	0.20	0.17	0.16	0.18
dl13	Annual min 30-day moving average/median entire record	-	0.41	0.33	0.32	0.34	0.26	0.25	0.27
dl14	Low exceedence flows (75% Exceedence)	-	0.56	0.54	0.52	0.57	0.48	0.46	0.50
dl15	Low exceedence flows (90% Exceedence)	-	0.35	0.32	0.30	0.34	0.29	0.27	0.30
dl16	Low flow pulse duration (events below 25th percentile)	#days	9.25	9.35	8.09	9.89	11.53	9.58	13.50
dl17	Variability in low pulse duration	m^3/s	64.54	37.55	31.41	41.03	32.97	29.71	35.99
tl1	Julian date of annual minimum	-	197	175.00	174.00	176.00	57.67	53.00	59.00
tl2	Variability in date of annual minima	m^3/s	58.88	39.07	38.84	39.50	51.31	50.21	53.65

6.4.8 *Lake Levels*

Under all the modelled natural outlet scenarios the average surface water level of Ennerdale Water would decrease. This reflects the changing form and location of the lake outlet in response to a sudden change in gradient. As demonstrated by the geomorphic modelling in Chapter 5, the upper Ehen would cut upstream into Ennerdale Water and over time reduce the gradient of the channel. This would also result in a narrowing of the channel when compared to the initial outlet profile designs. As a demonstration of the impact of outlet width on lake level, stage duration curves were created comparing three outlet profiles of the same initial elevation but with varying width (Figure 127). For the widest outlet lake levels were modelled consistently lower than the narrower outlets due to the larger conveyance capacity of the former outlet. The narrowest outlet, Profile 1, maintained the highest values at the lower end of the stage duration curve as water would be released downstream more slowly due to the lower conveyance capacity.

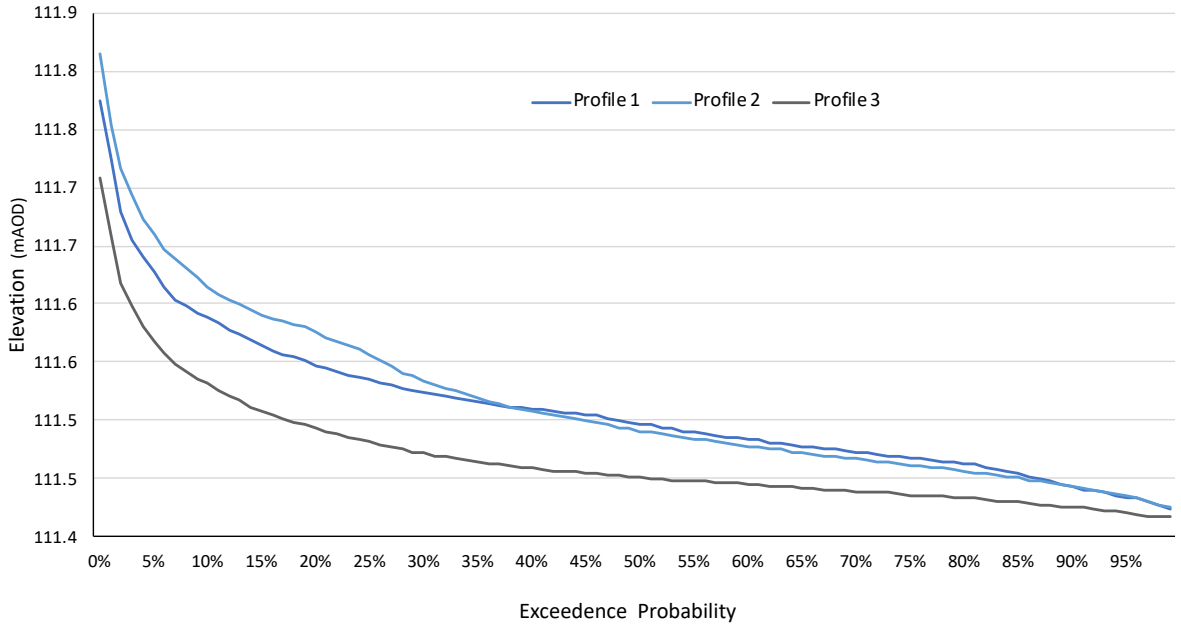


Figure 127. Stage duration curve for lake outlet Profiles 1,2 and 3 using modelled inflow data from 1995 – 2019 to demonstrate the impact of outlet width on lake stage. Profiles1, 2 and 3 have initial widths of 14m, 18m and 26m respectively.

In addition to a lower average surface water levels, the modelling results show that the range of observed water levels will be reduced (Figure 128). Under the current regime, water levels are both artificially increased through the impounding infrastructure and drawn down beyond their natural limit as a result of abstraction.

During the observed period 1995-2019, surface water levels in Ennerdale Water reached a maximum of 113.39 mAOD and a minimum elevation of 111.26, which is a range of 2.13m. The average water level during this period was 112.32 mAOD which is 0.01m above the crest of Ennerdale weir. The final outlet profiles from the 30-year geomorphic modelling runs were used to generate discharge and lake level data, for the same period, 1995-2019. The average lake level across all model runs reduced to 111.37 mAOD, which is a decrease of 0.95m. The maximum surface water elevation obtained across all model runs was 112.42 mAOD and the minimum was 111.24 mAOD, which is a range of 1.31m.

Using stochastic inflow data to generate a timeseries for two lake outlets (2a and 3) the minimum and maximum elevations were 111.9 mAOD and 111.08 mAOD respectively observed in outlet 2a. The average lake levels were approximately 111.35 mAOD in Profile 2a and 111.36 mAOD in Profile 3.

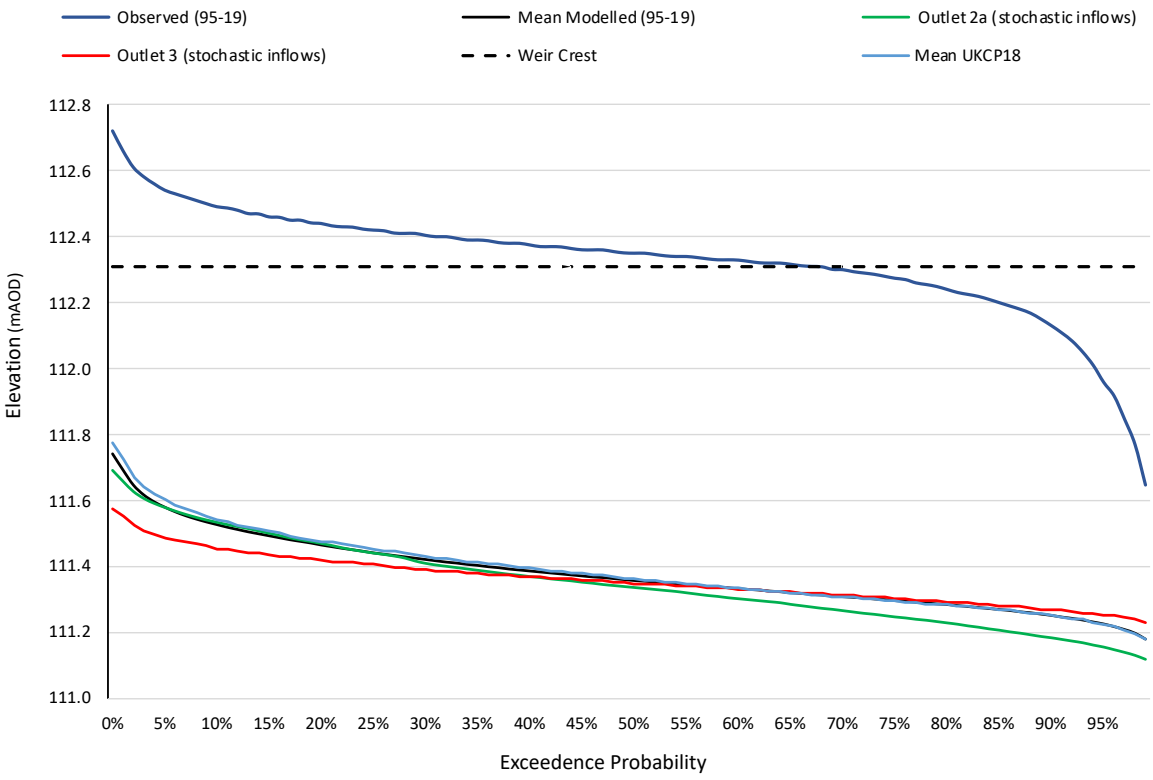


Figure 128. Stage duration curves under for all modelled scenarios.

Based on its current average water level Ennerdale Water has a surface area of approximately 3.04 km^2. This would reduce to around 2.89 km^2 based on an average water level of 111.37 mAOD. This equates to reduction in size of 0.15 km^2 or a 4.9 % reduction.

6.4.9 *Summary*

Chapter 6 has presented a range of model scenarios for a multiple flow alteration scenarios. The comparison of previous inflow timeseries with the newly generated SHETRAN inflows provides new information on the hydrological behaviour of the upper Ehen catchment whilst UKCP18 data provides an insight into hydrological changes under a high-emission scenario.

Twelve initial engineered outlet profiles and twelve modelled outlet profiles provide information on the sensitivity of the lake and downstream hydrology to lake outlet form and show how the hydrology may change in response to an evolving lake outlet.

Mean lake inflows generated by the SHETRAN model do not differ greatly to inflows previously generated by the Environment Agency for the period from 1995-2012. The main variation observed is in the variation between monthly flows, with the SHETRAN model producing greater seasonal variation.

Maximum flow values are also greater in all months using the SHETRAN inflows. For the two-year observation period, described in Chapter 3, there is substantial evidence to suggest the modelled flows are more representative of the range of hydrological behaviours experienced in the Upper Ehen Catchment.

Under all of the natural outlet scenarios, minimum flows values are lower than under the current managed regime due to the absence of compensation flows and flow regulation. On average the number of days between low flow events increases slightly under a natural flow regime. Impounding infrastructure has historically prevented low flow pulses reaching downstream. Lake levels need to be near weir crest in order for changes to the inflow hydrograph to be observed downstream.

Lake levels show a decrease in average elevation which is directly proportional to the changing elevation of the lake outlet. Under all scenarios the average lake level reduces considerably: however the range of lake levels also reduces. This is likely to have important implications for the freshwater ecology of the lake as it will provide a more stable environment. This is a potentially important implication for the wider integrity of the SSSI in Ennerdale Water. Modelling indicates the surface area of the lake would be expected to decrease by approximately 5%.

Chapter 7 - Discussion

This chapter discusses the key findings from Chapters 3 to 6. The significance of the findings for the site-specific case study and for the wider scientific and engineering community is outlined. The original objectives set out in the introductory chapter form the basis of the structure for this chapter. These were to:

1) Characterise the flow regimes of UK freshwater pearl mussel rivers.
2) Estimate the natural flow regime of the Upper Ehen catchment and quantify the level of alteration from the current managed hydrological regime.
3) Compare the current and future flow regime of the River Ehen with the flow regime preferences of freshwater mussels.
4) Characterise substrate properties of the outflow at Ennerdale Water, understand the natural geomorphology of the lake outlet and estimate geomorphic response to weir removal.
5) Quantify the sensitivity of lake storage and the downstream flow regime to changes in the morphology of the lake outflow.
6) Discuss the implications of the above findings for the future management of Ennerdale Water and the Upper Ehen catchment and in the wider context of ecological restoration of aquatic environments.

In addition to these objectives, the wider socio-economic implications and considerations for weir removal projects are discussed and a business case for implementing the results of this research is presented.

7.1 Characterisation of flow regimes in the UK's freshwater pearl mussel rivers

The characterisation of the UK freshwater pearl mussel rivers was carried out and discussed in Chapter 2. This is referred to in subsequent sections of this chapter along with a more detailed analysis of the hydrological metrics identified in Chapter 2.

7.2 The Upper Ehen natural flow regime

The two main hydrological features of the Upper Catchment are the River Liza and Ennerdale Water. Inflows into the lake have been previously modelled but observational data was limited to a handful of instantaneous flow measurements taken from various locations within the River Liza. However, bathymetric surveys of the Ennerdale Water and monitoring of the lake level provide useful additional information about the volumes of water entering and leaving the lake under the managed regime.

The neighbouring River Irt catchment (Figure 129) provides a useful point of comparison with the modelled natural discharge of the Upper Ehen. As geographical neighbours, both catchments have very similar physical properties and are subject to similar climatic conditions. Both catchments drain the same surface area, and both have lakes that attenuate discharge.

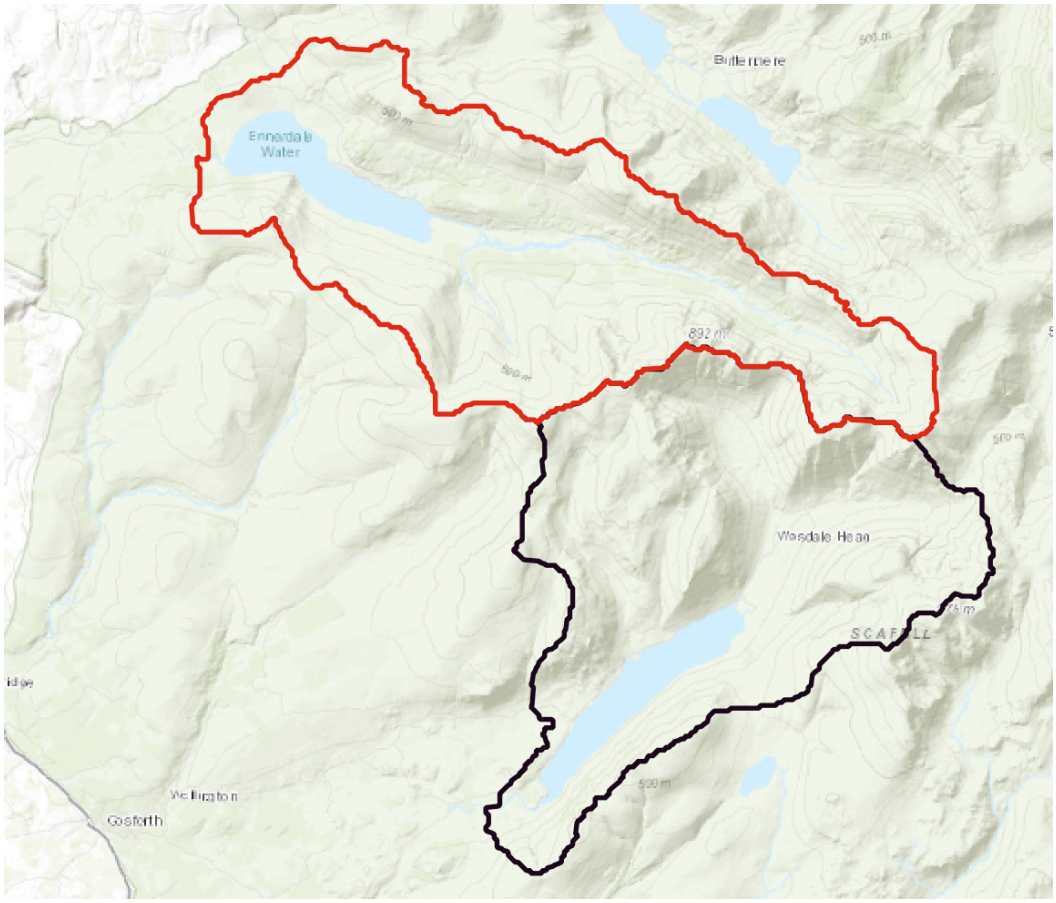


Figure 129. Upper Ehen (red) and River Irt (black) catchment boundaries with topographic detail

Using the 2017-2019 period for comparison the modelled flows for the Ehen behave in a very similar manner to the observed flows in the River Irt (Figure 130). Average flows in the Irt are slightly higher than in the Ehen and despite having a higher average channel gradient than

the Upper Ehen, the recessional limb of the hydrograph is typically less steep. This may be due to a combination of both the shape of the drainage basin (Figure 129), and the attenuating properties of Wast Water

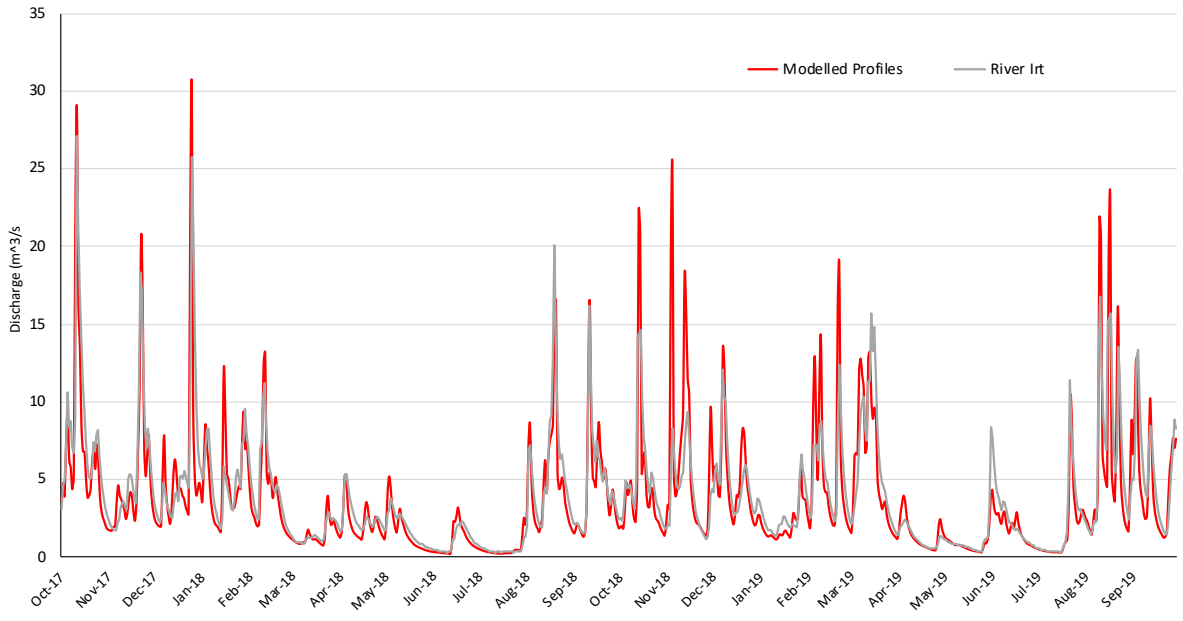


Figure 130. Mean modelled flows for the Upper Ehen (red) catchment compared to observed flows for the neighbouring River Irt catchment(grey).

7.2.1 *Assessing the Impact of Impoundment and Abstraction on Mean Flows*

Flow regulation can alter flow regimes in both consistent and unique ways according to how the weirs and dams are operated in a variety of ways. Furthermore, local topographical, geomorphological and biological variations mean that hydrological alterations may manifest themselves in a different ways (Rolls and Bond, 2017).

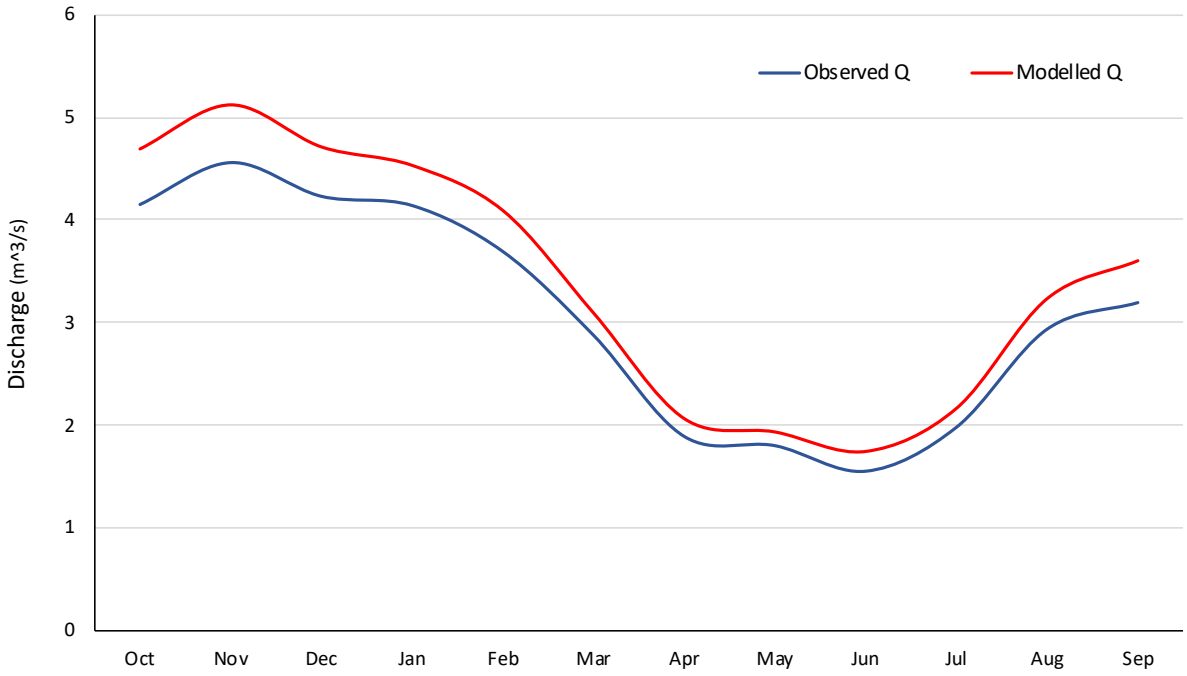
Flow regulation and abstraction have impacted the hydrology of the Upper Ehen for more than 150 years. However, regular and sustained hydrological monitoring only began in the 1970s and this was limited to flows downstream of Ennerdale Water.

Water abstraction has steadily declined during the 1995-2019 period. Initial abstraction rates in the 1995-1996 hydrological year were around 0.52 $m^3/s/d$ or a daily volume of 44.93 Ml/d. This has fallen to a present-day average of 0.22 m^3/s or an equivalent volume of 19 Ml/d. The mean abstraction over this period is 0.32 m^3/s (27.65 Ml/d). This is reflected in the modelled natural flow regime, which estimates a mean daily flow rate of 3.41 m^3/s compared to the observed value of 3.08 m^3/s.

Abstraction remains relatively constant throughout the year, with monthly average rates varying between 0.34 and 0.32 m^3/s. The effect on the average monthly hydrograph is slightly

less evenly distributed. The greatest changes to mean monthly flows under a natural lake outlet with no abstraction are shown to occur in June, September and October with a 13% increase compared to the historic average. The smallest increases are observed in March and May with flows increasing on average, by 8%.

Based on the reconstruction of a natural flow regime for the 1995-2019 period, the range in monthly mean average flows would increase slightly from the current managed regime. The range in average monthly flows is currently 4.56 m^3/s in November to 1.54 m^3/s in June; a range of 3.02 m^3/s. under a natural regime this is predicted to increase, with the model results predicting mean monthly flows of 5.12 m^3/s and 1.74 m^3/s in November and June respectively; a range of 3.38 m^3/s.



	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Annual
Observed Q (m^3/s)	4.15	4.56	4.23	4.14	3.69	2.87	1.88	1.79	1.54	1.97	2.93	3.19	**3.08**
Modelled Q (m^3/s)	4.69	5.12	4.71	4.53	4.09	3.09	2.06	1.93	1.74	2.16	3.23	3.60	**3.41**
% Change	13%	12%	11%	9%	11%	8%	10%	8%	13%	9%	10%	13%	**11%**

Figure 131.Mean monthly flows changes from historic to modelled natural flows.

In Chapter 2, a number of flow regime metrics describing the average flow conditions were identified as distinguishing features between rivers sustaining healthy population of freshwater mussels and those with struggling populations (Section 2.4.2). These metrics were MA11 (Spread in daily flows), MA36 (variability across monthly flows) and MA42 (variability across annual flows).

Metric MA11 is calculated as a ratio of the difference between the 75th and 25th percentile of the logs of the flow data and the log of the median flow for the entire flow record (Henriksen *et al.*, 2006). A larger number indicates greater range in daily flows from the minimum to maximum discharges. Under a natural flow regime, the spread in daily flows would be lower than under the current managed regime. For the observed 1995-2019 period, the score for this metric was 2.52 compared to 1.55 under natural conditions (Table 27). A direct comparison with the average for UK mussel rivers as assessed in Chapter 2 is not possible as, due to the availability of flow data, the lengths of the analysed time series differ. However, the findings suggest that a lower spread in daily flows is associated with healthier mussel populations. This reduced spread is an indication of a less changeable flow regime which implies that under a regime with no abstraction and a natural lake outlet the range of daily flows in the Upper Ehen would be expected to be less extreme. The variability of monthly flows is also predicted to be lower under a natural or unmanaged flow regime despite the range of mean monthly flows having been shown to increase. Metric MA36 is calculated by subtracting the minimum monthly flow from the maximum monthly flow and dividing this by the median monthly flow (Henriksen *et al.*, 2006). This decrease from 4.12 under the managed regime to 3.54 for the estimate natura flow regime. Again, indications from the national-scale study suggest that the healthiest mussel populations are found in rivers where monthly flow variability is lower.

Table 27. Flow regime metrics identified in the UK-wides study as distinguishing between healthy and failing mussel populations. Modelled values for the River Ehen are obtained from natural outlet profiles generated in Chapter 5.

Metric	Units	Mean Values UK Rivers				River Ehen	Modelled		
		NV	V	NV - N	V - N	Observed	Mean	Max	Min
DH13	-	5.5	3.9	5.7	4.02	4.45	3.55	3.71	3.41
DH19	days	1.75	1.51	1.57	1.4	1.5	1.61	1.77	1.44
DL9	%	38.3	28.3	34.6	35.1	26.21	38.73	39.59	37.76
MA11	-	1.11	0.59	1.49	0.66	2.52	1.55	1.70	1.44
MA36	-	5.05	3.84	5.24	3.67	4.12	3.54	3.54	3.53
MA42	-	0.57	0.48	0.57	0.48	0.88	0.63	0.63	0.63
MH16	-	4.36	3.4	4.37	3.67	4.07	3.27	3.50	3.14
MH22	days	10.58	5.77	9.58	5.79	8.19	4.88	5.19	4.43
ML13	%	71.8	56.1	68.4	60.6	53.37	62.35	65.06	59.68
ML22	m^3/s/km^2	0.004	0.007	0.004	0.005	0.012	0.008	0.009	0.008
RA5	-	0.32	0.36	0.32	0.35	0.36	0.34	0.35	0.34

The main reasons behind the reduction in this ratio are twofold. Firstly, the median flow value in the absence of abstraction is predicted to increase. Data for 1995-2019 show an observed median flow value of 2.74 m^3/s compared to a modelled median value of 3.07 m^3/s. Secondly,

the maximum mean monthly flow value as predicted by the model is 10.9 m^3/s compared to an observed value of 11.6 m^3/s. There are several possible causes for this variation. Firstly, the model may be underrepresenting peak flows during this month either because of the rainfall data or the way the model was set up. The model was calibrated to the lower flow end of the hydrograph so this may also be a factor. Another potential cause of the variation is due to the flow gauging infrastructure at Bleach Green. Flows above 38 m^3/s or a river stage of 1.5m are extrapolated and therefore subject to uncertainty. Whilst the model slightly underestimated the maximum mean monthly flow, the maximum monthly flow in both the modelled and the observed timeseries occurred in October 2008. The second and third highest mean monthly flow values also coincided with the corresponding months in the observed timeseries.

The final average flow metric identified in the national-scale study was metric MA42, a measure of annual flow variability. This is calculated in the same way as metric MA36 using annual values instead of monthly values. Both the modelled and observed minimum annual flows occurred in the hydrological year 1995-96. The observed mean flow for this year was 1.58 m^3/s compared to an estimated natural flow of 2.15 m^3/s. This variation is due to abstraction, which averaged 0.52 m^3/s that year. The maximum mean annual flow in the observed time series of 4.19 m^3/s occurred in the 2015-16 hydrological year. The modelled natural flow for this year was 4.22 m^3/s and the modelled maximum occurred in 2011-12 with an estimated average flow of 4.28 m^3/s. During the observed period, abstraction has steadily decreased and combined with compensation flow regulation this has helped reduced the minimum mean annual flows. Climate is an obvious additional factor affecting low flows as despite compensation flow regulation the lowest flows are likely to occur in the months with the lowest rainfall. The four driest years, based on mean flow across the year, coincide exactly between the observed and modelled data. However, the influence of abstraction means that the minimum flows are lower in the observed data compared to corresponding modelled data.

7.2.2 *Impact of Impoundment and Abstraction on Low Flows*

In the absence of flow regime management and with a natural lake outlet, minimum flow values are predicted to be considerably lower than under the current regime. Despite this, the predicted Q_{95} under a natural hydrological regime is predicted to remain the same at 0.5 m^3/s. This is since there will be more water overall in the system in the absence of abstraction. Given the same inflow conditions, periods of low flows under a natural regime would be shorter.

The absence of impounding structure would mean that inflow pulses into Ennerdale Water would be translated to the downstream discharge timeseries. This is evident when comparing the hydrographs of modelled natural flows against observed natural flows (Figure 130).

Low flow pulses and low flow variability serve an important ecological function. Studies in rivers where baseflows have been artificially elevated have shown that this can have negative ecological consequences. For example, Ponsatí et al., (2015) demonstrated that low flow stability increased the metabolic activity of biofilms and chlorophyll productivity in Mediterranean rivers. In the Upper Ehen, surveys of benthic algal communities have been undertaken since 2012 as biomass accumulation on the river substrates and freshwater mussel beds was an area of concern. The cyclical nature of this accumulation in autumn has resulted in a number of hypotheses for the unexpected pattern (i.e higher accumulation after summer than in summer). The link between the stable and artificially elevated baseflow may therefore be an important contributing factor to this. Additionally, the lack of low flow pulses that can serve to cleanse substrate and remove biofilm accumulation by physical means may further contribute to the observed patterns of algal growth.

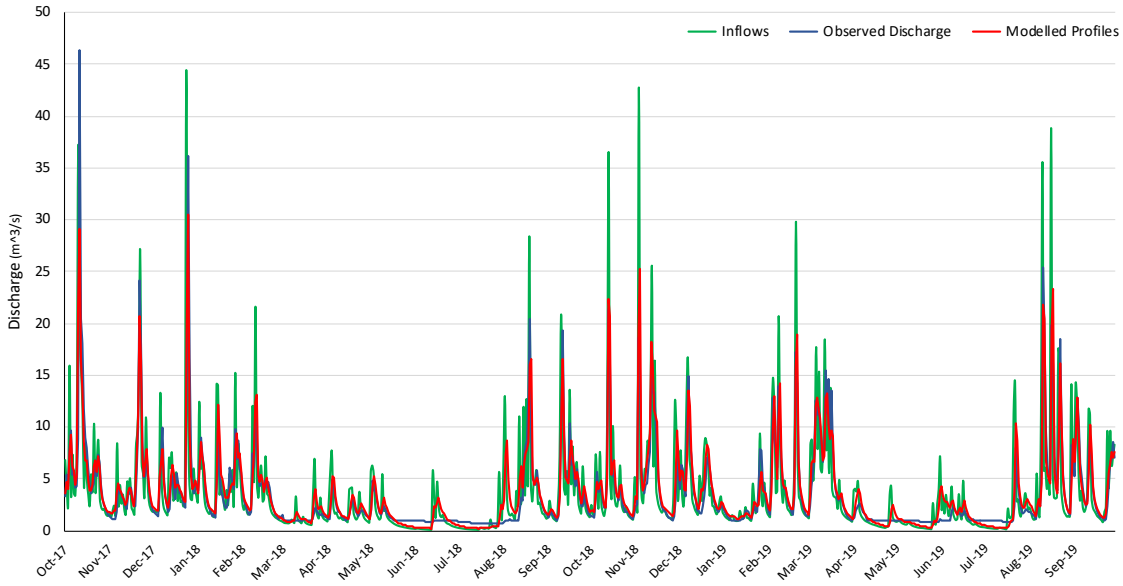


Figure 132. Modelled inflows and modelled vs obrved discharges for the 2017-2019 period.

Minimum 30-day moving average flow

Low flow metrics identified in the national-scale study were DL9, ML13 and ML22. Metric DL9 describes the variability of the annual minimum 30-day moving average flow. This is obtained by calculating the standard deviation of the minimum 30-day moving average across all years. This is then multiplied by 100 and divided by the mean. The results for this metric in the national-scale study were mixed. When looking solely at rivers with natural or

unmodified flow regimes the scores for this metric were very similar between the healthy and failing mussel populations (Table 27). However, the difference between rivers with modified flow regimes was more marked. Mean values ranged between 38.3 and 28.3. In the River Ehen, the variability of the annual minimum 30-day moving average flow under a managed regime obtained a score of 26.1, as compared to 38.7 under natural conditions. This change can be attributed, in part, to the absence of compensation flow regulations that have provided a stable minimum flow in more recent years. However, it doesn't evidence that annual moving 30-day minimum flow will be consistently lower than present. Removing abstraction is likely to increase minimum flows except for more extreme periods of extended dry weather and drought. Such conditions occurred in both 2018 and 2019 and the impact of compensation flow regulation is clearly evident on the hydrographs. However, whether the lack of pulse flows during these periods is more detrimental to downstream ecology than lower flow values interspersed with pulses requires further investigation. The ecological impacts of this type of flow regime change are difficult to monitor in long-lived species such freshwater mussel and requires in-depth analysis from specialists with the appropriate level of expertise. Nevertheless, the information gathered here provides further useful evidence when deciding on whether to proceed with weir removal and what if any additional mitigating measures may need to be implemented to minimise the risk of these changes. The implications of these flow regime changes for the future restoration of the River Ehen are discussed further in section 7.4.

Minimum monthly flow

Metric ML13 provides a measure of variability across minimum monthly flow values. ML13 is calculated in the same way as DL9 but using fixed time periods of calendar months rather than a moving 30-day average. Although the metrics are similar, the correlation analysis of the entire set of metrics did not show a high level of correlation between the two. The variability of minimum monthly flows is predicted to be higher under a natural flow regime changing from 53.37 to 62.35. Again, this is due to the reasons outlined for the changes to metric DL9. The range of mean values for the wider pool of catchments in the UK were between 71.8 and 68.4 for failing mussel populations and 60.6 and 56.1 for the healthier populations suggesting the lower variability provides more suitable conditions for freshwater mussels. However, it should be noted that the lowest scores for this metrics were observed in rivers with regulated flow regimes and that these metrics were calculated from shorter timeseries.

Average annual minimum flows

The final low flow metric, ML22, is a measure of specific mean annual minimum flow. This is calculated by obtaining the mean annual minimum flows and dividing this by the catchment area (Henriksen *et al.*, 2006). Under the managed regime the mean annual minimum flow between 1995 and 2019 was 0.55 m^3/s. Under a natural flow regime this is estimated to be 0.38 m^3/s. Dividing these values by the catchment area (44.5 km^2) produces specific mean annual discharges of 0.012 $m^3/s/km^2$ and 0.008 $m^3/s/km^2$ respectively. Whilst flow management has ensured this value remains high, the discharge under a natural flow regime is still higher than the average for the national-scale study. This reflects the high average rainfall for the catchment and flow attenuating properties of Ennerdale Water.

Minimum annual flows range from 0.35 to 0.98 m^3/s under the managed regime across the 25-year observation period. This reduces to 0.19 and 0.65 m^3/s under natural flow conditions. This shows that although minimum flows will be lower the range of minimum flows will be reduced and there is slightly less inter annual variation in minimum flows. Whether this more consistent but lower range of flows is better for the downstream ecology is likely to depend on the lifecycles and habitat requirements of individual species.

7.2.3 *Impact of Impoundment and Abstraction on Peak Flows*

Low flows have been highlighted as being of particular concern to freshwater mussels for reasons outlined in Chapter 2 (Moorkens and Killeen, 2014). However, high flows play an important ecological role and drive mechanisms that can have adverse impacts on aquatic species and their habitats. Under all the modelled scenarios peak flows are generally lower than the observed peak flows. This is not the case for all individual high flow events, but the maximum daily and monthly flows are predicted to be lower based on the modelled natural discharge timeseries. This is somewhat unexpected as the literature widely reports the peak flow attenuating property of dams (Magilligan and Nislow, 2005; Aldrick *et al.*, 2009). However, Ennerdale Water and the impounding weir are different to the majority of dams and reservoirs for water storage as Ennerdale water is a natural lake only partially modified by the weir installation to increase storage. The impounding infrastructure, for the most part, functions as a run of river weir. Average lake level is slightly above weir crest meaning that when high flows are experienced the weir has no peak flow attenuating properties and it is only the attenuating properties of the lake which influence the flood hydrograph. At lake stages below weir crest, the infrastructure is able to dampen and even completely eradicate sharp rises in the downstream hydrograph. This is most evident after extended dry periods

when due to continual abstraction and compensation flow release lake level continue to fall below natural levels.

Hydrological metrics identified as distinguishing between healthy and failing mussel populations were DH13, DH19, MH16 and MH22. Metric DH13 describes the ratio between the mean annual maximum 30-day moving average and the median flow across the complete discharge timeseries (Henriksen *et al.*, 2006). Under a managed regime the River Ehen has a value of 4.45 (dimensionless). This is below the mean value for rivers in the UK with declining mussel populations; however, it is higher than the average for mussel rivers with healthy populations. Under a natural regime this value would decrease to 3.55 (dimensionless) which is below the mean for healthy mussel populations, suggesting that this particular component of the flow regime may provide more preferable conditions than those currently experienced in the Upper Ehen.

Hydrological metric DH19 is a measure of the duration of high flow events above a given threshold. In this instance a high flow event is classified as a flow seven times greater than the median flow value of the complete discharge time series (Henriksen *et al.*, 2006). The median flow value under the current managed regime is 1.74 m^3/s compared to 2.27 m^3/s for the natural time series, thus thresholds for the classification of high flow events were 12.18 m^3/s and 15.89 m^3/s respectively. The average duration of events under the managed regime was 1.5 days whilst the duration of flow events under the natural conditions was 1.6 days. This is somewhat unexpected as the threshold is higher for the metric under a natural flow regime so it could be predicted that these events would be less frequent or shorter in duration. However, it may be that the attenuating properties of Ennerdale Water are greater under natural conditions and thus high flows are released more gradually from the lake resulting in a shallower falling limb of the hydrograph.

Metric MH16 is a high flow discharge index which is calculated by dividing the 10 percent exceedance value by the median flow for the entire record (Henriksen *et al.*, 2006). The 10 percent exceedance value or Q_{10} for the River Ehen is 7.1 m^3/s based on the 1995-2019 observed data and 7.4 m^3/s according to the modelled natural time series. The values for this metric were 4.07 for the observed data and 3.27 for the natural discharge timeseries. The national scale study of freshwater mussel rivers suggests that a lower score for this metric is more conducive to sustaining healthy populations of mussels. The scores for the failing mussel populations for this metric were 5.5 in the rivers with modified flow regimes and 5.7 in the rivers with unaltered regimes compared to scores of 3.4 and 3.7 in the rivers with healthy mussel populations.

The final high flow metric highlighted for comparison was MH22. This measures the average volume of high flows above a threshold three times the median flow for the entire record. This volume is then divided by the median flow of the complete timeseries (Henriksen *et al.*, 2006). Lower values represent less change in the discharge hydrograph or a less flashy hydrological regime. Mean values for this dimensionless metric were 10.58 and 9.58 in modified and unmodified flow regimes of rivers with declining mussel populations studied in Chapter 2. This contrasted to values of 5.77 and 5.79 for healthy mussel populations in rivers with modified and unmodified flow regimes. The River Ehen in comparison has a value of 8.19 under a managed flow regime, however this reduces to 4.88 when calculated using the natural flow regime timeseries.

7.2.4 *Upper Ehen Natural Flow Regime Summary*

Analysis of standard hydrological statistics and the key metrics identified in Chapter 2 highlights several key differences between the managed and natural flow regimes for the River Ehen. Furthermore, comparing the modelled discharges with the neighbouring River Irt catchment provides a useful visual representation of how the catchment hydrology will behave after weir removal. As expected, minimum flow values are predicted to be lower under a flow regime lacking regulation. Despite a greater abundance of water in the system due to the lack of abstraction, flows from the lake outlet can no longer be controlled and thus continue to reduce in extended dry periods. Whilst this may appear a negative and potentially harmful consequence, these low flow periods play an important ecological role (Rolls *et al.*, 2012). The model predicts that maximum flows will be slightly reduced however the overall impact to Q_5 values is less pronounced. Mean monthly flows are broader across the year due to an increased availability of water in the system without abstraction.

Low flows have been highlighted as being of particular concern to freshwater pearl mussels however, it is important to consider the full range of hydrological alteration caused by weir removal. Focusing on one aspect of the flow regime or a specific hydrological variable such as Q_{95} may reduce the chances of restoring ecological integrity to the site (Magilligan and Nislow, 2005).

7.3 River Ehen flows and flow regimes in the UK's freshwater mussel rivers

Periods of low flows are of particular concern for the conservation of Freshwater Mussel and have been sighted as the main cause of juvenile decline in the River Ehen (Kileen and Moorkens, 2012). Extended low flows in summer create conditions that induce low levels of oxygen and heat stress (Skinner *et al.*, 2003). Conversely, research in the River Kerry, Scotland, has shown that juvenile mussel recruitment closely matched historic precipitation from 1957 to 1991 and that in general mussels recruit well during wet years (Hastie *et al.*, 2003a).

Absolute minimum flows are predicted to be lower than under the current managed regime due to the absence of flow regulation and compensation flow releases. However, the severity of extended low flow periods is predicted to diminish.

Both the mean flows of the 30-day minima and 90-day minima increase in a scenario without abstraction and impoundment. This would suggest the conditions of low oxygen and heat stress caused by low summer flows would be less severe and thus provide preferable conditions for juvenile mussels. However, this will depend on future climate conditions and whether extended dry periods become more severe and frequent or if future land use changes in the catchment impact upon the hydrology.

The River Irt provides a useful reference for flow regime comparison with the River Ehen due to is geographical proximity and similar physical characteristics. The Irt also has minimal flow regime alteration from abstraction and Wast Water has a natural outlet with no impoundment or flow regulation structures. The River Irt is approximately 22km long (West Cumbria Rivers Trust) compared to around 27km for the Ehen (West Cumbria Rivers Trust). Both catchments drain a similar area with steep topography and have mean daily flows of and respectively.

In terms of providing a comparison for identifying optimum conditions for freshwater mussels the Irt is less useful and further investigation is required to understand the large differences in mussel population between the two locations. Despite having a relatively unmodified flow regime, mussel populations in the Irt are extremely low, with as few as five individuals recorded during the last survey of the river. Furthermore, the discharge time series suffers from gaps in the data and for this reason it was not included in the national scale study presented in Chapter 2.

This focus of this EngD is flow regime and physical catchment characteristics and a full before-after-control-impact study beyond the scope of the project. However, expanding on the significance of the life cycle of the freshwater pearl mussel, consideration of the host species and wider ecosystem conditions upon which they depend is key. Further assessment of the predicted hydrological and geomorphic changes will therefore be required.

The flow regime data and hydrological metrics generated from this study will provide the project steering group with additional information upon which to base future management decisions.

7.4 Substrate Characteristics and Geomorphic Response

7.4.1 *Fine Sediment Volumes and Composition*

Fine sediment has been highlighted as a major concern for freshwater mussel species and for a variety of aquatic organisms. It is therefore an essential consideration for dam and weir removal projects. In a synthesis of common management concerns associated with dam removal, Tullos *et al.*, (2016) attempted to identify the most frequent and uncertain concerns. They found that contaminated sediment was a concern at a number of sites in their analysed pool of dam removals however it was recognised that this is most likely to occur in water bodies where historic activities may be conducive to contamination and fine sediments are prevalent (Tullos *et al.*, 2016). The Upper Ehen has some evidence of historic activities that may have led to sediment contamination; however, these are predominantly downstream of Ennerdale Water. The bleaching weir approximately 400m downstream of Ennerdale Water, was the site of fabric processing involving the use of dyes and chemicals. The Chlorine used in these processes is known to react with minerals and elements to form toxins including dioxins, furans and polychlorinated dibenzodioxins which can persist in water or sediment and breakdown very slowly (Alvarez-Codesal and Sweeting, 2015). Whilst there is no evidence of these processes being carried out at the lake outlet, the analysis of the fine sediment in this area helps to reduce concerns of this potential risk. Tullos *et al.*, (2016) state that water managers need to be able to confirm the sediments are clean and within the biologically and legally accepted limits.

Annual average total sediment volume recorded across all model runs was 24 m^3, however this was strongly linked with peak flow events and ranged between 0.01m^3 and 345 m^3.

In the model input data fine sediment (< 2 mm) accounted for 10% of the total sediment. Of this, 6.7% was composed of particle sizes below 2 mm and the remaining 3.3% being of sediment particles below 0.5mm.

This would appear to suggest that fine sediment represents a limited risk to the freshwater mussel population after weir removal. However, there is substantial uncertainty as to the true nature of the sediment composition across a wider area of the site due to the limited number of samples obtained in the field surveys. Furthermore, previous studies have highlighted initial pulses of fine sediment immediately after dam removal and sediment transport diminishing over time (Major et al., 2017). Fine sediments, if present, are likely to be more easily mobilised and thus make up are larger proportion of sedimented transported in the earliest stages of channel adjustments. These finer sediments commonly travel further downstream and in general, the initial spatial extent of downstream impact is more limited when sediments have higher concentrations of coarse material (Major et al., 2017).

The spatial extent and magnitude of sediment transport will also depend on what mitigation measures or engineering works are carried out in the event of weir removal. If present in higher concentrations than suggested by the available data, removing fines sediments from the area behind the weir and replacing them with coarse substrates more characteristic of unmodified reaches of the River Ehen could be one such option to minimise fine sediment risk.

The sediment data gathered in this study and ongoing research at the site provide an initial insight into to sediment characteristics at the site. However, further surveys to build on the existing data are recommended and these should fully explore the physical and chemical characteristics of any impounded sediments to assist with weir removal and mitigation planning (MacBroom and Schiff, 2013). Additional sediment samples from the surface complimented by deeper core samples would be advantageous if time and resources permitted. The geomorphic modelling results give and indication of the upstream area likely to undergo channel erosion which can be used as a guide when planning the spatial extent of these future studies.

7.4.2 *Geomorphic Evolution of the lake outlet and Upper Ehen Channel*

Characterising the geomorphology of a river and its floodplains is important as this represents the physical template for flow and its physical and hydraulic conditions, in turn helping drive the distribution and functioning of ecosystems and their biota (Vietz and Finlayson, 2017). Conversely, flow shapes the geomorphic characteristics or rivers and their floodplains and

thus the interaction between the two is in constant feedback. Results from the geomorphic modelling (5.7) show that the predominant channel response was the upstream movement of a knickpoint as the channel adjusted to a new longitudinal gradient. This has been demonstrated in both conceptual and numerical modelling studies of dam removal (Doyle *et al.*, 2002; M. and Crowe, 2012; Poeppl *et al.*, 2019). The average longitudinal retreat of the knickpoint across the twelve model runs was approximately 40m over a 30-year period. Highest erosion rates occurred in the first 6 months to 5 years of the model runs. Doyle et al., (2005) reported that, based on observation of dam failures and experimental releases of sediment, the major geomorphic adjustments after dam removal occur with the first 1-5 years. However in this study sediment transport loads were highly correlated to maximum annual flows, supporting the notion that channels predominately undergo major reshaping at bank-full discharges (Zeiringer *et al.*, 2018). Lateral movement is also an important consideration for geomorphic responses to dam removal (Poeppl *et al.*, 2019). Lateral channel migration occurred in all model runs with distance ranging from 4 m to 26 m across the 30-year period. Evidence from the Ground Penetrating Radar data (5.5.1) has shown signs of fluvial activity below the current lakebed at a location further north of the current outlet. Relic meanders downstream of the weir provide further evidence of the extent of channel migration in the upper section of the River.

After completion of the 30-year model runs it was possible to measure the gradients of the newly formed river channels. The average gradient across all 12 model runs was 0.008, ranging between 0.006 and 0.01. This number shows the increase in elevation in metres for every metre of longitudinal distance. This additional information in conjunction with projected rates of change and channel dimensions can serve as a useful starting point for planning the weir removal. Decisions will need to be made about whether to allow the River Ehen to naturally erode its way upstream into the lake or whether to engineer a channel and riparian area that represents an expected future natural morphology post weir removal and system response to that action.

Engineering a channel would have the benefit of removing volumes of sediment that might otherwise be rapidly flushed downstream with unknown consequences. Conversely it would also lead to a more sudden change in lake levels and hydrological regime which would have significant implications for the lake and downstream ecology. In any case, removing the weir at Ennerdale is likely to have an immediate and abrupt impact, so deciding on the best approach to managing this transition requires considerable thought. Evidence from the sediment transport volumes in the geomorphic modelling chapter also suggest that the

simultaneous removal of two additional downstream barriers will lead to considerable increases in downstream sediment loads. In some cases, the model predicts almost double the volumes of sediment being transport from the modelling domain over the 30-year period.

Research focussing on geomorphic restoration by Brierley and Fryirs (2009) proposed three principles that form the basis for rehabilitation planning;

1. *Respect geomorphic diversity—appreciate and work with the natural range of variability in river character and behaviour in any given catchment.*
2. *Differentiate the behavioural regime of a river reach (its capacity for adjustment) from the potential for the reach to change to a different type of river.*
3. *Analyse downstream patterns of rivers and landscape (dis)connectivity to interpret the trajectory of river adjustment and potential for geomorphic recovery of each reach within a catchment.* (Brierley and Fryirs, 2009)

Previous experience of restoration from the site has shown that designing channels and expecting to accurately predict the hydromorphological responses is not a straightforward task. The reconnection of Ben Gill in 2014, was carried out by designing a channel based on the historic location prior to its diversion. Immediately after its reconnection, an extreme rainfall event produced dramatic changes to the channel resulting in unintended sediment loads being transported downstream. The implications of a similar event occurring after weir removal would possibly be far more severe with the potential to transport even larger volumes of sediment downstream. Whilst lessons can be learned from this and other dam or weir removal projects it is necessary to consider the range of possible responses and their downstream implications whilst respecting the geomorphic diversity of the site.

Impoundments impact rivers in a number of ways over a range of temporal and spatial scales. Petts (1984) proposed 3 nested orders of impacts when describing the long-term effects of impoundment. In this classification first order impacts describe the effect on an environmental process (e.g. flow regime alteration, sediment changes); second order impacts are changes to form (channel changes or invertebrate population changes); and third order impacts reflect feedbacks between morphological and ecological change (Petts, 1984). Using this understanding of the impacts of impoundment and examples from observations of dam removal studies the following conceptual diagram of expected changes

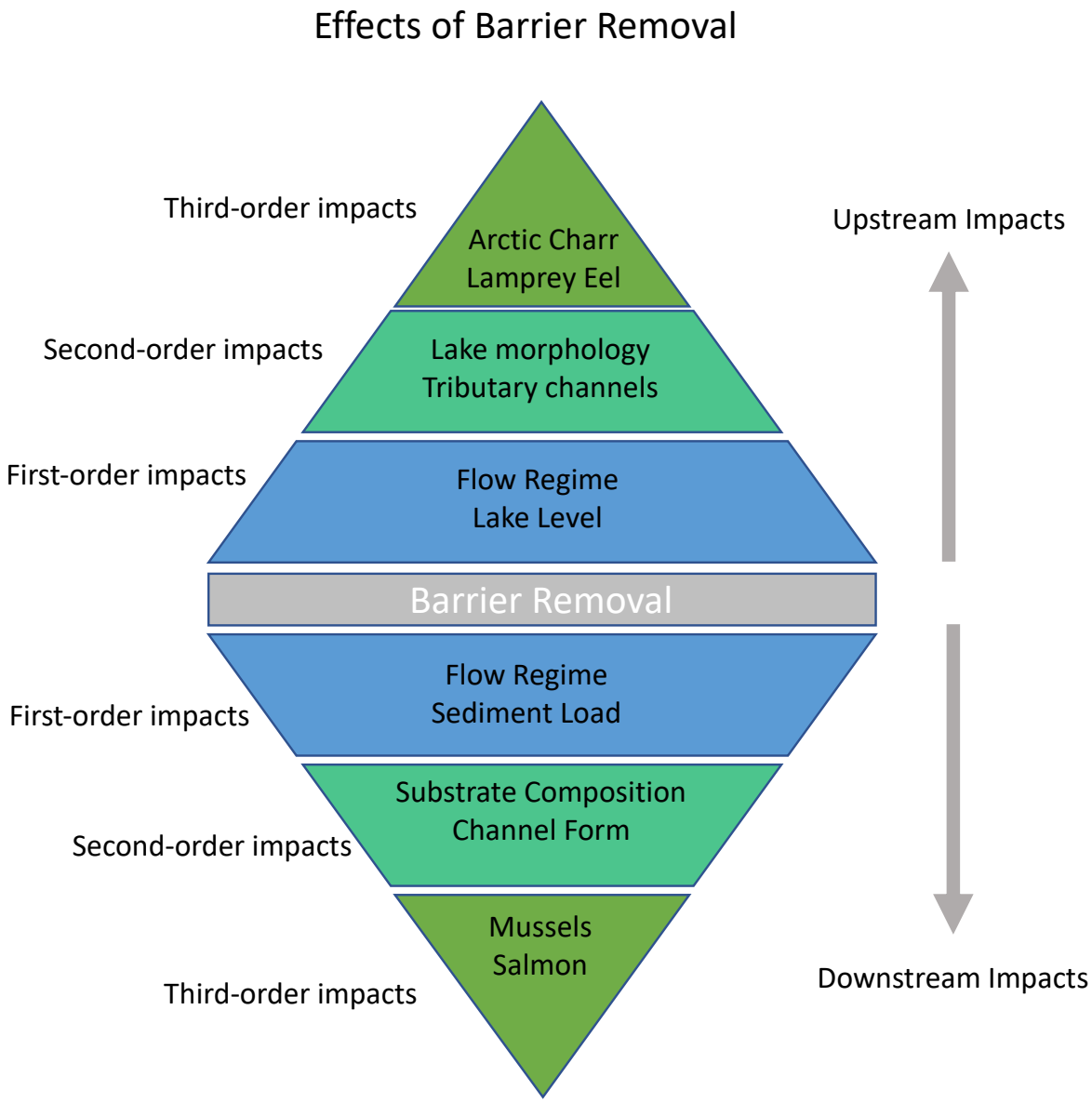


Figure 133. Effects of barrier removal conceptualised for Ennerdale Water and the River Ehen, adapting the model of Petts (1984). *Effects upstream are also expected to be seen for the River Liza, but the distance upstream is open to conjecture at this stage.*

Geomorphic impacts of weir removal will vary in space and time, with the impacts likely diminishing as both increase. In many cases, barrier removal results in the loss of the upstream water body, however in Ennerdale this will not be the case. Reductions to the lake level are therefore likely to have geomorphic impacts on the catchment over a wider spatial scale than might be observed in typical dam removals. The River Liza, the main inflow into Ennerdale Water is approximately a further 3.7km further upstream of the weir. The drop in mean lake level by almost 1m will expose new areas of shoreline resulting in corresponding adjustments to the lake tributary channels.

Both upstream and downstream changes will have impacts on the riverine ecology which in some cases will have feedbacks on the physical characteristics of the catchment. The rate and

type of vegetation colonisation around the lake will impact the geomorphic evolution of the newly exposed shoreline and lake outlet, whilst the mussel populations may respond to changes in the river substrate by moving upwards or deeper within the substrate column as conditions change. This will in turn impact localised bed hydraulics and geomorphic conditions in certain river reaches. First order impacts will be observed immediately after weir removal whilst second order impacts may be observed within the first year of barrier removal or within several decades. Third order impacts will have a considerable time lag in relation to first order impacts (Petts, 1984). In the case of freshwater mussels, this lag may be exacerbated due to the length of their life cycle and reproductive strategy.

Summary of Geomorphic Analysis and Modelling

Evidence from previous studies and the field data gathered as part of this study shows that fine sediment makes up a small proportion of the sediment in the Upper Ehen and the upper most sediment layer in the lakebed. Pockets of fine sediment are present in the man-made channels at the lake outlet, but analysis of their composition suggests they do not contain high levels of harmful substances. Their impact on downstream biota may however be considerable if released in a sudden or uncontrolled manner and thus mitigating this risk would be a desirable management option prior to removal of the weir.

The average final bed elevation of the modelled natural river channels at their most upstream extent was 111.08 mAOD. The average river channel gradient was 0.0076 across all 12 model runs, this is an important consideration for the site, as the analysis of the engineered channels has shown that gradient exerts a strong influence on the hydrological behaviour of the system. The predominant movement of the River Ehen channel is predicted to be the upstream migration of the channel knickpoint, however lateral movement will also occur. This information provides an indication of the final geomorphology of the lake outlet under natural conditions and thus helps to better inform the hydrological predictions and subsequent geomorphic feedback. However, it is important to emphasise the role of Ben Gill and its own evolution in the years after weir removal. Extreme high-flow events may alter the location of the confluence between Ben Gill, the River Ehen and Ennerdale Water with the potential to substantially influence the lake outlet form and evolution.

The conceptual understanding of responses to weir removal suggests that the largest changes will occur in closer spatial and temporal proximity to the weir however impacts are likely to be observed at wider catchment scale and potentially for several decades.

7.5 Lake outlet sensitivity

The following section provides an analysis of the sensitivity of the downstream hydrology to changes in the lake outlet form. This considers both the cross-sectional profile of the outlet channel and the gradient. Similarly, the impact of the aforementioned characteristics on the lake stage is also assessed.

7.5.1 *River Ehen Hydrological Sensitivity to Lake Outlet Form*

The model was initially tested with 12 designed lake outlets (Table 28). These varied in width, initial elevation and channel gradient. Channels with highest initial elevations had the highest channel gradients. Therefore, to directly compare the influence of channel shape on discharge the outlets were compared against outlets of the same initial elevation. Using channels 1,2 and 3 as examples it is evident that minimum flows were highest in the widest channel with no variation in channels 1 and 2. However, the Q_{95} was marginally higher in the narrowest channel design and gradually decreased from 0.47 m^3/s in profile 1 to 0.45 m^3/s n profile 3.

Table 28. Impact of initial outlet design on downstream hydrology for selected metrics.

	Units	Observed	1	1a	1b	1c	2	2a	2b	2c	3	3a	3b	3c	Mean
Max	m^3/s	80.20	63.04	63.67	63.54	65.09	63.88	63.56	64.35	65.27	63.59	63.74	64.66	64.22	64.05
Min	m^3/s	0.35	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.19	0.32	0.32	0.35	0.45	0.22
Q95	m^3/s	0.51	0.47	0.47	0.47	0.55	0.46	0.48	0.46	0.46	0.45	0.45	0.44	0.45	0.47
Q70	m^3/s	1.00	1.29	1.29	1.28	1.27	1.31	1.29	1.31	1.33	1.25	1.30	1.23	1.22	1.28
Q50	m^3/s	1.74	2.17	2.13	2.11	2.17	2.16	2.08	2.16	2.06	2.02	2.05	2.01	1.96	2.09
Q10	m^3/s	7.09	7.94	7.80	7.85	7.94	7.42	7.92	7.87	7.55	8.12	7.87	7.30	8.84	7.87
Q5	m^3/s	9.53	10.71	10.67	11.18	10.96	10.37	10.63	10.78	10.76	10.65	10.83	10.44	11.03	10.75

Initial outlet design had a limited impact on maximum flows across all designs, including when considering channel slope. Maximum flows ranged from 63.04 m^3/s to 65.27 m^3/s. The Q_5 values varied somewhat with a minimum value of 10.37 m^3/s and a maximum value of 11.18 m^3/s.

Under the modelled natural outflow forms the range of maximum flows varied more markedly (Table *29*). Profiles generated from the DEMs of modelled runs 1-1c are narrower and have steeper banks which form a more V-shaped profile whilst the remaining profiles have wider more U-shaped profiles.

With a more V-shaped lake outlet profile, maximum flows are significantly reduced however the Q_5 values are very similar across the 12 model runs. The mean Q_5 values are also lower

under natural outlet profiles than the initial designed outlets. Minimum flow values show less variability with natural channel outlets, ranging from 0.17 m^3/s to 0.21 m^3/s.

A one sample t-test was used to assess the difference between the observed values from the historic time series and the mean of the modelled results for Min, Max, Q95, Q7, Q50, Q10 and Q5. These metrics provide representation across the full range of flows (low, average and high).

Table 29. Impact of natural outlet form on downstream hydrology for 12 modelled natural outlet profiles

	Units	Observed	1	1a	1b	1c	2	2a	2b	2c	3	3a	3b	3c	Mean
Max	m^3/s	80.20	30.19	31.04	22.73	28.10	60.75	60.76	61.71	61.91	61.34	61.99	61.29	61.77	50.30
Min	m^3/s	0.35	0.19	0.19	0.21	0.20	0.20	0.18	0.19	0.18	0.17	0.17	0.17	0.19	0.19
Q95	m^3/s	0.51	0.51	0.52	0.57	0.54	0.51	0.54	0.52	0.53	0.49	0.51	0.51	0.53	0.52
Q70	m^3/s	1.00	1.36	1.43	1.41	1.41	1.35	1.50	1.42	1.43	1.34	1.37	1.42	1.47	1.41
Q50	m^3/s	1.74	2.29	2.26	2.21	2.34	2.18	2.28	2.36	2.24	2.17	2.24	2.28	2.35	2.27
Q10	m^3/s	7.09	7.48	7.41	7.45	7.44	7.47	7.19	7.55	7.49	7.62	7.33	7.17	7.38	7.41
Q5	m^3/s	9.53	10.04	9.70	10.07	9.99	10.28	9.71	10.03	10.02	10.41	9.66	9.87	9.63	9.95

In the case of Q95, the results from the one sample t-test showed that no significant change occurred between the observed values and the modelled flows with natural outlet profiles. The Q_{95} values vary range between 0.57 m^3/s and 0.49 m^3/s under the 12 different natural outlet profiles. However, the mean of these values is not significantly lower than the under the managed regime. The differences between the observed maximum, minimum, Q70, Q50, Q10 and Q5 and the modelled values were all statistically significant, return P values below 0.0001 in all cases.

Although, minimum flows are predicted to be lower, however these flow conditions will only be of concern for short periods of the year. Furthermore, evidence from the hydrographs (Figure 132) shows that periods of lows flow which occur after extended periods of low rainfall will be interspersed with low flow pulses which would otherwise not be observed with the presence of infrastructure. Whether this increased variability of low flows under a natural regime will have a negative or positive impact on the downstream ecology remains to be seen and should only be one of a number of considerations when assessing the overall impact of removing the weir.

7.5.2 *Lake Stage Sensitivity to Outlet Form*

Stage duration curves (SDCs) were plotted for the initial outlet designs to assess the sensitivity of the lake level to outlet design. These artificial outlets provided a starting point for the individual geomorphic model runs but also serve to compare lake level responses. The largest range of lake stages were observed in model runs using outlets 1, 2 and 3. These have a minimum elevation of 111.4 mAOD and thus a shallower downstream gradient. In profile 1,

lake stages maximum and minimum stages varied by 0.92m (Table 30). This compared to 0.9m and 0.84m for profiles 2 and 3 which had wider outlet channels.

Comparing the influence of channel slope on the lake level was possible by comparing the stage response of the sets of four profiles with the same cross-sectional profile but different initial elevations (e.g. 1, 1a, 1b, 1c). This shows that across all model runs, an increased downstream channel slope leads to a decrease in lake stage variation.

Table 30 Range of lake stages for initial and natural modelled outlet profiles generated from 1995-2019 inflow data

Outlets	Initial Outlets				Natural Outlets				Change	
	Maximum	Minimum	Mean	Range	Maximum	Minimum	Mean	Range	Lake Level	Range
	mAOD	mAOD	mAOD	m	mAOD	mAOD	mAOD	m	m	m
Profile 1	112.34	111.42	111.51	0.92	111.79	111.17	111.34	0.62	-0.25	-0.30
Profile 1a	112.38	111.52	111.60	0.86	111.80	111.11	111.31	0.69	-0.40	-0.18
Profile 1b	112.40	111.61	111.70	0.78	111.78	111.19	111.41	0.59	-0.43	-0.19
Profile 1c	112.49	111.71	111.80	0.78	111.78	111.16	111.36	0.62	-0.55	-0.16
Profile 2	112.32	111.42	111.51	0.90	112.24	111.19	111.38	1.05	-0.23	0.14
Profile 2a	112.33	111.52	111.61	0.81	112.18	111.13	111.36	1.05	-0.38	0.23
Profile 2b	112.41	111.62	111.70	0.79	112.33	111.20	111.40	1.12	-0.41	0.33
Profile 2c	112.48	111.72	111.78	0.77	112.31	111.19	111.38	1.13	-0.53	0.36
Profile 3	112.26	111.42	111.47	0.84	112.19	111.25	111.37	0.95	-0.17	0.11
Profile 3a	112.29	111.52	111.57	0.77	112.28	111.22	111.37	1.06	-0.30	0.29
Profile 3b	112.26	111.62	111.63	0.65	112.33	111.18	111.37	1.14	-0.43	0.49
Profile 3c	112.41	111.72	111.75	0.69	112.42	111.22	111.43	1.20	-0.49	0.50
Mean	112.36	111.57	111.64	0.80	112.12	111.18	111.37	0.93	-0.38	0.14

Under natural conditions the largest range of lake stages was 1.2m across the 24-year period. The lowest range was 0.59m in profile 1b. Outlets 1-1c used to model discharge and stage time series have very different forms to the outlets from 2 to 3c. The former profiles are narrower and have steeper banks which form a more V-shaped profile whilst the remaining profiles have wider more U-shaped form. This predicted reduction in the range of lake level fluctuations is consistent with evidence in the literature stating that unregulated lakes show smaller water levels fluctuations and fluctuations follow a more regular seasonal pattern (Brauns *et al.*, 2008).

Information on the expected mean and range of lake level fluctuations is important as these changes will have ecological effects. Information on the expected drop in lake level may also influence whether the weir is fully removed in one engineering operation or gradually reduced over time.

The ecological effects of water-level fluctuations (WLF) in lakes are varied. Habitats, breeding grounds and food sources can be gained or lost as the impacts of phenomena such as light, waves and water temperature change (Wantzen *et al.*, 2008). However, whilst WLF are natural and essential for the survival of a number species, helping increasing biodiversity and

productivity (Gafny *et al.*, 1992), future water availability and physical changes to Ennerdale Water may disrupt these natural processes.

In Ennerdale Water, both Arctic Charr and Lamprey use the River Liza and other lake tributaries for spawning and refuge. A sudden and permanent lowering of the mean lake level may impact littoral geomorphology and potentially limit access to a number of lake tributaries as theirs channels readjust. In the case of Artic Charr, access to the River Liza is required during winter months for spawning when river flows and lake levels are generally higher so this may be a less significant impact. However, the impact lake changes may have on other species and at different times of the year should not be overlooked.

With the use of detailed bathymetric data (3.2.5) for the site it is possible to predict which areas of the site are likely to be frequently exposed and which are likely to be inundated. With this information restoration and conservation of key riparian and aquatic vegetation can be carried out. Replanting protected or endangered aquatic plants from areas likely to be permanently or frequently exposed under unmanaged conditions to more suitable areas of the lakebed will help ensure their survival and recolonisation.

Furthermore, managed replanting of the newly exposed shoreline of Ennerdale Water may need to be carried out. Selecting species that are able to flourish in an environment constantly changing between submerged and dry may aid the overall restoration of the site. This management of vegetation is often important if the threat of invasive species colonising new areas is to be limited (Orr and Stanley, 2006). Furthermore, by selecting the most appropriate vegetation, stabilisation of the lake outlet may be encouraged to occur more quickly which could have the added effect of reducing the mobilisation and transport of fine sediments downstream (Cannatelli and Curran, 2012).

7.5.3 *Summary of Sensitivity Analysis*

The sensitivity of the downstream hydrology and lake level were assessed based on 24 different outlet profiles. The results of the hydrological analysis show that channel gradient has some influence on the Q_5 values for the downstream flows. Initial channel width had greatest impact on minimum flows, with the model predicting lowest flows for the narrower engineered channels. This information may help inform the future design of the lake outlet channel, should the decision be taken to use an engineered outlet instead of an unconstrained natural channel.

In the case of the sensitivity of the lake level to channel slope and geometry, greatest variation was shown in the modelled natural outlets. However, under all scenarios the maximum range

in lake stages was 1.2m compared to an observed range of over 2m with the current physical and hydrological conditions. Lake ecosystems are complex and consist of a range of habitats coupled by biological chemical and physical process (Leira and Cantonati, 2008). The results of this study therefore provide import information for planning the future of the site and managing the transition from the current system to a more natural one.

7.6 Implications for Weir Removal and River Restoration

Anthropogenic activities have disrupted natural low regimes and fragmented freshwater environments. These changes to flow dynamics have resulted in geomorphic and hydrological alterations and changes to in-channel morphology and behaviour which has limited their capacity for natural replenishment and thereby led to their degradation (Mondal and Patel, 2018). This is a major threat to aquatic ecosystems worldwide.

River restoration aims to reverse these trends. Broadly speaking, approaches to river restoration planning involve four phases: identifying desirable objectives, selecting appropriate approaches to achieving specific objectives, assessing the current catchment characteristics and implementing the restoration actions (Beechie *et al.*, 2008). In this context, the current thesis addresses the assessment of current catchment characteristics. However, the research goes further by attempting to predict what impact the proposed restoration actions will have and whether this will help achieve the specific restoration objectives.

In *Riverine Ecosytem Management*, Hohensinner *et al.*, (2018) suggest that planners and authorities carrying out restoration activities have a tendency to focus on specific aquatic habitats or the state of short stretches of rivers where issues are most evident. However, they state that flood dynamics and other connected hydrological processes at catchment scale need to be considered to achieve truly long-term sustainable restoration (Hohensinner *et al.*, 2018). In that context, this project addresses these concerns. Gilvear *et al.*,(2012) define catchment-scale restoration as *'a river restoration activity that singly, or in combination, restores natural catchment processes and a naturally functioning ecosystem which brings benefit or environmental services to the whole catchment and not just to the site of restoration'.* Using this definition, weir and dam removal easily falls into the category of catchment-scale restoration measures. Whilst the focus is on the lake outlet and its influence on sediment transport and hydrology, the work considers a range of hydrological processes beyond river-reach scale that will be affected by the proposed removal of the weir. The lake behaviour and upper catchment hydrology has been captured and monitoring of sub catchments provides a

more detailed insight into the range of physical properties and subsequent hydrological responses within. In addition to the evolving form of the lake outlet under weir removal scenarios, the volumes of sediment leaving the lake outlet have also been captured for an extended reach of the river. This information will provide the project steering group with valuable additional evidence to inform the future management of the site and plan the most appropriate weir removal process and associated mitigation measures. Furthermore, the evidence of the hydrological and geomorphic modelling highlights that the upstream and downstream impacts of the weir removal will need to be considered. Changes to the lake level will impact aquatic organisms and the channels of rivers and streams flowing into Ennerdale Water as they adjust to the new lowered average lake levels. Downstream channel geometry, physical habitats and local flow conditions will also be impacted by the changing hydrological regime and sediment supply.

The focus of this thesis and the principal driver for the changing management of the Upper Ehen is the conservation of the freshwater mussels. Freshwater pearl mussels are a keystone species, and their conservation can help maintain the integrity of aquatic ecosystems (Quinlan *et al.*, 2015a). However, there is clearly a need to consider the wider impact of weir removal on other species and the wider ecosystem.

The flow regime data and metrics generated in this study will serve to aid any future analysis on the wider impact of weir removal and a return to an unmanaged flow regime. This data could be used to apply habitat modelling, which can be conducted in a variety of ways and focus on a different species or groups of species. Habitat modelling, (sometimes called species distribution modelling or habitat suitability modelling) can be described as "*a method for predicting the suitability of a location for a species, or group of species, based on their observed relationship with environmental conditions*" (Rowden *et al.*, 2017).

In addition to freshwater mussels, species of particular relevance to the site are the atlantic salmon and brown trout as their presence is critical to the life cycle of the freshwater pearl mussel. However, this is not to suggest that other species such as aquatic invertebrates should be excluded from future studies as these also form an important part of the wider ecosystem and can serve as indicators of an ecosystem's overall health.

Examples of research in this area have looked at general river habitat modelling (Booker and Dunbar, 2004; Mouton *et al.*, 2007) or examined uncertainty in the application of the models (Habersack *et al.*, 2014). However other studies have looked at modelling for specific species.

Roy and Le Pichon, (2017) modelled functional fish habitat connectivity in rivers with a specific focus on river restoration for brown trout.

In the case of freshwater mussel species, Jähnig *et al.*, (2012) used linked hydrological, hydraulic and speices distribution models to predict habitat suitability for the European fingernail claim (sphariеum corneum), a freshawter bivalve molusc. Their study used hydroligcal modelling data as the input into hydraulic models to simulate water levels and sediment discharge. This aproach could be applied at Ennerdale to gain further understanding of the wider implications of weir removal.

It is also imortant to consider the whole life cyle of the mussels as, juveniles have different habitat requirements to adults. Habitat modelling shoul factor in the need for a range of habitat tyes for the species throughtout its lifecycle and additionally the requirements of other species. Furthermore, the work of Degerman *et al.,* (2013) highlights the need to consider, water chemistry and wider catchment scale factors in additon to the physical in-stream conditions when assessing or predicting habitat sutiability.

An additional consideration, when analysing the impact of future management changes is the life cycle of the freshwater mussel and the longevity of the species. Not all species respond to conservation measures at the same rate and it is possible that the impact of future changes to on the species may not be visible for several years, often referred to as an ecological time lag. It is also possible that an initial abrupt change in flow or physical conditions results in some loss of the species before it begins to recover across its population age structure. This also has implications for other species and global conservation efforts as recent research has shown that "*lags in species' responses to conservation action could be masking our ability to observe progress towards conservation success*"(Watts *et al.*, 2020). This emphasises the need to account for ecological time lags when assessing the future conservation success of weir removal and flow regime restoration at Ennerdale and elsewhere.

7.7 Uncertainty of Modelling Results

The term uncertainty, in the context of the following section refers to "any departure from the unachievable ideal of complete determinism" (Walker *et al.*, 2003). This can be broken down into three types of uncertainty; epistemic uncertainty, ambiguity and natural variability (Lowe *et al.*, 2017). Epistemic uncertainty stems from a lack of knowledge or information on a subject; ambiguity can come from either or both the way in which information is

communicated or the way in which it is interpreted; and natural variability stems from the inherent randomness and unpredictability of natural systems (Lowe *et al.*, 2017).

Any method of generating data for assessing levels of hydrological alteration or estimating the natural hydrological regime such as hydrological modelling, discharge timeseries naturalisation or transferring data from a reference catchment, present great uncertainty (Acreman *et al.*, 2009). Communicating the uncertainty of results is therefore an essential element of any modelling study. Whilst the hydrological modelling software used in this study has been shown through the use of performance indicators to be capable of representing the managed regime, there is no means of validating future predictions. Comparing the predicted results to similar catchments provides the level of assurance that the model is producing results that might be expected and capturing hydrological behaviours observed in physically similar and geographically close catchment.

SHETRAN has been shown in a number of studies to accurately represent a range of hydrological behaviours at a range of scales (Lewis *et al.*, 2018) whilst level pool routing is a well-established method for capturing the flow attenuating properties of lakes (Chin, 2000) where available data allows its application. This is also not to suggest that the models are without uncertainty, in fact, the recognition of the uncertainty surrounding hydrological models is well established. The mathematician George Box stated that *"all models are wrong, but some are useful"* and in some cases such as predicting future changes, models often represent the only reliable option. The previous application on the modelling tools and techniques used in this study and analysis of their performance suggest that they are fit for purpose. Yet, employing these techniques with data sets based on modelled data or future climate projections adds an additional level of uncertainty. This in no way undermines the veracity of the modelling carried out in this study as SHETRAN + LPR has be shown to accurately replicate the current managed regime thereby providing evidence of its suitability for this study. Rather, the uncertainty highlighted in the following sections is an inherent product of natural variability within hydrological systems and provides further information for consideration in future decisions on the appropriate management of the Upper Ehen. Furthermore, approaches to reduce these sources of uncertainty and further strengthen the findings of this study are proposed in Chapter 8 of the thesis.

Comparing the performance of the chosen modelling tools with alternative models was beyond the scope for this study. However, other models may have performed equally as well as the SHETRAN and the LPR model combination used in this study. A further area for study which could further highlight or reduce existing uncertainties would be to model natural flows

in the Upper Ehen using other modelling tools or techniques for comparison. Additionally, combining the results of multiple model outputs instead of those of a single hydrological model has been promoted in hydrological studies as a means of improving river discharge estimates (Shamseldin *et al.*, 1997; Arsenault *et al.*, 2017) and may be an avenue for further investigation. However, the additional time requirements for setting up and running additional models need to be weighed up against the potential gains in overall prediction performance.

7.7.1 *Uncertainties with Existing Lake Inflow Data*

The stochastically generated inflow data used in this thesis was used taken from a water resources modelling study for the entire United Utilities supply region carried out by the consultancy Atkins. The principal aim of the Atkins study was to provide data for long term water resources planning and predict supply deficits based on forecasts of future water demand. Its suitability for application in a hydrological study concerned with daily flow metrics is therefore questionable. However, for understanding the long-term hydrological behaviour of the catchment at monthly time-scales and above, the model provides statistically robust data.

The first step in the Atkins study was to use a weather generator which created a synthetic time series of monthly rainfall data with statistically similar properties to historical data from 1911 to 1988. The approach used was an adaptation of the methodology presented by Serinaldi and Kilsby, (2012) but used a Generalised Linear Model to describe the historic rainfall data (Atkins, 2017). The monthly data was converted to daily data by looking at historical daily distributions for each month. This was then used to run a CATHMOD rainfall-runoff model to generate lake inflows for the different catchments in the UU region including the Upper Ehen.

The stochastic inflow data provided from the Atkins study was then run through the Level Pool Routing model presented in Chapter 4 using two natural outlet profiles (Profiles 2a and 3) that showed the upper and lower extremes of the values for the low flow metrics (Section 6.4.6). The results from these model runs predicted mean flows of 3.05 m^3/s and Q_{95} values of 0.35 and 0.24 m^3/s. Despite the length of these model runs there is still some uncertainty around the inflow data used. Firstly, the inflow time series has generated zero flow values. Under extreme conditions this may be possible however, a situation where no water is entering the lake requires further investigation to assess the probability of this occurring. Another area of uncertainty with this data is the range of inflow values. The maximum inflow value from the stochastically generated time series of 1,134 years was just 30 m^3/s. Based on

the observation of both the historic discharge in the River Ehen and the gauged flows from the River Liza during the 2017-2019 monitoring campaign this raises several doubts. The maximum flow in the River Ehen between 1995-2019 based on measurements from the NRFA gauging weir was 80 m^3/s (20/11/2009) although there is some uncertainty around this figure as flows above 38 m3/s at this gauging station are extrapolated. Based on observations of the site and understanding of the attenuating effect of lakes it is fair to assume that inflow values would have been higher than the downstream observed discharge. The SHETRAN model predicted inflows into Ennerdale Water of 70 m^3/s and the level pool routing model predicted a downstream discharge of 61 m^3/s on the same date. During the period 1995-2019, the SHETRAN modelled predicted inflows greater than 30 m^3/s on 28 occasions.

Another area of uncertainty is that the inflow time series was generated using a weather generator software which relied on historic data from 19 rain gauges to provide coverage for the entire UU region and used disaggregated average monthly rainfall distributions. Using this technique, it is difficult to capture localised events and daily extremes in rainfall events, due to the highly heterogenous terrain and localised weather patterns in the North-west of England. Finally, the data used to calibrate the model was also not specifically tailored to the Upper Ehen catchment and thus cannot be expected to closely resemble the wider range of flows and rainfall that the historic data sets show.

The stochastic inflow data was generated using historic rainfall time series from 1911 up to 1998 and as such might be expected to capture a greater range of hydrological extremes, than seen in the modelling period of 1995 to 2019. However, this was not the case. The maximum flow generated from this data was 30.5 m^3/s in a time series over 1,000 years in length. Observational records for the River Ehen, from 1973 to 1998 alone, show that annual maximum flows exceeded 30 m^3/s on 15 occasions, and above 40 m^3/s on 6 occasions, reaching with an upper limit of 49 m^3/s.

This discrepancy may be due to the fact that the inflow data was generated from disaggregated monthly rainfall and consequently underrepresents the daily extremes. As a result, its suitability for assessing daily flow characteristics is questionable.

7.7.2 *Future Flows and Geomorphic Responses*

In general, predicating hydrological, thermal and water quality responses to river restoration measures under changing climatic conditions remains an area of great uncertainty (Arthington *et al.*, 2018). The results from the future climate scenarios are somewhat surprising. Looking at the FDC in isolation, one might think that the overall hydrology of the Upper Ehen

catchment may not have change substantially, even under a more extreme emissions scenario. However, when looking at the shift in the distribution of annual rainfall and subsequent impact of the flow regime the results show a substantial seasonal shift.

Based on the uncertainties surrounding future flows, the geomorphic evolution of the lake outlet is another area of potential uncertainty as this is primarily driven by the hydrological regime. The geomorphic modelling results showed similar predicted responses to weir removal that have been observed and documented in previous weir and dam removal studies (Doyle *et al.*, 2003b; Cannatelli and Curran, 2012; Poeppl *et al.*, 2019) which provides encouraging information for decision makers on the possible movements and form of the future lake outlet. The model also reflects the relationship between high discharges and rates of channel change. However, predicting the full range and frequency of future flows that the natural lake outlet will be subject to is challenging.

The estimates of both the natural flow regime and the geomorphic evolution provide an indication of the rates and ranges of change under a more natural regime. A number of extreme events, within the observation period have captured hydrological and geomorphic behaviours under both high and low flow conditions. Continued monitoring and research may help to further reduce epistemic uncertainty (Lowe *et al.*, 2017), however, additional data can also lead to additional uncertainty if previously unknown unknowns are brought to light. Furthermore, whilst certain sources of uncertainty such as natural variability may be hard to eliminate, acknowledging their existence is important in the decision-making process as it may help better constrain expectations or overly optimistic predictions of what may or may not occur as a result of management decisions. In the case of the Upper Ehen, suggestions for reducing the uncertainties outlined in this section are proposed in Chapter 8.

7.8 Implications for United Utilities

An additional requirement of the EngD programme is to produce a report which reflects upon the impact of this research on the sponsoring organisation and considers the business case for utilising the findings and tools developed during the research project. A ~7,000-word unpublished report (Carver, 2019) and its findings were presented to the United Utilities prior to submission of the final thesis.

In addition to that report, this final section of the discussion chapter briefly considers what the potential consequences of weir removal on the freshwater mussel population might be along with their implications for United Utilities. A discussion of how United Utilities can use cost

benefit analysis to assess the impact of future management options on its business operations is also presented.

7.8.1 *Management Options*

For the purposes of this analysis, three management options have been considered. These were chosen based on discussion with United Utilities and the wider project steering group.

1. Cease abstraction and continue flow regulation with existing infrastructure in place.
2. Remove infrastructure
    a) Construction phase
    b) Operational phase
3. Remove all infrastructure but maintain supplementary pumping.

In reality a number of additional options may also exist however the crux of the issue is focussed around removing or maintaining the weir. Furthermore, additional measures (e.g. monitoring, maintenance), are likely to be incorporated as part of the above options.

Maintaining the infrastructure without flow regulation was not considered as an option as the conditions of the impoundment license stipulate that compensation flows must be implemented. Furthermore, the design of the infrastructure means that lake levels could drop below the level by which water can flow under gravity (through the fish pass) into the River Ehen, which would be an unacceptable scenario for all parties.

7.8.2 *Implications for Freshwater Mussels and United Utilities*

Firstly, it is important to recognise, that all of the above options pose risks to the mussel population. However, there are subtle differences in the liabilities to United Utilities.

A major risk of associated with leaving the infrastructure in place (as with all other options) is that the overall health of the mussel population continues to deteriorate or shows no sign of juvenile recruitment. Mitigation options for United Utilities under this option are limited as the status quo is being maintained. Additionally, the company would be required to continue meeting its legal obligations under the impoundment license and other regulatory obligations such as the Reservoir Act requiring ongoing and significant operating expenditure with periodic additional investment to maintain asset health.

A major risk associated with removing weirs (options 2a and 3) is the mobilisation of large volumes of fine sediment, which in many cases is contaminated. However, the analysis of sediments at Ennerdale and historical knowledge of the site suggests that contamination is

unlikely to be a significant risk in this instance. However, mussels are sensitive to even relatively small amounts of fine sediment.

The risk of sediment releases can be mitigated in a variety of ways depending on the extent and volumes of sediment present. In some sites, mitigation, through sediment removal prior to weir or dam decommissioning may be required. This can be carried out using various techniques however the use of conventional earth moving equipment is often sufficient (MacBroom and Schiff, 2013). Additionally, sediment traps located downstream of the removal site may also be used to mitigate the release of fine sediments (Evans, 2015). Further techniques can be employed depending on the approach taken when removing the weir or dam. If a phased draw down of the reservoir is carried out this can allow vegetation to establish, helping to stabilise any sediment surrounding the site (Evans, 2015).

Further analysis of the substrate behind the weir (as advocated in chapter 5.8) could add further certainty around the composition, abundance and location of fine sediments and thus inform the extent, scale and most appropriate approaches for any required mitigation.

A risk associated with completely removing the existing infrastructure and any form of flow regime control measures is the potential for extended periods of dry weather or severe drought to occur after weir removal and mussel populations suffers as a result. However, this decision is likely to be approved by the regulatory bodies overseeing the project and thus the liability should not be placed on United Utilities if this is the case.

The third management option of retaining some form of flow control without the weir in place could mitigate this risk however United Utilities remains with an obligation to manage flows in perpetuity despite no longer having any use for the site. This also goes against the idea of restoring a natural flow regime which would be subject to the hydrological fluctuations imposed by climatic variations.

The risks or implications to United Utilities of undertaking any of the possible management options is primarily that the overall condition of the SAC and mussel population deteriorates. This would likely have negative reputational and financial consequences for the company. Stakeholder engagement and transparency over the decision-making process of future management decisions may go some way to mitigating this risk however this may be difficult to achieve if there is strong public opinion against the chosen management options. This is particularly pertinent, in the area surrounding Ennerdale as United Utilities, has previously had attempts to remove obsolete water resources infrastructure blocked due to strong public opinion opposing the engineering works (for further details see Friends of the Lake District,

2015). This case varies in that the infrastructure is in a Special Area of Conservation and subject to greater regulatory pressures however, opposition to infrastructure removal will still need to be taken into account.

7.8.3 *Cost benefit analysis*

Cost-benefit analysis (CBA) is an economic tool which can help businesses or decision makers chose from a range of policy (Boardman *et al.*, 2017) or investment options. Within CBA there are a number of approaches, however four of the more commonly used are hedonic pricing, travel cost, stated preference valuation: contingent valuations and stated preference valuation: choice modelling (Vallesi *et al.*, 2019).

Existing literature on CBA for dam removal has historically predominately derived from U.S. case studies and literature (Whitelaw and MacMullan, 2002; Doyle *et al.*, 2003a; Smith, 2009). However, more recently the AMBER project has produced a study quantifying the economic costs and benefits of river infrastructures which looked at different CBA approaches and assessed the costs of barrier removal in the UK, Spain and the U.S.A. (Vallesi *et al.*, 2019). In most cases however, whether from the U.S. or Europe, the general approach to cost-benefit analysis in these studies tends to look at the overall project rather than taking the point of view of a single stakeholder such as the dam owner or operator.

United Utilities

Although some of the wider issues of CBA are mentioned, the focus of this current discussion is based the implications for United Utilities who own and manage the weir and associated assets at Ennerdale.

The initial decision to revoke the abstraction licence at Ennerdale Water led to the incorporation of the West Cumbria Water Resource Zone (WRZ) with the integrated WRZ through a transfer pipeline from Thirlmere reservoir. This will become operational in 2022 and leave United Utilities with a set of obsolete assets which will continue to have an impact on the environment in which they are situated but serve little or no economic benefit to the company. The legal obligations of monitoring and maintenance of the infrastructure will also result in ongoing costs and liabilities. Although, the final decision whether to remove the weir is likely to be made by the regulator and advisory body (Environment Agency and Natural England), a cost benefit analysis allows United Utilities to plan its operations and finances under these different options.

In simple terms, United Utilities would need to assess the ongoing operational expenditure (OPEX) of its assets versus the capital expenditure (CAPEX) in removing or replacing it. In the case of ageing assets this decision point becomes clearer as they approach their end of their excepted lifespan and costs of maintenance begin to grow.

The OPEX at Ennerdale would be based on maintaining and repairing the weir, fish pass and the regulatory requirements of the compensation flow regime or the safety requirements of the Reservoirs Act. Additional assets such as pumps that may be in place to ensure compensation flows during droughts also have an operational life so would eventually need to be replaced. The cost of maintaining, operating and replacing these would therefore also need to be factored in.

Estimating the CAPEX for removing the weir would need to consider the cost of any feasibility studies, environmental impact studies, flood risk assessments, enabling works (e.g. access works, public right of way diversions), the infrastructure removal engineering works including construction phase mitigation measures (likely to be both bespoke and significant) and any ongoing site monitoring or maintenance after removal which would, given the long life cycle of freshwater mussels would be likely to be in excess of five years. This would involve a far greater initial financial outlay for United Utilities however this should also be framed in the wider context of the £300 million already invested by the company to integrate the West Cumbria WRZ and the cost and liability involved with maintaining the assets in perpetuity.

According to research by AMBER (Vallesi *et al.*, 2019), the engineering cost of barrier (dam or weir) removal is highly variable reflecting the variety of engineering options used in their construction and the locational settings. However, the research also shows that sediment volume and type are the most important determinants on the cost of barrier removal as this dictates the need for sediment removal and relocation (Vallesi *et al.*, 2019).

7.9 Summary

The UK is home to thousands of obsolete weirs and in-stream barriers, of which many are privately owned. The environmental impacts of these structures are well-documented and for water companies they are a costly and undesirable liability. Removing them can produce many ecological benefits yet this represents just one of a suite of many potential restoration activities and in isolation shouldn't be viewed as a panacea. Furthermore, where other barriers are present in a stretch of water, the removal of a single weir will have limited impact.

A large number of hydrological metrics have been generated for a range of scenarios and physical outlet forms. These provide an abundance of information for the wider project steering group. The national-scale study has highlighted that a small number of these metrics potentially offer key insights into the flow regime requirements of freshwater mussels however these results need to be examined and interpreted along with the wider set of hydrological metrics by the appropriate experts in freshwater ecology.

Most freshwater ecosystems and the ecology they sustain are subject to pressure resulting from the complex interaction between natural processes and man-made factors (Laizé *et al.*, 2014) Despite the potential for ecological enhancement of rivers and their associated floodplains (Bednarek 2001; Doyle et al. 2005), not all schemes are successful in producing positive or the desired ecological outcomes (Stanley and Doyle, 2003). Whilst removal is very effective at enhancing some aspects of riverine ecology it should be viewed as only one aspect of wider catchment-scale management (Stanford *et al.*, 1996). Truly effective weir removal projects will most likely need to be combined with additional conservation and restoration techniques (Hart *et al.*, 2002). Furthermore, in addition to scientific and technical expertise, dam removal requires strong coalitions across multiple levels and a willingness from regulators and policy makers (Foley *et al.*, 2017b). This is particularly pertinent for United Utilities, as despite the clear economic case for decommissioning obsolete infrastructure, the final decisions on the future of the weir at Ennerdale Water a will also hinge on satisfying the social and environmental case for its removal.

Chapter 8 - Conclusions

The main aim of this research project was to estimate the natural flow regime of the Upper Ehen catchment and examine the effect of weir removal scenarios on both the River Ehen and Ennerdale Water. This was achieved through the targeted collection of new lake inflow data and the reconstruction of a natural flow regime using hydrological and geomorphic modelling tools tailored to the site. Novel approaches to lake sediment data analysis and a bespoke modelling tool that enables rapid hydrological assessment, provide new insights into the possible future hydromorophology of the site post weir removal.

The modelling approaches and tool developed in this study are widely applicable to future weir and dam removal case studies elsewhere and in catchments where historic undisturbed flow regime data is lacking.

8.1 Key Conclusions

- Freshwater mussel populations are found in both regulated and unregulated flow regimes. Although flow regime is not the only factor affecting mussel population health, 11 hydrological metrics were identified that show a statistically significant difference (t-test and Wilcoxon-Mann-Whitney scores) between the flow regimes of rivers with healthy mussel populations and those with declining numbers (see section 2.3.2). These metrics form a baseline of comparison for flow regime attributes in altered and unmanaged river catchments in the UK and for predicted future flow in the River Ehen.
- SHETRAN + Level Pool Routing provides an effective tool for modelling catchments heavily influenced by the flow attenuating properties of lakes and water management infrastructure.
- Given the same inflow conditions, the natural flow regime of the Upper Ehen post weir removal would have higher mean flows across all months however, minimum flows below Q_{95} would be reduced.
- In the absence of impounding infrastructure, and under a natural flow regime Ennerdale Water would have an average surface elevation of 111.37 mAOD. This represents a reduction of 0.95m from present day levels and would equate to a surface area reduction of 5% or 0.16 km^2.

- Ground Penetrating Radar has shown evidence of historic outlet channels to the North of the current outlet, providing an insight into the possible future extent and direction of channel migration in the absence of retaining infrastructure at the lake outlet. The GPR survey and coring samples have located and characterised the composition of pockets of fine sediment in the area behind the weir.
- After weir removal, geomorphic modelling predicted an average 30 m^3 of sediment would be transported downstream across a 30-year period. Sediment transport rates would be highest immediately after weir removal and during extreme flood events.

The UK-wide study has identified a small pool of hydrological metrics that distinguish between catchments with failing and healthy mussel populations. These are tentative relationships as flow regime is a key, but not isolated, factor in ecological health of rivers. However, the results provide a useful comparison for the River Ehen in its current impounded form and the more natural state that would ensue post weir removal. Under a natural flow regime, the value for 7 of the 11 identified metrics, move towards scores found in rivers with healthy mussel populations.

Results from the data collection and modelling have demonstrated the suitability of the chosen approaches presented in this study. SHETRAN + Level Pool Routing was shown to replicate the managed regime with a NSE of 0.8 for the period from 1995-2019. This compared to a NSE of 0.64 when only using SHETRAN. For the calibration period 2017-2019 a NSE of 0.88 was obtained. However, with hourly inflow data, SHETRAN and a hydraulic lake outflow model achieved a NSE of 0.94. Both model arrangements demonstrate the suitability of SHETRAN to accurately represent the natural hydrology of the Upper Catchment. In addition, despite some limitations of the monitored discharge data, it has facilitated effective model calibration and produced previously unseen empirical evidence of the flow attenuating properties of Ennerdale Water. Rainfall data has also matched the observed water balance for the catchment for the 2017-2019 period and compared well with existing datasets for the short overlapping period.

Minimum flow values will be lower under a natural regime decreasing to an average of 0.19 m^3/s compared to under the managed regime. Mean values for Q_{95} are predicted to be 0.52 m^3/s compared to an observed Q_{95} of 0.51 m^3/s

The average channel slope is predicted to be 0.008 with upstream migration channel migration ranging between 35 and 40m. Lateral channel movement is expected to occur but be minimal in the first 30-years post weir removal. Under a natural flow regime with reduced lake level, peak flow will be reduced however mean Q_5 values show a slight increase

changing from 9.85 m^3/s to 9.95 m^3/s in the absence of abstraction and flow regulating infrastructure. Sediment transport rates and peak maximum discharge values were highly correlated producing an R^2 value of 0.75. The maximum annual sediment volume transported in a single year was 309 m^3. This occurred in a year when the highest 24-hour peak flow (80 m^3/s) was observed.

In response to these geomorphic changes, the average lake levels are predicted to decrease by around 0.95m after 30-years of natural geomorphic processes operating. The final decision on how the lake outlet would be engineered after weir removal remains to be made. However, the model outputs provide valuable evidence upon which to base the design of an engineered outlet. All of the modelled natural outlets showed a lower range of lake level fluctuation however this varied between 0.6m and 1.2m compared to a range of 2.13m for the observed lake levels under the managed regime between 1995-2019.

Evidence from the GPR survey suggests lateral migration of the lake outlet has occurred in the past. This highlights the dynamic nature of the system and supports findings from other dam removal studies that emphasise the importance of lateral channel movement (Poeppl *et al.*, 2019). The location of the relic channels in the GPR gives an indication of the direction and extent of this lateral migration. Model runs showed lateral movement of 4 m to 26 m across the 30-year period, whilst the relic channel identified with GPR was approximately 20m north of the current outlet. This information should be considered in planning for and understanding the future characteristics of the lake outlet.

The principal sources of uncertainty in the study are the stochastic inflow timeseries from the United Utilities Water Resource Management Plan and the inflows generated from UKCP18 8.5 emissions scenario. Comparison of the modelled natural flow regime for the 1995-2019 period against the observed data was seen to provide the most reliable indication of the natural hydrological regime of the Upper Ehen.

8.2 Recommendations for Further Research

Based on the findings from this thesis, four priorities for further research have been identified. They are all proposed with the aim of adding greater certainty to the predictions of hydrological and geomorphic responses to weir removal.

8.2.1 *Monitoring of Surface and Sub-surface Lake Inflows*

The research in this thesis has highlighted some discrepancies between the modelled flows and the observed inflows into Ennerdale Water in the lower River Liza. Hypotheses for these discrepancies proposed in Chapter 4 (Section 4.6.1) suggest a complex mix of surface and sub-surface flows within this area of the catchment. To test this hypothesis, a more targeted and sustained monitoring of the surface and subterranean hydrology of this area of the catchment is required. This should consist of two elements. Firstly, an appropriate method of continually monitoring flow in lower reaches of the River Liza should be researched and implemented. Discharge measurements underpin our understanding of hydrological systems (Coxon *et al.*, 2015) and reliable measurements are essential for accurate hydrological modelling and water resource management decisions. The chosen approach for measuring discharge would need to overcome certain constraints that affected the present study and provide accurate and continuous measurements to help reduce uncertainty over the volumes and timings of inflows into the lake.

A non-intrusive method for flow discharge estimation is desirable due to the need to protect the aquatic ecology, natural geomorphology and visual integrity of the Upper Ehen catchment. Several techniques have been developed in recent years that are based on image velocimetry. This relies on the detection of visibly or thermally distinct surface water features and estimates of their velocity as they move downstream. This can then be converted to a depth-averaged velocity and supplemented with topographic and bathymetric data to estimate discharge. Whilst this is an emerging field, recent advances in video technology have made this technique more accessible (Perks, 2020). Continual tracer injection methods are also well-established and can be programmed with simple, compact computers such as the Raspberry Pi, to be activated at certain river stage thresholds to ensure full coverage of the river flow. Several tracers, including dissolved salt as demonstrated in Chapter 3, have no significant environmental impacts and are proven to perform well in challenging mountainous stream conditions.

In addition to continual surface water measurements, groundwater movements in the lower flood plain should be investigated. Again, a number of techniques are available for this. Ground Penetrating Radar as described in Chapter 5 has been shown to provide information on groundwater levels and the composition of surface geology and geological features. In the lower floodplain this would provide further evidence of relic river channels which may be acting as conduits for water in the Ennerdale Water as well as providing data on the depth and rates of change of the water table. In combination, monitoring these two aspects of the Upper

Ehen hydrology would add greater certainty to the model result and allow for further calibration of the model.

8.2.2 *Flood Attenuation and Downstream Flood Risk*

Under a natural flow regime peak flows are predicted to be dampened. Downstream flood risk is a common management concern in weir and dam removal projects and thus making predictions on changes to flood risk requires high degrees of certainty as the social and economic implications of underestimating this risk are great. A detailed study of the changes to downstream flood risk is therefore an important area for further research. This project has looked at the full range of flow conditions and expected changes under a natural lake outflow and the absence of abstraction. However, the focus has been on low flows and the model has been calibrated to closely replicate observed low flows.

To increase understanding of what, if any, impact there is on the model's ability to represent high-flows and in order to further reduce uncertainty around the high-flow predictions, a study that uses specific flood modelling software should be used. The DEMs generated from the geomorphic modelling study will provide data for a more detailed flood-risk study however, the modelling domain would need to be extended to cover the entire lower catchment to include populated areas and critical infrastructure.

8.2.3 *Detailed Future Climate Studies*

A third area that will have implications for the two previously proposed areas for further study and any future hydrological investigations of the site is the future local climate of the catchment. A more detailed study of future climate projections for the site and the surrounding region would provide valuable information on the range of expected climatic variables and thus help constrain the natural hydrological conditions. The present study used an extreme emissions scenario to capture the upper limits of expected changes; however this data is part of a wider secondary data set and thus not specifically tailored to capture the localised catchment-scale variations in rainfall and other significant meteorological phenomena.

8.2.4 *Nature Based Solutions for Managing Flows*

The Upper Ehen catchment is unique. It has minimal development and its less accessible location on the Western edge of the Lake District National Park means that it still retains a feeling of wilderness. Flow regime management using the existing infrastructure and

modification to it, has been implemented to limit the impact of abstraction and impoundment on the downstream ecology. Whether some form of low flow management could or should remain in the future is debateable. Questions over the sustainability of such an approach and where responsibility lay for maintaining any system would need to be answered. However, further research into nature-based solutions for low flow management could be investigated. Tree planting is also high on the Government's environmental agenda and conducting modelling investigations into the impact of extensive woodland re-planting on the hydrology of the upper Ehen would provide further evidence of its effectiveness as a tool for flow regime management.

Whether these measures could be implemented upstream of Ennerdale Water and thereby reduce the flashiness of the upper catchment should be investigated. A modelling study that incorporated one or a suite of nature-based solutions and assessed their potential impact on specific aspects of the flow regime may be an area for future research.

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Appendix A

Hydrological metrics and their methods of calculations as published in (Henriksen *et al.*, 2006). Imperial units of measurement and American spellings are from the original source and not used in this thesis.

MA1	Mean of the daily mean flow values for the entire flow record (cubic feet per second – temporal).
MA2	Median of the daily mean flow values for the entire flow record (cubic feet per second – temporal).
MA3	Mean (or median - Use Preference option) of the coefficients of variation (standard deviation/mean) for each year. Compute the coefficient of variation for each year of daily flows. Compute the mean of the annual coefficients of variation (percent - temporal).
MA4	Standard deviation of the percentiles of the logs of the entire flow record divided by the mean of percentiles of the logs. Compute the log10 of the daily flows for the entire record. Compute the 5th 10th 15th 20th 25th 30th 35th 40th 45th 50th 55th 60th 65th 70th 75th 80th 85th 90th and 95th percentiles for the logs of the entire flow record. Percentiles are computed by interpolating between the ordered (ascending) logs of the flow values. Compute the standard deviation and mean for the percentile values. Divide the standard deviation by the mean (percent - spatial).
MA5	The skewness of the entire flow record is computed as the mean for the entire flow record (MA1) divided by the median (MA2) for the entire flow record (dimensionless - spatial).
MA6	Range in daily flows is the ratio of the 10 percent to 90 percent exceedence values for the entire flow record. Compute the 5 percent to 95 percent exceedence values for the entire flow record. Exceedence is computed by interpolating between the ordered (descending) flow values. Divide the 10 percent exceedence value by the 90 percent value (dimensionless – spatial).
MA7	Range in daily flows is computed like MA6 except using the 20 percent and 80 percent exceedence values. Divide the 20 percent exceedence value by the 80 percent value (dimensionless - spatial).
MA8	Range in daily flows is computed like MA6 except using the 25 percent and 75 percent exceedence values. Divide the 25 percent exceedence value by the 75 percent value (dimensionless – spatial).
MA9	Spread in daily flows is the ratio of the difference between the 90th and 10th percentile of the logs of the flow data to the log of the median of the entire flow record. Compute the log10 of the daily flows for the entire record. Compute the 5th 10th 15th 20th 25th 30th 35th 40th 45th 50th 55th 60th 65th 70th 75th 80th 85th 90th and 95th percentiles for the logs of the entire flow record. Percentiles are computed by interpolating between the ordered (ascending) logs of the flow values. Compute MA9 as (90th – 10th) /log10(MA2) (dimensionless – spatial).
MA10	Spread in daily flows is computed like MA9 except using the 20th and 80th percentiles (dimensionless – spatial).
MA11	Spread in daily flows is computed like MA9 except using the 25th and 75th percentiles (dimensionless – spatial).
MA12 - MA23	Means (or medians - Use Preference option) of monthly flow values. Compute the means for each month over the entire flow record. For example MA12 is the mean of all January flow values over the entire record (cubic feet per second – temporal).
MA24 – MA35	Variability (coefficient of variation) of monthly flow values. Compute the standard deviation for each month in each year over the entire flow record. Divide the standard deviation by the mean for each month. Average (or median - Use Preference option) these values for each month across all years (percent – temporal).
MA36	Variability across monthly flows. Compute the minimum maximum and mean flows for each month in the entire flow record. MA36 is the maximum monthly flow minus

	the minimum monthly flow divided by the median monthly flow (dimensionless – spatial).
MA37	Variability across monthly flows. Compute the first (25th percentile) and the third (75th percentile) quartiles (every month in the flow record). MA37 is the third quartile minus the first quartile divided by the median of the monthly means (dimensionless – spatial).
MA38	Variability across monthly flows. Compute the 10th and 90th percentiles for the monthly means (every month in the flow record). MA38 is the 90th percentile minus the 10th percentile divided by the median of the monthly means (dimensionless – spatial).
MA39	Variability across monthly flows. Compute the standard deviation for the monthly means. MA39 is the standard deviation times 100 divided by the mean of the monthly means (percent – spatial).
MA40	Skewness in the monthly flows. MA40 is the mean of the monthly flow means minus the median of the monthly means divided by the median of the monthly means (dimensionless – spatial).
MA41	Annual runoff. Compute the annual mean daily flows. MA41 is the mean of the annual means divided by the drainage area (cubic feet per second/square mile – temporal).
MA42	Variability across annual flows. MA42 is the maximum annual flow minus the minimum annual flow divided by the median annual flow (dimensionless – spatial).
MA43	Variability across annual flows. Compute the first (25th percentile) and third (75th percentile) quartiles and the 10th and 90th percentiles for the annual means (every year in the flow record). MA43 is the third quartile minus the first quartile divided by the median of the annual means (dimensionless –spatial).
MA44	Variability across annual flows. Compute the first (25th percentile) and third (75th percentile) quartiles and the 10th and 90th percentiles for the annual means (every year in the flow record). MA44 is the 90th percentile minus the 10th percentile divided by the median of the annual means (dimensionless – spatial).
MA45	Skewness in the annual flows. MA45 is the mean of the annual flow means minus the median of the annual means divided by the median of the annual means (dimensionless – spatial).
ML1 – ML12	Mean (or median - Use Preference option) minimum flows for each month across all years. Compute the minimums for each month over the entire flow record. For example ML1 is the mean of the minimums of all January flow values over the entire record (cubic feet per second – temporal).
ML13	Variability (coefficient of variation) across minimum monthly flow values. Compute the mean and standard deviation for the minimum monthly flows over the entire flow record. ML13 is the standard deviation times 100 divided by the mean minimum monthly flow for all years (percent – spatial).
ML14	Compute the minimum annual flow for each year. ML14 is the mean of the ratios of minimum annual flows to the median flow for each year (dimensionless – temporal).
ML15	Low flow index. ML15 is the mean of the ratios of minimum annual flows to the mean flow for each year (dimensionless – temporal).
ML16	Median of annual minimum flows. ML16 is the median of the ratios of minimum annual flows to the median flow for each year (dimensionless – temporal).
ML17	Base flow. Compute the mean annual flows. Compute the minimum of a 7-day moving average flow for each year and divide them by the mean annual flow for that year. ML17 is the mean (or median - Use Preference option) of those ratios (dimensionless – temporal).
ML18	Variability in base flow. Compute the standard deviation for the ratios of 7-day moving average flows to mean annual flows for each year. ML18 is the standard deviation times 100 divided by the mean of the ratios (percent – spatial).
ML19	Base flow. Compute the ratios of the minimum annual flow to mean annual flow for each year. ML19 is the mean (or median - Use Preference option) of these ratios times 100 (dimensionless – temporal).

ML20	Base flow. Divide the daily flow record into 5-day blocks. Find the minimum flow for each block. Assign the minimum flow as a base flow for that block if 90 percent of that minimum flow is less than the minimum flows for the blocks on either side. Otherwise set it to zero. Fill in the zero values using linear interpolation. Compute the total flow for the entire record and the total base flow for the entire record. ML20
ML21	Variability across annual minimum flows. Compute the mean and standard deviation for the annual minimum flows. ML21 is the standard deviation times 100 divided by the mean (percent – spatial).
ML22	Specific mean annual minimum flow. ML22 is the mean (or median - Use Preference option) of the annual minimum flows divided by the drainage area (cubic feet per second/square mile – temporal).
MH1 – MH12	Mean (or median - Use Preference option) maximum flows for each month across all years. Compute the maximums for each month over the entire flow record. For example MH1 is the mean of the maximums of all January flow values over the entire record (cubic feet per second – temporal).
MH13	Variability (coefficient of variation) across maximum monthly flow values. Compute the mean and standard deviation for the maximum monthly flows over the entire flow record. MH13 is the standard deviation times 100 divided by the mean maximum monthly flow for all years (percent – spatial).
MH14	Median of annual maximum flows. Compute the annual maximum flows from monthly maximum flows. Compute the ratio of annual maximum flow to median annual flow for each year. MH14 is the median of these ratios (dimensionless – temporal).
MH15	High flow discharge index. Compute the 1 percent exceedence value for the entire data record. MH15 is the 1 percent exceedence value divided by the median flow for the entire record (dimensionless – spatial).
MH16	High flow discharge index. Compute the 10 percent exceedence value for the entire data record. MH16 is the 10 percent exceedence value divided by the median flow for the entire record (dimensionless – spatial).
MH17	High flow discharge index. Compute the 25 percent exceedence value for the entire data record. MH17 is the 25 percent exceedence value divided by the median flow for the entire record (dimensionless – spatial).
MH18	Variability across annual maximum flows. Compute the logs (log10) of the maximum annual flows. Find the standard deviation and mean for these values. is the standard deviation times 100 divided by the mean (percent – spatial).
MH19	Skewness in annual maximum flows. Use the equation: MH19 = N2 x sum(qm3)-3N x sum(qm) x sum(qm2) + 2 x (sum(qm))3N x (N-1) x (N-2) x stddev3 where: N = Number of years qm = Log10(annual maximum flows) stddev = Standard deviation of the annual maximum flows (dimensionless – spatial).
MH20	Specific mean annual maximum flow. MH20 is the mean (or median - Use Preference option) of the annual maximum flows divided by the drainage area (cubic feet per second/square mile – temporal).
MH21	High flow volume index. Compute the average volume for flow events above a threshold equal to the median flow for the entire record. MH21 is the average volume divided by the median flow for the entire record (days – temporal).
MH22	High flow volume. Compute the average volume for flow events above a threshold equal to three times the median flow for the entire record. MH22 is the average volume divided by the median flow for the entire record (days - temporal).
MH23	High flow volume. Compute the average volume for flow events above a threshold equal to seven times the median flow for the entire record. MH23 is the average volume divided by the median flow for the entire record (days - temporal).
MH24	High peak flow. Compute the average peak flow value for flow events above a threshold equal to the median flow for the entire record. MH24 is the average peak flow divided by the median flow for the entire record (dimensionless – temporal).
MH25	High peak flow. Compute the average peak-flow value for flow events above a threshold equal to three times the median flow for the entire record. MH25 is the average peak flow divided by the median flow for the entire record (dimensionless – temporal).

MH26	High peak flow. Compute the average peak flow value for flow events above a threshold equal to seven times the median flow for the entire record. MH26 is the average peak flow divided by the median flow for the entire record (dimensionless – temporal).
MH27	High peak flow. Compute the average peak flow value for flow events above a threshold equal to 75th percentile value for the entire flow record. MH27 is the average peak flow divided by the median flow for the entire record (dimensionless – temporal).
FL1	Low flood pulse count. Compute the average number of flow events with flows below a threshold equal to the 25th percentile value for the entire flow record. FL1 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FL2	Variability in low pulse count. Compute the standard deviation in the annual pulse counts for FL1. FL2 is 100 times the standard deviation divided by the mean pulse count (percent – spatial).
FL3	Frequency of low pulse spells. Compute the average number of flow events with flows below a threshold equal to 5 percent of the mean flow value for the entire flow record. FL3 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FH1	High flood pulse count. Compute the average number of flow events with flows above a threshold equal to the 75th percentile value for the entire flow record. FH1 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FH2	Variability in high pulse count. Compute the standard deviation in the annual pulse counts for FH1. FH2 is 100 times the standard deviation divided by the mean pulse count (number of events/year – spatial).
FH3	High flood pulse count. Compute the average number of days per year that the flow is above a threshold equal to three times the median flow for the entire record. FH3 is the mean (or median – Use Preference option) of the annual number of days for all years (number of days/year – temporal).
FH4	High flood pulse count. Compute the average number of days per year that the flow is above a threshold equal to seven times the median flow for the entire record. FH4 is the mean (or median - Use Preference option) of the annual number of days for all years (number of days/year – temporal).
FH5	Flood frequency. Compute the average number of flow events with flows above a threshold equal to the median flow value for the entire flow record. FH5 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FH6	Flood frequency. Compute the average number of flow events with flows above a threshold equal to three times the median flow value for the entire flow record. FH6 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FH7	Flood frequency. Compute the average number of flow events with flows above a threshold equal to seven times the median flow value for the entire flow record. FH6 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FH8	Flood frequency. Compute the average number of flow events with flows above a threshold equal to 25 percent exceedence value for the entire flow record. FH8 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FH9	Flood frequency. Compute the average number of flow events with flows above a threshold equal to 75 percent exceedence value for the entire flow record. FH9 is the average (or median - Use Preference option) number of events (number of events/year – temporal).

FH10	Flood frequency. Compute the average number of flow events with flows above a threshold equal to median of the annual minima for the entire flow record. FH10 is the average (or median - Use Preference option) number of events (number of events/year – temporal). Note - 1.67-year flood threshold (Poff 1996) - For indices FH11 DH22 DH23 DH24 TA3 and TH3 compute the log10 of the peak annual flows. Compute the log10 of the daily flows for the peak annual flow days. Calculate the coefficients for a linear regression equation for logs of peak annual flow versus logs of average daily flow for peak days. Using the log peak flow for the 1.67 year recurrence interval (60th percentile) as input to the regression equation predict the log10 of the average daily flow. The threshold is 10 to the log10 (average daily flow) power (cubic feet/second).
FH11	Flood frequency. Compute the average number of flow events with flows above a threshold equal to flow corresponding to a 1.67-year recurrence interval. FH11 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
DL1	Annual minimum daily flow. Compute the minimum 1-day average flow for each year. DL1 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DL2	Annual minimum of 3-day moving average flow. Compute the minimum of a 3-day moving average flow for each year. DL2 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DL3	Annual minimum of 7-day moving average flow. Compute the minimum of a 7-day moving average flow for each year. DL3 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DL4	Annual minimum of 30-day moving average flow. Compute the minimum of a 30 day moving average flow for each year. DL4 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DL5	Annual minimum of 90-day moving average flow. Compute the minimum of a 90 day moving average flow for each year. DL5 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DL6	Variability of annual minimum daily average flow. Compute the standard deviation for the minimum daily average flow. DL6 is 100 times the standard deviation divided by the mean (percent – spatial).
DL7	Variability of annual minimum of 3-day moving average flow. Compute the standard deviation for the minimum 3-day moving averages. DL7 is 100 times the standard deviation divided by the mean (percent - spatial).
DL8	Variability of annual minimum of 7-day moving average flow. Compute the standard deviation for the minimum 7-day moving averages. DL8 is 100 times the standard deviation divided by the mean (percent - spatial).
DL9	Variability of annual minimum of 30-day moving average flow. Compute the standard deviation for the minimum 30-day moving averages. DL9 is 100 times the standard deviation divided by the mean (percent - spatial).
DL10	Variability of annual minimum of 90-day moving average flow. Compute the standard deviation for the minimum 90-day moving averages. DL10 is 100 times the standard deviation divided by the mean (percent - spatial).
DL11	Annual minimum daily flow divided by the median for the entire record. Compute the minimum daily flow for each year. DL11 is the mean of these values divided by the median for the entire record (dimensionless – temporal).
DL12	Annual minimum of 7-day moving average flow divided by the median for the entire record. Compute the minimum of a 7-day moving average flow for each year. DL12 is the mean of these values divided by the median for the entire record (dimensionless – temporal).
DL13	Annual minimum of 30-day moving average flow divided by the median for the entire record. Compute the minimum of a 30-day moving average flow for each year. DL13 is the mean of these values divided by the median for the entire record (dimensionless – temporal).
DL14	Low exceedance flows. Compute the 75 percent exceedance value for the entire flow record. DL14 is the exceedance value divided by the median for the entire record (dimensionless – spatial).

DL15	Low exceedence flows. Compute the 90 percent exceedence value for the entire flow record. DL14 is the exceedence value divided by the median for the entire record (dimensionless – spatial).
DL16	Low flow pulse duration. Compute the average pulse duration for each year for flow events below a threshold equal to the 25th percentile value for the entire flow record. DL16 is the median of the yearly average durations (number of days – temporal).
DL17	Variability in low pulse duration. Compute the standard deviation for the yearly average low pulse durations. DL17 is 100 times the standard deviation divided by the mean of the yearly average low pulse durations (percent – spatial).
DL18	Number of zero-flow days. Count the number of zero-flow days for the entire flow record. DL18 is the mean (or median - Use Preference option) annual number of zero flow days (number of days/year – temporal).
DL19	Variability in the number of zero-flow days. Compute the standard deviation for the annual number of zero-flow days. DL19 is 100 times the standard deviation divided by the mean annual number of zero-flow days (percent – spatial).
DL20	Number of zero-flow months. While computing the mean monthly flow values count the number of months in which there was no flow over the entire flow record (percent – spatial).
DH1	Annual maximum daily flow. Compute the maximum of a 1-day moving average flow for each year. DH1 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DH2	Annual maximum of 3-day moving average flows. Compute the maximum of a 3day moving average flow for each year. DH2 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DH3	Annual maximum of 7-day moving average flows. Compute the maximum of a 7 day moving average flow for each year. DH3 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DH4	Annual maximum of 30-day moving average flows. Compute the maximum of 30 day moving average flows. Compute the maximum of a 30-day moving average flow for each year. DH4 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DH5	Annual maximum of 90-day moving average flows. Compute the maximum of a 90day moving average flow for each year. DH5 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DH6	Variability of annual maximum daily flows. Compute the standard deviation for the maximum 1-day moving averages. DH6 is 100 times the standard deviation divided by the mean (percent – spatial).
DH7	Variability of annual maximum of 3-day moving average flows. Compute the standard deviation for the maximum 3-day moving averages. DH7 is 100 times the standard deviation divided by the mean (percent – spatial).
DH8	Variability of annual maximum of 7-day moving average flows. Compute the standard deviation for the maximum 7-day moving averages. DH8 is 100 times the standard deviation divided by the mean (percent – spatial).
DH9	Variability of annual maximum of 30-day moving average flows. Compute the standard deviation for the maximum 30-day moving averages. DH9 is 100 times the standard deviation divided by the mean (percent – spatial).
DH10	Variability of annual maximum of 90-day moving average flows. Compute the standard deviation for the maximum 90-day moving averages. DH10 is 100 times the standard deviation divided by the mean (percent – spatial).
DH11	Annual maximum of 1-day moving average flows divided by the median for the entire record. Compute the maximum of a 1-day moving average flow for each year. DL11 is the mean of these values divided by the median for the entire record (dimensionless – temporal).
DH12	Annual maximum of 7-day moving average flows divided by the median for the entire record. Compute the maximum daily average flow for each year. DL12 is the mean of these values divided by the median for the entire record (dimensionless – temporal).
DH13	Annual maximum of 30-day moving average flows divided by the median for the entire record. Compute the maximum of a 30-day moving average flow for each year.

	DL13 is the mean of these values divided by the median for the entire record (dimensionless – temporal).
DH14	Flood duration. Compute the mean of the mean monthly flow values. Find the 95th percentile for the mean monthly flows. DH14 is the 95th percentile value divided by the mean of the monthly means (dimensionless – spatial).
DH15	High flow pulse duration. Compute the average duration for flow events with flows above a threshold equal to the 75th percentile value for each year in the flow record. DH15 is the median of the yearly average durations (days/year – temporal).
DH16	Variability in high flow pulse duration. Compute the standard deviation for the yearly average high pulse durations. DH16 is 100 times the standard deviation divided by the mean of the yearly average high pulse durations (percent – spatial).
DH17	High flow duration. Compute the average duration of flow events with flows above a threshold equal to the median flow value for the entire flow record. DH17 is the average (or median - Use Preference option) duration of the events (days – temporal).
DH18	High flow duration. Compute the average duration of flow events with flows above a threshold equal to three times the median flow value for the entire flow record. DH18 is the average (or median - Use Preference option) duration of the events (days – temporal).
DH19	High flow duration. Compute the average duration of flow events with flows above a threshold equal to seven times the median flow value for the entire flow record. DH19 is the average (or median - Use Preference option) duration of the events (days – temporal).
DH20	High flow duration. Compute the 75th percentile value for the entire flow record. Compute the average duration of flow events with flows above a threshold equal to the 75th percentile value for the median annual flows. DH20 is the average (or median - Use Preference option) duration of the events (days – temporal).
DH21	High flow duration. Compute the 25th percentile value for the entire flow record. Compute the average duration of flow events with flows above a threshold equal to the 25th percentile value for the entire set of flows. DH21 is the average (or median - Use Preference option) duration of the events (days – temporal).
DH22	Flood interval. Compute the flood threshold as the flow equivalent for a flood recurrence of 1.67 years. Determine the median number of days between flood events for each year. DH22 is the mean (or median - Use Preference option) of the yearly median number of days between flood events (days – temporal).
DH23	Flood duration. Compute the flood threshold as the flow equivalent for a flood recurrence of 1.67 years. Determine the number of days each year that the flow remains above the flood threshold. DH23 is the mean (or median - Use Preference option) of the number of flood days for years in which floods occur (days – temporal).
DH24	Flood-free days. Compute the flood threshold as the flow equivalent for a flood recurrence of 1.67 years. Compute the maximum number of days that the flow is below the threshold for each year. DH24 is the mean (or median - Use Preference option) of the maximum yearly no-flood days (days – temporal).
TA1	Constancy. Constancy is computed via the formulation of Colwell (see example in Colwell 1974). A matrix of values is compiled where the rows are 11 flow categories and the columns are 365 (no February 29th) days of the year. The cell values are the number of times that a flow falls into a category on each day. The categories are: log(flow) < .1 x log(mean flow) .1 x log(mean flow) <= log(flow) < .25 x log(mean flow) .25 x log(mean flow) <= log(flow) < .5 x log(mean flow) .5 x log(mean flow) <= log(flow) < .75 x log(mean flow) .75 x log(mean flow) <= log(flow) < 1.0 x log(mean flow) 1.0 x log(mean flow) <= log(flow) < 1.25 x log(mean flow) 1.25 x log(mean flow) <= log(flow) < 1.5 x log(mean flow) 1.5 x log(mean flow) <= log(flow) < 1.75 x log mean flow) 1.75 x log(mean flow) <= log(flow) < 2.0 x log(mean flow) 2.0 x log(mean flow) <= log(flow) < 2.25 x log(mean flow) log(flow) >= 2.25 x log(mean flow). The row totals column totals and grand total are computed. Using the equations for Shannon information theory parameters constancy is computed as: 1- (uncertainty with respect to state) log (number of state)(dimensionless – spatial).
TA2	Predictability. Predictability is computed from the same matrix as constancy (see example in Colwell 1974). It is computed as: 1- (uncertainty with respect to interaction

of time and state - uncertainty with respect to time log (number of state) (dimensionless – spatial).

TA3 Seasonal predictability of flooding. Divide years up into 2-month periods (that is Oct-Nov Dec-Jan and so forth). Count the number of flood days (flow events with flows > 1.67-year flood) in each period over the entire flow record. TA3 is the maximum number of flood days in any one period divided by the total number of flood days (dimensionless – temporal).

TL1 Julian date of annual minimum. Determine the Julian date that the minimum flow occurs for each water year. Transform the dates to relative values on a circular scale (radians or degrees). Compute the x and y components for each year and average them across all years. Compute the mean angle as the arc tangent of y-mean divided by x-mean. Transform the resultant angle back to Julian date (Julian day – spatial).

TL2 Variability in Julian date of annual minima. Compute the coefficient of variation for the mean x and y components and convert to a date (Julian day – spatial). Note - 5-year flood threshold (Poff 1996) – For TL3 and TH3 compute the log10 of the peak annual flows. Compute the log10 of the daily flows for the peak annual flow days. Calculate the coefficients for a linear regression equation for logs of peak annual flow versus logs of average daily flow for peak days. Using the log peak flow for the 5-year recurrence interval (20th percentile) as input to the regression equation predict the log10 of the average daily flow. The threshold is 10 to the log10 (average daily flow) power (cubic feet per second).

TL3 Seasonal predictability of low flow. Divide years up into 2-month periods (that is Oct-Nov Dec-Jan and so forth). Count the number of low flow events (flow events with flows <= 5 year flood threshold) in each period over the entire flow record. TL3 is the maximum number of low flow events in any one period divided by the total number of low flow events (dimensionless – spatial).

TL4 Seasonal predictability of non-low flow. Compute the number of days that flow is above the 5-year flood threshold as the ratio of number of days to 365 or 366 (leap year) for each year. TL4 is the maximum of the yearly ratios (dimensionless – spatial).

TH1 Julian date of annual maximum. Determine the Julian date that the maximum flow occurs for each year. Transform the dates to relative values on a circular scale (radians or degrees). Compute the x and y components for each year and average them across all years. Compute the mean angle as the arc tangent of y-mean divided by x-mean. Transform the resultant angle back to Julian date (Julian day – spatial).

TH2 Variability in Julian date of annual maxima. Compute the coefficient of variation for the mean x and y components and convert to a date (Julian days - spatial).

TH3 Seasonal predictability of nonflooding. Computed as the maximum proportion of a 365-day year that the flow is less than the 1.67-year flood threshold and also occurs in all years. Accumulate nonflood days that span all years. TH3 is maximum length of those flood-free periods divided by 365 (dimensionless – spatial).

RA1 Rise rate. Compute the change in flow for days in which the change is positive for the entire flow record. RA1 is the mean (or median - Use Preference option) of these values (cubic feet per second/day – temporal).

RA2 Variability in rise rate. Compute the standard deviation for the positive flow changes. RA2 is 100 times the standard deviation divided by the mean (percent – spatial).

RA3 Fall rate. Compute the change in flow for days in which the change is negative for the entire flow record. RA3 is the mean (or median – Use Preference option) of these values (cubic feet per second/day – temporal).

RA4 Variability in fall rate. Compute the standard deviation for the negative flow changes. RA4 is 100 times the standard deviation divided by the mean (percent – spatial).

RA5 Number of day rises. Compute the number of days in which the flow is greater than the previous day. RA5 is the number of positive gain days divided by the total number of days in the flow record (dimensionless – spatial).

RA6 Change of flow. Compute the log10 of the flows for the entire flow record. Compute the change in log of flow for days in which the change is positive for the entire flow record. RA6 is the median of these values (cubic feet per second – temporal).

RA7	Change of flow. Compute the log10 of the flows for the entire flow record. Compute the change in log of flow for days in which the change is negative for the entire flow record. RA7 is the median of these log values (cubic feet per second/day – temporal).
RA8	Number of reversals. Compute the number of days in each year when the change in flow from one day to the next changes direction. RA8 is the average (or median - Use Preference option) of the yearly values (days - temporal).
RA9	Variability in reversals. Compute the standard deviation for the yearly reversal values. RA9 is 100 times the standard deviation divided by the mean (percent – spatial).