Using SHETRAN-Reservoir to assess the hydroecological impacts of climate change, reservoir decommissioning and increased native tree cover in the Upper Ehen catchment


Photo Courtesy of Environment Agency

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Declaration

I declare that the work reported in this dissertation was devised and carried out by myself and has not been accepted in any previous application for a degree. All information drawn from other sources, and any assistance received has been acknowledged in the appropriate place.

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Abstract

Climate change is projected to make flooding and droughts more common in the UK, which has considerable hydroecological implications. To help manage issues around water availability and environmental degradation, an increasing number of reservoirs are being decommissioned in the US and Europe, whilst nature-based solutions such as trees are increasingly being implemented in catchments. However, the majority of UK studies consider the hydrological impacts of climate change at a regional or national scale, rather than at the catchment scale, despite UK policy calling for a catchment-based approach to river management. Furthermore, little is known about the impacts of decommissioning reservoirs on UK river flows. Elsewhere, research has shown that trees can exacerbate low river flows. Recently developed hydrological modelling software SHETRAN-Reservoir was used to investigate how climate change, reservoir decommissioning and increased forest cover in the Upper Ehen catchment could affect discharge and freshwater ecology in the River Ehen SAC. UKCP18 emissions scenarios were used to simulate the effect of climate change on discharge. To assess the impacts of weir removal at Ennerdale Water, results from SHETRAN-Reservoir simulations were compared with those from SHETRAN-Standard simulations. Simulations were also run with current catchment forest cover (~19% and mainly coniferous) and planned future forest cover (~40% and mainly broadleaf). The emissions and weir removal scenarios resulted in more extreme discharge. The former caused a statistically-significant reduction in summer discharge, as well as the lowest 50% and 5% ($Q \geq Q95$) of river flows. The latter also resulted in a statistically significant decline in $Q \geq Q95$. The future forest cover scenario resulted in a slight decrease in discharge across the board. Though weir removal and increased forest cover would have ecological benefits, a decline in low flows and a change to river regime could have significant negative impacts on freshwater ecology. These results indicate that nature-based solutions which increase low flows are required in the Upper Ehen catchment alongside increased native forest cover which can reduce flood risk, sequester carbon and improve biodiversity. Additionally, it is recommended that greater weight is placed on drought amelioration in UK policy, as well as river flow management post-reservoir decommissioning. Integrated research that considers the impact of climate and land-use change on river flows and the wider hydroclimatological-hydroecological process chain is suggested, as well as that which models the impact of natural low flow management solutions and different tree species on catchment hydrology.

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1. Introduction

The 21st Century has so far been hotter than the three previous centuries and all areas of the UK are forecast to be even warmer by the end of the century (Met Office, 2019). Many of the most significant effects of climate change, caused by such warming, may involve water (Stern, 2006) and rivers could one of the most affected ecosystems (Millennium Ecosystem Assessment, 2005; Ormerod, 2009; Kernan *et al.*, 2010). How best to manage water and rivers as the climate change intensifies is therefore amongst the most crucial contemporary environmental and societal issues.

Other factors including land cover and reservoir operations are also likely to affect future river flows (Kay *et al.*, 2021). Therefore, planting trees and decommissioning reservoirs to allow rivers and lakes to return to more natural states could solve issues regarding climate change and water availability. The benefits of planting trees include flood mitigation (Broadmeadow and Nisbet, 2010); carbon sequestration (Cantarello *et al.*, 2011; Nijnik *et al.*, 2013) and improved biodiversity (Croci *et al.*, 2008; Lindenmayer *et al.*, 2010; Timonen *et al.*, 2010). Though reservoirs offer benefits such as controlling and managing water resources (Lehner *et al.*, 2011), in North America and Europe the rate of dam construction is slowing down (Chao *et al.*, 2008; Winemiller *et al.*, 2016). In the US, the rate of dams being decommissioned is increasing (O'Connor *et al.*, 2015) and across Europe it is estimated that 4000-5000 dams and weirs have been decommissioned (Gough *et al.*, 2018). With many dams built in the mid-20th Century, the cost to repair the aging infrastructure is greater than removal costs; there are concerns around their safety; and there is growing interest in restoring degraded ecosystems (Magilligan *et al.*, 2016).

This study used the recently developed modelling software SHETRAN-Reservoir to assess the impact of UK Climate Projections 18 (UKCP18) emissions scenarios, reservoir decommissioning and increased forest cover on the hydroecology of the Upper Ehen catchment, with a particular focus on low flows. The Upper Ehen catchment is located in Ennerdale in the western English Lake District and is the focus of a rewilding project overseen by the Wild Ennerdale Partnership. Climate change is expected to result in wetter winters and drier summers in the northern UK (Chan *et al.*, 2018). Meanwhile, Ennerdale Water, a reservoir impounded by a weir within the catchment, is due to be decommissioned in 2022 by United Utilities (Jacobs, 2017) and native forest cover in the catchment is to be doubled by the end of the century (Wild Ennerdale, 2018). The River Ehen downstream of the weir is a Special Area of Conservation (SAC) due to its population of *Margaritifera margaritifera* and *Salmo Salar* (JNCC, 2021). Due to this designation, the river is protected from low flows via environmental flow releases from the weir (Ricardo Confidential, 2020). Therefore, the catchment provides a useful context to investigate the impact of climate change, weir removal and increased forest cover on low river flows in the and the wider hydroecology of the catchment.

The study is of importance for the several reasons relating to research, management and policy:

- There are very few catchment-scale studies concerned with the impacts of climate change on river flows. UK water management policy calls for a catchment-based approach (CaBA; Robins *et al.*, 2017) but most studies have focused on regional and national-scale impacts (e.g. Christierson *et al.*, 2012; Prudhomme *et al.*, 2012; Sanderson *et al.*, 2012; Rudd *et al.*, 2019), the results of which do not necessarily translate to individual catchments. Furthermore, few studies have employed the UKCP18 used in this study.
- The development of SHETRAN-Reservoir allows for more accurate modelling of catchments which include reservoir operations (Hughes *et al.*, 2021) and can therefore be used to assess the impacts of reservoir decommissioning and weir removal on river flows in a UK context, an under-researched area. By using the software this study can address this gap, aid decision-making, inform environmental and water resource policy and test a recently developed hydrological model.
- Though trees are commonly used as a NbS to water management issues, they have been found to exacerbate low flows (e.g. Marc and Robinson, 2007; Bathurst *et al.*, 2018). This study can contribute to this research field and inform decision-making around targeted tree planting and NbS implementation.
- Greater weight in UK environmental policy is put on flood mitigation than drought amelioration and little policy refers to catchment management post-reservoir decommissioning. This study can contribute to policy by highlighting the need for greater emphasis to be put on drought mitigation, as well as the need to monitor river flows in catchments post-reservoir decommissioning.

Table 1 outlines the structure of the report.

Table 1 - Summary of the report structure

Chapter	Description
2. Aims and objectives	A statement of the study aims and objectives.
3. Literature Review	A review of current research regarding the hydroecological impacts of climate change; the hydroecological implications of reservoir decommissioning along with the current state of reservoir modelling; and the role of trees as NbS and how they affect the hydrological cycle. The chapter finishes with a overview of current UK climate change, water management and forest policy before identifying research gaps.
4. Methods	A detailed account of the study catchment, methodology and statistical analyses.
5. Results	An analysis of model performance, as well as the presentation of study results in relation to each of objectives from chapter 2.
6. Discussion	A discussion of the results in relation to the study aim and objectives, other research, recommended management practice and UK environmental policy, along with areas for further research arising from the study.
7. Conclusion	Concluding remarks are given.

2. Study Aim and Objectives

Figure 2 presents the study aim and objectives.

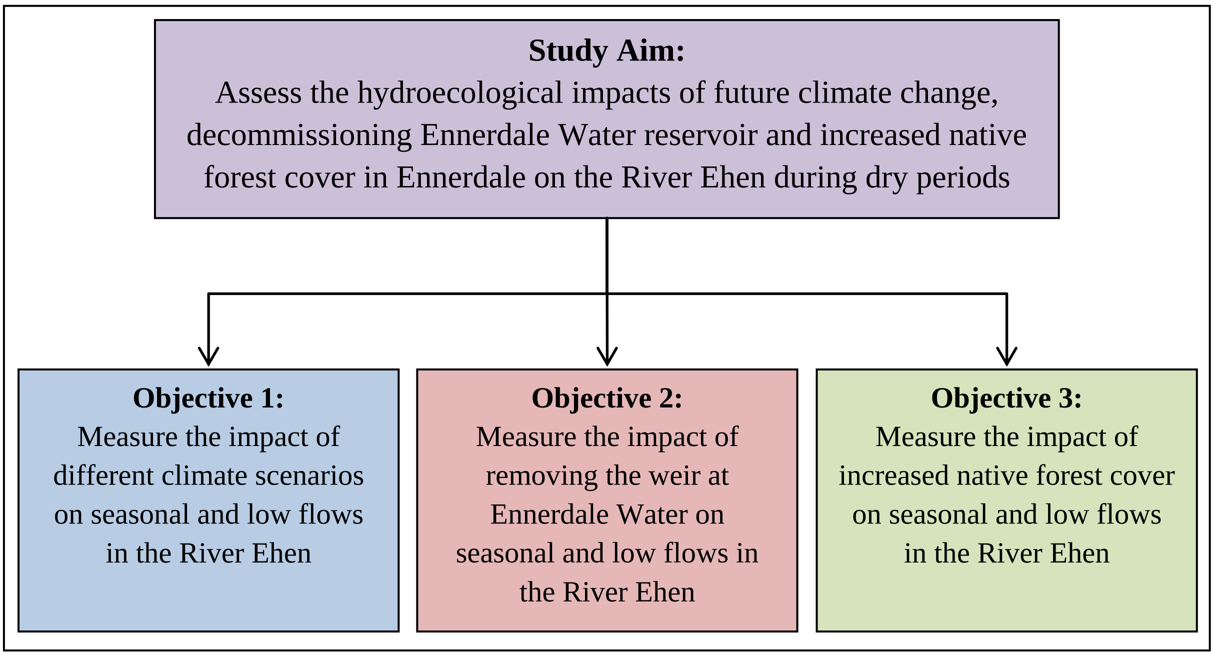

Figure 2 - Study aim and objectives

3. Literature Review

The following chapter reviews current research and policy related to: 1) the hydroecological implications of climate change in the UK; 2) hydrological modelling and the hydroecological impacts of decommissioning reservoirs; and 3) forests as nature-based solutions to hydrological issues caused by climate change. It concludes with research and policy gaps that this study aims to address.

3.1 Future climate change in the northwest England: hydroecological implications

According to the Met Office (2019), the average temperature from 2009 to 2018 across the UK was 0.3°C warmer than the 1981-2010 average from and 0.9°C higher than the 1961-1990 average. Furthermore, each of the ten warmest years since 1884 have occurred post-2002. The Central England Temperature dataset (figure 3.1) shows that 2009-2018 was approximately 1°C warmer than the pre-industrial period (1850-1900), which is in line with observed changes to global temperatures.

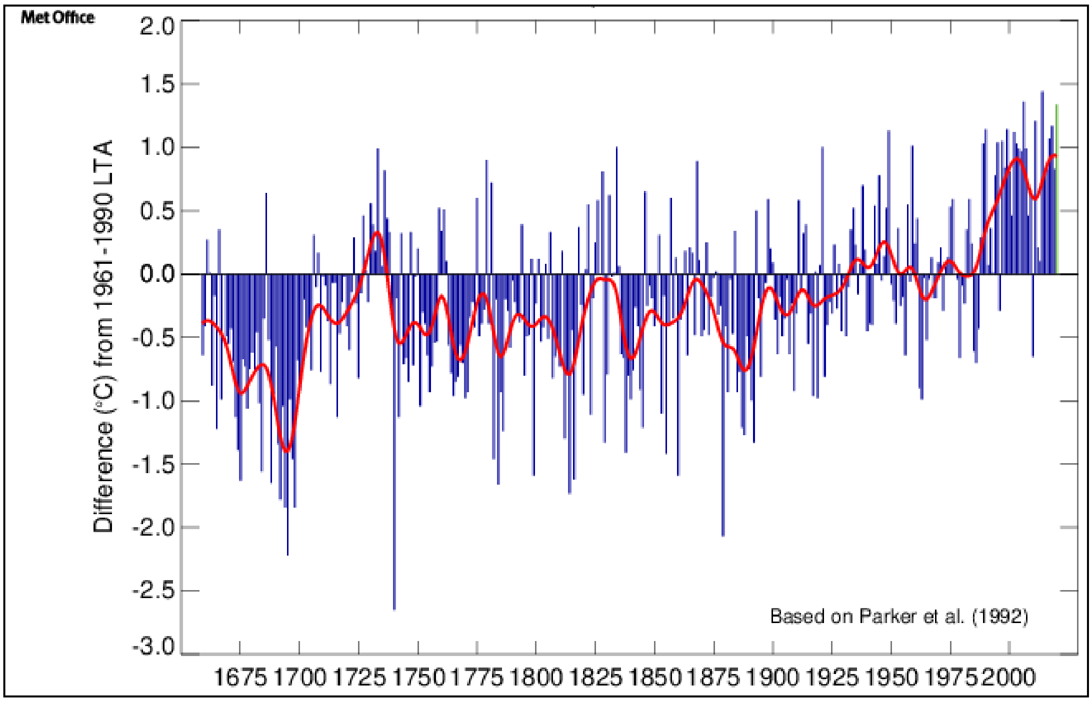

Figure 3.1 - Mean Central England Temperature annual anomalies, 1659 to November 2020 (Met Office, 2020)

Climate change caused by such warming affects the hydrological cycle through greater atmospheric water content and altered precipitation patterns (Bates *et al.*, 2008; Allan, 2011). Even under conservative emissions scenarios, future warming is expected to cause the global hydrological cycle to become more extreme, resulting in more periods of flooding and drought (Bates *et al.* 2008, Rockström *et al.*, 2009, Giorgi *et al.*, 2011) and greater water resources stress (Arnell *et al.*, 2011). It is therefore imperative that the hydroecological impacts of climate change are understood by land managers.

3.1.1 Projected changes to precipitation and evapotranspiration in northwest England

Together, precipitation and evapotranspiration are the foundation of the water balance and discipline of hydrology (Blackie and Eeles, 1985). UK Climate Projection 2018 (UKCP18; Lowe *et al.*, 2018; Met Office, 2019) precipitation projections for 2080-2099 in northwest (NW) England indicate a decrease in summer and an increase in winter (table 3.1.1.1). Chan *et al.* (2018) predict that summer mean daily precipitation will decrease by 30-50% by the end of the 21st Century in northern UK, although the intensity of extreme rainfall events will increase. In winter, an increase of 10-35% is forecast. UKCP 18 forecasts made for NW England are wider-ranging than this because they refer to the 10th-90th percentile spread (Lowe *et al.*, 2018).

Table 3.1.1.1 - UKCP18 predicted percentage changes to mean seasonal rainfall in NW England for low and high emission scenarios, 10th-90th percentile spread (Low *et al.*, 2018)

Season	Emission scenario	Mean rainfall percentage change
Summer	Low	-40% to +10%
	High	-60% to -10%
Winter	Low	-10% to +30%
	High	-10% to +50%

Whilst potential evapotranspiration (PE; the amount of water lost to the atmosphere assuming an unlimited supply of water; Federer *et al.*, 1996) projections depend highly on the calculation method (Prudhomme and Williamson, 2013), most predict annual PE across the UK will increase (Kay *et al.*, 2013; Prudhomme and Williamson, 2013). In Prudhomme and Williamson's (2013) study, the FAO56 method (Allen *et al.*, 1998) used to calculate projected changes to PE for NW England by 2070 (table 3.1.1.2) was deemed the most accurate and fell within the range of uncertainty.

Table 3.1.1.2 - Projected changes to potential evapotranspiration in northwest England by 2070 calculated from the FAO56 method (Prudhomme and Williamson, 2013)

Month	Percentage change
January	0-20
April	10-30
July	10-20
October	0-20

3.1.2 Projected changes to river flows in northwest England

Most current research has used the UK Climate Projections 2009 (UKCP09; Murphy *et al.*, 2009) to model UK river flows under future climate change (Kay *et al.*, 2021). Table 3.1.2.1 shows results of a selection of these that used Regional Climate Models (RGMs) for northwest (NW) England. Projected UK river flows derived from different climate models vary spatially and temporally due to studies employing different models and methodologies (Garner *et al.*, 2017). However, general trends are evident: increasing winter flows, declining summer flows and variability in spring and autumn flows.

Some studies have focused on droughts using UKCP09 projections. Harris *et al.* (2013) and Gosling (2014) found that climate change is likely to result in greater water scarcity in North Staffordshire and Scotland respectively. Rudd *et al.* (2019) projected hydrological drought severity (duration multiplied by intensity) to increase in Britain by the 2080s, though drought peak intensity will only worsen in the south east. Kay *et al.* (2018) investigated projected changes to 7- and 30-day low flows of 2- and 20-year return periods across Britain and found them to reduce, again particularly in the southeast. In Dobson *et al.*'s (2020) study, in which they employed a water resource model covering England and Wales, extreme droughts were projected to worsen by 2070-2099, with streamflows declining most in western regions of England and Wales. Collet *et al.* (2018) studied potential changes to floods and droughts in 281 catchments across Britain and with projected to increase, especially in western England and Wales and in northeastern Scotland.

Table 3.1.2.1 - Summary of changes to river flows in NW England presented in studies using UKCP09

Study	Season	Projected change to NW England river flows	Notes
Christierson *et al.* (2012) - used UKCP09 climate predictions (medium emissions) to model river flows across 70 sites in the UK in the 2020s.	Spring	-10% to +10%	A 30% decrease in median flows in August compared to the 1961-1990 baseline was the largest change.
	Summer	-10% to -40%	
	Autumn	-10% to +5%	
	Winter	5-15% increase	
Prudhomme *et al.* (2012) - projected changes across the UK by the 2050s (medium emissions) using 11 RGMs.	Spring	-20% to +40%	Summer decreases most extreme in northwest and autumn declines most common in southeast Annual flows remain the same.
	Summer	0% to -60%	
	Autumn	-80% to +60%	
	Winter	-20% to +20%	
Sanderson *et al.* (2012) - projected changes across the UK by the 2080s.	Spring	-5% to +15%	As above apart from increases to winter flows are projected to be larger than decreases to summer flows meaning, overall, annual flow averages will increase.
	Summer	-5 to -15%	
	Autumn	-5% to +5%	
	Winter	5-15% increase	
Charlton and Arnell, 2014 - projected changes to high (Q5) and low (Q95) flows in 6 catchments by the 2080s, including the Eden in east Cumbria.	Q95 could decline by 10-70% by the 2080s, with a 100% chance of decline. Q5 could change by -10% to +20%, with an 80% chance of an increase.		

Fewer studies have employed UKCP18 to forecast changes to river flows in the UK. Kay *et al.* (2021) investigated low and high flows of 5- and 20-year return periods in river basin districts across Britain. Projections for NW England (table 3.1.2.2) were in line with studies which have employed UKCP09: by 2076 high flows will increase whilst low flows will decrease.

Table 3.1.2.2 - Projected changes to scaled magnitude of high and low flows in NW England by 2076 (Kay *et al.*, 2021)

		5-year return period		20-year return period	
		1996	**2076**	**1996**	**2076**
High flows		1.3	1.5-1.7	1.6	2.0-2.3
Low flows	**7-day**	0.6	0.4-0.2	0.4	0.25-0.15
	30-day	0.6	0.3-0.2	0.3	0.2-0.1

3.1.3 Ecological implications of changing river flows

The impact of projected climate change and changing river flows on freshwater ecosystems is understudied (Garner *et al.*, 2017). Despite this, they should be considered one of the ecosystems most susceptible to climate change (Durance and Ormerod, 2007) because they are influenced by interrelated factors such as thermal regime, light, nutrient levels, discharge, species interactions, habitat connectivity, and land and water management practices (Laize *et al.*, 2014): the hydroclimatological-hydroecological process chain (Garner *et al.*, 2017). Numerous studies have found (the alteration of) flow regimes (long-term annual and monthly means, low and peak flows, daily and inter-annual variability, and the timing of flows; Döll and Zhang, 2010) to hold major influence over the biotic composition, structure, function and diversity of river ecosystems (Poff and Ward, 1989; Arthington and Pusey, 1993; Matthews and Marsh-Matthews, 2003; Poff and Zimmerman, 2010). Altered flow regimes also affect other abiotic elements of freshwater ecosystems, such as water quality and temperature and sediment transport (Döll and Zhang, 2010). For example, water temperature has been found to increase with lower flows (Pandolfo *et al.*, 2010).

UK-based studies concerned with the relationship between climate change and freshwater ecology have only been small-scale (Garner *et al.*, 2017) and tend to focus on water temperature. Climate change has been found to be a factor in determining macroinvertebrate abundance and assemblage (Durance and Ormerod, 2007); local extinctions (Durance and Ormerod, 2010); and declines in numbers of juvenile salmonids (Clews *et al.*, 2010).

International studies have considered the impacts of climate change on freshwater mussels with respect to water temperature and discharge. Changes to aquatic habitat thermal regime is associated with overall species survival (Pandolfo *et al.*, 2010) and can also affect mussels' resilience to additional stressors such as water pollution (Van Hattum *et al.*, 1993) and introduced non-native species (Ferreira-Rodríguez *et al.*, 2018). Low flows associated with severe droughts can reduce species richness and cause local extinction (Haag and Warren, 2008; Sousa *et al.*, 2018). Flooding due to extreme precipitation can scour mussel beds (Strayer, 1999) and disrupt recruitment (Haag, 2012).

3.1.4 UK climate change policy

UK policy calls for climate change mitigation through emissions reductions via the 2015 Paris Agreement and UK Climate Change Act 2008 (Committee on Climate Change, 2016; Fankhauser *et al.*, 2018) climate change adaptation. The 2010 Convention on Biodiversity (CBD) provides a framework to inform national policy on biodiversity (DEFRA, 2011a). Target 15 of the CBD highlights the need to improve ecosystem resilience and contribution to carbon stocks in order to help nature both adapt to and mitigate climate change. On a national scale, the Climate Change Act aims to improve climate change adaptation in the UK (Fankhauser *et al.*, 2018). The Act suggests a continual adaptation planning approach over a five-year cycle. This begins with a Climate Change Risk Assessment (CCRA), which is followed a year later by a National Adaptation Programme (NAP) that sets out how to manage risks. The most recent CCRA was in 2017, which was followed by the 2018-2023 NAP. Key risks in CCRA2017 which can be associated with low flows, as outlined by DEFRA (2018b), are:

- species and habitats unable to adapt to climate change;
- water scarcity and flooding affecting agriculture and wildlife; and
- increased seasonal aridity and wetness damaging soils.

NAP2018 highlights actions needed to respond to these risks and build ecological resilience to climate change, in tandem with DEFRA's (2018a) 25 Year Environmental Plan (25YEP; table 3.1.4).

Table 3.1.4 - Climate change adaptation objectives of NAP2018 and 25YEP (DEFRA 2018a; 2018b)

Conservation area	Actions	Links to 25YEP
Land, lakes and rivers	1. Protect and improve designated sites and important habitats. 2. Restore damaged ecosystems.	By 2042: 1. 75% of terrestrial sites in favourable condition. 2. Create or restore 500,000ha of habitat.
Water quality and abstraction	Reform approach to water abstraction in order to enhance natural resilience to drought and improve ecological health.	As part of the Water Industry National Environment Programme, water companies must address unsustainable abstraction.
Soils	Improved soil management and health.	By 2030: • England's soils must be managed sustainably; and appropriate soil metrics and management approaches must be developed. • Design a soil health index and protect peatlands by halting the use of peat in horticulture.
Forestry	Create new woodland and improve and connect existing woodland to improve resilience.	Increase woodland management and tree planting, and incentivise planting on private land.

3.2 Reservoir decommissioning in the UK: hydroecological implications

Dams are *"barriers used to obstruct the flow of water and create reservoirs, [which are] artificially created bod[ies] of water"* (Schmutz and Moog, 2018, p. 113). Dam construction dates back 5000 years but the majority of dams worldwide have been built since the Second World War to promote economic development (Schmutz and Moog, 2018). Globally the number of dams continues to rise (Couto and Olden, 2018) and there are nearly 59,000 registered large dams (580 of which are in the UK), which store over 14,000km^3 of water

(ICOLD, 2020). It is thought that there are also around 16 million small dams worldwide (Lehner *et al.*, 2011).

Reservoirs are created to meet the specific needs of a community, such as drinking water, industry, flood mitigation, hydropower, irrigation and recreation (World Commission on Dams, 2000; Schmutz and Moog, 2018). The primary reservoir operations - direct abstractions and releases through the use of controls such as sluice gates and overspills - alter catchment hydrology, particularly reservoir stage dynamics and river flow regimes (Hughes *et al.*, 2021). Table 3.2 presents the positive and negative impacts of dams.

Table 3.2 - Advantages and disadvantages of dams

Advantages	Disadvantages
• Enable the control and management of water resources (Lehner *et al.*, 2011). • Contribute to food production and provide energy through hydropower (World Commission on Dams, 2000). • Can act as barriers that prevent the spread of invasive species (Rahel, 2013). • Reservoirs can provide novel refuges for aquatic organisms during periods of low flows (Beatty *et al.*, 2017). • Can be used to maintain downstream flows during dry periods, e.g. UK water bodies are required to	• Can cause negative environmental impacts upstream, downstream and within reservoirs (Beck *et al.*, 2012) and have adverse impacts on freshwater biodiversity (Birnie-Gauvin *et al.*, 2017). • Act as barriers, preventing the downstream transportation of sediment and nutrients, as well as the migration of aquatic organisms such as fish (Lessard and Hayes, 2003; Meixler *et al.*, 2009). The upstream movement of organisms is crucial to healthy river ecosystem function (Pringle, 1997). 63% of rivers longer than 1000km worldwide are considered non free-flowing, with dams and reservoirs being the main cause of this fragmentation (Grill *et al.*, 2019). • Aquatic organisms often depend on specific thresholds of water level, velocity, timing, water temperature and dissolved oxygen to function so the altered flow rates caused by dams can have detrimental impacts (Bunn and Arthington, 2002; Lessard and Hayes, 2003; Dugan *et al.*, 2010). • Can also lead to increased greenhouse gas emissions through biomass and soil decomposition (St. Louis *et al.*, 2000; Mäkinen and Khan, 2010).

maintain river flows above a specified threshold set by the Environment Agency, in accordance with River Basin Management Plans (DEFRA, 2018b).	• Reduced flood frequency results in a decrease in downstream deposition crucial for the fertility of riparian areas (Shoemaker *et al.*, 2001; Lebel *et al.*, 2005), which leads to the loss of biodiversity (Winemiller *et al.*, 2016), as well as increased erosion (Kondolf, 1997).

3.2.1 Decommissioning reservoirs

In tandem with increasing numbers of reservoirs being decommissioned, there is a growing research field around dam removal (Bellmore *et al.*, 2017), particularly in North America. Ecological impacts of dam removal vary spatially and temporally (Bellmore *et al.*, 2019) and the geographical context of each dam is unique, which means environmental responses to removal are also likely to be unique (Foley *et al.*, 2017a). Despite this, the ecological responses to dam removal are controlled by universal physical and biological links and occur upstream, downstream and within the former reservoir (Bellmore *et al.*, 2019; table 3.2.1).

The impacts of decommissioning reservoirs on discharge is not as well-researched, and most studies focus on large dams, despite the majority of dam removals involving small dams (Hart *et al.*, 2002). In their review of the outcomes of small dam removal in the US, Tonitto and Riha (2016) found the effects on flooding to be mixed. Dam removal has been found to increase (Endreny and Higgins, 2008), decrease (Nislow *et al.*, 2002) or cause little change to flood risk (Roberts *et al.*, 2007; Wyrick *et al.*, 2009). Over a four-year period, Sullivan *et al.* (2018) found that median daily discharge from May to September increased by 89% after dam removal in the Olentangy River, USA.

Little has been written on the impacts of dam removal on low flows, however. In a review of 29 dammed rivers in the US, Williams and Wolman (1984) found the impacts of dam construction on mean daily and low flows to vary depending on dam release policy. Other studies have found dams to generally increase the frequency, magnitude and duration of low flows owing to controlled releases (Magilligan and Nislow, 2005). It can be inferred from this that dam removal also has a variable effect on low flows, with a tendency to decrease the magnitude of low flows. However, it is necessary to assess the context-specific impacts of dam removal, particularly small dams, on low flows in catchments and there is a dearth of such studies in the UK.

Table 3.2.1 - Summary of the ecological benefits of dam removal within a catchment

Catchment section	Impacts of dam removal

Upstream of reservoir	• Restored longitudinal connectivity, which enables the recolonisation of upstream habitats by fish, invertebrates and commensal microorganisms (Service, 2011; O'Connor *et al.*, 2015; Bellmore *et al.*, 2019).
	• Improved longitudinal connectivity also contributes to upstream increases in:
	  ○ species richness (Burdick and Hightower, 2006; Catalano *et al.*, 2007; Burroughs *et al.*, 2010; Magilligan *et al.*, 2016);
	  ○ life history diversity (Morita *et al.*, 2000; Pascual *et al.*, 2001; Pess *et al.*, 2008; Quinn *et al.*, 2017); and
	  ○ the delivery of nutrients and organic matter (Tonra *et al.*, 2015).

Within reservoir	• An increase in riparian vegetation (Shafroth *et al.*, 2002; Orr and Stanley, 2006) as the reservoir level declines and is replaced by a river. • A change from lentic-adapted fish to lotic fish and from pelagic (e.g. plankton) to benthic producers and consumers, resulting from a decrease in water depth and an increase in water velocity (Smokorowski *et al.*, 2011; Foley *et al.*, 2017b).
Downstream of reservoir	• Increased river connectivity and a return to a more natural flow regime results in an increase in riparian and aquatic species (Poff *et al.*, 1997; Wohl *et al.*, 2015). • In the short-term, the release and deposition of sediment (which may contain toxins or heavy metals) can bury benthic and riparian organisms (Bednarek, 2001; Stanley *et al.*, 2002; Sethi *et al.*, 2004; Orr *et al.*, 2006) as well as increase water turbidity to the detriment of fish species (Kjelland *et al.*, 2015). • Over time this initial deposit is eroded and the release of reservoir sediment increases the amount of nutrients and organic matter in the water, which can benefit riparian vegetation and benthic producers and consumers in the long-term (Bellmore *et al.*, 2019).

Overall, it is not a simple case of saying that if reservoirs are decommissioned, equilibrium will necessarily be restored. Dam removal involves many complex, multi-scale trade-offs, and this must be taken into account by decision makers (Roy *et al.*, 2018). Climate change is projected to alter river discharge in every populated basin across the world (Palmer *et al.*, 2009), which adds further complexity because reservoirs do provide hydroecological benefits (Beatty *et al.*, 2017), as well as improved water security (Ehsani *et al.*, 2017) in the face of a changing climate. Greater consideration of the impacts of climate change is needed by decision makers in terms of dam removal (Beatty *et al.*, 2017; Hughes *et al.*, 2021), particularly in the UK.

Few studies consider the hydrological impacts of climate change alongside reservoir decommissioning. Battle *et al.* (2016) found that the number of salmon increased with dam removal in Taiwan due to the increased capacity for them to migrate upstream, whilst increased typhoon rainfall intensity associated with climate change had a detrimental impact. Dugdale *et al.* (2017) modelled the impacts of climate change and dam decommissioning a dam on water temperature in the Saint John River, Canada. Whilst the former saw an increase in temperature, which has important ecological implications for fish, the latter resulted in cooler, more variable temperatures. However, there is little research on the impacts of dam removal on downstream discharge or low flows under climate change in the UK.

3.2.2 UK water management policy

As Robins *et al.* (2017) explain, until replaced by new legislation post-Brexit, water planning and policy in the UK is currently guided by the EU directives, namely the Water Framework Directive (WFD; 2000/60/EC, European Commission, 2000), the Floods Directive (2007/60/EC, European Commission, 2007) and, to a lesser extent, the Habitats Directive (92/43/EC, European Commission, 1992). The WFD sets water quality standards based on ecosystem health and how the water is ultimately used (Robins *et al.*, 2017). It provides a framework for greater integration of water and land management with regards to improving the aquatic environment (DEFRA, 2014a). The primary means of achieving this is via River Basin Management Plans (RBMPs), which provide details on current status and set targets for the 15 River Basin Districts (RBDs; Robins *et al.*, 2017). To ensure the requirements WFD and Floods Directive are met, water policy was reformed in 2011 (DEFRA, 2011b, 2011c), particularly regarding the 'catchment-based approach' (CaBA) and abstractions (Robins *et al.*, 2017). The adoption of the CaBA was intended to lead to more appropriate RBMPs (Robins *et al.*, 2017) in response to criticisms (Starkey and Parkin, 2015). RBMPs now involve a broader cross-section of stakeholders and integrate plans and processes (DEFRA, 2014a). The scale of planning and engagement has also moved away from the 10 RBDs to 93 individual catchments (Robins *et al.*, 2017).

The 2011 reforms to abstraction management included flow-based controls to protect the environment and catchment abstraction reviews set in the context of overall catchment management (DEFRA, 2016). Several policy documents, summarised in table 3.2.2, outline the UK's plans to improve its water management, though this is through updated abstractions licences as opposed to decommissioning reservoirs. Indeed, although decommissioning reservoirs is included in the England and Wales Reservoirs Act 1975, as amended by the Water Management Act 2010 (British Dam Society, 2016), very little UK policy relates directly to decommissioning reservoirs. Unlike flooding, there is no exclusive EU directive for drought and no UK policy equivalent to the Flood Risk Management Plans produced for each RBD under the Floods Directive (Robbins *et al.*, 2017), despite drought in the UK having had significant environmental, agricultural and societal impacts in the past (Kendon *et al.*, 2013). Furthermore, drought has been calculated to cost the EU and UK region 7.4-14.2 €billion/year (Cammalleri *et al.*, 2020).

Table 3.2.2 - Summary of UK water abstraction plans

Biodiversity 2020 (DEFRA, 2011a)	Calls for: • a river basin planning approach, in alignment with the EU Water Framework Directive (WFD), in order to protect water quality and quantity; and • reform to water abstraction in order for abstractors to better meet water needs and protect ecosystem functioning, as well as reduce unsustainable abstractions.
Water Abstraction Plan 2017 (DEFRA, 2020)	• Sets out the government's plans to reform water abstraction in order to conserve the environment and improve water access, calls echoed in DEFRA's 25YEP and NAP (see section 3.1.4). • Calls for a CaBA that has a more localised focus and will produce updated abstraction licensing strategies. Such strategies include: ○ controls on licences to protect the environment, particularly during dry periods;

	○ limiting licences to prevent over-abstractions; ○ fine-tuning the use of surface and groundwater sources for more efficient water use; and ○ supporting water trading where it is most required to allow abstractors to share water access.

3.2.3 Reservoir modelling

Given the debate surrounding reservoirs under climate change and recent updates to UK reservoir policy, understanding their role in the hydrological cycle is crucial. Reservoir models offer a solution in this regard. Fundamentally, reservoir modelling is about routing flows through reservoirs and calculating storage and outflow. Reservoir processes (figure 3.2.3) and related models generally fall into three categories: hydrological, hydraulic and water resource models (Hughes *et al.*, 2020). Table 3.2.3 reviews the current range of reservoir models.

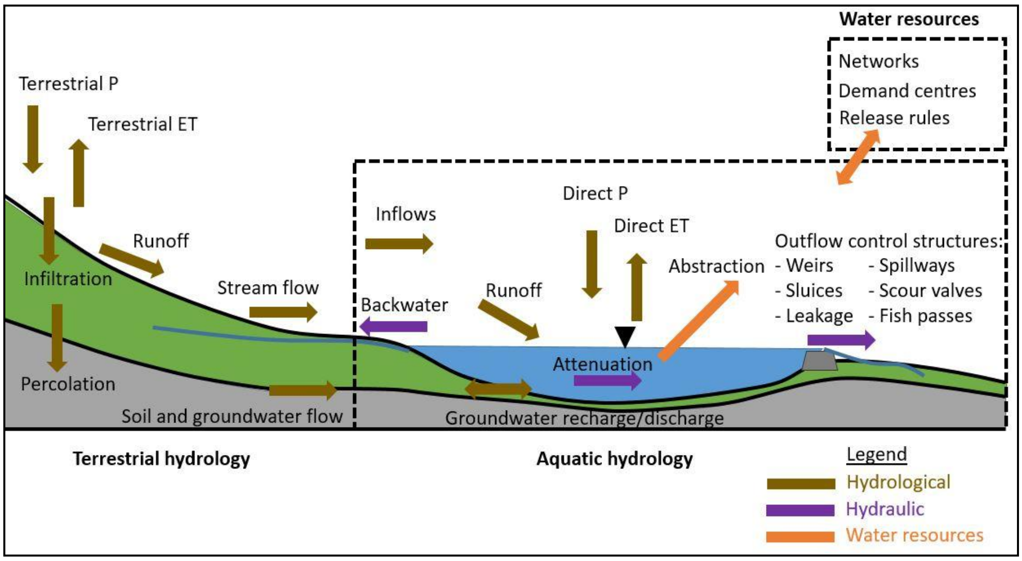

Figure 3.2.3 - Diagram illustrating reservoir hydrology processes (Hughes *et al.*, 2021)

Table 3.2.3 - Review of current reservoir models

Model type	Advantages	Disadvantages	Examples
Simple hydraulic	• Does not rely on detailed and expensive bathymetric data (Goodell and Wahlin, 2009).	• Take processes or groups of processes in isolation (Hughes *et al.*, 2020). • If not fully coupled, important feedbacks from the reservoir (such as storage, backwater effects and groundwater) are unaccounted for (Fleischmann *et al.*, 2019).	Goodell and Wahlin (2009) - level pool routing.
Simple hydrological	• Can be used to generate inflows that are put through a reservoir model which simulates stores and outflows (Wen *et al.*, 2014). • Can simulate storage and downstream discharge (Hughes *et al.*, 2020).		Wen *et al.* (2014) - Honghu Lake, China - TOPMODEL.
Basic lumped or distributed hydrological/ hydraulic	• With their fixed storage-discharge relationships, are useful for lakes or unmanaged reservoirs (Hughes *et al.*, 2020).	Do not consider: • the complex hydrological processes at play in a catchment; or • the dynamic storage-discharge relationships associated with managed	HBV - semi-distributed model - routes runoff into lakes and reservoirs and uses storage-discharge relationship to calculate outflows, (Bergström, 1992). • e.g. Mendez and Calvo-Valverde (2016) - three subcatchments of the Aquacaliente River, Costa Rica - inconsistent results.

Complex lumped or distributed hydrological/ hydraulic	Can simulate: • thermodynamics, evaporation, and outflows using empirical stage-discharge relationships; and • hydraulic connections between lakes in the form of backwater effects (Gronewold *et al.*, 2017).	reservoirs (Hughes *et al.*, 2020).	Advanced Hydrological Prediction System for the American Great Lakes (Gronewold *et al.*, 2017).
Complex lumped or distributed reservoir	• Able to reflect - to some extent - operations that transfer water in and out of a reservoir, therefore changing storage-discharge dynamics, do exist (Hughes *et al.*, 2020).	• based on only basic mechanisms of hydrological processes, as well as a limited number of examples of reservoir operations (e.g. Zhang *et al.*, 2012).	Some versions of the lumped model SWAT simulate reservoir operations, such as SWAT2005 (e.g. Zhang *et al.*, 2012) Todorović *et al.* (2019) - 3DNet-Catch semi-distributed model.
Reservoir suite	• When coupled can simulate all of the necessary reservoir hydrology processes (Hughes *et al.*, 2020).	• Highlights the need for a single, integrated programme (Hughes *et al.*, 2020).	MIKE SHE generates runoff; MIKE HYDRO River simulates operations (e.g. sluice gates); and MIKE HYDRO Basin simulates reservoir operations (DHI, 2017a; 2017b; 2017c).

Water resource	• Include reservoir processes, such as context-specific control rules, as well as more detailed formulations based around reservoir outflows, e.g. leakage, spillage, pumping ad sluice flow (Hughes *et al.*, 2020).	• Tend to rely on simplified catchment hydrology (Ampitiyawatta, 2020; Hughes *et al.*, 2020).	Klipsch and Hurst, (2013) - HEC-ResSim - includes a watershed setup module to generate inflows from precipitation series. • e.g. Ampitiyawatta (2020) - Qingjiang Cascade Reservoirs, China.

Table 3.2.5 highlights the need for a model that simulates both catchment hydrology and reservoir processes in detail. SHETRAN-Reservoir (SHETRAN v.4.4.6; Newcastle University, 2020a) has been developed to provide an integrated, distributed model package that includes reservoir operations involving dynamic outflow structures (Hughes *et al.*, 2021). The model emulates other models by using an elevation-discharge relationship to simulate weir outflows but it also includes a function that enables the addition of more than one discharge relationship to reflect sluice operations. It was found that the model improves the SHETRAN-Standard model performance (SHETRAN v.4.4.5; Newcastle University, 2020b), which does not have the capacity to simulate reservoir operations, particularly during dry spells where outflows are dominated by reservoir operations. The model's ability to answer questions related to the decommissioning of reservoirs, climate change and ecological conservation could therefore be significant (Hughes *et al.*, 2021).

3.3 Forests: a nature-based solution for low-flow hydrology?

Through direct interception and the evapotranspiration of precipitation (Neal *et al.*, 1991; Iroumé and Huber, 2002; Calder *et al.*, 2003), trees reduce the amount of water that reaches the ground (Ray and Nicoll, 1998; Nisbet, 2005; Dixon *et al.*, 2014).

Infiltration is the process by water enters the ground via tree roots (Christen, 2007). Pathways in the soil are opened up, which increases the amount of water that can infiltrate it and be stored, thus decreasing the occurrence of rapid surface runoff (Thomas and Nisbet, 2016). Protection afforded by tree cover, the build-up of soil organic matter and healthy soil structure from leaf litter (Bischoff *et al.*, 2015) improve macroporosity and encourage infiltration (Bird *et al.*, 2003). Infiltration rates in areas of forest have been found to be greater than that of grasslands (Carroll *et al.*, 2004; Marshall *et al.*, 2009, 2014).

To varying extents, trees have been found to decrease surface runoff in a catchment (Blackie, 1993; Marshall *et al.*, 2009, 2014). Hydrological roughness - caused by trunks, branches, roots and debris - filters and impedes flood water, which in turn increases water storage and slows down flow rates (Thomas and Nisbet, 2007).

3.3.1 Forests as nature-based solutions for water management

Due to their role in the hydrological cycle, trees are commonly used as a nature-based solution (NbS) in water management. The International Union for Conservation of Nature's defines NbS as *"actions to protect, sustainably manage, and restore natural or modified ecosystems, that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits"* (Cohen-Shacham *et al.*, 2016, p. 5). The socio-environmental benefits of forests are both crucial and manifold (Costanza *et al.*, 1998). These benefits include: flood mitigation (Broadmeadow and Nisbet, 2010); helping to preserve people's physical and mental wellbeing (Ward Thompson *et al.*, 2005; O'Brien and Morris, 2014); providing ecosystem services such as shelter and food (Schütz *et al.*, 2010); carbon sequestration (Cantarello *et al.*, 2011; Nijnik *et al.*, 2013) and improved biodiversity (Croci *et al.*, 2008; Lindenmayer *et al.*, 2010; Timonen *et al.*, 2010).

This array of benefits has seen forests increasingly used as a NbS to water management issues under climate change. Forests and woodland management practices have long been associated with affecting both the quantity and timing of stream flows (McCulloch and Robinson, 1993). In a UK context, the majority of studies concerned with forests as NbS to water-related issues tend to revolve around flood mitigation (e.g. Linstead and Gurnell, 1999; Robinson *et al.*, 2003; O'Connell *et al.*, 2004; Sakals *et al.*, 2006; Nisbet and Thomas, 2008; Dixon *et al.*, 2016). Whilst it is widely accepted that forests can reduce flood magnitude, they can exacerbate low flows during droughts (McCulloch and Robinson, 1993) due to increased interception and evapotranspiration losses (Vogt and Somma, 2000; Robinson *et al.*, 2003); a lowered water table (Robinson *et al.*, 2003); and reduced soil moisture reserves (Hudson, 1988; Robinson and Cosandey, 2002). Two UK-based studies of upland catchments - one in

Wales (Marc and Robinson, 2007) and one in northern England (Bathurst *et al.*, 2018) found that found that annual runoff is 18% and 24% less in a catchment with 100% forest cover than 100% grassland respectively. Another study of European forests, including four in the UK, found that low flows decline as forests grow (Robinson *et al.*, 2003). Other studies in the US and UK have found that low flows are increased after forests are cleared, particularly during summer (Hornbeck *et al.*, 1993; Johnson and Black, 1997; Robinson and Dupeyrat, 2003).

Forest type has been found to influence the extent to which forests affect low flows. Whilst conifers respond quickly during dry periods by regulating stomatal opening, broadleaves sustain high levels of transpiration when leaves are present, regardless of soil moisture (Chirino *et al.*, 2011; Link *et al.*, 2014). However, whilst some studies have found no difference in discharge (e.g. Hisada *et al.*, 2012), others have found it to be higher in conifer forests (e.g. Nainar *et al.*, 2020) or in broadleaf forests (e.g. Fahey and Jackson, 1997; Komatsu *et al.*, 2009). Furthermore, though it is not well understood, physiological and morphological differences within and between tree species has also been found to play a significant role in forest ecosystem water balance, which has implications for forest management in light of climate change (Aranda *et al.*, 2012).

3.3.2 UK forest policy in relation climate change and water

Forestry in the UK is based on the central theme of sustainable forest management, a concept that has arisen from an international context (table 3.3.2.1) and an approach underscored by the UK Forestry Standard (UKFS; Forestry Commission, 2017). Nationally, the devolved governments set their own forest programmes and strategies. In England, DEFRA sets the forest programme and incentives, and Forestry England is responsible for its implementation. Further refinement takes place at the local and regional level (Forestry Commission, 2017).

Table 3.3.2.1 - Summary of international commitments relevant to UK forest management (Forestry Commission, 2017)

International commitment	Relevance to UK forestry
1992 Rio 'Earth Summit'	Concept of sustainable development - which forms the basis of forest management globally - introduced. Nations agreed on three legally-binding conventions agreed, all of which have relevance to forest management: • UN Framework on Climate Change (UNFCC). • UN Convention on Biological Diversity (UNCBD). • UN Convention to Combat Desertification (UNCCD). Also introduced the UN Statement of Forest Principles.
United Nations Forum on Forests (UNFF)	Established in 2000. In 2006, four global forest objectives set (table 3.3.2.2) for nations to have made progress towards by 2015. The 2007 UNFF session brought about a non-legally binding instrument for all forest types.
UN Sustainable Development Summit	In 2015, 17 Sustainable Development Goals agreed, of which Goal 15 (*Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss)* relates directly to forests.

Table 3.3.2.2 - 2006 UNFF global objectives on forest (source: Forestry Commission, 2017)

Reverse the loss of forest cover worldwide through sustainable forest management, including protection, restoration, afforestation and reforestation, and increase efforts to prevent forest degradation.
Enhance forest-based economic, social and environmental benefits, including by improving the livelihoods of forest-dependent people.
Increase significantly the area of protected forests worldwide and other areas of sustainably managed forests, as well as the proportion of forest products from sustainably managed forests.
Reverse the decline in official development assistance for sustainable forest management and mobilise significantly increased new and additional financial resources from all sources for the implementation of sustainable forest management.

To combat climate change, the UK government has committed to planting 30 million trees by 2025 in order to reach Net Zero emissions (Ares and Uberoi, 2020). The UKFS has specific guidance related to forest management and water stemming from the WFD, which is applied to legislation in England and Wales through the Water Environment Regulations 2003 (Forestry Commission, 2011). The vast majority of guidance relates to water quality and flood mitigation. However low flow management is included through the following: *"Where new woodlands are proposed, the sensitivity of downstream water bodies and wetlands to a reduction in water quantity should be considered; where this is an issue, advice should be sought from the water regulatory authority and conservation agency."* (Forestry Commission, 2011, p. 26).

3.4 Research gap

Though there is uncertainty surrounding projections, climate change in the UK is likely to have significant hydroecological impacts. One of these is an increase in the frequency and magnitude of drought and low flows across the UK, which could have negative consequences for freshwater ecology. There is therefore scope for studies to investigate these impacts at the catchment scale in order to correspond with the CaBA and inform catchment management. Furthermore, few studies have applied UKCP18 climate projections in order to assess the impacts of climate change on river flows and update forecasts.

Against this backdrop of climate change, as well as an increase in the number of reservoirs being decommissioned in Europe and North America, there is also a need for studies to consider the hydroecological implications of dam removal in the UK, which is under-researched. Recent advancements in the effectiveness of reservoir modelling through SHETRAN-Reservoir offer a new opportunity to model future changes to restored catchments under climate change conditions, as well as develop an understanding of the effectiveness and applications of the model.

Given the benefits that reservoirs can provide during periods of drought via environmental flows, there is also scope for studies to consider how land managers can replace these through NbS after reservoirs have been decommissioned. Trees are widely accepted to regulate water transfer and therefore offer a solution. However, research has found them to exacerbate low flows. It is therefore necessary to explore how increased forest cover affects low flows in restored rivers under climate change.

Finally, international and UK policy is committed to improving hydroecological resilience to climate change. This study provides an opportunity to explore how catchment and forest restoration can achieve this. However, given that the majority of policy is concerned with water quality or flood mitigation, low flow hydrology is under-represented. This study can help to address this by considering whether there is a need for greater low flow management and policy.

4. Methods

This chapter details the study methodology by describing 1) the study catchment, 2) catchment model setup, 3) scenario development, 4) statistical analyses and 5) study constraints.

4.1 Study catchment

The Upper Ehen catchment (described in table 4.1) is located in the west of the Lake District National Park in northwest England (figure 4.1.1).

Table 4.1 - Features of the Upper Ehen catchment

Catchment	• Area = 44.2km² (NRFA, 2021). • Encompasses a glacial valley, resulting in steep relief.
Geology	• The entire catchment is underlain by completely impervious bedrock (of the Skiddaw Slate and Borrowdale Volcanic groups; NRFA, 2021). • Superficial deposits (alluvium, till and glacial deposits) account for ~13% of the total area (NRFA, 2021).
Land cover	• Grassland constitutes ~70% of the catchment. • Forest covers ~19% (Wild Ennerdale, 2018). • Lakes, tarns and watercourses: 8% (Wild Ennerdale, 2018).
Hydrology	• The catchment outlet is the River Ehen to the west, which is gauged at Bleach Green gauging station (NRFA, 2021). • The Ehen drains the catchment's glacial lake - Ennerdale Water. The lake's area is 3km² (~6.8% of the catchment's area) and has a mean depth of ~17.8m (Environment Agency, 2021). At its deepest, the lake is ~43m (Ricardo Confidential, 2020). • Ennerdale Water's main inflow is the River Liza, which flows into the lake's north-eastern corner (Ricardo Confidential, 2020).
Reservoir operations	• Ennerdale Water is a raised lake owing to the broad-crested weir located at its confluence with the River Ehen on its western edge, a structure which consists of a weir crest, fish pass with penstock and wing walls (figure 4.1.2). • United Utilities are the reservoir operator and operations comprise environmental flow releases and abstractions:

	◦ Environmental flows are maintained at 60,000-80,000 m^3/d (~0.69-0.93 m^3/s) depending on lake level. They are released by gravity through the fish pass and controlled by an automated penstock. If the lake level is too low for the flows to be maintained solely by gravity, additional water is abstracted and released by pumps from the pumping station downstream of the weir (Ricardo Confidential, 2020). ◦ Abstraction occurs via a submerged gravity-fed intake ~250m inside the lake perimeter. The abstractions licence permits abstractions up to 47,420m^3/d (~0.55m^3/s) down to lake levels of 110.61 metres above ordnance datum (mAOD; or 1.70m below weir crest level (bwcl; Ricardo Confidential, 2020). On average 25,000m^3/d (~0.29m^3/s) are abstracted and, in dry periods, this is offset by abstractions from local boreholes (Ricardo Confidential, 2020).
Designations	• The River Ehen is a Special Area of Conservation (SAC) and Ennerdale is a Site of Special Scientific Interest (SSSI; United Utilities, 2018). • The river's population of *Margaritifera margaritifera* is the primary reason for the SAC designation; whilst the presence of *Salmo salar* is cited as a qualifying feature (JNCC, 2021).

In 2013 the Environment Agency revoked the Ennerdale Water abstraction licence and abstractions are due to cease in 2022 upon completion of the Thirlmere transfer scheme. The decision followed a series of reviews through the Habitats Directive 'Review of Consents' process that determined current reservoir operations as having *"potentially significant negative impacts on interest features of the River Ehen SAC"* (United Utilities, 2018, p. 2). For this reason, understanding the impacts of removing the weir and halting environmental flows at Ennerdale Water is important.

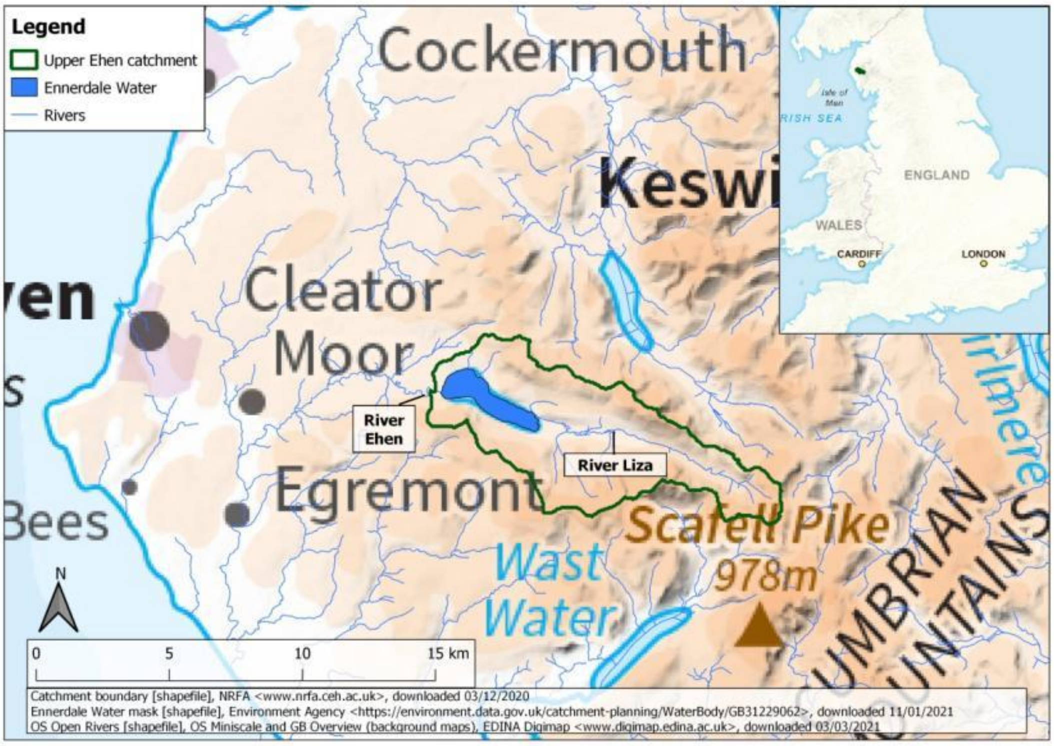

Figure 4.1.2 - Location of Upper Ehen catchment and Ennerdale Water

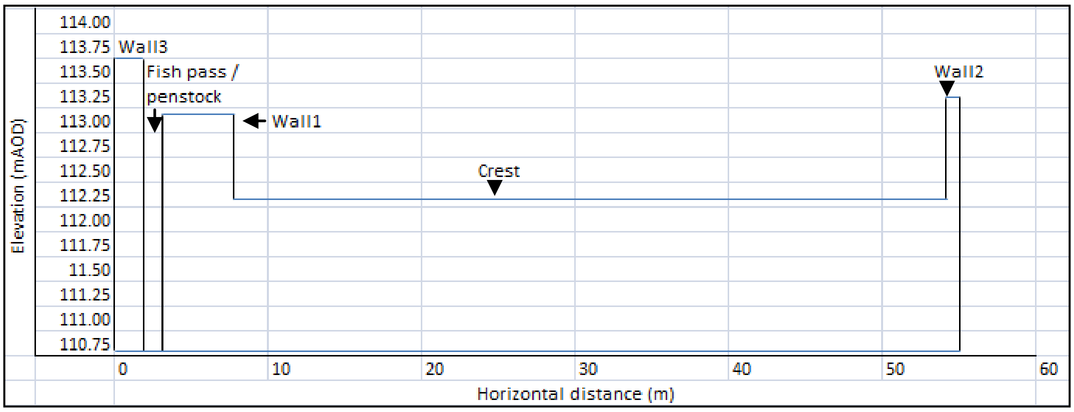

Figure 4.1.3 - Cross-section of the weir at Ennerdale Water (created by author)

4.2 Catchment model setup

In order to meet the study aims and objectives, a catchment model was built (figure 4.2). Because the study is assessing the impact of future climate and land-use change, a modelling approach was deemed most appropriate. Building the catchment model consisted of 1) setting up a standard SHETRAN model; 2) developing a 'lake' model; and 3) including reservoir operations to run the model through SHETRAN-Reservoir (Newcastle University, 2020a).

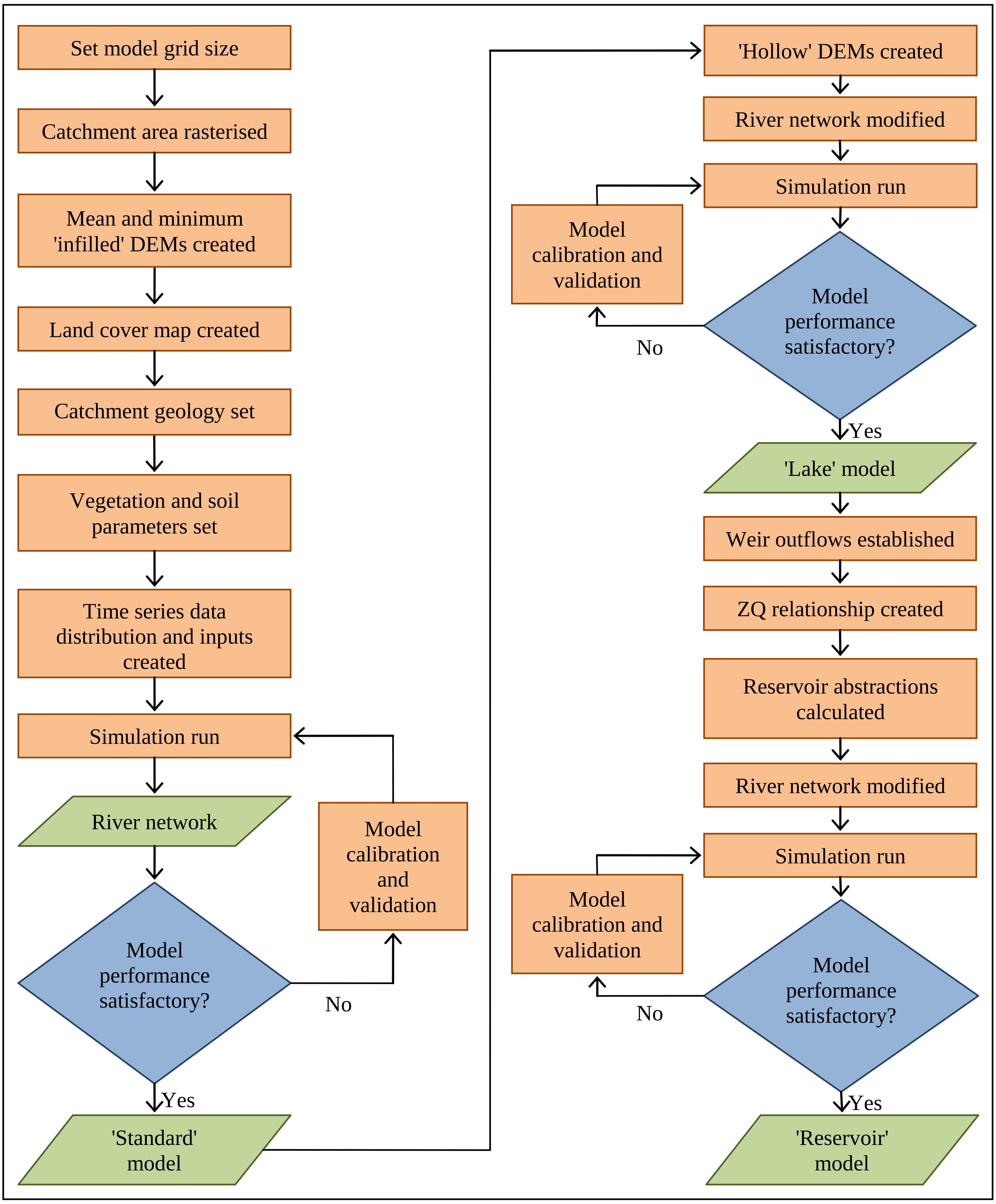

Figure 4.2 - Flow diagram of model construction

4.2.1 'Standard' model

Initially, SHETRAN-Standard (Newcastle University, 2020b) was used to develop the model. A 500m grid size was chosen as it sufficiently represented the stream network and lake area, without causing the computational effort of a smaller grid size. Increased model complexity does not necessarily mean improved performance (Orth *et al.*, 2015). The catchment mask was rasterised using GIS software so that SHETRAN could determine which grid squares were within the catchment. The rasterised catchment was adjusted to ensure that the area was accurate (figure 4.2.1.1). Spatial data inputs (figure 4.2.1.2) included: mean and minimum digital elevation models (DEMs; sinks were removed from the former; Ordnance Survey, 2021); land cover (Morton *et al.*, 2020); and gridded rainfall and potential evapotranspiration (PE; Robinson *et al.*, 2016; Tanguy *et al.*, 2019).

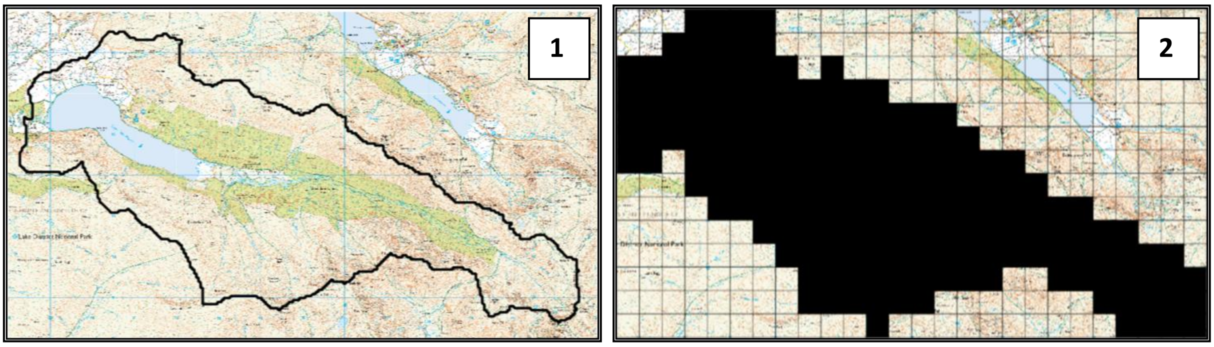

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Figure 4.2.1.1 - Creation of the model catchment area

1) catchment mask (NRFA, 2021); 2) rasterised catchment area; 3) text file of adjusted catchment area ('1' denotes grid squares inside the catchment, '0' denotes grid squares outside catchment).

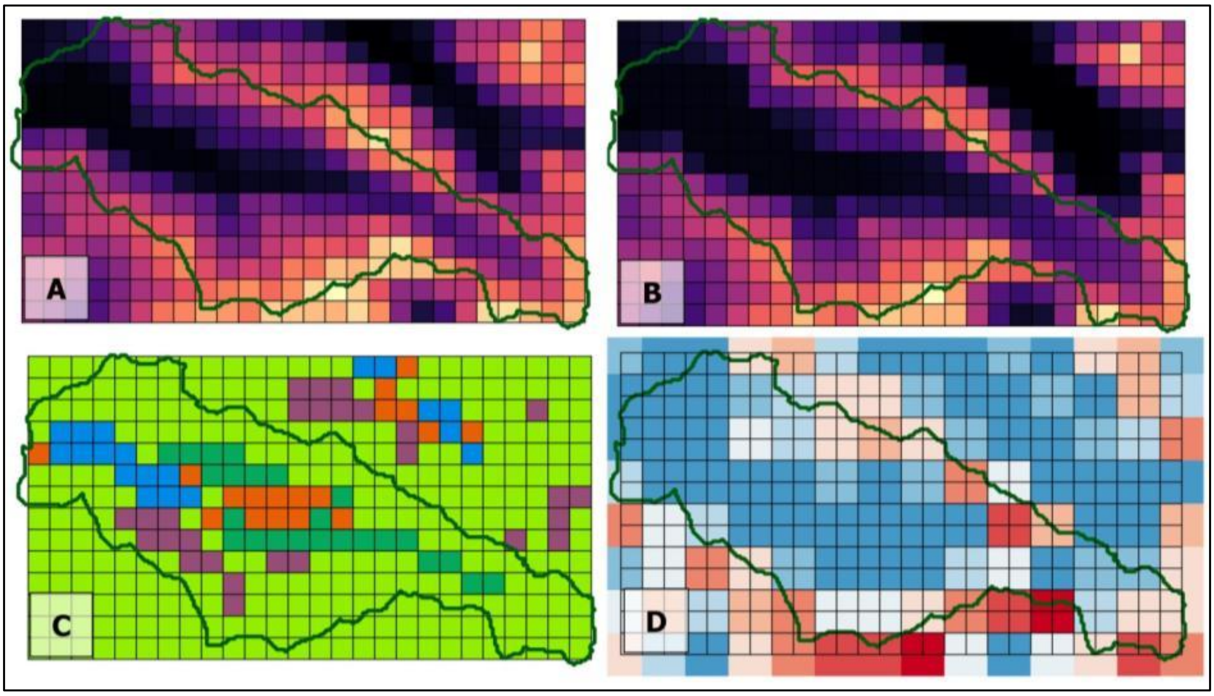


Figure 4.2.1.2 - Spatial data inputs for the 'standard' model
A) mean DEM; B) minimum DEM; C) land cover; and D) rainfall and PE distribution

For the 'standard' model, the DEMs were 'infilled', meaning that ground elevation at the lake represented the water surface elevation, as opposed to the lake bed elevation. This is because LiDAR does not penetrate water well (Newcastle University, 2020c).

Each grid square was assigned a land cover type by using the modal land cover from the 25m rasterised land parcels. It was ensured that the percentage of the model catchment under forest cover closely matched that of the actual catchment (~19%; Wild Ennerdale, 2018). The Land Cover Map (LCM) classifications were then converted to corresponding SHETRAN land cover types (figure 4.2.1.3). Parameters (table 4.2.1.1) were then assigned to each vegetation type. Parameters were adapted from Breuer *et al.* (2003), Birkinshaw (2016) and Hughes *et al.* (2021).

Table 4.2.1.2 shows the steps taken to set up the spatially-distributed rainfall and PE (figure 4.2.1.4).

LCM Classification	Corresponding SHETRAN land cover type	Number on SHETRAN vegetation map
Broadleaved wood	DeciduousForest	4
Coniferous wood	EvergreenForest	5
Improved grassland	Grass	3
Acid grassland		
Heather	Shrub	6
Freshwater	Water	2

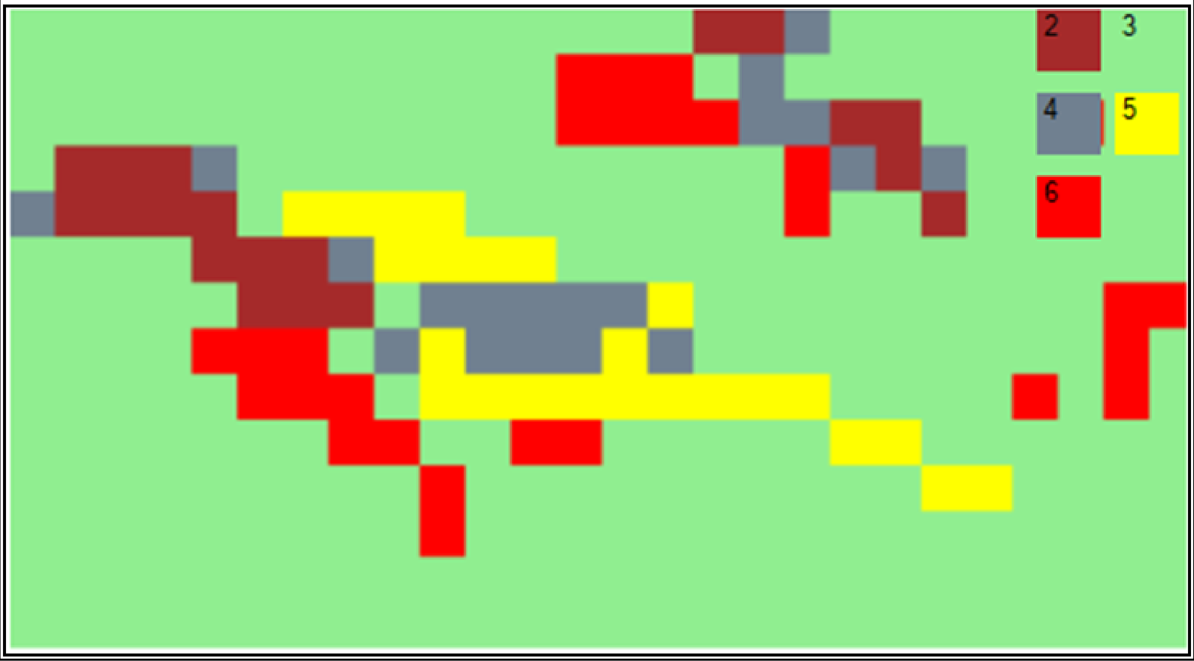


Figure 4.2.1.3 - LCM land cover types and their corresponding SHETRAN land cover type

Table 4.2.1.1 - Model vegetation parameters

Vegetation type	Canopy storage (mm)	Leaf area index	Maximum rooting depth (m)	AE/PE at field capacity	Strickler runoff coefficient
Freshwater	0	0	0.1	0.4	5
Grass	1.5	1 (min) / 3 (max)	1	0.6	1
Deciduous Forest	5	1 / 6	1.6	0.8	0.5
Evergreen Forest	5	1 / 1	2	0.8	0.5
Shrub	1.5	1 / 3	1	0.4	1

Table 4.2.1.2 - Summary of the steps taken to set up the spatially-distributed rainfall and PE

Step 1	Daily rainfall data up to 2017 (Tanguy *et al.*, 2019) and daily PE data up to 2015 (Robinson *et al.*, 2016) on a 1km grid were used.
Step 2	Each 1km grid square was assigned a number from 1 to 112 on a 8x14 grid.
Step 3	A CSV file was produced with a column for each grid number. Each row had a rainfall and PE value (mm) for each day in 2015.
Step 4	The 26x14 500m grid was applied to the 14x8 1km grid, meaning that, for each of the 112 grid squares, there were four 500m squares (with the exception of outside rows and columns).
Step 5	A text file was produced for the 26x14 500m grid containing the numbers that correspond with the 1km grid and the columns in the CSV file.

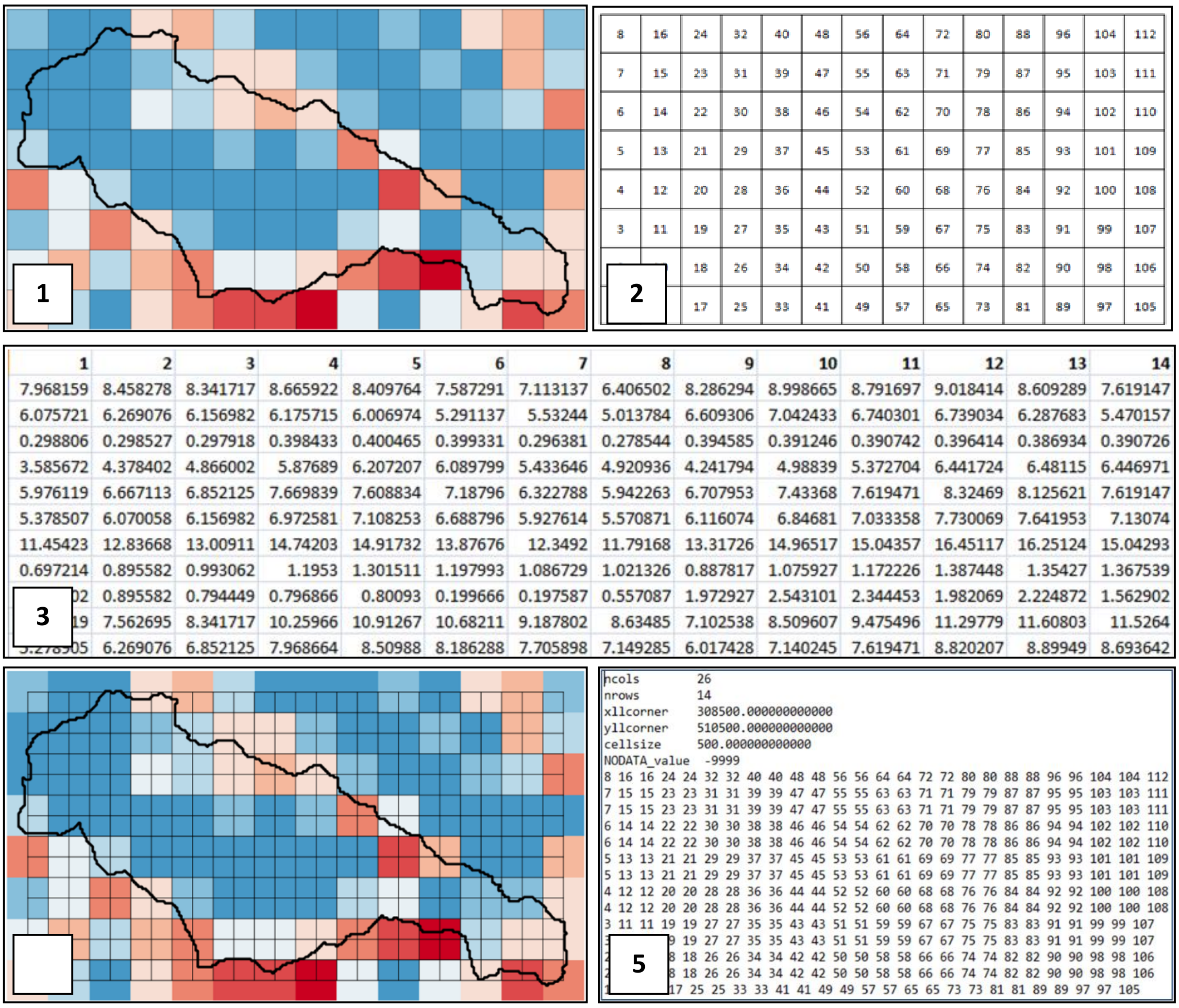


Figure 4.2.1.4 - Spatially-distributed rainfall and PE setup

1) 1km gridded data; 2) 1km grid with numbered grid squares; 3) extract of CSV file of rainfall data; 4) 1km gridded data with 500m catchment grid applied; 5) text file showing the 26x14 500m model grid containing the numbers that correspond with the 1km grid and the columns in the CSV file.

model catchment geology was made homogenous throughout because the entire catchment is underlain by completely impervious bedrock with only a small amount of superficial deposits (see table 4.1). Soil parameters (table 4.2.1.3) were taken from Hughes *et al*. (2021), with the exception of the saturated conductivity of the bedrock (denoted by *), which was reduced during model calibration to improve simulation of peak and low flows.

Table 4.2.1.3 - Model soil parameters

Soil layer	**Soil type**	**Depth at layer of base (m)**	**Saturated water content**	**Residual water content**
1	Coarse (18% clay; 65% sand)	0.4	0.403	0.025
2	Bedrock	2	0.3	0.2
Soil layer	**Soil type**	**Saturated conductivity (m/day)**	**vanGenuchten-alpha (cm-1)**	**vanGenuchten -n**
1	Coarse (18% clay; 65% sand)	30	0.0383	1.3774
2	Bedrock	0.001*	0.01	5

4.2.2 'Lake' model

The 'lake' model was adapted from the 'standard' model. 'Hollow' mean and minimum DEMs (figure 4.2.2.1) - which accounted for the bathymetric depths of Ennerdale Water - were produced to replace the 'infilled' DEMs used in the 'standard' model via the steps shown in table 4.2.2.

Table 4.2.2 - Summary of the steps taken to create the 'hollow' DEMs

Step 1	Contours were created using GIS software to represent the height of the land beneath the water's surface, based on the 1:50,000 OS map bathymetric contours, and these were incorporated with the contours for the surrounding catchment.
Step 2	The contours were interpolated into a 5m DEM.
Step 3	The 5m DEM was converted into 500m gridded DEMs - mean and minimum - and the height values for the reservoir grid squares inserted into the 'infilled' DEMs to replace the flat water surface heights. Here it was ensured that the average depth of the model lake matched that of the actual lake.

River links that ran through the lake in the 'standard' model were removed (figure 4.2.2.2) and the channel height of the remaining links were adjusted to reflect the ground elevations of the 'hollow' minimum DEM.

In essence, the 'lake' model was identical to the 'reservoir' model (section 4.2.3) except it did not include reservoir operations. It was therefore used to simulate catchment hydrology if the weir were removed and reservoir operations ceased.

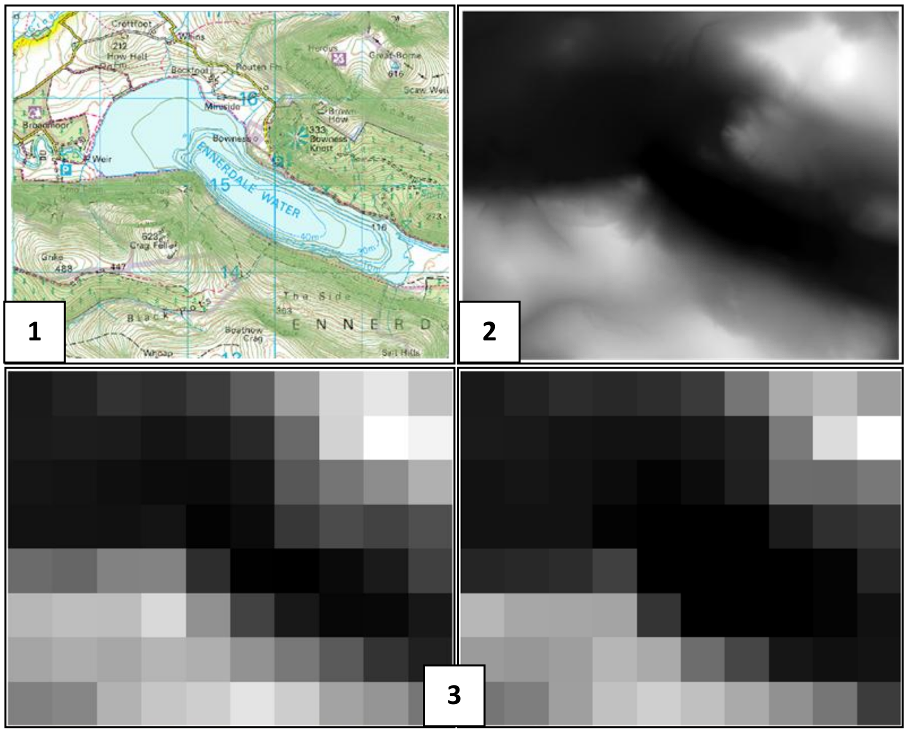


Figure 4.2.2.1 - Creation of the 'hollow' DEMs

1) Bathymetric contours used to create the DEMs; 2) 5m DEM interpolated from the contours; 3) mean and minimum 'hollow' 500m DEMs.

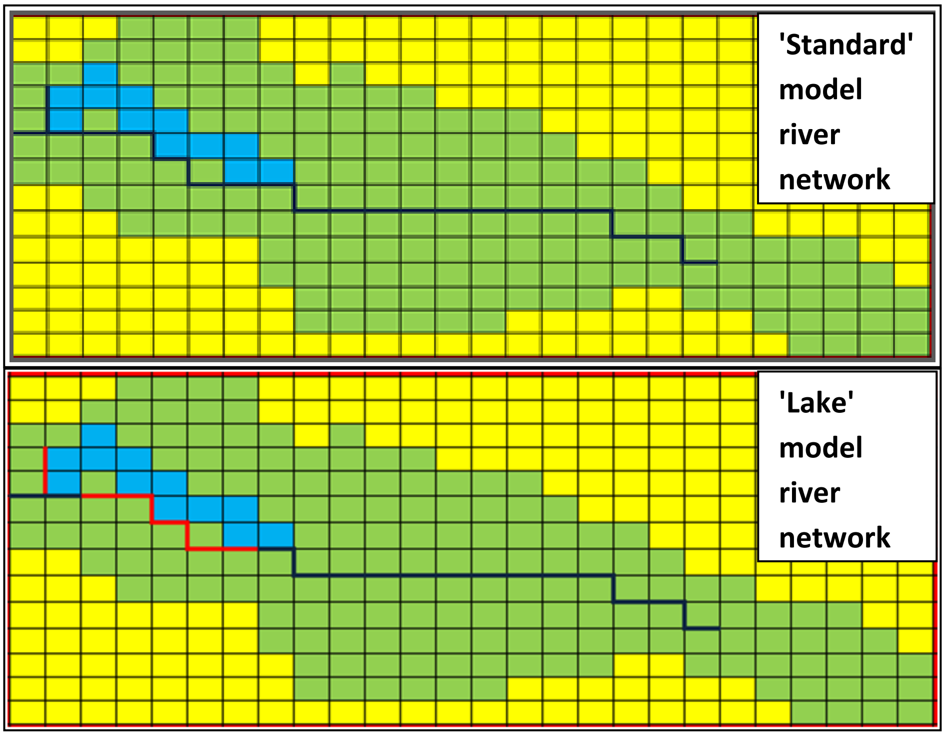


Figure 4.2.2.2 - Modifications made to the river network for the 'lake' model

N.B. Dark blue lines represent active channel links. Red lines represent channel links that were removed for the 'lake' model.

4.2.3 'Reservoir' model

The 'reservoir' model, which was adapted from the 'lake' model, included reservoir operations (abstractions and weir outflows) to create a more accurate replica of catchment hydrology. The model was run using SHETRAN-Reservoir. Table 4.2.3.1 summarises the steps taken to build the 'reservoir' model.

Table 4.2.3.1 - Summary of the steps taken to build the 'reservoir' model

Step 1	Theoretical discharge (m^3/second, 'cumecs' herein) past the weir at Ennerdale Water was calculated using an equation for each relevant weir component (table 4.2.3.2): • Manning's open channel equation was used for the fish pass (Manning, 1891). The roughness and channel slope coefficients were established from photos of the fish pass and during model calibration. • For the weir crest, an equation for critical flow over broad-crested weirs was used. This equation was used by Carver (2021) to develop a theoretical stage-discharge (ZQ) relationship for the weir at Ennerdale Water, which achieved a good fit to the observed ZQ relationship (figure 4.2.3.1).

Step 2	Using the weir equations and details regarding environmental flow releases when reservoir stage is bwcl (Ricardo Confidential, 2020), a ZQ relationship was created, to be read by SHETRAN-Reservoir via a text file (figure 4.2.3.2).
Step 3	A text file detailing daily abstractions was also created (figure 4.2.3.3). A weighted daily mean was used to reflect reduced abstractions for ~25% of the year, during which water is abstracted from local boreholes instead (Ricardo Confidential, 2020).
Step 4	A more complex river network was generated and modified (figure 4.2.3.4). This was achieved by changing the flow accumulation value from 20 (used in the 'standard' and 'lake' models) to 5. This meant that a river link was generated during simulation each time flow accumulation exceeded 5 grid squares. This resulted in improved accuracy during model calibration and validation compared with the simplified river network used for the 'lake' model.

Table 4.2.3.2 - Weir equations used in the 'reservoir' model

Weir component	**Minimum elevation (mAOD)**	**Equation**	**Reference**
Fish pass	110.79	Manning's open channel equation: $\left(\frac{1}{n}\right)$*A*$R^{1.5}$*$S^{0.5}$ Where: n = roughness coefficient (0.015) A = flow area R = hydraulic radius S = channel slope (0.006)	Manning (1891)
Crest	112.31	Critical flow over broad crested weir: 1.6*b*$H^{1.5}$ Where: b = width H = hydraulic head	Carver (2021)

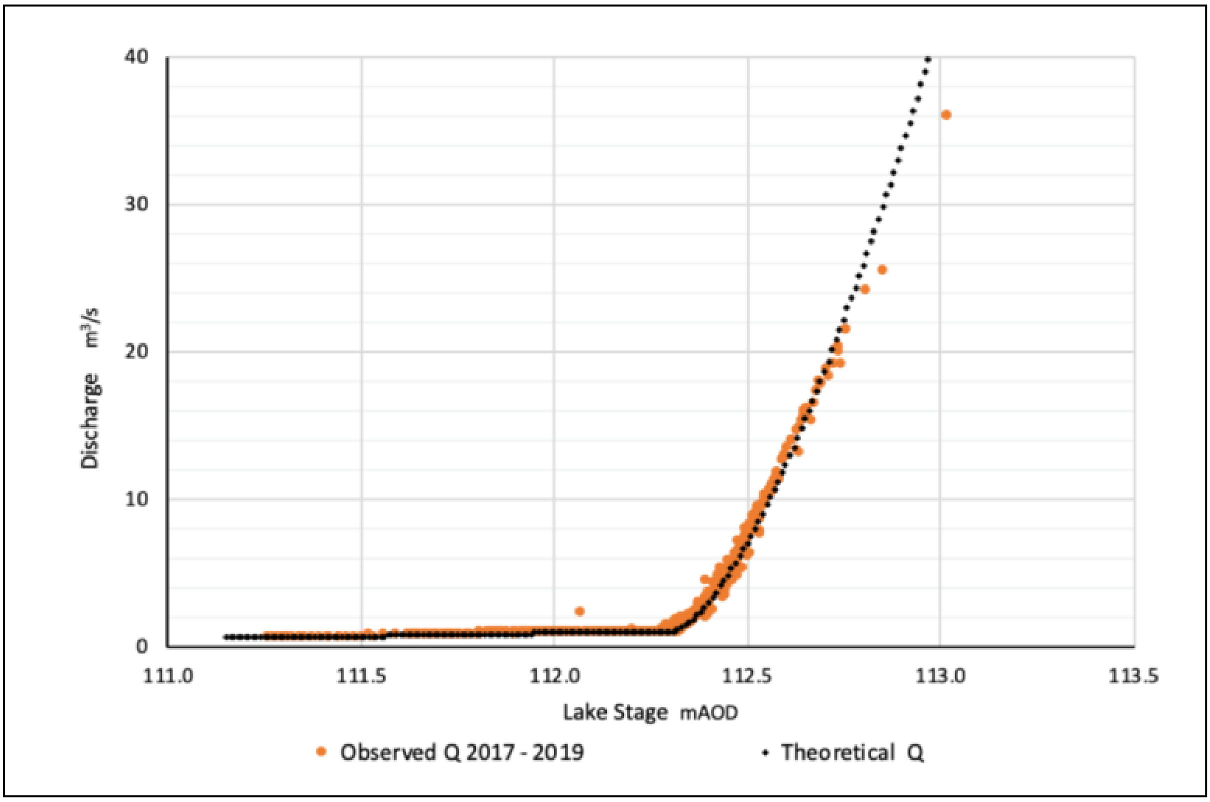
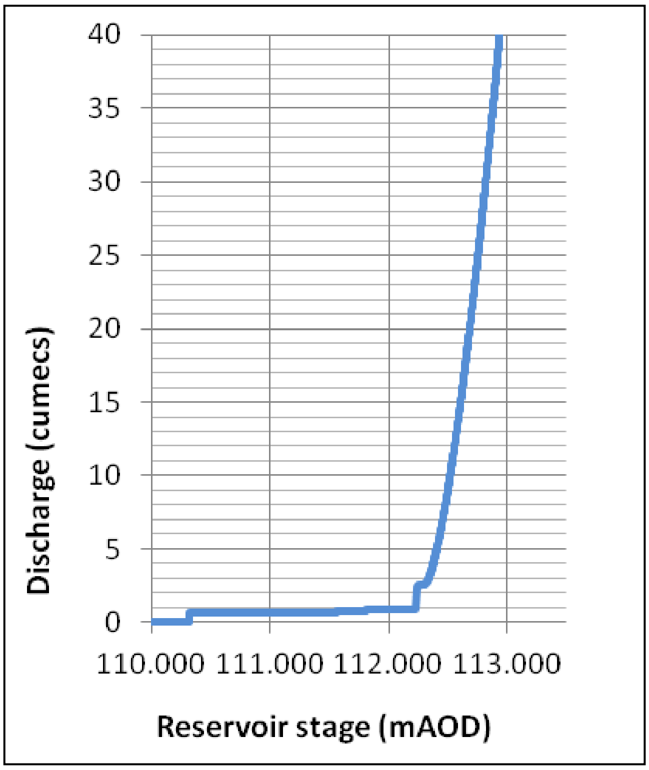


Figure 4.2.3.1 - Observed stage-discharge relationship at Ennerdale Water compared to a theoretical stage-discharge relationship based on weir component dimensions and environmental flow rules (Carver, 2021)




```
: NUMBER OF ZQ TABLES NEEDED
1
: ZQ TABLE REFERENCE NUMBER
1
: ZQ TABLE LINK NUMBER
19
: ZQ TABLE FACE NUMBER
3
: ZQ TABLE OPERATION HOUR
12
: ZQ TABLE NUMBER OF ROWS
2931
 ZQ TABLE1 Ennerdale
Z ZQ>110.00
110.000 0
110.001 0
110.002 0
110.003 0
110.004 0
110.005 0
110.006 0
110.007 0
110.008 0
110.009 0
```

Figure 4.2.3.2 - a) Theoretical stage-discharge relationship developed for the 'reservoir' model; b) extract of the text file for the ZQ relationship read by SHETRAN-Reservoir

```
Well abstraction for each category (m3/s) constant since
the previous breakpoint. Y,M,D,H,min,Qn.
2015    1    1     0    0    0.240
2015    1    2     0    0    0.240
2015    1    3     0    0    0.240
2015    1    4     0    0    0.240
2015    1    5     0    0    0.240
2015    1    6     0    0    0.240
2015    1    7     0    0    0.240
2015    1    8     0    0    0.240
2015    1    9     0    0    0.240
2015    1    10    0    0    0.240
2015    1    11    0    0    0.240
2015    1    12    0    0    0.240
2015    1    13    0    0    0.240
2015    1    14    0    0    0.240
2015    1    15    0    0    0.240
2015    1    16    0    0    0.240
2015    1    17    0    0    0.240
2015    1    18    0    0    0.240
2015    1    19    0    0    0.240
2015    1    20    0    0    0.240
2015    1    21    0    0    0.240
2015    1    22    0    0    0.240
2015    1    23    0    0    0.240
2015    1    24    0    0    0.240
```

Figure 4.2.3.3 - Extract of the text file detailing daily abstractions to be read by SHETRAN-Reservoir

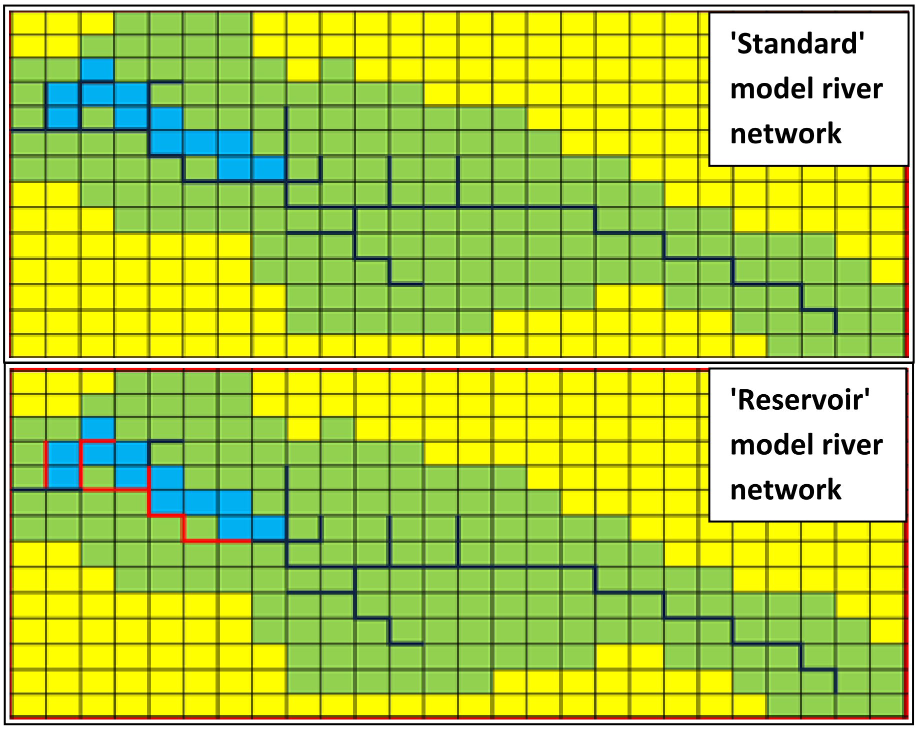


Figure 4.2.3.4 - Modifications made to the 'reservoir' model river network

N.B. Dark blue lines represent active channel links. Red lines represent channel links that were removed for the lake model.

4.3 Scenario development

20 model scenarios (table 4.3.1) were developed and run through the 'lake' and 'reservoir' models. This allowed for the simulation and comparison of mean daily discharge in the River Ehen at Bleach Green gauging station ('discharge' herein) over the course of a year under a range of emissions scenarios; with and without the weir and abstractions; and with current and future forest cover.

The future land cover input layers (figure 4.3) were adapted from the Wild Ennerdale Stewardship Plan (Wild Ennerdale, 2018) and GIS information provided by Forestry England (G Browning 2020; 2021, personal communication, 12 November; 26 January). Future forest cover was increased to ~40%. Of this, conifers comprised ~25% and broadleaves accounted for ~75. Without the weir, lake area was reduced by 25% (inferred from the GIS information) and lake cells replaced by other land cover types were returned to their 'infilled' DEM elevations. Replacement land cover types were inferred from the Stewardship Plan.

Table 4.3.1 - Overview of model scenarios

Weir Scenario	Forest Scenario	Emissions Scenario
With weir and abstractions ('reservoir' model)	Current forest cover	1990 baseline
		RCP2.6
		RCP4.5
		RCP6.0
		RCP8.5
	Future forest cover	1990 baseline
		RCP2.6
		RCP4.5
		RCP6.0
		RCP8.5
Without weir and abstractions ('lake' model)	Current forest cover	1990 baseline
		RCP2.6
		RCP4.5
		RCP6.0
		RCP8.5
	Future forest cover	1990 baseline
		RCP2.6
		RCP4.5
		RCP6.0
		RCP8.5

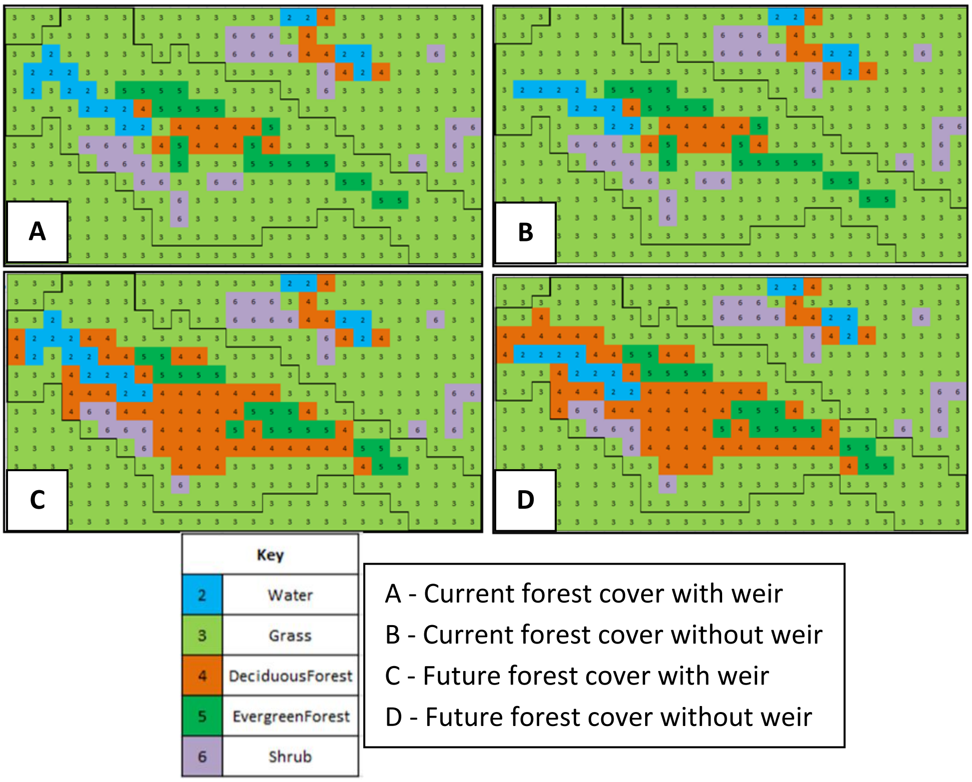


Figure 4.3 - Model scenario land cover input layers

The future climate scenarios (table 4.3.2) were developed by altering rainfall and PE model inputs. Future rainfall amounts were derived from applying precipitation rate anomalies for the catchment from the UKCP18 emissions scenarios (Met Office, 2021) to 1990 rainfall data (Tanguy *et al.*, 2019). All rainfall percentages refer to the 50th percentile for probabilistic (25km) projections for the catchment. The 50th percentile represents predicted changes most likely to occur (Met Office, 2018a) and therefore reduced model uncertainty. As per UKCP18, the rainfall seasons corresponded with the following months:

- Winter - December, January and February
- Spring - March, April and May
- Summer - June, July and August
- Autumn - September, October and November

PE scenarios were based on Prudhomme and Williamson's (2013) forecasted changes to PE derived from the FAO56 method (see section 3.1.1). In their study, changes are given as a range for each month. To reflect this uncertainty, the lower limit of each range was made to correspond with the UKCP18 lower emissions scenarios (RCP2.6 and 4.5), whilst the upper limit corresponded with the higher emissions scenarios (RCP6.0 and 8.5). Percentage changes were applied to each month of the 1990 PE data (Robinson *et al.*, 2016).

1990 was selected as the baseline year because it falls within the UKCP18 and Prudhomme and Williamson's (2013) study baselines (1981-2000 and 1961-1990 respectively). 2070 was selected as the future scenario year because future forest cover is expected to be established by then, forecasted changes to PE apply to 2041-2070, and 2070 was selected as the time slice for UKCP18 precipitation rate anomalies.

Table 4.3.2 - Model climate change scenarios

<table>
<tr><th>Emissions scenario</th><th>Rainfall Percentage Change</th><th>PE Percentage Change</th></tr>
<tr><td>RCP2.6</td><td>Winter: +10.5%
Spring: -2.8%
Summer: -11.8%
Autumn: +11.1%</td><td rowspan="2">December: +10%
January: 0%
February: +10%
March: +10%
April: +10%
May: +10%
June: 0%
July: +10%
August: +20%
September: +10%
October: 0%
November: +10%</td></tr>
<tr><td>RCP4.5</td><td>Winter: +12.1%
Spring: -3%
Summer: -13.5%
Autumn: +13.7%</td></tr>
<tr><td>RCP6.0</td><td>Winter: +12.2%
Spring: -2.8%
Summer: -13.8%
Autumn: +13.9%</td><td rowspan="2">December: +30%
January: +10%
February: +20%
March: +30%
April: +20%
May: +20%
June: +10%
July: +20%
August: +30%
September: +20%
October: +10%
November: +20%</td></tr>
<tr><td>RCP8.5</td><td>Winter: +17.6%
Spring: -1.3%
Summer: -18.3%
Autumn: +15.4%</td></tr>
</table>

4.4 Statistical analyses

The 'lake' and 'reservoir' models were run using 2015 rainfall and PE inputs and validated against discharge measured at Bleach Green gauging station in 2015 (NRFA, 2021). This year was chosen because it was the most recent year for which all time series data was available. Model performance was measured using Nash Sutcliffe efficiency (NSE), Pearson's coefficient of determination (R^2) and percent bias (PBIAS).

NSE (equation 1) is a normalised statistic that establishes the relative size of variance between simulated and observed data (Nash and Sutcliffe, 1970). 1 is the optimal value (Moriasi *et al.*, 2007).

$$NSE = 1 - \left[\frac{\sum_{i=1}^{n}(y_i^{obs} - y_i^{sim})^2}{\sum_{i=1}^{n}(y_i^{obs} - y^{mean})^2}\right] \tag{1}$$

R^2 (equation 2) describes collinearity between simulated and observed data by quantifying the proportion of variance. It ranges from 0 to 1, with values closer to the latter signifying less error variance (Moriasi *et al.*, 2007).

$$R^2 = 1 - \left[\frac{\sum(y_i - \hat{y}_i)^2}{\sum(y_i - \bar{y})^2}\right] \tag{2}$$

PBIAS (equation 3) measures the average tendency for simulated data to be less than or greater than measured data. The optimal value is 0, whilst positive values indicate overestimation bias and negative values indicate underestimation bias (Gupta *et al.*, 1999).

$$PBIAS = 100 * \left[\frac{\sum_{i=1}^{n}(y_1^{sim} - y_1^{obs})}{\sum_{i=1}^{n}(y_1^{obs})}\right] \tag{3}$$

For annual catchment-scale models, performance can be adjudged to be satisfactory if NSE>0.5, R^2>0.6 and PBIAS<15% (Moriasi *et al.*, 2015). See section 5.1 for model performance results.

To assess the impacts of climate and land-use change on the discharge of the River Ehen at low flows, flow-duration curves were constructed. Flow-duration curves provide the duration of occurrence across the whole range of river flows and can therefore provide a measure of flow deficiency (Shaw *et al.*, 2011). Subsequently, the flow that is exceeded 95% of the time (Q95; Shaw *et al.*, 2011) was established for each scenario to quantify their impact on low flows.

In order to test the statistical significance of any variation in discharge, Kruskal-Wallis tests (Kruskal and Wallis, 1952) were carried out (table 4.4). The Kruskal-Wallis test is the non-parametric equivalent of a one-way ANOVA and can be used to compare two or more datasets (Ostertagová *et al.*, 2014). It assumes that populations are not normally-distributed, are independent and are of the same distribution (Ostertagová *et al.*, 2014), assumptions which applied to the simulated discharge data in this study (figure 4.4). For a Kruskal-Wallis test, the null hypothesis (H_0) is that the population medians are equal, the alternative being that there is a difference between at least two of them (Bewick *et al.*, 2004). It was therefore used to test the significance (for α=0.05) of any variation in discharge caused by the five emissions scenarios, the weir removal scenario *or* the future forest cover scenario. In the case of the latter two, only the baseline emissions scenario was used to ensure the impact of emissions scenarios did not skew results. The tests were used to check the statistical significance of any variation between low flows, seasonal flows, as well as across the range of river flows.

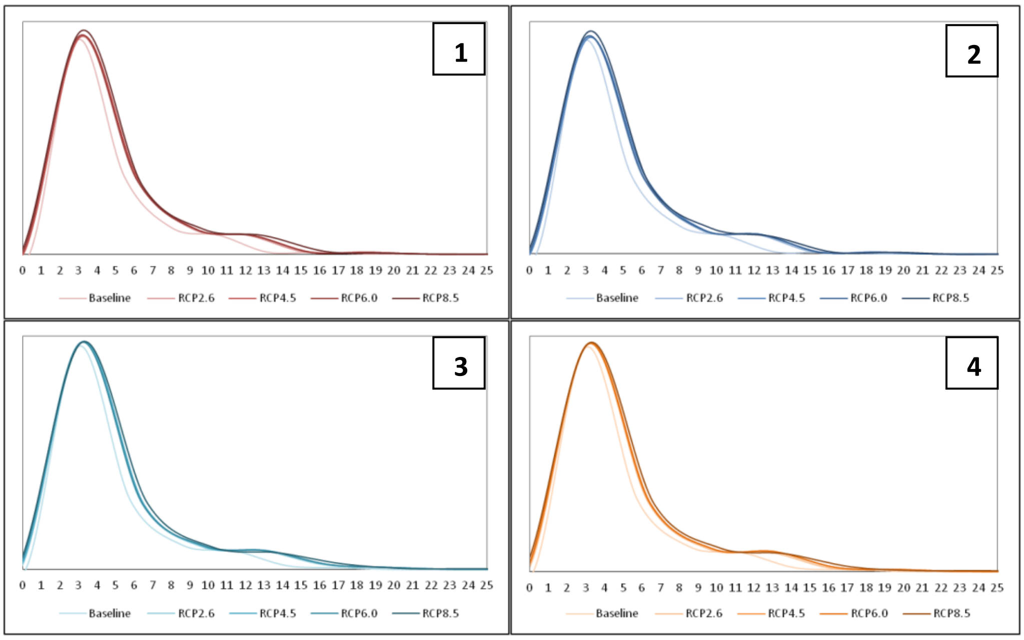


Figure 4.4 - Distribution of simulated discharge data

1) With weir, current forest cover; 2) with weir, future forest cover; 3) without weir, current forest cover; 4) without weir, future forest cover.

Table 4.4 - Kruskal-Wallis tests carried out during results analysis

Test description	Null hypothesis (H_0)
Compared seasonal discharge (annual, winter, spring, summer and autumn) across the five emissions scenarios (with the weir and current forest cover).	*There will be no difference in seasonal flows across the emissions scenarios.*
1) Compared the lowest 50% of discharge values across the five emissions scenarios (with the weir and current forest cover). 2) Compared the lowest 5% of discharge values across the five emissions scenarios (with the weir and current forest cover).	*There will be no difference in low flows across the emissions scenarios.*
Compared seasonal discharge with and without the weir (current forest cover).	*There will be no difference in seasonal flows with and without the weir.*
1) Compared the lowest 50% of discharge values with and without the weir (current forest cover). 2) Compared the lowest 5% of discharge values with and without the weir (current forest cover).	*There will be no difference in low flows with and without the weir.*
1) Compared seasonal discharge with current and future forest cover (with the weir). 2) Compared seasonal discharge with current and future forest cover (without the weir).	*There will be no difference in seasonal flows with current and future forest cover.*
1) Compared the lowest 50% of discharge values with current and future forest cover (with the weir). 2) Compared the lowest 50% of discharge values with current and future forest cover (without the weir). 3) Compared the lowest 5% of discharge values with current and future forest cover (with the weir). 4) Compared the lowest 5% of discharge values with current and future forest cover (without the weir).	*There will be no difference in low flows with current and future forest cover.*

4.5 Constraints

SHETRAN is a physically-based spatially-distributed (PBSD) model (Ewen *et al.*, 2000; Birkinshaw *et al.*, 2010), which represents component hydrological processes (Pechlivanidis *et al.*, 2011). Factors that contribute to uncertainty around the accuracy of results from PBSD models include:

- The physics of PBSD models tend to be based on laboratory or field experiments and are therefore affected by the nature of the experiments themselves (Pechlivanidis *et al.*, 2011).
- There is the assumption that such physics can be applied to all catchments regardless of scale and characteristics (Beven, 2004).
- Catchments tend to have a high level of spatial heterogeneity, which can be costly (and time-consuming) to replicate, particularly with regard to subsurface processes (Pechlivanidis *et al.*, 2011).
- In theory PBSD model parameters are quantifiable but in practice it is impossible at the scale of model application (Wheater, 2002). Therefore, parameters are averaged at the grid or element scale, which are larger than the scale of variation in reality (Pechlivanidis *et al.*, 2011).

Table 4.5 summarises study-specific constraints. Uncertainty is an inherent part of hydrological modelling and must be taken into account. Model calibration and validation were used to minimise this uncertainty, so too was the implementation of a number of future scenarios that represent a range of prospective catchment conditions. The results of the study should not be thought of as prescriptive, but rather as information to aid decision-making. Indeed, any model biases were assumed to be systematic and it is advised that conclusions are drawn from relative rather than absolute changes to river flows. To this end, the study remains of use to researchers, land managers and policy makers.

Table 4.5 - Summary of study constraints

Model area	Constraints
Model physics	• SHETRAN's physics will have imperfectly represented the actual physics of the Upper Ehen catchment.
Spatial inputs	• 500m grid size means that all spatial inputs were averaged/simplified. • Use of 'stock' parameters - rather than parameters collected in the field - likely increased model inaccuracy.
Time series inputs	• CHESS PE dataset is calculated using the Penman-Monteith equation, which creates uncertainty (Tanguy *et al.*, 2018). Inaccuracies in calculated PE can be negated during model calibration (Peng *et al.*, 2016; Seiller and Anctil, 2016). • CEH-GEAR interpolated rainfall data (Tanguy *et al.*, 2019) is less accurate in mountainous areas (the north and west UK) due to orographic enhancement of frontal or pre-frontal rainfall, which varies with altitude (Keller *et al.*, 2015). Hydrological models have been found to be more sensitive to errors in rainfall than PE, particularly in temperate climates (Paturel *et al.*, 1995; Bastola *et al.*, 2011; Guo *et al.*, 2017). • When river stage at Bleach Green gauging station is greater than 1.5m (or discharge is greater than 34cumecs), readings are extrapolated due to flow bypass (NRFA, 2021). This will have affected model calibration, validation and performance. In 2015 discharge exceeded 34cumecs in November and December (figure 4.5).
Reservoir operations	• A perfect fit between theoretical discharge (calculated using weir equations) and measured discharge is impossible. • The weighted average used for abstractions will have caused discrepancies in simulated and observed reservoir stage and downstream discharge. • The current environmental flow regime was introduced post-2015. Model calibration and validation using time series data from 2015 will have therefore resulted in inaccuracies. • Environmental flows are informed by discharge measured at Bleach Green and not reservoir stage (Ricardo Confidential, 2020). Therefore, the ZQ relationship was unable to perfectly match environmental flows and a QQ relationship would have been more appropriate.

Future scenarios	• UKCP18 climate projections (and therefore rainfall scenarios) are based on future emissions assumptions; the estimated ranges for future climate are based on a series of assumptions; and the predictions do not capture all possible outcomes (Met Office, 2018b). • Similarly, PE estimations, including FAO65, often involve simplifying assumptions (Westerhoff, 2015). • The extent and structure of future forest cover, whilst planned, is uncertain, as is lake area if the weir were removed. Assumptions were therefore made in designing the future land cover input layers.

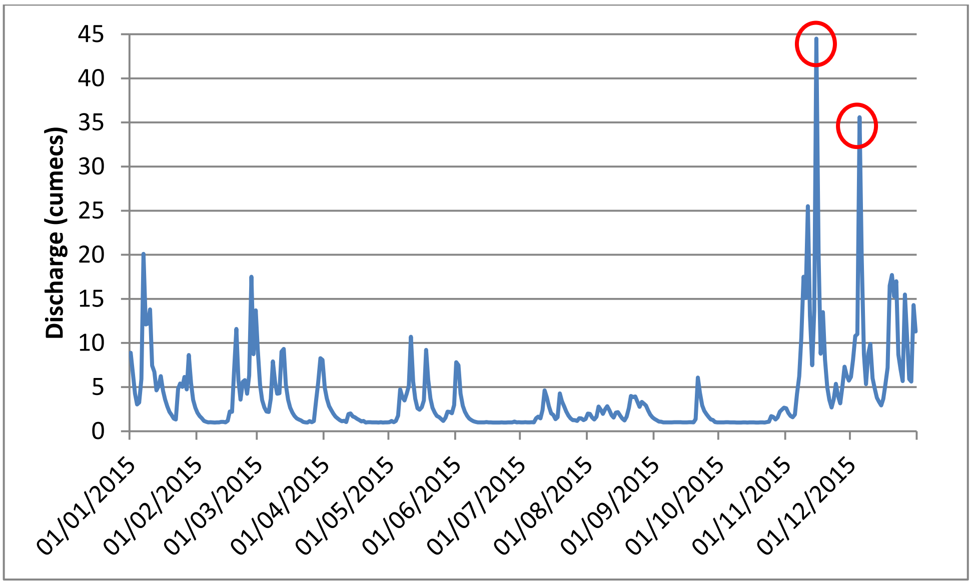


Figure 4.5 - Mean daily discharge at Bleach Green gauging station in 2015

N.B. Extrapolated discharge readings (Q>34cumecs) are circled in red.

5. Results

This chapter presents results related to: 1) model performance and the impact of the 2) emissions scenarios, 3) weir removal scenario and 4) future forest cover scenario on seasonal and low flows. Appendix A contains data tables for each of the graphs presented from sections 5.2 to 5.4. Appendix B contains annual hydrographs showing mean daily discharge for each scenario.

5.1 Model performance and uncertainty analysis

Both models are above the thresholds for each measure for catchment-scale models (outlined in section 4.1) and therefore simulated discharge with satisfactory accuracy (table 5.1). The 'reservoir' model simulated discharge with more accuracy, though with a slightly higher PBIAS deficit.

Table 5.1 - Results of model performance measurements

Model	NSE	R^2	PBIAS
Lake	0.81	0.73	-5.31%
Reservoir	0.86	0.90	-6.53%

There was a tendency for the overestimation of low flows and underestimation of peak flows. Therefore, absolute discharge values are unlikely to be 100% accurate. As stated in section 4.5, any model biases are likely to have been systematic. Relative change in discharge between the scenarios is therefore considered.

5.2 The impact of the emissions scenarios on discharge

5.2.1 Seasonal flows

The emissions scenarios result in an increase in mean annual discharge from the baseline (figures 5.2.1.1 and 5.2.1.2). Percentage increase in the annual mean daily discharge is greater with higher emissions scenarios, with the exception of RCP6.0, which is lower than RCP4.5. Across all emissions scenarios, mean daily winter discharge is above the annual mean and the season with the highest mean daily discharge. The four future emissions scenarios increase winter discharge. Like mean annual discharge, winter discharge percentage increase is greater as the emissions scenarios become more extreme, with the exception of RCP6.0. Mean daily autumn discharge is also above the annual mean across the emissions scenarios. Compared with the baseline, mean daily discharge in autumn increases. The greatest increase occurs for RCP4.5. Otherwise, percentage increase is greater with higher emissions scenarios. For all emissions scenarios, mean daily discharge is lowest during summer and below the annual mean. The greatest percentage change to discharge is experienced during summer and, without exception, summer discharge decreases as the emissions scenarios increase when compared to the baseline. Finally, mean daily spring flow is less than the annual mean across all emissions scenarios. RCP6.0 results in the biggest decrease in discharge compared with the baseline but, in general, mean spring discharge decreases with higher emissions scenarios.

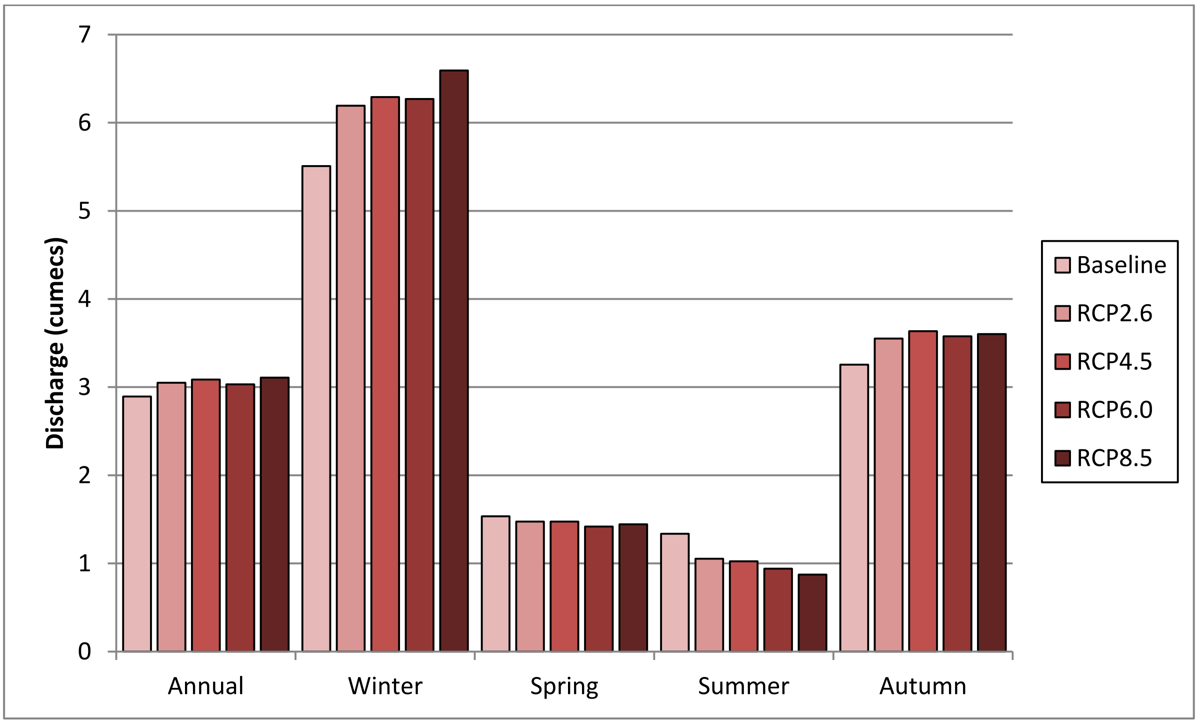


Figure 5.2.1.1 - Mean daily discharge by season for the emissions scenarios

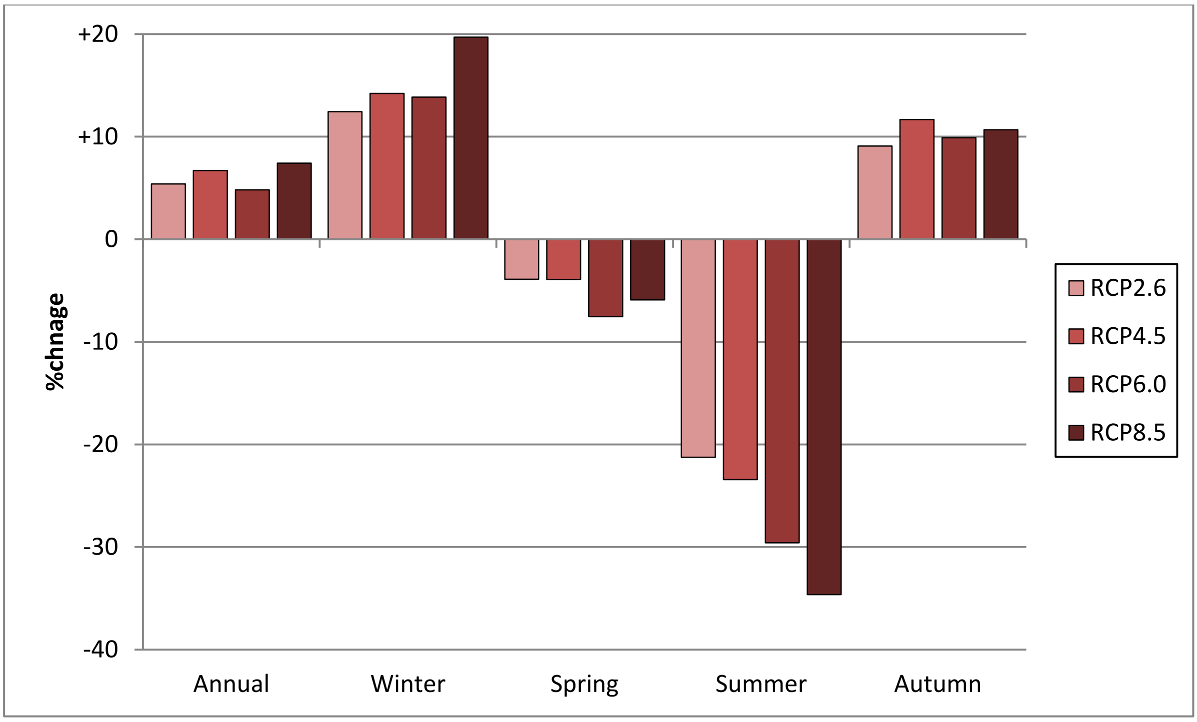


Figure 5.2.1.2 - Percentage change to mean daily discharge by season for the emissions scenarios, compared to baseline emissions

Kruskal-Wallis test results (table 5.2.1) for each season show that, because P>α, there is no significant difference for annual discharge, as well as for discharge in winter, spring and autumn across the emissions scenarios. H_0 (there is no difference between groups) is therefore accepted. However, there is significant difference in discharge across the emissions scenarios in summer (P≤α) and H_0 is therefore rejected.

Table 5.2.1 - Kruskal-Wallis test results comparing seasonal discharge across the emissions scenarios

	Annual	**Winter**	**Spring**	**Summer**	**Autumn**
α	0.05	0.05	0.05	0.05	0.05
Critical value	9.49	9.49	9.49	9.49	9.49
H value	3.21	4.88	3.11	29.03	0.37
P	0.524	0.3	0.539	0.00	0.985
Confidence level 100-(P*100)	47.6%	70%	46.1%	>99.9%	1.5%
H_0	Accepted	Accepted	Accepted	Rejected	Accepted

5.2.2 Low flows

As the emissions scenarios become more extreme, discharge becomes flashier (i.e. the magnitude of low and peak flows increases; figures 5.2.2.1 and 5.2.2.2). The range of discharge values increases with higher emissions scenarios. Similarly, the interquartile range (IQR) increases. Each of the median, lower quartile (Q_{1}) and minimum values decrease as the emissions scenarios increase.

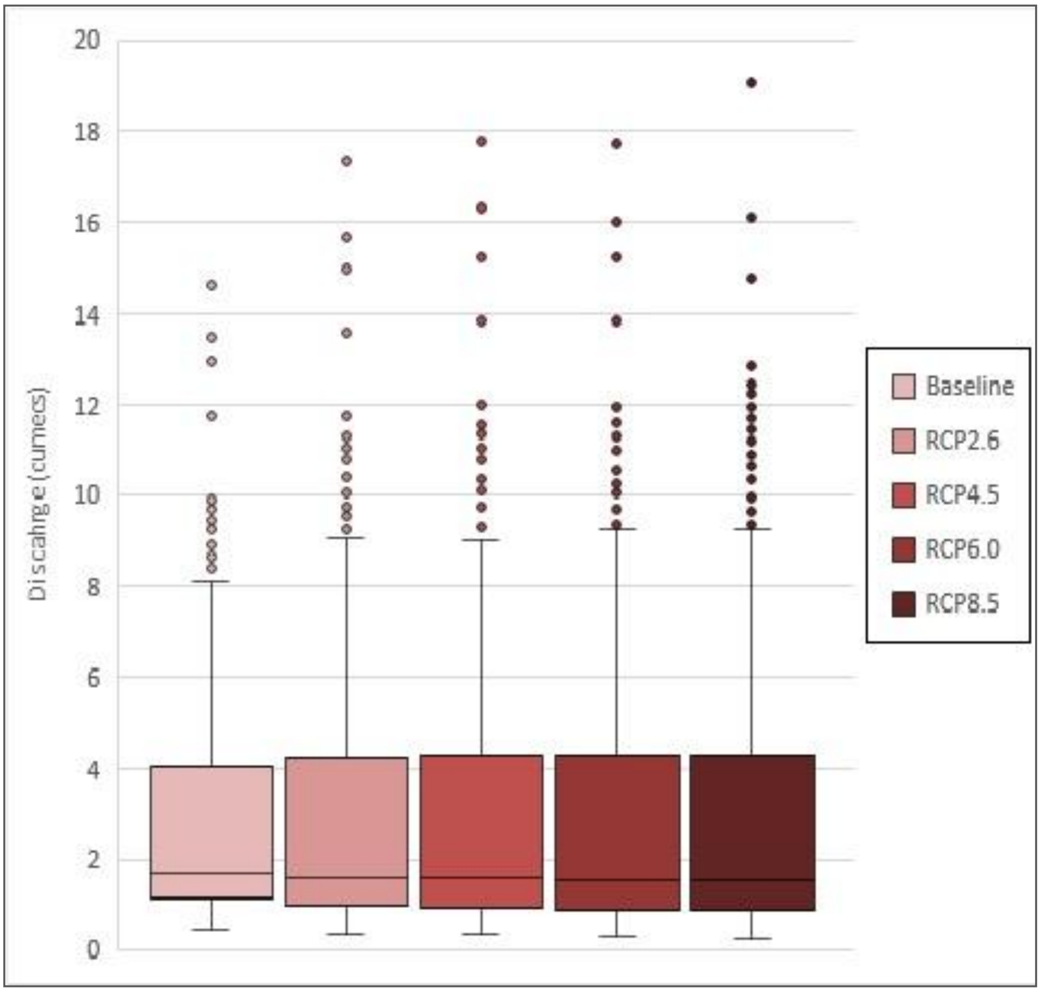


Figure 5.2.2.1 - Box plots of mean daily discharge for the different emissions scenarios (current forest cover, with weir, excluding outliers)

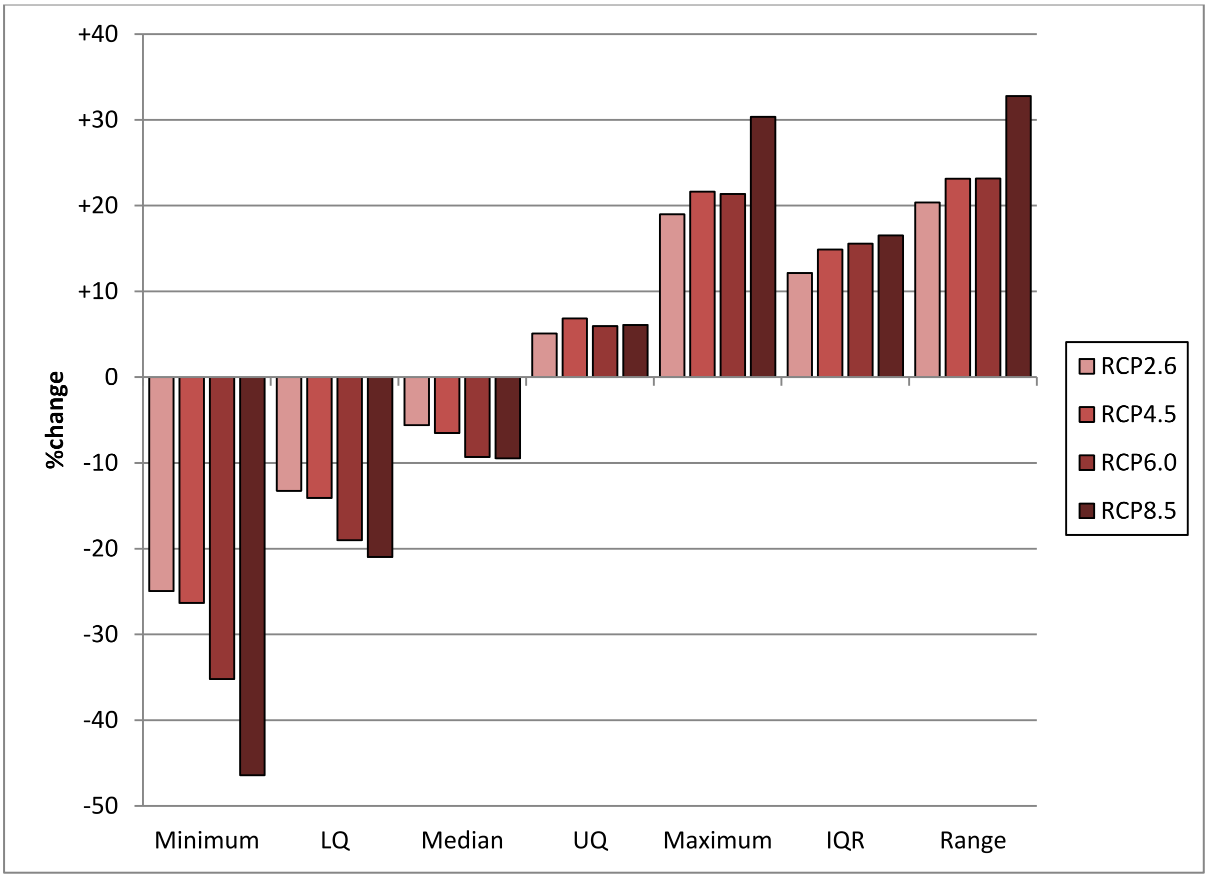


Figure 5.2.2.2 - Percentage change to measures of dispersion and the median mean daily flow for the emissions scenarios compared to the baseline (current forest cover, with weir)

The flow-duration curve for the emissions scenarios (figure 5.2.2.3) paints a similar picture. As the emissions scenarios increase, the magnitude of both low and high flows grows (signified by steeper curves), and the flows occurring at 50% exceedance (Q50 or median) and 75% exceedance (Q75 or Q_1) decrease with higher emissions. The emissions scenarios results in a greater proportion of discharge decreasing (~60%) than increasing. Relatively, the decrease in discharge becomes more pronounced with lower flows, illustrated by greater divergence between the lines towards the 100% exceedance value.

Q95 decreases as a result of the emissions scenarios (figures 5.2.2.4 and 5.2.2.5). As the emissions scenarios increase, the decline in Q95 becomes greater.

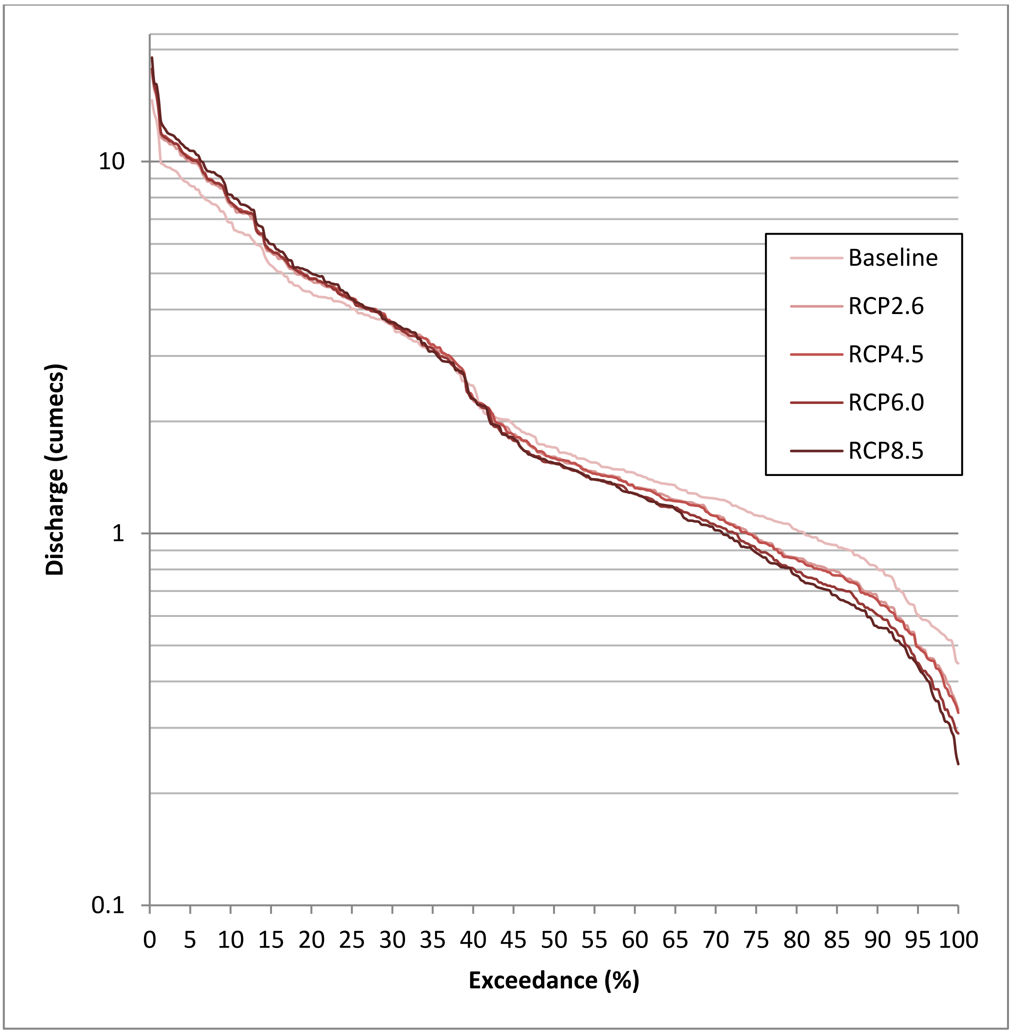


Figure 5.2.2.3 - Flow-duration curve for different emissions scenarios (current forest cover, with weir)

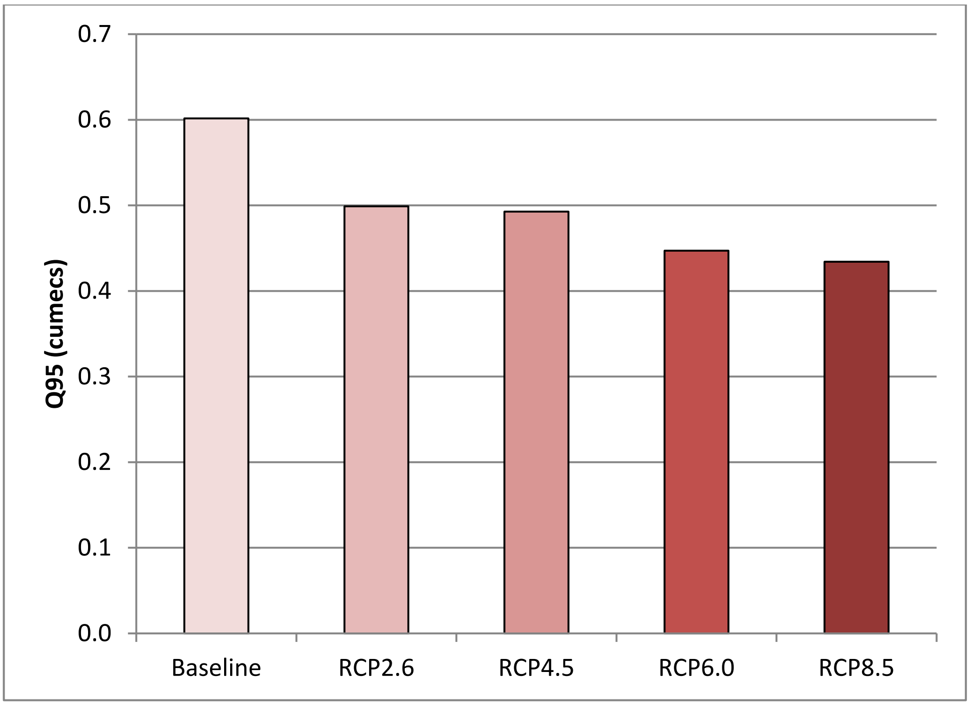


Figure 5.2.2.4 - Q95 for the different emissions scenarios (current forest cover, with weir)

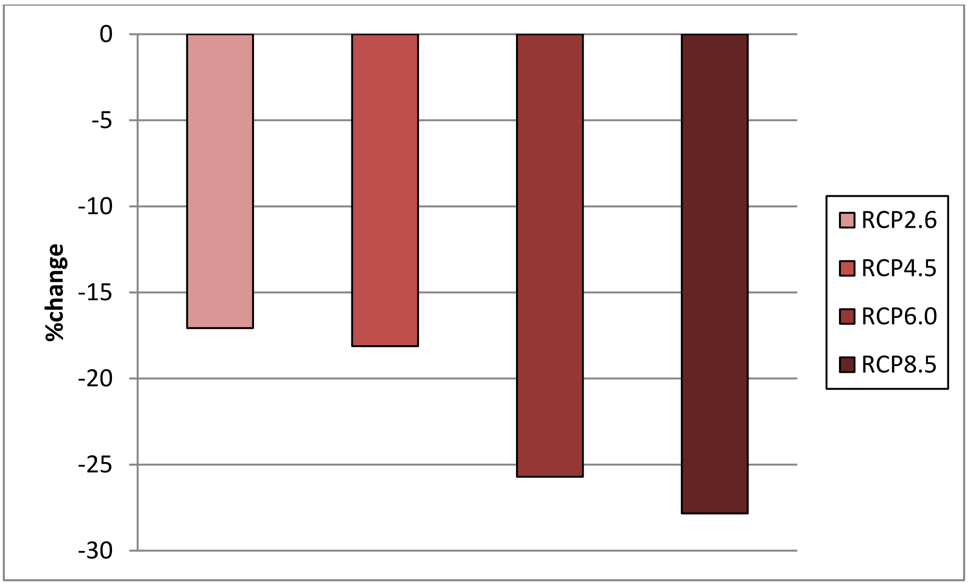


Figure 5.2.2.5 - Percentage change to Q95 for each emissions scenario compared to the baseline (current forest cover, with weir)

Kruskal-Wallis test results (table 5.2.2) indicate that there is statistically-significant variation in both Q_1 (Q>Q50 herein) and for the lowest 5% of flows (Q≥Q95 herein). In both cases H_0 is therefore rejected. This reiterates that low flows become more extreme as the emissions scenarios increase. A comparison of the H values from each of the tests in which H_0 is rejected suggests that the most significant variation in flows occurs when Q≥Q95, indicated by a bigger H value. Variation in summer flows and Q>Q50 are almost identical, with summer flows having a marginally bigger H value.

Table 5.2.2 - Kruskal-Wallis test results comparing low flows across the emissions scenarios

	Q>Q50	Q≥Q95
α	0.05	0.05
Critical value	9.49	9.49
H value	28.29	54.98
P	0.000	0.000
Confidence level	>99.9%	>99.9%
H_0	Rejected	Rejected

5.3 The impact of the weir removal scenario on discharge

5.3.1 Seasonal flows

Compared to the change in seasonal mean daily discharge resulting from the emissions scenarios (figure 5.2.1.2), the weir removal scenario has a smaller impact. The weir removal scenario results in an overall increase in the annual mean daily discharge (figure 5.3.1) compared to the equivalent 'with weir, current forest cover' scenarios. The impact of the 'without weir' scenario becomes smaller with higher emissions scenarios, however. This is also the case for both winter and summer mean daily discharge. Mean daily discharge also increases in autumn. However, here the impacts of the 'without weir' scenario increases with higher emissions scenarios, with the exception of the baseline emissions scenario. Spring is the only season that sees a decrease in mean daily discharge in the weir removal scenario and the decrease caused by the weir removal scenario grows with higher emissions scenarios.

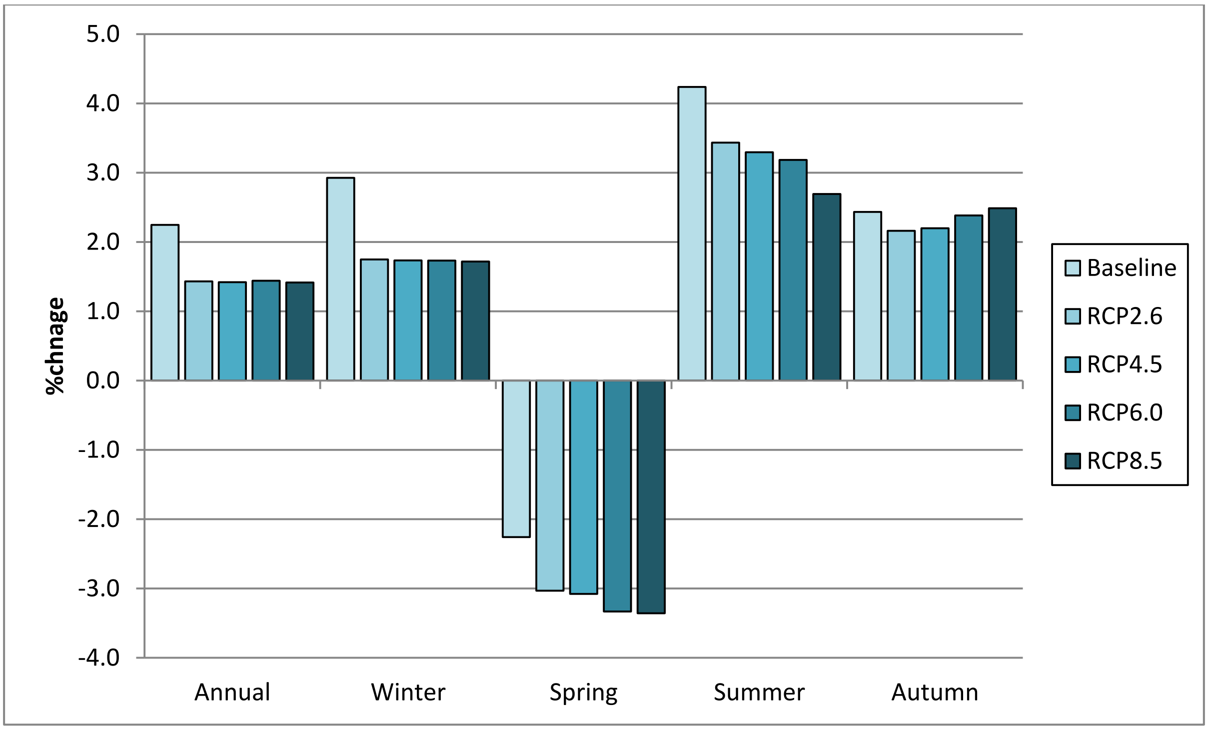


Figure 5.3.1 - Percentage change to mean daily discharge by season without weir (current forest cover) compared with the equivalent 'with weir, current forest cover' emissions scenarios

The Kruskal-Wallis tests indicate that there is no significant variation between seasonal discharge with and without the weir (table 5.3.1). The weir removal scenario causes the greatest variation in summer flows, which is in agreement with the results shown in figure 5.3.1. When compared to the P values from the corresponding tests for the emissions scenarios (table 5.2.1), it is evident that there is less variation in mean daily discharge across the seasons resulting from the weir removal scenario.

Table 5.3.1 - Kruskal-Wallis test results comparing seasonal discharge with and without the weir (current forest cover, baseline emissions)

	Annual	Winter	Spring	Summer	Autumn
α	0.05	0.05	0.05	0.05	0.05
Critical value	3.84	3.84	3.84	3.84	3.84
H value	0.09	0.003	0.07	0.26	0.07
P	0.768	0.959	0.797	0.612	0.786
Confidence level	23.2%	4.1%	20.3%	38.8%	21.4%
H_0	Accepted	Accepted	Accepted	Accepted	Accepted

5.3.2 Low flows

In the weir removal scenario, discharge again becomes flashier (figures 5.3.2.1 and 5.3.2.2). The range of discharge increases in the baseline emissions scenario compared to the same emissions scenario for the 'with weir' scenario. However, for the higher emissions scenarios the percentage decrease caused by the weir removal scenario is smaller. The IQR, however, is reduced in the weir removal scenario, suggesting less variation in the middle 50% of discharge, but greater dispersion either side of this. Again, the impact of weir removal scenario on the IQR diminishes as the emissions scenarios become more extreme. In the weir removal scenario, the median mean daily flow increases, which again suggests an overall increase in discharge across the range of flows. The percentage increase caused by the 'without weir' scenario peaks for RCP6.0, with the smallest increase for RCP8.5. Q_1 decreases in the weir removal scenario, with the exception of RCP2.6, which sees a small increase. In general, the decrease in Q_1 caused by the weir removal scenario grows as the emissions scenarios increase. Finally, the weir removal causes a decrease in minimum flows and this decline grows with higher emissions scenarios, with the exception of RCP8.5, which sees a smaller decrease than RCP6.0.

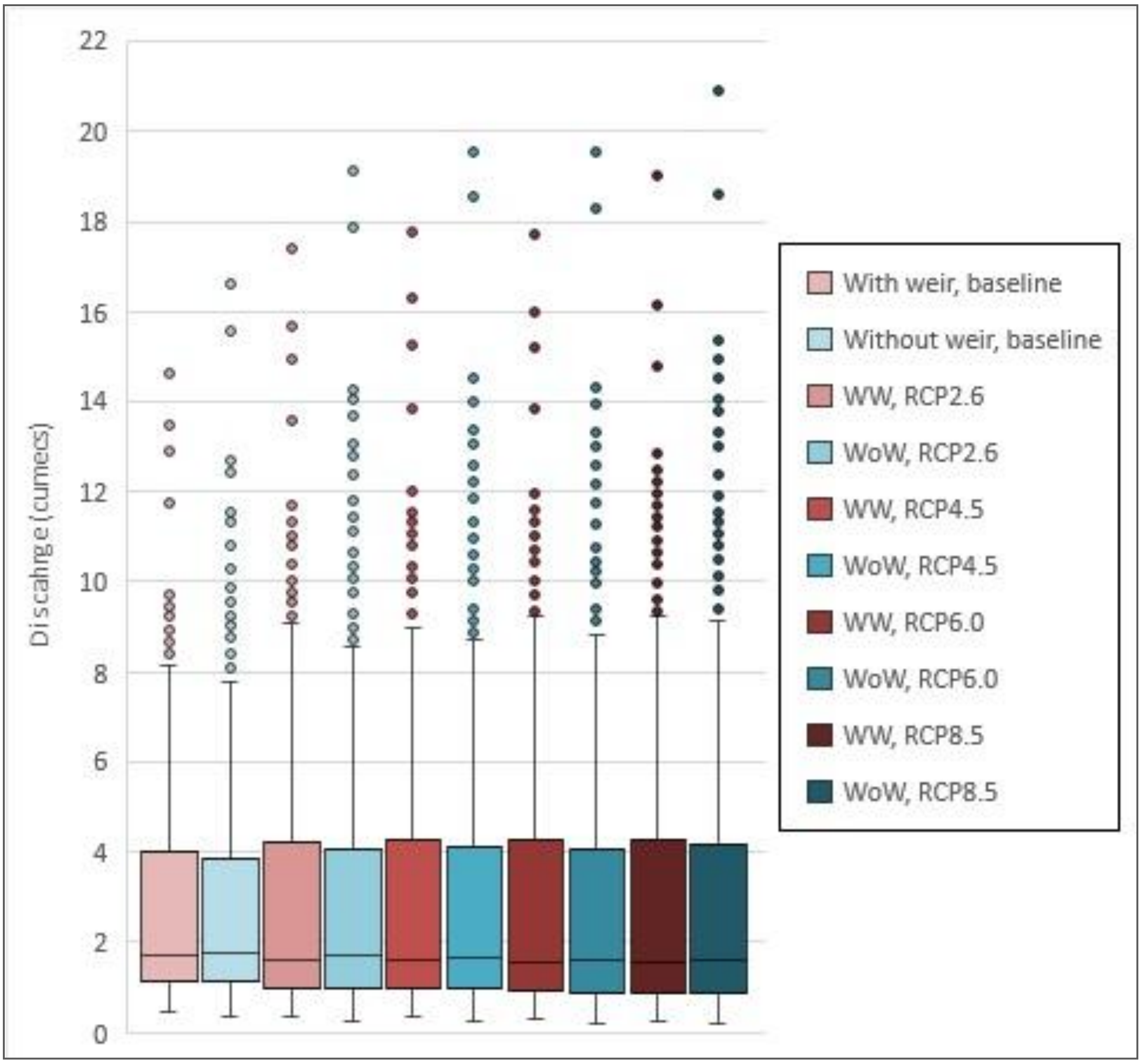


Figure 5.3.2.1 - Box plot of mean daily discharge with and without the weir (current forest cover)

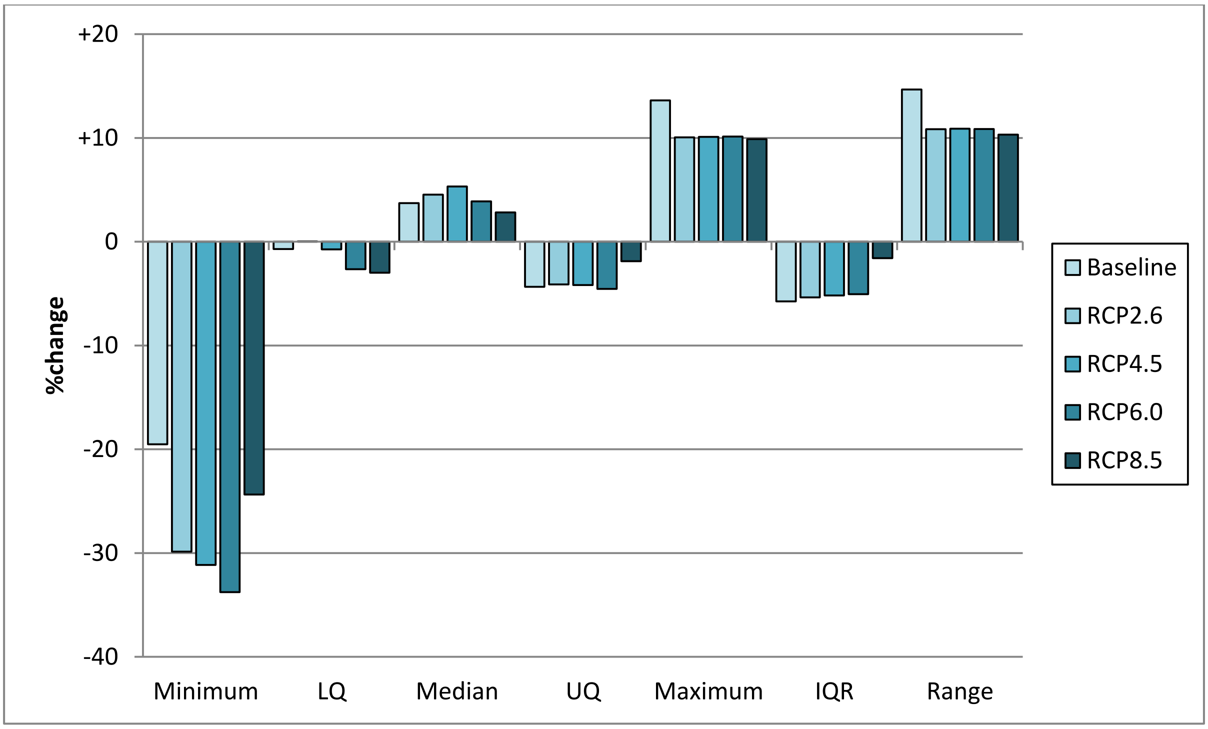


Figure 5.3.2.2 - Percentage change to measures of dispersion and the median mean daily discharge (current forest, without weir) compared to the equivalent 'current forest, with weir' emissions scenario

The flow-duration curve (figure 5.3.2.3) illustrates how, overall, discharge also becomes flashier in the weir removal scenario. As with the emissions scenarios, the magnitude of both low and peak flows increases, indicated by the steeper curves. However, discharges within ~20-70% exceedance remain similar in the with and without weir scenarios. Therefore the weir removal scenario appears to mainly affect only the lowest ~30-40% of discharge values. Higher emissions scenarios appear to somewhat negate the impact of the weir removal scenario on low flows, with only the lowest ~30% of discharge values affected. Conversely, for the baseline emissions scenario, the lowest ~40% decrease with weir removal. Like with the emissions scenarios, the decrease in low flows becomes relatively more pronounced towards 100% exceedance.

Q95 is reduced by the weir removal scenario (figures 5.3.2.4 and 5.3.2.5). This impact grows as the emissions scenarios become more extreme.

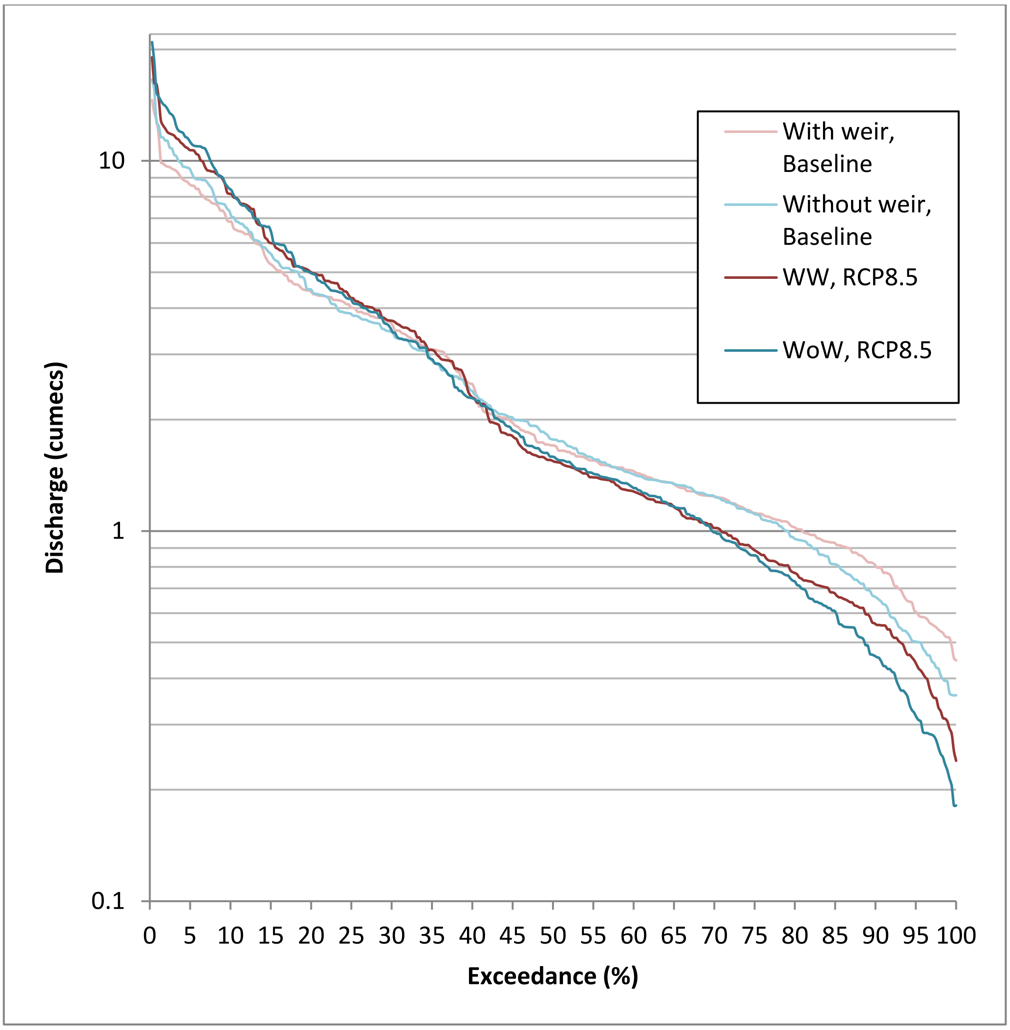


Figure 5.3.2.3 - Flow-duration curve: with and without weir (current forest cover)

N.B. Only the discharge for baseline emissions and RCP8.5 are included to indicate the range of flows across the emissions scenarios and to avoid overloading the graph.

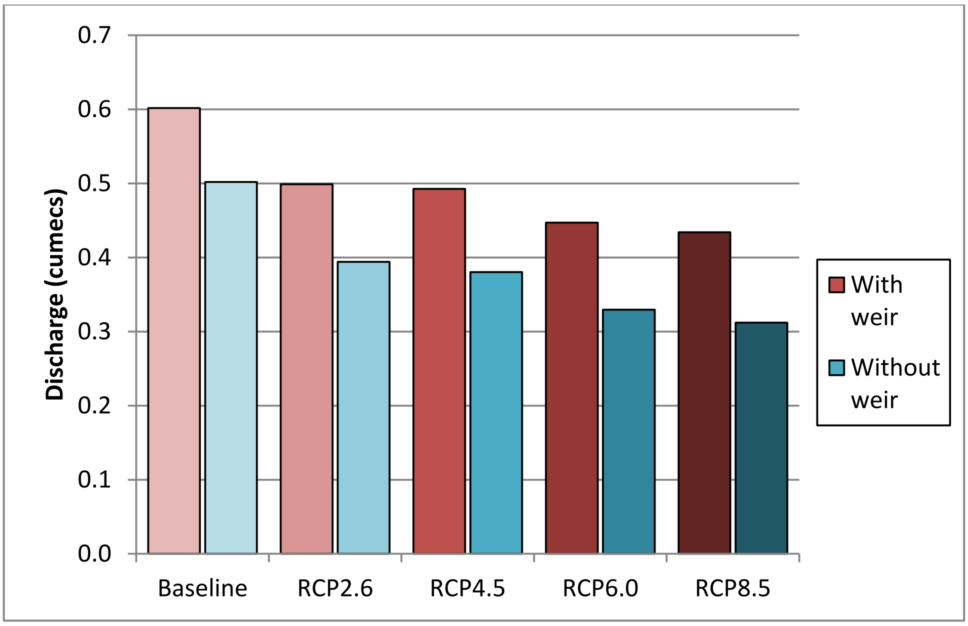


Figure 5.3.2.4 - Q95 with and without weir (current forest cover)

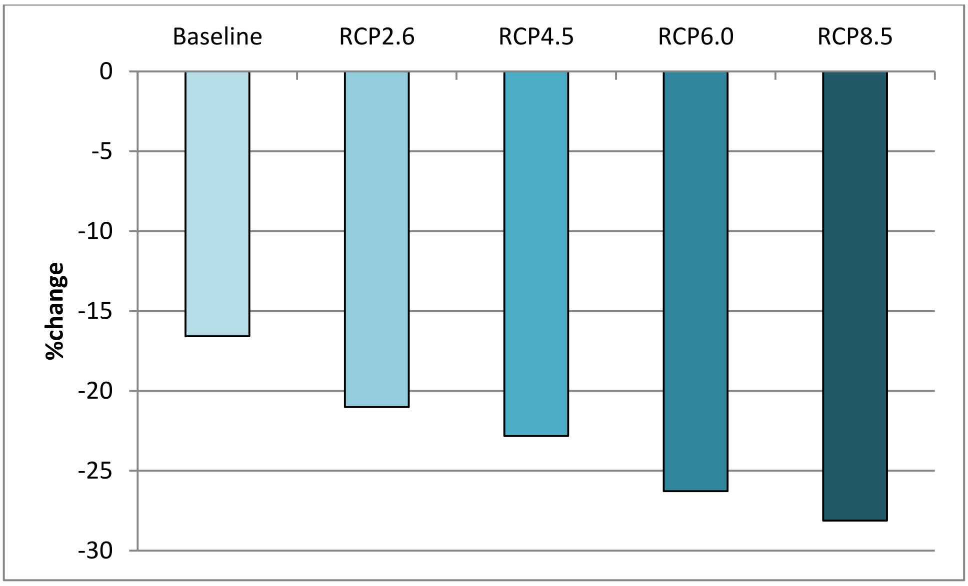


Figure 5.3.2.5 - Percentage change to Q95 (without weir, current forest) compared to the equivalent 'with weir, current forest' emissions scenario

Results for the Kruskal-Wallis tests (table 5.3.2) indicate no significant difference for Q>Q50 caused by the weir removal scenario. H_0 is therefore accepted. However, there is a statistically-significant variation in Q≥Q95 and H_0 is therefore rejected. This reinforces the flow-duration curve (figure 5.3.2.3), which showed that the weir removal scenario only has a significant impact on the lowest discharge values. When comparing the P values with those from the corresponding Kruskal-Wallis tests for the emissions scenarios (table 5.2.2), the impact of the weir scenario on Q≥Q95 is just as significant.

Table 5.3.2 - Kruskal-Wallis test results comparing low flows with and without the weir (current forest cover, baseline emissions)

	Q>Q50	Q≥Q95
α	0.05	0.05
Critical value	3.84	3.84
H value	0.10	21.93
P	0.318	0.000
Confidence level	68.2%	>99.9%
H_0	Accepted	Rejected

5.4 The impact of the future forest cover scenario on discharge

5.4.1 Seasonal flows

Across the whole year and each season, the future forest cover scenarios results in a decrease in mean daily discharge (figure 5.4.1). With higher emissions scenarios, the future forest scenarios cause a more pronounced decline, almost without exception. The decrease in discharge across the year resulting from the increased forest cover scenarios is almost identical in the 'with weir' and 'without weir' scenarios. There is a slight decrease in winter discharge, and the impact of the future forest scenario is more pronounced in the 'without weir' scenario. Conversely, the future forest scenario causes a larger decrease in spring under the 'with weir' scenario. The future forest scenarios cause the largest decrease to summer discharge and, like winter, the decrease is larger for the weir removal scenario. Finally, like summer the decrease in autumn is greater than that of the year as a whole, and the impact of the future forest scenario is larger in the 'with weir' scenario compared with the weir removal scenario.

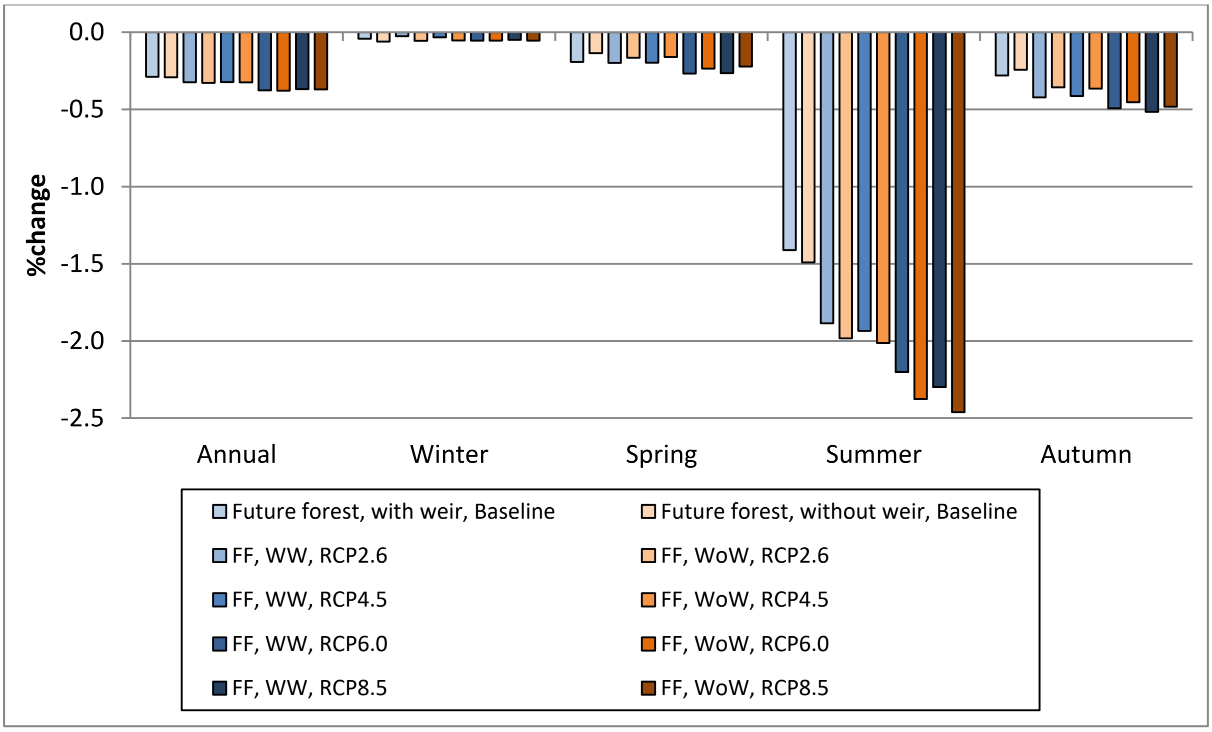


Figure 5.4.1 - Percentage change to mean daily discharge by season resulting from the 'future forest' scenario (with and without weir) compared to the equivalent 'current forest' weir and emissions scenarios

The Kruskal-Wallis tests (table 5.4.1.1 and 5.4.1.2) indicate that for all seasons, the increased forest cover scenarios cause no significant variation in discharge when compared with the scenarios with current forest cover. Comparing the H values, there is more variation over the whole year, and in spring and summer, for the 'future forest cover, with weir' scenario, and greater variation in winter and autumn for the 'future forest cover, without weir' scenario. The most significant change in discharge occurs in summer. When comparing the results with the corresponding emissions and weir scenario tests (tables 5.2.1 and 5.3.1) it is evident that the increased forest scenarios have the least impact on seasonal discharge.

Table 5.4.1.1 - Kruskal-Wallis test results comparing seasonal discharge with current and future forest cover (with weir, baseline emissions)

	Annual	**Winter**	**Spring**	**Summer**	**Autumn**
α	0.05	0.05	0.05	0.05	0.05
Critical value	3.84	3.84	3.84	3.84	3.84
H value	0.006	0.002	0.003	0.06	0.000002
P	0.939	0.963	0.956	0.803	0.999
Confidence level	6.1%	3.7%	4.4%	19.7%	0.1%
H_0	Accepted	Accepted	Accepted	Accepted	Accepted

Table 5.4.1.2 - Kruskal-Wallis test results comparing seasonal discharge with current and future forest cover (without weir, baseline emissions)

	Annual	Winter	Spring	Summer	Autumn
α	0.05	0.05	0.05	0.05	0.05
Critical value	3.84	3.84	3.84	3.84	3.84
H value	0.00001	0.003	0.0004	0.02	0.004
P	0.997	0.957	0.985	0.881	0.952
Confidence level	0.3%	4.3%	1.5%	11.9%	4.8%
H_0	Accepted	Accepted	Accepted	Accepted	Accepted

5.4.2 Low flows

The increased forest cover scenario results in a small decrease in the range of discharge (figures 5.4.2.1-5.4.2.3). Discharge is therefore less flashy. This is the case for both the 'with weir' and 'without weir' scenarios. The impact of the 'future forest cover, without weir' scenario on the range is greater, though the impact diminishes with higher emissions scenarios. Across all emissions scenarios, the future forest scenario results in the largest range (discharge is flashiest) under the 'without weir' scenario. The 'future forest cover, with weir' scenario has a variable effect on the IQR across the emissions scenarios. Three out of the five emissions scenarios result in a percentage decrease and reductions are greater than increases, however. For the 'future forest cover, without weir' scenario, the IQR increases. This suggests greater variation in the middle 50% of discharge values, despite the smaller range overall. Median flows are reduced by both future forest cover scenarios. The future forest scenario has a bigger impact on median flows under the 'without weir' scenario. With the exception of RCP6.0 ('with weir' scenario) and baseline emissions ('without weir' scenario), the increased forest cover scenarios causes a decrease in Q_1. Lastly, the increased forest cover scenario cause the largest decrease in minimum discharge, and this decline occurs across all of the emissions scenarios, and for both the 'with weir' and 'without weir' scenarios. The lowest minimum discharges are experienced in the 'future forest cover, without weir' scenario.

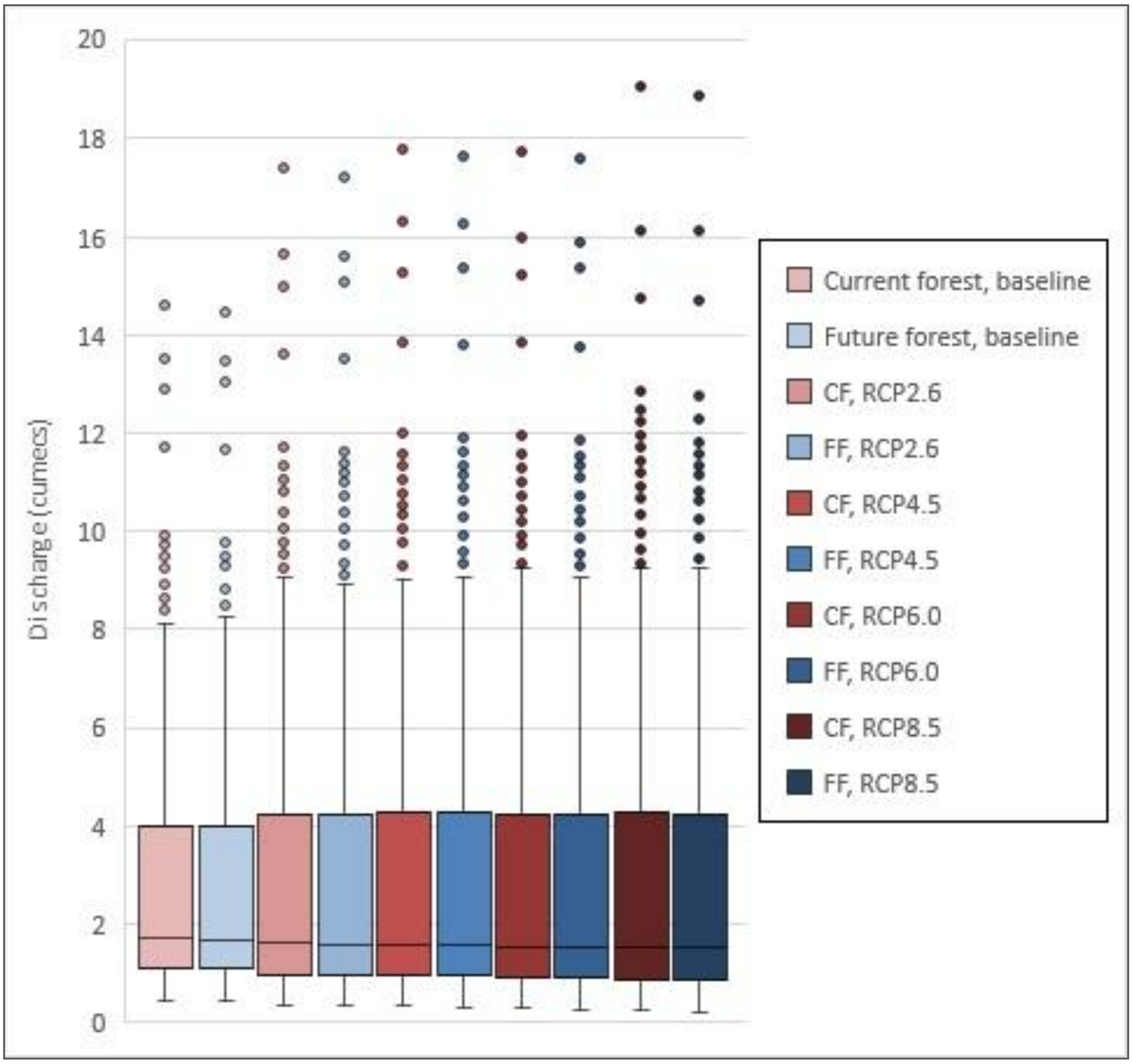


Figure 5.4.2.1 - Box plot of mean daily discharge with current and future forest cover (with weir)

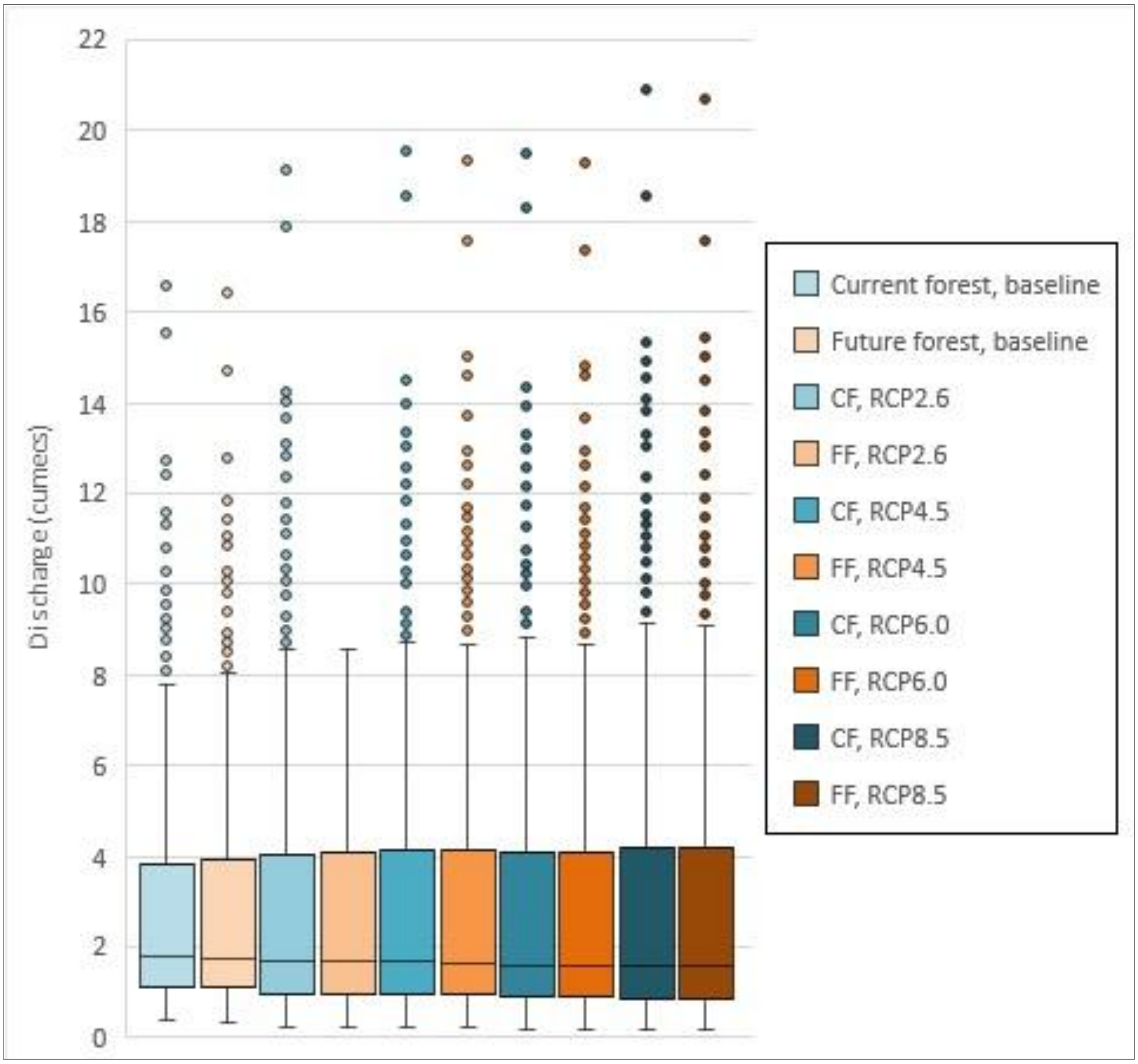


Figure 5.4.2.2 - Box plot of mean daily discharge with current and future forest cover (without weir)

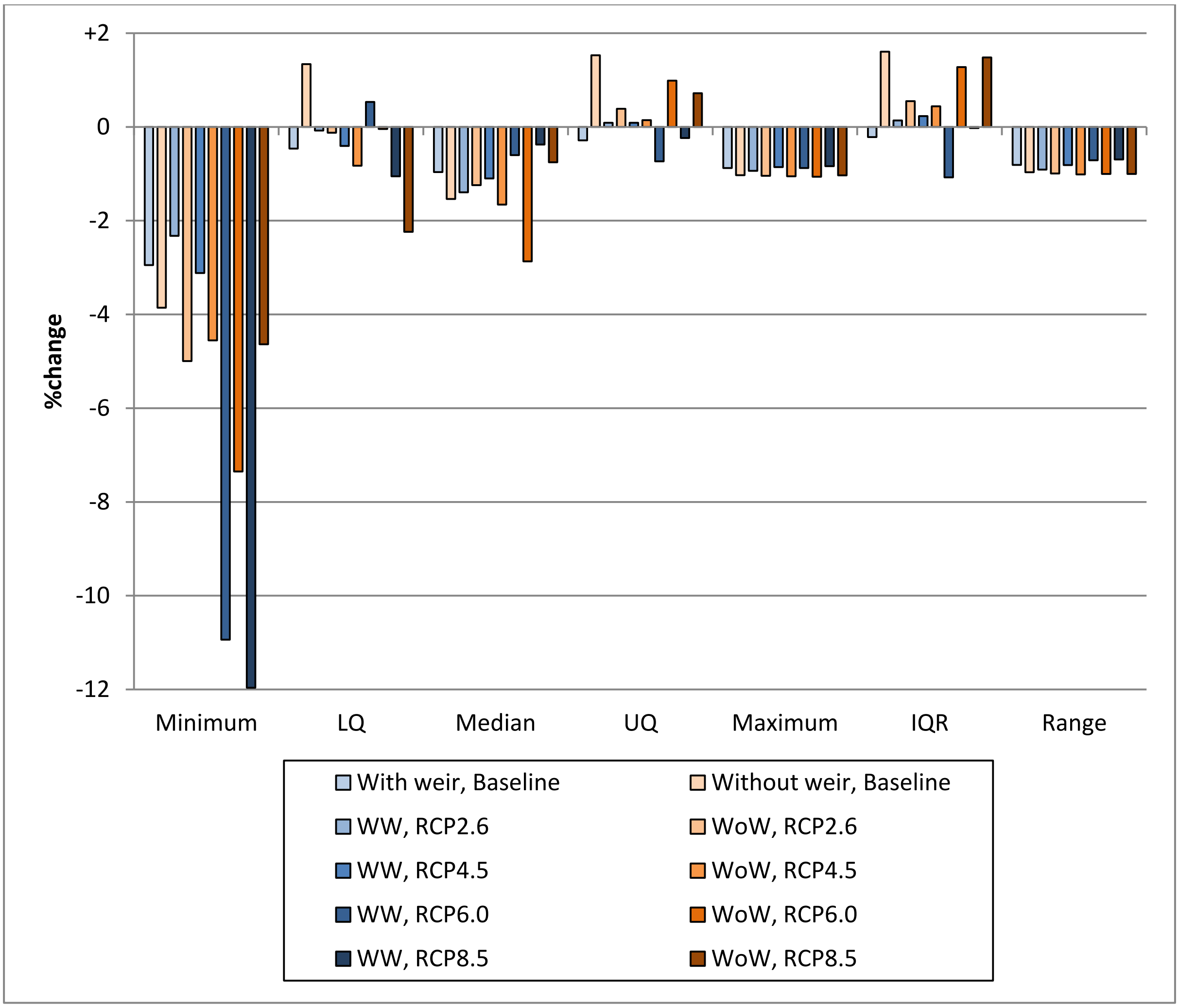


Figure 5.4.2.3 - Percentage change to measures of dispersion and the median mean daily discharge resulting from the 'future forest' scenario (with and without weir) compared to the equivalent 'current forest' weir and emissions scenario

The flow-duration curves (figures 5.4.2.4 and 5.4.2.5) show that the increased forest cover scenarios result in very little change to discharge across the range of flows. For lower flows, there is a slight decrease in discharge caused by the increased forest cover scenarios, particularly with higher emissions, indicated by greater divergence between the RCP8.5 lines.

Q95 (figures 5.4.2.6-5.4.2.8) is reduced by the future forest cover scenarios, with the exception of the baseline emissions scenario ('with weir'), which results in a slight increase. The future forest cover scenario causes a greater decline in Q95 under the 'without weir' scenario.

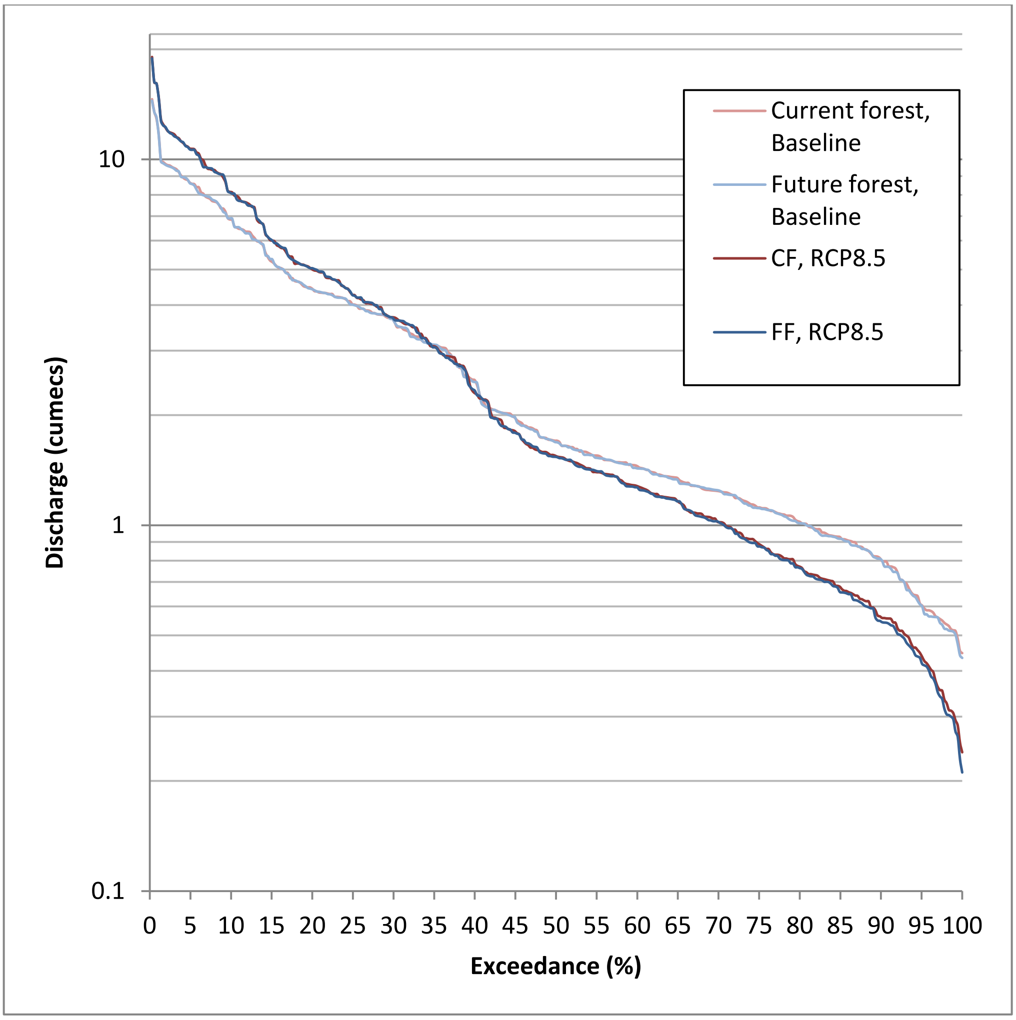


Figure 5.4.2.4 - Flow-duration curve for current and future forest cover (with weir)

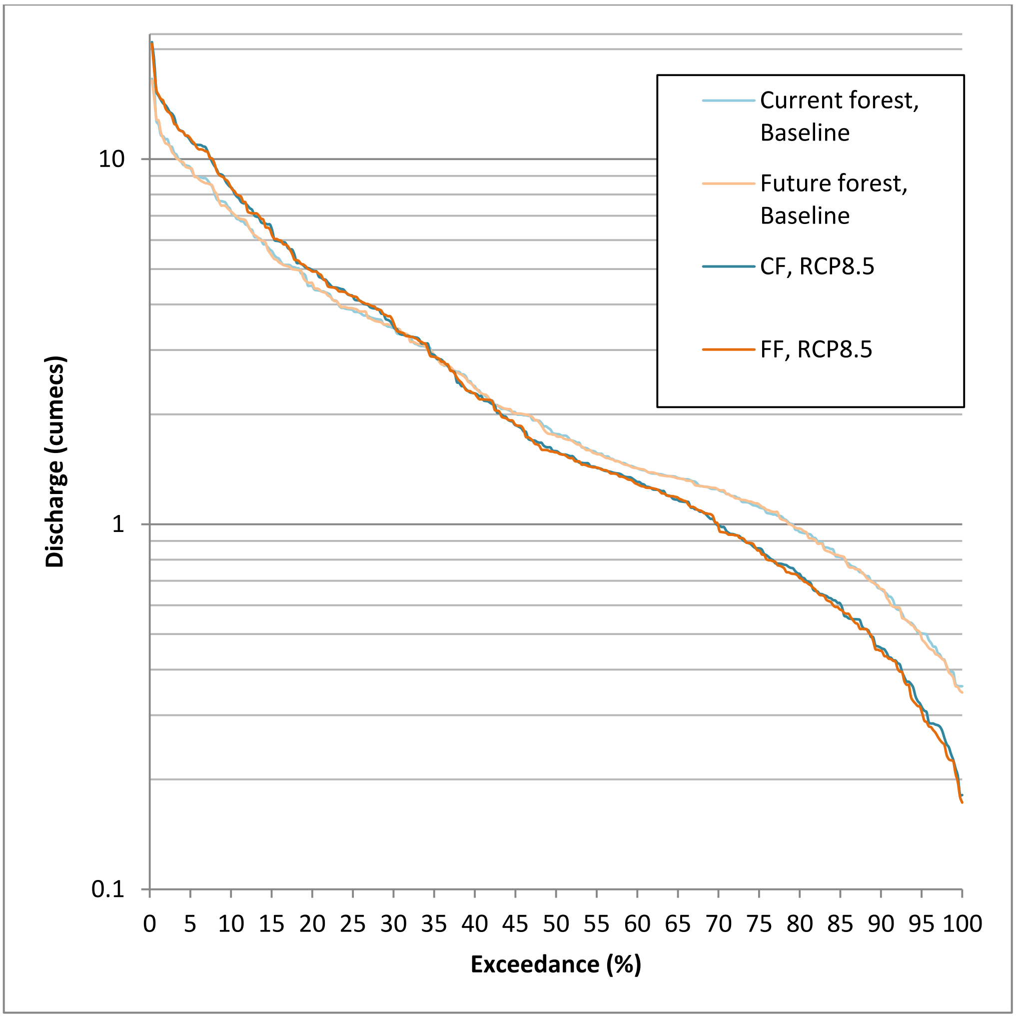


Figure 5.4.2.5 - Flow-duration curve for current and future forest cover (without weir)

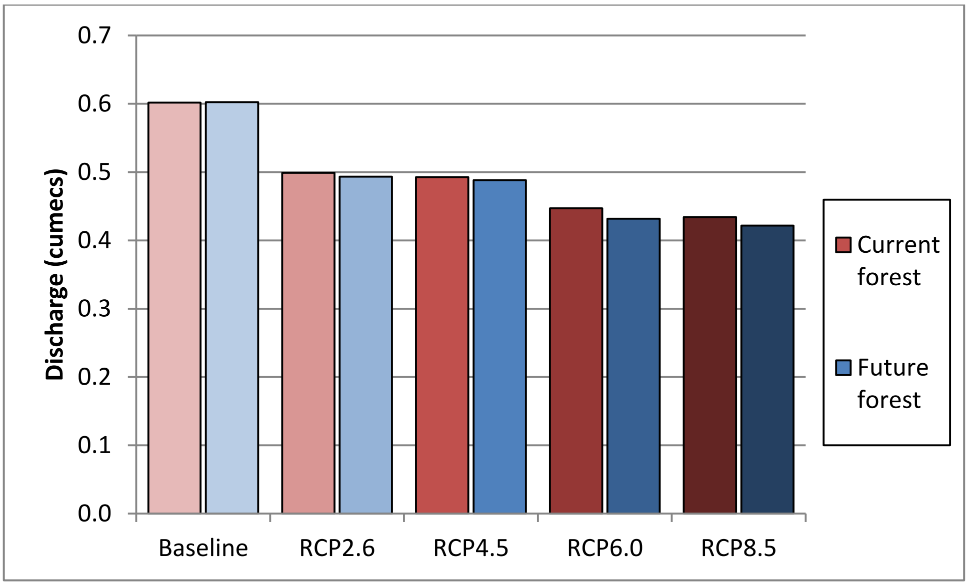


Figure 5.4.2.6 - Q95 for current and future forest cover (with weir)

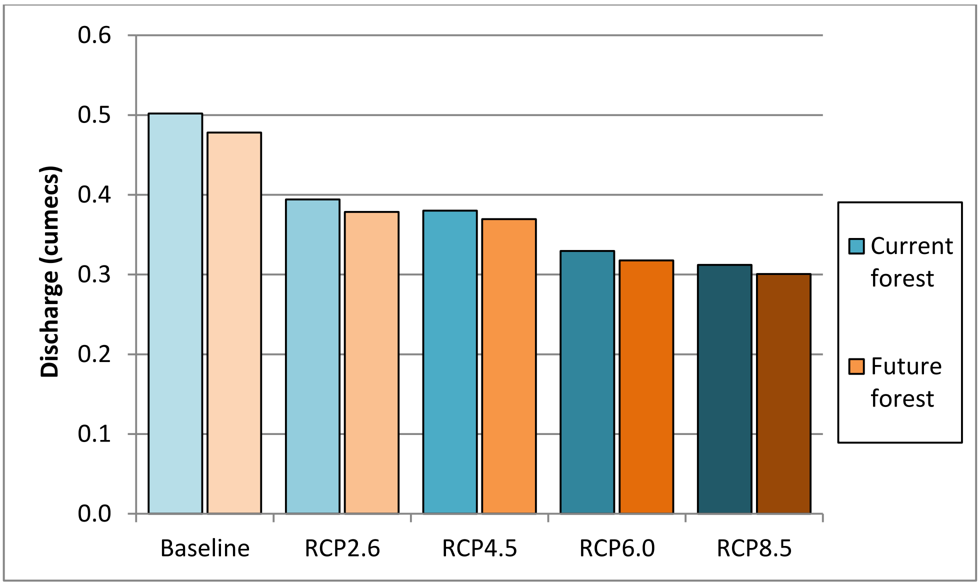


Figure 5.4.2.7 - Q95 for current and future forest cover (without weir)

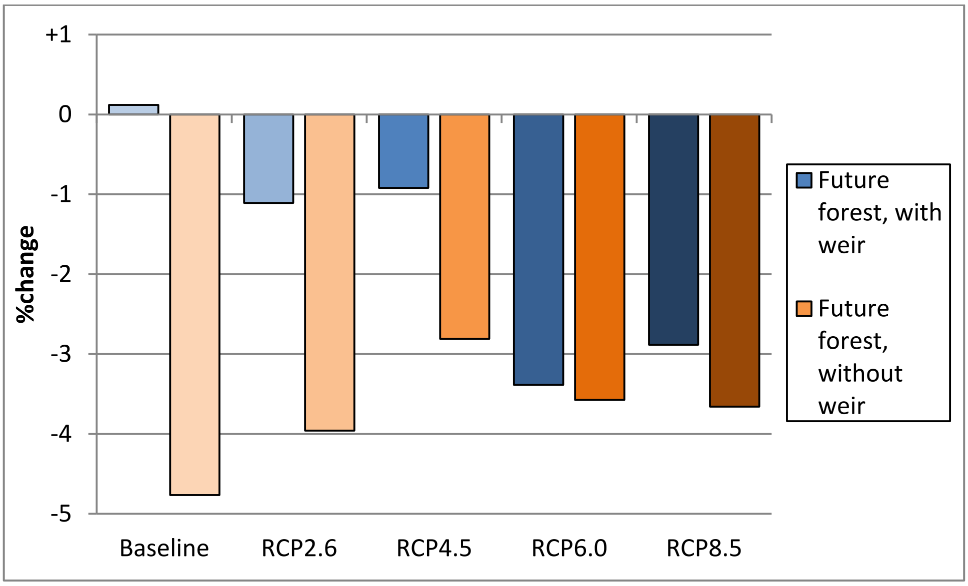


Figure 5.4.2.8 - Percentage change to Q95 resulting from the 'future forest' scenario (with and without weir) compared to the equivalent 'current forest' weir and emissions scenario

Results for the Kruskal-Wallis tests (table 5.4.2) indicate that the increased forest cover scenarios have no significant effect on low flows. H_0 is therefore accepted. This reiterates that the 'future forest' scenarios have minimal impact on low flows, and less impact compared to the emissions and weir removal scenarios. The 'future forest cover, with weir' scenario has a bigger impact on both Q>Q50 and Q≥Q95 than the 'future forest cover, without weir' scenario.

Table 5.4.2 - Kruskal-Wallis test results for discharge with future forest cover (baseline emissions)

	With weir		Without weir	
	Q>Q50	**Q≥Q95**	**Q>Q50**	**Q≥Q95**
α	0.05	0.05	0.05	0.05
Critical value	3.84	3.84	3.84	3.84
H value	0.03	1.09	0.001	0.63
P	0.866	0.296	0.975	0.429
Confidence level	13.4%	70.4%	2.5%	57.1%
H_0	Accepted	Accepted	Accepted	Accepted

5.5 Summary

Overall, the emissions scenarios have the greatest impact on discharge and cause a significant decrease in summer flows, as well as the lowest 50% and 5% of flows. In general, the weir removal scenario results in an increase in discharge but causes a significant decrease in the lowest 5% of flows. The increased forest cover scenarios have the least and no significant impact on discharge, but do cause a slight decrease in flows across the board, particularly low flows.

6. Discussion

The following chapter 1) evaluates SHETRAN-Reservoir as a tool to model the impacts of reservoir decommissioning; 2) discusses the present study's findings on the impacts of climate change, reservoir decommissioning and increased forest cover on river flows in relation to the wider research base; considers the implications of these findings for the 3) hydroecology and 4) management of the Upper Ehen catchment, as well as 5) UK policy; and 6) concludes with suggestions for further research arising from the study.

6.1 Evaluation of SHETRAN-Reservoir

The relative performance of the 'lake' and 'reservoir' models (see section 5.1) backs up Hughes *et al.*'s (2021) assertion that SHETRAN-Reservoir enables more accurate modelling of catchments that include reservoir operations, particularly low and peak flows. This has important implications for hydroecologists and reservoir operators (Hughes *et al.*, 2021). Furthermore, comparing simulations with those produced by SHETRAN-Standard allows for the hydrological impacts of (the cessation of) reservoir operations to be assessed. This can help catchment and water resource managers to make more informed decisions in terms catchment restoration and water resource planning. Against the backdrop of climate change this development this is even more crucial.

There is a wide variety of reservoir operating rules used in modelling (Lund, 1996), which reflect the myriad contexts in which reservoirs exist. SHETRAN-Reservoir makes the assumption that reservoir operations such as environmental flow release is a function of reservoir stage (Hughes *et al.*, 2021). SHETRAN-Reservoir could therefore be developed to encompass a broader range of reservoir operations and control structures in order to improve model accuracy. The weir at Ennerdale Water contains an automated penstock and pump system, which responds whenever discharge at the downstream Bleach Green gauging station (rather than reservoir stage) falls below certain thresholds. As stated in table 4.5, a QQ function that informs environmental flow releases past the weir would be more appropriate in this context. Furthermore, abstractions from Ennerdale Water are offset by abstractions from nearby boreholes during dry periods. A function that enables abstraction rules to be set for such cases, especially when historic abstraction data is unavailable, would result in more accurate simulation of reservoir stage and, as a result, downstream discharge.

6.2 The impact of climate change on seasonal and low flows in the River Ehen

Changes to seasonal flows in the River Ehen arising from climate change fall broadly in line with findings from national-scale studies (e.g. Christierson *et al.*, 2012; Prudhomme *et al.*, 2012; Sanderson *et al.*, 2012; see table 3.1.2.1). These changes can be put down to the seasonal variations in rainfall and PE shown in table 4.3.2. However, the changes presented in this study are more extreme than that of Sanderson *et al*. (2012) whose projections for 2080 are the closest match to this study in terms of timeframe. Like Prudhomme *et al.* (2012) found, the greatest percentage change was found to occur in summer but the greatest absolute change in discharge was experienced in winter, which explains the overall annual increase in mean daily discharge. In this regard, this study's results correspond more closely to those of Sanderson *et al.* (2012) than of Prudhomme *et al.* (2012), the latter of which found annual mean discharge to remain similar because changes to seasonal discharge (an increase in winter and decrease in summer) evened out. This highlights how different methodologies and model scales can generate discrepancies in results, which could results in inappropriate management planning and policy making.

Many studies assert that climate change is going to cause the hydrological cycle to become more extreme, with more frequent floods and droughts in general (e.g. Bates *et al.*, 2008; Rockström *et al.*, 2009; Giorgi *et al.*, 2011) and NW England (e.g. Collet *et al.*, 2018; Kay *et al.*, 2018, 2021; Rudd *et al.*, 2019; Dobson *et al.*, 2020). This study has demonstrated that this is also likely to be the case in the Upper Ehen catchment. This can be explained by an increase in annual temperatures in NW England, which is expected to result in wetter winters and drier summers (Lowe *et al.*, 2018), as well as an increase in PE (Prudhomme and Williamson, 2013). Even under the most conservative emissions scenario it can be expected that discharge will become flashier and significant decreases in low flows due to climate change are highly probable, with both Q_1 and Q95 declining. The latter finding is in agreement with Charlton and Arnell (2014) and Kay *et al.* (2021) who also found that climate change will cause low flows to decrease in NW England. However, this finding is at odds with that of Rudd *et al.* (2019) who found that the median river flow drought intensity in NW England is projected to decrease by 6% in the 2070-2099 time-slice compared with the 1970-1999 baseline. Again, this illustrates how discrepancies in results are caused by methodological differences.

6.3 The impact of weir removal on seasonal and low flows in the River Ehen

The removal of the weir at Ennerdale Water is likely to see an increase in mean daily discharge in the River Ehen, as well the median mean daily discharge. This can be put down to the cessation of abstractions (as part of the Thirlmere transfer scheme; Jacobs, 2017), a process which takes water out of the catchment's hydrological cycle that could otherwise leave the catchment via the River Ehen. An unexpected result was the increase the summer discharge, when mean daily discharge generally being at or below 1 cumec (see figure 9.2.2 in appendix B), despite weir removal causing a decline in discharge during dry periods. This can be explained by the fact that mean discharge (and therefore percentage change to mean discharge) is skewed by bigger values (Maity, 2018). Consequently, peak flows will have had a larger effect on seasonal means than low flows. There was a peak flow (with discharge up to 10 cumecs) in each of the three summer months of the simulation year and weir removal resulted in an increase in peak flows (see section 5.3.2), which is likely to have caused the overall increase in mean summer daily discharge. Changes to seasonal mean daily flows in any given year are therefore largely a product of the weather patterns of that particular year and are not necessarily a reliable indicator of longer-term changes to river flows.

In the case of weir removal, flow-duration analysis is likely to offer more insights than seasonal analysis. Weir removal could see discharge in the River Ehen become flashier. The River Ehen SAC designation sees it protected by mandatory environmental flow releases and there is a high probability that ceasing these will see low flows decline during dry periods and compound the decrease resulting from climate change. This is in accordance with Magilligan and Nislow (2005), who found that dam construction in the US caused increased low flows due to controlled releases, inferring that the inverse is the case with dam removal. This can be explained by Williams and Wolman's (1984) assertion that the impact of dams (and therefore their removal) on low flows depends on the dam release policy. At the other end of the spectrum, the absence of an impounding structure could result in an increase to peak flows. This provides further evidence that dams reduce downstream flood risk (e.g. World Commission on Dams, 2000; Nislow *et al.* (2002); Magilligan *et al.* (2003); Schmutz and Moog, 2018).

6.4 The impact of increased native forest cover on seasonal and low flows in the River Ehen

Overall, the impact of increased native cover on discharge in the River Ehen is small and less significant than that of climate change and weir removal. This backs up the assertion that the impact of forest management on extreme river flows at the catchment scale are minimal (Robinson *et al.*, 2003).

Increased forest cover could slightly reduce mean daily discharge in the River Ehen both annually and seasonally. This backs up research suggesting that trees reduce surface runoff (e.g. Blackie, 1993; Marshall *et al.*, 2009, 2014) through increased infiltration, interception and evapotranspiration (e.g. Bird *et al.*, 2003; Dixon *et al.*, 2014; Bischoff *et al.*, 2015; Thomas and Nisbet, 2016). A lowered water table (Robinson *et al.*, 2003) and reduced soil moisture reserves (Hudson, 1988; Robinson and Cosandey, 2002) caused by an increase in trees in the catchment could also be a factor. This rise in forest cover from ~19% to ~40%, a change which would largely see grassland replaced by trees, backs up the assertion that discharge is lower in catchments with high levels of forest cover rather than majority grassland cover (e.g. Marc and Robinson, 2007; Bathurst *et al.*, 2018). However, the decrease in annual discharge seen in this study is less than that of the aforementioned studies. This is likely due to those studies compared catchments with 100% forest and 100% grassland cover.

Like other studies found (e.g. Hornbeck *et al.*, 1993; Johnson and Black, 1997; Robinson and Dupeyrat, 2003), the season that experienced the greatest decrease in discharge with increased native forest cover was summer. This can be explained by the change in forest structure from being largely coniferous to mainly broadleaves. Summer is when leaf cover in broadleaf forest is greatest, resulting in increased interception and evapotranspiration (Wehr *et al.*, 2016). On the other hand, mean winter daily discharge only slightly decreases, suggesting that the loss of leaf cover may only result in minimal flood mitigation.

Discharge could become less flashy with increased native forest cover, illustrated by a decrease in the range of discharge values and peak flows. This backs up studies that have found forests to decrease flood risk (e.g. Linstead and Gurnell, 1999; Broadmeadow and Nisbet, 2010; Dixon *et al.*, 2016). However, for the same reason, increased forest cover could see a decline in Q95 by up to 5% and minimum flows by 12%. This corresponds with findings from other studies (e.g. Hornbeck *et al.*, 1993; Johnson and Black, 1997; Robinson and Dupeyrat, 2003; Robinson *et al.*, 2003). Reduced runoff and increased evapotranspiration due to greater forest cover could be a factor here. The increase in the percentage cover of broadleaves in the catchment could also be responsible, given that low flows are most common in summer, when leaf cover peaks.

6.5 The hydroecological impacts of future climate change, decommissioning Ennerdale Water and increased native forest cover in Ennerdale on the River Ehen during dry periods

More extreme river flows caused by climate change could have significant impacts on the hydroecology of the River Ehen. Discharge is an important constituent of the hydroclimatological-hydroecological process chain (Garner *et al.*, 2017). More extreme low (and peak) flows, and associated changes to water temperature, could cause a decline in *Margaritifera margaritifera, Salmo salar* (as well as other catchment salmonids such as *Salvelinus alpinus* and *Salmo trutta)* and macroinvertebrates (as per Strayer, 1999; Clews *et al.*, 2010; Durance and Ormerod, 2010; Pandolfo *et al.*, 2010; Poff and Zimmerman, 2010; Haag, 2012; Sousa *et al.*, 2018).

Assessing the impacts of decommissioning Ennerdale Water and increasing native forest cover in the Upper Ehen catchment is more complex. As stated in table 3.2.3, weir removal will provide an array of hydroecological benefits to the freshwater ecosystem of the Upper Ehen catchment, such as improved longitudinal connectivity, an increase in riparian habitat and greater nutrient and organic matter content in the River Ehen, all of which improves habitat quality and increases biodiversity (Poff *et al.*, 1997; Wohl *et al.*, 2015; Bellmore *et al.*, 2019). More native woodland is needed to reverse and improve nature's resilience to climate change, as well as improve biodiversity (Reid *et al.*, 2021). Increased native forest cover in the catchment would be in accordance with recent NAP2018 and 25YEP in that it would constitute new and improved woodland, which has the potential to restore degraded ecosystems, (re)introduce wildlife-rich habitat which is more resilient to climate change, improve soil quality and reduce downstream flood risk for nature and communities such as Ennerdale Bridge. Increased native forest cover would also contribute towards meeting the UK Government's target to plant 30 million trees by 2025 in order to reach Net Zero emissions (Ares and Uberoi, 2020).

However, dam removal involves complex trade-offs (Roy *et al.*, 2018). Removing the weir at Ennerdale Water could exacerbate low and peak flows, as well as the associated changes to thermal regime, compounding the aforementioned hydroecological issues caused by climate change. Additionally, as Beatty *et al.* (2017) assert, a reduction in the size and depth of Ennerdale Water caused by weir removal would mean the loss of a refuge for the catchment's salmonids during dry periods. Increased forest cover could also intensify low flows and this should be considered, as per the UK Forestry Standard (Forestry Commission, 2017). On balance, because the changes to river flows caused by increased forest cover were not found to be statistically significant in this study and because the benefits are otherwise manifold, it is advised that native forest cover should increase, though management considerations are presented in section 6.6.

6.6 Suggestions for land managers of the Upper Ehen catchment

It is clear that the combined positive and negative hydroecological effects of reservoir decommissioning and increased forest cover against a backdrop of climate change make for a complex, multifaceted management issue. Any potential solution requires an approach which carefully considers each impact and aims to find balance between the benefits of catchment restoration and the negative consequences of climate and land-use change. The key challenges facing land managers and suggested management options are shown in figure 6.7.

Climate change is expected to have the most significant impact on the hydroecology of the Upper Ehen catchment. It is therefore unlikely that land-use changes in the catchment alone will provide the solution to a more extreme catchment hydrology. As well the need for a catchment-scale approach to restoration, this study highlights the importance of a multi-scale strategy to tackle the global issue of climate change. Without this, more localised efforts to adapt to and mitigate climate change will enjoy only limited success.

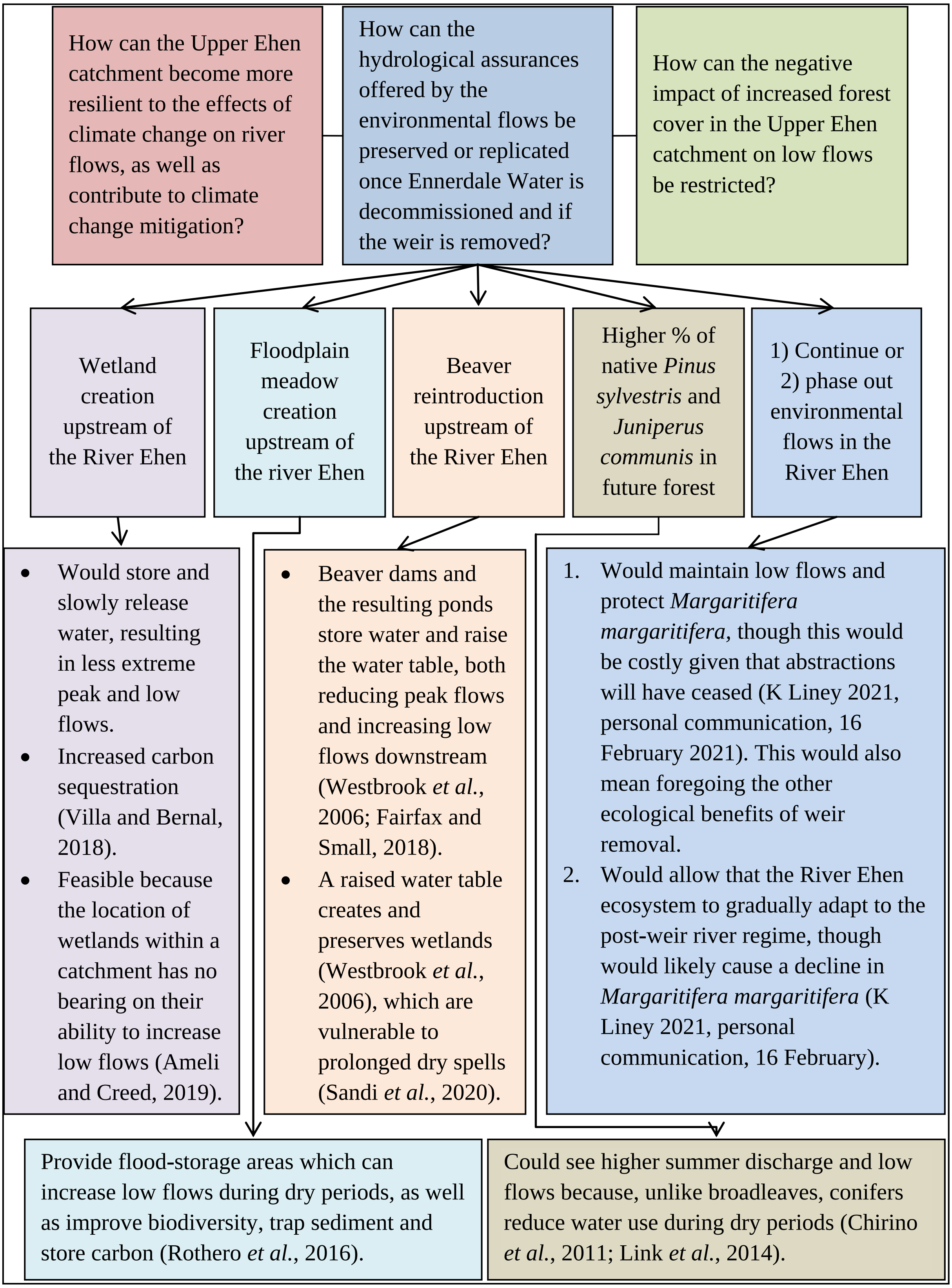


Figure 6.6 - Recommendations for the land managers of the Upper Ehen catchment

6.7 Policy implications

This study highlights two strengths of current UK environmental policy. Firstly, discrepancies between the changes to river flows caused by climate change found in this catchment-scale study and national-scale studies (e.g. Christierson *et al.,* 2012; Prudhomme *et al.*, 2012; Sanderson *et al.*, 2012; Rudd *et al.*, 2019) justifies the catchment-based approach (CaBA) present across UK water management policy. Differences between annual means, seasonal runoff and extreme runoff (Q5 and Q95) for large-scale hydrological models and catchment hydrological models do occur (Gosling *et al.*, 2011), which can have implications for water management. Secondly, integrated policy stemming from the WFD that considers the needs of land, water and people, such as the 25YEP, CaBA and UKFS is necessary to tackle issues around climate change, water security and biodiversity because land, water and people are all elements of the same system. In the case of the Upper Ehen catchment, the terrestrial ecosystem, aquatic ecosystem and people of West Cumbria will all be affected by climate change, as well as land-use changes like decommissioning Ennerdale Water and increased native forest cover. Therefore land managers must consider these in tandem, which is more likely given the integrated nature of current environmental policy.

However, though the UK National Risk Register of Civil Emergencies includes droughts (Cabinet Office, 2017), current UK policy does not cite the issue of drought and low flows as explicitly as it does flooding, despite causing significant social, economic and environmental impacts in the UK and Europe (Kendon *et al.*, 2013; Cammalleri *et al.*, 2020). This study has reiterated that there is a significant chance that climate change will result in more extreme low flows in the UK. Each catchment is required to have a Flood Risk Management Plan under the Floods Directive (Robins *et al.*, 2017) but there is no equivalent directive for drought nor a requirement to produce 'Drought Risk Management Plans'. This policy gap is reflected in management priorities. In a review of 229 case studies in the EU, the majority (56%) of NbS in the EU focused on flood mitigation, with <5% aimed at drought mitigation (Debele *et al.*, 2019). There is an opportunity after Brexit for the UK to introduce legislation and policy focused solely on managing drought risk and address this imbalance.

With the rising number of reservoirs being decommissioned and reformations to water management policy regarding abstractions, policy must also address requirements when reservoirs are decommissioned. This is particularly pertinent for reservoirs that have been required to ensure the provision of environmental flows following the 2011 policy reformations (DEFRA, 2011b, 2011c). Currently, in terms of meeting the requirements of the WFD, river flow standards are only used for identifying the risk posed to the ecological quality of rivers by abstractions and quantifying necessary improvements needed to meet objectives (DEFRA, 2014b). They are not used as a means of measuring ecological quality of rivers unaffected by abstractions, which could have implications for the River Ehen once abstractions are ceased and if the weir is removed.

6.8 Further research

Similar studies in different catchments could explore the hydrological impacts of climate change, reservoir decommissioning and forest management in different geographical contexts. This would allow for more robust evidence to be fed into decision-making and policy. Other potential future research related to climate change, reservoir decommissioning, as well as trees and NbS that would build on this study's findings, develop the research base and aid management practice and policy-making are shown in table 6.8.

Table 6.8 - Recommendations for further research related to climate change, decommissioning reservoirs and nature-based solutions

Climate change	• Modelling a wider range of possible climate change scenarios (such as the 10th and 90th percentiles of each UKCP18 emissions scenario) would cover a greater degree of the uncertainty surrounding potential hydrological impacts of future climate change. • To address uncertainty around the seasonal changes presented in this study, a multi-year study to identify clearer seasonal (and other) trends could be carried out. This would give a more representative idea of how climate change (and land-use) change could on average affect catchment hydrology.
Decommissioning reservoirs	• Due to the interrelated elements of the hydroclimatological-hydroecological process chain, Hart *et al* (2002) argue for the need for integrated studies that consider the variety of impacts of weir removal alongside discharge, such as sediment loads, water temperature, biogeochemical impacts and biotic factors. Such a study would provide a deeper and more detailed understanding of the impacts of removing the weir at Ennerdale Water. • Flow-duration curves give no indication of sequences of low flows,

	which is important because the greatest stress to water supply (and the environment) arises from extended periods of low flows (Shaw *et al.*, 2011). Therefore, an investigation into changes to the timing and duration of extreme events resulting from climate and land-use change could also provide a useful insight for land managers. • Using SHETRAN-Reservoir to model the hydrology of a range of catchments with reservoirs would: 1. develop a deeper understanding of its strengths and weaknesses as a tool for water resource managers and researchers; and 2. improve understanding of the impacts of (decommissioning) reservoirs in a greater number of contexts.
Trees and nature-based solutions	• As section 3.3.2 attests, there is limited understanding of the comparative water use of conifers and broadleaves, as well as inter- and intra-species. In order to gain a better understanding of the water use of different tress and how to achieve optimal results from afforestation, a study that considers the impacts of different tree species, forest and structures and percentage cover could be investigated. This would allow for the comparison of the impacts of forest management on hydrology at the catchment and sub-catchment scale. • Similarly, studies could model the hydrological impacts of other NbS, such as those outlined in section 6.6, in the Upper Ehen catchment and other catchments to ascertain the extent to which they can solve hydrological issues associated with climate and land-use change.
Other	• A more detailed representation of the geology of the Upper Ehen catchment - and how this might change with and influence land-use - could potentially produce more accurate simulations.

7. Conclusion

This study used the recently developed hydrological model SHETRAN-Reservoir to simulate the impacts of future climate change, weir removal and increased native forest cover on discharge in the River Ehen in the English Lake District during dry periods, in order to assess the hydroecological implications of projected changes. The emissions and weir removal scenarios caused discharge in the River Ehen to become more extreme. The emissions scenarios caused a statistically significant decline to summer flows, as well as the lowest 50% and 5% ($Q \geq Q95$) of flows. The weir removal scenario also resulted in a statistically significant decline in $Q \geq Q95$. The increased forest cover scenarios caused a further, albeit smaller decline in discharge, particularly low flows. Though weir removal and increased forest cover have well-documented ecological benefits, such changes to the River Ehen's hydrological regime could have negative hydroecological implications, particularly for the catchment's protected species, *Margaritifera margaritifera* and *Salmo salar*.

A growing number of reservoirs are being decommissioned in the US and UK and, alongside this, UK water management policy has moved towards a catchment-based approach (CaBA) in recent years. For this reason, this study simulated discharge in the Upper Ehen catchment using two models: a standard SHETRAN model without reservoir operations and a SHETRAN-Reservoir model that did include reservoir operations, in order to ascertain the impacts of weir removal on discharge under different UKCP18 emissions scenarios and with current and future forest cover. The 'reservoir' model produced more accurate simulations than the standard 'lake' model, whilst the 'lake' model provided an insight into how the catchment's hydrology might operate when reservoir operations in Ennerdale Water are ceased and if the weir were removed. The development of SHETRAN-Reservoir therefore enables the investigation of the impacts of reservoir decommissioning on catchment hydrology, which has important implications for land managers and policy makers. The software could be developed to allow for a wider range of reservoir management regimes to be more accurately modelled, improving its scope.

The study findings raise interesting questions regarding the impacts of catchment restoration and current UK policy emphasis. A much greater proportion of policy is concerned with flooding than drought, despite both being likely to become more extreme due to climate change. This study reiterates this and therefore it is recommended that greater weight is put on addressing the social, economic and environmental impacts of drought and low flows in

UK policy, especially in catchments in which reservoirs are to be decommissioned. Furthermore, it is apparent that the removal of a weir that has been required to provide environmental flows in order to protect the river downstream does not necessarily guarantee positive hydroecological results. Nor does increased native forest cover. Therefore, it is recommended that land managers consider ways to increase low flows and negate the potential impacts of climate change, weir removal and increased native forest cover on the hydroecology of the River Ehen.

The CaBA organises water management in the UK at the catchment scale. Despite this, the majority of UK studies concerned with climate change and river flows focus on regional and national-scale models, which provide results that not do not directly translate to catchment-scale management. This study has therefore provided a catchment-scale perspective on the impacts of climate change and catchment restoration proposals on seasonal and low flows, as well as a methodology that can be applied to other catchments in order to develop understanding of the hydroecological impacts of climate and land-use change, and aid decision and policy making in line with the CaBA. Furthermore, it has provided insight into the impact of weir removal on river flows in a UK context, similar studies of which are uncommon despite the growing number of reservoirs being decommissioned. Integrated studies that consider the impacts of changing river flows post-weir removal in the wider hydroclimatological-hydroecological process chain are needed to develop a deeper understanding of the impacts of weir removal on freshwater ecosystems.

This study has also provided insight into the impacts of forests on seasonal and low flows, which is important given the UK Government's pledge to plan 30 million trees by 2025. Though further research is needed to establish agreement over the comparative hydrological impact of different tree species, it is recommended that a wider range of nature-based solutions are utilised in order to allow nature to adapt to the full range of hydroecological impacts of climate and land-use change.

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9. Appendices

9.1 Appendix A - data tables

Table 9.1.1 - Mean daily discharge (cumecs) by season and percentage change from the 1990 baseline for the different emissions scenarios (current forest cover, with weir)

	Baseline	RCP2.6		RCP4.5		RCP6.0		RCP8.5	
Annual	2.89	3.05	+5.39%	3.09	+6.70%	3.03	+4.82%	3.11	+7.42%
Winter	5.51	6.19	+12.44%	6.29	+14.22%	6.27	+13.85%	6.59	+19.70%
Spring	1.53	1.47	-3.89%	1.47	-3.92%	1.42	-7.55%	1.44	-5.91%
Summer	1.34	1.05	-21.25%	1.02	-23.43%	0.94	-29.60%	0.87	-34.64%
Autumn	3.25	3.55	+9.09%	3.63	+11.67%	3.58	+9.90%	3.60	+10.68%

Table 9.1.2 - Measures of dispersion and median mean daily discharge for different emissions scenarios (current forest cover, with weir)

	Minimum	Q_1	Median	Q_3	Maximum	IQR	Range
Baseline (cumecs)	0.45	1.12	1.70	4.02	14.61	2.90	14.16
RCP2.6 (cumecs)	0.34	0.97	1.61	4.22	17.38	3.25	17.05
%change	-24.95	-13.25	-5.62	+5.09	+18.98	+12.16	+20.37
RCP4.5 (cumecs)	0.33	0.96	1.59	4.29	17.77	3.33	17.44
%change	-26.34	-14.08	-6.51	+6.83	+21.63	+14.89	+23.14
RCP6.0 (cumecs)	0.29	0.90	1.54	4.25	17.73	3.35	17.44
%change	-35.22	-19.02	-9.32	+5.95	+21.38	+15.56	+23.17
RCP8.5 (cumecs)	0.24	0.88	1.54	4.26	19.04	3.38	18.80
%change	-46.43	-20.99	-9.48	+6.09	+30.36	+16.52	+32.79

Table 9.1.3 - Flow-duration for different emissions scenarios (current forest cover, with weir)

Percentile	Baseline (cumecs)	RCP2.6 (cumecs)	RCP4.5 (cumecs)	RCP6.0 (cumecs)	RCP8.5 (cumecs)
5	8.62	10.01	10.24	10.17	10.69
10	6.85	7.63	7.77	7.75	8.14
15	5.26	5.70	5.78	5.76	6.00
20	4.43	4.78	4.85	4.83	5.00
25	4.02	4.23	4.29	4.26	4.26
30	3.62	3.63	3.70	3.66	3.70
35	3.10	3.17	3.22	3.14	3.08
40	2.49	2.28	2.31	2.30	2.30
45	1.96	1.85	1.83	1.78	1.80
50	1.70	1.61	1.59	1.54	1.55
55	1.55	1.47	1.45	1.40	1.40
60	1.45	1.33	1.33	1.28	1.28
65	1.34	1.23	1.22	1.17	1.16
70	1.24	1.12	1.11	1.05	1.02
75	1.12	0.98	0.97	0.91	0.89
80	1.02	0.86	0.85	0.79	0.77
85	0.92	0.79	0.77	0.71	0.68
90	0.81	0.67	0.66	0.60	0.56
95	0.60	0.50	0.49	0.45	0.43
100	0.45	0.34	0.33	0.29	0.24

Table 9.1.4 - Q95 for different emissions scenarios (current forest cover, with weir)

	Baseline	RCP2.6	RCP4.5	RCP6.0	RCP8.5
Q95 (cumecs)	0.60	0.50	0.49	0.45	0.43
Q95 %change	n/a	-17.07	-18.12	-25.71	-27.84

Table 9.1.5 - Mean daily discharge by season with and without the weir (current forest cover)

		Baseline	RCP2.6	RCP4.5	RCP6.0	RCP8.5
Annual	With weir (cumecs)	2.89	3.05	3.09	3.03	3.11
	Without weir (cumecs)	2.96	3.09	3.13	3.08	3.15
	%change	+2.25	+1.43	+1.42	+1.44	+1.42
Winter	With weir (cumecs)	5.51	6.19	6.29	6.27	6.59
	Without weir (cumecs)	5.67	6.30	6.40	6.38	6.71
	%change	+2.93	+1.75	+1.73	+1.73	+1.72
Spring	With weir (cumecs)	1.53	1.47	1.47	1.42	1.44
	Without weir (cumecs)	1.50	1.43	1.43	1.37	1.40
	%change	-2.26	-3.03	-3.08	-3.33	-3.36
Summer	With weir (cumecs)	1.34	1.05	1.02	0.94	0.87
	Without weir (cumecs)	1.39	1.09	1.06	0.97	0.90
	%change	+4.24	+3.43	+3.30	+3.18	+2.69
Autumn	With weir (cumecs)	3.25	3.55	3.63	3.58	3.60
	Without weir (cumecs)	3.33	3.63	3.71	3.66	3.69
	%change	+2.44	+2.16	+2.20	+2.38	+2.49

Table 9.1.6 - Measures of dispersion and median mean daily discharge with and without weir (current forest cover)

	Minimum	Q_1	Median	Q_3	Maximum	IQR	Range
With weir, Baseline (cumecs)	0.45	1.12	1.70	4.02	14.61	2.90	14.16
Without weir, Baseline (cumecs)	0.36	1.11	1.77	3.84	16.60	2.73	16.24
%change	-19.51	-0.71	+3.71	-4.35	+13.61	-5.75	+14.66
WW, RCP2.6 (cumecs)	0.34	0.968	1.61	4.22	17.38	3.25	17.05
WoW, RCP2.6 (cumecs)	0.24	0.969	1.68	4.05	19.13	3.08	18.89
%change	-29.86	+0.04	+4.54	-4.12	+10.06	-5.36	+10.85
WW, RCP4.5 (cumecs)	0.33	0.96	1.59	4.29	17.77	3.33	17.44
WoW, RCP4.5 (cumecs)	0.23	0.95	1.68	4.11	19.56	3.16	19.34
%change	-31.14	-0.74	+5.32	-4.18	+10.10	-5.18	+10.88
WW, RCP6.0 (cumecs)	0.29	0.90	1.54	4.25	17.73	3.35	17.44
WoW, RCP6.0 (cumecs)	0.19	0.88	1.60	4.06	19.53	3.18	19.34
%change	-33.77	-2.65	+3.88	-4.54	+10.13	-5.06	+10.86
WW, RCP8.5 (cumecs)	0.24	0.88	1.54	4.26	19.04	3.38	18.80
WoW, RCP8.5 (cumecs)	0.18	0.86	1.58	4.18	20.93	3.32	20.75
%change	-24.35	-2.99	+2.83	-1.88	+9.89	-1.59	+10.32

Table 9.1.7 - Flow-duration with and without weir (current forest cover)

Percentile	WW, Baseline (cumecs)	WoW, Baseline (cumecs)	WW, RCP8.5 (cumecs)	WoW, RCP8.5 (cumecs)
5	8.62	9.44	10.69	11.24
10	6.85	7.19	8.14	8.36
15	5.26	5.61	6.00	6.47
20	4.43	4.49	5.00	4.97
25	4.02	3.85	4.26	4.20
30	3.62	3.45	3.70	3.48
35	3.10	2.89	3.08	2.91
40	2.49	2.39	2.30	2.28
45	1.96	2.02	1.80	1.87
50	1.70	1.77	1.55	1.59
55	1.55	1.57	1.40	1.43
60	1.45	1.42	1.28	1.31
65	1.34	1.34	1.16	1.16
70	1.24	1.24	1.02	0.99
75	1.12	1.11	0.89	0.86
80	1.02	0.95	0.77	0.73
85	0.92	0.81	0.68	0.60
90	0.81	0.66	0.56	0.46
95	0.60	0.50	0.43	0.31
100	0.45	0.36	0.24	0.18

Table 9.1.8 - Q95 with and without the weir (current forest cover)

Emissions scenario	Q95 (cumecs) with weir	Q95 (cumecs) without weir	%change
Baseline	0.60	0.50	-16.58
RCP2.6	0.50	0.39	-21.01
RCP4.5	0.49	0.38	-22.82
RCP6.0	0.45	0.33	-26.28
RCP8.5	0.43	0.31	-28.12

Table 9.1.9 - Mean daily discharge by season with current and future forest cover (with weir)

		Baseline	RCP2.6	RCP4.5	RCP6.0	RCP8.5
Annual	Current forest (cumecs)	2.89	3.05	3.09	3.03	3.11
	Future forest (cumecs)	2.88	3.04	3.08	3.02	3.10
	%change	-0.29	-0.32	-0.32	-0.38	-0.37
Winter	Current forest (cumecs)	5.51	6.19	6.29	6.27	6.59
	Future forest (cumecs)	5.51	6.19	6.29	6.27	6.59
	%change	-0.04	-0.03	-0.03	-0.05	-0.05
Spring	Current forest (cumecs)	1.53	1.47	1.47	1.42	1.44
	Future forest (cumecs)	1.53	1.47	1.47	1.41	1.44
	%change	-0.19	-0.20	-0.20	-0.27	-0.27
Summer	Current forest (cumecs)	1.34	1.05	1.02	0.94	0.87
	Future forest (cumecs)	1.32	1.03	1.00	0.92	0.85
	%change	-1.41	-1.89	-1.93	-2.20	-2.30
Autumn	Current forest (cumecs)	3.25	3.55	3.63	3.58	3.60
	Future forest (cumecs)	3.25	3.54	3.62	3.56	3.58
	%change	-0.28	-0.42	-0.41	-0.49	-0.52

Table 9.1.10 - Mean daily discharge by season with current and future forest cover (without weir)

		Baseline	RCP2.6	RCP4.5	RCP6.0	RCP8.5
Annual	Current forest (cumecs)	2.96	3.09	3.13	3.08	3.15
	Future forest (cumecs)	2.95	3.08	3.12	3.06	3.14
	%change	-0.29	-0.33	-0.33	-0.38	-0.37
Winter	Current forest (cumecs)	5.67	6.30	6.40	6.38	6.71
	Future forest (cumecs)	5.67	6.30	6.40	6.38	6.70
	%change	-0.06	-0.06	-0.05	-0.06	-0.05
Spring	Current forest (cumecs)	1.50	1.43	1.43	1.37	1.40
	Future forest (cumecs)	1.50	1.43	1.43	1.37	1.39
	%change	-0.14	-0.17	-0.16	-0.24	-0.22
Summer	Current forest (cumecs)	1.39	1.09	1.06	0.97	0.90
	Future forest (cumecs)	1.37	1.07	1.04	0.95	0.87
	%change	-1.49	-1.98	-2.01	-2.38	-2.46
Autumn	Current forest (cumecs)	3.33	3.63	3.71	3.66	3.69
	Future forest (cumecs)	3.33	3.61	3.70	3.65	3.67
	%change	-0.24	-0.36	-0.37	-0.45	-0.48

Table 9.1.11 - Measures of dispersion and median mean daily discharge for current and future forest cover (with weir)

	Minimum	LQ	Median	UQ	Maximum	IQR	Range
Current forest, Baseline (cumecs)	0.45	1.12	1.70	4.02	14.61	2.90	14.16
Future forest, Baseline (cumecs)	0.43	1.11	1.69	4.00	14.48	2.89	14.05
%change	-2.95	-0.46	-0.96	-0.29	-0.88	-0.22	-0.81
CF, RCP2.6 (cumecs)	0.34	0.97	1.61	4.22	17.38	3.25	17.05
FF, RCP2.6 (cumecs)	0.33	0.97	1.58	4.22	17.22	3.26	16.89
%change	-2.32	-0.08	-1.39	+0.09	-0.94	+0.14	-0.91
CF, RCP4.5 (cumecs)	0.33	0.96	1.59	4.29	17.77	3.33	17.44
FF, RCP4.5 (cumecs)	0.32	0.96	1.57	4.29	17.62	3.34	17.30
%change	-3.12	-0.41	-1.10	+0.09	-0.86	+0.23	-0.81
CF, RCP6.0 (cumecs)	0.29	0.90	1.54	4.25	17.73	3.35	17.44
FF, RCP6.0 (cumecs)	0.26	0.91	1.53	4.22	17.58	3.31	17.32
%change	-10.94	+0.53	-0.60	-0.73	-0.88	-1.08	-0.71
CF, RCP8.5 (cumecs)	0.24	0.88	1.54	4.26	19.04	3.38	18.80
FF, RCP8.5 (cumecs)	0.21	0.87	1.54	4.25	18.88	3.38	18.67
%change	-11.97	-1.05	-0.38	-0.24	-0.84	-0.02	-0.70

Table 9.1.12 - Measures of dispersion and median mean daily discharge for current and future forest cover (without weir)

	Minimum	LQ	Median	UQ	Maximum	IQR	Range
Current forest, Baseline (cumecs)	0.36	1.11	1.77	3.84	16.60	2.73	16.24
Future forest, Baseline (cumecs)	0.35	1.12	1.74	3.90	16.43	2.78	16.08
%change	-3.86	+1.34	-1.54	+1.53	-1.03	+1.61	-0.97
CF, RCP2.6 (cumecs)	0.24	0.97	1.68	4.05	19.13	3.08	18.89
FF, RCP2.6 (cumecs)	0.22	0.97	1.66	4.06	18.93	3.09	18.71
%change	-4.99	-0.12	-1.24	+0.39	-1.04	+0.55	-0.99
CF, RCP4.5 (cumecs)	0.23	0.95	1.68	4.11	19.56	3.16	19.34
FF, RCP4.5 (cumecs)	0.22	0.94	1.65	4.12	19.36	3.17	19.14
%change	-4.55	-0.83	-1.66	+0.15	-1.05	+0.44	-1.01
CF, RCP6.0 (cumecs)	0.19	0.88	1.60	4.06	19.53	3.18	19.34
FF, RCP6.0 (cumecs)	0.18	0.88	1.56	4.10	19.32	3.22	19.14
%change	-7.35	-0.04	-2.87	+0.99	-1.06	+1.28	-1.00
CF, RCP8.5 (cumecs)	0.18	0.86	1.58	4.18	20.93	3.32	20.75
FF, RCP8.5 (cumecs)	0.17	0.84	1.57	4.21	20.71	3.37	20.54
%change	-4.64	-2.24	-0.75	+0.72	-1.03	+1.48	-1.00

Table 9.1.13 - Flow-duration with current and future forest cover (with weir)

Percentile	CF, Baseline (cumecs)	FF, Baseline (cumecs)	CF, RCP8.5 (cumecs)	FF, RCP8.5 (cumecs)
5	8.62	8.60	10.69	10.64
10	6.85	6.90	8.14	8.09
15	5.26	5.34	6.00	6.02
20	4.43	4.42	5.00	5.04
25	4.02	4.01	4.26	4.25
30	3.62	3.63	3.70	3.67
35	3.10	3.12	3.08	3.07
40	2.49	2.46	2.30	2.33
45	1.96	1.96	1.80	1.79
50	1.70	1.69	1.55	1.54
55	1.55	1.53	1.40	1.41
60	1.45	1.43	1.28	1.27
65	1.34	1.32	1.16	1.16
70	1.24	1.25	1.02	1.02
75	1.12	1.11	0.89	0.88
80	1.02	1.01	0.77	0.76
85	0.92	0.92	0.68	0.66
90	0.81	0.81	0.56	0.55
95	0.60	0.60	0.43	0.42
100	0.45	0.43	0.24	0.21

Table 9.1.14 - Flow-duration with current and future forest cover (without weir)

Percentile	CF, Baseline (cumecs)	FF, Baseline (cumecs)	CF, RCP8.5 (cumecs)	FF, RCP8.5 (cumecs)
5	9.44	9.43	11.24	11.38
10	7.19	7.20	8.36	8.38
15	5.61	5.46	6.47	6.26
20	4.49	4.58	4.97	4.93
25	3.85	3.90	4.20	4.21
30	3.45	3.49	3.48	3.57
35	2.89	2.90	2.91	2.87
40	2.39	2.36	2.28	2.28
45	2.02	2.01	1.87	1.88
50	1.77	1.75	1.59	1.57
55	1.57	1.56	1.43	1.43
60	1.42	1.42	1.31	1.29
65	1.34	1.34	1.16	1.18
70	1.24	1.24	0.99	0.98
75	1.11	1.13	0.86	0.85
80	0.95	0.97	0.73	0.71
85	0.81	0.82	0.60	0.58
90	0.66	0.67	0.46	0.45
95	0.50	0.48	0.31	0.30
100	0.36	0.35	0.18	0.17

Table 9.1.15 - Q95 with current and future forest cover (with weir)

Emissions scenario	Q95 (cumecs) current forest	Q95 (cumecs) future forest	%change
Baseline	0.6017	0.6024	+0.12
RCP2.6	0.50	0.49	-1.11
RCP4.5	0.493	0.488	-0.92
RCP6.0	0.45	0.43	-3.39
RCP8.5	0.43	0.42	-2.88

Table 9.1.16 - Q95 with current and future forest cover (without weir)

Emissions scenario	Q95 (cumecs) current forest	Q95 (cumecs) future forest	%change
Baseline	0.50	0.48	-4.76
RCP2.6	0.39	0.38	-3.96
RCP4.5	0.38	0.37	-2.81
RCP6.0	0.33	0.32	-3.57
RCP8.5	0.31	0.30	-3.66

9.2 Appendix B - hydrographs

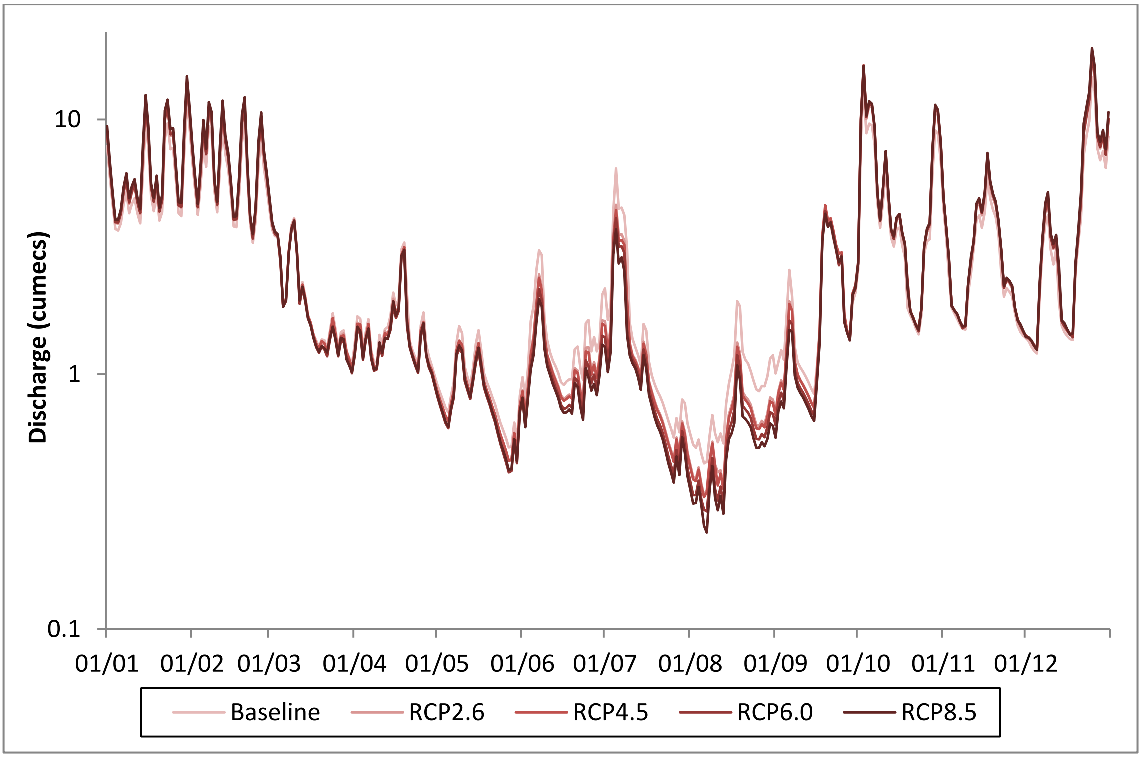

Figure 9.2.1 - Hydrograph showing discharge under the emission scenarios (current forest cover, with weir)

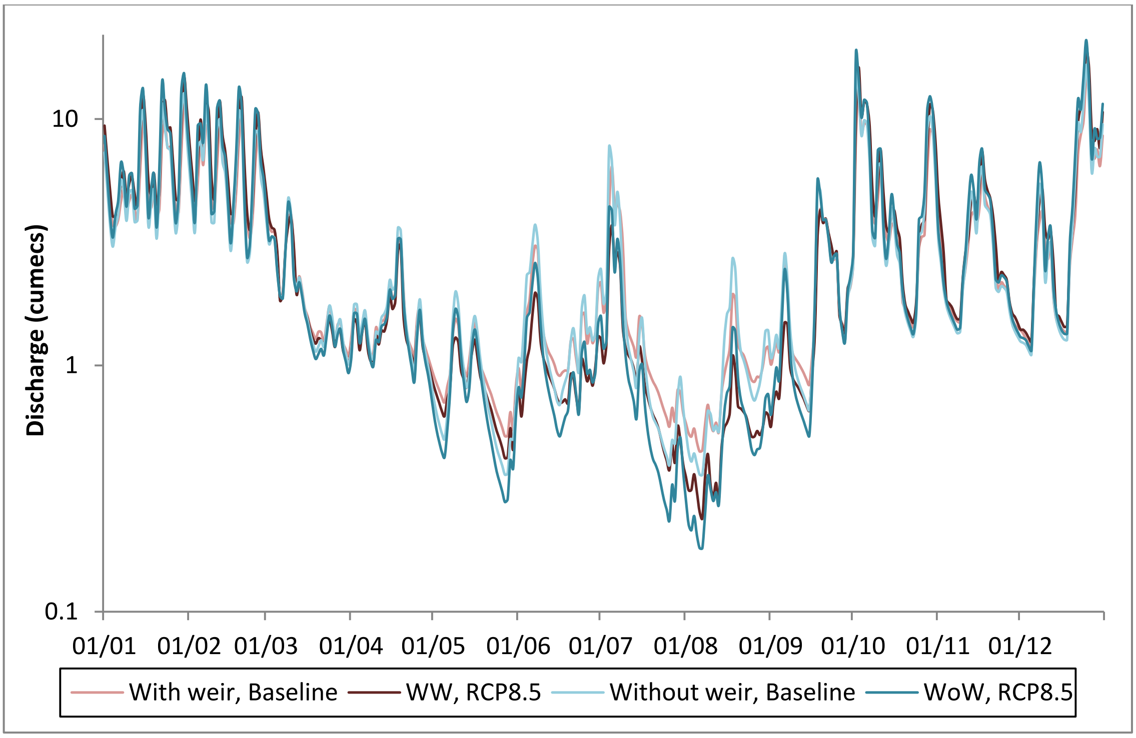

Figure 9.2.2 - Hydrograph showing discharge with and without weir (current forest cover)

N.B. Only the discharge for baseline emissions and RCP8.5 are included to indicate the range of flows across the emissions scenarios and to avoid overloading the graph.

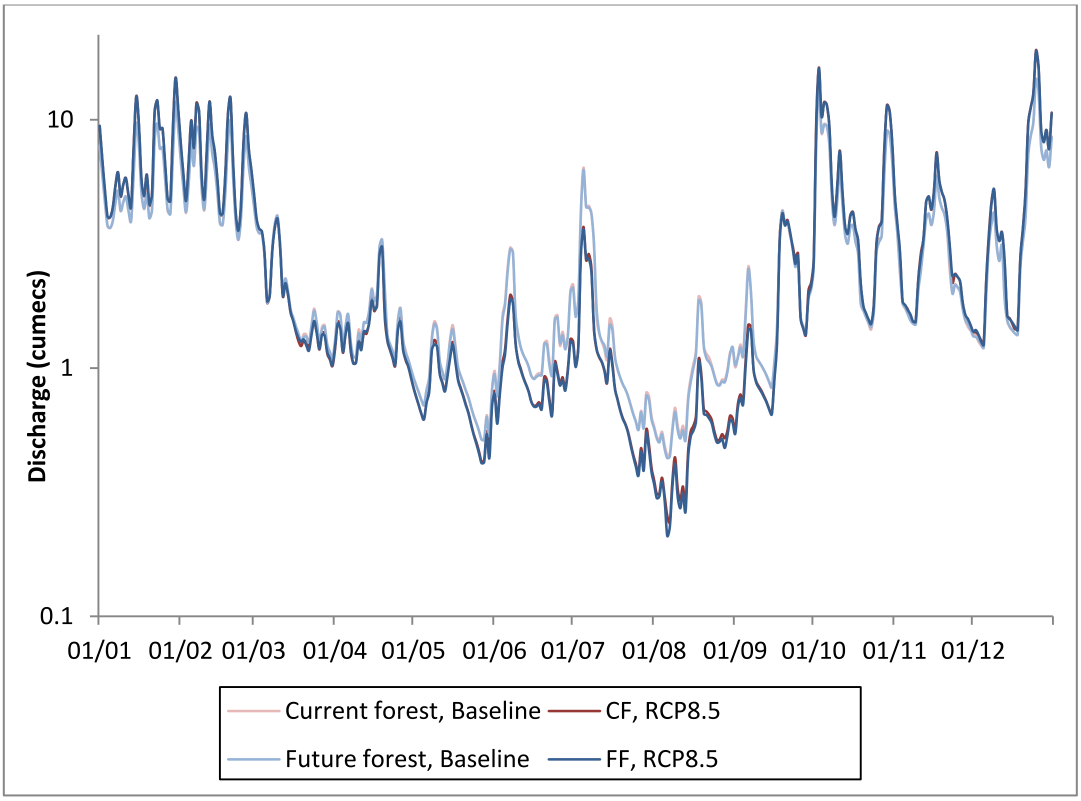

Figure 9.2.3 - Hydrograph showing discharge with current and future forest cover (with weir)

N.B. Only the discharge for baseline emissions and RCP8.5 are included to indicate the range of flows across the emissions scenarios and to avoid overloading the graph.

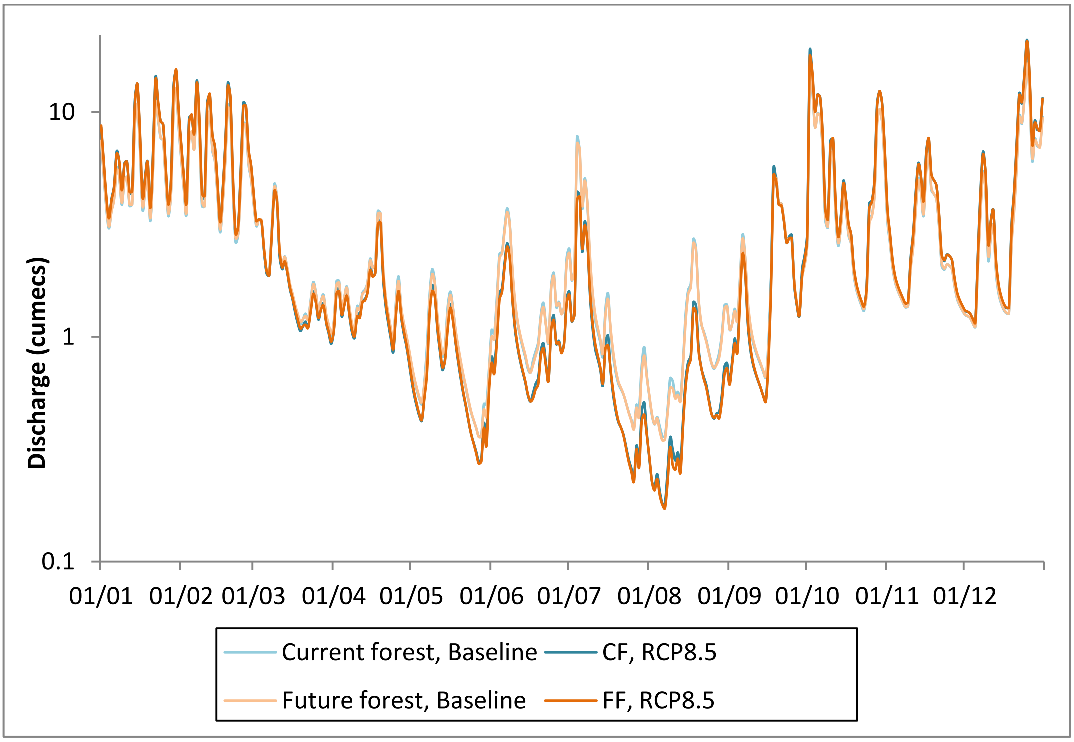

Figure 9.2.4 - Hydrograph showing discharge with current and future forest cover (without weir)

N.B. Only the discharge for baseline emissions and RCP8.5 are included to indicate the range of flows across the emissions scenarios and to avoid overloading the graph.