Reservoir compensation releases and the ecology of the River Derwent, Northumberland.

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Abstract

River regulation is commonplace in England and much of the UK. Regulation for the purposes of public water supply causes flows downstream of a reservoir to be attenuated and the flow regime of the channel to be altered. The impact of channel impoundment on a small, upland UK river, has been assessed and methods for mitigation of ecological impacts explored. The method utilised a unique macroinvertebrate data set for pre- and post-impoundment periods to quantify the impact of Derwent Reservoir and the steady, continuous compensation release into the River Derwent, Northumberland. Impacts on the hydrological regime were also investigated and links drawn between changes in flow regime and changes in macroinvertebrate richness and diversity as a result of impoundment. In response to the claim that the impoundment has caused a change in flow regime and had deleterious effects on fish and macroinvertebrates, a compensation redesign tool (CRAB: Compensation Release Assessment at the Broad scale) was employed to design new compensation release regimes from the reservoir which account for the seasonal flow requirements of a number of key fish species. The impact of impoundment on the current flow regime was modelled and the impacts of predicted new regimes were predicted, using a 1D hydrodynamic model (HEC-RAS), as part of a modelling process known as CRAM (Compensation Release Assessment at the Meso-scale). Depth and velocity were the foci of the analysis as they are the two habitat requirements most well documented for the fish species in question, they could be modelled using HEC-RAS and they can act as surrogates for other habitat parameters such as temperature and substrate. The suitability of the depth and velocity combinations predicted using the HEC-RAS model were assessed using fuzzy rule-based modelling, which allowed the habitat quality of a given parameter combination to be quantified.

Based on the results of the investigation it was concluded that there has been a change in flow regime and in ecological community structure since impoundment. The flow regime of the River Derwent has become less flashy with fewer extreme events, while macroinvertebrate richness and diversity have increased. The new flow regimes that were designed by CRAB, based on the depth and velocity requirements of brown trout, grayling and Atlantic salmon were predicted through CRAM to have minimal benefits for the fish populations of the River Derwent and it was concluded that no changes to flow regime should be made based solely on the assessment of habitat for fish. Impacts for the macroinvertebrate communities must also be considered as well as the impacts on other aspects of fish habitat including temperature, substrate and cover. A more detailed, micro-scale investigation into the effects of changing flow regime would be required to warrant a change in compensation release regime from Derwent Reservoir.

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Contents

List of Figures

Chapter Six

Chapter Seven

Chapter Eight

List of Tables

Chapter Six

Chapter Seven

Chapter Eight

Chapter One

Introduction

1.1 Aim and research questions

Aim

To investigate the effects of channel impoundment on hydrology and ecology and design a new way of managing compensation releases to improve habitat quality and availability within regulated rivers

Research Questions

1. What are the long-term effects of channel impoundment and river regulation (for water supply purposes) on flow regime and ecological populations of a channel?

2. Can a compensation release regime be redesigned and optimised to meet the habitat requirements of a number of species and how should its components be introduced?

3. What are the possible impacts of a proposed new flow regime on habitat: at what scale should analysis be conducted and using which methods?

1.2 Context

River regulation, in its many forms, is now commonplace across the globe. River regulation can occur for a number of purposes, the most common being flood defence, water supply and power generation. Dams and reservoirs are regular features on rivers and are often used to control the flow in a river to deliver the above mentioned services. However, this regulation of flow can lead to 'unnatural' hydrograph shapes. For example, impoundment for the purpose of hydro-electric power (HEP) may cause very low flows and very sudden high flows over short timescales, whereas impoundment for water supply and flood defence purposes dampens all features of the hydrograph to produce a steady, less flashy flow. Amidst the implementation of the EU Water Framework Directive (WFD) and in an aim to achieve 'good ecological status' for most of the UK's rivers by 2015, assessing the impact of river regulation on ecology, a long-standing field of research, is becoming increasingly important. In the UK, this is partially due to the requirements of the statutory body responsible for the health of riverine biology (the Environment Agency) to submit River Basin Management Plans to the Secretary of State for Environment, Food and Rural Affairs and the Welsh Assembly by the end of 2009. More widely, the increase in research interest has come as a result of climate change. With water resources becoming more scarce in many parts of the world due to reduced rainfall and/or increased temperatures (as well as increasing demand as living habits change and populations grow), the strategy of river impoundment for water supply purposes is frequently adopted. When such a change is applied to a channel, the implications on the available resource for the ecology and the water users must be considered. Furthermore, for existing impounded rivers, the compensation release volumes set at the time of impoundment are often based on criteria which are no longer relevant (e.g. based on needs of abstractors or mill owners downstream, Maddock *et al.,* 2001). For example compensation release regimes designed to meet the needs of industrial river uses may now be higher than necessary. If this is the case, the focus of the river management should be updated from industrial purposes to delivering environmental and water resource benefits in accordance with more recent river management initiatives such as the EU Water Framework Directive.

Altering channel baseflow can affect hydrology, morphology and ecology, including:

- Depth, velocity and wetted perimeter altered (Crisp, 1995)
- Reduced variability of flow (necessary for healthy biological populations (Richter *et al*., 1997))
- Loss of extreme low and high flows
- Second-order impacts: aggradation, degradation, armouring, reduced bed stability

All of these changes have implications for habitat availability, and therefore the fish and macroinvertebrate population dynamics of a channel. Impacts of channel impoundment can occur on a number of scales, ranging from adjustments of channel width (Petts, 1980; Gurnell, 2005) to changes in microtopography within pools and riffles.

The investigation will address a number of issues which are currently under-researched. Firstly, there are few studies of the long-term impacts of regulation on ecology, due to difficulty of obtaining long-term ecological data. This project, therefore, will assess the effects of river regulation on the hydrology and subsequently the ecology of a river through comparison of ecological populations before and after impoundment. It has been hypothesised that the effects of impoundment may be mitigated through management of compensation release regimes (e.g. Maddock *et al.,* 2001) but the optimisation of one new flow regime for the benefit of multiple species with different flow requirements has been rarely addressed. Thus, the combined impacts of a number of flow regimes will be analysed for the key species within the river. The introduction of spate flows may be an efficient method of improving the substrate habitat conditions within a channel as well as providing the variability in flows required by some species for the initiation of activities such as spawning and migration. However, there is little information available on how to introduce such flows and what impact they may have on the channel and its ecology. This will be considered as part of the investigation.

1.3 Explanation of research questions

Q1 What are the long-term effects of channel impoundment and river regulation (for water supply purposes) on flow regime and ecological populations of a channel?

By comparing pre- and post-impoundment data on hydrology and ecology, a comparison can be made between the two time periods and the effects of impoundment inferred. An idea of expected impacts will inform remediation decisions and allow subsequent methodologies and management strategies (e.g. Maddock *et al.,* 2001) to be focused on the aspects of flow and ecology that are most significantly affected by impoundment. This question will be addressed in Chapters Three (hydrology) and Four (ecology).

Q2 Can a compensation release regime be redesigned and optimised to meet the habitat requirements of a number of species and how should its components be introduced?

In order to provide improved habitat (area and quality) for fish and macroinvertebrates, it is essential to know what conditions they require and therefore how to change flows in order to achieve those conditions. The outcome of Q1 will determine the importance of flow variability for ecological populations and will quantify the effects of regulation on flow variability. Q2 aims to account for change in flow variability by designing compensation releases that might deliver the variability of flows required, as identified in question one. The answers to Q2 will provide a range of new compensation release regimes that may deliver natural features of the hydrograph, to be tested through Q3 and an 'optimum' regime suitable for a number of species will be developed. Q2 will be addressed in Chapter Five.

Q3 What are the possible impacts of a proposed new flow regime on habitat: at what scale should analysis be conducted and using which methods?

Q3 aims to predict the impact of prescribed flow regimes developed in answering Q2. The new understanding of the long-term effects of impoundment (from Q1) will allow an evaluation of the new compensation flow designs by attempting to assess the impacts of a number of different combinations and timings of compensation releases throughout a year, in terms of impact to flow depth and velocity (important habitat controls for ecology). Assessment of depth and velocity combinations will provide an indication of the habitat suitability under a range of conditions. The applicability of hydrodynamic and fuzzy modelling methods for determining habitat suitability will be considered. Scale will be considered by comparing the prescribed flows (based on a set of depth and velocity requirements) with the depths and velocities predicted in the broad scale assessment. Results from the 1D, river-scale assessment of this study will be compared to the findings of a similar study conducted at the 2D, reach scale (Mould, 2006). Chapters Six and Seven will address this research question.

1.4 Case study site and justification

Numerous investigations have been conducted on the effect of channel impoundment on flow and morphology (Petts and Gurnell, 2005). However, identification of whether a population is flow limited or habitat limited and whether the impoundment is the controlling factor in changes experienced cannot be inferred without considering the specific population and the specific channel. Therefore, the focus of this study is to investigate the impacts of the impoundment of the river Derwent, North East England on its hydrology and ecology and develop the new methodology for compensation flow redesign on this river.

The River Derwent lends itself to this study for a number of reasons: (i) there are abundant baseline data for both hydrology and ecology from before and after reservoir construction; (ii) the river supports fish and macroinvertebrate populations which are thought to have been affected by the changes in flow regime; and (iii) without any significant tributaries, the river is sensitive to the impact of releases from Derwent Reservoir. It is expected that the effects of impoundment will be most strongly felt near to the point of release as this will be the most artificial flow, and that through accretion of natural flows from runoff and tributaries, a more natural regime will be experienced in the downstream reaches. Therefore, the river will be considered from the reservoir release, to the most downstream gauged point. The area of focused study lies between Eddy’s Bridge and Rowlands Gill because these two sites have gauged flow data and macroinvertebrate data exist for areas between these two sites. The existence of flow and macroinvertebrate data from both before and after impoundment will allow a unique analysis of the long term effects of regulation and will also be used to test the efficiency of the modelled flows in the hydrodynamic modelling.

The River Derwent is important within the region for ecological, water resources and recreational purposes. It is therefore considered as an important site for investigation. Further detail about the river and catchment are included in Chapter Two.

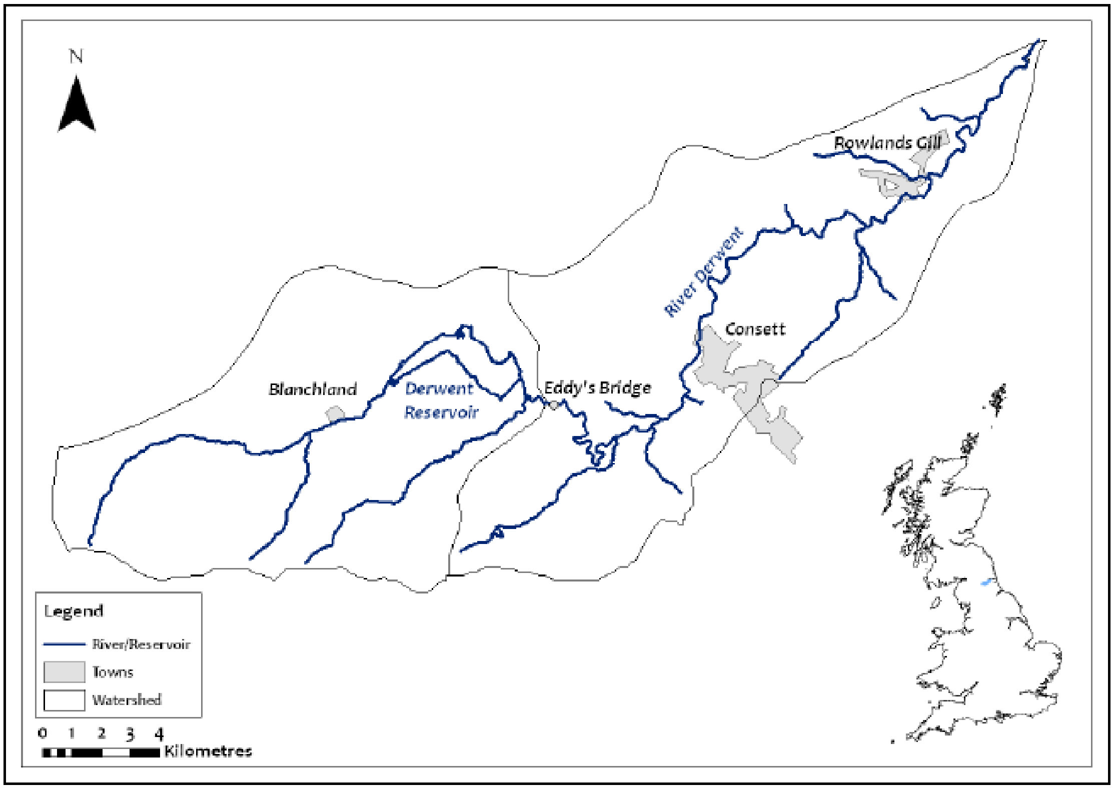

Figure 1.1 Location of the Derwent catchment in a national context

1.5 Thesis structure

This thesis is structured around the investigation of current and hypothetical impacts of regulation. Chapter Two introduces the study catchment. Chapters Three and Four investigate the impacts of regulation on flow and ecology, respectively. Chapter Five develops potential new flow regimes and Chapters Six and Seven attempt to assess the impact of the new regimes through hydrodynamic and fuzzy modelling approaches (respectively). Findings and suggestions are summarised in Chapter Eight.

Chapter Two

Study Site Characteristics

2.1. Catchment characteristics

This Chapter introduces the hydrological, geomorphological and ecological characteristics of the Derwent catchment and develops the justifications outlined in Chapter One as to the choice of this area for study. Four main characteristics will be addressed: hydrology (2.1.1); geomorphology (2.1.2), ecology (2.1.3) and history (2.1.4). Section 2.2 will address the management of the River Derwent. Figure 1.1 shows the catchment's location in a UK context and the position of the reservoir within the river basin. As this study is focused upon the effects of the impoundment on the channel, the length of river downstream of the dam will be the focus of this study site section.

2.1.1. Hydrology

The Derwent catchment covers an area of 266 km^2 and water fed to the reservoir drains from an area of 110 km^2. The catchment contains a steep, V-shaped valley as far as Shotley Bridge, through which the 50 km long River Derwent flows. The Derwent Catchment receives on average 812 mm of rainfall a year and has a long-term low flow value (Q95) of 70.8 Ml/d at the most downstream gauged site of Rowlands Gill (flow record runs from 1962 to present). The river Derwent is currently characterised by stable and less flashy flows as a result of regulation (from the dam, built in 1966). However, there are occasions when large flood peaks are experienced. Discharge of the River Derwent is gauged in two locations by EA gauging stations (Eddy's Bridge and Rowlands Gill, see Figure 2.1) and an additional river level monitoring station at Blackhall Mill.

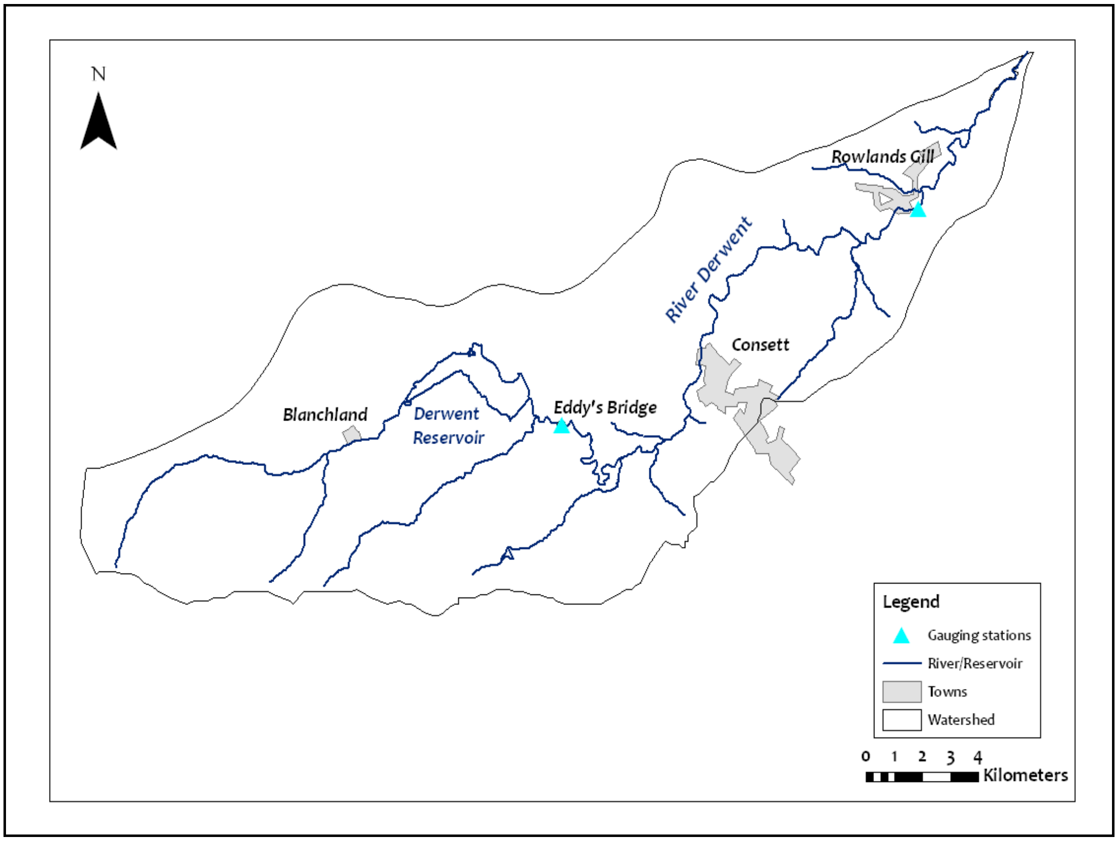

Figure 2.1 Flow gauging station locations in the Derwent catchment

Derwent Reservoir, constructed between 1960 and 1966, is the second largest of Northumbrian Water Limited's (NWL) reservoirs and is located 14.5 km from the source of the River Derwent, near the small town of Blanchland (Figure 2.1). The 3.5 km long reservoir has a surface area of 4 km^2 and a maximum capacity of 50,000 Ml. Water drains to the reservoir from the upper River Derwent and its tributaries, and is subsequently piped 3.5 kms to NWL's water treatment plant at Mosswood where it is filtered and gravitated 43 km to the town of Washington for distribution. A compensation release from the reservoir augments flow to the impounded River Derwent, at a rate of 8706 Ml year, equating to 25 Ml/d between April and September and 22.7 Ml/d between October and March. There is a further 827 Ml allocated to spate releases in any one year. Currently, there is little variation in the release rate across the year (a range of only 2.3 Ml/d for the whole year). On occasion, when the reservoir is at capacity, overtopping of the dam wall occurs, increasing the amount of water discharged to the River Derwent for a short period. This provides a form of freshet (a short period of high discharge). The earth dam is approximately 1 km long and 36 m high. For the purposes of this thesis, the volumetric unit of Ml/d (megalitres per day) will be used because this is the unit frequently used when discussing reservoir releases and is used in the first piece of modelling software applied in the investigation (N.B. 86.4 Ml/d = 1 $m^3\ s^{-1}$).

There are seven weirs along the course of the River Derwent, between the reservoir and the confluence with the River Tyne, which all have an effect on flow (e.g. depth, velocity) and water quality (e.g. aeration). Weirs reduce the velocity of flow upstream, increase the depth and may cause aggradation to occur as the water loses energy. Overtopping of the dam wall at Derwent Reservoir occurs occasionally and may provide a form of freshet (short period of high discharge) to the river. However, this happens infrequently (EA Tyne CAMS, 2005) and cannot be relied upon to ensure optimum habitat for riverine species.

Groundwater in the catchment is limited as the aquifers in the area are minor. However, where there are groundwater sources, connectivity between the aquifer and the river is good. It can therefore be assumed that any abstraction from a groundwater source would have a similar effect on the River Derwent as an abstraction from the main river would (EA Tyne CAMS, 2005).

The hydrological characteristics of the river will be discussed further in Chapter Three.

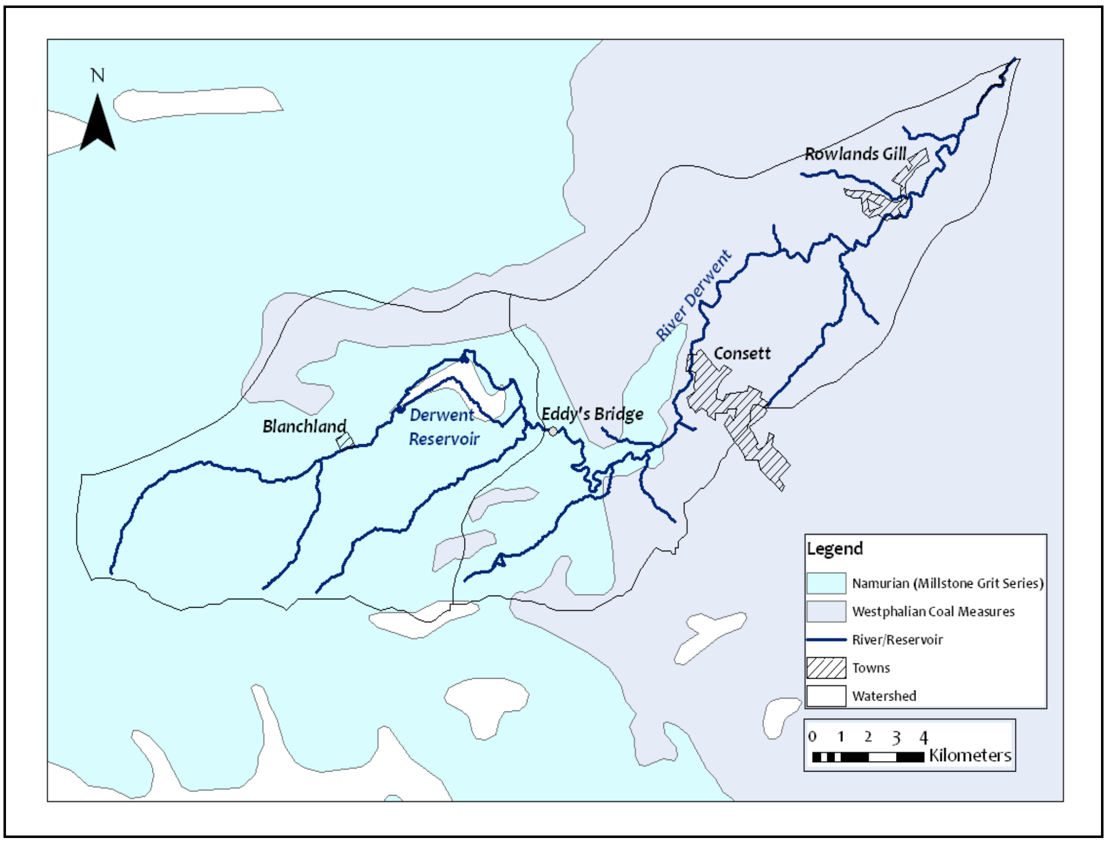

Figure 2.2 Geology of the Derwent catchment

2.1.2. Geomorphology

Geomorphology is an important control on flow. Channel morphology affects depth of flow and wetted perimeter, through controls on channel width and shape. On a more localised scale, morphology and flow interact through processes of aggradation, degradation and sediment transport. Type of sediment transport is largely controlled by sediment type and can affect the sediment size composition of a channel. For example, selective transport will cause a fining of sediment with distance downstream. Sediment supply is largely affected by impoundment with the dam acting as a sediment transport discontinuity for both coarse and fine sediment. Flow velocity affects aggradation and degradation patterns and therefore sediment composition of a reach.

Channel morphology is itself controlled by factors such as geology. The geology of the Derwent catchment (Figure 2.2) consists primarily of a Stainmore formation, of Namurian age (Millstone Grit Series), in the upper reaches, and Westphalian coal measures, downstream of Allensford. As a result of the geological composition of the Derwent catchment, the River Derwent has a varying form ranging from bedrock reaches with vertical rock walls to sandy-bed areas with wide floodplains. Between the reservoir outlet and Allensford, the gravel bed river follows a tightly meandering course and is dominated by pool-riffle sequences. Downstream of Allensford, meanders become wider and further apart, and there is an increase in the number of gravel bars, while pool-riffle sequences remain frequent. Towards the confluence with the River Tyne, the Derwent becomes wider, deeper, straighter and consists of much finer sediment where cobbles and pebbles are still present, but is dominated by sand. For much of its length, the Derwent Valley is narrow and confined with only a narrow floodplain and many reaches with large boulder/bedrock sections. This characteristic will affect flow and channel form by limiting lateral migration potential of the river. It will also decrease the sensitivity of the channel to the sediment transport discontinuity caused by the dam because there is less potential for the deposition of sediment on the floodplain. The floodplain widens at around Blackhall Mill, but there are still frequent reaches which are confined by steep valley walls on at least one side of the river. Figure 2.3 shows how channel form changes with distance downstream.

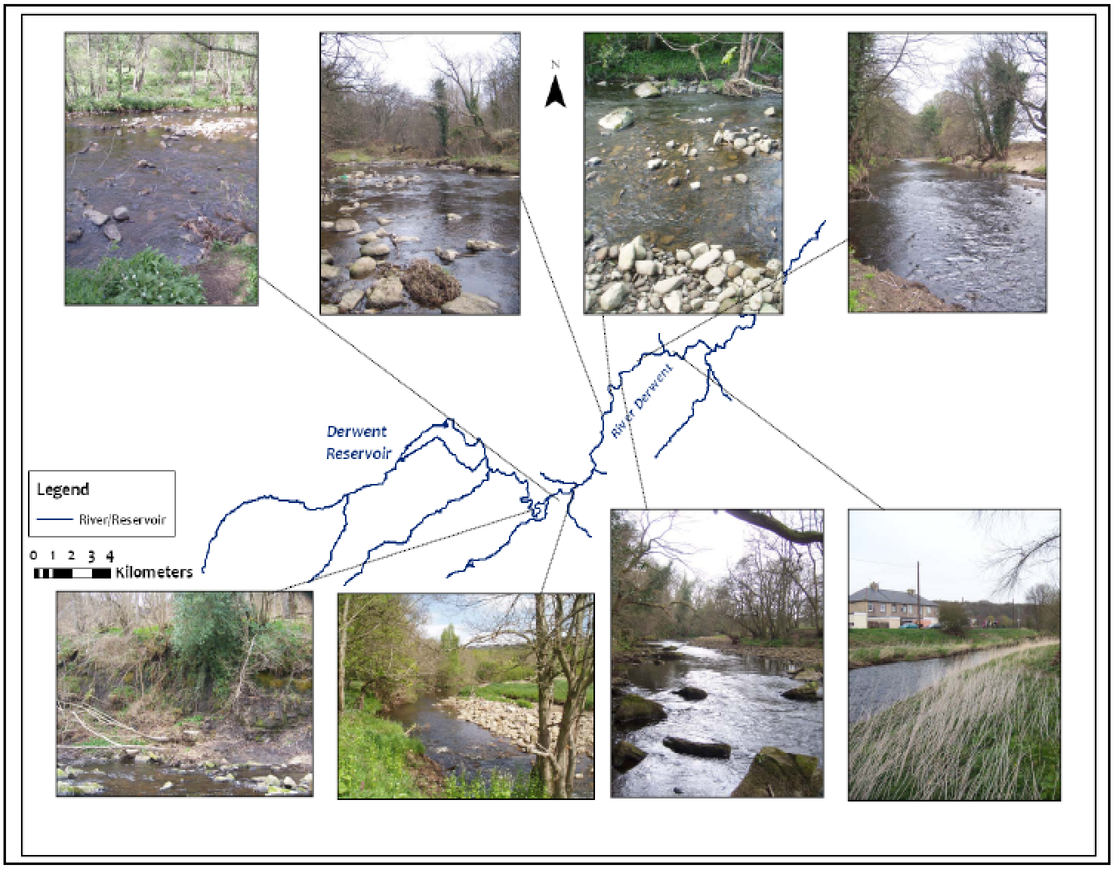

Figure 2.3 Sediment size distribution and geomorphological characteristics of the River Derwent

Channel width ranges from 50 m at the confluence with the River Tyne, to around 17 m for much of the middle reaches between Rowlands Gill and Allensford, to around 5 m at Eddy's Bridge, just downstream of the reservoir. The mean river width, used for broad scale modelling purposes in this research and discussed in Section 5.4, is 11.5 m. For the more detailed hydrodynamic modelling (HEC-RAS, Chapter Six), the morphology of the river will be represented using a series of cross-sections. The numerous bridges and weirs on the River Derwent have an impact on channel morphology and flow. Water backs up behind weirs causing a localised increase in width and depth. Width also increases downstream of the weir. Width of flow is important as it determines the wetted perimeter, effectively the area of usable habitat available for riverine ecology.

2.1.3. Ecology

The River Derwent supports a number of different flora and fauna species, including fish, macroinvertebrates and macrophytes. The main fish populations in the river are brown trout and grayling while some of the most common macroinvertebrates at present include Baetidae, Heptageniidae, Ephemerellidae, Leuctridae, Elmidae, Hydropyschidae and Chironomidae (according to Environment Agency macroinvertebrate data, collected between 1995 and 2007).

Currently, salmon and sea trout are unable to negotiate the major weirs of the river and therefore are mostly absent from the river. However, the Environment Agency is in consultation with local funders to discuss the possibility of installing fish passes on the four main obstructions. An indication from this project on whether the river could support salmon and sea trout (in terms of habitat) under a new flow regime would be very informative to this proposal.

In the 2005 Tyne Catchment Abstraction Management strategy, the River Derwent is classed as having Environment Agency general Quality Assessment scores (for ecological quality) of good and very good, for the whole river.

Habitat for fish and macroinvertebrates is affected by flow, morphology and sediment size distribution. Large pebbles and cobbles provide a matrix for fish to deposit eggs, channel shape determines wetted perimeter which defines usable habitat area and flow controls depth (and therefore wetted perimeter), and velocity (which triggers life events such as spawning and migration). Each of these factors will be investigated in detail later in the study. Temperature, also important for events in a fish's life cycle, is affected by channel and environmental conditions. There is dense riparian tree cover for much of the length of the river between Derwent Reservoir and Rowlands Gill (locations in Figure 2.1). Tree cover provides a canopy, important in providing shade for fish species such as brown trout (Raleigh *et al.,*1986).

There are a number of ecological designations within the Derwent catchment, each one aimed at preserving a particular habitat or ecological feature, which are summarised in Table 2.1.

Designation	Area	Details
SPA	Covering large areas of the catchment to the south of Derwent Reservoir and as far east as Crooked Oak	**Special protection area –** part of the Natura 2000 network designated under the Eurpoean Habitats and Birds Directive. Provides protection to birds and their nests, eggs and habitats.
SAC	Covering large areas of the catchment to the south of Derwent Reservoir and as far east as Crooked Oak	**Special area of conservation–** part of the Natura 2000 network designated under the Eurpoean Habitats and Birds Directive. Contributes to the maintenance and restoration of habitats and species.
SSSI	Covering small parts of the catchment near Crooked Oak and further downstream near Rowlands Gill	**Site of special scientific interest –** a prerequisite to designation as SAC or SPA. This is a national level designation overseen by Natural England. A SSSI can be an area of land that is of special interest by reason of its flora, fauna or geological or physiographical features (as defined by Natural England).
SNCI	Covering small parts of the catchment near Crooked Oak, muggleswick and Ebchester	**Site of Nature Conservation Interest/ Importance –** a county scale designation. Overseen by the County Wildlife Trust and provides protection to wildlife and habitats.
AONB	Covering all of the catchment upstream of Allensford	**Area of outstanding natural beauty** – managed by local authorities, organisations and community groups, the aim of an AONB is to draw special attention to an area because of its flora, fauna, historical/cultural associations or scenic views.

Table 2.1 Designations within the Derwent catchment.

The ecology of the River Derwent will be examined further in Chapter Four.

2.1.4. Land Use and History

Land use in the Derwent Catchment is primarily arable, with large areas of grassland and moorland. There are also some wooded areas and a number of small urban settlements (EA Tyne CAMS, 2005). The main use of the river and reservoir today is for recreation (including angling) and a few small industries are located along the banks. The primary abstractor of water is Northumbrian Water Limited, associated with public water supply. Small, local abstractions are made for domestic and agricultural uses. In the past, there were a number of mill wheels which used water from the river. Consett Iron Works was the major industry in the valley (EA Tyne CAMS, 2005). Regulation of the channel has affected baseflows, creating a more stable, less flashy regime. However, because of the proximity of the main tributary, Burnhope Burn, to the outlet, severe flood and drought events are still easily detected within the hydrograph.

Water quality is affected by a number of inputs, including those from sewage treatment works, water treatment plants (e.g. Mosswood) and industry. Inputs to the river are monitored and controlled by the Environment Agency to ensure minimum impact on the quality of the river. In the 2005 Tyne CAMS, the River Derwent was assessed as having generally good water quality.

2.2. River regulation and catchment management

The ecology, quality and water resources of the Derwent catchment are managed in a number of ways by the Environment Agency and in conjunction with Northumbrian Water Limited, the owners and operators of Derwent Reservoir. Table 2.2 summarises the management strategies for the river and what they aim to do.

Strategy	Purpose
Derwent Water Order (1957)	Designed to manage the water impounded by and released from the River Derwent by NWL. Regulation of the River Derwent began in 1966. The compensation flow rate is set at a total rate of 9533 Ml/y (23.85 Ml/d and a spate allowance of 827 Ml in one year). It is permissible, by law, to vary the volumes of water released, provided that the annual minimum volume is released.
CAMS (Catchment Abstraction Management Strategy, 2005)	Used to assess the current stresses on water resources and develop sustainable licensing strategies to balance needs of the ecology water users. Revised release regimes from the reservoir must be able to meet the abstraction requirements of current licence holders. CAMS also governs how Northumbrian Water Ltd can abstract water from the River Derwent and Derwent Reservoir, in line with the 1957 Derwent Water Order.
CFMP (Catchment Flood Management Plan)	Outlines areas at risk from flooding, types of flood risk and management practices in place or proposed to prevent and/or accommodate flooding in the Derwent Catchment. Any increases in flow under new flow regimes must not increase the chances of flooding, as outlined in the CFMP.
Tyne Salmon Action Plan	Includes area of River Derwent. There are no salmon currently in the River Derwent, however, should salmon be able to access the Derwent, future Salmon Action Plans would include assessment of stocks and management practices for salmon in the River Derwent.
Summary of significant water management issues (2007): Northumbria River basin District (part of the EU WFD)	Highlights issues such as physical modification and minewater pollution. Definition of these issues is used to set environmental objectives for each water body within each EU WFD River Basin District. Under the EU WFD these objectives must be met unless significant reason can be given to justify their failure. The aim of this is to ensure that water bodies are brought to a good standard across Europe over the next 10-20 years.
NWL Water Resources Management Plan	Outlines how NWL intend to balance supply and demand over the next 25 years
Asset Management Plans (AMPs)	Outline the management of infrastructure and other assets maintained by water companies

Table 2.2 Management strategies for the River Derwent

2.3 Summary

The Derwent catchment is a relatively small one, but highly regulated through river impoundment and weirs. There is a history of channel alteration for industry in the catchment which has more recently given way to use as a recreational river and the water abstracted is primarily used for supply within the region. The River Derwent supports a number of ecological groups including fish and macroinvertebrates although some species (such as Atlantic salmon) cannot populate the river due to anthropogenic boundaries. Chapters Three and Four will aim to assess the impact that flow regulation has had on the flow and the ecology within the Derwent catchment, and subsequent Chapters will aim to suggest amelioration measures to these changes, should it be concluded that they are necessary.

Chapter Three

Assessing the impacts of impoundment on downstream hydrology

3.1 Introduction

This Chapter aims to assess the impacts of channel impoundment and river regulation on a number of rivers through the review of published literature, and more specifically on the River Derwent through analysis of hydrological data which exist from the pre- and post-impoundment periods. An understanding of the implications of flow regulation will inform subsequent decisions on methods of habitat improvement.

The first part of this Chapter (Section 3.2) will outline the effects of impoundment on flow, morphology and temperature. The review of literature seeks to assess the current view of such impacts and assess which aspects require further investigation. It also serves to place the current study in context and for this reason, the main focus is upon the effects of rivers dammed for water supply purposes, where appropriate. Drawing on the research needs identified in the literature, research questions and methodologies have been designed and detailed in Section 3.3. Section 3.4 presents the results of data analysis and the implications of these results are discussed in Section 3.5. Section 3.6 summarises the findings of the Chapter and outlines the direction the project will take as a result of the findings in this Chapter.

3.2 Overview of current literature

3.2.1 Introduction

Dams now exist on many rivers across the UK and throughout the world. Reasons for such installations include flood management, water resource management and energy production (e.g. hydro-electric energy). When a river is impounded, flow in the channel is interrupted and the river dries up. To mitigate this, operators generally allow a certain amount of water to be released through pipes or valves in the dam wall. This is known as a compensation flow and serves to provide a baseflow to the channel downstream of a reservoir. When a reservoir reaches its full capacity, water will overtop the dam wall and cause an increased level of flow. As a result of the nature of flow released from the reservoir, hydrological regimes are altered. In the case of the River Derwent, flow is released at a relatively constant rate for the whole year (which varies by a very small amount between summer and winter). The manipulation of discharge impacts upon a number of aspects, including flow dynamics (Petts and Pratts, 1983; Armitage, 1995; Isik *et al*., 2008; Jorde *et al*., 2008), channel morphology (Petts, 1980; Gilvear *et al*., 2002; Jacobson and Galat, 2006; Brown and Pasternack, 2008), and other ecological aspects of the river environment such as temperature (Petts, 1984; Crisp, 1995; Jackson *et al*., 2007).

3.2.2 Flow

Dams and river impoundment affect hydrology (Isik *et al.*, 2008). Regulation of flow is known to affect a number of hydrological aspects: depth (Armitage, 1995; Crisp *et al.*, 1995; Jorde *et al.*, 2008), velocity (Crisp *et al.,* 1995), and wetted area (Armitage, 1995; Crisp, 1995; Jorde *et al.*, 2008). Impoundment can also affect the shape of the hydrograph by dampening extremes such as spring floods and summer low flows (Petts and Pratts, 1983; Ibañez *et al.,* 1995; Isik *et al.,* 2008). Flood frequency/magnitude (Bradley and Smith, 1984; Brown and Pasternack, 2008; Isik *et al.,* 2008; Jorde *et al.,* 2008), overall quantity and timing of flow (Petts, 1980, Jorde *et al.,* 2008), rate of change of flow (Brown and Pasternack, 2008) and increased and decreased summer baseflows are known to be affected by damming in some rivers (e.g. the The Ebro River, Spain, Batalla *et al.,* 2004). Such alterations to the hydrograph have been documented by a number of authors. For example, Ibañez *et al.,* (1995) found that on regulated rivers such as the Ebro in Spain, volume of water released from reservoirs in dry years is often less than the discharge measured in periods before impoundment (i.e. there is a reduction in mean annual discharge and fewer flood events). Batalla *et al*., (2004) supported this by noting Dams on the Ebro river caused a reduction in flood magnitude with Q_2 and Q_{10} values reduced by up to 30%. Crisp (1977) found that regulation of the River Tees, UK, has smoothed fluctuations in flow and eliminated the river's characteristic very low and very high discharges. Bradley and Smith (1984), studying the Milk River, USA, found that flows in excess of 100 $m^3\ s^{-1}$ occurred once every 1.8 years before impoundment and once every 4.5 years (on average) after impoundment.

3.2.3 Morphology

There are a range of associations between process changes and morphological changes below dams. Channel morphology, unlike flow or water quality, has a slow response time to river impoundment (Petts 1980, Petts and Pratts, 1983,). Once the change in flow has been accommodated a new, but steady state, may be established. For example, Petts (1984) writes of the new stability found after channel degradation and armouring. Jorde *et al.,* (2008) note that downstream of Libby Dam there is an absence of sediment replenishment from upstream sources. However, the channel is also less able to carry coarse sediment, leading to stability of the channel. Channel response to dams is also controlled by input of sediment from unregulated tributaries (Petts, 1980).

Altered river flow regimes affect morphology (Gilvear *et al.,* 2002; Jacobson and Galat, 2006; Brown and Pasternack, 2008). Regulated rivers are often characterised by reduced channel widths (Petts 1980; Petts and Pratts, 1983; Church 1995; Gilvear 2000; Gilvear *et al.,* 2002, Petts and Gurnell, 2005). For instance, Gilvear (2000) found that reduced flood magnitudes on the River Spey below the Spey Dam caused narrower channels, reducing available habitat for fish.

Bradley and Smith (1984) showed that downstream of a dam, the Milk River, Alberta, narrows and there is bed degradation near the dam. Petts and Pratts (1983) found a decrease in channel width following reservoir installation on the River Ter, Essex, UK.

Jowett and Duncan (1990) suggest that flow variability causes a variation in stream morphology. Variations of mean cross-section and depth with velocity indicated that rivers with less flow variation were longitudinally more uniform, particularly at low flows, than rivers with high flow variability. Their field observations confirmed that stable flow rivers tended to be confined to well-defined channels and lacked the characteristic pool/riffle structure of gravel bed rivers. The difference may be related to the physical processes of scour and fill which help maintain the pool/riffle structure of gravel bed rivers during floods (Andrews, 1979).

River impoundment can also cause sediment aggradation and/or degradation (Bradley and Smith, 1984; Petts, 1984; Gilvear *et al*.,2000, Isik *et al.,* 2008), formation of large tributary confluence bars and siltation of channel substrates (Gilvear *et al.,* 2002). The primary driving factors in the changes to sediment dynamics and channel morphology are the blockage of the sediment flow path through the river (Petts, 1980, Isik *et al.,* 2008) and the change in the carrying capacity of the river (i.e. the regulation of flow). Leopold *et al.,* (1964) state that reservoirs trap in excess of 95% of sediment transported by a river. Reduced flow downstream of a dam can cause sedimentation when the sediment input by a tributary cannot be transported by the new flow (Petts, 1984). Conversely, immediately downstream of a reservoir, the channel may be eroded (Isik *et al.,* 2008), producing an incomplete 'armoured' layer of coarse material, which may be too large for spawning requirements (Petts, 1984). Such an effect was observed by Milhous (1982) when there was a decrease in Chinook salmon after impoundment below the Shasta Dam on the Sacramento River, California.

3.2.4 Temperature

Although beyond the scope of this project, flow regulation and river impoundment also impact upon water temperature, and therefore it will be considered briefly here. Epilimnial (water surface) and hypolimnial (deep water) releases can determine the nature of the effect of reservoir water releases on temperature. The main parameters affected by the different releases are water temperature and chemistry. River thermal regime is naturally very variable (on seasonal, diurnal and spatial scales). Releases from reservoirs may dampen these extremes (Petts, 1984; Crisp, 1995; Webb and Walling, 1996; Jackson *et al.,* 2007), as found by Ward (1974, in Petts, 1984) below the Cheeseman Dam, Colorado, USA where winter temperatures were higher and summer temperatures lower than normal, with less seasonal fluctuation. Temperature can be affected by reservoir releases by raising mean water temperature, eliminating freezing conditions, depressing summer maximum values, delaying the annual cycle and reducing diurnal fluctuation (Webb and

Walling, 1996). If stratification of temperature within a reservoir occurs, this is thought to greatly increase the impact of reservoir storage on river water temperatures (Crisp, 1995). At Kielder Reservoir in northeast UK, water is drawn from both eplilmnial and hypolimnial regions and mixed in order to minimise the effects of temperature regulation (Armitage, 1977; Crisp, 1977). Temperature change from stored waters lasts a relatively short distance (Gore, 1977), about 10-30 km, downstream of the release (Crisp, 1995) as inputs from tributaries and adjustment to the local air temperature mitigate effects.

3.3 Methods

3.3.1 Data available

Environment Agency gauging stations have recorded 15 minute flows at Eddy's Bridge (416815 558094) and Rowlands Gill (416815 558094, see Figure 2.1) between 1982 and 2008. There are also daily mean flow data for both of these sites. For Eddy's Bridge, this dates back to 1954 and at Rowlands Gill, to 1962. Both of these data sets extend to before reservoir construction, allowing an evaluation of the effects of river impoundment on the hydrograph for a location just downstream of the reservoir and also a site some 27 km downstream. The provision of data for the two sites is important in determining the magnitude of impoundment effects across the catchment scale.

3.3.2 Data analysis methods

Long-term hydrographs and flow duration curves were produced for each of the gauged sites as well as other statistical analyses:

First, long-term hydrographs were produced using daily mean flow data from Eddy's Bridge and Rowlands Gill gauging stations. Long-term hydrographs can be used to illustrate the continuous variation in flow over time and how a river responds to rainfall events. Gordon *et al*., (1992) suggest that hydrographs are important for biological classification of rivers as the biota are affected by how quickly a river rises and falls, a characteristic which can be deduced from hydrograph analysis. In this study, the hydrograph can be used to highlight patterns in flow before and after the River Derwent was regulated.

Second, flow Duration Curves can be used to determine the flashiness of a river. They display the relationship between streamflow and the percentage of time that it is exceeded (Gordon *et al*., 1992). Flow Duration Curves for Eddy's Bridge and Rowlands Gill flow data display the complete range of river discharges from low flows to floods and illustrate the variation of flow around the median. The curves were produced by ranking all of the flows measured over each time period (i.e. the period before impoundment and the period after impoundment) and assigned a percentage value according to their rank, with the highest flows attracting a low percentage value (because flow of that volume or greater are experienced for a small proportion of time). The percentage values were plotted against the flow magnitude to create the flow duration curves.

Third, basic descriptive statistics were derived for the pre- and post-impoundment daily mean flow data in order to quantify the change in characteristics of flow following regulation. The statistics created for each time period were Q95, Q5, Q1, median, minimum and maximum discharges. Q95 is the amount of flow that is exceeded for 95% of the time and represents a 'low flow'. Q5 and Q1 are flows that are exceeded for 5% and 1% of the time, respectively, and represent very high and exceptionally high flows. A comparison of these values pre- and post-impoundment will provide an indication of whether high and low flows have changed as a result of impoundment. Minimum and maximum values can be compared to assess the impacts of impoundment on the most extreme flows and median indicates the change in more frequent flows in the River Derwent.

Fourth, the Base Flow Index (BFI) can be used as a measure of the river's flow that is derived from stored sources (Gustard *et al*., 1992). The BFI of the River Derwent at Eddy's Bridge was examined to determine whether impoundment affected proportion of water contributed to the river through geological and stored sources. As groundwater and other sources from storage tend to be released to the river at a steady rate, a high proportion of baseflow would result in a less flashy, more stable flow. Gustard *et al.,* (1992) suggest that the BFI can be useful in studies of low and high flows. BFI can be calculated using smoothing and separation rules on a hydrograph within a computer program (the program used for this study was developed by the Institute of Hydrology). The index is defined as a ratio:

flow under separated hydrograph:flow under total hydrograph

Fifth, a flashiness index (Baker, *et al.,* 2004) was calculated in order to assess the change in river response to rainfall events as a result of regulation. The quantification of flashiness was achieved through the following equation:

$$R - BIndex = \frac{\sum_{i=1}^{n} |q_i - q_{i-1}|}{\sum_{i=1}^{n} q_i} \qquad \text{Equation 3.1}$$

where: *R-BIndex* is the Richards-Baker Flashiness Index and *q* is the mean daily flow

The index compares daily flows to total annual flow to determine a measure of variability for each year. The method is taken from Baker *et al.,* (2004).

Finally, the distributions of annual maximum and minimum flows were compared for pre- and post impoundment daily average flow data in order to determine the effects of impoundment on the occurrence of the extreme flows. This will highlight whether impoundment buffers flow regimes in all conditions, or whether there is a limit to the amount of change it can imply.

3.4 Results

3.4.1 Long-term hydrograph

Figures 3.1 and 3.2 show a change in hydrograph shape after the impoundment of the River Derwent. Frequency of high flows decreases considerably. This effect, although present at the downstream site of Rowlands Gill, is more pronounced at Eddy's Bridge, situated only one kilometre from the impoundment. After impoundment, at Eddy's Bridge, the river rarely reaches the peaks that were seen most years before the reservoir was built. High flows do occur, but less regularly. Indeed, the very highest flows at both sites occur after impoundment, demonstrating that once the reservoir is full, its 'buffering' effects no longer operate as excess water overtops the dam wall. Base flows have also been affected by impoundment, with the baseflow becoming elevated by river regulation at the upstream site of Eddy's Bridge. However, at the downstream site of Rowlands Gill the opposite effect has occurred and base flows at this site are lower following impoundment than those occurring before the reservoir was built.

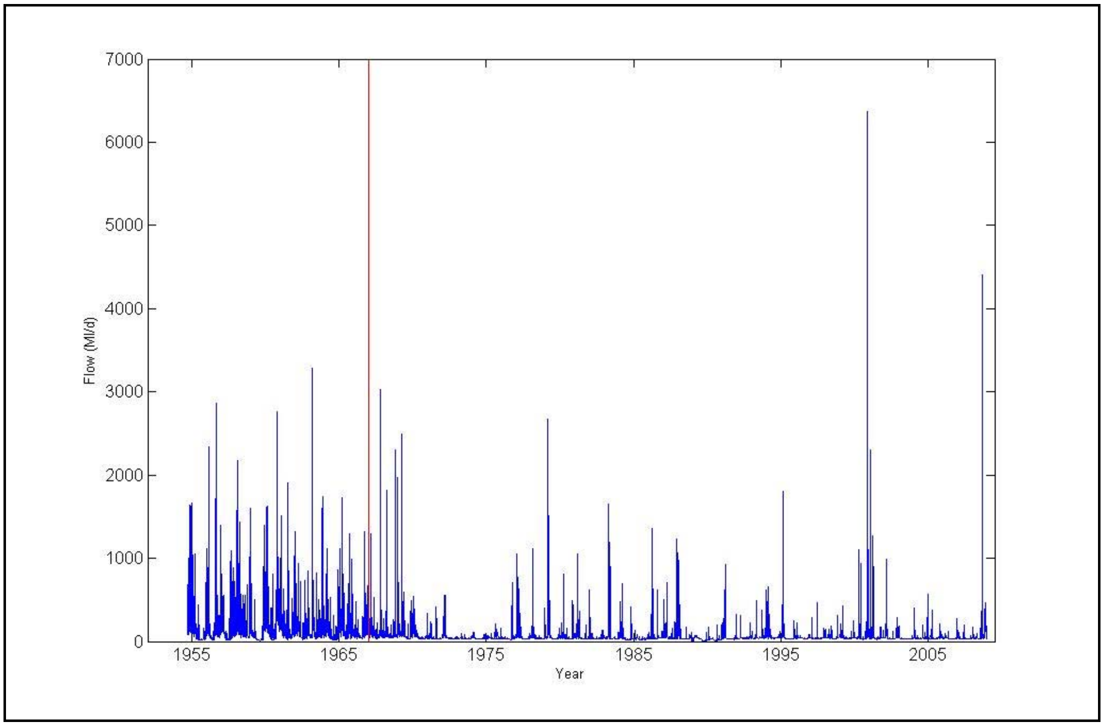

Figure 3.1 Long term hydrograph for Eddy's Bridge daily mean flow data

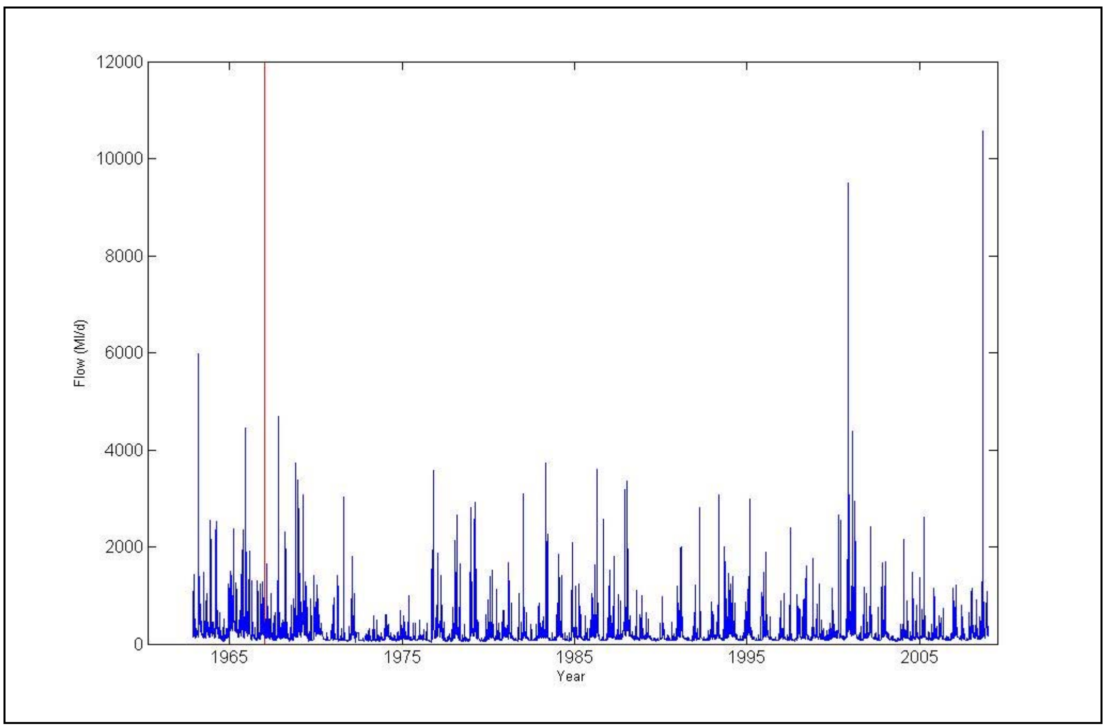

Figure 3.2 Long term hydrograph for Rowlands Gill daily mean flow data

3.4.2 Flow Duration Curves

At both sites, the flow duration curve becomes smoother and flatter following impoundment, although the difference is again more pronounced at the upstream site of Eddy's Bridge. The contrast between the two curves in Figure 3.3 shows that low and high flows occur much less frequently from 1967 onwards. Figure 3.4 shows that high flows at the upstream site of Rowlands

Gill also occur less frequently, but the lowest flows at Rowlands Gill are actually lower and occur more frequently following impoundment than the flows experienced prior to impoundment. This is supported by the Q95 values for Rowlands Gill (Table 3.1). Table 3.1 shows that the pre-impoundment Q95 of 18.1 Ml/d for the River Derwent at the upstream site of Eddy's Bridge occurs in excess of 97% of the time following impoundment and the pre-impoundment high flow (Q5) value of 570.1 Ml/d (for Eddy's Bridge), was exceeded for only 0.7% of the time since impoundment. For the downstream site of Rowlands Gill (Figure 3.4), the level of flow that was exceeded for 95% of the time (93.5 Ml/d) is only exceeded for 69% of the time post-impoundment (i.e. baseflows have fallen, contrary to what occurred at Eddy's bridge). The Q5 value of 1052.4 Ml/d is only exceeded 1.7% of the time after impoundment (i.e. high flows have also fallen, the same as occurred at Eddy's Bridge).

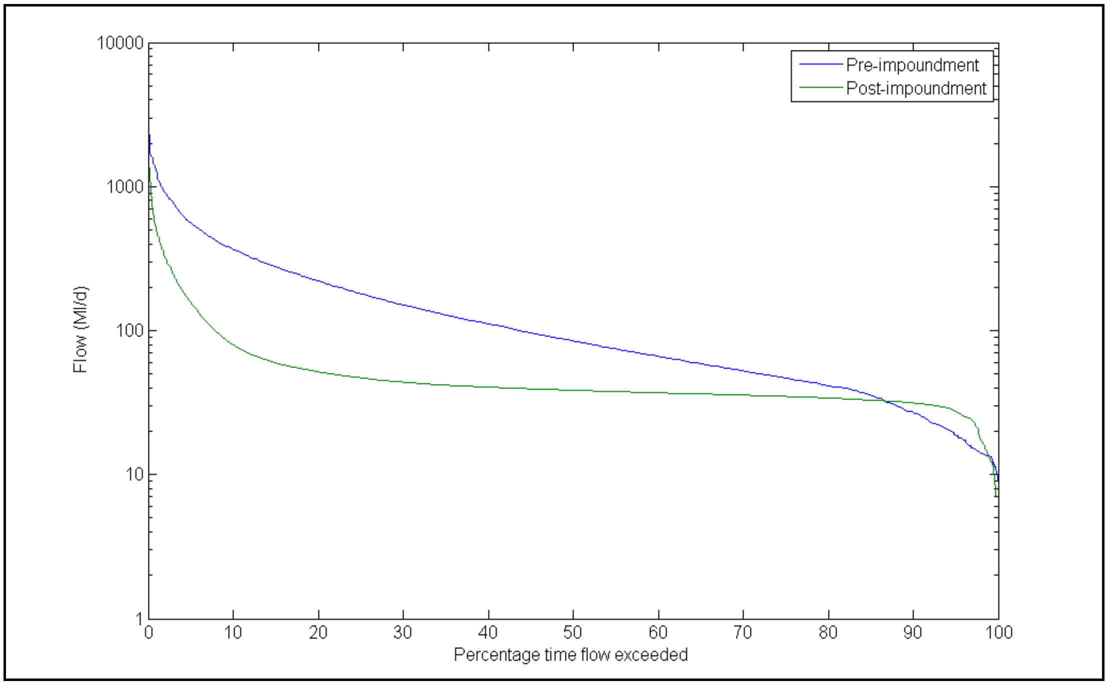

Figure 3.3 Flow Duration Curves for Eddy's Bridge (1 km from reservoir release) before and after channel impoundment (pre-impoundment data run from 1954 to 1966 and post-impoundment data run from 1967 to 2008).

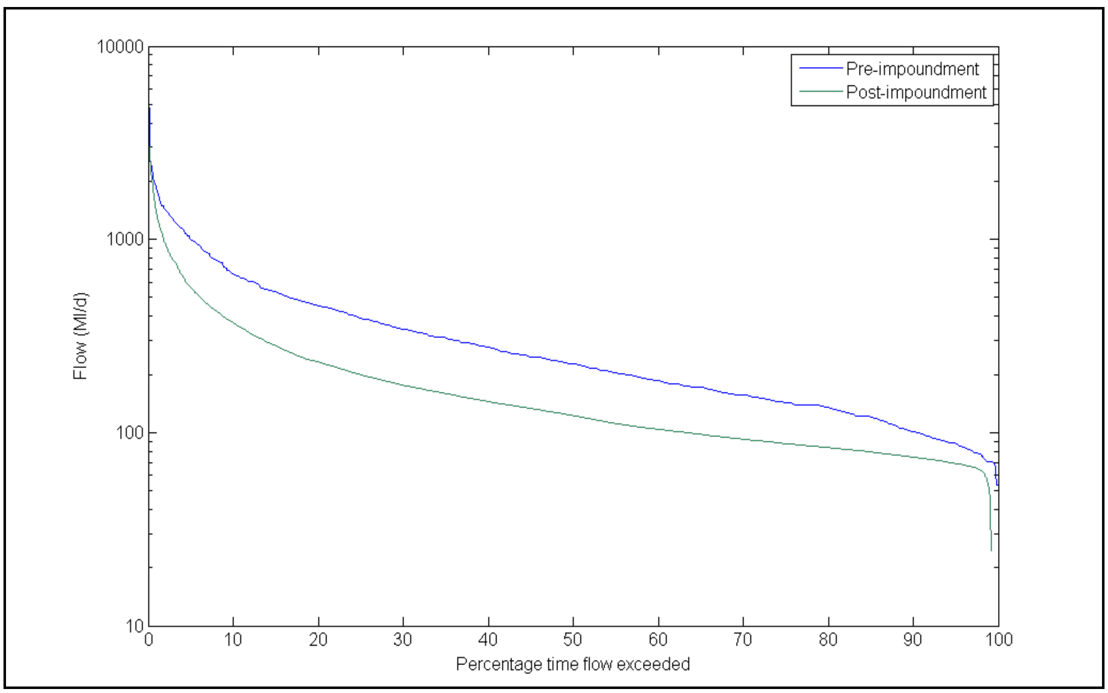

Figure 3.4 Flow Duration Curves for Rowlands Gill (27 km from reservoir release) before and after channel impoundment (pre-impoundment data run from 1962 to 1966 and post-impoundment data run from 1967 to 2008).

3.4.3 Statistical analysis

Table 3.1 shows the descriptive statistics calculated for daily mean flows in regulated and non-regulated periods, at Eddy's Bridge and Rowlands Gill.

	Eddy's Bridge		Rowlands Gill	
Statistic	1954-1966 (Pre-impoundment)	1967-2008 (Post-impoundment)	1954-1966 (Pre-impoundment)	1967-2008 (Post-impoundment)
Q95 (Ml/d)	18.1	28.3	93.5	70.2
Q5 (Ml/d)	570.1	163.9	1052.4	575.9
Q1 (Ml/d)	1311.6	501.6	1886.1	1375.5
Median (Ml/d)	87.7	38.7	244.7	123.8
Maximum Q (Ml/d)	3278.8	9281.6	5969.4	10573.6
Minimum Q (Ml/d)	9.1	7.2	53.8	24.5
Base Flow Index	0.43	0.63		

Table 3.1 Statistical analyses performed on pre- and post-impoundment average daily flow volumes, at two sites.

The low flow (Q95) and high flow (Q5) statistics support the patterns shown by the graphs in Figures 3.1 to 3.4. The flow supported by the river for 95% of the time increases by 62% following impoundment at Eddy's Bridge. However, this effect is not repeated at Rowlands Gill, where there is a decrease in Q95 of 25%. This suggests that the effect of regulation is depleted with distance from the point of release and that the data pre-impoundment are taken from a generally wetter period. Unfortunately, the effect of rainfall cannot be quantified as rainfall data for the catchment do not exist on a scale which is fine enough to compare to daily flows (rainfall totals are available for months only and this time scale is too coarse to be able to determine the response of the river to rainfall and individual rainfall events cannot be identified from within the data). The rainfall data that do exist do not extend back to the pre-impoundment period. The effect of the reservoir release will be 'diluted' with distance downstream as a result of inputs from the overland flow in the catchment and from tributary inputs. This can be explained by watershed area. The catchment area for the River Derwent as far as Eddy's Bridge is 118 km^2, but to Rowlands Gill, this area more than doubles, to 242 km^2.

Flows exceeded 5% of the time fall considerably at both sites: by 72% at Eddy's Bridge and by 55% at Rowlands Gill. Median (Q50) and very high (Q1) flows also decrease at both sites following river regulation. Interestingly the most extreme events (concerning both high and low flows) have occurred, for both sites, since impoundment. A decrease in magnitude of extreme flows supports the theories drawn from the shape of the flow duration curves shown in Figures 3.3 and 3.4. Baseflow index, another indicator of flashiness and of the role of groundwater in a river, rises from 0.43 to 0.63. This shows that there has been an increase in the ratio of base flow to total flow (Calow and Petts, 1992, p42) – i.e. storage is greater in the catchment now than it was before impoundment. This suggests that the River Derwent becomes more stable following impoundment and that a greater proportion of the flow may come from the geology, or more likely, a water store (i.e. the reservoir).

3.4.4 Flashiness index

The Richards-Baker flashiness index was used to quantify the assumptions made about flashiness of flow in the previously mentioned methods of analysis. Figure 3.5 shows the results of the Richards-Baker flashiness index analysis performed on pre- and post- impoundment data collected at Eddy's Bridge and Rowlands Gill. At the upstream site of Eddy's Bridge, the index values fall into a much tighter range for the pre-impoundment years than for the post-impoundment (range of 0.2 pre-impoundment and 0.5 post-impoundment). It can be seen that the effect is more pronounced at Eddy's Bridge and that extreme events, such as the floods of 2000 and 2007 cannot be 'buffered' by the current form of river regulation because when the reservoir is full, the water overtops the dam wall and creates unusually high flows within the channel. Surprisingly, flashiness appears to be generally higher at Eddy's Bridge and varies more between years at Eddy's Bridge than at Rowlands Gill with post-impoundment flashiness indices for Eddy's Bridge ranging from 0.049 to 0.55 and Rowlands Gill ranging from 0.14 to 0.46.

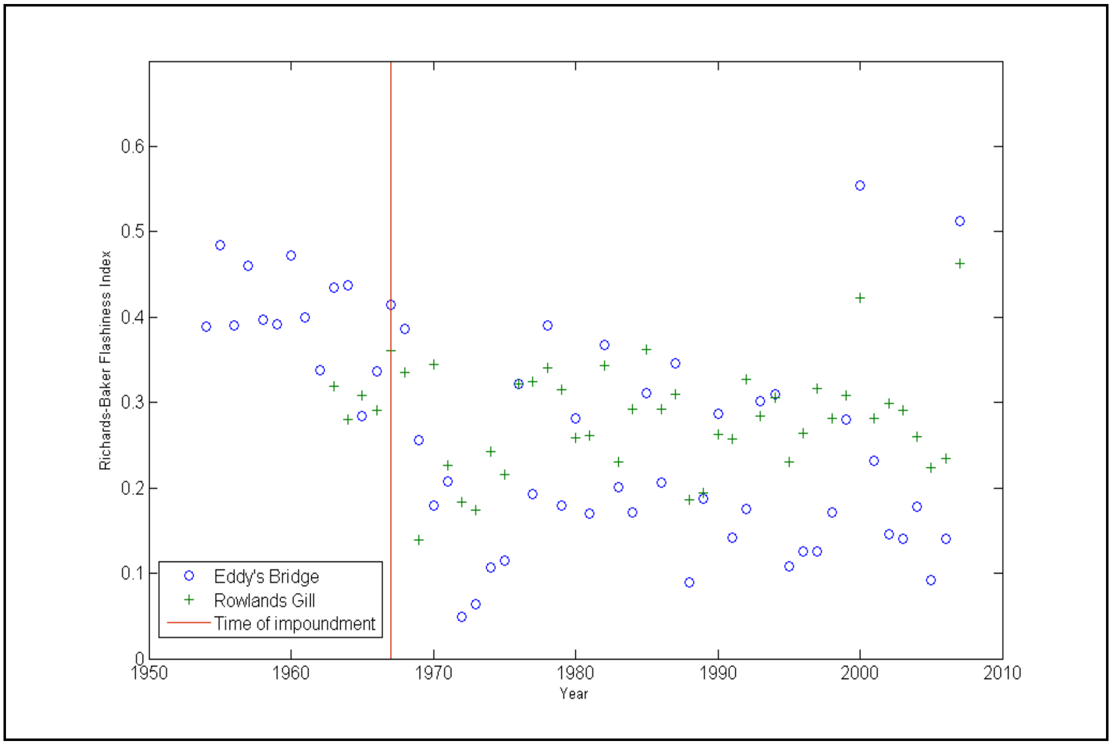

Figure 3.5 Richards Baker Flashiness Indices for each year at Eddy's Bridge and Rowlands Gill daily mean flow data, pre- and post-impoundment

3.4.5 Annual maximum and minimum flow rates

Figures 3.6 and 3.7 show the distribution of annual maximum and minimum flows, based on daily mean flow values for hydrological years (1 October to 30 September) at Eddy's Bridge and Rowlands Gill gauging stations. It can be seen that at Eddy's Bridge, annual maximum flows generally decrease following impoundment, although there is still some variability from year to year and very high flows are still evident despite river regulation. Annual minimum flows at Eddy's Bridge are slightly raised following impoundment, although very low flows are still experienced. Although there are fewer pre-impoundment data for comparison at Rowlands Gill, it appears that the impacts of regulation reach this downstream point with annual maxima being generally lower post-impoundment than pre-impoundment. However, as discussed for baseflow in the previous analyses, annual minima appear to be similar or lower than pre-impoundment annual minima.

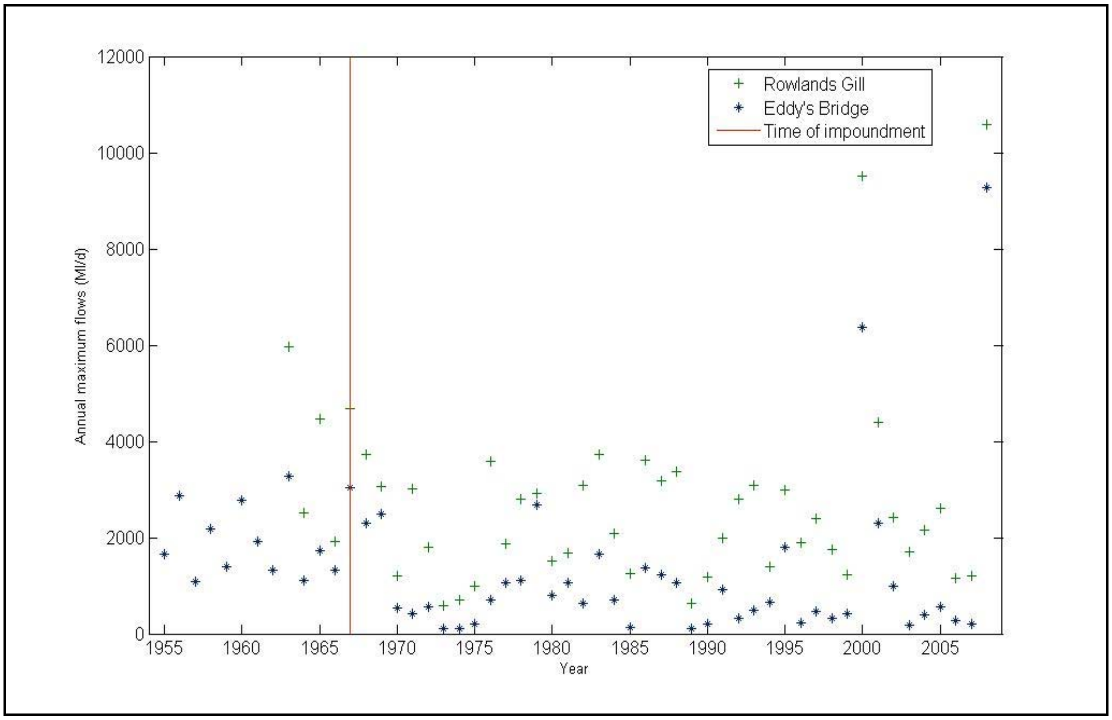

Figure 3.6 Annual maximum flows for Eddy's Bridge and Rowlands Gill

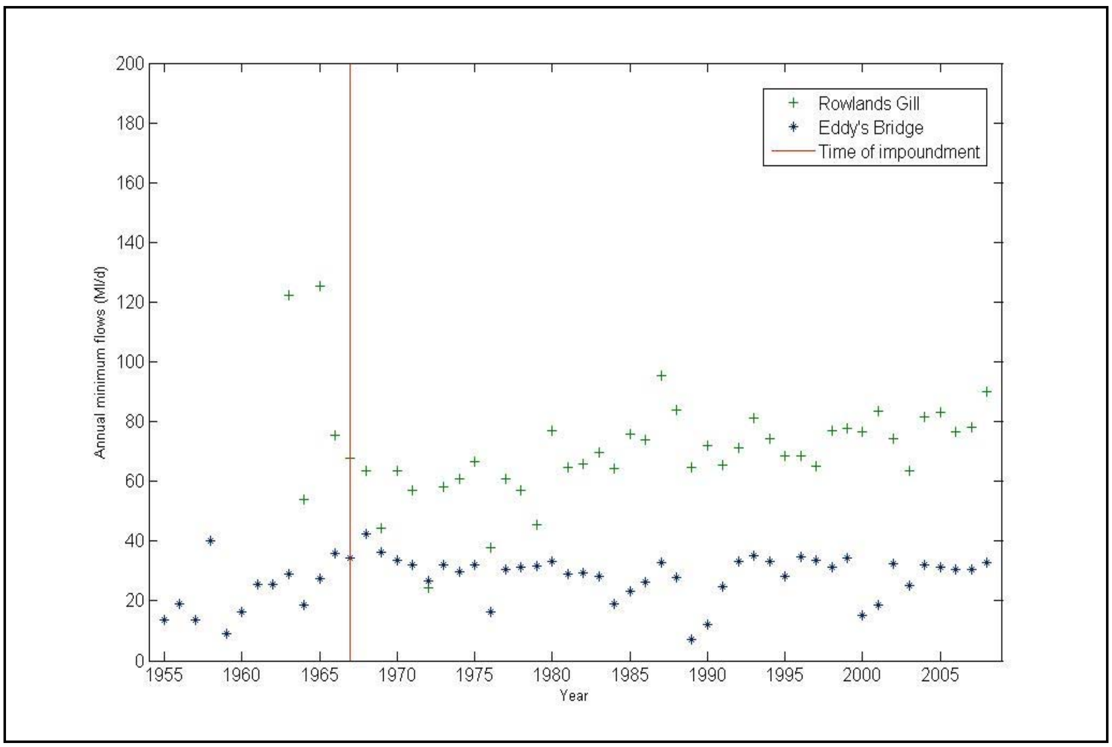

Figure 3.7 Annual minimum flows for Eddy's Bridge and Rowlands Gill

3.5 Discussion

It is clear that the flow regime of the River Derwent has been affected by impoundment through increased low flows, decreased high flows and less frequent extreme events. However, the most extreme events, particularly at high flows, cannot be prevented through the current level of river regulation. The fall in Q95 at Rowlands Gill, despite the provision of a baseflow from the reservoir indicates that the impacts of the impoundment decrease with distance downstream from the dam wall and it is expected that this is as a result of other inputs along the channel including unregulated tributaries and groundwater. This trend is supported by the long-term hydrograph which shows that flow regime is less markedly changed following impoundment at Rowlands Gill than at Eddy's Bridge. The overall effect of impoundment has been a 'dampening' of the hydrograph, as seen on many impounded rivers, including the Ebro River in Spain (Ibañez *et al.*, 1995, Batalla *et al.,* 2004), the River Tees, North East England (Crisp, 1977) and the Milk River, USA (Bradley and Smith, 1984).

According to Baker *et al's*., (2004) criteria for the definition of stable/flashy river, the average flashiness index of the pre-impoundment period (0.4) would place the River Derwent at Eddy's Bridge on the stable/flashy boundary, while the average flashiness index post-impoundment (0.2) for Eddy's Bridge would classify the river as 'stable'. There are a number of factors supporting the theory (e.g. Baker *et al*., 2004) that the flashiness of the river may decrease following impoundment. These include the smoother, gentler flow duration curves (which show less variation on the mean flow for the period studied), the increase in baseflow index and the Richards-Baker flashiness index results. As flashiness is a function of deviation from the mean flow, raising low flows and lowering high flows will naturally cause a reduction in flashiness. It could be expected that, being so close to the reservoir release (and therefore having less influence from unregulated tributaries), flows at Eddy's Bridge would generally be less flashy than at Rowlands Gill and would be more consistent over time. However, according to the Richards-Baker flashiness index results, this is not the case. There is one unregulated tributary that flows into the River Derwent (Burnhope Burn) before the gauged point at Eddy's Bridge (but after the point of the reservoir release). This sub-catchment of 24 km^2 constitutes 20% of the Derwent's catchment area upstream of Eddy's Bridge. Therefore, natural variability of flows in this tributary are likely to affect the variability of flows and flashiness of the gauged flow at Eddy's Bridge. Furthermore, Baker *et al*., (2004) suggest that their flashiness index values tend to decrease with increasing watershed area which would explain the higher index values produced for the smaller of the two areas studied here. The absolute flashiness index may also need to be converted to allow for watershed size before comparing two rivers (or an upstream and downstream reach). According to Baker *et al's*., (2004) stable/flashy definition criteria, the area of the watershed at Rowlands Gill would fall into the same category as that of Eddy's Bridge, meaning that the two sets of flashiness indices can be compared. Baker and Richards (2000) also suggest that high flashiness values can be expected in smaller watersheds "as a consequence of hydrograph mixing accompanying flood routing through stream networks and other scale dependent factors". Land use and management may also affect flashiness (Baker *et al.*, 2004). Land uses in the Burnhope Burn catchment are different to those found in the reaches between Eddy's Bridge and Rowlands Gill – with the Burnhope Burn catchment consisting of steeper terrain, elevated moorlands and grassland while the catchment area between Eddy's Bridge and Rowlands Gill is more low lying and a considerable length of the river is lined by riparian woodland. Further afield land use is arable or urban.

Descriptive statistics show that in most cases, extreme flows have become less extreme (e.g. the rise in the Q95 low flow level). It is expected that this is a result of the compensation release of (on average) 23.85 Ml/d which provides a more stable and uniform baseflow for the channel. The amount of water released from Derwent Reservoir each day is greater than the pre-impoundment low flow value (18.1 Ml/d) at Eddy's Bridge. This helps to augment flows in the channel at times when there is low natural input from rainfall. Similarly, the pre-impoundment high flow (Q5) value of 570.1 Ml/d is considerably greater than the post-impoundment Q5 of 163.9 Ml/d because high flows from upstream are stored in the reservoir. The Q5 value remains much higher than 23.85 Ml/d as a result of inputs from large rainfall events which reach the River Derwent via Burnhope Burn (upstream of Eddy's Bridge) and also due to overtopping events when the reservoir reaches its full capacity (see Figure 2.1 for a map of the catchment). The greater differences between the current compensation release rate of 23.85 Ml/d and flow values at Rowlands Gill (compared to Eddy's Bridge) are due to dilution of impoundment effects by flow accumulation through inputs from lower order streams. The effects are similar to those upstream, but less severe because the reservoir compensation release constitutes a greater proportion of the flow found at Eddy's Bridge than it does at Rowlands Gill. For example, the average daily release of 23.85 Ml/d is 131% of the Eddy's Bridge pre-impoundment Q95 value and only 25% of the pre-impoundment Q95 at Rowlands Gill.

Caution should be exercised when interpreting the flow duration curve results as Gordon *et al.*, (1992) suggest that in long-term flow duration curves, as opposed to short-term ones (such as daily flow duration curves) extremes in flow are smoothed out through averaging. Therefore, the length of the records used in this analysis may cause an under-representation of the most extreme flows and the difference in records used for the pre- and post-impoundment periods may also cause the smoothing to occur to different degrees, meaning that comparisons may not be entirely reliable. This could be solved by producing flow duration curves for shorter timescales, but these would not represent the distribution of the range of flow magnitudes over the time period being investigated.

3.6 Conclusion

Impoundment of the River Derwent has caused a change in the flow regime which is most pronounced near the source of the release (Eddy's Bridge) and which continues downstream but is generally dispersed by the downstream point of Rowlands Gill. There appears to be some impact on high flows at Rowlands Gill, but not on low flows. The changes caused by the regulation include the damping of hydrograph features, particularly high and low flow events and reduced flashiness of flow. However, the level of regulation currently offered by the reservoir does not extend to the most extreme events (e.g. the 2000 and 2007 floods) in which the reservoir was overtopped and flows in the channel reached higher than any recorded pre-impoundment flow. Knowing that the regulation of the River Derwent has affected the flow regime, it can be assumed that the changes in flow regime have had subsequent impacts on ecology through changes to habitat availability and connectivity. The next Chapter will examine pre- and post-impoundment macroinvertebrate data, in conjunction with the findings of the current Chapter, in an attempt to determine whether there has been any change to ecological diversity, and if so, what. Subsequent Chapters will then aim to investigate ways of improving (if necessary), the habitat conditions created by the current flow regime.

Chapter Four

Assessing the impacts of channel impoundment on ecology

4.1 Introduction

Channel impoundment not only affects the physical features of a river, but also impacts upon the flora and fauna inhabiting that river through alteration of the abiotic environment upon which flora and fauna depend. It has been suggested that the macroinvertebrate and fish populations of the River Derwent have been impacted upon by changes in flow regime following the regulation of flows. This Chapter aims to assess the theoretical impacts of impoundment on ecology through a review of literature and previously observed examples and to evaluate the effect of impoundment on macroinvertebrate and fish populations in the River Derwent through analysis of extant ecological data. Section 4.2 reviews the literature that exists on the impacts of regulations on ecology. Section 4.3 discusses the methods used in this part of the project. Section 4.4 presents the results of the data analysis and the results are discussed in Section 4.5. Section 4.5 will also discuss the links found between changes in flow and changes in ecology on the River Derwent. Conclusions and further work will be summarised in Section 4.6.

4.2 Current understanding of the impacts of regulation on ecology

4.2.1 Introduction

As discussed in Chapter Three, channel impoundment will cause changes to river flow, morphology and physical properties (such as temperature). These variables are determinants of habitat for river ecology and when they are altered, the habitable environment available to fish and macroinvertebrate populations is also affected. This Section aims to outline the effects of channel impoundment on ecology with reference to flow, morphology and temperature.

4.2.2 Macroinvertebrates

It is widely acknowledged that flow is a major control on macroinvertebrate species, families and populations (e.g. Gore, 1977, 1978; Extence, 1999). The variability and frequency of flow events dictate disturbances which may cause changes to communities. A disturbance, as described by Sousa (1984, pg 356), is “a discrete, punctuated killing, displacement, or damaging of one or more individuals (or colonies) that directly or indirectly creates an opportunity for new individuals (or colonies) to become established”. Therefore, both low and high flows can be considered as disturbances as it is the pattern and magnitude of such variability that is said to affect macroinvertebrate populations in streams (Clausen and Biggs, 2000).

Low flows

A decrease in discharge will cause decreases in water depth and width and therefore colonisable area (Cowx *et al.,* 1984). Reduced depths cause dynamic flow stability within rivers whereas at increasing depths, eddy formation occurs resulting in less favourable conditions for filter feeders (Ulfstrand, 1967). Additionally, reduced wetted bed area will have implications because many adults lay eggs in fast-flowing or broken water (Sawyer, 1950) and implications can be enhanced if period of drought overlaps the time for hatching (Hynes, 1941).

Drought can also increase water temperature through higher air temperatures and lower volumes of water, and it can affect water chemistry through decreased dilution potential (Extence, 1981). An increase in temperature may cause faster growth and earlier reproductive activity in some invertebrates (Extence' 1978). For example, Extence (1981) found that *Erpobdella octoculata* (freshwater leech), which usually hatches in July in the River Roding, hatched in May of the drought year of 1976. Increased temperatures can also lead to increased algal blooms and availability of plant detritus which are food sources for macroinvertebrates.

Disturbance events such as low flows and droughts are thought to initially cause a reduction in species abundance and diversity but it is generally accepted that recovery times are short (Ladle and Bass, 1981; Death and Winterbourn, 1995; Wood and Petts, 1999) and may be within several life cycles according to Cowx *et al.,* (1984). After recovery from low-flow events, invertebrate populations have been seen to improve in density and diversity (e.g. Extence, 1981). Extence (1981), in his study of the macroinvertebrate response to the 1976 drought, found an increase in invertebrate populations on the Roding River as a result of increased temperatures and food sources and decreased wetted area (for some species). However, certain Groups were eliminated (e.g. cased caddisfly larvae and prosobranch molluscs) due to stranding and chemical changes to their environment.

Extence (1981) concluded that most invertebrate species not only survived drought conditions but were able to exploit the modified environment and increased in number. Few species were adversely affected. It was also argued that should such conditions be recreated, naturally or anthropogenically (e.g. through reservoir construction and regulation of flow) similar changes in invertebrate communities would predictably follow.

High flows

The role of high flows in macroinvertebrate disturbance is simpler than that of low flow, is focused around simple physical processes and is well documented (Clausen and Biggs, 1997). Poff and Ward (1989) suggest that floods act as 'reset mechanisms' and Wood *et al.,* (2000) claim that disturbance by floods is the main control on community structure. Death of individuals or communities occurs through scouring, crushing and downstream export of individuals (Poff and Ward, 1989; Jowett and Duncan, 1990). Clausen and Biggs (1997) described the processes of removal by shear stress, abrasion and physical movement of gravels as reasons for the major impact that floods have on invertebrates. High flows also increase wetted bed area which may increase usable habitat area. Wood *et al.,* (2000) found a strong correlation between flow and number of individuals which they attribute to the increased wetted bed area. Catastrophic drift of macroinvertebrates may occur when natural freshets or regulated flows increase flow rate in a river and large quantities of invertebrate material can be carried (Crisp *et al.*, 1995), as found by Brooker and Hemsworth (1978) when freshets were released from Caban Coch reservoir on the Elan River. Reductions in macroinvertebrate densities were seen to occur when flows exceeded the median discharge by three times in a New Zealand river, in a study by Sagar (1986). Quinn and Hickey (1990) found that flooding can reduce biomass and richness in New Zealand rivers. Allan (1995) claims that increased frequency and/or duration of high flows may displace velocity sensitive organisms including young fish and macroinvertebrates.

Stability

Because of the importance of low and high flow events for macroinvertebrate community structures, it follows that channel and hydrological stability are determinants of habitat availability. Time since disturbance event allows colonisation to occur and macroinvertebrate density and diversity are known to recover from disturbances quickly (Ladle and Bass, 1981; Death and Winterbourn, 1995; Wood and Petts, 1999). Therefore, in streams with regular disturbances, community densities and diversities are likely to be low, while stable environments will offer the opportunity for more abundant and diverse communities to develop. A number of studies have found a decline in species number as stream stability decreased (e.g. Death and Winterbourn, 1981; Doeg *et al.,* 1989; McElravy *et al.,* 1989; Scrimgeour and Winterbourn 1989; Jowett and Duncan, 1990). Death and Winterbourn (1995) found species *richness* increased linearly with time after disturbance but species *evenness* (or diversity) peaked at periods of intermediate disturbance. Extence (1981) found that a reduction in number of spate flows would increase bed stability and the associated levels of flora and fauna. This will lead to greater potential for colonisation and an increase in invertebrate production during a drought period.

Sediment

Sediment composition within a channel affects community structures by determining where habitat is usable. Deposition of fine sediment can be associated with low-velocity flows. Fine sediment deposition will restrict hyporheic flow between coarser sediments which affects respiration as water cannot flow through coarse sediment and become oxidised (Wood and Armitage, 1997). Wood and Armitage (1997) also noted that filter feeders will be adversely affected by fine sediment suspended within the water which reduces amount of available prey. Extence (1981) found that silted conditions were detrimental to some species e.g. *Potamopygrus jenkinsi* and *Elmis aenea*. In a study by Wood and Petts (1999), areas with high sedimentation rates, such as those induced by river regulation were found to house more lentic associated species especially aquatic Gastropoda, Coleoptera and Corixidae. In such cases, areas that do have high flow probably act as refugia for macroinvertebrates. Invertebrate abundance and richness were lowest in rivers with beds of silt or sand, or cobbles overlain with sand deposits, in a study of invertebrate communities in 88 New Zealand rivers, by Quinn and Hickey (1990). It was noted by Mould (2006) that within the range of coarse sediment, size of clast has little impact on habitat availability. Cobb *et al.,* (1992) in Gibbins *et al.,* (2001) noted that substrate stability determines the effect of flood events on benthic invertebrates and sedimentation brought about by reduced flow may cause a change in insect populations and potentially a loss of fish food (after Petts, 1984). Saunders and Smith (1962) found a change in fauna in the Colorado River when accumulated sediment caused Trichoptera to be replaced by dipteran larvae.

Reservoir effects

The impacts of reservoir installation and river impoundment on physical characteristics of the river such as flow and morphology were discussed in Chapter Three, but their implications extend into the biology of a river and the changes to flow and morphology will impact upon habitat and ecology. As a result of impoundment, very low flows and droughts which are initially detrimental to invertebrate communities are largely prevented. Similarly, very high flows which strip the river of invertebrates, are mostly prevented. Intermediate flows, which were shown (above) to provide the most stable environment and the greatest diversity of invertebrates, occur for the majority of the time. There have been a number of studies directly into the impacts of impoundments and regulation on macroinvertebrates. Maddock *et al.,* (2001) note that the macroinvertebrate community overall, downstream of the dam on the River Derwent, Derbyshire, is slightly reduced (compared to that upstream) but with some species in greater abundance. Species preferring high velocities and depths were less abundant downstream of the reservoir. This supports the theory that habitat availability is less of a control on population diversity than hydrological conditions such as depth and velocity although Brown and Pasternack (2008) found that an increase in minimum flows caused a decrease in high quality habitat for most species and while Armitage (1995) suggests that an actual decrease in habitat area is less of an issue for invertebrates than fish, Crisp (1977) found that, under higher than normal discharge and velocity rates, flow altered the invertebrate community on the Tongue River, Montana, USA. Gibbins *et al.,* (2001) found little evidence of changes to invertebrate communities downstream of Kielder Reservoir during times of transfer of water to the River Wear.

According to Petts (1984), several species of Ephemeroptera and Plecoptera appear to be particularly sensitive to such changes in thermal regime. More generally, hypolimnial (low level) releases from Cow Green Reservoir on the Tees caused a reduction in diversity and an increase in biomass just below the reservoir (Armitage, 1988).

Chapter Three has shown that there are numerous effects of channel impoundment on stream characteristics and these can be summarised as: i) increase in baseflow (i.e. reduction of low-flow events); ii) prevention or dampening of high flow events; iii) increased hydrological and morphological stability of the channel. The impacts of these changes are closely related to channel parameters such as slope, geometry and roughness. Therefore, discussion of the impacts must be done with consideration of the specific stream in question, deeming the current study vital in investigation of the impacts that impoundment has had upon the River Derwent.

4.2.3 Fish

Depth

Depth requirements vary between species and within species, according to life stage. Different types of flow are required, for example, during the spawning period. It is recognised that salmonids generally require flow depths that are at least the height of the body, although there are exceptions (Armstrong *et al.,* 2003). Mitchell *et al.,* (1998) suggest that depth selected increases with flow, possibly due to fish holding station as flow increases. It is generally acknowledged that older salmonids prefer deeper water than younger ones (Heggennes, 1989; Greenberg *et al.*, 1996; Armstrong *et al.,* 2003). Juvenile and adult brown trout and Atlantic salmon have much lower depth requirements than grayling, for example, the minimum requirements are rarely below 0.2 m for brown trout and Atlantic salmon (e.g. Moyle *et al.,* 1983; Rimmer *et al.,* 1984; Morantz *et al.,* 1987; Heggennes, 1988; Heggennes and Saltveit, 1990; Maki-Petays *et al.,* 1997) while grayling have minimum depth requirements in the range of 0.4 to 0.8 m (e.g. Sempeski and Gaudin, 1995; Greenberg *et al.*, 1996; Mallet *et al.,* 2000).

In a review of published salmonid habitat requirements, Armstrong *et al.,* (2003) conclude that there have been a wide range of preferred depths suggested for both brown trout and Atlantic salmon, at all age ranges, and this may be for a number of reasons. First, fish may develop genetic adaptation to local habitats allowing them to thrive in that specific stream. Second, tolerances for these species are broad and it is possible for them to exist in a range of environments. Third, between-study variation in habitat use may result from differences in ecology. At low densities, most fish may use the best habitat but at high densities, fish may use sub-optimum habitat. Finally, fish do not select areas of habitat that have a certain depth independent of a certain velocity. They populate areas based on the best *combination* of factors (e.g. depth and velocity) (Shirvell and Dungey, 1983). Therefore, all variables need to be considered together.

Velocity

Velocity requirements also vary according to species and life stage. Shirvell and Dungey (1983) suggest that, for trout, velocity is more important in habitat selection than depth. Hynes (1970) notes that velocity dictates energy expenditure required by a swimming fish and therefore the shape of a fish will determine its capability to move efficiently and the velocities which it is able to resist. Specific velocities are also required to condition the sediment in which eggs are lain. For instance, salmonids often create redds in areas of accelerated flow so that downwelling currents will force streamflow into and through substrate (Alonso *et al.,* 1996). As seen with depth requirements, adult and juvenile brown trout and Atlantic salmon require similar velocities, often quoted in the range of 0 to 0.2 m s^{-1} (e.g. Bovee, 1978; Heggennes, 1988; Heggennes, 1990; Heggennes and Saltveit, 1990; Bird *et al.,* 1995). However, in the spawning period, preferred velocities for Atlantic salmon are greater than those for brown trout (0.4m s^{-1} and upwards for salmon (e.g. Beland *et al.,* 1982; Moir *et al.,* 1998) and 0.1-0.2 m s^{-1} for brown trout (e.g. Shirvell and Dungey, 1983; Raleigh *et al.,* 1986; Johnson *et al.,* 1995)). Grayling have higher velocity requirements which often start with a minimum of 0.2 m s^{-1} but have been seen to range from 0.2 to 1 m s^{-1} in both juvenile and spawning seasons (e.g. Sempeski and Gaudin, 1995; Mallet *et al.,* 2000; Nykanen *et al.,* 2004).

Velocity can affect shelter as turbulence and white-water areas determine predator visibility (Petts, 1984). Patrick (1970) notes that predators are important in maintaining species diversity. Therefore, a change in velocity may lead to a change in diversity and community composition (e.g. low predation pressure may produce a benthic community dominated by a few species with high densities).

Jowett and Duncan (1990) found that as variability of flow increased, mean water velocity decreased. Velocity determined how weighted usable area (WUA) varied with flow variability. WUAs with lower optimum velocities tended to increase with flow variability, whereas WUA with higher optimum velocity decreased.

The points made in the previous Section by Armstrong *et al*., (2003), explaining the reasons for the observation of a broad range of habitat requirements, also apply to velocity requirements.

Morphology and Substrate

Gravels are preferred as a bed material by salmonids and there are a range of size preferences observed within the literature. Fish create redds (spawning nests) within the gravel and lay their eggs in them. It is important that the spaces between the gravels do not become clogged by fine sediment as this will reduce intragravel flow velocities (Petts, 1984; Crisp, 1995). This throughflow is essential for a number of reasons. It is needed to remove toxic metabolic wastes produced by the eggs. A lack of hyporheic flow also leads to lower levels of dissolved oxygen in the water near the bed and these are important for fish whose metabolisms rise rapidly with temperature (this includes trout) (Hynes, 1970). As temperature rises, oxygen is needed to support the increasing metabolic rate. Low levels of oxygen have also been said to be required to prevent late emergence and lowered survival rates of fry (Petts, 1984; Raleigh, 1986; Alonso *et al.,* 1996). Excess amounts of fine sediment can also trap alevins as they attempt to emerge after hatching (Crisp, 1995), or prevent fish from dislodging gravel to recover eggs (Hausle & Coble, 1976; Witzel & MacCrimmon, 1983; Petts, 1984, Crisp, 1995, Brown and Pasternack, 2008). Substrate of cobble (for juvenile) and boulder (for adult) (Greenberg *et al.,* 1996) size is also important in providing shelter and resting areas for fish (Hynes, 1970). Milhous (1998) suggests that flushing flows are needed to remove fine sediment from fish spawning habitat.

Fish are affected by change in channel width as this controls the area of usable habitat and by shape because this controls the velocity of water under a given discharge (Petts, 1984). Brown and Pasternack (2008) note that width can also affect flow velocity. In fact, area of channel inundated can affect area for spawning, area for food availability and water temperature (Fraser, 1972). Changes in channel shape, bed morphology and composition are significant for fish habitats (Petts, 1980). Vaux (1968), claims morphology and particle size composition of the channel bed are of major importance in the determination of stream habitats. This is reflected in Petts' (1980) statement that quantity, velocity and quality of flows within substrate and interchange between substratum flow and stream water are determined by surface profile of bed, thickness of deposits and substrate permeability. As channel shape can control velocity, and velocity is important territorially to some fish (Petts, 1984), change in channel shape may cause a redistribution of areas inhabited by fish species. Reyes-Gavilan *et al.,* (1995) suggest that bank habitat, which is advantageous for brown trout, can be reduced through increased channel width as the bank length/channel surface ratio is reduced.

Pools and heterogeneous shallow areas are required for shelter and food (Petts, 1984, Bowen *et al.,* 2003) and inundation by increased flows may flood pools and shallow areas, removing these features. Riffles provide primary spawning areas for fishes, including salmonids, and increased flows due to regulation can inundate riffles (Petts, 1984), effectively stripping the river of this feature. This is important as brown trout prefer riffles and favour large blocks in their habitats for shelter (Reyjol *et al.,* 2001 and Baran *et al.,* 1995), and Atlantic salmon require bar features and heterogeneous areas for spawning (Petts, 1980). Similarly, brown trout density, biomass per square metre and mean age have been positively correlated with the presence of large boulders (Reyes-Gavilan *et al.,* 1995).

Ecology is dependent on substrate size (Richter, 1997). This was demonstrated by Brown and Pasternack's (2008) study in which Chinook Salmon were found to require a certain range of sediment size to spawn as well as space between clasts for egg deposition. Gibbins and Acornley (2000) note that a study by Capra *et al.,* (1995) found that if spawning habitat dropped below 80% of the optimum for a continuous period in excess of 20 days during the spawning season, recruitment was reduced (concerning brown trout). Moir and Pasternack (2008) comment on the fact that shallow, moderately flowing areas with sediment smaller than cobble size were used for spawning by Chinook Salmon on the Lower Yuba River, California.

Temperature

Water temperature is one of the most important controls on fish growth. Variations in temperature affect a number of life stages (Crisp, 1981, 1988) among fish communities, but embryos and young, emerging fish appear to be the most susceptible to changes in temperature. This may be attributable to the fact that water temperature requirements for embryonic development are more stringent than the range tolerated by adults and that high water temperatures are deleterious to embryos (Armstrong *et al.,* 2003). Temperature directly affects survival of eggs and rate at which alevins develop (Crisp 1993, 1996). Elliott *et al.,* (1998) found that mortality of Atlantic salmon eggs increases as temperatures fall below 4^{o}C. Because colder rivers cause longer incubation periods, adult salmon have been found to spawn earlier in order to ensure their progeny emerge at optimum times (Elliott *et al.,* 1988). Trout eggs are susceptible to *rising* temperatures, as illustrated by Jowett (1992) and Crisp (1993), when mortality of embryos was found to increase as temperatures exceeded 9-10^{o}C.

Feeding and growth rates are known to be affected by changes in temperature (Crisp, 1993). For example, Power (1981) notes that growing stage for salmon is during the period of days that water temperatures exceed 7^{o}C. Similarly, Allen (1940, 1941) and Hynes (1970) claim that salmon do not grow in winter when temperatures are too low. Elliott and Hurley (1997) found that growth of trout is negligible below 3.6^{o}C. Stress is known to become present in salmon at 22^{o}C and upper limits are quoted to be in the range of 25-28^{o}C (Elliott, 1981; Elliott and Hurley, 1997) while most fishes in temperate environments can withstand temperatures as low as 0^{o}C (Hynes, 1970). Adult brown trout stop feeding at 19.5^{o}C and mortality begins at temperatures of 25^{o}C (Elliott, 1981, Elliott and Hurley, 1997).

Spawning (Reyes-Gavilan *et al.,* 1995), and stimulation of migration back to sea (Crisp, 1995) are also controlled by water temperature. Furthermore, temperature is important in determining dissolved oxygen content whereby cold water regimes can provide refuge for fish in need of oxygen in warm streams (Burkholder *et al.,* 2008).

Clark *et al.,* (1999) note that, for poikilotherms, the availability of thermal heterogeneity in a river system can provide potential energetic advantages, or refugia in more extreme temperature regimes. Locally cooler areas that could be used as refugia do occur as a consequence of depth variations and shading by vegetation.

Variability

The controls on fish communities discussed earlier in this Section (depth, velocity, substrate, temperature) are all affected by environmental variability (e.g. variations in rainfall and discharge) and the impact that variability has on these variables has implications for riverine ecology. As discussed in the invertebrate Section, flow variability is an important determinant of microhabitat community and it is generally agreed that maximum diversity occurs at intermediate flows. The role of variability for fish populations, however, is more contested, with claims that variability is essential (e.g. Hynes, 1970; Jowett, 1990; Minshall, 1988) and also that it is detrimental (e.g. Swales and Harris, 1995; Stalnaker, 1996). Richter *et al.,* (1997) stated: "Hydrological variation is now recognised as the primary driving force within river ecosystems because fluvial processes maintain a dynamic mosaic of channel and floodplain habitat structures". Swales and Harris (1995) suggest that habitat diversity decreases with flow extremes.

Jowett (1990) identified flow variability as one of the major factors influencing trout distribution and abundance. It has been acknowledged by a number of authors that high and low flow disturbances play a central role in structuring stream communities (Hynes 1970; Schlosser, 1987; Delucchi, 1988; Minshall, 1988; Resh *et al.,* 1988).

Lobon-Cervia and Rincon (2004) also found that environmental variability in the form of year-to-year variation in discharge was the major determinant of population dynamics of brown trout in Rio Chabatchos (Spain) during their study between 1987 and 1999. Environmental variability operated through a limiting factor namely the availability of suitable micro-habitats for juveniles shortly after emergence which in turn was influenced by climatic factors such as rainfall. This was a variable that occurred on a regional scale.

Stalnaker *et al.,* (1996) argue that riverine fish often exhibit high inter-annual variability in reproductive success that corresponds to variations in hydrological conditions (Mills and Mann, 1985; Schlosser, 1985). This can mean dampening flow extremes can be detrimental to fish populations. Eliminating or dampening flow variability may eliminate a non-equilibrium mechanism that contributes to diversity in many river systems (Moyle and Li, 1979). Stalnaker *et al.,* (1996) suggest that elimination or significant dampening of inter-annual variability in streamflow can reduce early life history success and thus alter the biodiversity in free-flowing sections.

Floods and temporal variations in flow are needed to maintain channel geomorphology and substrate transport as well as suitable substrate conditions for the benthic organisms on which salmon feed (Gilvear *et al.,* 2002), as demonstrated on the River Devon, Clackmannanshire where stable compensation flows have been known to cause silt and algae on the river bed and a deterioration in the condition of the fishery on the river.

Further to requiring annual and inter-annual variability in flows, the *predictability* of varied flows is important to some communities. Some species are dependent on infrequent exceptional flow years (e.g. floods, droughts) for successful reproduction (Bain and Boltz, 1989). Vondracek (1985) suggested that the high predictability of non-flood periods in montane western streams serves as a reliable environmental cue for native samlonid species whose flood-susceptible fry are present only during the flood-free summer season. Similarly, John (1963) found that cyprinid life histories in the desert south west are linked to the timing of floods, which can be used as reproductive cues.

While the role of variability has been accepted within the literature, there are cases to suggest that its effects vary depending on location and population in question, and impacts can sometimes be mitigated through natural processes. Lobon-Cervia and Mortensen (2005) found that climatic variation caused variation in discharge, and temperature and was a major regulating factor in the size of brown trout populations in two streams. However, in one stream (Bisballe Baek, Norway), the recruitment decreased curvilinearly with increasing discharge whereas in the Rio Chaballos (Spain), an optimum discharge was found when recruitment increased for a time with increasing discharge and then decreased as discharge continued to increase. They also noted that the interaction of discharge and temperature was an important control where water temperature acts as a buffer to low discharges, allowing persistence of species through some environmental variability. Lobon-Cervia and Rincon (2004) found that despite the brown trout populations being driven by environmental variability, there was a large amount of persistence and maximum recruitment occurred at the most frequent discharge conditions. Extremes at both ends of the spectrum resulted in decreased recruitment. Stalnaker *et al.,* (1996) found that habitat stability during early life history could be directly translated into year-class-strength for some fish species, including brown trout.

Swales and Harris (1995) suggest that habitat diversity in most rivers is at a maximum at intermediate flows and decreases towards flow extremes – suggesting that reduced hydrological variation, which dampens extreme flows, is not necessarily detrimental. According to Poff and Ward (1989), long periods of low flow can cause high vagility among fishes and physiological tolerance of low dissolved oxygen (Matthews, 1987). These may lead to high persistence and stability.

Poff and Allan (1995) found that variable habitats tend to support resource generalists and stable habitats supported more specialist species. This would suggest that different levels of flow variation would be beneficial to different types of community. Therefore, it is difficult to say whether varied flows are generally beneficial to fish populations. This presents the question of whether a specific species (or form of species) should be focused upon when designing habitat improvements, or whether the best approach is to make changes to the hydrological regime and observe how the form of species responds (e.g. McDonald *et al.,* 2004). In this project, the aim is to improve habitat quality for the fauna as a whole through a change in hydrological regime by using the flow requirements of three key fish species as a guide to the kind of flows necessary.

The coexistence of some species and high community diversity frequently depend on habitat heterogeneity caused by environmental disturbances (papers in Pickett and White, 1985), but large scale or frequent exotic disturbances may impose habitat homogeneity (Denslow, 1985). If a disturbance of naturally high magnitude, but artificial frequency occurs, it can exceed the ability of most species to exploit the variability.

Changes to flow variability can also have indirect impacts. The natural drift of food organisms can be disturbed by changes in daily and seasonal flow which may affect fish feeding habits and reproduction (Petts, 1984) although Fraser (1972) argues that artificial flows can contribute to feeding by dislodging benthic organisms.

Impacts of reservoirs

Given the impacts of river impoundment on channel characteristics such as depth, velocity, temperature and morphology (Chapter Three), and given the influence that these parameters have upon fish communities, the impacts of river regulation on fish populations can be considerable and have been well documented.

Hynes (1970) and Richter *et al.,* (1997) note that migration and spawning are often stimulated by rising flows, therefore the more uniform flow created by storage reservoirs would not encourage migration.

Gibbins and Acornley (2000) found that spawning habitat for Atlantic salmon on the River Tyne below Kielder Reservoir was reduced to about one third of that available at optimum discharge by the compensation flow (i.e. 1.3 $m^3 s^{-1}$ of 4 $m^3 s^{-1}$). It is suggested that in this river, 2 $m^3 s^{-1}$ is needed to cover most of the bed with water – an important requirement for the spawning season. The regulation reduces the time that 2 $m^3 s^{-1}$ is exceeded in the spawning season from

90% to 60%. By regulating the river and reducing normal flow to the compensation flow rate, Gibbins and Acornley (2000) predict that up to 60% of the former spawning habitat area available would become stranded. Gibbins *et al.,* (2001) found that, on the same river, compensation flows were sufficient to support older fish (0+), but they limited availability of spawning habitat.

Crisp (1977) studied the effects of the Cow Green reservoir and dam on the River Tees, UK. He found that discharge fluctuations were smoothed considerably but that the more significant impacts arose from the change in thermal regimes. The water released from the reservoir was generally at the same temperature as the water stored in the middle of the reservoir and caused a reduction in diel temperature fluctuations and a delay in the spring rise of water temperature of 20-50 days. It is suggested that this would delay the annual peak of metabolism in brown trout by about one month, but impact on growth rate would be negligible. Reyes-Gavilan *et al.,* (1995) found that brown trout spawning times were delayed in colder areas. This may be an impact arising from temperature regulation in some streams. Holden and Stalnaker (1975) found a reduction in native fish abundance below the cold release of the dams on the Colorado River. Diapause eggs in Lehmkuhl's 1972 study would not hatch in areas influenced by hypolimnial release.

Petts (1984) states that anadromous fish species such as salmonids have been particularly affected by impoundment and claims that the *rate* of increase or decrease in flow is arguably the most important factor for fauna. This is perhaps more important in rivers controlled for purposes such as Hydro-Electric Power (HEP). Nevertheless, a rapid rise in discharge will erode spawning gravels and remove benthic invertebrates (a source of fish food) (Petts, 1984) and it is less likely that fish will be able to adapt and cope with rapid changes than with slow, steady changes in flow and habitat.

In summary, it has been shown that ecology is affected in the long-term both directly and indirectly by changes in flow regime. The possession of a long-term macroinvertebrate data set will allow a unique analysis of the impacts of Derwent Reservoir on macroinvertebrate communities.

4.3 Methodology used to assess impact of river impoundment on ecology in the River Derwent

4.3.1 Data available

Macroinvertebrates

Two forms of analysis were conducted on macroinvertebrate data. These were:

1. Comparison of pre- and post-impoundment macroinvertebrate populations at the same site (Allensford) on the River Derwent. This was done using macroinvertebrate data collected before impoundment by Professor H. B. N. Hynes and after impoundment by the Environment Agency.

2. An ergodic approach to allow an unregulated catchment with similar physical characteristics to act as a proxy for the Derwent catchment prior to impoundment. The site of Allenbanks on the River Allen was used for this analysis. More detail on each of the data sets is given below.

Macroinvertebrate counts for a number of species and taxa have been collected at a number of locations on the River Derwent (Figure 4.1). Between 1955 and 1962 (pre-impoundment), samples were collected by Professor Hynes on behalf of Consett Steel Works. It is believed that a kick sampling method was used for this purpose, in which the river bed is disturbed through kicking and the species found in a certain amount of time are recorded. Environment Agency (EA) routine sampling has been conducted at a number of locations between 1990 and 2007, using the kick-sample method. The Hynes data have not been previously analysed and provide a unique opportunity to assess the macroinvertebrate conditions of the River Derwent before the onset of flow regulation. Professor Hynes has produced numerous publications on instream ecology (e.g. 1960; 1964; 1970) and is considered a leading figure in the investigation of stream ecology. His data therefore provide a reliable image of the conditions of the River Derwent between 1955 and 1962. EA data categorise number of species/taxa at an abundance level, rather than an actual count, allocating scores of 3, 33 or 333, depending on presence of species/taxa. The data also needed to be filtered as some samples were collected to species level and some to taxa level. The site used for comparison of the two time periods was Allensford as this is the only site sampled by Hynes (pre-impoundment) and the Environment Agency (post-impoundment). The site lies upstream of the main industrial areas in the Derwent catchment. Therefore, risk of skewing of results from poor water quality was reduced. The location of the sampling sites, with Allensford highlighted, is shown in Figure 4.1. Environment Agency macroinvertebrate data were also obtained for the unregulated catchment of the River Allen, another tributary of the Tyne which lies to the west of the River Derwent (Figure 4.1). The two catchments are physically similar

(Table 4.1) and the data for the Allen (taken at Allenbanks) were used as a comparison to the Derwent. The methods used are discussed in Section 4.3.2.

Characteristic	Allensford (River Derwent)	Allenbanks (River Allen)
Altitude (m)	115	85
Discharge category	5	5
Slope (m/km)	10	9.5
Distance from source (km)	26	24
Width (m)	15.7	18
Depth (cm)	17.3	25.4
Alkalinity (mg/l)	35.8	95
Substrate: clay/silt (%)	1	0
Substrate: sand (%)	5	0
Substrate: pebble/gravel(%)	31	24
Substrate cobble/boulder (%)	63	76

Table 4.1 Comparison of physical data for the sampling sites of Allensford (River Derwent) and Allenbanks (River Allen). The Allen is ungauged and therefore a comparison of flow data is not possible. However, long term average width and depth can be used as a substitute for this.

Fish

There are limited fish count data available for the catchment. There are some counts of brown trout, but none of grayling and counts were not taken on a regular basis or at regular sites. The value of these data, for this project, are therefore limited. The sampling locations for brown trout fish counts are shown in Figure 4.2.

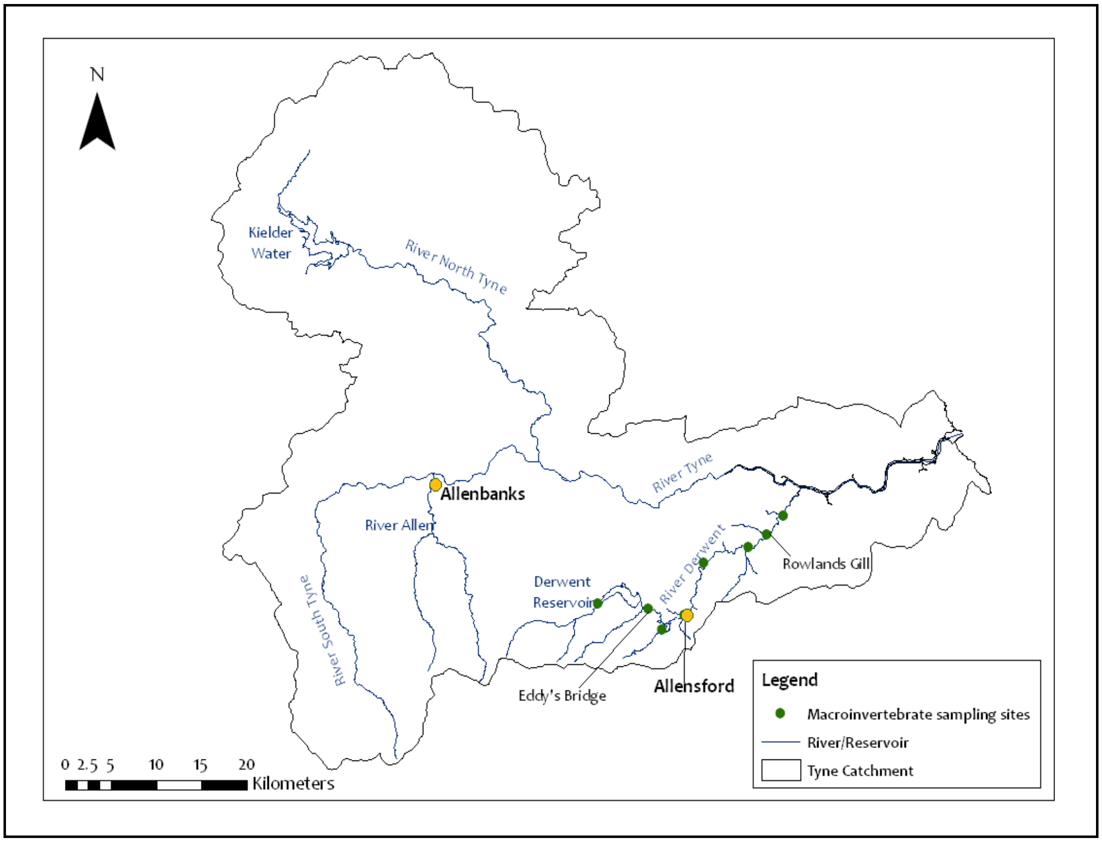

Figure 4.1 Macroinvertebrate sampling site locations on the River Derwent and River Allen, with the sites of Allensford and Allenbanks highlighted

4.3.2 Data analysis methods

Macroinvertebrates

Macroinvertebrate data were filtered in order to ensure that species counts were not duplicated within family counts and that Hynes' data were in a format comparable to the EA data. Statistical analyses were performed on the data sets to determine changes in population composition before and after impoundment.

- Observations of **families present** and absent in each of the time periods (pre- and post-impoundment).

- **Biological scores** were calculated for the macroinvertebrate data from the pre- and post-impoundment periods at the site of Allensford (Figure 4.1). The scores were used as an indication of the flow level and water quality characteristics of the River Derwent. The following scores were calculated:

    - **NTAXA** is simply a count of the number of individuals belonging to each taxanomic group within the sample. It was used as a measure of the number of different taxa found in each sample. This method is sensitive to sampling effort.
    - **BMWP** (Biological Monitoring Working Party) system is a method of assessing the biological quality of a river, based on water pollution. It uses binary data and relies on taxonomic resolution only. Pollution-intolerant families are assigned high scores and pollution-tolerant families, low scores (Johnson *et al.,* 1993). The scores are totalled to provide the BMWP score (Momo *et al.,* 2006). A table of the scores that should be assigned to each species/family are listed in Appendix One. The result is sensitive to sample size and sample processing efficiency (Hawkes, 1997). Therefore, the BMWP was converted to the **ASPT** by dividing the BMWP score by the number of taxa in order to standardise the results. From hereon, only analysis of ASPT will be discussed as it is a more representative measure.
    - **LIFE** (Lotic-invertebrate Index for Flow Evaluation) scores, developed by Extence *et al.,* (1999), are used by the Environment Agency to relate macroinvertebrate data to flow regime. Each taxum is assigned a Group reflecting the type of flow it inhabits and another Group reflecting its abundance within a sample (scores are listed in Appendix Two). These two values are combined to produce a score from a pre-defined list. The listed score is used in the equation:

      LIFE = Sum(fs)/n **Equation 4.1**

      where: sum(fs) = sum of the individual taxon flow scores for the whole sample

      n = number of taxa used to calculate the sum(fs)

      Higher flows should result in higher LIFE scores.

- **Statistical analyses** were performed on the biological scores from pre- and post-regulation periods. The Mann-Whitney U Test was used to determine whether ASPT scores from the pre-impoundment period differed significantly from the post-impoundment ASPT scores. This was repeated for the LIFE scores, BMWP scores and NTAXA. ASPT and LIFE scores are relatively insensitive to sampling method and effort. Therefore, they can be considered as reliable. The Mann-Whitney U test was considered appropriate for the comparison of biological data sets over time because it is an effective method of comparing unmatched samples and can be conducted on data sets with as few as four observations in each (Fowler *et al.,* 1998, p166). The test can also be used on data sets which are not normally distributed (Fowler *et al.,* 1998, p166), making it ideal in comparison of counts or diversity indices.

Richness and Diversity tests

Tests were conducted to quantify family richness and population diversity at Allensford across the two time periods. Not all invertebrates were identified to species level. To prepare the data for analysis, the family counts could have been assigned to a species (or number of species) within that family, without knowing the actual abundance of certain species within the family, or the family could have been added to the species list as a measurement artefact. To remove the error associated with either of these options, the species data were aggregated to family level. Further, Lane *et al.,* (2007) note that the aggregation of species data to family level allows a fairer comparison of data sets collected through different sampling methods (e.g. kick sampling and Surber sampling) or when sampling efforts may vary. Magurran (1988, pg 73) notes that both the Shannon and the Simpson indices, (as well as the species richnesss indices) are sensitive to sample size, with diversity index increasing as sampling effort/size increase. Alatalo and Alatalo (1977) also note that results can be biased according to sample size but do suggest that using more than one assessment can help to do justice to the complicated notion of richness and diversity (i.e. more than one test should be applied to ascertain parsimony of results).

With the data in a format suitable for analysis, the following methods were applied to the family level data:

Richness can be quantified using a simple count, or through the indices developed by

- **Margalef** (Clifford and Stephenson, 1975):

$$D_{Mg} = (S-1)/\ln N \qquad \textbf{Equation 4.2}$$

where: S = Number of species recorded (in this case, number of families recorded)

N= Total number of individuals summed over all species (families)

- and **Menhinick** (Whittaker, 1977):

$$\mathbf{D_{Mn} = S/\sqrt{\mathit{N}}} \qquad \textbf{Equation 4.3}$$

The main advantage of these richness analyses is the ease of calculation (Magurran, 1988). However, the number of species is insensitive to the distribution of individuals across a species. Well distributed species are less likely to be lost to random loss effects (i.e. local species extinctions). Therefore, if an understanding of the *diversity* and *distribution* of a population is required, the indices of Shannon (H) and Simpson (D) are more appropriate:

- **Shannon's Diversity index** was calculated using the following equation:

$$H^{'} = \sum_{i=1}^{S} \frac{n_i}{N} \left| \log \frac{n_i}{N} \right|$$

Equation 4.4

where: H' = Diversity

N_i = Count of individuals of species *i*

N = Total count of all individuals of all species

While the data used were not in the most suitable format for this type of analysis (EA data in some cases were given as an abundance category rather than a finite number, e.g. 3; 33; 333 and counts were at family level rather than species), the analysis gives an *indication* of the change in population diversity within the River Derwent following impoundment. Despite its limitations (e.g. it does not reflect sample size and is insensitive to rare species (Peet, 1974)), the Shannon index is a useful tool in comparing diversity between numerous sites or periods of time as it incorporates both richness and evenness (Peet, 1974; Miserendino *et al.,* 2008).

The Shannon index of diversity is based on two assumptions, the first being that samples are taken from a population that is indefinitely large and the second that all species are represented in the sample. This may cause an error in the output (Magurran, 1988). Within this calculation, $\log_2$ is usually used, although Magurran (1988) notes that any log base may be adopted, provided that this is consistent throughout analysis.

- **Simpson's diversity index** was used as a comparison to the results obtained through the Shannon index, although output values are inverse to the Shannon results (a higher Simpson's D value denotes lower diversity, a higher Shannon's value denotes higher diversity), one set of results can be used to support (or oppose) the conclusions drawn from the other. Simpson's Index (D) is used as a measure of the relative concentration of dominance (Peet, 1974). The index was derived through this equation:

$$D = \sum_{i=1}^{S} \frac{n_i(n_i - 1)}{N(N-1)} \qquad \textbf{Equation 4.5}$$

where: n_i = number of individuals in the *i*th species

N = total number of individuals

It should be noted that Simpson's index is heavily weighted towards the abundance of the most common species in a sample, while being less sensitive to species richness (Magurran, 1988). Therefore, results should be interpreted with caution.

- **Comparison against a 'natural' population.** The macroinvertebrate population in the Derwent (at Allensford) was compared to what may be a 'natural' population in a similar, but unregulated catchment. The River Allen, which also flows into the Tyne, has similar physical characteristics such as slope and catchment area (at the point of sampling) to the Derwent, but differs in aspects such as sediment size composition, alkalinity and width. These are factors considered to be impacted upon as a result of impoundment. The macroinvertebrate population data from the two sites were compared with respect to BMWP, ASPT, number of taxa and LIFE scores to determine whether the population found in the Derwent was significantly different to that of the similar ungauged river. The method used was the Mann-Whitney U test, the merits of which have been previously described.

Flow and ecology

Analyses of flow patterns were coupled with macroinvertebrate population analyses to infer impacts of flow on populations. Correlation and linear regression analyses were performed to quantify the relationships between flow characteristics and macroinvertebrate richness/diversity. The flow characteristics investigated were flashiness and change in range of annual flow between pre- and post-impoundment periods.

Analysis of downstream changes in species diversity/abundance were not conducted due to the wide range of possible influences on populations over the length of the river. As species present may change as a result of factors other than flow regime with distance downstream (e.g. discharges from industrial functions such as Consett Steel Works), little can be determined about the effect of the reservoir through spatial analysis. Therefore temporal analysis was the focus within this study.

Fish

Figure 3.9 shows the extent of the fish data available. The most they can be used to show is that brown trout are present in the river. Because there is no consistency between timing and location of sampling sites between each visit, no temporal or spatial pattern can be deduced from the information available. A lack of counts at any location and on any date could be due to the absence of sampling rather than the absence of fish. No data exist on fish populations before impoundment and therefore the effect of impoundment cannot be quantified. There exists much speculation over the effect of the reservoir on fish populations in the River Derwent but this extends only to anecdotal evidence and cannot be quantified. However, there is a keen interest in the fish populations of the river, both from an ecological perspective (e.g. the Environment Agency and Northumbrian Water Ltd) and from a recreational perspective (e.g. anglers). Therefore, fish populations in the River Derwent, and the potential to improve their habitat, must be considered key aspects of this study and the data that are available will be considered briefly in Sections 4.4 and 4.5.

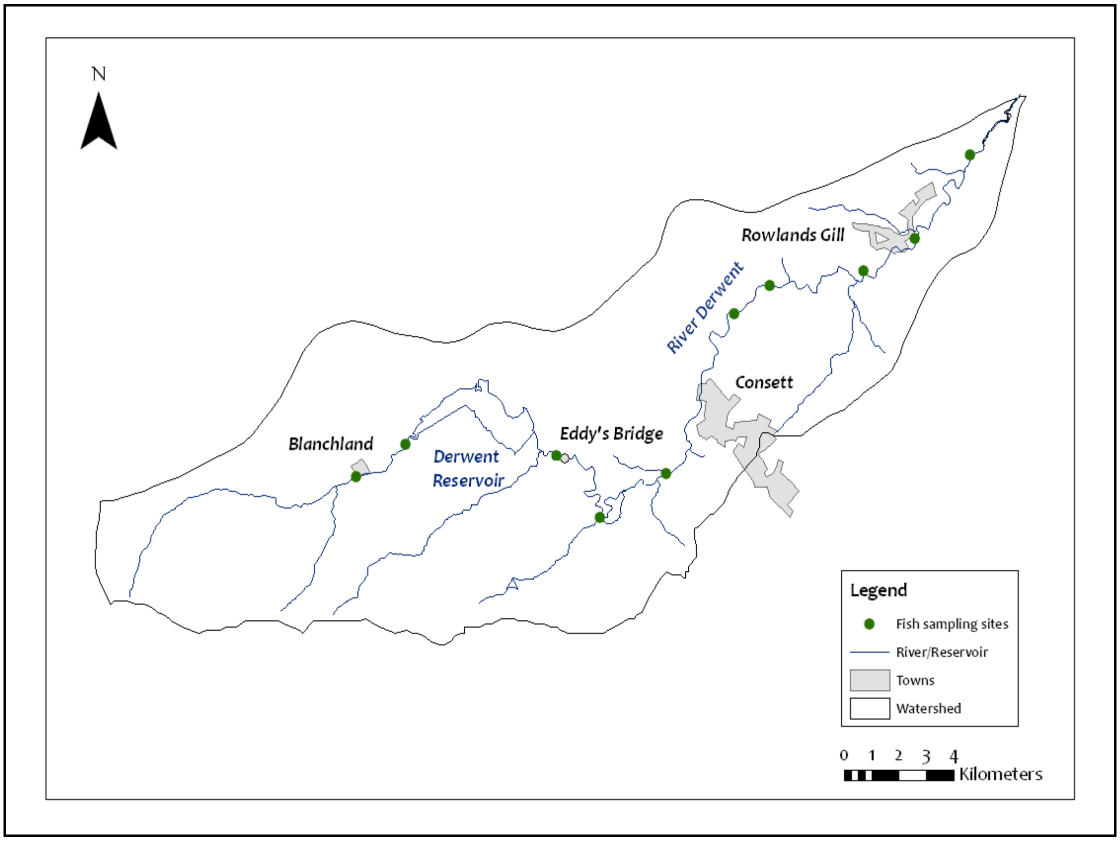

Figure 4.2 Brown trout sampling sites on the River Derwent

4.4 Results

4.4.1 Ecology

Macroinvertebrates

Table 4.2 summarises the presence and absence of families from the pre- and post-impoundment time periods. It is evident that there are fewer families present in the River Derwent in the pre-impoundment period. The families present in each time period are largely different with only nine families (out of 52 observed in total over the whole time period) being present both pre- and post-impoundment. Even fewer families are present in the pre-impoundment period only (three out of a total of 52 families observed).

Family	Present Pre-impoundment	Present Post-impoundment	Family	Present Pre-impoundment	Present Post-impoundment
Hydrobiidae	n	y	Psychomyiidae	n	y
Ancylidae	n	y	Lepidostomatidae	n	y
Sphaeriidae (Pea mussels)	n	y	Odontoceridae	n	y
Asellidae	n	y	Leptoceridae	n	y
Gammaridae	n	y	Tipulidae	n	y
Baetidae	n	y	Psychodidae	n	y
Heptageniidae	n	y	Chironomidae	n	y
Leptophlebiidae	n	y	Rhagionidae	n	y
Ephemeridae	n	y	Empididae	n	y
Ephemerellidae	n	y	Muscidae	n	y
Caenidae	n	y	Planariidae	n	y
Taeniopterygidae	n	y	Pediciidae	n	y
Nemouridae	n	y	Enchytraeidae	y	n
Leuctridae	n	y	Naididae	y	n
Perlodidae	n	y	Haliplidae	y	n
Perlidae	n	y	Gyrinidae	y	y
Chloroperlidae	n	y	Hydraenidae	y	y
Dytiscidae	n	y	Glossosomatidae	y	y
Hydrophilidae	n	y	Polycentropodidae	y	y
Scirtidae	n	y	Hydropsychidae	y	y
Elmidae	n	y	Limnephilidae	y	y
Sialidae	n	y	Sericostomatidae	y	y
Rhyacophilidae	n	y	Ceratopogonidae	y	y
Hydroptilidae	n	y	Simuliidae	y	y

Table 4.2 Macroinvertebrate family presence and absence in the pre-impoundment (data from 1955-1962) and post-impoundment (data from 1990-2007) time periods according to Hynes and Environment Agency macroinvertebrate samples. Green shading denotes presence and red shading denotes absence.

When calculating a Mann-Whitney U score to compare two sets of data with the hypothesis that the two sets of data are taken from different populations, the lowest Mann-Whitney score from the two sets should be used for analysis (Ebdon, 1965, p 61). Figure 4.3 shows that the ASPT scores increased post-impoundment and this change is significantly different according to the Mann Whitney U test (Table 4.3). However, there has been little change in the LIFE scores and the two data sets are not significantly different. Number of taxa found within a sample have also increased since impoundment, with a statistically significant difference (Figure 4.4 and Table 4.3), but it should be noted that this result is dependent on sampling effort and the level to which macroinvertebrates are identified by individual samplers. As discussed in Section 4.3, the effort employed in the sampling by Hynes is uncertain and the Environment Agency used abundance categories rather than actual counts in some of their samples. Therefore, results on the number of taxa (NTAXA) should be taken with caution. Because higher ASPT scores are given to families with lower tolerances of water quality and hydromorphological stresses (Armitage *et al.,* 1983), the increase in ASPT post-regulation suggests that the species inhabiting the river are less tolerant of poor water quality than the species that were present prior to regulation. Therefore, in order for these species to populate the river, the water quality must have improved over time. There are two years in the pre-impoundment period with particularly low ASPT scores (1960 and 1961). 1959 experienced some very low flows. In fact, the lowest minimum flow of any of the years both pre- and post-impoundment was for 1959. In this year, the period of low flow (usually April to June in the years 1956 to 1963 (Figure 3.1)) lasted longer than usual, until October. Low flows may have caused a deterioration in water quality as pollutants and impurities may not be as well diluted. This is particularly important for areas which lie downstream of sewage and water treatment works releases, such as Allensford, the site for which the ASPT scores have been calculated. As macroinvertebrate populations can take at least one life cycle (a year or more) to recover from such a disturbance, this may explain the lower ASPT scores found in the subsequent years. A higher LIFE score suggests a higher flow rate. There is very little change in the LIFE scores for pre- and post-impoundment periods which would indicate little change in flow rate. As Chapter Three shows that this is not the case, it could be assumed that the overall flow rate has not changed, although the extremes have become less frequent, or that morphological effects are dominant over flow effects in determining habitat suitability.

	ASPT	LIFE	NTAXA
Lowest Mann-Whitney U score (working on the hypothesis X≠Y)	6	32	0.5
Critical value, based on number of samples (at 0.05 confidence level)	15	15	15
Result	Calculated value < than critical value, reject H_0. **Samples are significantly different at 0.05 confidence level.**	Calculated value > than critical value, cannot reject H_0. **Samples are not significantly different at 0.05 confidence level.**	Calculated value < than critical value, reject H_0. **Samples are significantly different at 0.05 confidence level.**

Table 4.3 Statistical analysis of macroinvertebrate data pre- and post-regulation

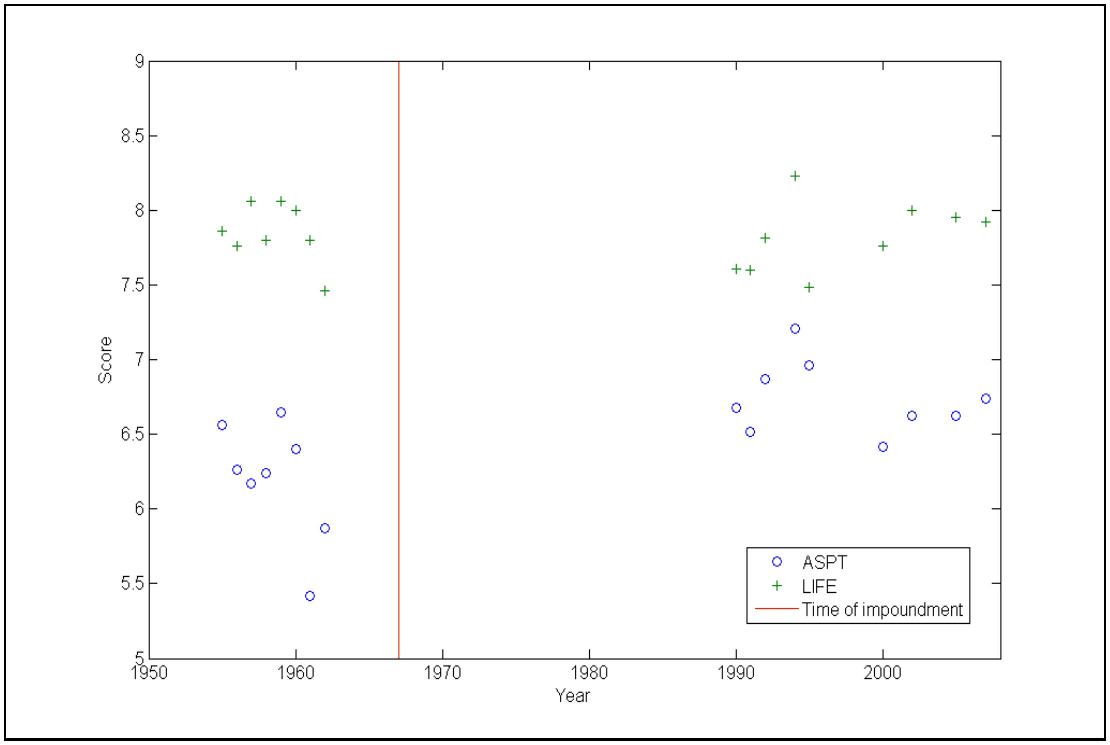

Figure 4.3 ASPT and LIFE scores for pre- and post-impoundment periods at Allensford

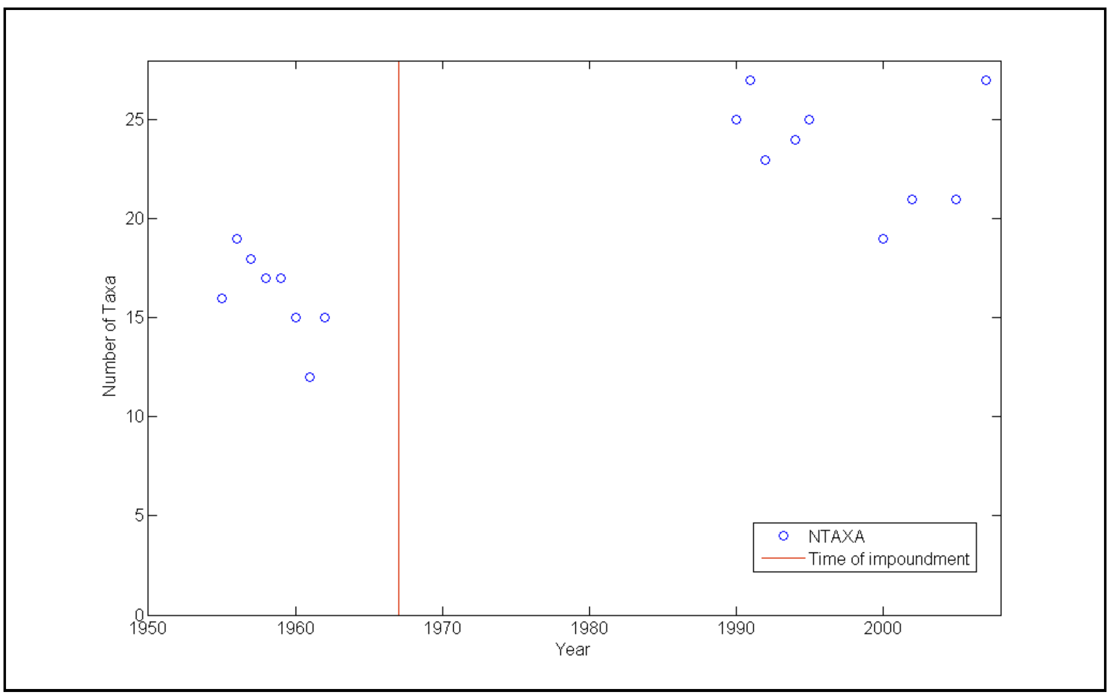

Figure 4.4 Number of taxa for pre- and post-impoundment periods at Allensford

Diversity Indices

Richness appeared to increase considerably according to the Margalef index, but to a lesser extent when quantified through the Menhinick equation (Figure 4.5). The simple count method showed a considerable increase in richness post-impoundment with roughly three times as many families being present in the samples in recent years.

The Simpson and Shannon Diversity Index results both suggest that diversity has increased since the River Derwent became regulated. A Simpson's value of 0 denotes infinite diversity, and 1 denotes no diversity. Therefore, the decrease in Simpson's value over time suggests that diversity has improved. Shannon's index is converse to this and an increase in Shannon's index defines an increase in diversity. Therefore, as Figure 3.8 shows, both tests concluded that diversity has increased. It should be noted, however, that an increase in diversity does not guarantee an increase in quality and the diversity observed may be different to what is deemed 'natural'. Table 4.4 shows that there has been little change in the mean and median Menhinick index values between the pre- and post-impoundment periods. There has been a decrease in variability of those scores since impoundment. The mean and median Margalef scores have increased considerably, but there has been virtually no change to the variation of scores between the two time periods. This trend is reflected in the Shannon mean and median results for each time period, although variability has increased slightly. There has been a decrease in Simpson mean and median scores (which denotes an overall increase in diversity) and a slight decrease in variability in the post-impoundment time period. The change in the Margalef, Simpson and Shannon results suggest that overall diversity is greater in the post-impoundment period than in the pre-impoundment period. As discussed previously, if there has been a change in diversity, the Menhinick results appear to be insensitive to it.

Ecological score →	**Menhinick**		**Margalef**		**Simpson**		**Shannon**	
Time period →	**Pre-impound-ment**	**Post-impound-ment**	**Pre-impound-ment**	**Post-impound-ment**	**Pre-impound-ment**	**Post-impound-ment**	**Pre-impound-ment**	**Post-impound-ment**
Mean	1.04	1.08	0.92	2.63	0.42	0.16	0.49	1.01
Median	1.06	1.06	0.88	2.63	0.45	0.15	0.47	0.95
Standard deviation	0.53	0.31	0.43	0.42	0.16	0.09	0.17	0.23

Table 4.4 Descriptive statistics for richness and diversity indices for the pre- and post-impoundment periods

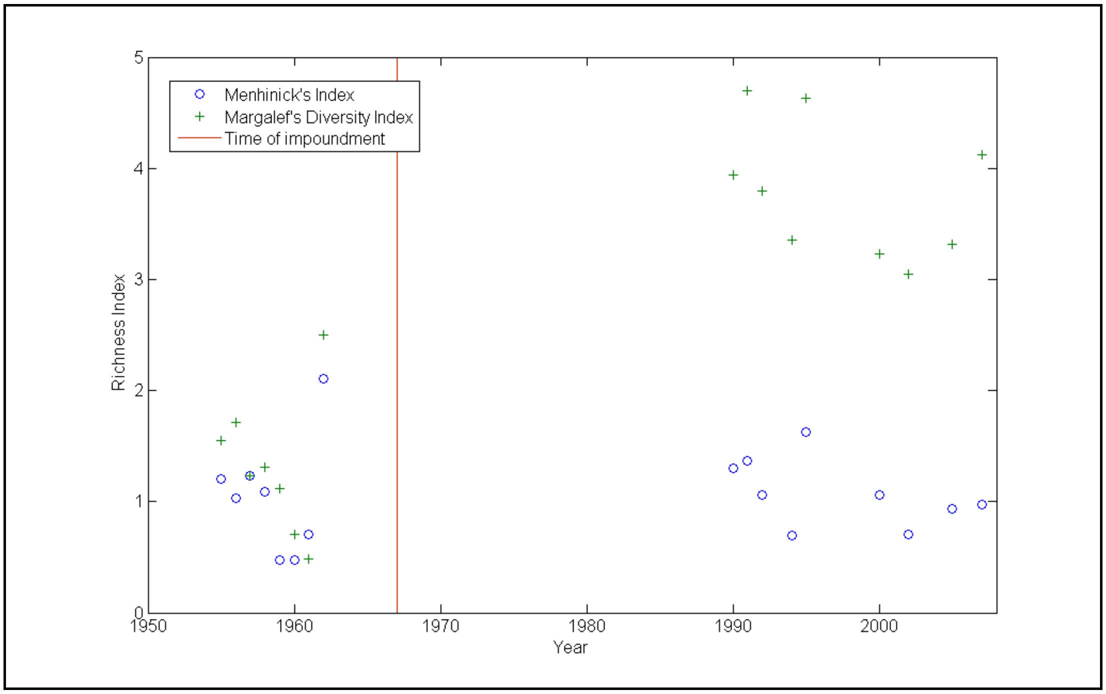

Figure 4.5 Quantification of richness according to Menhinick's Index and Margalef's Diversity Index, based on family level data

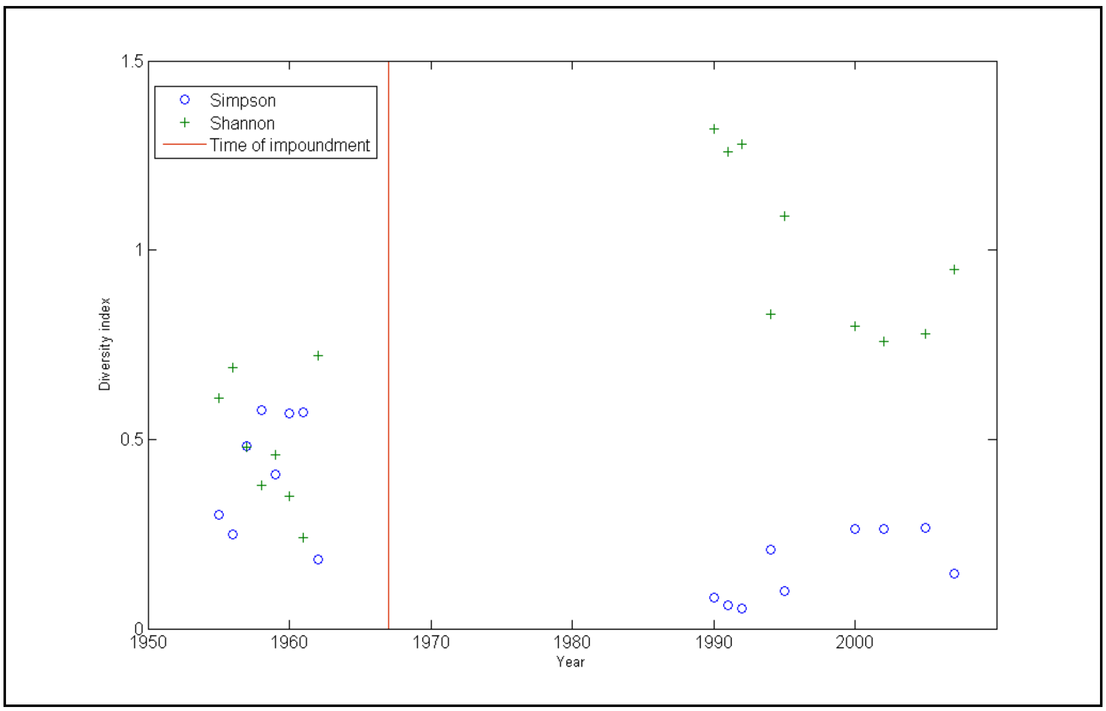

Figure 4.6 Simpson and Shannon diversity index results for pre- and post-impoundment years

Comparison against a 'natural' population

The Mann-Whitney U test results (Table 4.5) show that there is no significant difference in the number of taxa between the Derwent (post-impoundment) and the Allen (at present), in the ASPT scores for the Derwent and the Allen or in the LIFE scores between the Derwent and the Allen. The mean LIFE scores, however, are greater in the Allen than in the Derwent, suggesting that the species populating the Derwent prefer lower flows to those in the Allen. If the species present have been altered by flows and therefore reflect the level of flow in the channel, this would support the observation that the overall flow level (or the flow that occurs for most of the time), is lower in the Derwent than in the Allen, as a result of regulation.

	ASPT	**LIFE**	**NTAXA**
Lowest Mann-Whitney U score	22	20	15.5
Critical value, based on number of samples (at 0.05 confidence level)	10	10	10
Result	Calculated value > than critical value, cannot reject H_0. **Samples are not significantly different at 0.05 confidence level.**	Calculated value > than critical value, cannot reject H_0. **Samples are not significantly different at 0.05 confidence level.**	Calculated value > than critical value, cannot reject H_0. **Samples are not significantly different at 0.05 confidence level.**

Table 4.5 Statistical analysis of macroinvertebrate data at the regulated site and a ‘natural’ site

4.4.2 Flow and ecology

Flow and ecology

It appears that the effect of the controlled flow regime in the River Derwent may have had an impact on the ecology of the river. The effect on fish populations cannot be quantified here and results for macroinvertebrates must be read with caution as a result of discrepancies in sample data between the two sampling periods, as discussed in Section 4.3. However, Figures 4.7 and 4.8 attempt to show that there may be a relationship between flow (flashiness) and macroinvertebrate richness/diversity as a result of channel impoundment. As previously discussed, there is a general decrease in flashiness of the water following impoundment and an increase in richness/diversity and number of species which are more sensitive to poor water quality (possibly due to a decrease in fine sediment load as a result of impoundment, which will cause water to be clearer). As Figure 4.7 and 4.8 show, this can be seen as a trend over time, although attributing one factor to another in a specific year is more difficult (and comparisons should not be made to specific years as macroinvertebrates take time to adjust to changes in flow conditions and it would take at least one year to detect a response to a change in flow characteristic within a sample).

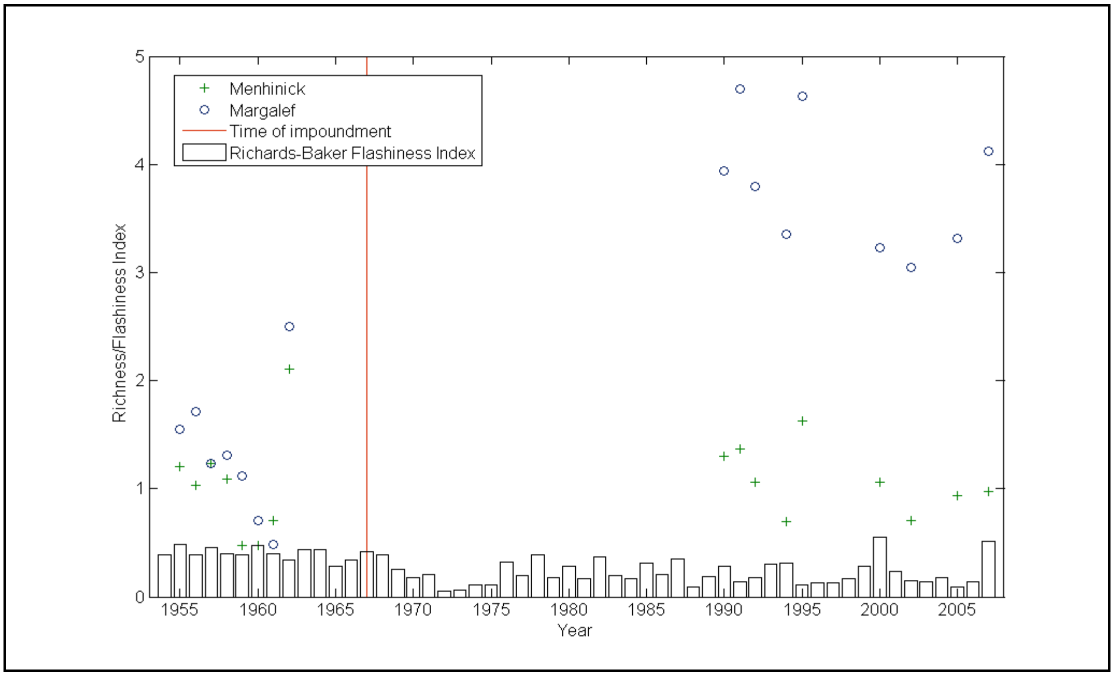

Figure 4.7 Coupling of richness indices and flashiness results to identify relationship between changes in flow and changes in macroinvertebrate populations

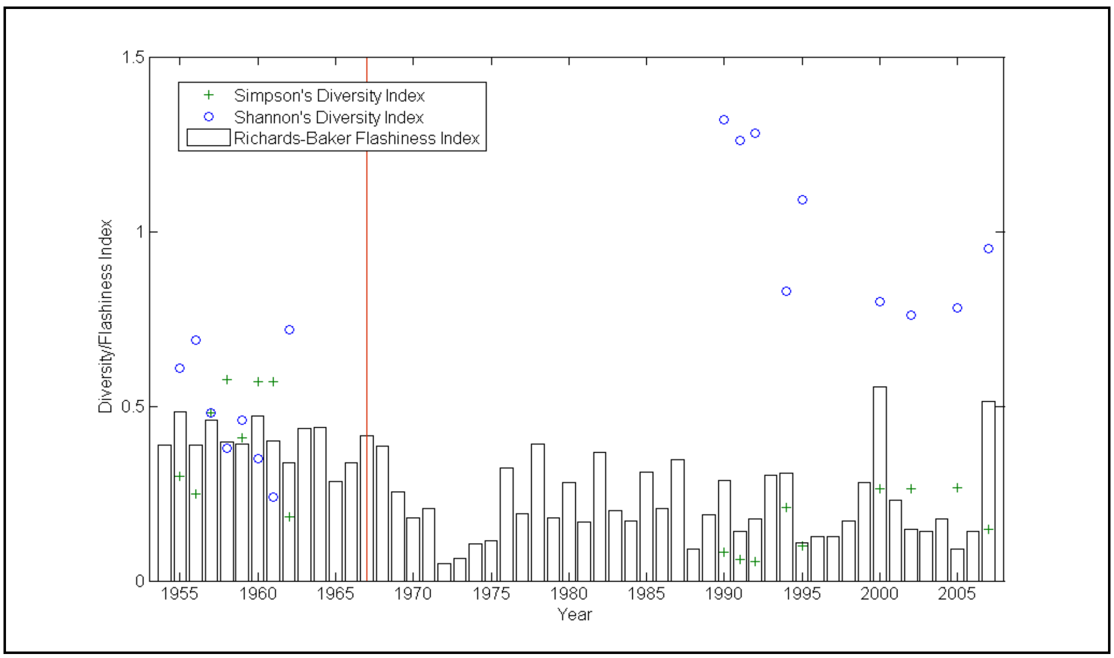

Figure 4.8 Coupling of diversity index and flashiness results to identify relationship between changes in flow and changes in macroinvertebrate populations

Table 4.6 shows the Mann-Whitney U test results for the comparison of the pre- and post-impoundment flashiness and richness/diversity indices. The flashiness values are statistically different between the pre- and post-impoundment periods. For all but the Menhinick analyses, the data suggest that there is a significant difference between the pre- and post-impoundment richness and diversity of macroinvertebrates in the River Derwent. Correlation analysis was used to determine whether there is a relationship between the change in flashiness and the change in richness/diversity. The results are summarised in Table 4.7. The correlation analysis shows that pre-impoundment, there may be a positive correlation between flashiness of flow and richness /diversity, but the post-impoundment figures suggest that there is a weak negative correlation between flashiness and richness/diversity (except when using the Simpson index, which is the inverse of the Shannon index, an increase in Simpson D value denotes a *decrease* in diversity). This denotes that there has been a change in the populations but that it is not necessarily as a result of changes in flow. However, none of the analyses were significant at the 95% confidence level and therefore a relationship does not exist. The absence of a relationship may be as much because the ecology will not be adjusted to flashiness on an annual scale (at which the flashiness index is calculated) than because of any process response (or lack of) to the change in longer term flow regimes.

	Flashiness	**Annual range**	**Menhinick**	**Margalef**	**Simpson**	**Shannon**	**Simple count**
Lowest Mann-Whitney U score	47	81	33	0	7	0	2
Critical value, based on number of samples (at 0.05 confidence level)	83	83	15	15	15	15	15
Result	Calculated value < than critical value, can reject H_0. **Samples *are* significantly different at 0.05 confidence level.**	Calculated value > than critical value, cannot reject H_0. **Samples *are not* significantly different at 0.05 confidence level.**	Calculated value > than critical value, cannot reject H_0. **Samples *are not* significantly different at 0.05 confidence level.**	Calculated value < than critical value, can reject H_0. **Samples *are* significantly different at 0.05 confidence level.**	Calculated value < than critical value, can reject H_0. **Samples *are* significantly different at 0.05 confidence level.**	Calculated value < than critical value, can reject H_0. **Samples *are* significantly different at 0.05 confidence level.**	Calculated value < than critical value, can reject H_0. **Samples *are* significantly different at 0.05 confidence level.**

Table 4.6 Mann-Whitney U Test results to determine statistically significant difference between pre- and post-impoundment richness and diversity scores

Relationship between:		Menhinick	Margalef	Simpson	Shannon	Simple Count
Flashiness	Pre-impoundment Correlation coefficient (*P value*)	0.41 (*0.31*)	0.48 (*0.23*)	0.37 (*0.36*)	0.27 (*0.51*)	0.27 (*0.52*)
	Post-impoundment Correlation coefficient (*P value*)	-0.22 (*0.56*)	-0.20 (*0.60*)	0.22 (*0.57*)	-0.23 (*0.55*)	-0.11 (*0.77*)
Annual range	Pre-impoundment Correlation coefficient (*P value*)	-0.42 (*0.28*)	-0.26 (*0.54*)	0.18 (*0.68*)	-0.12 (*0.77*)	-0.13 (*0.76*)
	Post-impoundment Correlation coefficient (*P value*)	0.09 (*0.82*)	-0.26 (*0.50*)	0.41 (*0.26*)	-0.36 (*0.35*)	-0.51 (*0.16*)

Table 4.7 Quantification of relationship (correlation coefficients) between flow characteristics and diversity, with significance value of each correlation: *P*=<0.05=significant at the 95% confidence level

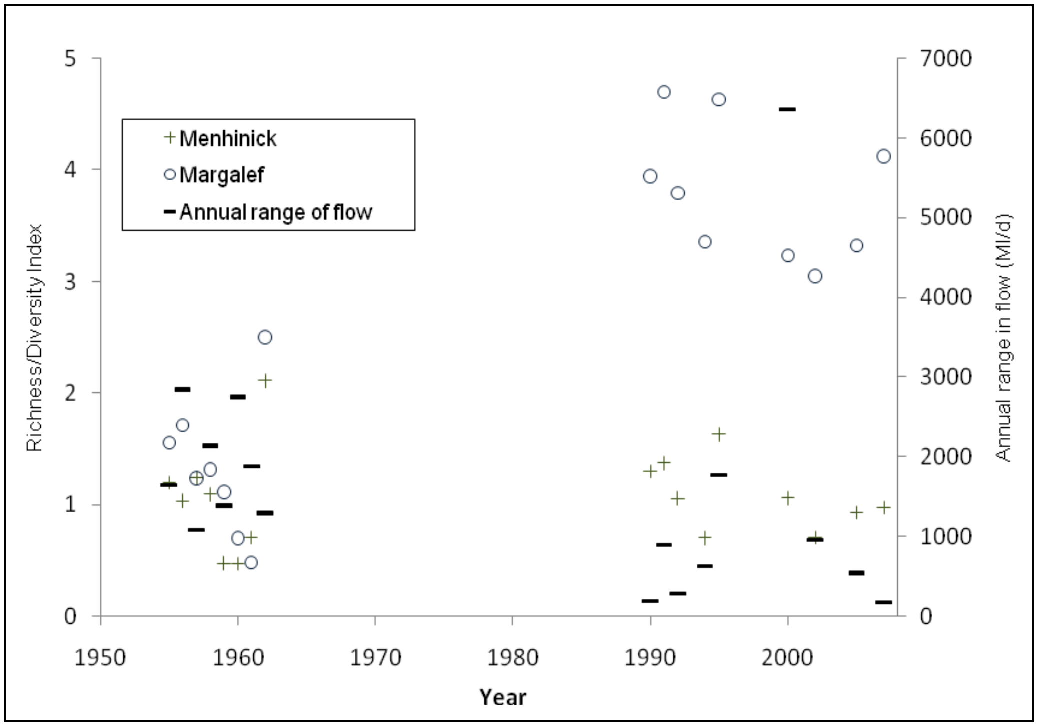

Figure 4.9 Comparison of richness indices with range of flows. Flow values taken from gauging station at Eddy's Bridge, invertebrate samples taken at Allensford

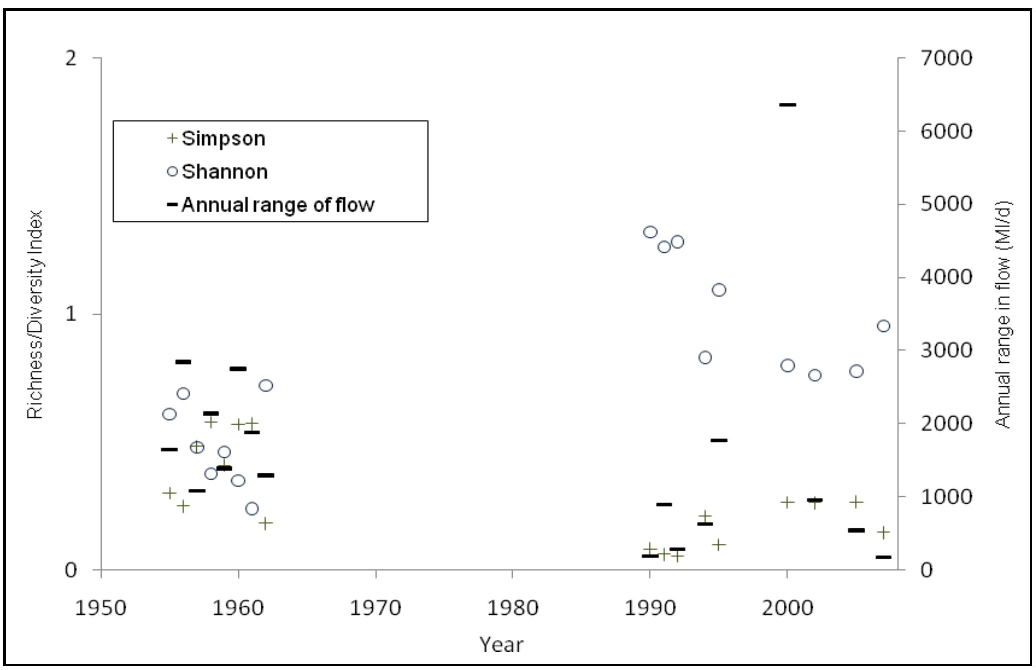

Figure 4.10 Comparison of diversity indices with range of flows. Flow values taken from gauging station at Eddy's Bridge, invertebrate samples taken at Allensford

Figures 4.9 and 4.10 illustrate the change in difference between maximum and minimum annual flows pre- and post impoundment. Overall, the range has fallen post impoundment (i.e. the difference between annual maximum and annual minimum is lower (Chapter Three). The diversity and richness values have been plotted alongside the annual range in flow data to allow a visual identification of any relationships that may occur. As range in annual flows has decreased following impoundment, variation in diversity and richness values appears to have increased. However, according to the correlation values in Table 4.7, range of annual flow (i.e. difference between maximum and minimum flows for each year) has little impact on diversity/richness. While a weak negative correlation coefficient is produced between most richness/diversity indices and annual ranges of flows, none of the tests are significant to the 95% confidence level. Table 4.6 shows that the range of flows pre- and post-impoundment are not statistically different according to the Mann-Whitney U test.

4.4.3 Fish

As discussed in Section 4.3, there are limited numerical data available to define the ability of the River Derwent to support fish populations. What Figure 4.11 does show is that brown trout have been found in the river for a number of years. An absence of data on the graph does not necessarily suggest an absence of brown trout as the results are heavily dependent on subjective choices made by the sampler concerning when and where to sample. It is known from Environment Agency sources and anecdotal data (primarily from members of the Upper Derwent Angling Association) that brown trout, and grayling both inhabit the River Derwent and have done so for many years. Atlantic salmon, however, are not present as they are unable to negotiate the larger weirs, such as that at Derwent Haugh.

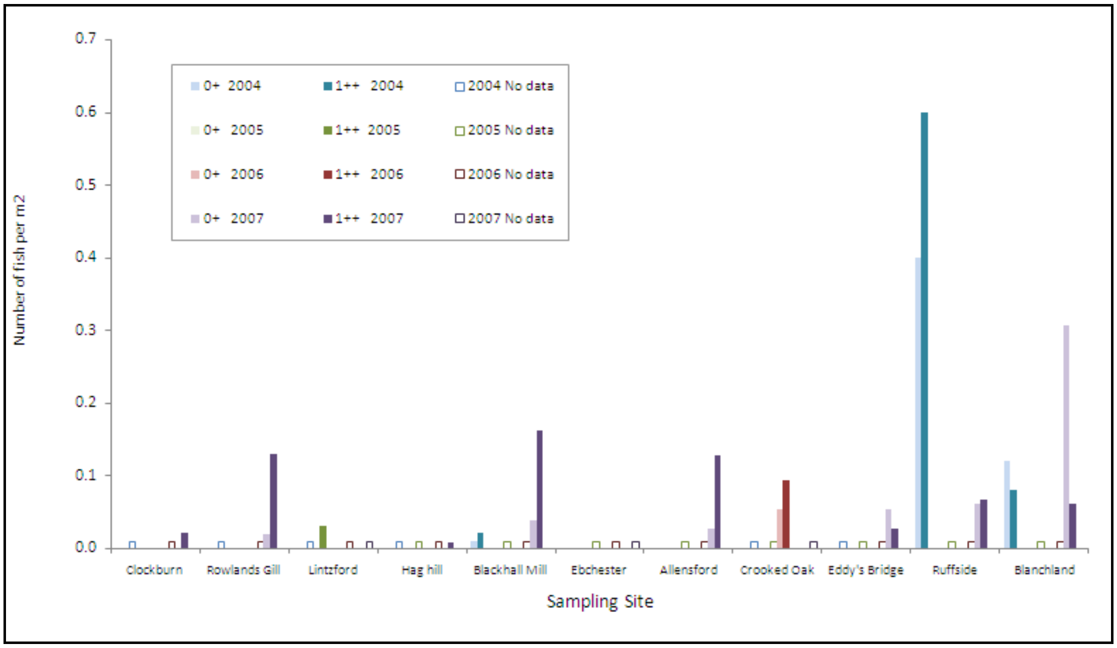

Figure 4.11 Brown trout counts in the River Derwent between 2004 and 2007, starting with the most downstream site at the left hand side of the figure, moving upstream towards the right

4.5 Discussion

Chapter Three demonstrated that there have been changes in flow characteristics following impoundment. Previous Sections of the present Chapter have illustrated the change in macroinvertebrate population dynamics between the pre- and post-impoundment periods, and compared the changes between flow and macroinvertebrate communities. The following Section will attempt to explain the trends found and explore the impacts that changes in flow may have on macroinvertebrate communities. Some consideration will also be given to brown trout populations and the impact of flow on them.

4.5.1 Macroinvertebrates

As reported in Section 4.4, there is a significant difference between the ASPT scores in the pre- and post-impoundment periods. It is evident that ASPT scores for Allensford, on the River Derwent, increase following channel impoundment. Whether they have increased as a *result* of impoundment is less apparent and this relationship will be considered further, below. According to Armstrong *et al.,* (1983), high ASPT values usually characterise clean, upland sites where there are many taxa with high BMWP scores. A high BMWP score is given to pollution-intolerant families, while pollution tolerant families receive lower BMWP scores. Therefore, an increase in ASPT value indicates an increased concentration of pollution-intolerant families at Allensford, in recent years. This would suggest that there has been an improvement in water quality between the two periods of data collection. It is possible that an improvement in water quality may be a result of river regulation. An increase in baseflow would guarantee a minimum level of pollutant dilution, (especially from the water treatment works discharge into the River Derwent at Mosswood, 1 km upstream of Allensford) and would decrease the fine sediment concentration at low flows, which would also create a more favourable habitat for macroinvertebrates. However, there are no data to link the increase in ASPT directly to a change in water quality, or a change in water quality to the change in flow regime. Consequently, a relationship between the new flow regime and ASPT value may be inferred here, but not quantified.

There has been no significant change in LIFE flow scores between the pre- and post-impoundment periods. Because LIFE scores are used to indicate flow requirements of macroinvertebrates (higher LIFE scores are assigned to Groups that prefer fast flowing rivers, Extence *et al.,* 1999), this finding could lead to the conclusion that there has been no change in flow pattern between the time periods. However, as demonstrated in Chapter Three, there has been a change in flow regime with a marked decrease in flow variability and extremes. The reason for this could be that the flow velocity is/was not the limiting factor in macroinvertebrate population structure and that something else has caused the change. However, there are no data to support this theory and so consideration must be given to what LIFE scores can actually tell us. All of the families found at Allensford, both pre- and post-impoundment, fall into three of the six Flow Classification Groups proposed by Extence *et al.,* (1999). The six Groups are shown in Table 4.8 and the Groups represented at Allensford are highlighted. Figure 4.12 shows the distribution of families from each Group for each year of study.

Group	Ecological flow association	Mean current velocity
I	Taxa primarily associated with rapid flows	Typically >100 cm/s^{-1}
II	Taxa primarily associated with moderate to fast flows	Typically 20-100 cm/s^{-1}
III	Taxa primarily associated with slow or sluggish flows	Typically <20 cm/s^{-1}
IV	Taxa primarily associated with flowing (usually slow) and standing waters	-
V	Taxa primarily associated with standing waters	-
VI	Taxa primarily associated with drying or drought impacted sites	-

Table 4.8 Benthic freshwater macroinvertebrate Flow Groups, those found at Allensford are highlighted (After Extence *et al.,* 1999)

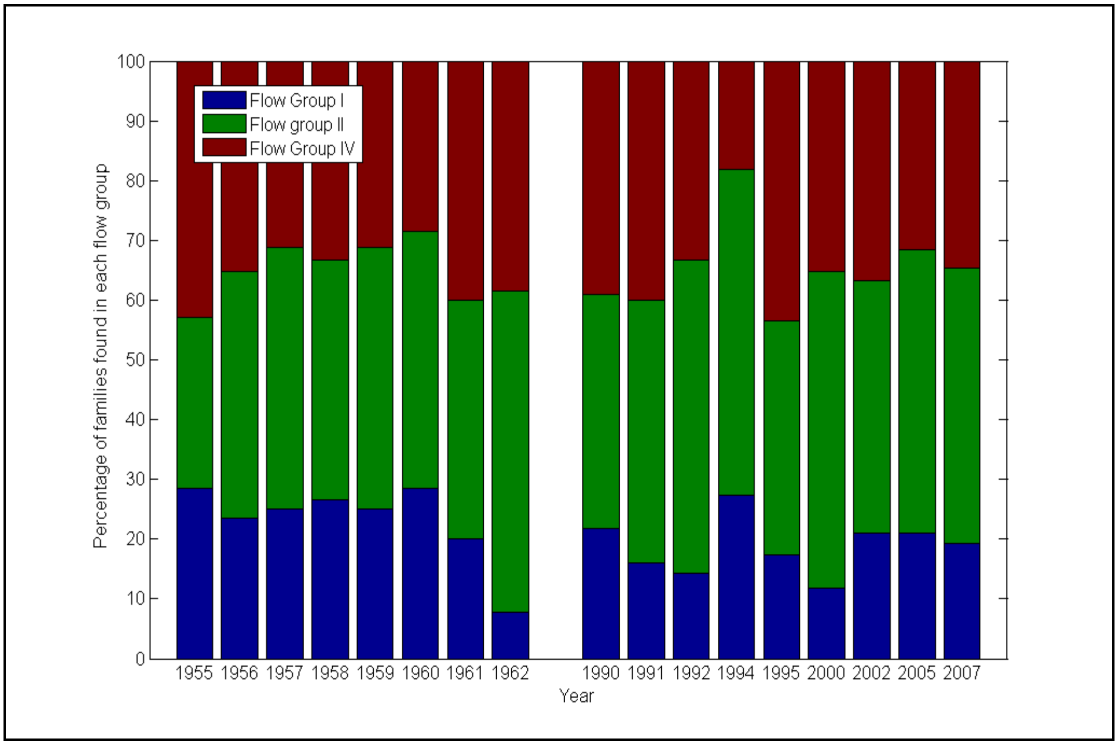

Figure 4.12 Proportion of families in each Flow Group at Allensford, represented as a percentage of all the families identified (note that years are not continuous – only sampled years are shown)

Table 4.8 and Figure 4.12 show that only Flow Groups I, II and IV are represented at Allensford. Groups I and II include macroinvertebrates that require relatively fast flows, while Group IV includes macroinvertebrates that live in slow and standing water. The distribution of macroinvertebrates from Groups II and IV is relatively even, with slightly fewer from Group I in each of the years for which there are data (Figure 4.12). This suggests that the range of flows experienced in the River Derwent at Allensford, at the time of sampling, is large and can accommodate a range of macroinvertebrates. It should be noted here that most of the families found in Group IV are classed as 'containing species/genera with variable flow requirements' (Extence *et al.,* 1999) and therefore the description for conditions required in Group IV cannot be taken as a definitive description of the type of creature found. The conditions described for Group III (slow or sluggish flows) are likely to be only found, for a river, in the most downstream reaches with low gradients, or in isolated areas which are not fully incorporated into the channel flow. Categories IV, V and VI are rare in most rivers and particularly in upland streams. Therefore, it can be assumed that most macroinvertebrates that would be found in any upland stream, will be categorised as Group I or II. This, therefore, means that investigating flow characteristics at the scale of this project (i.e. changes in flow regime and extremes, but at the same site in the same river) is too fine a scale for LIFE scores to be informative. What the scores do indicate is that flow has not changed from rapid/fast flows to slow, sluggish or standing flows, as a result of impoundment because the representation from each Group is consistent both before and after impoundment.

Number of taxa has increased since impoundment (Figure 4.4). Table 4.3 shows that there is a significant difference in the counts between the two time periods and this is reflected in the richness and diversity scores (Figures 4.5 and 4.6). Species richness (quantified through the Margalef and Menhinick indices) can offer a simple and clear expression of diversity and has been employed by a number of researchers including Abbott (1974); Karydis and Tsirtsis (1996); Saldana and Ibanez (2004).

The Margalef, Menhinick, Simpson and Shannon indices, and the simple count, all denote the same trend: an increase in diversity post-impoundment. This is supported by both the scores for each individual year and the mean/median scores for each time period. As demonstrated in Section 4.4, this increase in diversity may be attributed to an increase in flow stability which has occurred as a result of river regulation. It has been suggested that a change in flow regime can alter the ecological composition of a stream (Death and Winterbourn, 1981; McElravy *et al.,* 1989; Jowett and Duncan, 1990). However, the conditions which exist within that stream before the change takes place, will play a role in determining the response of the ecology. Connell (1978) suggests that a community is never at an equilibrium state and that there are a number of stages, which often occur on a cycle. A habitat which is frequently disturbed may have low species diversity as only the most resilient species can recover from mortality. At low rates of disturbance, diversity is reduced by competitive exclusion (Connell, 1978). It is, therefore, possible that the original, flashy, nature of the River Derwent was suitable for only these most resilient species. A reduction in flashiness and frequency of extreme events may have created a habitat in which a wider range of species may be sustained. If stability of flow was to increase further (for example, if there were fewer inputs from unregulated tributaries and runoff) and flow rates became more constant, it could, according to Connell's hypothesis, lead to a *reduction* in diversity, as more dominant species eliminate weaker competitors. Figure 4.13 illustrates how this process may take place. The results from this study suggest that the River Derwent is currently at the peak of the curve (Figure 4.13).

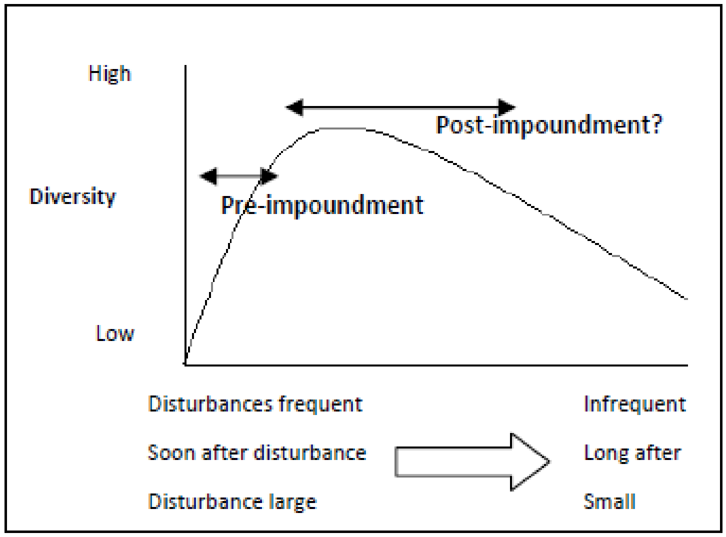

Figure 4.13 State of diversity in the River Derwent following impoundment (after Connell, 1978)

Huston (1994) presents a new conceptual model, the dynamic equilibrium theory, which builds upon Connell's intermediate disturbance hypothesis, but expands upon the parameters which may affect diversity. Huston agrees with the foundations of Connell's model – that frequency and magnitude of disturbance have an effect on diversity, but proposes that there are influences on diversity beyond disturbance – local influences which will determine the product of changes in disturbance rates (such as growth rate as well as mortality, reproduction failure and local extinction). He suggests that the same change in frequency or intensity of disturbance can have very different effects on diversity, depending on the current state of competitive displacement. For example, if rates of competitive displacement are low, an increasing disturbance frequency will result in a monotonically decreasing pattern of diversity. High rates of competitive displacement would cause an increase in disturbance frequency to result in an increase in diversity, as the more competitive species are removed through disturbance, allowing other species to grow. Rate of growth also plays an important role. If a species grows rapidly, then it is able to recover quickly and be more resilient to frequent disturbances. Therefore, the effect of a disturbance regime on the extinction rate depends on the growth rate of the population (Huston, 1994). In other words, it is the initial conditions and the context in which the change occurs that determine the response of the community. This means that, at Allensford, the initial conditions will have played a role in the way the macroinvertebrate population responded to the changes in flow regime. Because diversity increased following impoundment, it may be assumed that competitive displacement was high prior to impoundment. Unfortunately, growth rates and other local factors are unknown for both pre- and post-impoundment periods. Therefore, only assumptions can be made about which, if any, of the hypotheses are being supported here.

The pattern of lower diversity in the more unstable pre-impoundment period is concurrent with the findings of a number of authors (Death and Winterbourn., 1981; Clifford, 1982; Doeg *et al.,* 1989; McElravy *et al.,* 1989; Scrimgeour and Winterbourn, 1989; Jowett and Duncan, 1990). Death and Winterbourn (1995) found that species evenness (which can be equated with diversity, Magurran, 1988), peaked at times of intermediate stability. This has been mirrored in the current study because, while flows are now more stable than prior to impoundment, there are still fluctuations in flow (due to input from tributaries, rainfall and reservoir overtopping. Chapter Three). Death and Winterbourn (1995) and McCabe and Gotelli (2000) found that *richness* peaked in the most stable environments, which may explain why the increase in richness is less pronounced than the increase in diversity. Townsend *et al.,* (1997) correlated species richness with bed stability and presence of refugia. Dead space and large substratum is more frequently found in areas of stability and it affords shelter in which invertebrates can grow and be buffered from disturbances. The increased stability at Allensford is likely to have provided a more stable river bed with areas of refugia.

There have been investigations in which disturbance frequency has been found to have no impact on species diversity (e.g. Reice, 1985; Rader *et al.,* 2008). It must therefore be considered that disturbance frequency/intensity is not the only control on macroinvertebrate communities (as expressed by Huston, 1994) and that other factors will have an influence on population dynamics. In terms of river regulation and impoundment, there are changes to hydrological, morphological and physical aspects of the river aside from disturbance rate. Volume of flow, wetted area and temperature are all major controls on macroinvertebrate sustainability, as discussed in Section 4.2. Section 4.2 highlights the importance of changes in wetted area for colonisation of invertebrates. Section 4.4 shows that following impoundment there has been a reduction in the total volume of water flowing through the channel (which would cause an overall reduction in wetted area and increase in temperature, Extence, 1981)). This would result in a change to macroinvertebrate population structure and, according to Extence (1981) an overall increase in diversity and species abundance. It can be postulated that this is one of the reasons that overall diversity and richness has increased at Allensford. However, at this stage, changes in wetted area and temperature cannot be quantified but wetted area will be addressed in more detail in Chapter Six. As previously discussed, high flows and disturbance events brought about by such flows lead to the death and decline of individuals and communities (Poff and Ward, 1989; Jowett and Duncan, 1990; Clausen and Biggs, 1997). The reduction of high flows as a consequence of river regulation will have caused a decline in the frequency of the 'reset mechanisms' proposed by Poff and Ward (1989). This will have rendered the habitat at Allensford favourable to a greater number of species because there is less displacement and more time for recovery and growth (Allan, 1995).

Sediment size distribution data do not exist for pre- and post-impoundment periods and therefore the impacts of Derwent Reservoir on the sediment of the River Derwent cannot be quantified. However, Chapter Three shows that there is a general agreement that impoundment may cause an increase in the *proportion* of fine sediment (all sediment sizes are trapped by the reservoir, but fine sediment is more readily delivered to the channel from other sources such as the banks and in runoff) and armouring of the gravel layer (Milhous, 1982; Petts, 1984, Gilvear *et al.,* 2002; Isik *et al.,* 2008). There is much evidence to suggest that river beds with elevated amounts of sand and silt are detrimental to macroinvertebrates (Extence, 1981; Quinn and Hickey, 1990; Wood and Armitage, 1997). Therefore, because species richness and diversity has increased, it may be concluded that for the Derwent at Allensford: a) sediment size composition has less impact on species dynamics than other factors of flow such as magnitude and disturbance rates; or b) that the effects of impoundment on the sediment at this point have been counteracted by inputs from tributaries. Petts (1980, 1984) and Jorde *et al.,* (2008) write of the new-found stability in the substrate that occurs in the years following impoundment. They attribute this to the inputs of tributaries and the reduced carrying capacity of the river under the new flow regime. It may be that the River Derwent has now reached a new steady state in terms of sediment size composition and that the current situation supports a wide range of invertebrate species.

Rader *et al.,* (2008) found that temperature had a stronger control on biodiversity than the frequency of floods and that the effect that reservoir installation has on water temperatures may outweigh the effects on and changes caused by new disturbance frequencies and intensities. Therefore, it must be concluded that all components of the ecosystem that impact upon macroinvertebrate community structure must be considered before a final conclusion can be drawn on the effect that a change in each one may have.

Although the macroinvertebrate population of the River Derwent appears to have been altered by the effects of impoundment, this does not necessarily mean that the population existing in the Derwent is not a good one. Indeed, when compared to the control site at Allenbanks on the River Allen, there was no significant difference in any of the scores used to quantify populations (ASPT, NTAXA, LIFE). Data for Allenbanks do not exist for the 1955-1962 period and therefore a change over time cannot be proved or disproved, but the findings of this study suggest that the macroinvertebrate population at Allensford is comparable to the populations of a similar but unregulated river which is considered by the Environment Agency to be of ‘very good’ ecological quality (EA Tyne CAMS, 2005).

4.5.2 Fish

Despite the limited fish data for the Derwent, Figure 4.11 is able to tell us that in the face of the regulation and increased uniformity of flows on the River Derwent, brown trout are able to live in the river, at all sites sampled. It may be expected that fish populations at sites such as Eddy's Bridge, Crooked Oak and Allensford (Figure 4.2) would be most severely affected by impoundment, being located nearest to the point of release and thus having lower levels of input from tributaries and runoff. However, according to Figure 4.11, brown trout do populate these sites, and not in significantly lower densities than at other sites. It may be, then, that the conditions created by the regulated flows, at these sites, are favourable to brown trout or that brown trout are resilient to changes in flow depth and velocity and can adapt to a range of habitat conditions. It is known that fish have 'preferred' habitat conditions and 'usable' habitat conditions (Shirvell and Dungey, 1983; Mitchell *et al.,* 1998; Armstrong *et al.,* 2003). Fish are territorial and often return to the same place in the river each year, even if this means living in less than favourable conditions. If the flow regime of the River Derwent is less favourable now than it was prior to impoundment, the change may not be significant enough to have displaced species which have a broad tolerance of habitat conditions. It is also possible that the effects of the reservoir have been sufficiently diluted at all of the downstream sampling sites as the main tributary of the River Derwent, Burnhope Burn, lies upstream of the sampling point nearest to the reservoir outlet (Eddy's Bridge). Figure 4.14 shows how an ergodic approach can be used to substitute space for time. If distance from the point of release can be considered concurrent with time since release (or indeed downstream sites could be considered synonymous with pre-impoundemnt sites because the flow regime is largely natural), Figure 4.14 shows that distance from and time since, the compensation release have little impact on brown trout populations in the River Derwent because there is no pattern between the distance from the point of release and the number of fish observed. It may be, then, that the local morphology of the channel is more important than a change of flow at that point. The sample sites upstream of the reservoir (Ruffside, Blanchland) have similar morphology and sediment size characteristics as the sites just downstream of the reservoir (Eddy's Bridge, Crooked Oak, Allensford). If the sites that are unaffected by impoundment are considered to represent the downstream sites (Eddy's Bridge, Crooked Oak, Allensford) as they were before the channel was impounded, then Figure 4.14 would suggest that impoundment has caused a decrease in brown trout populations.

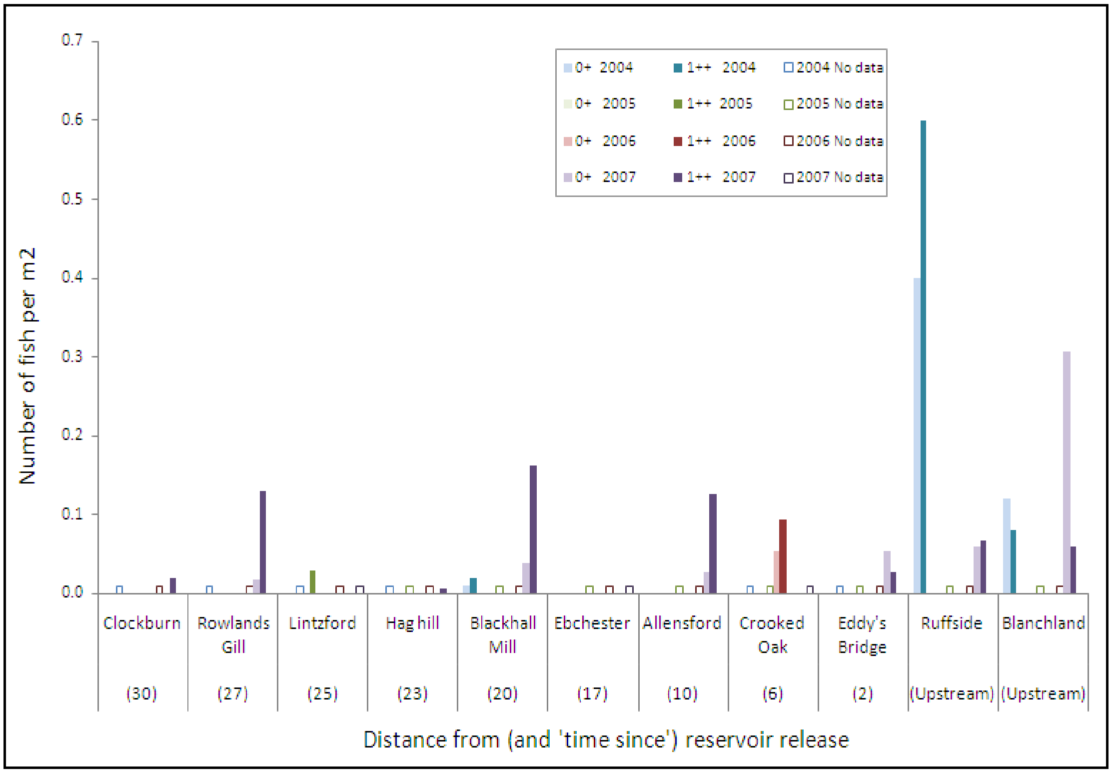

Figure 4.14 Brown trout counts with distance from release as a substitute for time. Distance from release(km) is included in ()

4.6 Conclusion

This Chapter aimed to quantify and to assess the changes in ecology in the River Derwent since impoundment occurred in 1966. It has been shown that there has been an increase in richness and diversity of macroinvertebrates since impoundment but that the interpretation of some scores (e.g. LIFE) should be taken with caution. Other environmental factors such as water temperature and competition must be considered when assessing the impacts of flow on ecology.

The next Chapter will consider the flow requirements of fish species and aim to propose compensation release regimes that support the different life stages of anadromous fish in the River Derwent. It is hoped that this will answer some of the questions created through analysis in this Chapter.

Chapter Five

Redesigning compensation flows

5.1. Introduction

Following the identification of the impacts of impoundment on the River Derwent in Chapters Three and Four, this chapter seeks to design new compensation flow release regimes in order to benefit the stream ecology through increased habitat availability and connectivity. A simple model, Compensation Release Assessment at the Broad Scale (CRAB) is employed to consider demands on water resources from the River Derwent along with depth and velocity requirements of three key fish species (brown trout, grayling and Atlantic salmon). Although the CRAB methodology has been applied previously (Mould, 2006), its application was limited to one species (brown trout). In this project, application of the model has been taken a step further by considering the flow requirements of three fish species (brown trout, grayling and Atlantic salmon) for the same river. Results are used to inform the methodology of Chapter Six (CRAM: Compensation Release Assessment at the Meso-scale) as well as provide an indication of the types of flow regime required for the ecology of the River Derwent.

5.2. Compensation Release Assessment at the Broad Scale (CRAB)

Compensation Release Assessment at the Broad Scale (CRAB) is a Durham-developed method of assessing how an annual compensation release volume could be redesigned on an annual basis in order to deliver specific discharge per unit width required by instream populations for optimum habitat availability and connectivity. It has been applied in a previous study into the investigation of reservoir compensation redesign in Yorkshire (Mould, 2006; Lane *et al*., 2006).

5.2.1. Theoretical basis of CRAB

The focus of the model is to consider current flow conditions, water resource requirements, and flow requirements at different times of the year, according to life stages of certain riverine species. The output of the model is an annual distribution of compensation release rates which include short term spates and longer term (1 to 3 month) elevations in flow as well as baseflow levels for the rest of the year, in order to introduce elements of the natural features of the hydrograph which are absent under current flow regimes.

The model is developed around a number of assumptions, as outlined by Mould (2006):

1. Habitat availability is not directly controlled by depth and velocity distributions, but these distributions are a result of discharge and its interaction with the shape of the river bed, its perimeter sedimentology (which at a very high resolution controls a component of the shape of

the river bed), and the delivery of sediment upstream. In turn, these influence the spatial patterns of flow velocity and depth as well as temperature and other variables of ecological significance. In order to predict channel responses to changes in discharge, a model must be able to account for these interactions (Lane *et al*., 2007). However, CRAB is not a predictive tool, rather a prescriptive one which can be used to help regulators decide *how* compensation flows should be redesigned in order to affect these interactions. The process should be informed by basic ecological information.

2. A width scaling (discharge per unit width) is used as the magnitude of flow release in order to allow comparison of smaller and larger streams because width and discharge are linearly related.

3. Organism-specific minimum and maximum requirements will exist for monthly defined habitat needs. CRAB allows these requirements to be met through three settings (spate flow, period of elevated baseflow, period of low baseflow) by specifying minimum velocity and depth requirements and using these to set minimum flows per unit width. This is a major approximation of instream flow needs for a number of reasons. First, a number of different velocity-width combinations can result in the same discharge per unit width, but not necessarily the same habitat conditions. Second, other processes such as stream temperature and sedimentation are of particular biological significance, but very difficult to model. Therefore, CRAB follows traditional habitat preference curve analysis in focusing on velocity and depth and has minimal recognition of biological understanding of instream ecological processes. This is one of the main the reasons why CRAB is not predictive, but a tool to aid in the redesign of compensation flows. There may be cases when CRAB suggests that there is insufficient flow to support a certain species' requirements, when in fact the species may have been inhabiting the river for some time (e.g. as with grayling on the River Derwent). In such cases, the spawning flow period can be set as 'sufficient' and flows altered in the rest of the year to reach a more realistic flow regime. This has been explored in section 5.4.2. Third, for the case of the spate flow, a warning is included when the proposed rate of change of flow is likely to be greater than is recommended. Fourth, when setting minimum velocity and depth requirements, consultation with biological experts is essential.

4. Response of new flows will depend upon the partitioning of flow changes between width, depth and velocity. CRAB does not have the ability to account for this as this would have high modelling and data requirements, which would detract from the simple tool it is designed to be.

5. There will always be limitations to the level of changes that can be made to flows. These include water licensed for abstraction downstream, legal requirements on volumes to be released and mechanical issues such as size of release pipe and dam safety. These are factored into the analysis by CRAB.

6. Inputs from overtopping, runoff and unregulated tributaries may drive the ecosystem response, masking the changes that occur due to changed compensation release. Magnitude and frequency of overtopping events may also be altered by changed compensation regime. This should be considered as timing and magnitude of these events may have a significant impact on stream ecology.

CRAB provides a broad scale assessment of flows. It is not a predictive tool. Rather it can be used to inform the user of how a total annual release requirement can be altered to reflect a more ‘natural’ or varied flow pattern. It works by taking a set annual release and apportioning amounts to each month, while considering information on abstraction demands, ecological flow demands and the shape of the river. It also includes a spate flow, to be run on one day, if required by the species and catchment in question. Once recommended flows have been obtained, settings such as scale of compensation flow and suitability of current actual flow can be altered to gain the required flow regime. The data required to run the model process are summarised in Table 5.1

Data	Purpose	Source
Current compensation release	To set a baseline from which changes/improvements in flow can be assessed and to examine current suitability of flow.	Current release rates were obtained from the 1957 Water Order for the River Derwent.
Changes to existing flows	To test and evaluate scenarios through CRAB.	Based on habitat requirements of species, from literature.
Current demands on river	So that requirements from current demands such as abstraction licences can be met.	EA Tyne CAMS (Catchment Abstraction Management Strategy) abstraction licence database.
Capability of reservoir to increase / decrease flows	To account for physical limitations on amount of water that can released at any one time (e.g. amount that can flow through release pipes or spate valves).	Advised by NWL technical staff.
Ecological data – habitat preferences of species of interest	Usually based on depth and velocity preferences, they are used to design suitable flows per unit width for different life stages (e.g. spawning, migration) of different species.	From current literature and EA fisheries experts. Fish were assessed in this method rather than macroinvertebrates because habitat preference data is more readily available. Brown trout, grayling and Atlantic salmon were investigated.
Morphology	The width of the river is required to indicate how volume of release relates to discharge in a given reach.	Only one width can be applied to CRAB. Therefore, a mean was taken from widths measured at upstream, midstream and downstream locations.

Table 5.1 Data required for the application of the CRAB methodology to a river or stream

5.3. Literature review – flow requirements

There is a wealth of literature prescribing flow depth and velocity requirements for brown trout, for all age groups and in a number of stream types. Atlantic salmon spawn at a similar time of year to brown trout and the two species tend to be considered together. Grayling, however, has been studied less and suggestions for depth and velocity requirements, particularly at the juvenile stage, are less well documented. Environment Agency official documents were used to supplement the findings from the literature review as they are UK focused, unlike many of the published articles that were obtained. Results were discussed with Environment Agency fisheries experts before being applied to the model. The flow requirement results for each of the three species investigated (brown trout, grayling and Atlantic salmon) are summarised in the following Sections. Flow requirements were investigated for fish in their spawning season because a change in flow level is often a trigger for spawning. Fish have often been observed using different levels of flow when spawning to the rest of the life cycle. Flow depths and velocities required for the rest of the year were based on observations of juveniles because these flows are more achievable than flows utilised by adult fish but still enable the user to reconsider one part of the ecology.

5.3.1 Brown trout

In the UK, brown trout generally spawn in the autumn, between September and November (Degerman *et al*., 2000). The depth and velocity requirements listed in Tables 5.2 and 5.3 are based on observations from rivers over a large area including the UK and Europe. Habitat preferences varied slightly with location but using a number of examples from different locations allows the habitat values to be transferable to other river systems. An average of all of the habitat preference ranges was calculated and any extremes (or case studies which were particularly different to the present project) were excluded. The depth and velocity requirements used in the CRAB model are summarised in Table 5.4.

Source	Depth requirements (m)	Velocity requirements (ms^{-1})
Shirvell and Dungey (1983)	0.6-0.82	0.15-0.75
Witzel and MacCrimmon (1983)	0.25	0.11-0.80
Crisp and Carling (1989)	-	0.15-0.20
Johnson *et al*., (1995)	0.20-0.50	0.20-0.40
Nelson (1986)	0.18-0.46	0.35-0.95
Grost *et al*., (1990)	0.12-0.18	0.24-0.37
Ottaway *et al*., (1981)	-	0.30-0.40
Raleigh *et al*., (1986)	>0.20	0.20-0.70

Table 5.2 Depth and velocity requirements for spawning brown trout

Source	Depth requirements (m)	Velocity requirements (ms^{-1})
Shirvell and Dungey (1983)	0.14-1.22	0-0.65
Maki-Petays *et al*., (1997)	0.40-0.75	-
Heggennes (1988)	>0.50	0.10-0.70
Bird *et al*., (1995)	-	<0.20
Baldes and Vincent (1969)	-	<0.20
Bachman (1984)	-	<0.20
Belaud *et al*., (1989)	0.30	<0.40 but 0 preferred
Bovee (1978)	<0.90	<0.50 but 0-0.10 preferred
Heggennes and Saltveit (1990)	0.30-0.60	0.05-0.30
Johnson *et al*., (1995)	0.25-0.55	0.15-0.60
Loar (1985)	0.30	<0.30 but 0 preferred
Moyle *et al*., (1983)	0.30	0-0.50
Shuler *et al*., (1994)	0.27-0.57	0.09-0.45 by day, 0.03-0.45 by night
Faush (1984)	-	<0.20

Table 5.3 Depth and velocity requirements for juvenile brown trout

	Depth (m)	**Velocity (ms^{-1})**
Spawning (3 months)	0.26	0.21
Juvenile (9 months)	0.31 (0.37 if 'maximum' value is included)	0.04 (0.09 if 'maximum' value is included)

Table 5.4 Depth and velocity requirements chosen for use in CRAB model when applied to brown trout

5.3.2 Grayling

Unlike brown trout (and Atlantic salmon), grayling tend to spawn in the spring (Varly, 1967, Woolland, 1972), and have higher depth and velocity requirements, as shown in Tables 5.5 and 5.6. Grayling require much greater flow depths in the time that they are not spawning (i.e. for around 9-10 months of the year). Much of the literature provided flow requirements for adult, rather than juvenile, grayling. This is highlighted in Table 5.6**.** It is possible that this may result in an over-estimation of depth and flow requirements. An average of all of the habitat preference ranges was calculated and any results for case studies which were particularly different to the present project were excluded. The depth and velocity requirements used in the CRAB model are summarised in Table 5.7.

Source	Depth requirements (m)	Velocity requirements (ms^{-1})
Beauchamp (1990)	>0.25	max 0.21
Sempeski and Gaudin (1995)	0.10-0.40	0.25-0.91
Darchambeau and Poncin (1998)	0.20-0.55	-
Nykanen *et al*., (2004)	>0.60	0. 40-0.70
Parkinson *et al*., (1999)	0.20-0.80	0. 25-1.0
Lucas and Bubb	0.10-0.80	0.30-0.80

Table 5.5 Depth and velocity requirements for spawning grayling

Source	Depth requirements (m)	Velocity requirements (ms^{-1})
Sempeski and Gaudin (1995)	0.40-0.60	0.20-0.40
*Nykanen *et al*., (2004)	1.4-3.0	0.40-1.0
Mallet *et al*., (2000)	0.80-1.20	0.70-1.10
*Vehanen *et al*., (2003) Adult Modified restored river	0.20-1.55	0.2-0.45
*Hubert *et al*., (1985) Adult	-	0.21-0.61
*Dyk (1984) Adult	-	0.24-0.5
*Greenberg *et al*., (1996) Adult	0.75-1.65	-

Table 5.6 Depth and velocity requirements for juvenile grayling. Sources marked * denote requirements for adults rather than juveniles

	Depth (m)	Velocity (ms^{-1})
Spawning (3 months)	0.24	0.28
Juvenile (9 months)	0.71	0.33

Table 5.7 Depth and velocity requirements chosen for use in CRAB model when applied to grayling

5.3.3 Atlantic salmon

Although there are currently no Atlantic salmon populating the River Derwent upstream of Derwent Haugh, their flow requirements are being considered here (Tables 5.8 and 5.9) because, if it can be shown that the river could support the species under the new flow regime, the possibilities of installing fish passes may be enhanced. An average of all of the habitat preference ranges was calculated and any results for case studies which were particularly different to the present project were excluded. The depth and velocity requirements used in the CRAB model are summarised in Table 5.10.

Source	Depth requirements (m)	Velocity requirements (ms^{-1})
Heggberget (1991)	Mean 0.50	Mean 0.40
Moir *et al*., (1998)	Mean 0.25	Mean 0.53
Beland *et al*., (1982)	0.17-0.76	0.35-0.80
Crisp and Carling (1989)	-	Minimum >0.15-0.20

Table 5.8 Depth and velocity requirements for spawning Atlantic salmon

Source	Depth requirements (m)	Velocity requirements (ms^{-1})
Symons and Heland (1978)	0.25-0.60	0.5-0.65 (Utilised preference)
Rimmer *et al*., (1984)	0.25-0.60	-
Morantz *et al*., (1987)	0.25-0.60	Maximum <1.20
Heggennes (1990)	0.20-0.70	0.1-0.65 (Utilised preference)
Heggennes *et al*., (1999)	-	0.20-0.60
*Heggennes and Saltveit (1990)	0.30-1.00	0.1-0.50

Table 5.9 Depth and velocity requirements for juvenile Atlantic salmon. Sources marked * denote requirements for adults rather than juveniles.

	Depth (m)	Velocity (ms^{-1})
Spawning (3 months)	0.31	0.36
Juvenile (9 months)	0.25	0.23

Table 5.10 Depth and velocity requirements chosen for use in CRAB model when applied to Atlantic salmon

5.3.4 Summary of depth and velocity requirements

CRAB assumes that there will be other inputs to the river, from reservoir overtopping and from unregulated tributaries. Therefore, the flow depth and velocity requirements to be set should be minimum values to allow for additional inputs into the system and to reduce the amount of water required for release by the operating water company. The minimum requirements used in the CRAB model (based on published literature) for brown trout, grayling and Atlantic salmon are summarised in Table 5.11.

	Brown trout		**Grayling**		**Atlantic salmon**	
	Depth (m)	Velocity (ms^{-1})	Depth (m)	Velocity (ms^{-1})	Depth (m)	Velocity (ms^{-1})
Spawning	0.26	0.21	0.24	0.28	0.31	0.36
Juvenile (Rest of year)	0.31 (0.37 if 'maximum' value is included)	0.04 (0.09 if 'maximum' values are included)	0.71	0.33	0.25	0.23

Table 5.11 Summary of minimum depth and velocity requirements in spawning season and rest of year, for brown trout, grayling and Atlantic salmon

5.4. Application of CRAB and results

5.4.1 Other data requirements in CRAB

Additional to the flow depth and velocity requirements outlined in Section 5.3, there are other data requirements in order to run CRAB (discussed in Section 5.2.2). These parameters are considered constants as they are not affected by the flow requirements of the fish species. The values used are outlined in Table 5.12.

Parameter	Value used	Source
Current compensation flow	23.85 Ml/d (an average of the two flow rates for the year)	Northumbrian Water Ltd Water Order, 1957
Current demands on river	2.57 Ml/d	Tyne CAMS (2005 + updates)
Maximum release capacity	2333 Ml/d	Northumbrian Water Ltd reservoir engineer
Geomorphology (average river width. It is assumed that the geomorphology is unresponsive to proposed flow changes)	11.5 m	Average of the widths of an upstream, midstream and downstream site of the cross-sections used in HEC-RAS (bank-full). Measurements were taken at geomorphologically simple areas to ensure mean width was not obscured by bars and split channels.

Table 5.12 Additional input data for CRAB

It is assumed that the geomorphology is unresponsive to the proposed flow changes. Accrual of flow with distance downstream will occur through inputs from tributaries. However, CRAB does not consider this impact because the aim of this assessment is to design minimum release rate requirements that can cope when the only flows are due to reservoir releases. The geomorphology was represented by an average river width from an upstream, midstream and downstream site.

The calculated minimum depth and velocity requirements for each fish species were applied to the CRAB model, as well as other required inputs, as discussed in Section 5.2.2. The months required for elevated baseflow were set and spate conditions (e.g. length, total volume) were set. The model was run to output compensation flow distributions for each species, starting with the values in Table 5.4. Other parameters such as length of elevated flow and daily release rates were then altered to achieve the closest possible fit for peak elevated and lower baseflows.

Channel width

A sensitivity analysis was conducted to determine the effect of width measurement on model output. Because CRAB is a simple model, designed to be operated with minimal data collection and user training, only one width value can be applied for the whole reach being studied. Therefore, a mean width was calculated for the 25 km stretch between the reservoir and the downstream study site of Rowlands Gill. This was calculated using three sites – one representative of the upstream area (site 14 – to allow the effects of the weir at Eddy's bridge to be dissipated), one midstream site (site 8 – lies 13 km, halfway, between the reservoir release and the end of the study reach. Site 9 lies slightly nearer to the halfway point but is just downstream of a weir and so site 8 was chosen to allow the effects of the weir to dissipate) and a downstream site (site 2 – the most downstream site before the end of the study reach; Site 1 was discounted because it is a weir). See Figure 6.2 for location of sites.

The CRAB model was run, based on the flow requirements for brown trout, with a number of different channel widths and with all other parameters as constants. Figure 5.1 shows the results of the simulations. It is evident that channel width has an impact on the discharge per unit width required to achieve certain depth and velocity requirements. This is because width affects the amount of water necessary to obtain a certain depth. An increase in width of 2 m caused an increase in required flow rate of 3 Ml/d for baseflow and up to 14 Ml/d for elevated flow (as a result of the non-linear relationship between width and depth). Only minimum *recommended* flows are included in Figure 5.1, but it follows that if these are altered by width, then discharges in the *suggested* flow regime will also be altered by change in width.

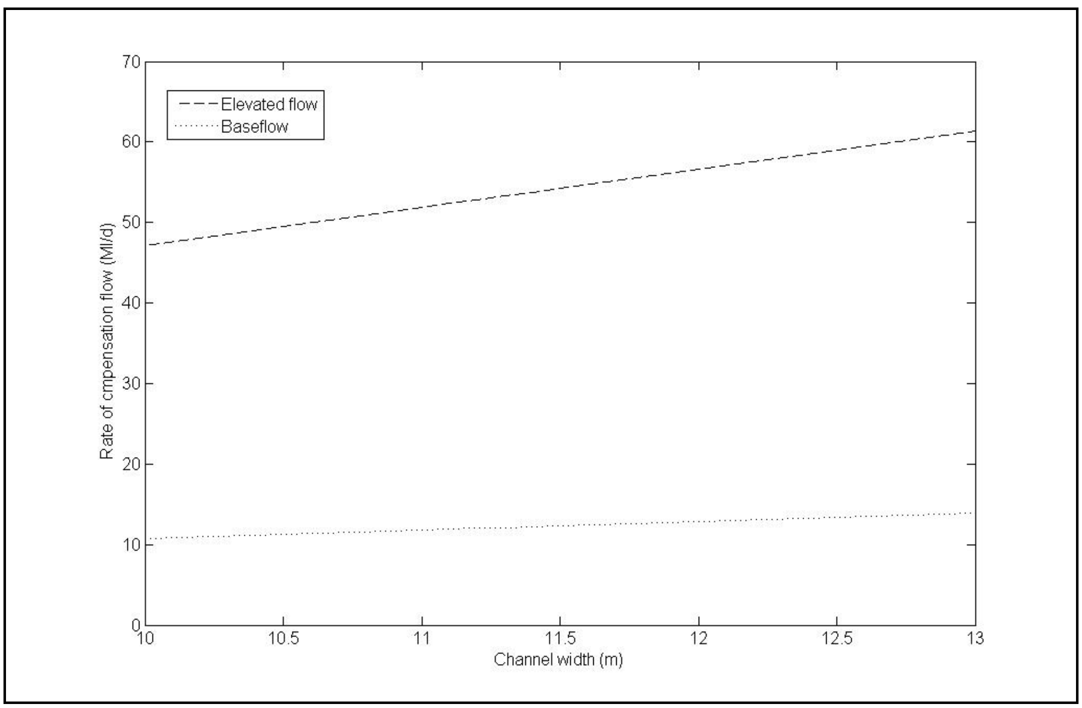

Figure 5.1 Discharge per unit width required to meet the depth and flow requirements of brown trout, for a number of different channel widths between 10 m and 13 m, as defined by CRAB

Suitability of depth and velocity requirements

Because many of the flow requirement figures derived from the literature were mean or maximum values, or were based on observation of adults rather than juveniles, the flow requirement values in Tables 5.6 and 5.9 are likely to be an over-estimation. In this case, they cannot be accommodated by the annual compensation release from Derwent Reservoir of 9533 Ml/y (183 days at a release rate of 25 Ml/d, 182 days at a release rate of 22.7 Ml/d plus 827 Ml allowed for spates). Therefore, the values used were revised to include only the minimum required values for juveniles, as described for each species in the respective Section. This helped to bring the discharges required closer to the volume available for release. It should also be noted that the volume of water that can be released from the reservoir is the very minimum and that there are other inputs to the river from unregulated tributaries and runoff, as well as any overtopping events that may occur. Therefore, if the required flows cannot be achieved, through compensation release only, it is likely that they will be augmented by other inputs. This is discussed further in Section 5.5. However, the aim of the application of CRAB in this case is to provide baseflows from the reservoir which protect the ecology when there is no other flow to the river. For this purpose, CRAB suggests that the annual reservoir release volume is insufficient to ensure sufficient supply of water for ecological requirements for the whole year. The option of focusing upon a time of year when flows are less reliable (e.g. autumn) is discussed in Section 5.5.

Model runs and results are discussed in Section 5.4.2. The model runs used, adjustments made to flow requirements and model results will be discussed separately for each species as inputs and approach varied for each species.

5.4.2. Model runs and outputs

Brown trout

The parameters from Tables 5.2 and 5.3 were applied to the CRAB model and the redefined compensation release regime for a 12 month period plotted against the current compensation release regime, as shown in Figure 5.2.

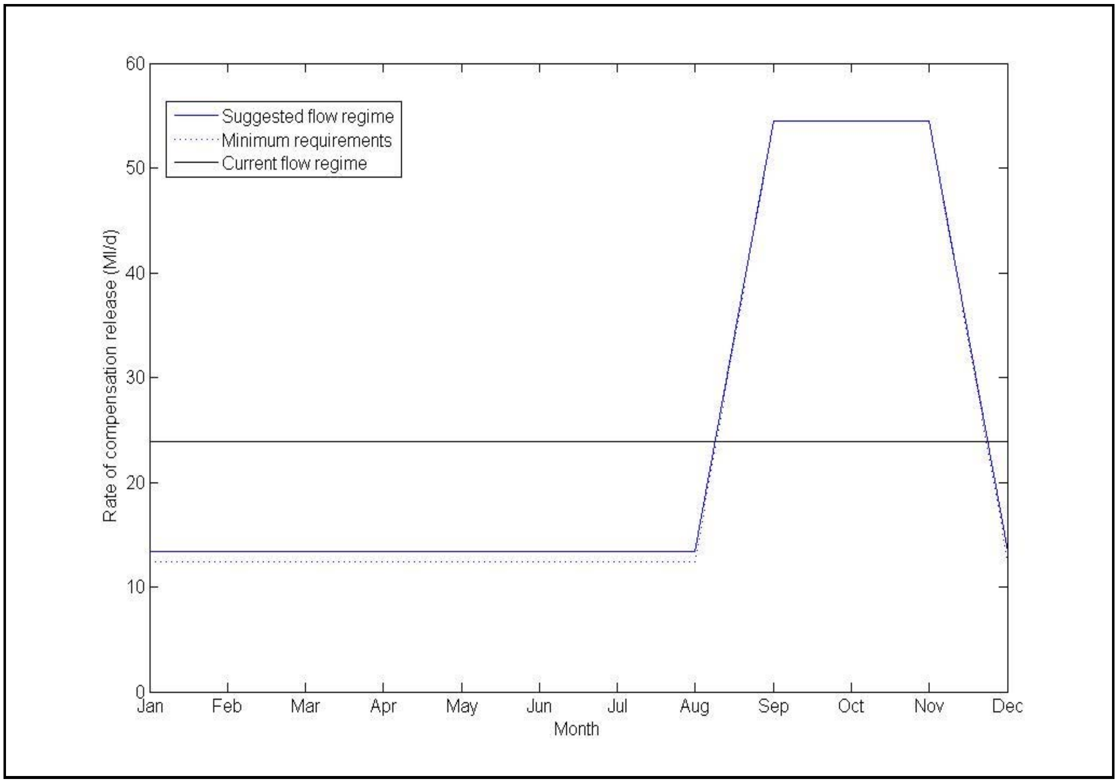

Figure 5.2 CRAB output for brown trout – suggesting minimum required flows and new compensation release regime for the River Derwent. Although a volume of 827 Ml of water is allowed within the annual release budget, the timing of this release (or indeed whether it takes place), is unknown and cannot be deduced from hydrology data

A spate flow was included in the redesign of flows. Spate flows are necessary for flushing through fine sediment and removing fine sediment from between gravel clasts. This is important for fish habitat because it creates sheltered areas in which eggs can be laid and allows oxygenation of water flowing through the matrix which is required for successful egg development. Spates would usually occur in the form of floods, but river regulation mitigates this effect and therefore regulators have a responsibility to provide an alternative. Reservoir overtopping may provide such a feature, but as overtopping occurs infrequently at Derwent dam (Chapter Three), there is a need to address this manually. The spate flow calculated by CRAB occurs on one day of the year and lasts for nine hours. The spate would begin with that month's standard daily release rate at 09.00 and rise rapidly over the course of one hour to a peak of 700 Ml/d and fall more gradually to return to the compensation flow rate eight hours later. The total volume of water released during the spate would be 74.53 Ml. A spate release peak of 700 Ml/d was chosen in order to balance the flushing effects of the flow with the stability of the channel and the ecology. A higher peak would be possible under the reservoir's capacity to release water. However, this could lead to too rapid a change in rate of flow which may be damaging to the channel topography and ecology (as suggested by Lane *et al.*, 2007). Therefore, 700 Ml/d was the greatest release rate possible without having detrimental effects on the ecology and morphology. The spate flow recommended is illustrated in Figure 5.3. The dynamics of the spate flow are unaffected by depth and velocity requirements or daily release rate.

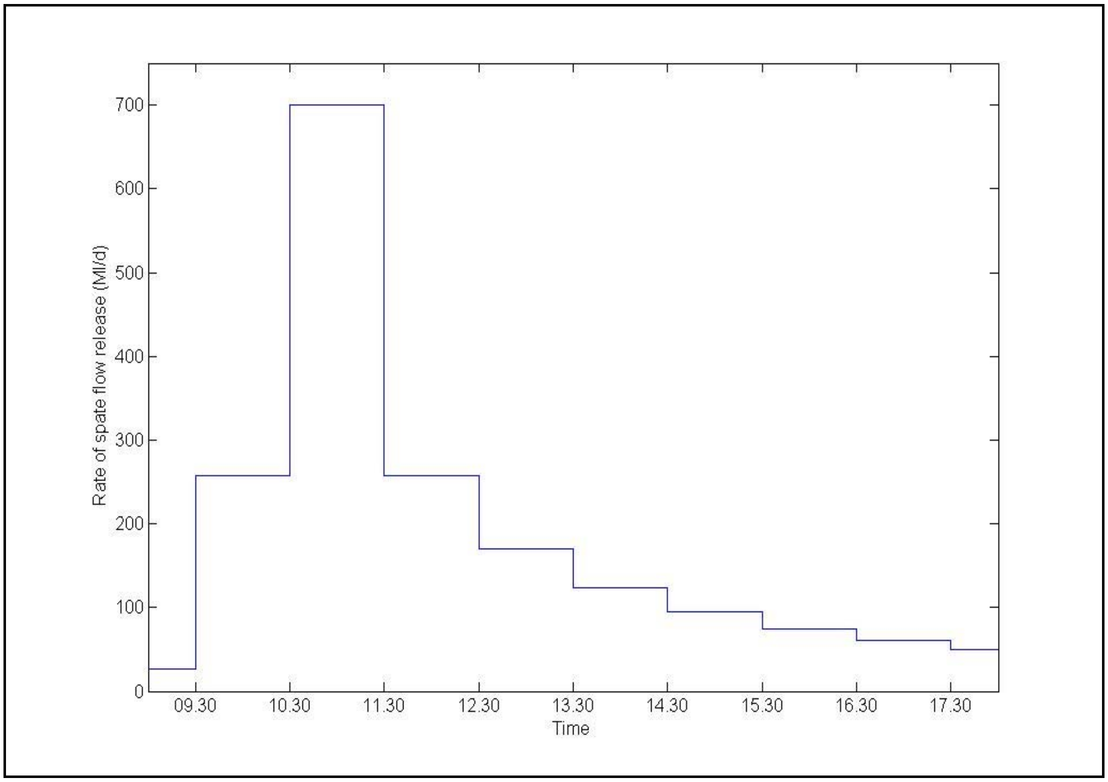

Figure 5.3 CRAB output for brown trout – suggesting one day spate flow for the River Derwent.

The CRAB output suggests that the minimum depth and velocity requirements for brown trout at different times of the year can be accommodated through the current allocated compensation volume (i.e. meet the discharge per unit width required for juvenile and spawning brown trout depth and velocity requirements, without needing to release any more water in a year than is already released from the reservoir).

Grayling

The original depth and velocity requirements for grayling produced very high discharge per unit width requirements which could not be achieved with the annual allowance of 9531 Ml (Figure 5.4). The depth and velocity requirements obtained for older fish were based primarily on observations of adults rather than juveniles. Parkinson *et al*., (1999) provided observations of one fish only and Nykanen *et al*., (2004) investigated a Finnish river which may differ in character to UK rivers. Due to the uncertainty with some of the data, Environment Agency published documents (*Seasonal movements and habitat use of grayling in the UK,* Lucas and Bubb, 2005, and *A Review of Grayling Ecology, Status and Management Practice; Recommendations for Future Management in England and Wales,* R&D Technical Report W245**,** Ibbotson *et al*., 2001) were consulted. These documents provided depth and velocity requirements based on their own findings and those of other published authors, and the two documents were in concordance. The flow requirements stated in these documents (shown in Table 5.13) were applied to CRAB because they are approved by the Environment Agency. The release regime based on these values (Figure 5.4) is more achievable than that first suggested by CRAB when using the initial flow requirements.

	Depth (m)	Velocity (ms^{-2})
Spawning	0.1	0.3
Juvenile (rest of year)	0.2	0.25

Table 5.13 Revised depth and velocity requirements for spawning and juvenile grayling, based on Lucas and Bubb, (2005) and Ibbotson *et al*., (2001).

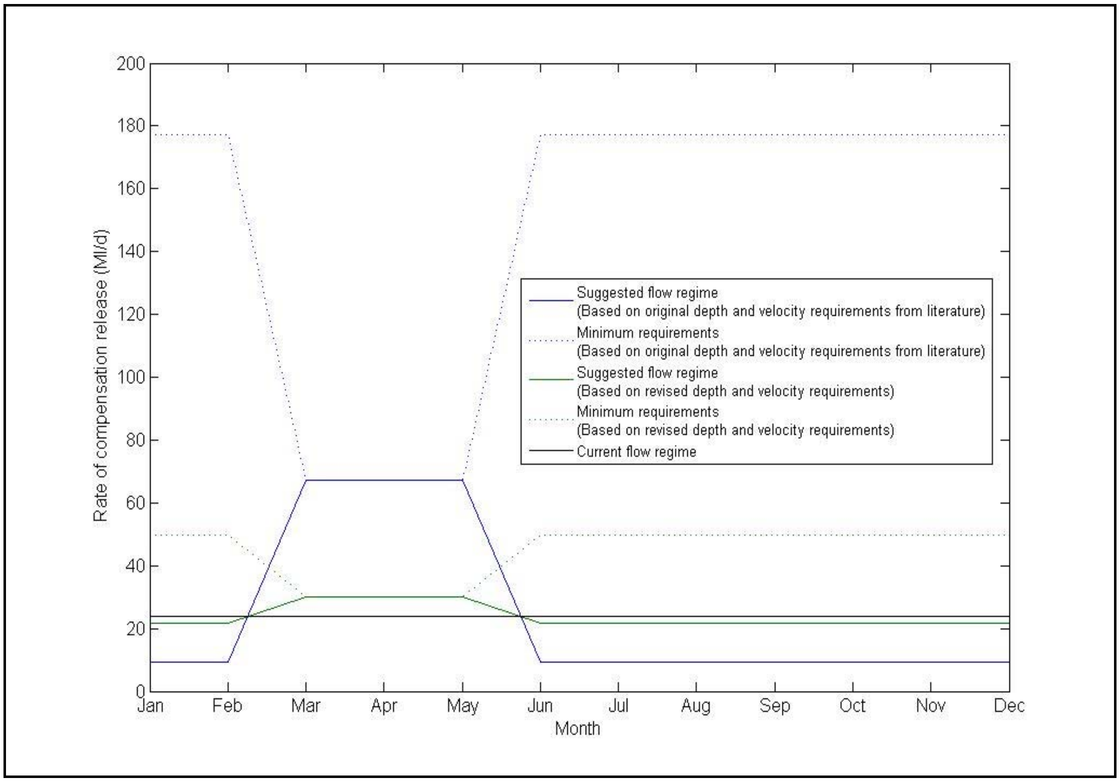

Figure 5.4 CRAB output for grayling – suggesting minimum required flows and new compensation release regime for the River Derwent, based on original depth and velocity requirements taken from literature and revised requirements taken from EA documents

The spate flow designed for grayling is the same as that for brown trout (see Figure 5.3) because it is independent of the depth and velocity requirements of each species. Because it is known that grayling currently inhabit the River Derwent, and have done so for over 100 years (anecdotal evidence), it must be assumed that the compensation flow regime from the reservoir is sufficient to support this species. Therefore, CRAB was re-run, with the setting applied to indicate that the current spawning flows in the River Derwent are sufficient. This allowed CRAB to design a compensation flow regime where the current compensation release is maintained for spawning periods and redistributed for the rest of the year. The results of this simulation are shown in Figure 5.5. It can be seen that little difference is made to the current release regime as a release rate of 23.85 Ml/d in the spawning months combined with the allocation for the spate flow, leaves enough water in the rest of the year for an average of just below 23.85 Ml/d. There is the option to scale the elevated baseflow so that more water can be allocated to the rest of the year. However, as the current release regime produces flows too low to meet any of the depth and velocity requirements prescribed in the literature, it was considered inappropriate to reduce this value further. If the brown trout regime was applied to the river, flows required by juveniles would be achieved in the autumn (i.e. the elevated period of flow for brown trout) but would decrease to far below grayling juvenile or spawning requirements in the rest of the year.

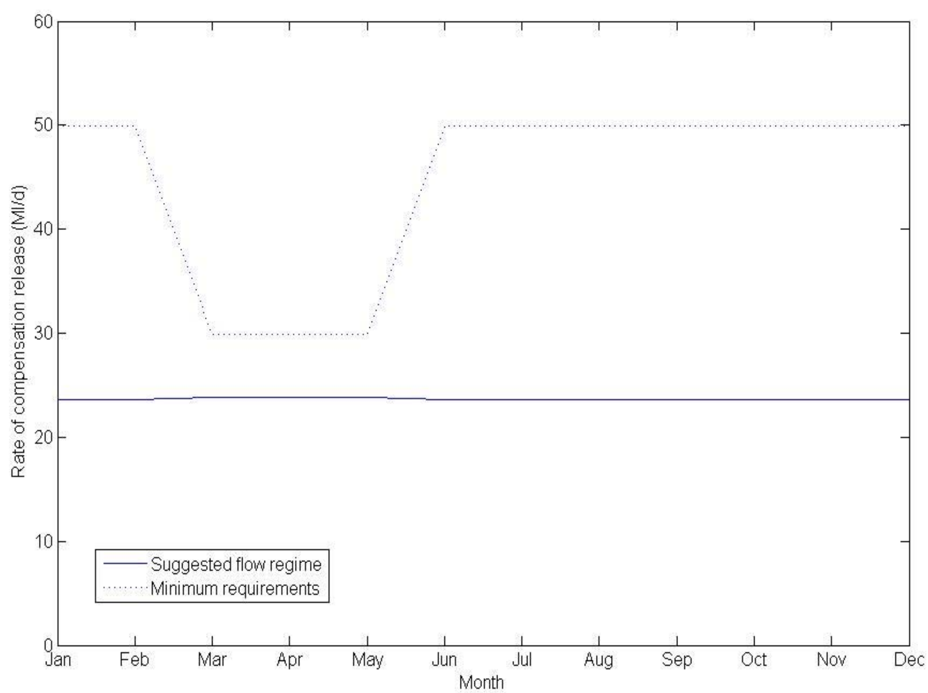

Figure 5.5 CRAB output for grayling when current spawning flows are sufficient – suggesting minimum required flows and new compensation release regime for the River Derwent

Atlantic Salmon

Many of the flow requirements for Atlantic salmon were based on mean and maximum values and 'utilised preferences'. Caution should be taken when using 'preference' values rather than 'used' values because they may differ greatly. A certain type of habitat may be preferred by fish, but the species may be able to exist in more extreme conditions if necessary. Therefore, mean, maximum and preference values were removed and a new set of flow requirements (Table 5.14) applied to CRAB. The results can be compared to the original CRAB compensation flow suggestion in Figure 5.6.

	Depth (m)	Velocity (ms^{-2})
Spawning	0.17	0.20
Juvenile (rest of year)	0.21	0.20

Table 5.14 Revised depth and velocity requirements for spawning and juvenile Atlantic salmon, based on Lucas and Bubb., (2005) and Ibbotson *et al*., (2001)

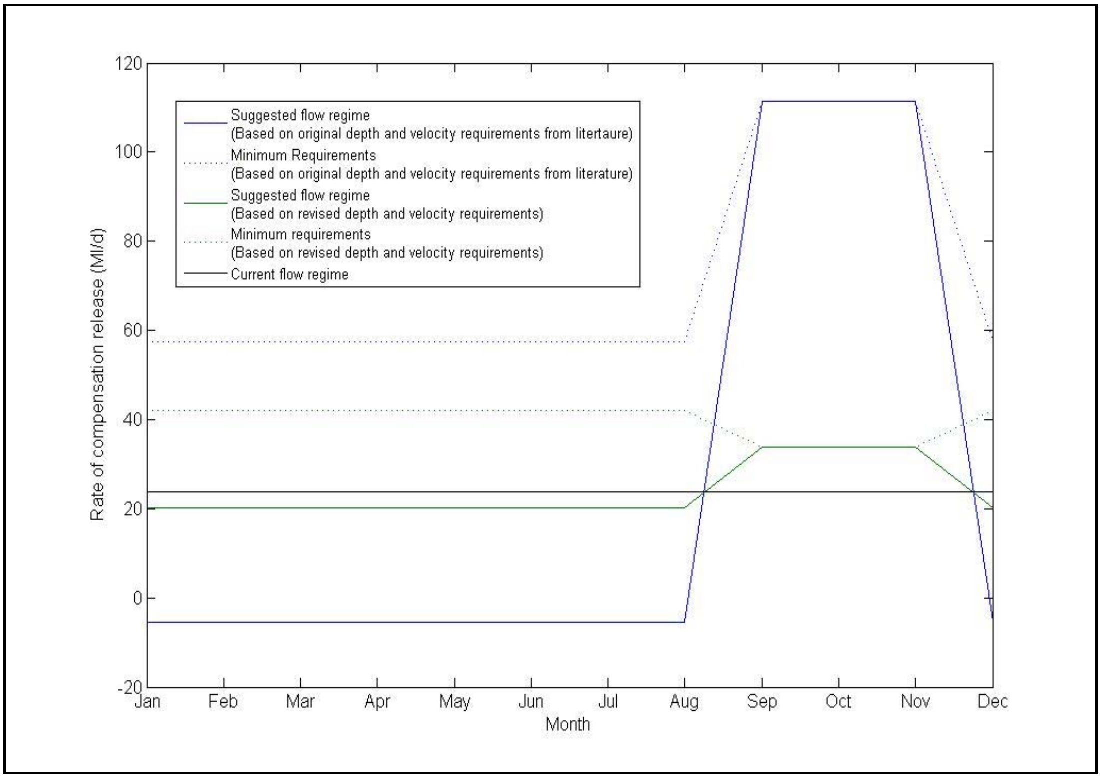

Figure 5.6 CRAB output for Atlantic salmon – suggesting minimum required flows and new compensation release regime for the River Derwent based on original depth and velocity requirements taken from literature and revised requirements based on means and juvenile observations only

The spate flow designed for Atlantic salmon is the same as that for brown trout (see Figure 5.3) because it is independent of the depth and velocity requirements of each species. Despite the revised depth and velocity preferences allowing the suggested flow regime to more closely match the minimum requirements, there is still a shortfall and the minimum requirements for Atlantic salmon cannot be met though the available reservoir compensation release only.

Summary of preliminary results

Table 5.15 provides a summary of the simulations described in Section 5.4.2, and the suggested redesign of flows.

		Month											
		Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**July**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Brown Trout	**Original Minimum requirements (Ml/d)**	12.3	12.3	12.3	12.3	12.3	12.3	12.3	12.3	*54.5*	*54.5*	*54.5*	12.3
	Original CRAB suggested flow regime (Ml/d)	13.4	13.4	13.4	13.4	13.4	13.4	13.4	13.4	*54.5*	*54.5*	*54.5*	13.4
	Revised Minimum requirements (Ml/d)	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Revised CRAB suggested flow regime (Ml/d)	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Grayling	**Original Minimum requirements (Ml/d)**	233	*67.1*	*67.1*	*67.1*	233	233	233	233	233	233	233	233
	Original CRAB suggested flow regime (Ml/d)	9.23	*67.1*	*67.1*	*67.1*	9.23	9.23	9.23	9.23	9.23	9.23	9.23	9.23
	Revised Minimum requirements (Ml/d)	49.9	*29.9*	*29.9*	*29.9*	49.9	49.9	49.9	49.9	49.9	49.9	49.9	49.9
	Revised CRAB suggested flow regime (Ml/d)	21.6	*29.9*	*29.9*	*29.9*	21.6	21.6	21.6	21.6	21.6	21.6	21.6	21.6

Atlantic Salmon	**Original Minimum requirements (Ml/d)**	57.4	57.4	57.4	57.4	57.4	57.4	57.4	57.4	*111*	*111*	*111*	57.4
	Original CRAB suggested flow regime (Ml/d)	-5.5	-5.5	-5.5	-5.5	-5.5	-5.5	-5.5	-5.5	*111*	*111*	*111*	-5.5
	Revised Minimum requirements (Ml/d)	41.9	41.9	41.9	41.9	41.9	41.9	41.9	41.9	*33.9*	*33.9*	*33.9*	41.9
	Revised CRAB suggested flow regime (Ml/d)	20.2	20.2	20.2	20.2	20.2	20.2	20.2	20.2	*33.9*	*33.9*	*33.9*	20.2

Table 5.15 Results of CRAB simulations for annual compensation release regimes for brown trout, grayling and Atlantic salmon based on original compensation release rates of 23.85 Ml/d. Grayling and Atlantic salmon require lower flows in the spawning period than in the rest of the year because of the combinations of depth and velocity requirements for each time period (see Tables 5.7, 5.9 and 5.10). For grayling and Atlantic salmon, depth and velocity requirements were revised to represent more realistic requirements than those originally suggested within the literature – results from both sets of data are included in this Table and are classed as 'original' and 'revised'

5.4.3. Sensitivity analysis

Elevated flow period

CRAB is programmed to divide flow between: 1) a one-day spate flow, 2) a three-month elevated flow and 3) a nine-month baseflow. It has been suggested that a spawning season of one month may be sufficient for the fish to spawn in the River Derwent (personal communication, Environment Agency fisheries expert). If the elevated flow needed to be sustained for one month rather than three, more water could be allocated to meet flow requirements for the rest of the year. Therefore, based on the minimum requirements output by CRAB, flow distributions were calculated for: 1) a one-day spate flow, 2) a **one**-month elevated flow and 3) an **eleven**-month baseflow. The results are summarised in Table 5.16.

	Brown trout		Grayling		Atlantic salmon	
	Minimum flow requirements based on CRAB's 3:9 month distribution	Suggested flows based on 11:1 month distribution	Minimum flow requirements based on CRAB's 3:9 month distribution	Suggested flows based on 11:1 month distribution	Minimum flow requirements based on CRAB's 3:9 month distribution	Suggested flows based on 11:1 month distribution
Spawning	54.49	54.49	29.94	29.94	33.93	33.93
Juvenile	12.37	21.11	49.90	23.31	41.91	22.95

Table 5.16 Summary of new flow regimes suggested based on a one month elevated spawning flow for brown trout, grayling and Atlantic salmon. Grayling and Atlantic salmon require lower flows in the spawning period than in the rest of the year because of the combinations of depth and velocity requirements for each time period (see Tables 5.8, 5.10 and 5.11)

Table 5.16 shows that permitting only one month for a spawning flow does not allow the minimum flow requirements to be met, except for brown trout, in which case the 3:9 month distribution is already adequate. Grayling and Atlantic salmon actually require lower discharge per unit width (but not necessarily depth AND velocity) in times of spawning than during the rest of the year, according to the reviewed literature. Consequently, assigning less water for one month would put a greater demand on volume of water needed for the remaining eleven months of the year, making the shortfall for this period larger.

Distribution of spate flow

According to the 1957 Derwent Water Order, Northumbrian Water Ltd are required to release 25 Ml/d from the reservoir between April and September and 22.7 Ml/d between October and March. In addition to this, they are allowed to release up to 827.4 Ml in any one year in the form of freshet flows, provided that the release in any one day does not exceed 90.0 Ml. This means that if a total freshet value of 74.53 Ml is released on one day, 752.87 Ml remains for release.

There are a number of options in dealing with this surplus. The freshet period could be extended to run for a number of days, or the remaining water could be apportioned to the standard daily flows. This Section investigates what impact these actions would have on the suggested release regimes in CRAB. Table 5.17 details the recommended release regime if the daily flow were to increase to 26.12 Ml/d (which takes the average current daily flow and includes the remainder of the freshet allowance) and the effect of allocating freshet flows to five days and then allocation of the remaining water to the standard daily flows. The analysis is conducted for each fish species and is based on the revised depth and velocity requirements (rather than original figures taken from the literature) for grayling and Atlantic salmon.

Daily release rate of (Ml/d): ↓		**Month**											
		Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**July**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Brown trout	**23.85**	12.4	12.4	12.4	12.4	12.4	12.4	12.4	12.4	*54.4*	*54.4*	*54.4*	12.4
	26.12	16.4	16.4	16.4	16.4	16.4	16.4	16.4	16.4	*54.4*	*54.4*	*54.4*	16.4
	25.22	15.2	15.2	15.2	15.2	15.2	15.2	15.2	15.2	*54.4*	*54.4*	*54.4*	15.2
Gray-ling	**23.85**	21.6	*29.9*	*29.9*	*29.9*	21.6	21.6	21.6	21.6	21.6	21.6	21.6	21.6
	26.12	24.6	*29.9*	*29.9*	*29.9*	24.6	24.6	24.6	24.6	24.6	24.6	24.6	24.6
	25.22	23.4	*29.9*	*29.9*	*29.9*	23.4	23.4	23.4	23.4	23.4	23.4	23.4	23.4
Atlantic salmon	**23.85**	20.2	20.2	20.2	20.2	20.2	20.2	20.2	20.2	*33.9*	*33.9*	*33.9*	20.2
	26.12	23.3	23.3	23.3	23.3	23.3	23.3	23.3	23.3	*33.9*	*33.9*	*33.9*	23.3
	25.22	23.4	23.4	23.4	23.4	23.4	23.4	23.4	23.4	*33.9*	*33.9*	*33.9*	23.4

Table 5.17 CRAB Suggested release regimes based on original release rate of 23.85 Ml/d, increased release rate of 26.12 Ml/d, and release rate of 25.22 Ml/d (based on five day rather than one day spate flow), for brown trout, grayling and Atlantic salmon. Spawning flows remain consistent as these are the most sensitive periods – flows in the rest of the year are changed according to different daily compensation release rates

It is clear that a daily flow rate of 26.12 Ml provides the most potential to meet the required discharges, although it is insufficient to meet the requirements of grayling and Atlantic salmon with even the most optimistic sets of depth and velocity requirements. Solutions to this issue will be discussed in Section 5.5.

5.5. Discussion and conclusions

5.5.1 General observations

The results have shown that for both spawning and juvenile brown trout, the required flow conditions are achievable on the River Derwent. Juvenile grayling and Atlantic salmon require considerably higher discharges per unit width than juvenile brown trout, even when a conservative figure is drawn from suggestions within literature. For grayling and Atlantic salmon, the depth and velocity requirements described in the literature are not theoretically achievable. However, because it is known that grayling already exist within the channel, it can be concluded that current flow conditions for this species are liveable, if not ideal. Changing the compensation flow for just one month is unlikely to benefit any of the species studied here. Changing the flow regime to provide three month changes in flow may be beneficial in nearing the optimum depth and velocity requirements. However, the results suggest that in practice, this would involve a *lower* flow in the spawning period for grayling and Atlantic salmon than during the rest of the year because grayling and Atlantic salmon require lower depths than juveniles of the respective species, according to literature. Because depth and velocity are used to determine the required discharge per unit width, this leads to a lower compensation discharge in spawning periods.

If a flow regime were to be designed based on the requirements of grayling and Atlantic salmon, spawning requirements for brown trout would not be met (although, again, it is known that this species already successfully populates the River Derwent). Applying a new flow regime based on the requirements of any one species is unlikely to benefit the other species significantly. Grayling require an elevation of flow in the spring to meet spawning requirements while spawning brown trout and Atlantic salmon require elevations of flow in the autumn. The flow regime designed for brown trout would benefit Atlantic salmon in the spawning period, but would not be suitable for juvenile Atlantic salmon as there would be a decrease in flow beyond the current release regime which would create discharge per unit width lower than they require and lower than already experienced. Chapters Six and Seven will investigate further which flow regimes may be applied to be of the most benefit to all species.

5.5.2 Variation in seasonal requirements

Not only is there variation in the amounts of discharge required by the different species studied, there are also variations in *when* and to what *magnitude,* changes in flow are required. Figure 5.7 illustrates how grayling require a change in flow in the spring (Varley, 1967, Woolland, 1972), in contrast to brown trout and Atlantic salmon which require changed flows in the autumn (Degerman *et al*., 2000, Armstrong *et al*., 2003). Atlantic salmon would require the most profound change in scale of flow and brown trout would require a different *direction* of change to grayling and Atlantic salmon (i.e. an increase in the spawning period). It should be noted that Figure 5.7 displays the optimum minimum *requirements*, based on the literature and not the new flow regimes suggested by CRAB. A combined optimum minimum *required* flow throughout the year for the three species would ideally be as that shown in Figure 5.8. The actual change in flows suggested by CRAB (Figure 5.9) indicate that there should be an upward change in flow volume (compared to the current compensation release regime) for the spawning period in all cases and that this should be compensated for by a decrease in flow for the rest of the year. This is because CRAB assumes that the spawning flow should be prioritised, although there are insufficient fish data on the River Derwent to prove that the population is spawning limited. There is the option to alter the flow for the rest of the year rather than the spawning flow but this would lead to flow remaining at 23.85 Ml/d all year, or the introduction of flows that are acceptable for different species in the spawning season (as discussed in Section 5.4.2 – Grayling).

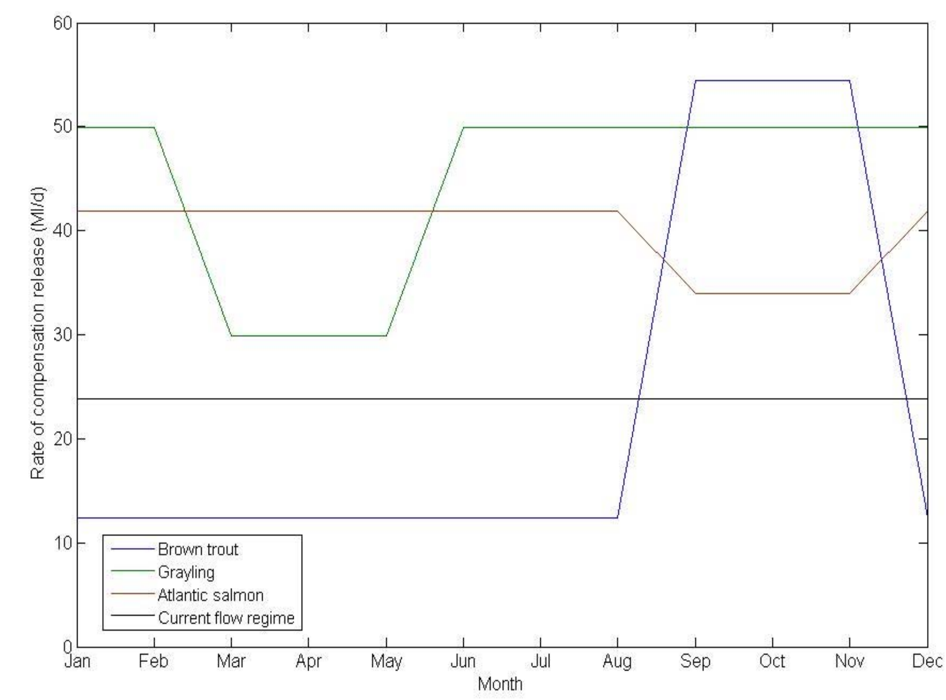

Figure 5.7 Timing and magnitude of *required* changes in flow to meet minimum depth and velocity requirements of brown trout, grayling and Atlantic salmon

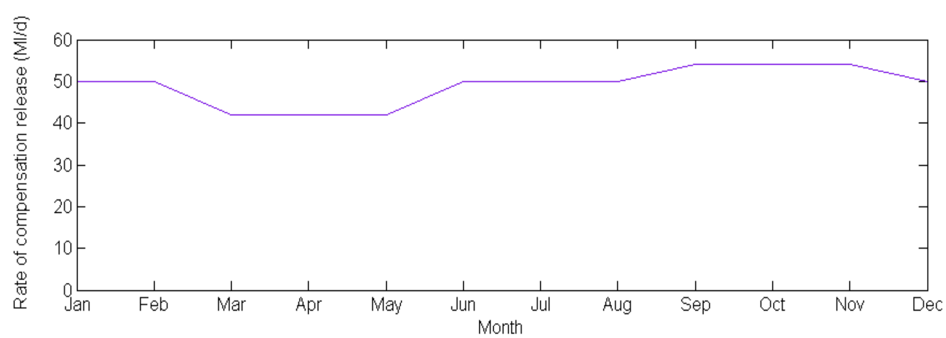

Figure 5.8 Combined *required* compensation necessary to meet the requirements of all three fish species

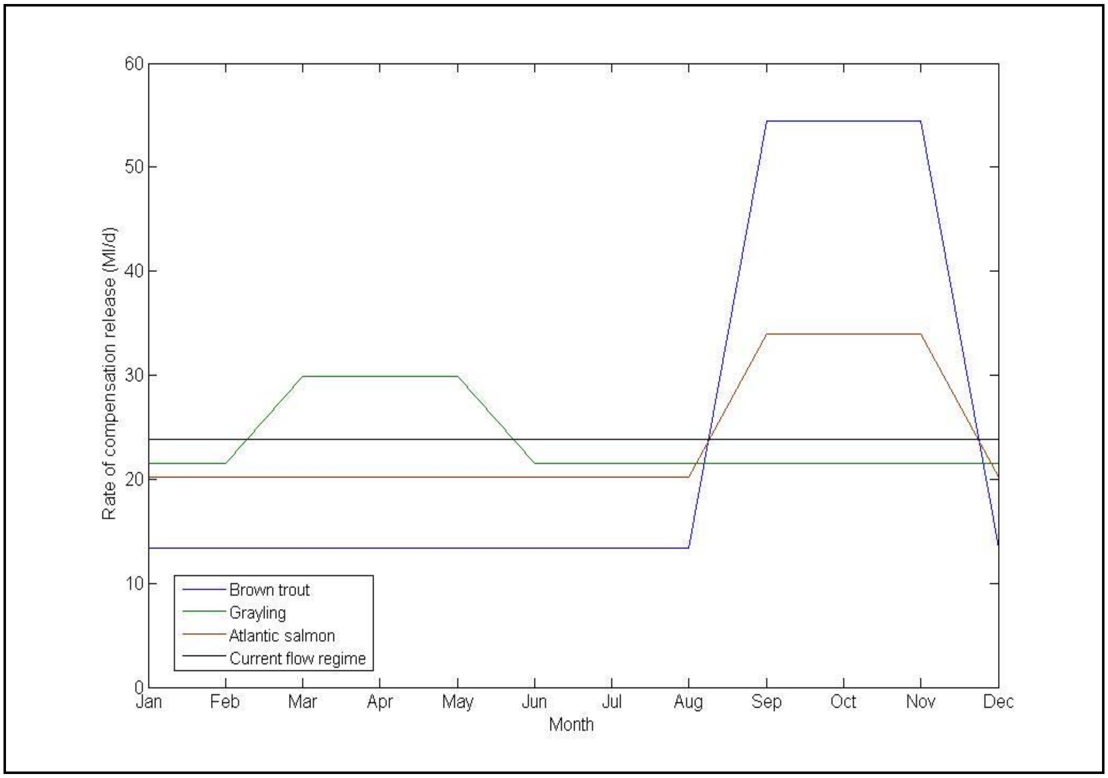

Figure 5.9 Timing and magnitude of *suggested* changes in flow for brown trout, grayling and Atlantic salmon

5.5.3 Shortfalls in required flows and the role of CRAB

Although the results suggest that the required flows cannot be achieved for two of the three species studied, encouragement should be taken from the fact that grayling already exist in the River Derwent, despite the suggestion that flows are inadequate. It should be reiterated that the flows used in the CRAB analysis are based solely on the release from the reservoir and that there are a number of other inputs to the river from sources such as tributaries. Indeed, only 1-2 km from the reservoir release lies the confluence of the River Derwent and Burnhope Burn, the Derwent's most significant tributary. The positioning and size of this tributary (the catchment area of Burhnhope Burn is 24 km^2 and the catchment area of the Derwent up to that point is 118 km^2, i.e. Burnhope Burn drains 20% of the Derwent catchment to that point) mean that flows are supplemented for most of the River's length downstream of the impoundment. Figure 5.10 illustrates how significant an input the Burnhope Burn has on the discharge of the River Derwent. It can be seen that for the majority of time, flow at Eddy's Bridge sits well above the daily discharge rate of Derwent Reservoir, suggesting that Burnhope Burn provides a significant proportion of the River Derwent's discharge. Times when flow falls below the daily reservoir release rate may be attributed to the implication of drought orders or errors in gauged data. It could be concluded, therefore, that the key to the project would be to focus on times of drought (i.e. Autumn, when there has been considerable reservoir drawdown over the summer months), when flow in the River Derwent is not supported by flows from Burnhope Burn. If this is the case, then it may be prudent to focus upon the suggested flow regimes for brown trout and Atlantic salmon as these are the ones in which the spawning period occurs in the driest period of the year.

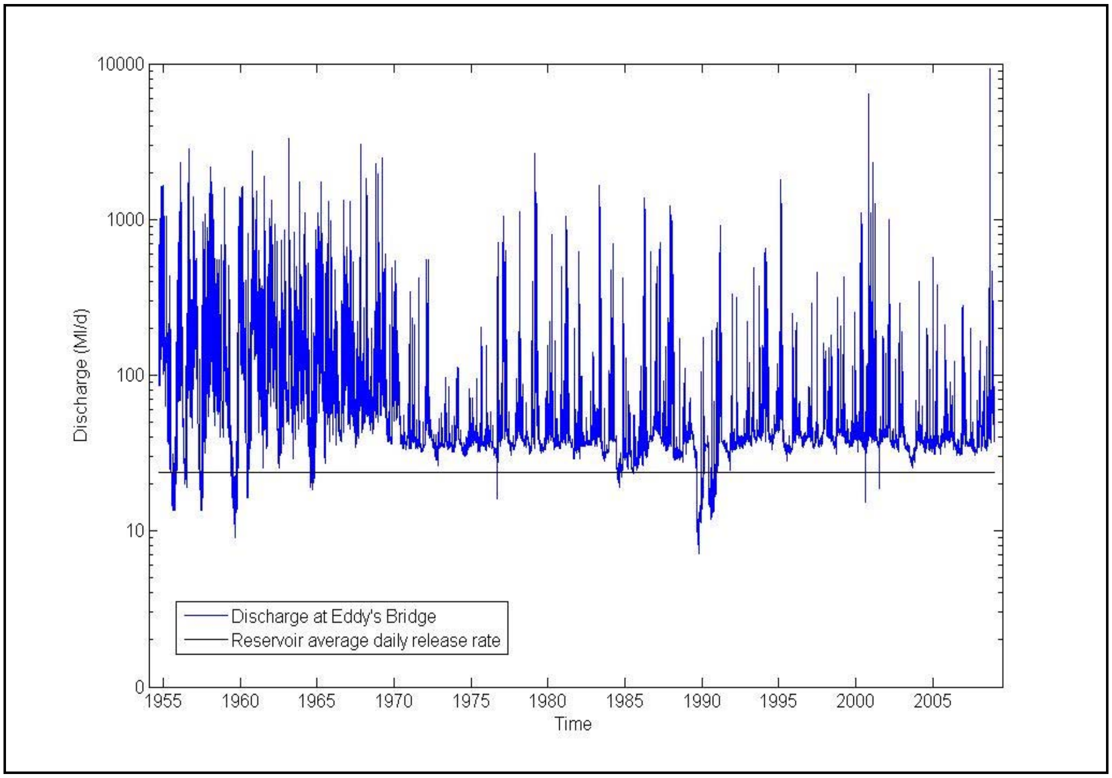

Figure 5.10 Hydrograph for Eddy's Bridge gauging station showing current average rate of compensation release from Derwent Reservoir

It should also be remembered that CRAB is not intended to be the end point of analysis, but an indication of how compensation flows may be redesigned to deliver certain hydrological properties (depth, velocity). Although CRAB cannot inform the reader of how certain species will respond to changes in flow regimes, it is a descriptive tool which can be used with minimal data input and user training in order to design flows for a specific species and a specific channel. As with any model output, the only true way of testing effectiveness is to monitor instream effects in response to the prescribed change. CRAB should be viewed as a starting point to the process of introducing new flow patterns and its effects should be monitored carefully and for a period of time suitable to detect any changes. There are scenarios in which the use of CRAB may be appropriate and when a more sophisticated approach, such as CRAM (Compensation Release Assessment at the Meso-scale) might be adopted. These are summarised below (Lane *et al.,* 2007).

CRAB is the appropriate tool to use when:

- The channel is dry
- It is accepted (through data or local knowledge) that current ecology is degraded or constrained by the current regime
- Where changing the flow will impact on the wider community (such as abstractors or species which have adapted to the current regime)
- The effects of a revised regime can be carefully monitored

CRAM may be more appropriate if:

- No clear redesign recommendation emerges
- More subtle changes in compensation flow need to be identified (e.g. changes to morphology and amount of usable habitat)
- Where the issue is contentious
- Where the current ecology is healthy and may be put at risk by a new regime.

Although it is accepted that the impoundment of the River Derwent has affected the ecology of the channel, and that changing the flow will impact upon the ecology, anglers, water abstractors and recreational users, it is the latter points which compel the author to recommend further, more detailed investigation into the redesign of compensation flows on the River Derwent. Although CRAB appears to prescribe improved flow conditions for brown trout, benefits for Atlantic salmon and grayling may be limited but will be explored further in Chapters Six and Seven. The issue of changing flows must be approached with delicacy. Water is a valuable resource and changing the compensation flow is likely to impact upon abstractors as well as recreational users. A redesigned flow such as that suggested in CRAB would reduce stream flows in summer months – when abstractors are most likely to take water. It is likely that CAMS (Catchment Abstraction Management Strategy) limitations (such as Hands-Off Flows) will come into place more often should flow decrease in some months. The Environment Agency must be seen to balance the needs of the users in a just manner. Currently, since there are brown trout and grayling residing in the River Derwent, great care must be taken not to degrade the populations already present.

The results from CRAB provide information that can be carried into the next stage of analysis – the hydrodynamic modelling, in which the impacts of the flow regimes suggested by CRAB can be examined on a number of more detailed and more localised scales.

5.6 Conclusions

Because of the limitations with the data and results used in this Section it would be inappropriate to suggest revised compensation release regimes based on CRAB analysis alone. Therefore, suggestions will be made following a more thorough examination, as follows in Chapter Six. The results from Chapter Five will, however, be used as a baseline for the CRAM analysis. Because CRAB is easier and quicker to run, it can be applied to a number of scenarios and the outcomes of these can be shortlisted for a more detailed investigation. Therefore, the flow regimes suggested in Figure 5.9 will be applied to the CRAM process.

Chapter Six

Hydrodynamic modelling of the River Derwent

6.1. Introduction

Chapter Five introduced the concept of redesigning new compensation release regimes by considering the depth and velocity requirements of certain species. This Chapter builds upon that approach and assesses the impacts of those new flow regimes on the depths and velocities at a number of sites, for a range of flows and over time. Chapters Six and Seven include the key components of the CRAM (Compensation Release Assessment at the Meso-scale) process, which combines the results from hydrodynamic (Chapter Six) and fuzzy (Chapter Seven) modelling approaches to determine the impact of flow regimes on habitat suitability. Chapter Six starts with a brief discussion on the advantages and disadvantages of using 1D and 2D hydraulic models (6.2). Section 6.3 introduces the hydraulic model to be used and the modelling process while Section 6.4 presents the results. These results and their implications for river ecology are discussed in Section 6.5 and concluded in Section 6.6.

6.2. Approach to modelling

A number of hydraulic models, operating at a range of levels of complexity, are available for the prediction of channel and floodplain flows. These range from simple 1-D systems such as HEC-RAS (US Army Corps), MIKE11 (Danish Hydraulic Institute, Denmark, 1997) and Isis (Halcrow and HR Wallingford), through 2-D models including FESWMS (U.S. Department of Transportation, Federal Highway Administration (Froehlich, 2002)), TELEMAC-2D (Horritt and Bates, 2001) and MIKE 21 (DHI, 2000) to 3-D approaches (including FLUENT, Fluent Inc., 1993 and PHOENICS e.g. Lane *et al*., 2002). There is debate within the literature as to whether a 1D, 2D or 3D approach is preferable when modelling open channel hydraulics. Tayefi *et al*., (2007) claim that it has been well acknowledged that *in-channel* flows may be satisfactorily described by a 1D representation, while the more complex 2D and 3D effects of *out of bank* flows cause topographic and topological characteristics of floodplains to strongly impact upon flow processes, thus reducing the applicability of a 1D approach. Similarly, Chatterjee *et al*., (2008) concluded that 1D models may be used to study water level and discharge reductions in the main river while a coupled 1D-2D model may be used to predict out of channel water levels (such as those on the floodplain and in storage areas). They note that 1D models are useful because they are simple to use and provide information on bulk flow characteristics but fail to provide detailed information on the flow field. 2D models have their drawbacks in that they require substantial computation time (Chatterjee *et al*., 2008) and data inputs. Horritt and Bates (2002) found that the 1D model HEC-RAS outperformed 2D models (TELEMAC-2D and LISFLOOD-FP) in some cases of flood event prediction. It is expected that the reason for this is due to a difference in response to the calibration process – HEC-RAS predicts well when calibrated against hydrometric data (Horritt and Bates, 2002).

A 1D approach looks at the stream as a number of cross-sections (Ghanem *et al*., 1996). Lane and Ferguson (2005) note that when using 1D models, there is the issue of the width-averaging of flow. This implies that bar-pool-riffle structures and the associated lateral variation in flow cannot be accurately represented and this spatial heterogeneity in upland streams is an important feature of fish habitat.

Downs and Thorne (2000) suggest that while new generation 2D and 3D models may overcome some of the problems of addressing flow and morphology that exist in 'traditional' models, they require advanced modelling capabilities and are therefore expensive to apply. They, too, found that when modelling flow processes in an ecological context, simple models such as HEC-RAS are often adequate. With specific reference to HEC-RAS, Downs and Thorne (2000) conclude that the model is an accepted tool amongst flood defence and drainage engineers with achievable data requirements at reach and basin scales and can represent a variety of more complex channel scenarios such as the reconciliation of flood defence with habitat diversity and geomorphological sustainability.

Lin *et al*., (2005) agree that 1D models are appropriate for modelling river/channel flows and hydraulic structures, but when floodplains need to be considered, a more detailed approach is preferable. Like Chatterjee *et al*., (2008), they suggest the coupling of a 1D and 2D model to assess flows and inundation over floodplains.

Lane and Ferguson (2005, p 218) comment on the suitability of 1D and 2D models in flow prediction and habitat assessment. They conclude that the use of 1D models is widespread in the assessment of reach-scale flow, morphodynamics and habitat problems and it is likely that they will continue to be widely used where the reach of interest is longer than many tens of channel widths (because of the computational and data demands of more complex 2D and 3D models). A number of channel routing and flood risk analysis problems are assessed through commercial modelling packages which use a standard set of equations (Equations 6.1 to 6.6). These include HEC-RAS, ISIS and MIKE-11. What this 1D approach cannot account for is within-reach flow variability in the vertical or cross stream direction (Lane and Ferguson, 2005 p 218). When such aspects of flow are considered important, Lane and Ferguson (2005) suggest the application of at least a two-dimensional analysis. They note that within some of the most commonly used habitat assessment models (e.g. PHABSIM, US Fish and Wildlife Service), the hydraulics component consists of a 1D treatment, limiting applicability to areas greater than $10m^2$, which is generally unsuitable for habitat modelling.

Given the above evaluation of 1D and 2D approaches to flow modelling, and the widespread use of 1D models, the application of a 1D model was considered suitable for the assessment of flow on the River Derwent. As consideration is to be given to the whole river and not a particular reach, and the fact that general indication of changes in depth and velocity were required for the 25km stretch, the data collection and computational demands of a 2D model were considered too great for this project. It is agreed by a number of authors that 1D models are suitable for in-channel assessment on a river scale (Chatterjee *et al*., 2008; Horritt and Bates, 2002; Tayefi *et al*., 2007) and it was therefore deemed suitable for the case in question.

6.3 Methodology

6.3.1 Introduction

The steps highlighted in Figure 6.1 were followed in order to ensure the correct model was chosen and developed effectively in order to predict new flow patterns for the River Derwent. This Section outlines the processes involved in model choice, development and application.

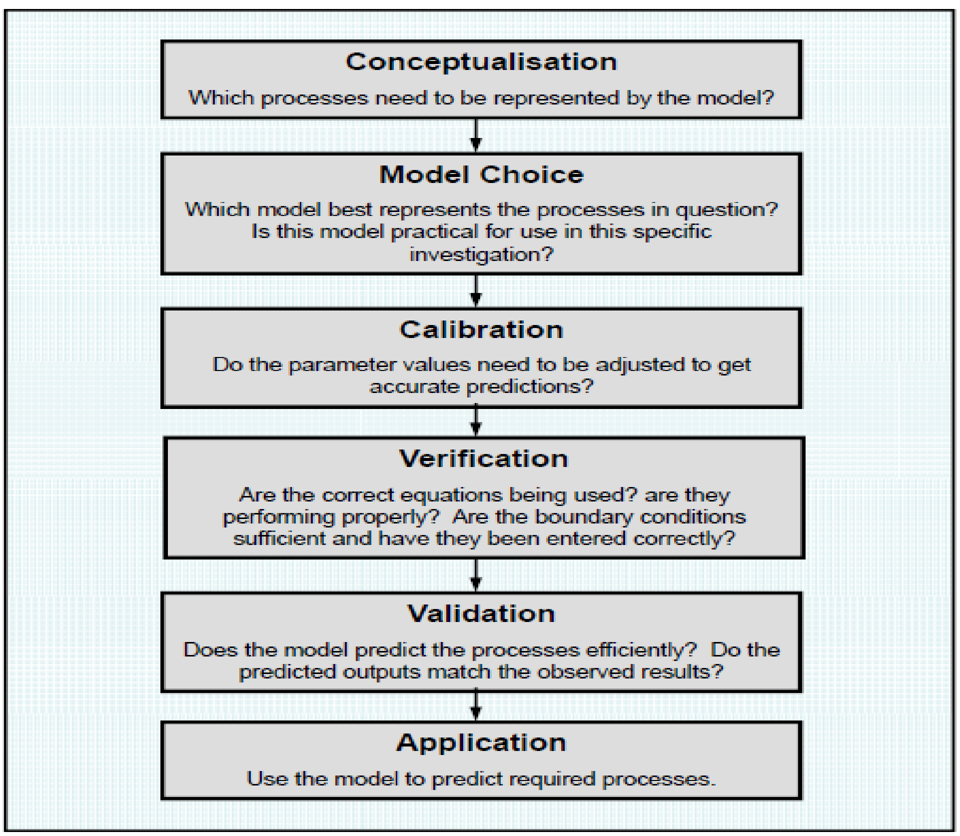

Figure 6.1 Stages involved in model development. (Source: adapted from Beven, 2001 p 4 and Lane, 2003 p 274)

Conceptualisation

In this step, thought was given to the tasks required by a modelling process. Defining required outputs enables the user to ensure the correct model is employed. In this step, the physical processes that needed to be considered in order to deliver the required outputs were identified and matched to the chosen model. Lane and Ferguson (2005) outline the basic flow equations used in 1D approaches to hydrodynamic modelling (Equations 6.1 to 6.6). In its simplest sense, discharge (Q, $m^3 s^{-1}$) can be described as a function of area (*A), (*m) and velocity (*v), (*m s^{-1}):

$$Q = vA \qquad \text{Equation 6.1}$$

Area and velocity may both vary with distance downstream. Because of this, if flow is steady (i.e. when discharge, depth and velocity are temporally constant), then mass conservation gives:

$$0 = -\frac{\partial (vA)}{\partial x} + i = -v\frac{\partial A}{\partial x} - A\frac{\partial v}{\partial x} + i \qquad \text{Equation 6.2}$$

where *i* is the input from (or, if negative, loss to) storage per unit distance downstream. For an unsteady flow, mass conservation would give the equation:

$$\frac{\partial A}{\partial t} = -v\frac{\partial A}{\partial x} - A\frac{\partial v}{\partial x} + i \qquad \text{Equation 6.3}$$

The same analysis was applied to momentum conservation. The rate of change of momentum through time at a point is governed by the spatial change of the momentum plus sources (i.e. the driving forces), which can be expressed as:

$$\frac{\partial (Av)}{\partial t} = -\frac{\partial (Av^2)}{\partial x} + sources \qquad \text{Equation 6.4}$$

The driving forces are: 1) pressure gradients; 2) potential energy; and 3) friction that causes energy expenditure:

$$\frac{\partial (Av)}{\partial t} + \frac{\partial (Av^2)}{\partial x} = -Ag\frac{\partial h}{\partial x} + gA(S_o - S_f) \qquad \text{Equation 6.5}$$

where h is mean flow depth (m), S_o is the bed slope of the channel (defining the potential energy term) (-), and S_f is the friction slope (i.e. the friction term) (-). This equation represents not only the effects of boundary resistance, but a whole set of other processes. These mean that momentum can be transformed into flow components that vary spatially within the cross-section (e.g. secondary circulation) and through time (turbulence processes). In most cases, the friction term is defined under the assumption that the flow is locally uniform, allowing a uniform flow equation to be used, such as the Darcy-Weisbach equation:

$$S_t = \frac{v^2 f}{8gR} \qquad \text{Equation 6.6}$$

where R is the hydraulic radius (m) and f is a 'friction parameter' (dimensionless).

A variation on the Darcy-Weisbach friction equation, and one frequently used in flow modelling, is the Manning's roughness equation:

$$v = k\frac{R^{2/3}s^{1/2}}{n} \qquad \text{Equation 6.7}$$

where k=1 (SI units), R is hydraulic radius (m), s is the slope of the energy gradient and n is the Manning's resistance coefficient (dimensionless) (Knighton, 1998).

These equations can be used to predict channel routing and are the basis of commercially available flood risk models such as HEC-RAS, ISIS and MIKE 11. The major limitation of using such equations is the inaccuracy in recreating within-reach flow variability (vertically or cross-stream), which often results from secondary circulation (Lane and Ferguson, 2005).

Thus, Equations 6.1 to 6.6 are the basis on which the model for this study is to be set. They provide a 1D method with which to calculate flow routing and details of flow characteristics (e.g. depth, velocity) at cross-sectional, reach and river scales and are widely accepted in the modelling community for this purpose.

6.3.2 Model choice and set-up

As discussed above, a 1D approach was chosen for this project and the flood inundation model HEC-RAS 4.0 (U.S. Army Corps River Assessment System) was chosen for a number of reasons. First, it operates using the flow Equations 6.1 to 6.6, which, as discussed are accepted as appropriate for 1D flow routing models. Second, it is a commercially used model, free and available for use and adaptation. Third, it has simple data requirements which are achievable from a few days' data collection (cross-sections) (hydrology data were already available from the Environment Agency to provide input and output boundary conditions). Fourth, roughness values can be input for each different cross-section and the model allows a distinction to be made between the channel and each of the banks. Fifth, the model uses the generally accepted flow equations as noted by Lane and Ferguson (2005) for the purposes of flood prediction. Finally, the software provides a simple and workable user interface to aid speedy model development.

HEC-RAS has a wide range of uses from flow modelling (e.g. Tayefi *et al.*, 2005) and channel restoration studies to modelling dynamics of glacial outburst floods (e.g. Baker *et al.,* 1993 and Benito, 1997, in Carrivick, 2007). The model is widely used by the UK Environment Agency as their primary tool for flood risk assessment.

The model is based upon the continuity and momentum equations, as described above (Equations 6.3 to 6.6) and these are supported by the energy equation (Equations 6.8 to 6.11, as described by Tayefi, 2005):

$$Y_2 + Z_2 + \frac{\alpha_2 V_2^2}{2g} = Y_1 + z_1 + \frac{\alpha_1 V_1^2}{2g} + h_e \qquad \text{Equation 6.8}$$

where Y_1 and Y_2 are the depth of water at cross-sections 1 and 2 (m), Z_1 and Z_2 are the elevation of the main channel at sections 1 and 2 (m), V_1 and V_2 are the average velocities (total discharge/total flow area) at sections 1 and 2 (m s^{-1}), α_1 and α_2 are the velocity weighting coefficients at sections 1 and 2 (-), g is the gravitational acceleration (-) and h_e is the energy head loss (-). This equation illustrates that energy cannot be created or destroyed but does change as flow moves from location to location (Heastad methods *et al.,* 2003) and is used to calculate the water surface elevation from one cross-section to the next. The energy head loss component quantifies the friction losses and contraction and expansion losses between two cross-sections and is used to account for the change in shape and change in friction between two points in the channel. It is calculated through:

$$h_e = L\overline{S}_f + C\left|\frac{\alpha_2 V_2^2}{2g} - \frac{\alpha_1 V_1^2}{2g}\right| \qquad \text{Equation 6.9}$$

where L is the discharge weighted reach length (m), $\overline{S}_f$ is the representative friction slope between the two sections (-) and C is the expansion and contraction loss coefficients (-), (Tayefi, 2005).

The distance-weighted reach length, *L,* is calculated as:

$$L = \frac{L_{lob}\overline{Q}_{lob} + L_{ch}\overline{Q}_{ch} + L_{rob}\overline{Q}_{rob}}{\overline{Q}_{lob} + \overline{Q}_{ch} + \overline{Q}_{rob}} \qquad \text{Equation 6.10}$$

where L_{lob}, L_{ch} and L_{rob} are the cross-section reach lengths specified for flow in the left overbank, main channel and right overbank respectively (m), and Q is the arithmetic average of the flows between sections for the channel and each of the banks ($m^3\ s^{-1}$).

The representative friction slope for a reach (along with distance weighted reach length) is used in HEC-RAS to evaluate the friction loss and is calculated thus:

$$\overline{S}_f = \left(\frac{Q_1 + Q_2}{K_1 + K_2}\right)^2 \qquad \text{Equation 6.11}$$

where subscripts 1 and 2 relate to variable values at the upstream and downstream end of the reach under study, and *Q* and *K* are the total discharge ($m^3\ s^{-1}$) and conveyance (-) for a cross-section, respectively. Channel conveyance has been defined as a measure of the discharge carrying capacity of a channel.

HEC-RAS uses the following equation to calculate contraction and expansion losses:

$$h_{ce} = C\left|\frac{\alpha_1 V_1^2}{2g} - \frac{\alpha_2 V_2^2}{2g}\right| \qquad \text{Equation 6.12}$$

where *C* is the contraction or expansion coefficient (-). By default, expansion is usually expressed as 0.1 and contraction as 0.3.

The next section will outline the steps of model development, as specified in Figure 6.1. All processes from this point forward were carried out with reference to HEC-RAS.

6.3.3 Hydrological boundary conditions

The boundary conditions required in HEC-RAS include: (1) upstream input (hydrology); (2) downstream input (hydrology); and (3) geometry (cross-sections and roughness). This Section will outline the importance of each data set and describe data collection methods.

Hydrology data are required at the input and output boundaries to define the quantity of flow running through a reach in a specified time. The hydrology data used for the input and output (or upstream and downstream) boundary conditions were obtained from existing Environment Agency flow records at the Eddy's Bridge (upstream) and Rowlands Gill (downstream) gauging stations. The location of these gauging stations can be seen in Figure 6.2. Data are available for 15-minute flow intervals as well as daily mean flows. For the purposes of the model data input, the 15-minute flow data were used as it is more accurate and more representative of the flow conditions on a finer timescale. Flow accretion from tributaries was not accounted for in the model as the tributaries are ungauged. However, consideration was given to this in the description of the calibration process and in the analysis of results.

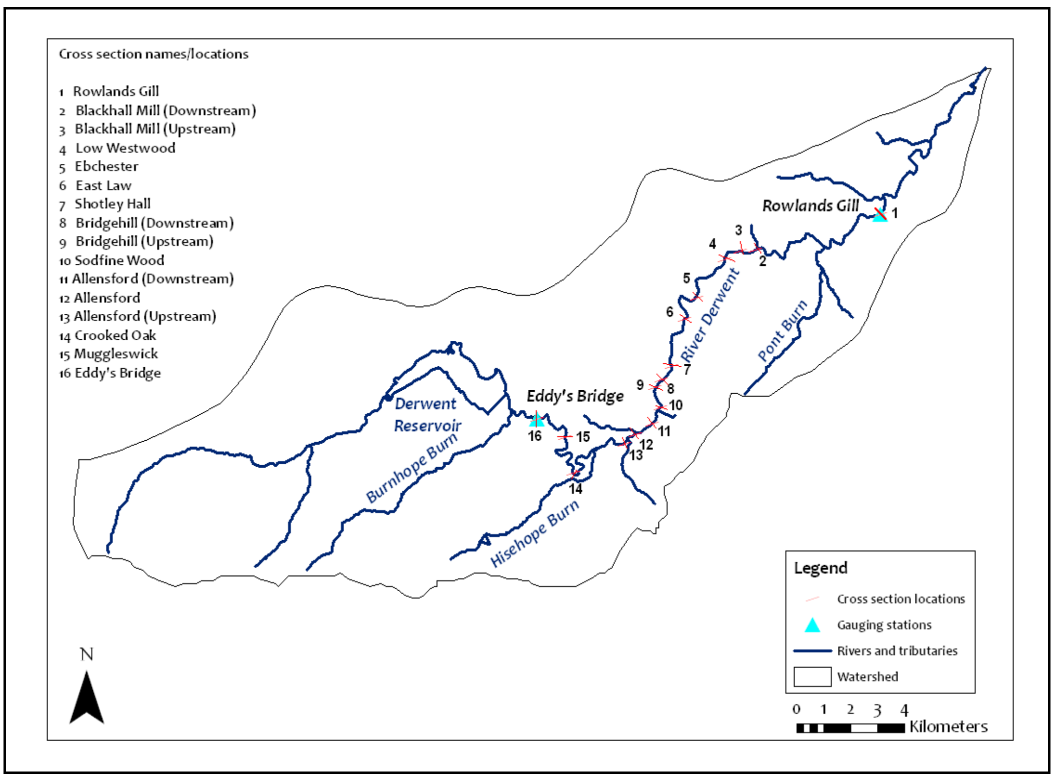

Figure 6.2 Cross-section locations on the River Derwent for use in HEC-RAS flow modelling

Geometry

The cross-sections are used to inform the model of how channel form and slope change with distance downstream. The geometry data define channel and floodplain capacity and are coupled with hydrology data to determine conveyance of water downstream.

Cross-sections were collected from 16 locations along the length of the river, with roughly 1 km spacing between each section. The locations of the sections are illustrated in Figure 6.2. It was decided that 16 cross-sections was an optimum balance between geometry data resolution and time available for data collection. Locations were chosen based on a number of criteria, including significant change in channel form and/or size, distance from last cross-section and accessibility. Downstream of site 2, (Figure 6.2), the river became too wide and deep to be accessible. Therefore, cross-sections did not extend beyond this point, although one was available at the downstream boundary from Environment Agency hydrology archives.

The cross-sections were collected using a Leica total station EDM. Points were recorded every 0.5-1 m, or where there was a significant change in slope. The location of the cross-sections and individual points were georeferenced by obtaining a grid reference using a GPS where a reading could be obtained, and extrapolating to the other points. Once obtained, the points from the EDM were processed using Leica GeoOffice software to produce a set of data containing distance and height information, for each point, with reference to the fixed station. These data were entered into the HEC-RAS Geometric data tool, along with distance between each cross-section (for channel, left bank and right bank), elevation (for each cross-section station) and roughness (Manning's n, for each cross-section: channel, left bank and right bank). Table 6.1 lists the name, location and relative position of cross-section measured.

				Roughness (Manning's n)			Distance to downstream cross-section (m)		
No	**Name**	**Grid reference**	**Elevation (mAOD)**	**LOB**	**Channel**	**ROB**	**LOB**	**Channel**	**ROB**
1	Rowlands Gill	416815 558094	26	0.07	0.011	0.07	0	0	0
2	Blackhall Mill (Downstream)	412173 556839	54	0.03	0.035	0.035	7104.4	6999.1	7064.8
3	Blackhall Mill (Upstream)	411612 556816	55	0.025	0.045	0.025	598.8	595.5	593.7
4	Low Westwood	411060 556490	60	0.025	0.035	0.08	795.2	794.1	805.1
5	Ebchester	409999 555112	70	0.04	0.045	0.025	2108.8	2057.5	2087.1
6	East Law	409559 554349	74	0.07	0.045	0.07	1625.6	1624.4	1641.8
7	Shotley Hall	409086 552696	90	0.07	0.045	0.07	2011.9	1992.4	2016.1
8	Bridgehill (Downstream)	408720 552190	95	0.035	0.045	0.04	763.4	686.6	686.4
9	Bridgehill (Upstream)	408480 551930	99	0.04	0.045	0.04	351	346.3	363.6
10	Sodfine Wood	4 08676 551196	112	0.025	0.05	0.025	888.3	840.9	861.7
11	Allensford (Downstream)	408349 550625	115	0.035	0.045	0.035	734.9	717.4	714.1
12	Allensford	407756 550254	116	0.025	0.045	0.04	758.6	748.1	754.2
13	Allensford (Upstream)	407340 549890	120	0.035	0.045	0.04	600.6	595.8	604.4
14	Crooked Oak	405456 548853	161	0.035	0.045	0.04	3472.1	3418.3	3458.3
15	Muggleswick	405140 550150	180	0.035	0.045	0.035	2629.3	2607.4	2631.5
16	Eddy's Bridge	404107 550791	183	0.025	0.011	0.08	1871.9	1808.3	2051.4

Table 6.1 Number, name, elevation, roughness and downstream distance for Left Over Bank (LOB), Chanel and Right Over Bank (ROB) of each site investigated

Type of Channel and description	*Minimun*	*Normal*	*Maximium*
A). Natural streams			
Main Channels			
a. Clean, straight, full no rifts or pools	0.025	0.030	0.033
b. Same as above, but more stones and weeds	0.030	0.035	0.040
c. Clean, winding, some pools and shoals	0.033	0.040	0.045
d. Same as above, but some weeds and stones.	0.035	0.045	0.050
e. Same as above, lower stages, more ineffective slopes and sections	0.040	0.048	0.055
f. Same ad 'd' but more stones	0.045	0.050	0.060
g. Sluggish reahes, weedy, deep pools.	0.050	0.070	0.080
h. Very weedy reaches, deep pools, or floodways with heavy stands of timber and brush.	0.070	0.1	0.150
2. Floodplains			
a. Pasture no brush			
i). Short grass	0.025	0.030	0.035
ii). High grass	0.030	0.035	0.050
b. Cultivated areas			
i). No crop	0.020	0.030	0.040
ii). Mature row crops	0.025	0.035	0.045
iii). Mature field crops	0.030	0.040	0.050
c. Brush			
i). Scattered brush, heavy weed	0.035	0.050	0.070
ii) Light brush and trees in winter	0.035	0.050	0.060
iii) Light brush and trees in summer	0.040	0.060	0.080
iv). Medium to dense brush, in winter	0.045	0.070	0.110
v). Medium to dense brush, in summer	0.070	0.100	0.160
d). Trees			
i). Cleared land with tree stumps, no sprouts	0.030	0.040	0.050
ii). Same as above, but heavy sprouts	0.050	0.060	0.080
iii). Heavy stands of timber, few down trees, little undergrowth, flow below branches	0.080	0.100	0.120
iv). Sames as above but with flow into branches	0.100	0.120	0.160
v). Dense willows, summer, straight.	0.110	0.150	0.200

Table 6.2 Descriptors used in the allocation of Manning's n values for each cross-section (Source: HEC-RAS user manual, 2009).

Roughness

Tables 6.1 and 6.2 list the roughness values used for each cross-section. The Manning's n values are used as a measure of roughness to inform the model of the friction that would be applied to flow at each section. The amount of friction exerted affects the conveyance of water and therefore the speed at which water can be moved through a channel. Combined with channel shape, this determines the depth of flow within a channel at a certain point. Application of Manning's n values contains an element of subjectivity, but there are standard values applied to certain types of channel and floodplain. These are well documented in HEC-RAS and other sources (e.g. USGS). By comparing photographs from USGS and Chow (1959) to observations and photographs of each reach, suitable Manning's n values were obtained, based on channel shape, substrate size and form, vegetation and floodplain characteristics. The values used are listed in Tables 6.1 and 6.2 and reflect the change in channel and floodplain form with distance downstream. For the purposes of this investigation, roughness values did not change as a function of stage, although it is noted by Ferguson (2007) that the impact of roughness will change according to water depth – with skin friction and large scale form resistance (e.g. bedforms) being the dominant physical form of flow resistance at high flows and form drag from large roughness elements being more important at lower flows. Ferguson (2007) proposed a Variable Power Equation (VPE) to allow a model to predict velocity for a wide range of flows, using a single equation. This could not be incorporated into the HEC-RAS software but is an important consideration for future investigations.

A sample flow event was chosen from the EA hydrology data for the input boundary (Eddy's Bridge) and applied to the model in the first instance. The output discharge data were then compared to the observed data at the downstream boundary (Rowlands Gill, cross-section 1). Sensitivity analyses for changing the roughness coefficient (n) were conducted on the data. It was found that, at this point, variation of the roughness coefficient had only a small impact on depth and velocity, except when it was elevated or decreased to such an extreme level that the model became unstable. An illustration of the effect of changing Manning's n is included in Figures 6.3 and 6.4, for four cross-sections.

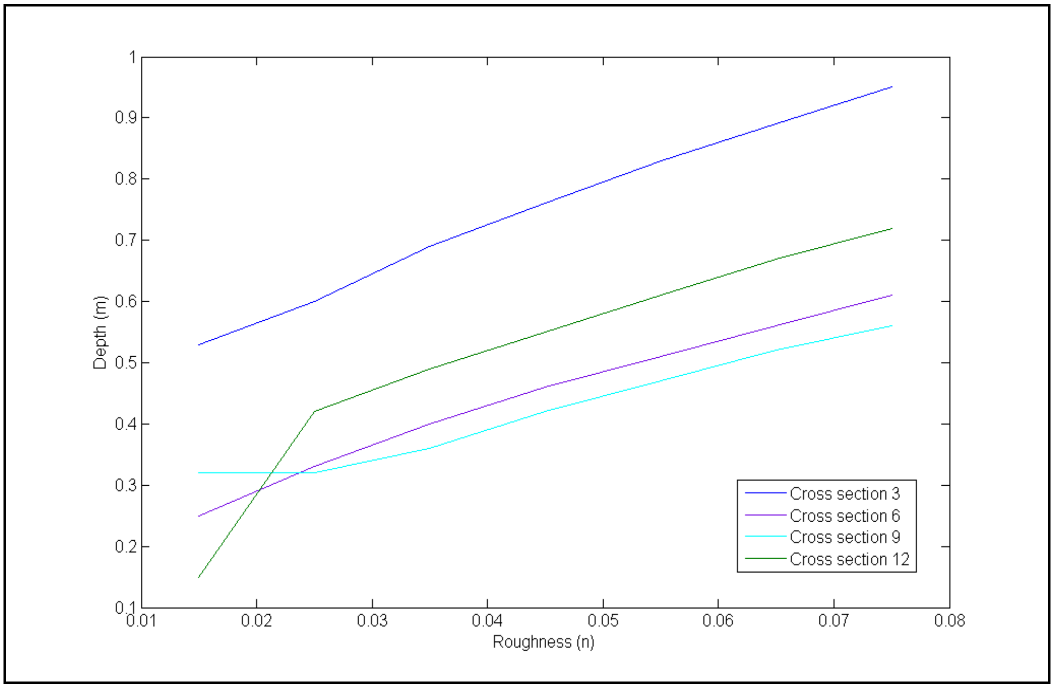

Figure 6.3 Sensitivity analysis of roughness coefficient on depth outputs for four cross-sections, based on a steady state simulation at a rate of 5 $m^3 s^{-1}$

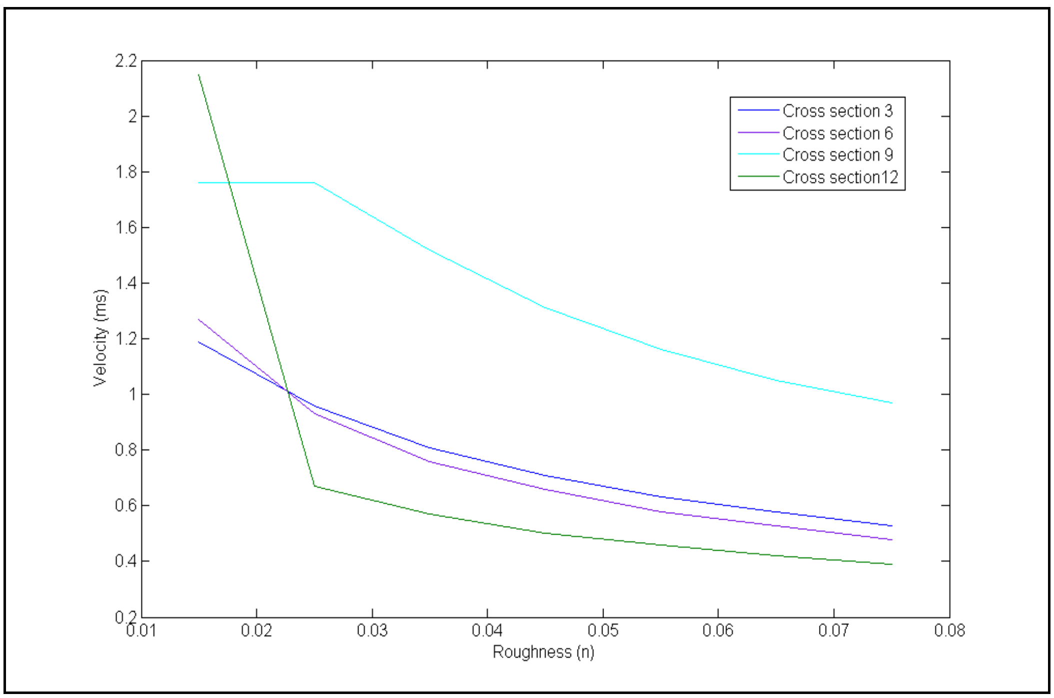

Figure 6.4 Sensitivity analysis of roughness coefficient on velocity outputs for four cross-sections, based on a steady state simulation at a rate of 5$m^3 s^{-1}$

6.3.4 Calibration

Haestad Methods *et al.,* (2003, p 159) defines calibration as "[adjustment] of model parameters (within reason) so that predicted system performance agrees with observed system performance over a range of conditions". Haestad Methods *et al*., (2003, p 154) note that calibration of Manning's n to gauged flow data is an important process in ensuring that appropriate roughness values have been applied to the model. It is also noted that output results are more sensitive to the in-channel n value than that of the floodplain. Therefore it is imperative that the in-channel n values are appropriate for the channel being investigated. For this purpose, the method described by Haestad Methods *et al.,* (2003, p 154) was followed, which is simply to adjust roughness values within allowable limits to reproduce the known stage for the measured discharge. It is also noted at this point that even with known roughness data, a perfect match between the model's output and the known river data should not be expected. An error of 5% can be expected, which may translate to a computed water-surface elevation error of 0.15 m or greater.

An attempt was made to calibrate HEC-RAS to the River Derwent by applying the boundary data and operating the model for a high flow event. The modelled output of discharge and stage data were compared to the observed equivalents at Rowlands Gill and the error calculated (Figure 6.5). As the model appeared to represent the shape of the hydrograph well, but encountered a major shortfall in *volume* of water, adjustment of the Manning's n for various cross-sections was carried out. Table 6.3 shows how the n values for the main channel, (as these are considered more important than those of the floodplain (Haestad Methods, 2003)), were altered. This was also carried out with left over-bank and right over-bank roughness values. Manning's n values (channel, left over-bank and right over-bank) for all cross-sections were altered simultaneously and then separately. This effect did not change the output hydrograph. Extensive adjustment of Manning's n made little difference to the volume of water that was being accumulated through the river (Figure 6.7 'roughness coefficient'), therefore, external factors were considered.

Cross-section	Original Manning's n value (channel)	n+0.01	n+0.02	n+0.03	n-0.01	n-0.02	n-0.03
1	0.011	0.021	0.031	0.041	0.035	0.0001	0.0001
2	0.045	0.045	0.055	0.065	0.025	0.015	0.005
3	0.045	0.055	0.065	0.075	0.035	0.025	0.015
4	0.045	0.045	0.055	0.065	0.025	0.015	0.005
5	0.045	0.055	0.065	0.075	0.035	0.025	0.015
6	0.045	0.055	0.065	0.075	0.035	0.025	0.015
7	0.05	0.055	0.065	0.075	0.035	0.025	0.015
8	0.045	0.055	0.065	0.075	0.035	0.025	0.015
9	0.045	0.055	0.065	0.075	0.035	0.025	0.015
10	0.045	0.06	0.07	0.08	0.04	0.03	0.02
11	0.045	0.055	0.065	0.075	0.035	0.025	0.015
12	0.045	0.055	0.065	0.075	0.035	0.025	0.015
13	0.035	0.055	0.065	0.075	0.035	0.025	0.015
14	0.045	0.055	0.065	0.075	0.035	0.025	0.015
15	0.035	0.055	0.065	0.075	0.035	0.025	0.015
16	0.011	0.021	0.031	0.041	0.035	0.0001	0.0001

Table 6.3 Manning's n combinations used in model calibration, with the most appropriate set of n values for the River Derwent highlighted

The distance between the upstream and the downstream boundaries is 25 km and, as there are no gauged tributaries, these were not added as a feature of the model. There are a number of tributaries which enter the river between Eddy's Bridge and Rowlands Gill (including Burnhope Burn (catchment area 24 km^2, Pont Burn, Mill Burn and Horsley Hope Burn, Figure 6.2) and although small, they will input water that has not been accounted for by HEC-RAS. Catchment area to Eddy's bridge (the upstream gauging station) is 118 km^2 and to Rowlands Gill (downstream boundary) is 242 km^2. Therefore, more than double the amount of water can be expected to pass through the downstream gauging station of Rowlands Gill as does through the upstream station of Eddy's Bridge (*more* than double because flows from upstream of Eddy's Bridge are attenuated by the reservoir). Additionally, there are a number of discharges to the main River Derwent from the water treatment works – these contribute water to the channel and are also absent from the model. Figure 6.8 highlights the extent of these inputs and relates them to the shortfall in flow. Further to this, it is known that there is an issue at Rowlands Gill gauging station with the accumulation of coarse sediment over the weir (http://www.environment-agency.gov.uk/hiflows/station.aspx?23007, 9/10/2009). The sediment accumulation will elevate the stage of the water surface and when a rating equation is used to calculate the discharge based on stage (Equation 6.13), there will be an over-estimation of discharge.

$$Q=K*((h+a)^p) \qquad \textbf{Equation 6.13}$$

where Q is discharge ($m^3\ s^{-1}$), K is the multiplier component of the rating equation (-); a is the intercept component of the rating equation (-); h is stage (m) and p is an exponent component of the rating equation (-). For the River Derwent, the various components are as detailed in Table 6.4.

Maximum stage (m)	***K***	***a***	***p***
0.16	13.723	0	1.294
0.59	53.46	0	2.028
5	44	0	1.658

Table 6.4 Multiplier, intercept and exponent components of the rating equation for the River Derwent, Northumberland

For example, in this equation, when stage is 1 m, the discharge is calculated as 3801 Ml/d (44 $m^3\ s^{-1}$). But if 1 m of sediment were to accumulate on the river bed, the water surface would be elevated to 2 m and the discharge would be calculated as 11997 Ml/d (138.9 $m^3\ s^{-1}$).

Consequently, the model cannot be validated with the boundary data available and therefore, the actual outputs of the model cannot be used to determine the exact effect of flow regimes in the River Derwent. However, the outputs from the model can be applied to provide an *indication* of what changes may occur to flow characteristics (e.g. depth, velocity) and to test the application of the fuzzy rule-based method to such an investigation (Chapter Seven).

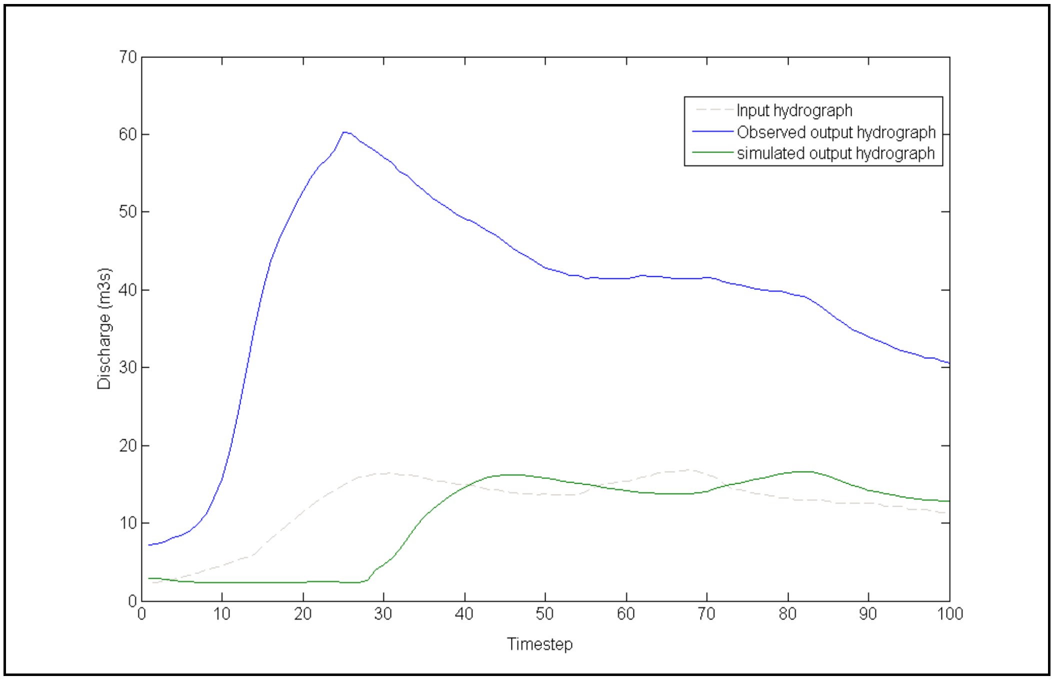

Figure 6.5 Initial HEC-RAS discharge prediction for a flow event using original cross-section and roughness data

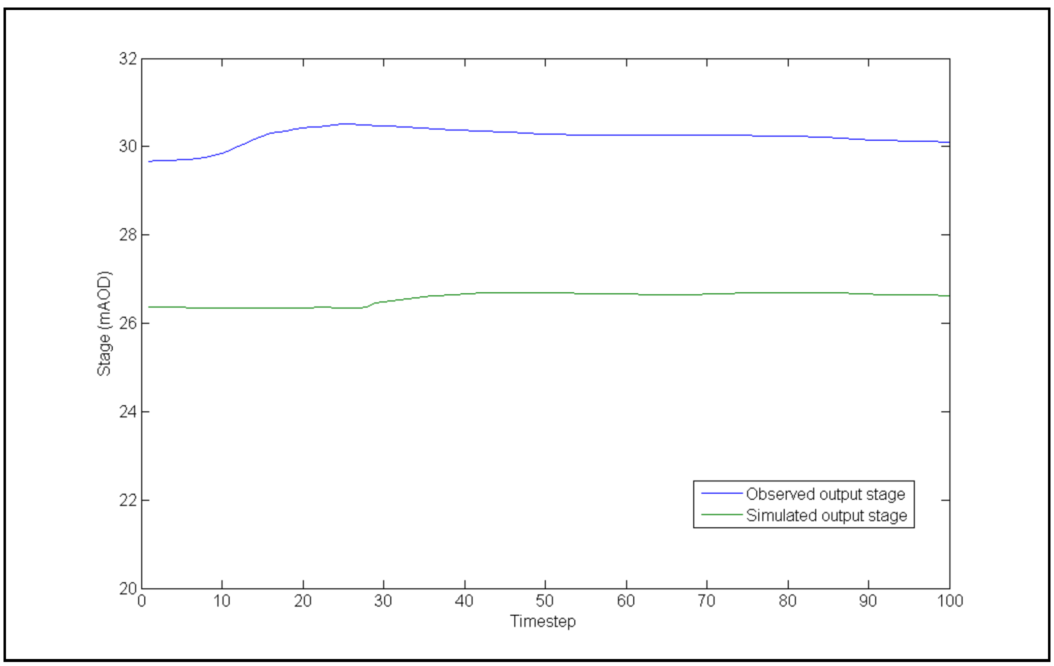

Figure 6.6 Initial HEC-RAS stage prediction for a flow event using original cross-section and roughness data

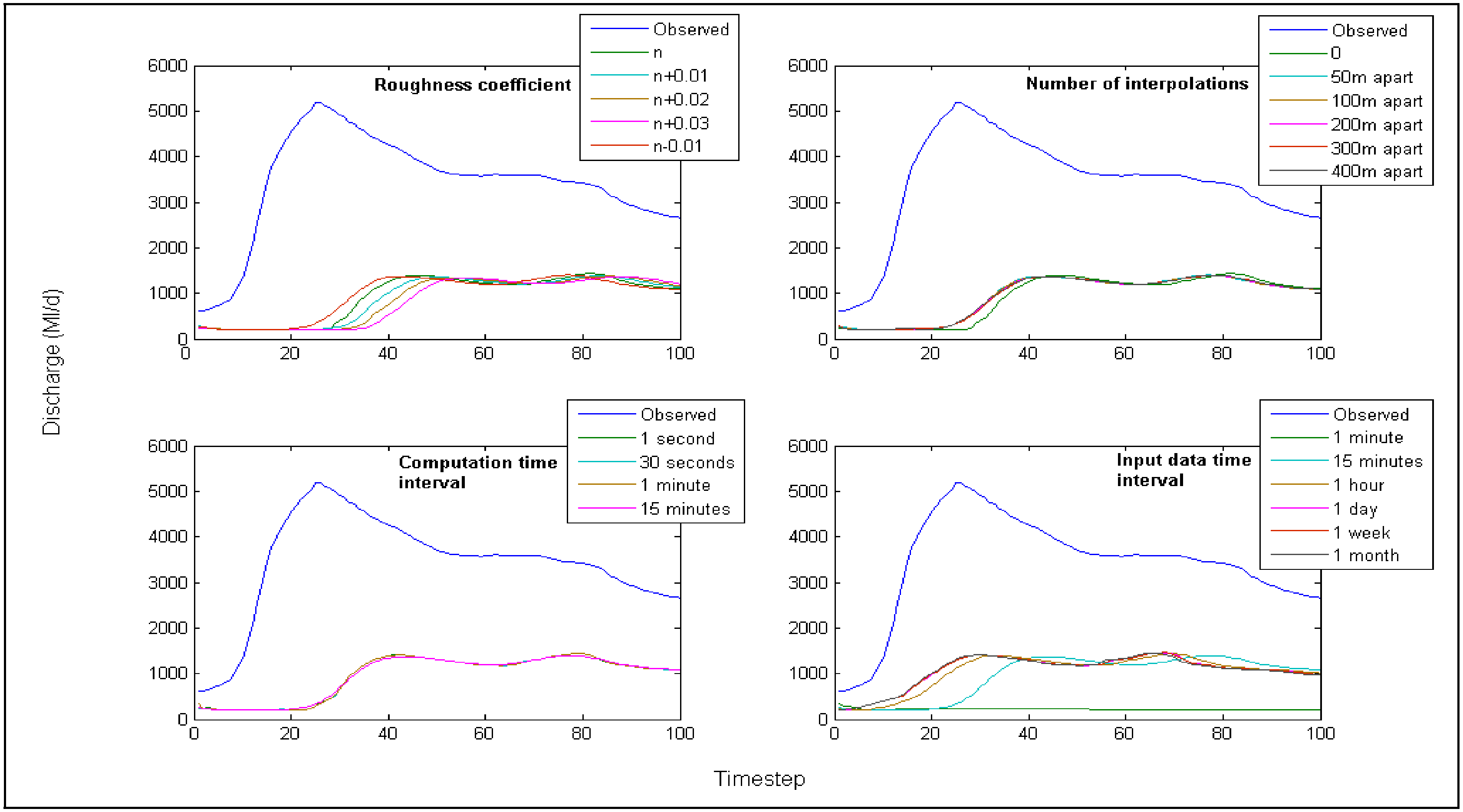

Figure 6.7 Calibration and verification of HEC-RAS for the River Derwent

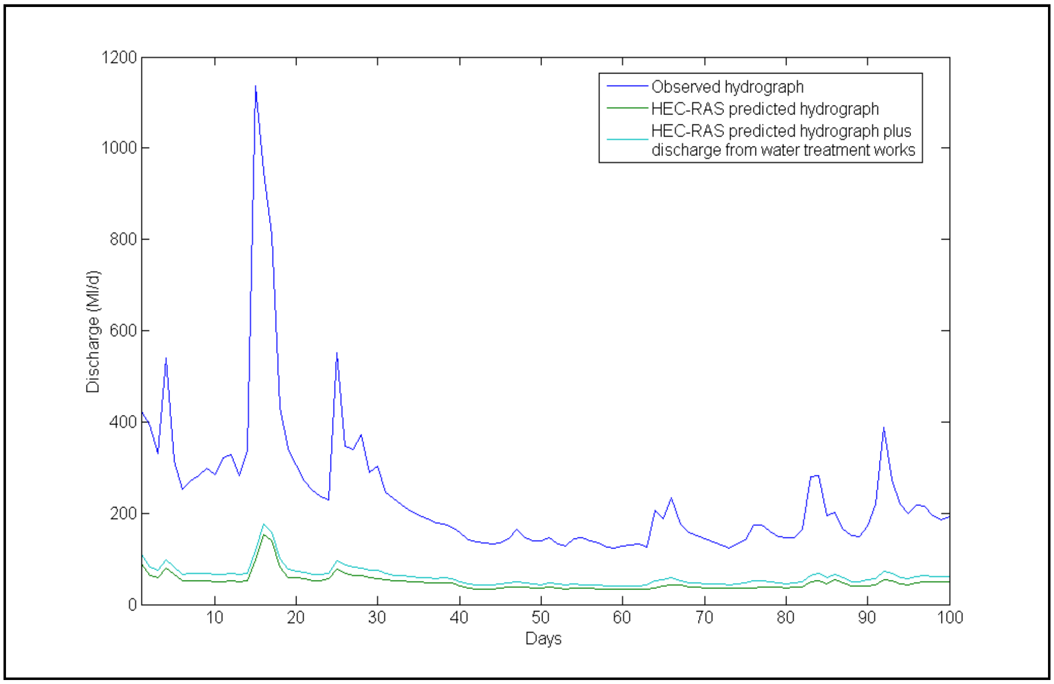

Figure 6.8 Impact of water treatment works discharges on hydrograph

6.3.5 Verification and validation

Verification

Verification can be defined as the correct solution of associated equations within a model (Lane and Richards, 2001) and this may done through the comparison of numerical solution with an analytical one (Oreskes *et al*., 1994). Lane *et al*., (2005) list a number of criteria important to the verification of a model (standard of reporting; level of solution accuracy in space; mesh independence testing; determination of solution convergence; solution accuracy in time; specification of boundary conditions; reporting of code, use of benchmark solutions and comparison with experimental results). These processes are considered necessary to ensure that equations are being solved correctly. Verification involves checking for minimisation of coding errors as well as errors associated with both spatial and temporal discretisation and numerical solution (Hardy *et al*., 2003). Therefore, a sensitivity analysis was conducted to ensure the model for this project was spatially and temporally discretisation independent. This involved the application of the model with some spatial or temporal scale used as a variable. For example, spatial discretisation was tested by altering the number of interpolations between cross-sections and comparing output hydrographs. Temporal scales were examined by varying input data time interval results. Sensitivity to computation time interval was also examined. The variables used in the sensitivity analysis can be seen in Table 6.5 and the results from the sensitivity analysis are included in Figure 6.7. It is often thought that increasing the spatial and temporal resolution will increase a model's predictive ability because of: i) expected improvements in solution stability as the grid spacing tends towards the true continuum level; ii) the ability of high-resolution models to facilitate complex, and thereby more realistic parameterisation of the code; and iii) a closer correspondence of field measurement model scales (Hardy *et al*., 1999). Based on this, when the model produced the same results, regardless of timestep, or number of interpolations used, then it can be deemed robust and suitable for application. In this case, the model appeared to be verified spatially, and temporally for the computation time interval, although there was some sensitivity to roughness and input data time interval (Figure 6.7).

Roughness coefficient (Manning's n)	**Maximum distance between interpolations (m)**	**Computation time interval**	**Input data time interval**
n	0	1 second	1 minute
n+0.01	50	30 seconds	15 minutes
n+0.02	100	1 minute	1 hour
n+0.03	200	15 minutes	1 day
n-0.01	300		1 week
Beyond +0.03 and -0.01 were unstable	400		1 month

Table 6.5 Variables used in sensitivity analysis for model verification

Validation

For validation purposes, a model is run with a new set of actual flow data that have not been used in calibration. No further alteration of the parameters such as roughness, is used. If the model predicts well the gauged flows, it can be considered suitable for application to all other flood events (Haestad Methods *et al.,* 2003). Two new flow events were chosen, including a low and high flow, and the model operated for these flows. The outputs were compared to the observed flows (Figure 6.9). As discussed in the above paragraph, the model was unable to effectively reproduce the flow data collected from the output boundary at Rowlands Gill and therefore it cannot be said that the model was validated. However, the output data can be used as a guide to what may happen and used as a tool on which to test the fuzzy modelling in Chapter Seven.

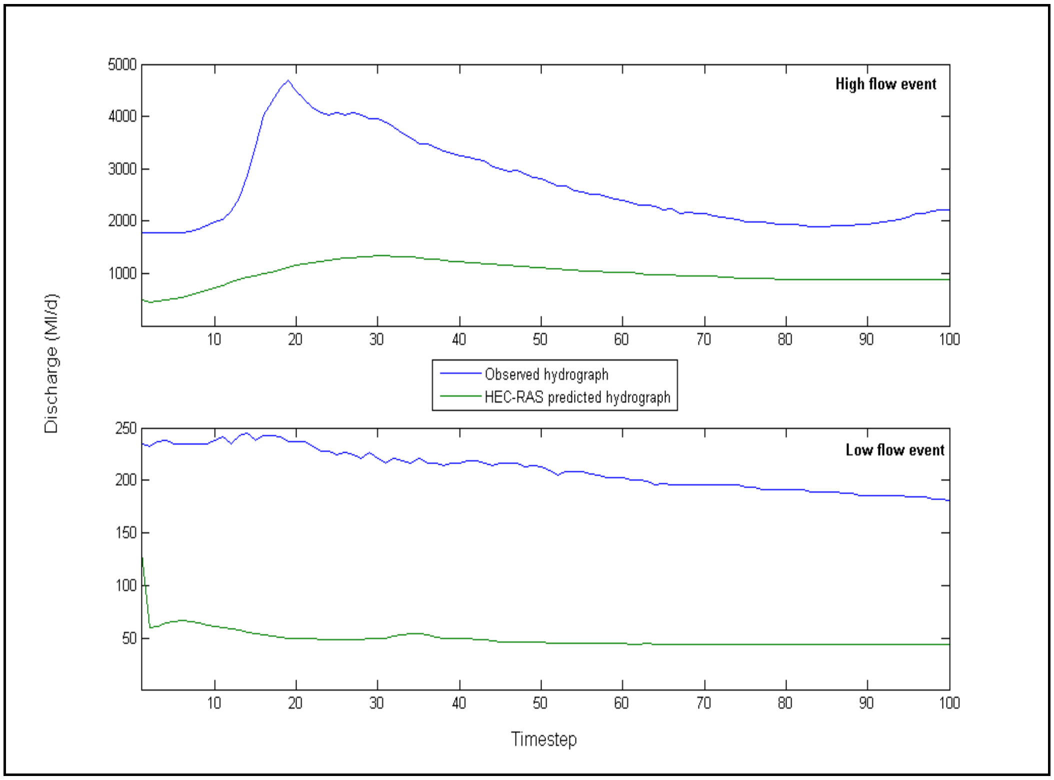

Figure 6.9 Validation test results for the River Derwent. Shape of hydrograph is being reproduced but magnitude of flow is underestimated

6.3.6 Application

The outputs from HEC-RAS can be used as a data set on which to conduct an investigation into the effectiveness of fuzzy rule-based modelling. The following Section describes how the model was applied and output data used.

Two of the most important variables to fish in habitat selection are depth and velocity as well as the flow patterns and channel forms which develop as a result of the combination of these two parameters. Other important features include sediment size and temperature; however, these could not be modelled within the version of HEC-RAS developed here and therefore, the focus of this Section will be upon the changes to depth and velocity as input boundaries are altered. Two forms of analysis were undertaken:

- **Steady state analysis** was applied to a number of different flow magnitudes. This was to examine the response of the channel to a number of hypothetical flows and was carried out as a steady state simulation because only one discharge was being investigated at a time. Once the three-month changes to flow were introduced, the simulation became an unsteady one, but with only two different discharges in the input data.

- **Hydrograph analysis** was conducted on the actual hydrograph for the water year 2002-2003 to investigate the effect of different flow regimes when applied to an actual hydrological situation. This was carried out as an unsteady simulation as discharge changed on a daily basis.

More details are provided in the relevant Sections as to how the model was applied for each specific scenario.

A pilot test was conducted to determine the effect of applying the model to different time scales. Flow data were entered as monthly, daily, hourly and minute data and the depth and velocity results assessed to detect any variation in output. The input-data timestep used did not impact upon results and therefore, smaller timesteps (daily) were used (instead of monthly), in order to reduce computation time required. Thus, the model was run for a period of 12 days rather than 12 months, and the results produced are for daily flow rates/depths/velocities. However, the daily results are representative of the whole month because the daily rate of flow remains constant in any one month.

The first assessment was that of changes in predicted depths and velocities as a result of impoundment. Low, median and high flows were considered in order to assess the effects both at the most extreme flows and at middle value flows. The Q95, Q50 and Q5 were calculated for both the pre- and post-impoundment periods. Pre-impoundment data exist only as daily means (rather than 15-minute discharges). Therefore, to allow a comparison between pre- and post-impoundment scenarios, daily means (rather than 15-minute flow data) were used for all analyses. Six simulations were conducted, with hydrological input boundaries as described in Table 6.6. The results for the pre- and post-impoundment simulations at each discharge magnitude (i.e. Q95, Q50 and Q5) were compared to define the difference in depth and velocity that can be expected from each flow regime.

	Pre-impoundment	Post-impoundment
Q95	18.1	28.3
Q50	87.7	38.7
Q5	570.1	163.9

Table 6.6 Input boundary conditions used to assess current impact of river impoundment (Ml/d)

Following the identification of changes to flow characteristics such as depth and velocity following impoundment, the effects of the new compensation release regimes were considered. The current daily compensation release volume was subtracted from each of the post-impoundment low, median and high flows, and replaced with the new flow rates suggested by CRAB (Chapter

Five), with the proposed juvenile suitable flow in 75 of the 100 input cells and the proposed spawning flow in 25 of the 100 input cells (to represent the elevated flow that occurred for three months, or 25% of the year). The spate flow was not included at this stage as its purpose is for short term flushing effects rather than an alteration of the depth or velocity. The variables used for these analyses are included in Table 6.7. Figure 6.10 illustrates the change in low, median and high flows across the year, when the new suggested compensation release regimes are applied.

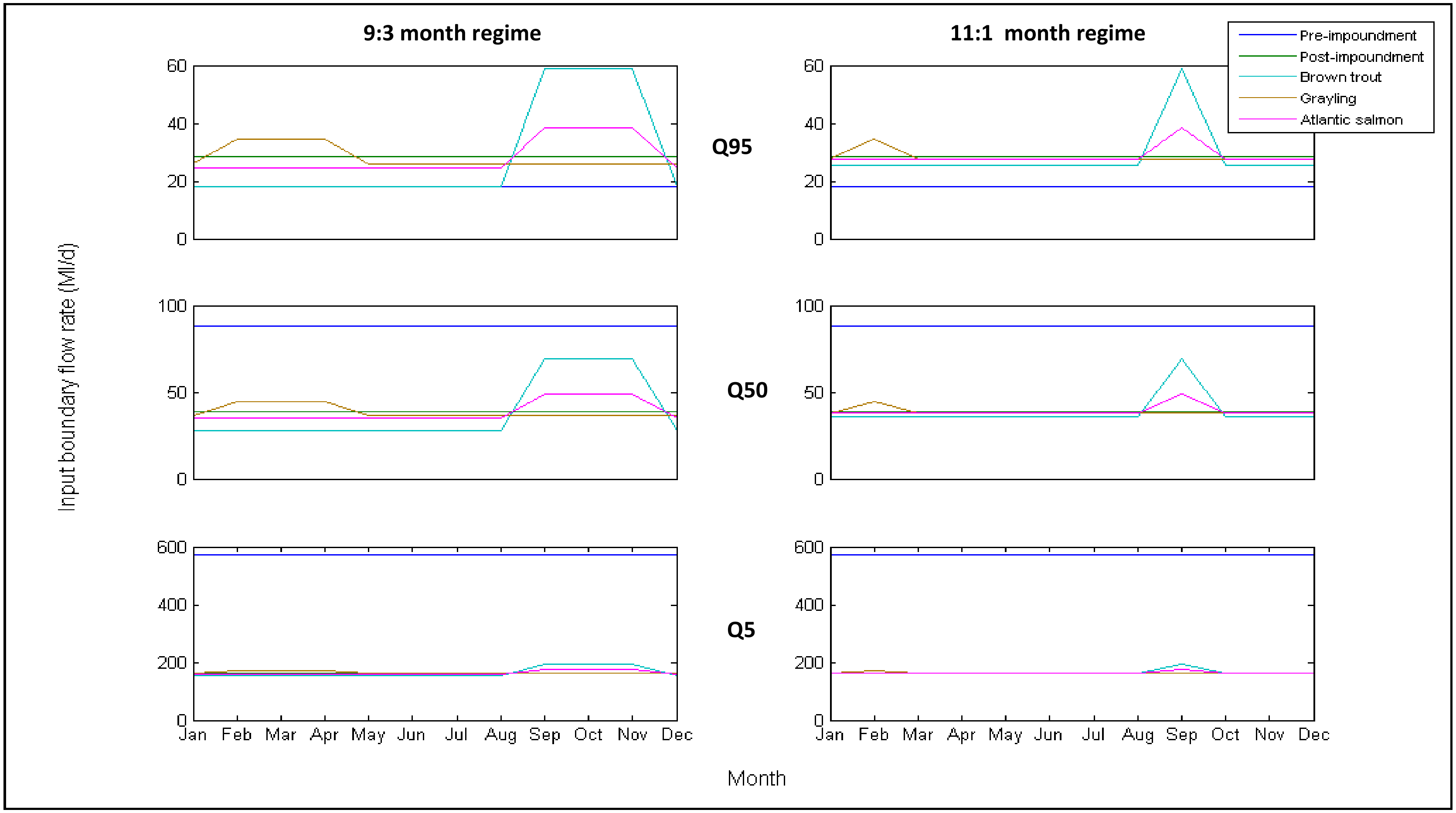

Figure 6.10 Input boundary hydrographs to test pre- post-impoundment and species specific flow regimes, for 9:3 and 11:1 months distribution

	Juvenile suitable flow			Spawning suitable flow		
	Brown trout	Grayling	Atlantic salmon	Brown trout	Grayling	Atlantic salmon
Q95	17.9	26.0	24.7	58.9	34.4	38.4
Q50	28.3	36.4	35.1	69.3	44.8	48.8
Q5	153.5	161.6	160.3	194.5	170.0	174.0

Table 6.7 Input boundary conditions used to assess impact of proposed new compensation release regimes (three months spawning flow, nine months juvenile suitable flow) (Ml/d)

As addressed in Chapter Five, consideration has been given to an elevated flow that lasts for one month instead of three – this would allow greater volumes of water to be released during the baseflow period and would cause a less severe change of environment for the fish and macroinvertebrates in the lower flow period. Therefore, the effects of these proposed flow regimes (one month spawning flow, 11 months juvenile suitable flow) on depth and velocity were also assessed using HEC-RAS and the variables used are shown in Table 6.8 and Figure 6.10.

	Juvenile suitable flow			Spawning suitable flow		
	Brown trout	Grayling	Atlantic salmon	Brown trout	Grayling	Atlantic salmon
Q95	25.6	27.8	27.4	58.9	34.4	38.4
Q50	36.0	38.2	37.8	69.3	44.8	48.8
Q5	161.2	163.4	163.0	194.5	170.0	174.0

Table 6.8 Input boundary conditions used to assess impact of proposed new compensation release regimes (one month spawning flow, 11 months juvenile suitable flow) (Ml/d)

Following the assessment of the theoretical flows, as described above, HEC-RAS was applied to an annual hydrograph, based on daily mean flow data from the water year 2002-2003. This year was chosen because it was the most recent year (to represent current conditions) in which there were good quality data available for the whole year for both the input boundary (Eddy's Bridge) and output boundary (Rowlands Gill) locations. Daily mean values (rather than 15-minute flow data) were used because of the limit in HEC-RAS to 100 data points at any time. Using 15-minute flow data for a whole year would have taken a great amount of computation time and it was felt that daily flows provided enough resolution for this task. The model was applied in four stages, assessing flows for three calendar months at a time. A pilot test was conducted to ensure that outputs for a certain time period were the same regardless of whether the time period occurred at the start, middle or end of the simulation. For instance, the model was run for the period October to December and then for December to February – the depth and velocity results for December from each run were compared and found to be the same. The method followed that used to assess the steady low, median and high flows, in which the current compensation release rate was subtracted from each daily mean and replaced by the new suggested compensation release rates for each species. The original 2002-2003 hydrograph is shown in Figure 6.11, as well as the hydrographs with the new suggested compensation release rates for each species. In the previous Section of analysis, the current compensation release rate was kept constant as the input was being treated as a steady flow. At this stage, because the flow was unsteady throughout the year, the true current compensation release rate could be used – as described in Chapter Two, this rate rises from 22.7 Ml/d between October and March to 25 Ml/d between April and September. Therefore, for the period October to March, 22.7 Ml/d were deducted from the daily mean and for the period April to September, 25 Ml/d were deducted. However, the new compensation release rates were applied for the juvenile period of December to August and the spawning flow period of September to November, for the brown trout and Atlantic salmon regimes (and spawning period for the grayling regime was applied between February and April). This is to create suitable habitat at different times of year (Chapter Five).

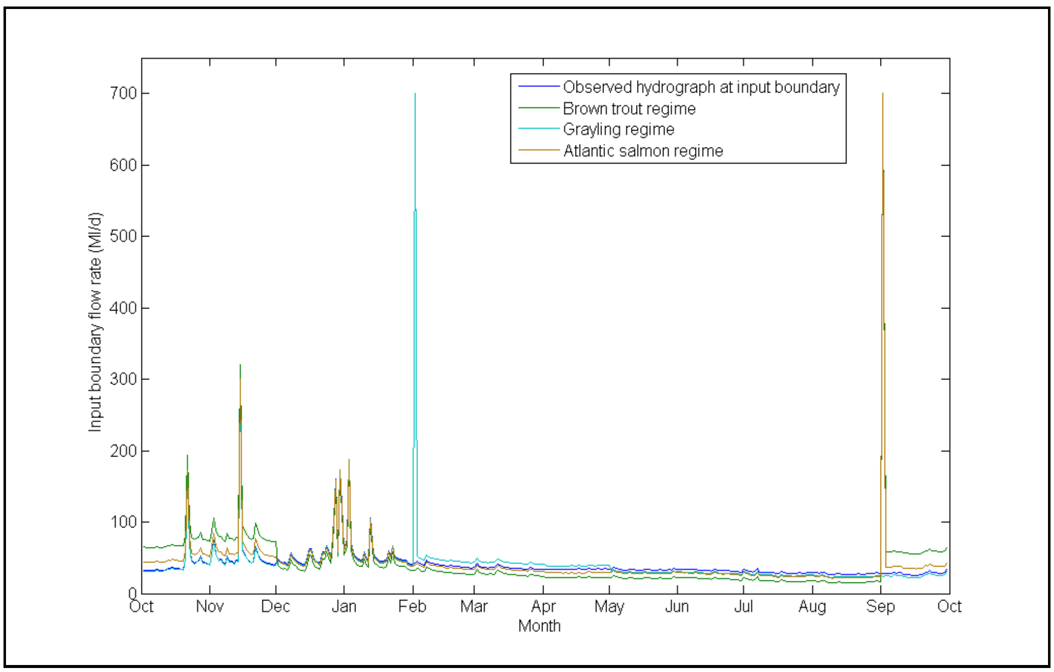

Figure 6.11 Input boundary flow data for current, brown trout, grayling and Atlantic salmon flow regimes, based on gauged discharge data from the water year 2002-2003 at Eddy's Bridge

The model outputs of flow depth and velocity at each cross-section for different times of year were analysed to determine the effects of the new compensation release regime on a typical year's flow and were also used in the fuzzy modelling component, to be addressed in Chapter Seven.

6.4 Results

6.4.1 Introduction

For the purposes of this assessment, the results of the most downstream site (site 1: Rowlands Gill) have not been considered as the weir causes an unnatural channel form and low resistance to occur which has caused depths and velocities to appear anomalous. In order to get a realistic reflection of depth and velocity distributions, site 1 was included in figures, but omitted from discussions and quantifications of changes to depth and velocity.

6.4.2 Steady state, hypothetical low, median and high flows

Effects of impoundment

The modelling study supports the evidence presented in Chapter Three that there has been a significant change to the flow regime as a result of impoundment and quantifies this for flows of Q95, Q50 and Q5 in terms of depth and velocity. At what is considered a low flow for the post-impoundment period (the Q95) there has been an increase in depth at all sites, with the exception of one site: cross-section 5. There has also been an increase in velocity, with the exception of the first, second and fifth sites (Figure 6.12). According to HEC-RAS, the differences between pre- and post-impoundment depths and velocities are slight: 0.01-0.19 m for depth and 0.01-0.1 m s^{-1} for velocity. However, it should be noted that HEC-RAS has a tendency to under-estimate flow magnitude and so it is possible that the actual differences are greater. At both median (Q50) and high (Q5) flows, depth and velocity have decreased post-impoundment. Differences in depth are more pronounced as flows increase. Q95 flow depths differ within the range of 0.01-0.06 m while the Q50 differences range from 0.04 to 0.18 m and the Q5 differences range from 0.13 to 0.47 m. A similar effect occurred with velocity where differences at low flow (Q95) range between 0 and 0.11 m s^{-1}, Q50 differences range from 0.01 to 0.27 and the Q5 differences range from 0.2 to 0.54 m s^{-1}. At median flows (Q50) there is one anomaly (an increase rather than a decrease) for both depth and velocity. For depth there is an increase of 0.04 m and this occurs at site 10 and for velocity there is an increase of 0.07 m s^{-1} at the next downstream site (site 9). There has been a negligible impact on spatial distribution of depth and velocity patterns. On the whole, the sites with the highest depths pre-impoundment tend to have the highest depths post-impoundment. The same is apparent for velocity. Again, this trend is more clearly defined at times of high flow (Q5), and less defined at lower flows (Q95 and to some extent Q50), during which time there is some change of sites that have the highest/lowest flow depths/velocities.

The difference in range of depths between sites has been most pronounced at high flows. At low and median flows, the range of depths between sites is around 0.4 m both pre- and post-impoundment, whereas at higher flows (Q5), the range of depths between different sites falls from 1.05 m pre-impoundment to 0.71 m post impoundment. For velocity, there is a much less clear pattern, with the biggest difference in velocity range between sites occurring at low flow (a range of 1.19 m s^{-1} pre-impoundment and 0.66 m s^{-1} post-impoundment). For median and high flows, the ranges remain within 0.2 m s^{-1} of each other pre- and post-impoundment.

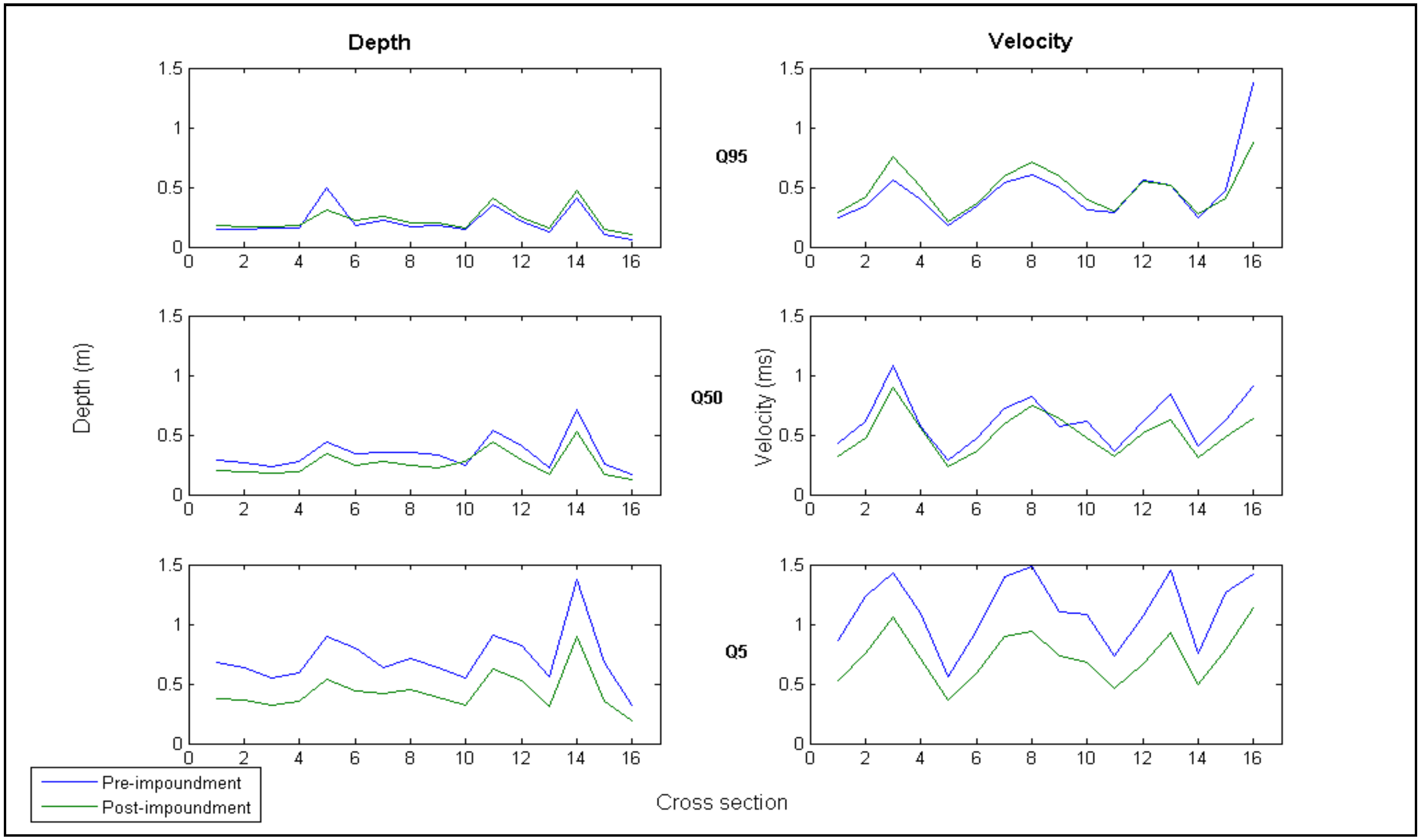

Figure 6.12 Depths and velocities produced at each site under low (Q95), median (Q50) and high (Q5) values of pre-and post-impoundment periods

Effects of new suggested compensation release regimes

The results for this Section can be split into three parts: the expected impacts of changing compensation release regime according to the depth and velocity requirements of: 1) brown trout; 2) grayling and 3) Atlantic salmon. This was done for the low, median and high (Q95, Q50 and Q5) flow values of the post-impoundment period (Section 6.3). Figures 6.13 to 6.18 show how HEC-RAS predicts the river would respond to changes in the compensation release regime, for each site, in comparison to the current compensation release regime and the pre-impoundment flow conditions.

Under brown trout flow requirements – 9:3 distribution

Figure 6.13 shows that under the regime suggested by CRAB, based on depth and velocity requirements of brown trout, depths during the baseflow period would closely follow the Q95 depth produced by the pre-impoundment Q95, which is lower than the Q95 produced by the current flow regime. As a result of the lowered baseflow, the elevated flow would create depths between 0.03 and 0.14 m greater than the depths produced under the current compensation release regime. A similar effect was predicted for velocity at times of low flow with the baseflow velocity being reduced to pre-impoundment levels and elevated flow velocity rising by up to 0.27 m s^{-1} above the velocities predicted by Q95 flows under the current compensation release regime. There does not appear to be any spatial pattern in the differences in depth/velocity over time (i.e. difference between elevated depth or velocity for elevated flow is no more prominent in any one reach of the river). As was discussed in the previous paragraphs, once median and high flows are reached, the regulation of the river causes the post-impoundment flows to become lower than the pre-impoundment flows. This is more exaggerated as flow increases, as is shown in Figure 6.12. By Q50, all of the flow scenarios (current compensation release regime, new suggested compensation release regime: both spawning and juvenile suitable flows) produce lower depths and velocities than the Q50 for the pre-impoundment period. At Q50, this is a very slight change, with the new suggested elevated flows producing depths at all sites within 0.06 m and 0.12 m s^{-1} of the depths produced by pre-impoundment flows. As discharge increases, there is a smaller difference in depth between the depths produced by suggested base flows and those produced by elevated flows. For example, the base flow depths at Q95 are on average 0.1 m lower than those at elevated flow. At Q5, the average difference between base flow and elevated flow depth is 0.04 m. This is true also for velocity, which has an average difference of 0.08 m s^{-1} between elevated and base flows at Q95 and an average difference of 0.05 m s^{-1} between elevated and base flows at Q5.

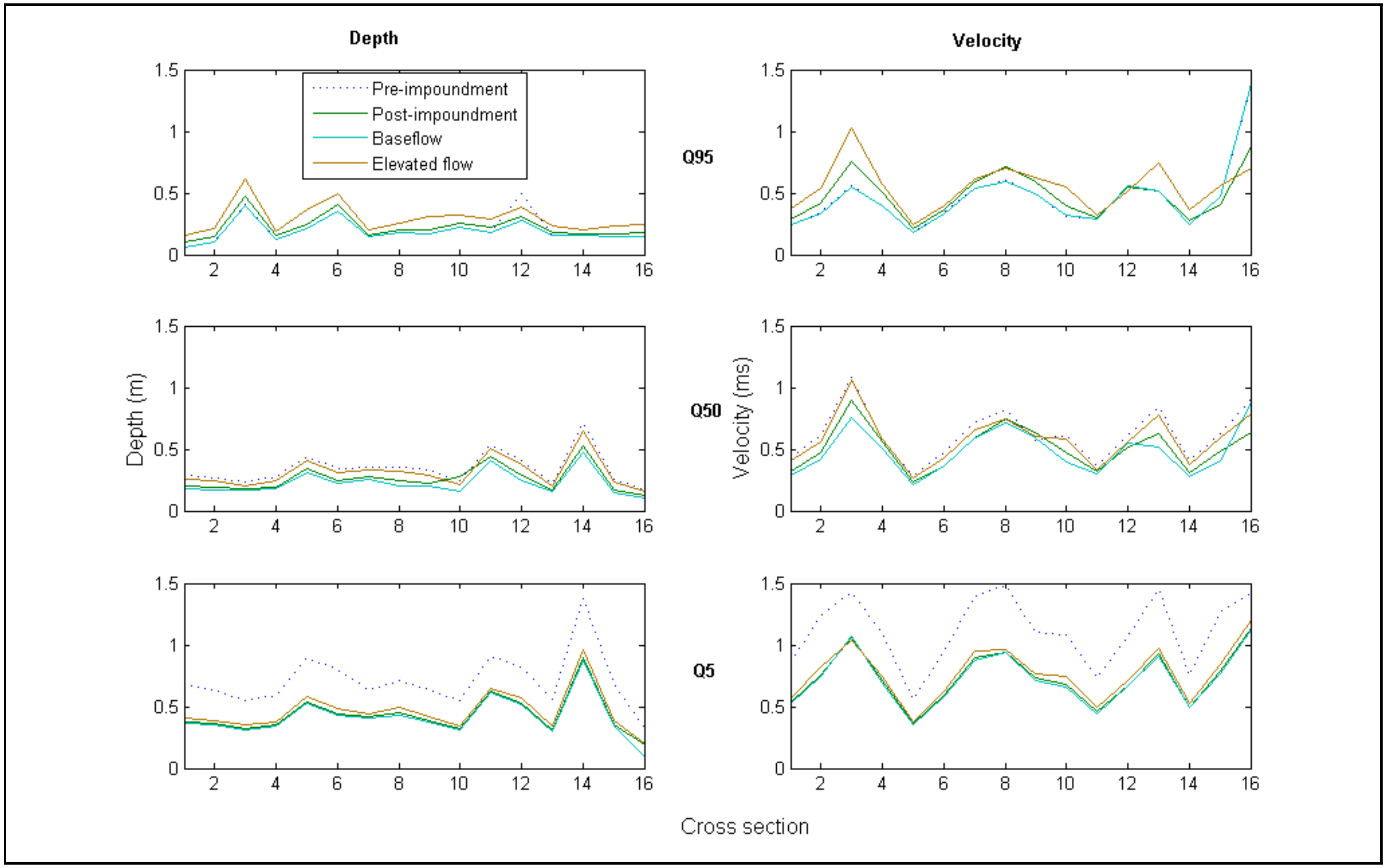

Figure 6.13 Depths and velocities produced at each site under low (Q95), median (Q50) and high (Q5) values, based on the new suggested regime designed for brown trout requirements, with pre- and post-impoundment depth and velocity results included as context

Under grayling flow requirements – 9:3 distribution

A similar *pattern* of events occurred under the compensation release regime designed for grayling, with all new flow depths and velocities being closest to the pre-impoundment depths and velocities for the low flow (Q95) scenario, and becoming lower than the pre-impoundment depths and velocities by the time median (Q50) and high (Q5) flows were reached (Figure 6.14). Contrary to the results produced using the brown trout suggested compensation release regime, baseflow depths and velocities at Q95 are all greater than those produced by the pre-impoundment low flow, with an average difference of 0.03 m between new suggested baseflow and pre-impoundment flow for grayling, and an average difference of 0.015 m when using the brown trout suggested compensation release regime. The key difference between the grayling compensation flow regime and the brown trout compensation flow regime lies in the magnitude of deviation from the current flow conditions. The elevated flow for grayling is lower than that for brown trout. Thus, both the elevated flow and the baseflow produce depths and velocities that differ little from those produced by the current compensation release regime, the difference becoming smaller as discharge increases. Indeed, at Q5, there is virtually no variation in depth and velocity produced by the new base and elevated flows, and those produced by the current compensation release regime (depth differences range from 0 to 0.02 m and velocity differences range only from 0 to 0.01 m s^{-1}).

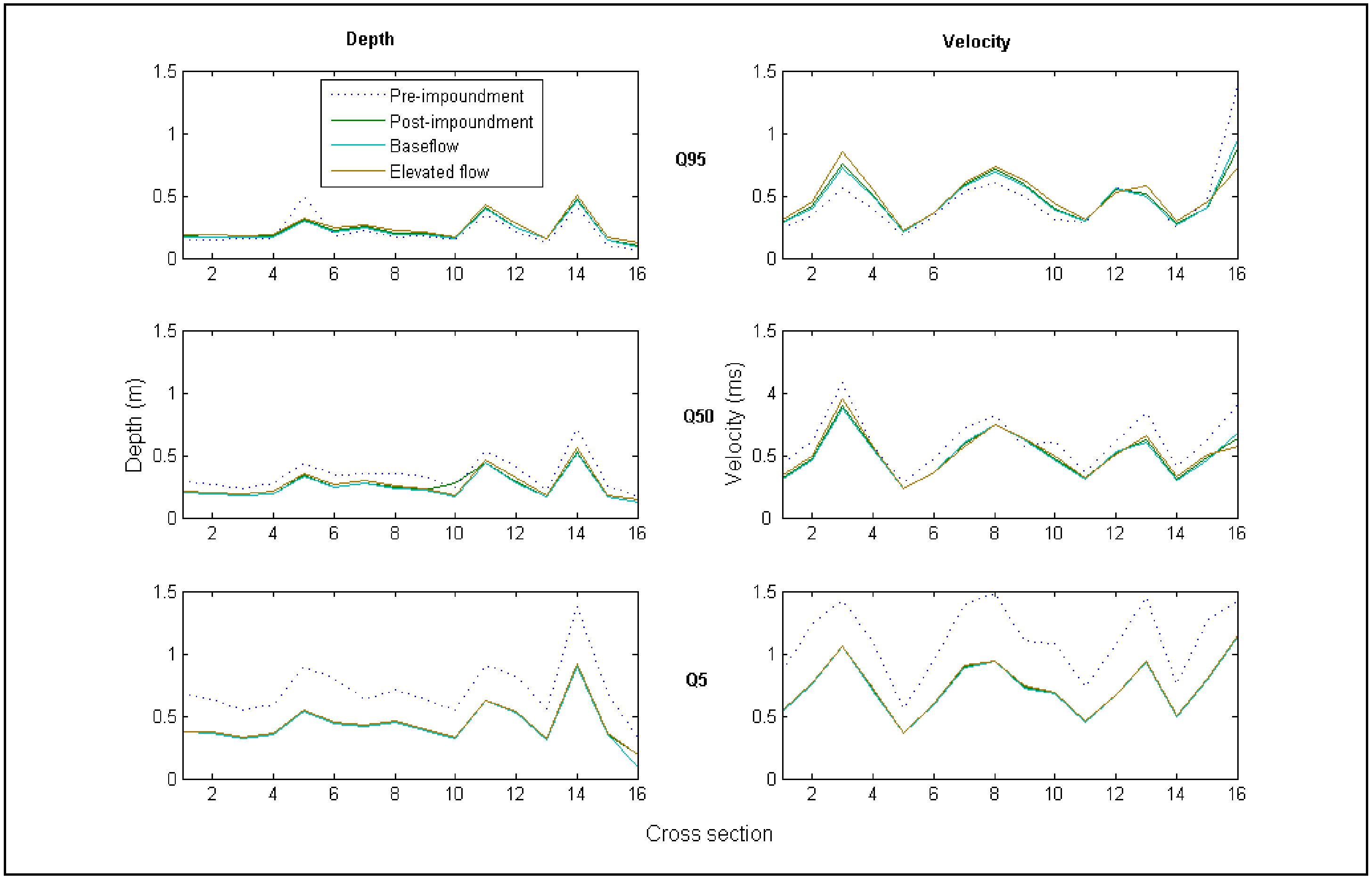

Figure 6.14 Depths and velocities produced at each site under low (Q95), median (Q50) and high (Q5) values, based on the new suggested regime designed for grayling requirements, with pre- and post-impoundment depth and velocity results included as context

Under Atlantic salmon flow requirements – 9:3 distribution

Because the baseflow/elevated flow distribution required to meet the depth and velocity demands of Atlantic salmon are very close to those of grayling (Table 6.10), the results from HEC-RAS were very similar to those produced when using the grayling suggested compensation release regime (Figure 6.15). At Q95, the depths and velocities produced using the Atlantic salmon suggested compensation release regime are greater than the depths and velocities produced pre-impoundment, whereas for median and high flows they are smaller than those produced under the pre-impoundment regime. Magnitude of deviation from depths and velocities produced under the current compensation release regime are of a similar scale to those found in the grayling study. Again, the difference between the depths and velocities produced for the Atlantic salmon suggested compensation release regime and those produced under the current regime at higher flows (Q5) are closer than at low flows (Q95), with virtually no difference in depth or velocity (maximum difference at any site is 0.01 m, 0.01 m s^{-1} and most sites have 0 m or 0 m s^{-1} difference).

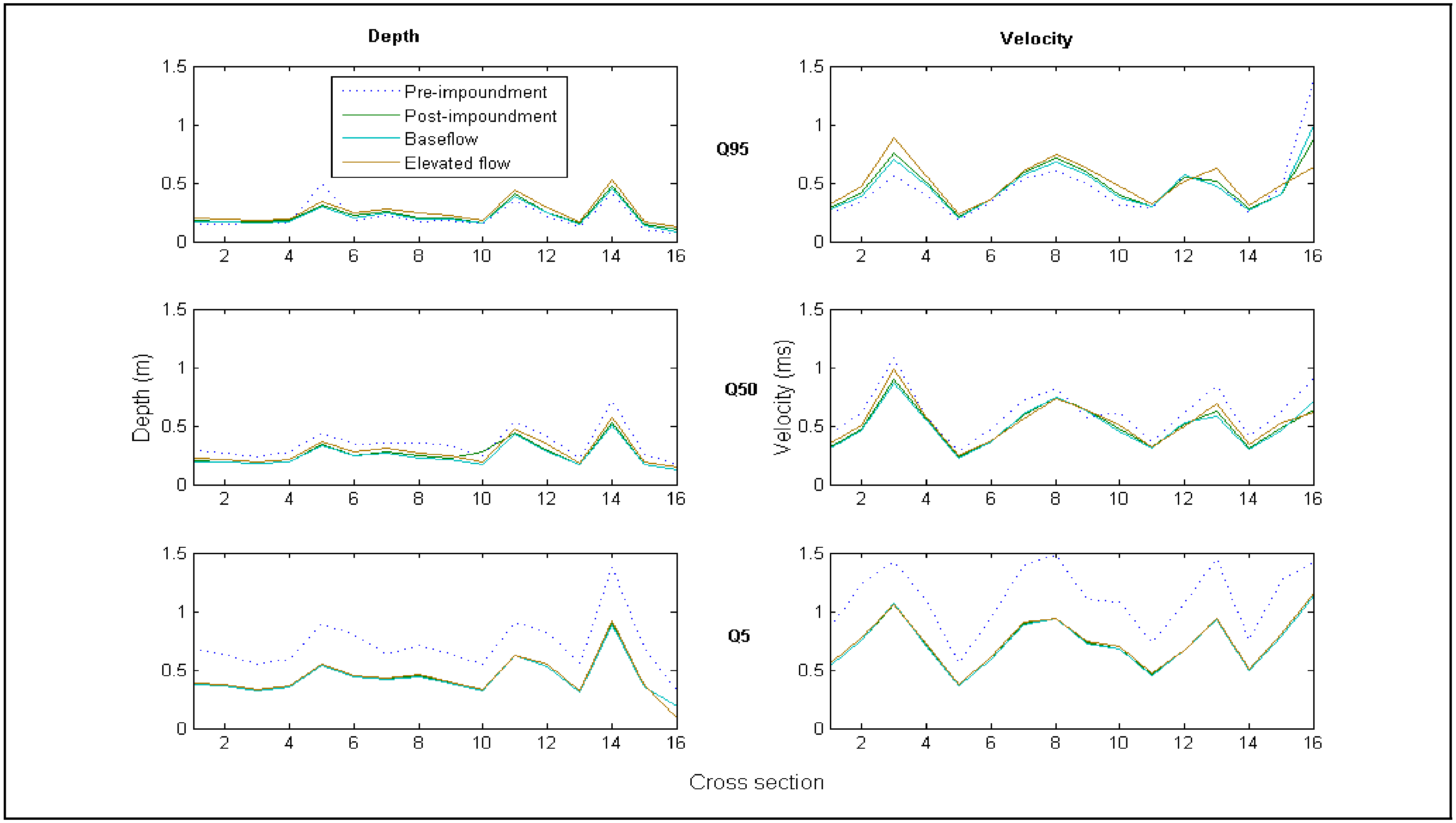

Figure 6.15 Depths and velocities produced at each site under low (Q95), median (Q50) and high (Q5) values, based on the new suggested regime designed for Atlantic salmon requirements, with pre- and post-impoundment depth and velocity results included as context

11:1 month distribution

When elevated compensation release was reduced to a duration of one month in each year, the amount of water available for release in the other 11 months of the year increased. In this assessment, the rate of elevated flow remained the same as originally suggested in Chapter Three and so predicted depths and velocities under the elevated flow remained as they were in the previous Section. However, as the compensation release rate for the rest of the year increased, the predicted depths and velocities increased (Figures 6.16 to 6.18). This approach is most effective when applied to the new suggested compensation release regime for brown trout as this is the one with the greatest elevated flow rate (and therefore there is a greater surplus to redistribute across the year). The effect is also most pronounced at lower flows. Using the brown trout suggested compensation release regime, when compared to the baseflow depths and velocities produced through the original 9:3 month distribution, the 11:1 distribution provides on average a 14.3% increase in depth and a 10.8 % increase in velocity, at Q95. This falls to a 2.6% increase in depth and a 1.7% increase in velocity at Q5.

When applied to the grayling-suggested new compensation release regime, the impact was much smaller and at best (Q95), depths were only raised by an average of 0.005 m (2.3%) and velocities by an average of 0.007 m s^{-1} (1.6%), across the 15 sites considered. At high flows (Q5), there was no change in depths from those found with the 9:3 month distribution, and there actually appeared to be a slight decrease in velocity (average of 0.007 m s^{-1}) at all sites. The same was found at low flows for the Atlantic salmon suggested new release regime with average increases in depth of 0.007 m (3.34%) and 0.014 m s^{-1} (3.31%) increase in velocity. At Q5 there was a negligible change in depth and velocity (an average increase of 0.1% in depth and 0.2% in velocity).

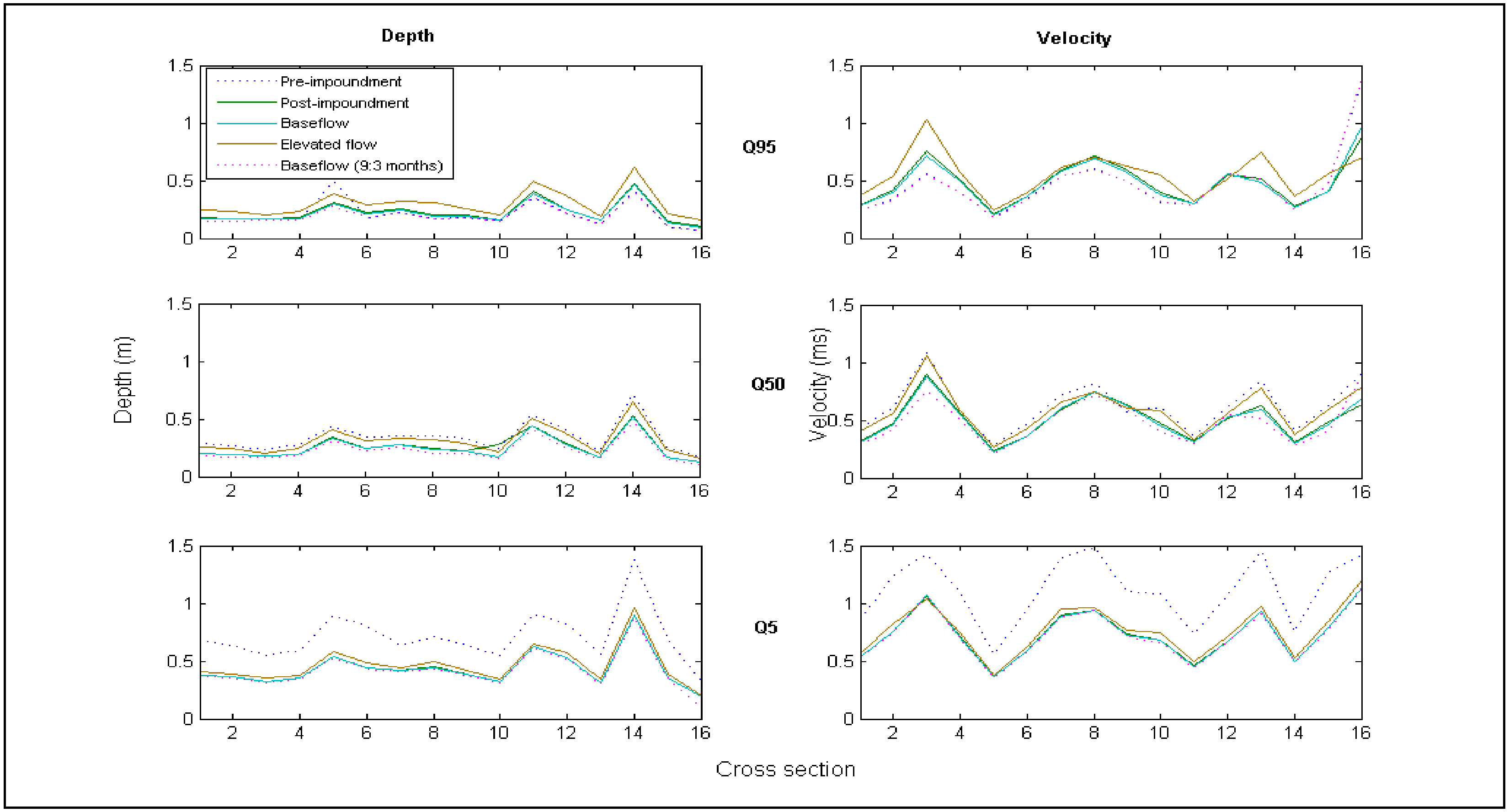

Figure 6.16 Depths and velocities produced at each site under low (Q95), median (Q50) and high (Q5) values, based on the new suggested 11:1 month regime designed for brown trout requirements, with pre- and post-impoundment depth and velocity results included as context

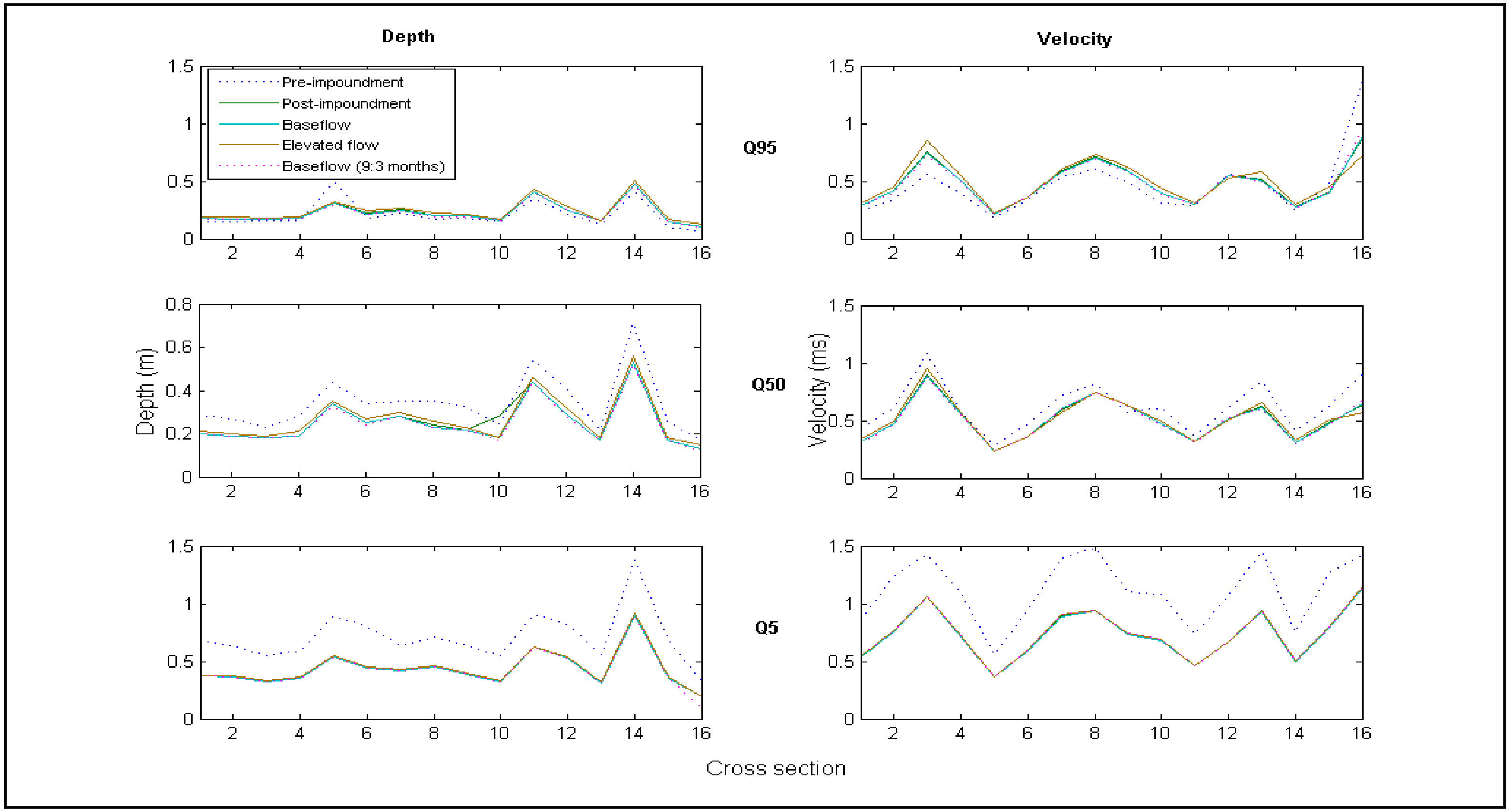

Figure 6.17 Depths and velocities produced at each site under low (Q95), median (Q50) and high (Q5) values, based on the new suggested 11:1 month regime designed for grayling requirements, with pre- and post-impoundment depth and velocity results included as context

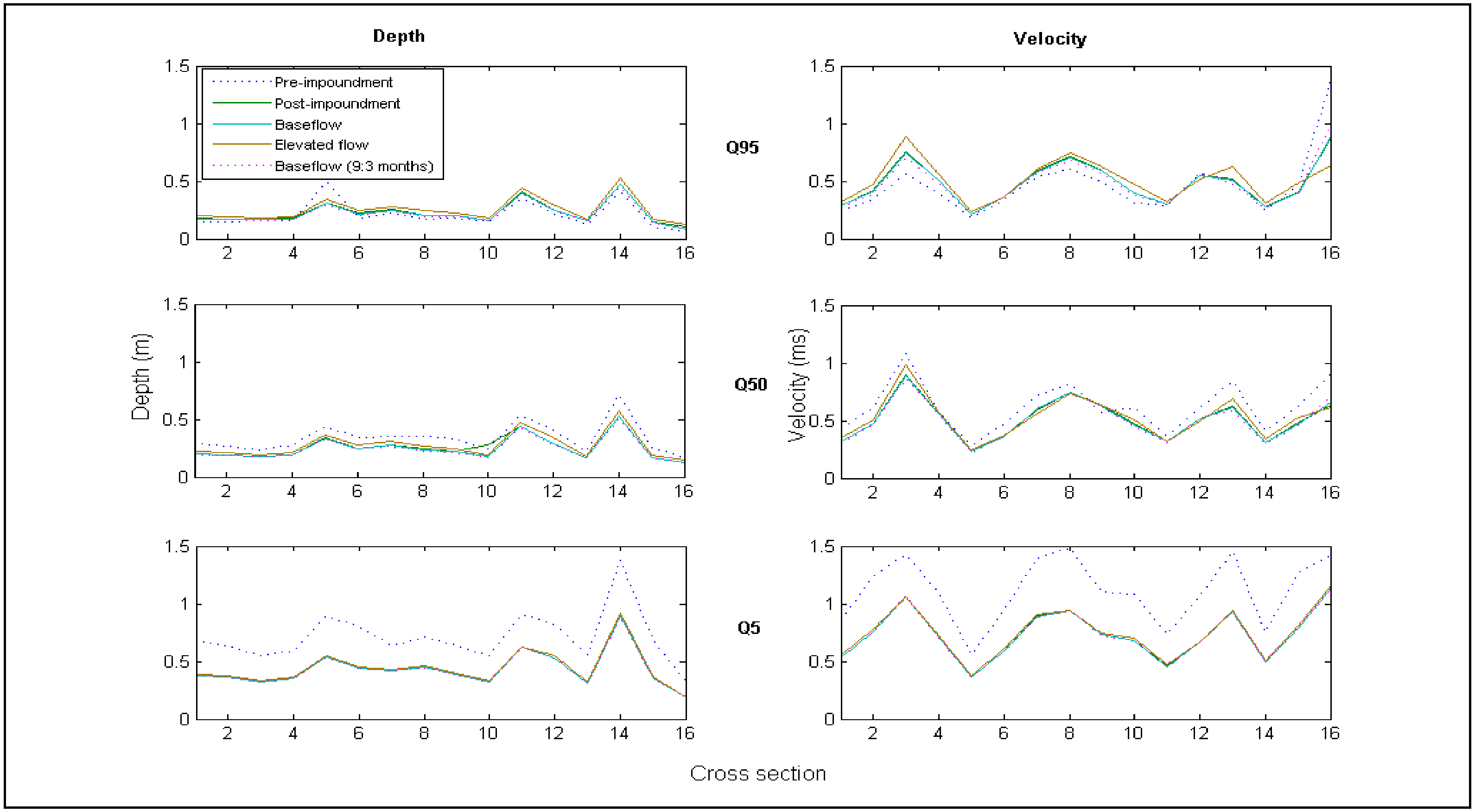

Figure 6.18 Depths and velocities produced at each site under low (Q95), median (Q50) and high (Q5) values, based on the new suggested 11:1 month regime designed for Atlantic salmon requirements, with pre- and post-impoundment depth and velocity results included as context

6.4.3 Unsteady simulations: annual hydrograph – 2002-2003

The results from the year-long hydrograph analysis show that the suggested new compensation release regimes for the different species would have produced a variation in the flow depths and velocities that were experienced at each cross-section, when compared to the flows experienced under the current compensation release regime. Figures 6.19 to 6.24 show the effect of the changes in compensation release regime on depth and Figures 6.25 to 6.30 show the effect on velocity.

Between September and November, when elevated flows would be required for brown trout and Atlantic salmon, the depths at each cross-section are greater than those produced under the current compensation release regime. However, under the grayling-suggested compensation release regime, between September and November, the depths produced are marginally lower than currently experienced. This is because the elevated flows for grayling are required in the spring months (February to April, see Section 4.3.2 for explanation). Therefore, a reduction in baseflow must compensate for this during the rest of the year. The difference between the flows predicted under the brown trout/Atlantic salmon regimes and those produced by the current regime and grayling suggested regime are apparent during the autumn months (brown trout/Atlantic salmon elevated flow period). However, during the spring (grayling period of elevated flow), the increases in depth caused by the grayling flow regime are less notable when compared to the depths produced by any of the other flow regimes (current, brown trout or Atlantic salmon). There is also variation within the elevated period of compensation flow for grayling – the difference between the depths produced by the grayling regime and those produced by all other regimes is greater in February and March than in April. This is a result of the current compensation release regime. In order to achieve the elevated flow required by grayling, there needs to be an increase in the current compensation release rates – between 19% and 32%, depending on time of year. As discussed in Chapter Three, the current compensation release regime varies slightly across the year and the period of elevated flow for grayling crosses the change in current release rates (currently, 25 Ml/d are released between April and September, 22.7 ml/d are released between October and March. The elevated period for grayling is from February to April). Therefore, there is a smaller difference between the February/March grayling compensation release rate and the current release rate than there is between the April grayling compensation release rate and the current April release rate. Compounding this, the elevated period for grayling is the lowest of the three elevated flow rates. The percentage increase for grayling is between 19% and 32%, whereas for Atlantic salmon it is between 35% and 45%, and for brown trout there is a 119 to 141 percentage increase on the current release rates.

What Figures 6.19 to 6.24 illustrate well is that there is a sharp and brief increase in depth at the beginning of September (under the brown trout and Atlantic salmon release regimes, in February for the grayling regime), when the one-day spate occurs, after which, depths remain elevated for a three-month period. The elevated (but not spate) flows are noticeably different to the depths experienced under the current compensation release regime, when using the brown trout/Atlantic salmon regimes, but less so when using the grayling regime, as discussed above. The choice of regime used does not affect the magnitude of the spate flow – this has been set at 700 Ml/d for all species. The baseflow depths, however, which are designed to reduce flow for the remainder of the year to compensate for the additional water released during the elevated flow, deviate much less from the conditions already experienced.

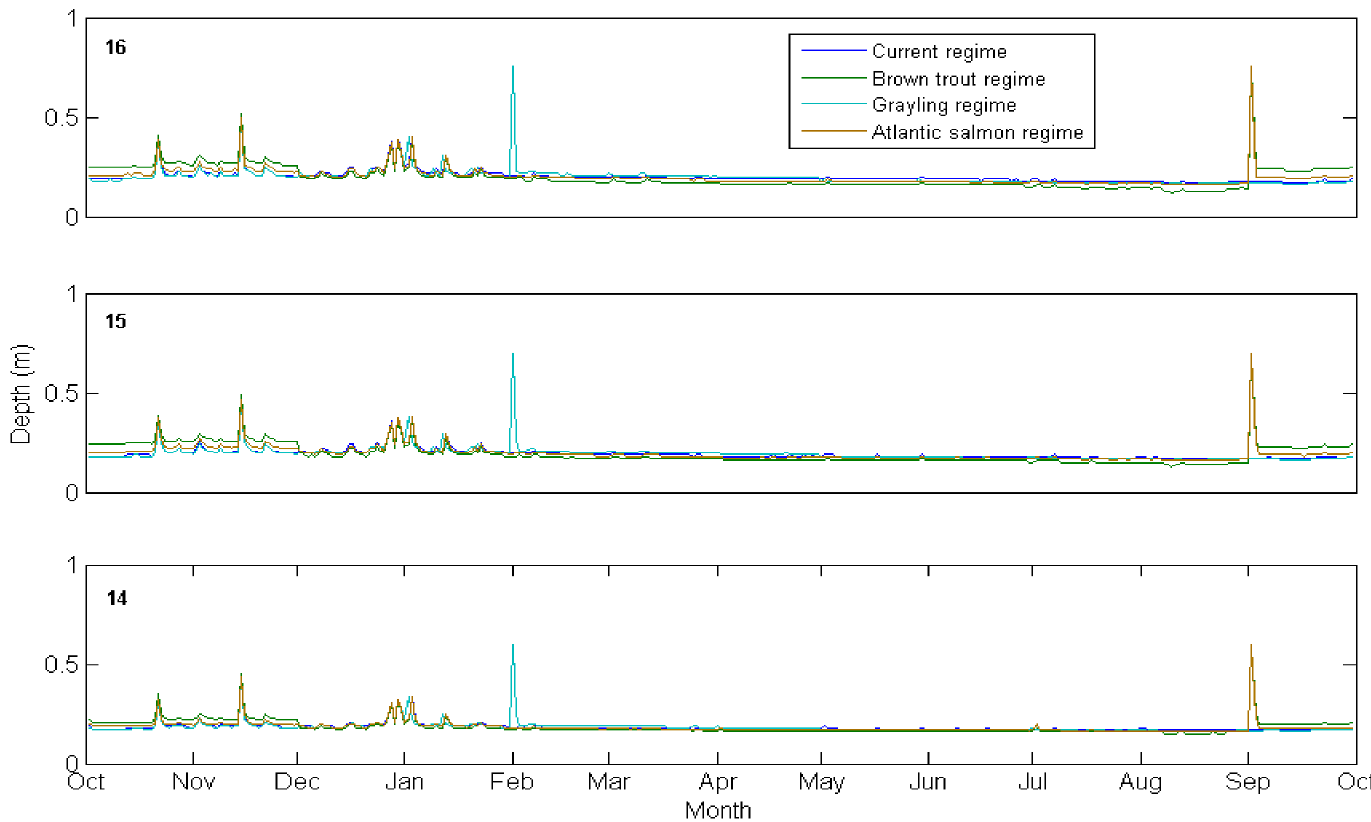

Figure 6.19 Depths produced during the water year 2002-2003 when the current, brown trout, grayling and Atlantic salmon regimes were applied to HEC-RAS, sites 16 (most upstream site: Eddy's Bridge) to 14

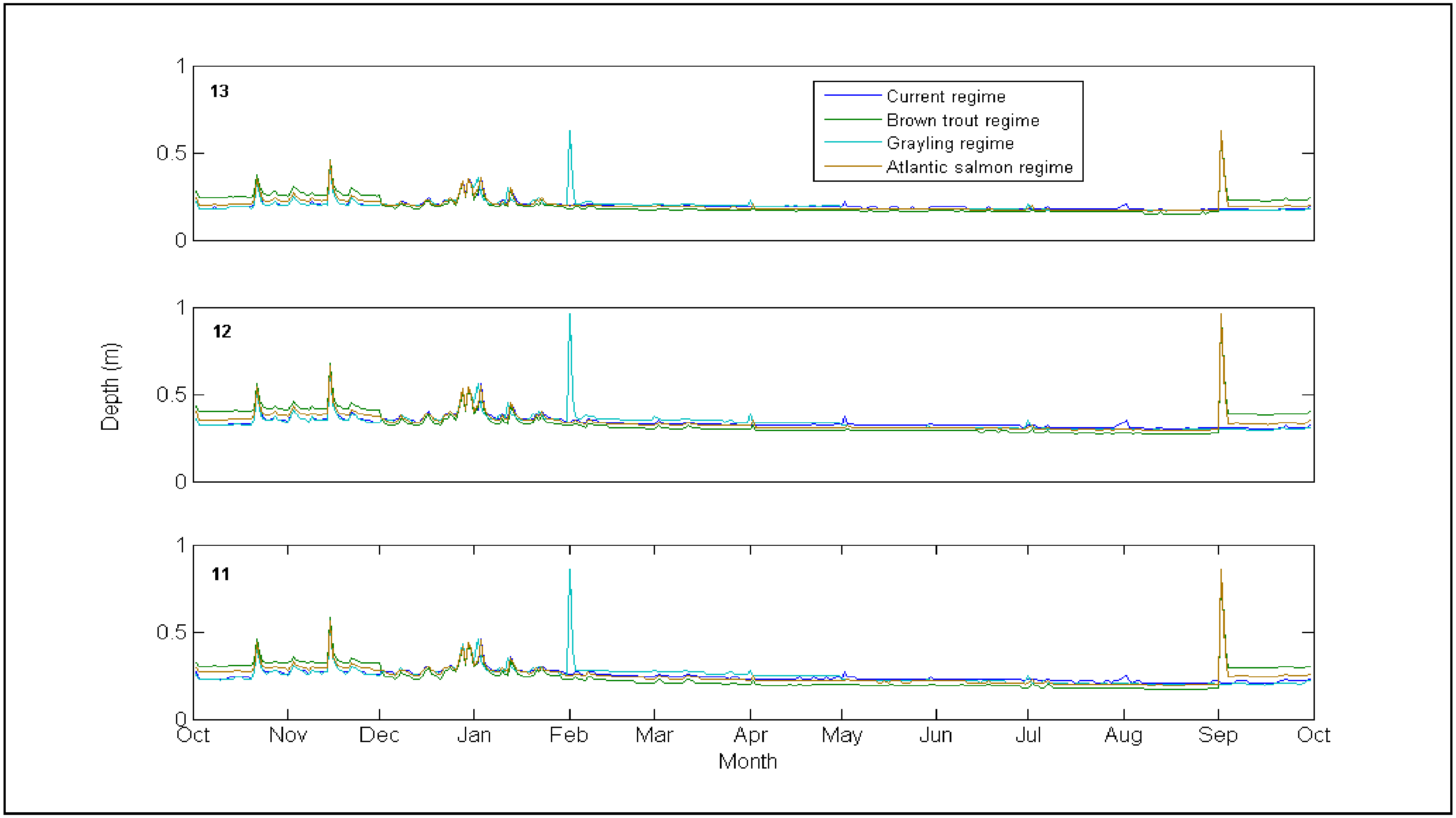

Figure 6.20 Depths produced during the water year 2002-2003 when the current, brown trout, grayling and Atlantic salmon regimes were applied to HEC-RAS, sites 13-11

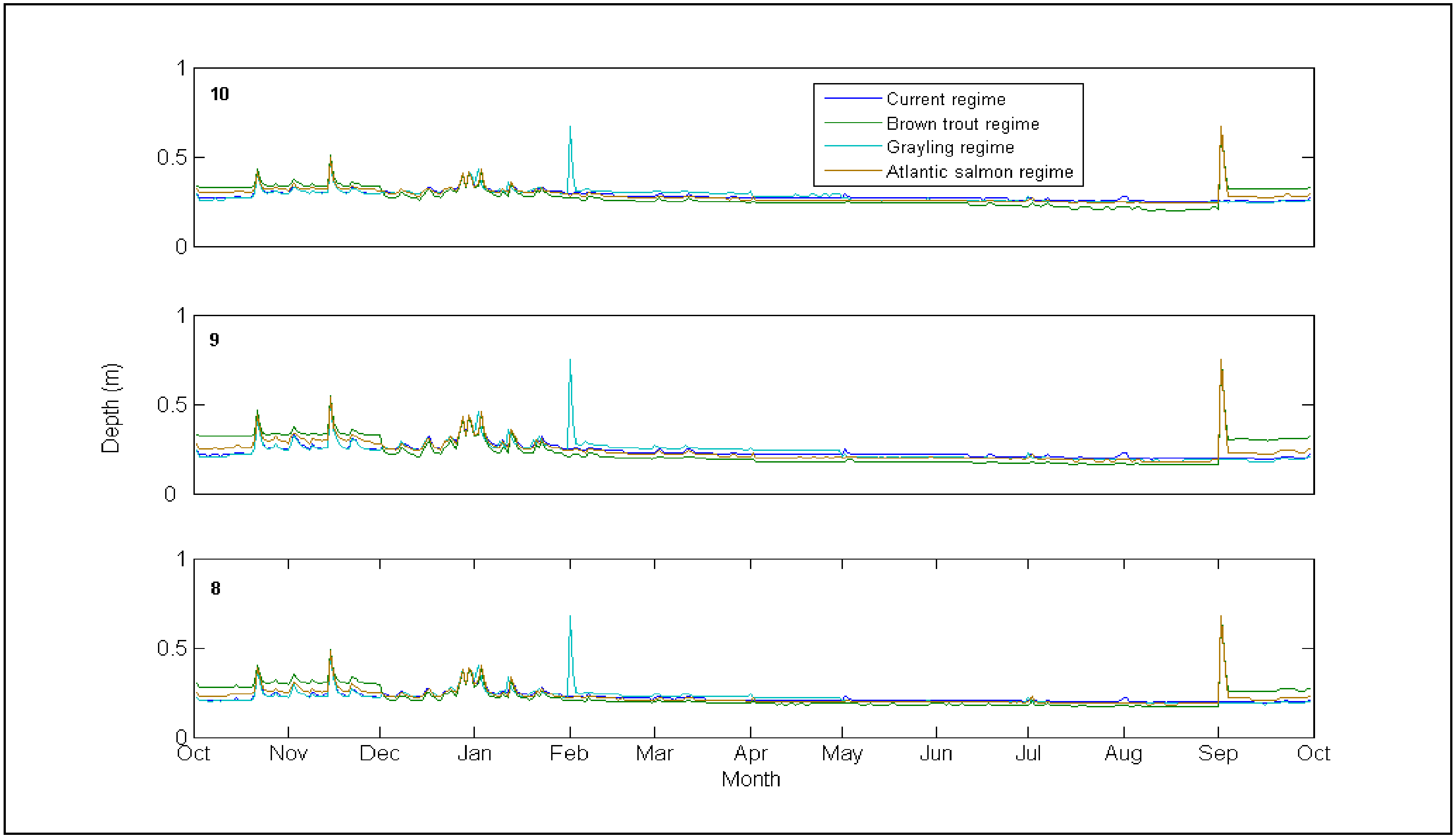

Figure 6.21 Depths produced during the water year 2002-2003 when the current, brown trout, grayling and Atlantic salmon regimes were applied to HEC-RAS, sites 10-8

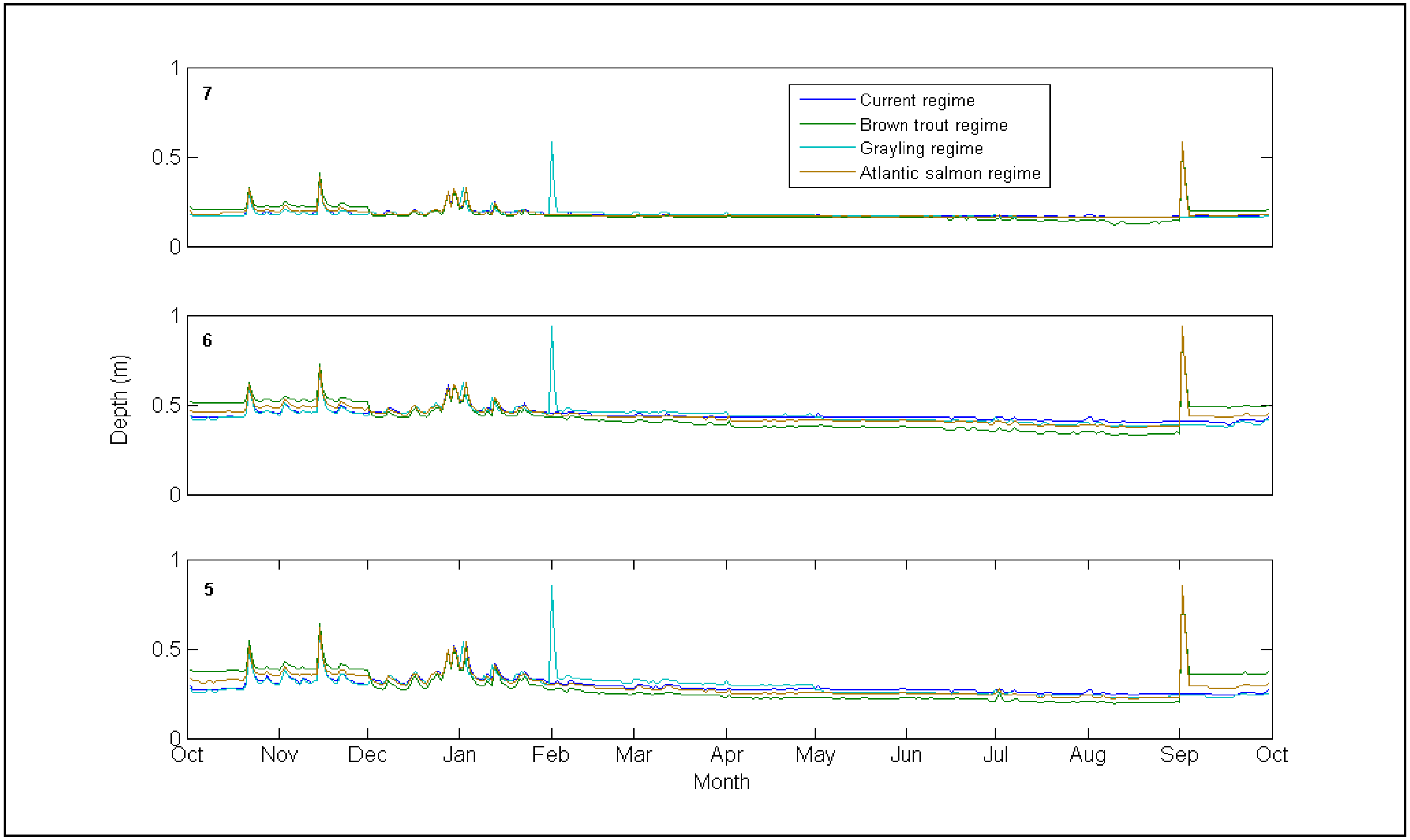

Figure 6.22 Depths produced during the water year 2002-2003 when the current, brown trout, grayling and Atlantic salmon regimes were applied to HEC-RAS, sites 7-5

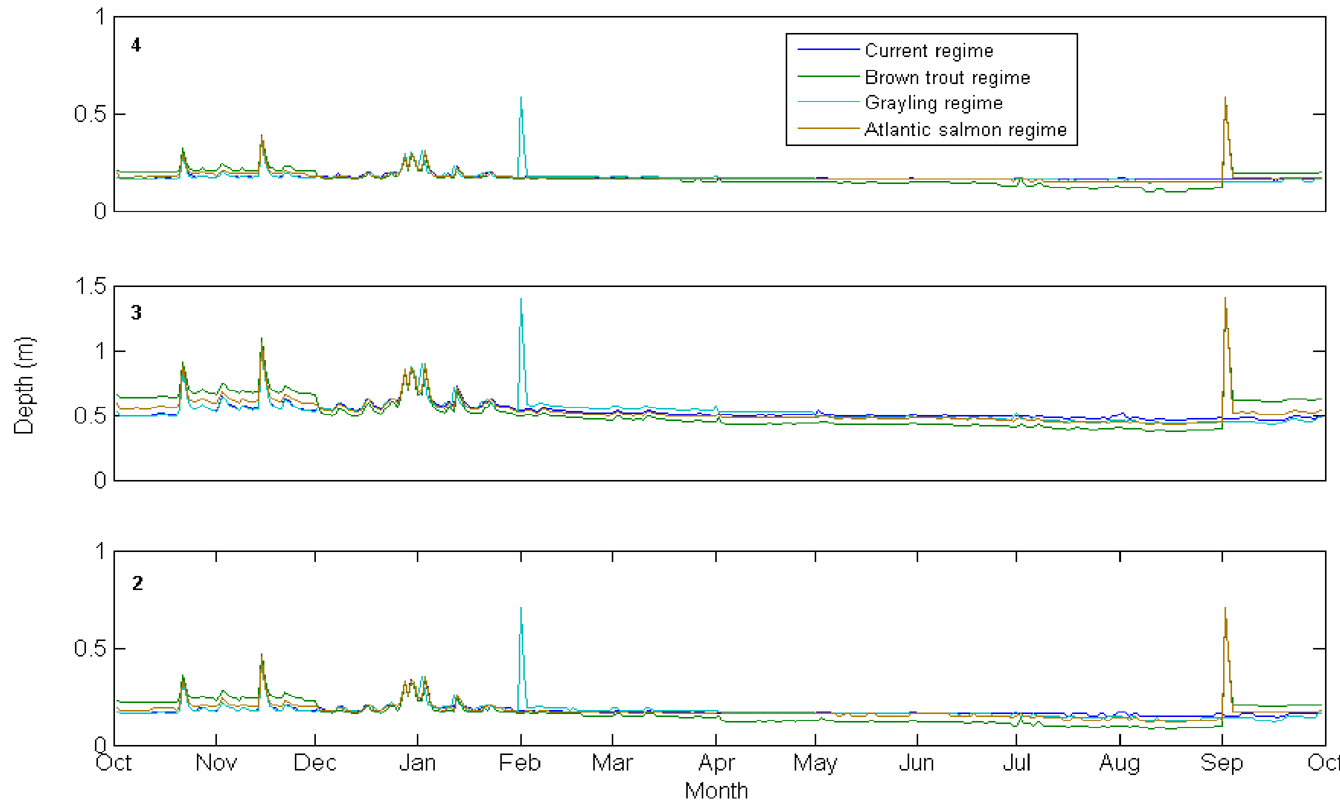

Figure 6.23 Depths produced during the water year 2002-2003 when the current, brown trout, grayling and Atlantic salmon regimes were applied to HEC-RAS, sites 4-2

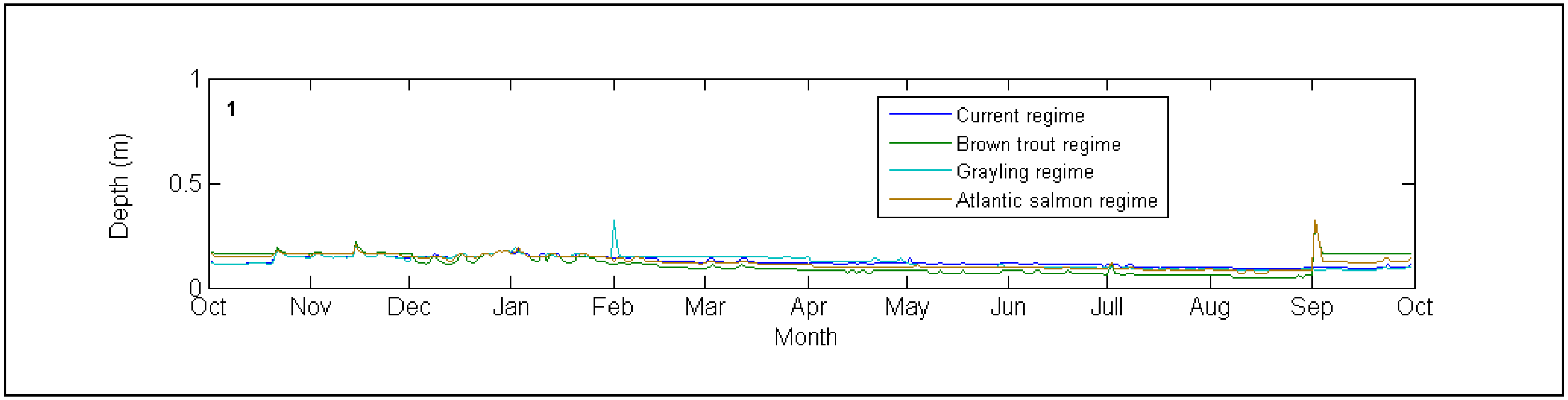

Figure 6.24 Depths produced during the water year 2002-2003 when the current, brown trout, grayling and Atlantic salmon regimes were applied to HEC-RAS, site 1

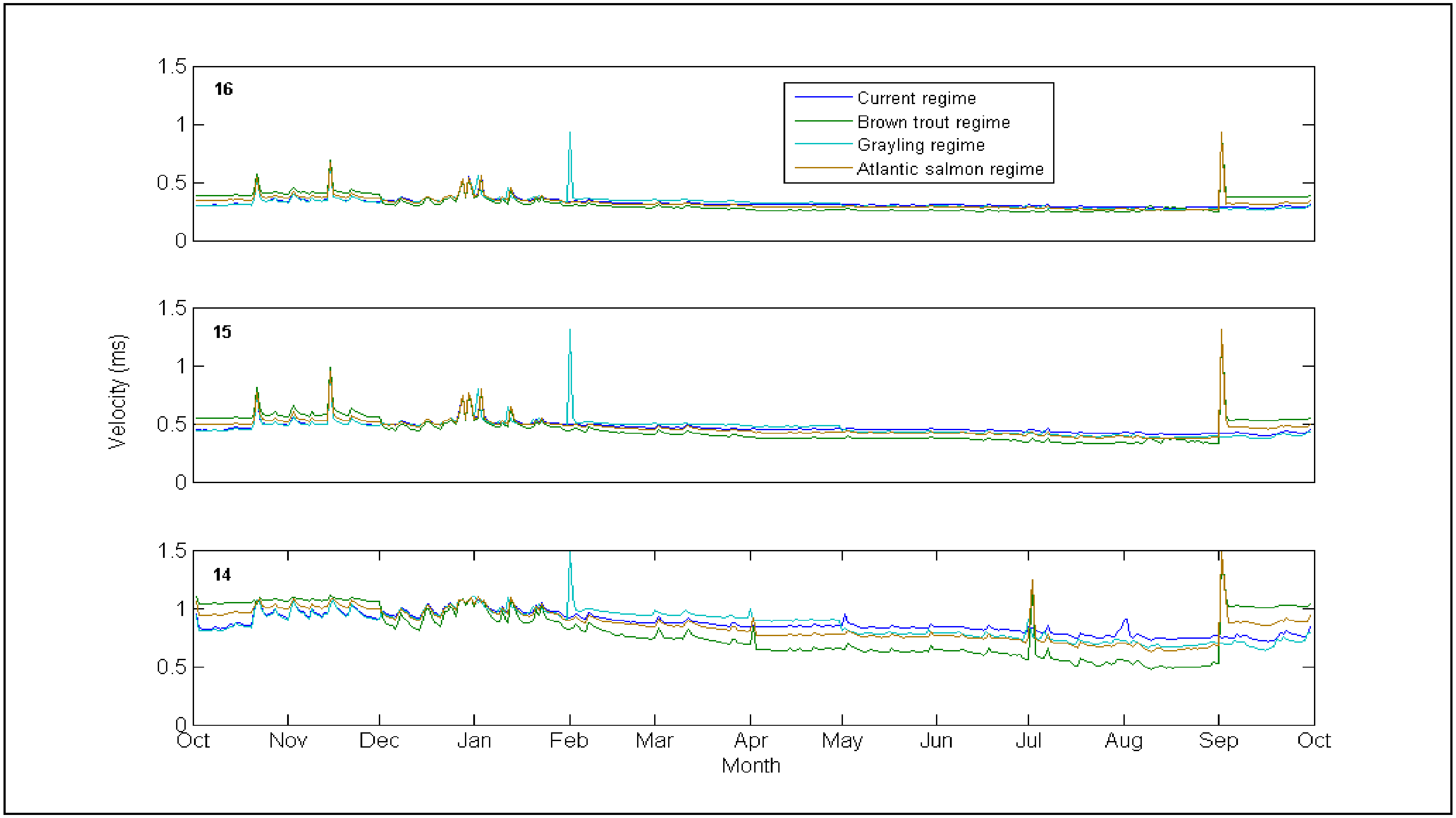

Figure 6.25 Velocities produced during the water year 2002-2003 when the current, brown trout, grayling and Atlantic salmon regimes were applied to HEC-RAS, sites 16 (most upstream site: Eddy's Bridge) to 14

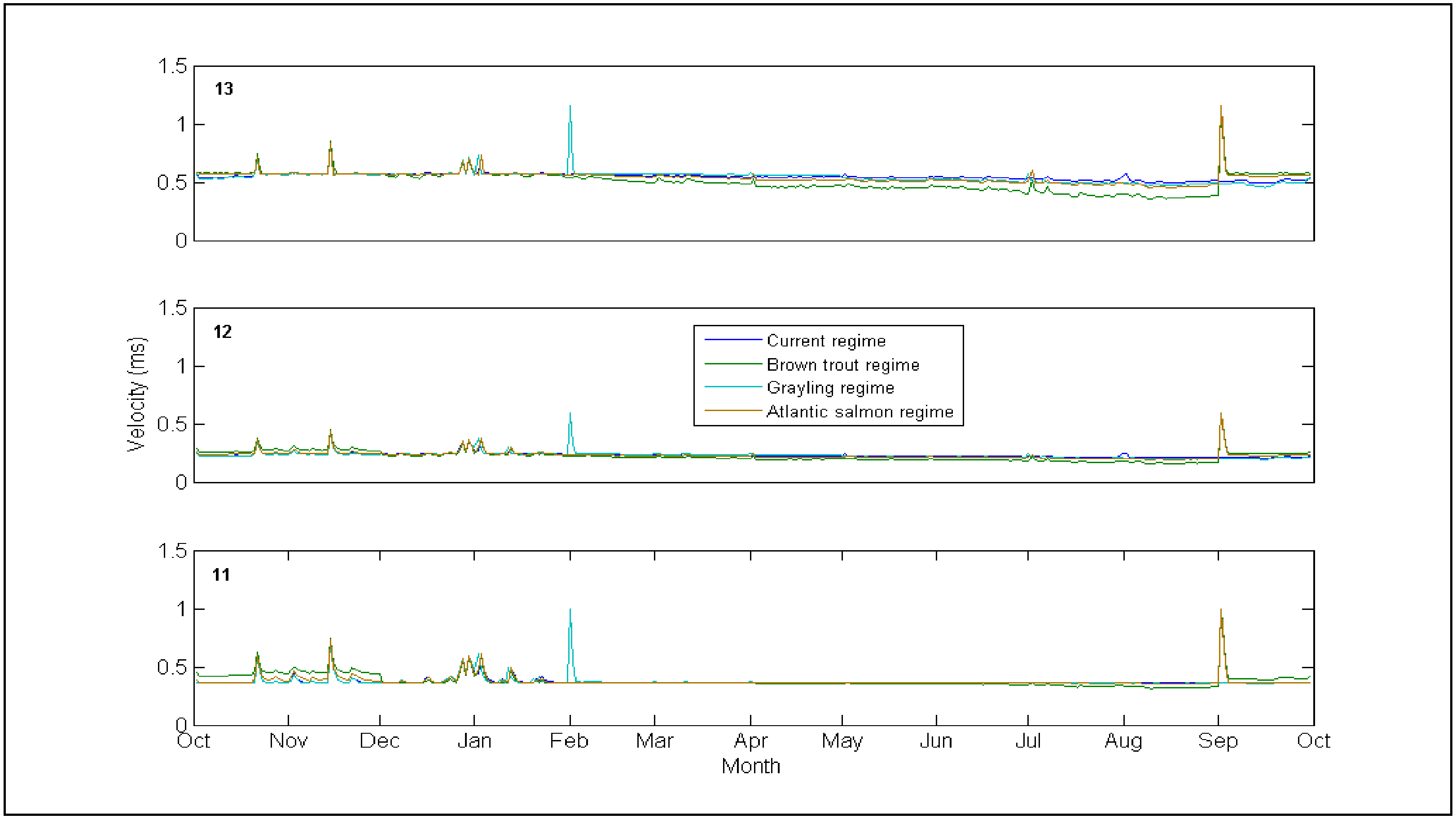

Figure 6.26 Velocities produced during the water year 2002-2003 when the current, brown trout, grayling and Atlantic salmon regimes were applied to HEC-RAS, sites 13-11

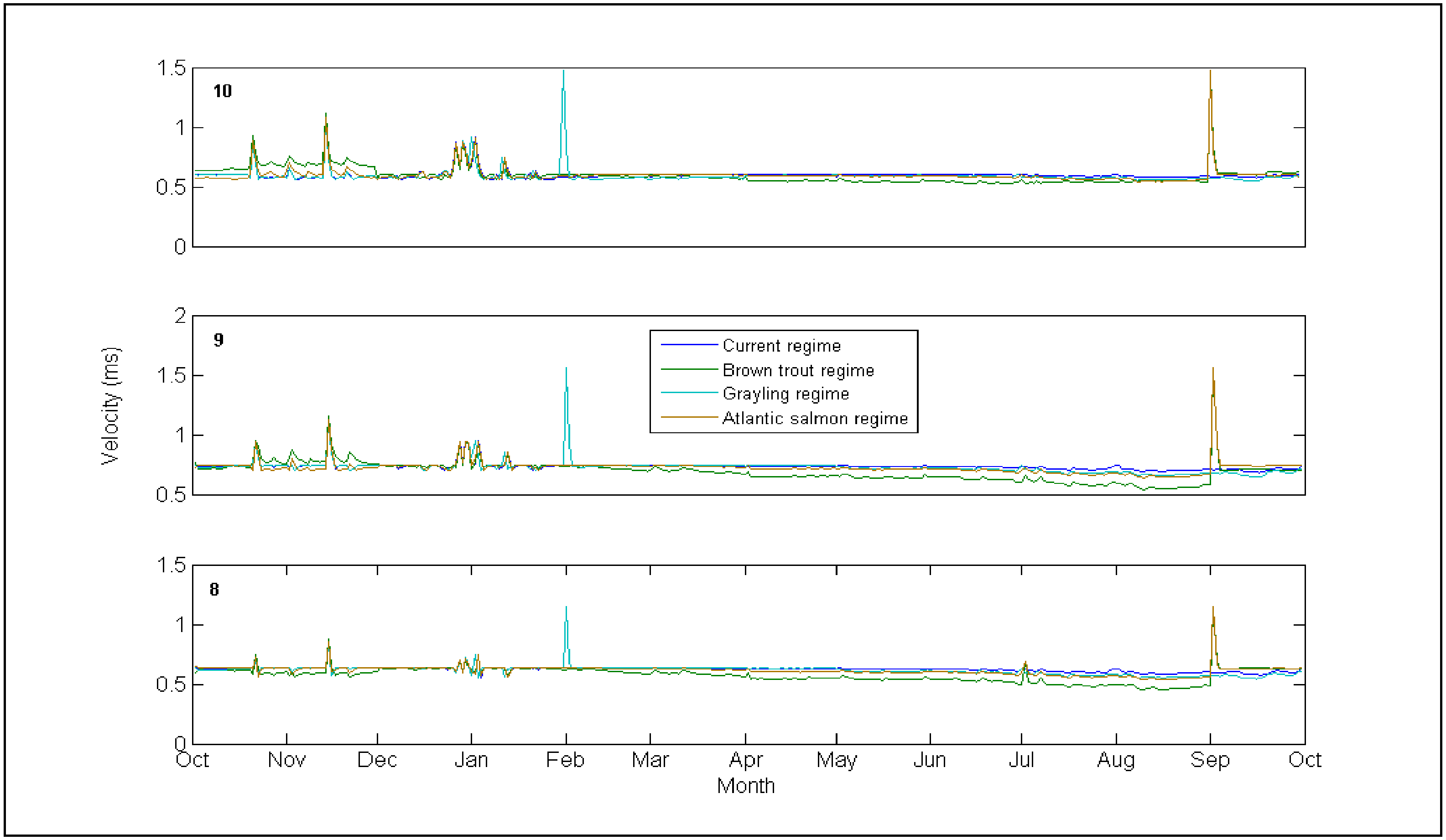

Figure 6.27 Velocities produced during the water year 2002-2003 when the current, brown trout, grayling and Atlantic salmon regimes were applied to HEC-RAS, sites 10-8

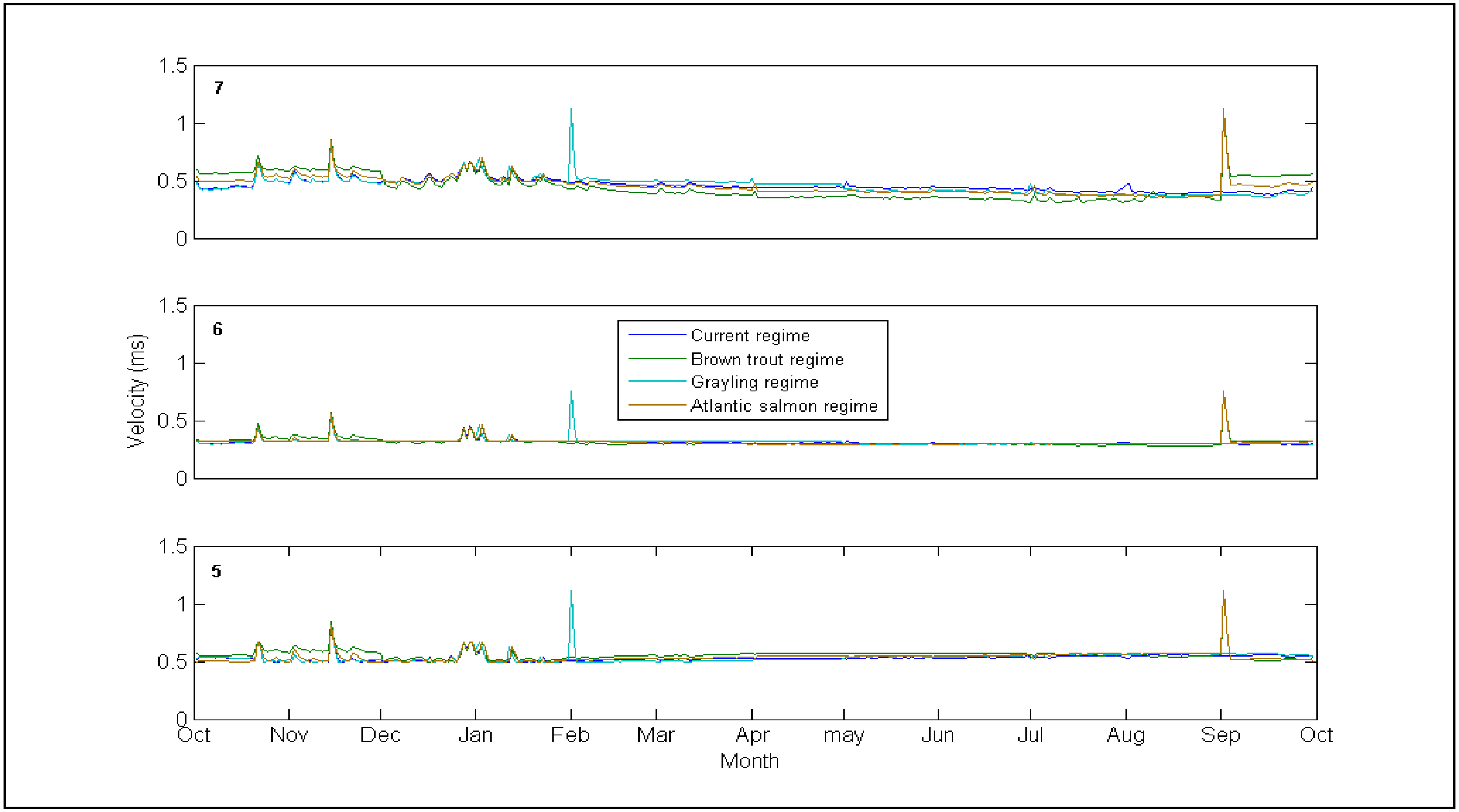

Figure 6.28 Velocities produced during the water year 2002-2003 when the current, brown trout, grayling and Atlantic salmon regimes were applied to HEC-RAS, sites 7-5

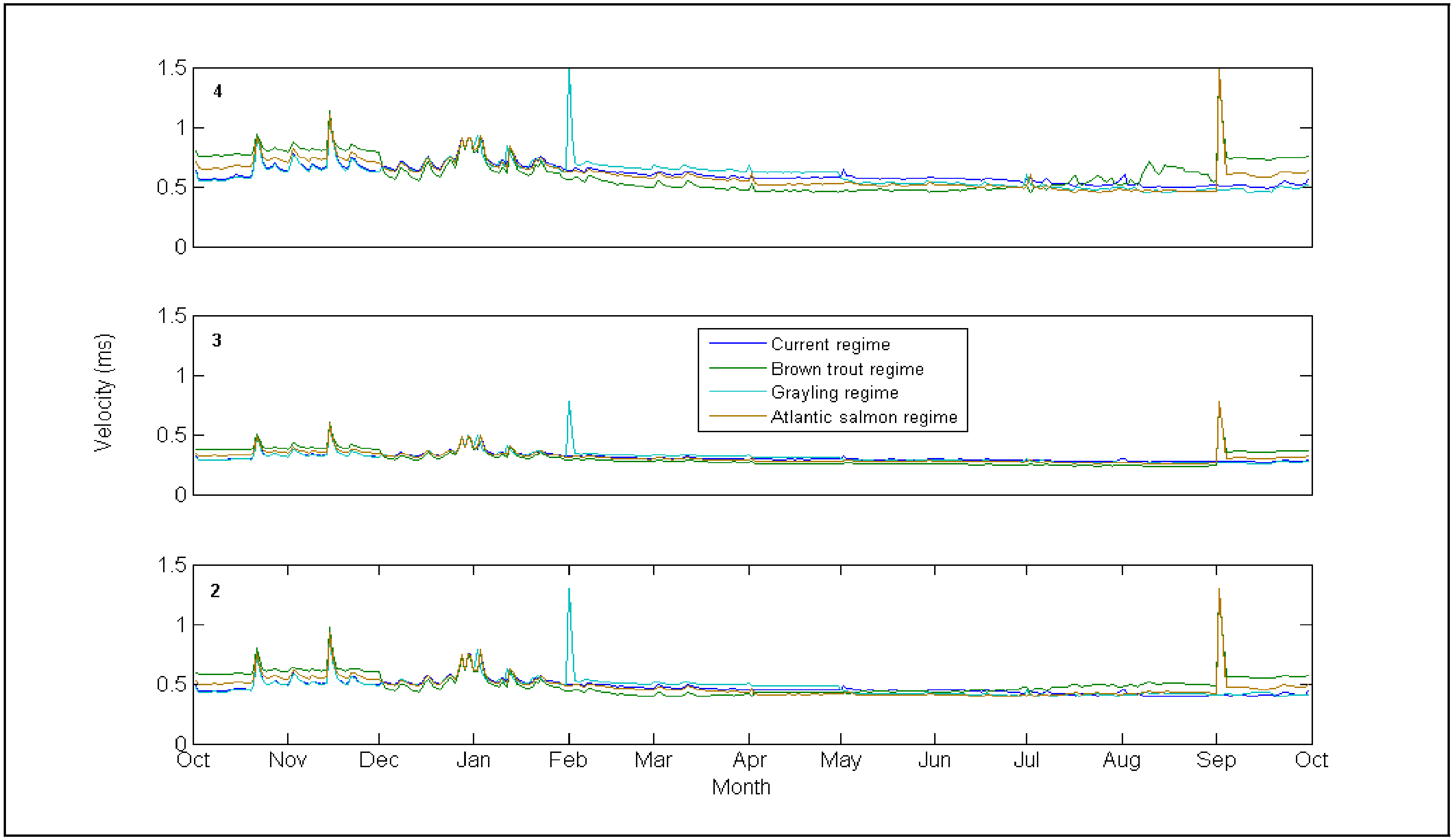

Figure 6.29 Velocities produced during the water year 2002-2003 when the current, brown trout, grayling and Atlantic salmon regimes were applied to HEC-RAS, sites 4-2

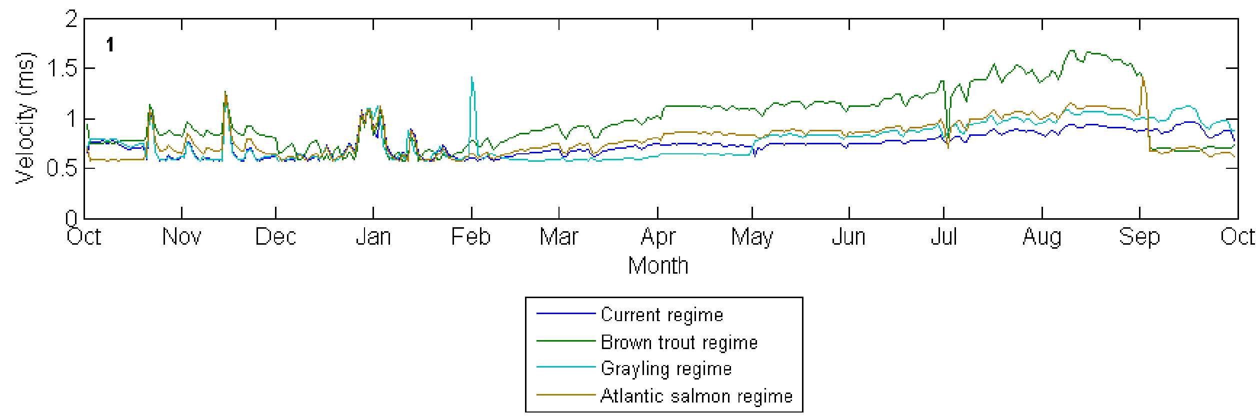

Figure 6.30 Velocities produced during the water year 2002-2003 when the current, brown trout, grayling and Atlantic salmon regimes were applied to HEC-RAS, site 1

Velocities were also generally increased during times of elevated flow and decreased during times of baseflow, when compared to the velocities produced under the current compensation release regime.

Table 6.9 shows the percentage difference in depth during baseflow and elevated flow periods for each flow regime, when compared to the current compensation release regime. It is evident that the brown trout impact has the most profound impact upon both baseflow depths and elevated flow depths, with up to 42% increase in depth during times of elevated flow and a decrease in depths by up to 28%. Grayling and Atlantic salmon have similar suggested new regimes and therefore produce changes in depth that are similar to each other, and both are lower than those produced under the brown trout regime.

Cross-section	Brown trout flow regime		Grayling flow regime		Atlantic salmon flow regime	
	Baseflow (December - August)	Elevated flow (September - November	Baseflow (February – March)	Elevated flow (April – January)	Baseflow (December – August)	Elevated flow (September – November)
1	-28.1	35.3	-6.6	15.8	-9.9	22.2
2	-21.1	36.3	-4	10.9	-5.4	15.9
3	-12	27	-3	8.1	-4.1	12
4	-11.8	23.7	-1.9	8.3	-2.3	10.8
5	-14.9	34.2	-3.6	11.8	-5.4	16.6
6	-10.8	17.8	-2.5	5	-3.6	8.1
7	-7.9	24.5	-1.4	8.3	-2.2	11
8	-10.7	33	-3	8.9	-3.8	13.3
9	-15.8	41.9	-3.9	12.6	-6	18.3
10	-11	20.5	-2.5	8	-3.5	10.1
11	-14.1	30	-3.3	11.8	-4.9	15.2
12	-8.8	23.7	-2.2	8.4	-2.9	10.9
13	-9.1	31.1	-2.1	7.6	-3.3	12.4
14	-6.4	21.1	-1.6	7.1	2.3	9.5
15	-10.9	32.5	-1.9	8.8	-3.8	13.5
16	-13.8	32.3	-3.3	8.9	-4.7	13.7

Table 6.9 Difference between depths produced by different flow regimes, expressed as a percentage of the depths produced by the current flow regime. A positive value denotes an increase in depth and a negative value a decrease

Changes in velocity are less pronounced, overall, than changes in depth, but are more sporadic. This is evident when considering the values in Table 6.10, percentage differences between velocities produced by new flow regimes and those produced under the current flow regime. However, the overall impact from each different regime remains the same, with brown trout causing the greatest changes and grayling the smallest. In some cases, the expected impact is reversed, for instance, at cross-section five, there is an overall increase in velocity during the baseflow period and an overall decrease during periods of elevated flow, according to the HEC-

RAS results. There is less variation in difference across the year, with velocity differences in the elevated periods being lower than those for depth.

Cross-section	Brown trout flow regime		Grayling flow regime		Atlantic salmon flow regime	
	Baseflow (December - August)	Elevated flow (September - November	Baseflow (February – March)	Elevated flow (April – January)	Baseflow (December – August)	Elevated flow (September – November)
1	41.1	13.1	8.0	-8.1	11.4	-4.2
2	-0.78	28.9	-0.88	9.8	-3.0	14.2
3	-10.7	25.1	-1.5	7.9	-3.5	11.1
4	-8.4	32.4	-4.2	10.2	-6.3	15.5
5	3.9	7.3	1.8	-0.92	2.1	-0.28
6	-2.4	9.9	0.20	5.3	-0.87	5.5
7	-14.5	27.3	-3.4	8.9	-5.5	13.2
8	-10.0	-0.19	-1.8	3.6	-2.8	3.4
9	-8.9	4.7	-1.4	1.9	-2.4	2.4
10	-4.8	13.6	-0.67	-0.74	-1.56	3.9
11	-3.6	20.0	0.29	3.0	-0.59	6.4
12	-10.9	19.9	-1.9	5.7	-3.4	8.2
13	-13.3	6.8	2.4	3.6	-3.8	5.4
14	-19.2	22.2	-4.3	7.7	-6.2	12.0
15	-13.7	22.9	-2.5	7.5	-4.7	11.3
16	-10.9	26.1	-1.9	7.6	-4.0	11.5

Table 6.10 Difference between velocities produced by different flow regimes, expressed as a percentage of the velocities produced by the current flow regime. A positive value denotes an increase in velocity and a negative value a decrease.

Like the depth results, the velocities produced by the brown trout and Atlantic salmon regimes diverge further from the velocities produced by current regime than those produced under the grayling regime. As discussed above, it is likely that this is due to the lower elevated flow required by grayling (a 19-32% increase compared to the brown trout increase of 119-141% and the Atlantic salmon required increase of 35-45%). As with depth, there is a step effect in the impact of the elevated flow which is brought about by the changing of the current compensation release regime during the elevated flow period. This happens for the compensation release regimes designed for all three species but is more noticeable for the grayling regime, perhaps because at lower flows changes are more pronounced, whereas at higher flows there is a drowning of subtle changes to flow, such as those brought about by the change in current compensation release regime across the year.

The spate flow, which occurs at the start of each elevated flow period, causes a short term but significant increase in velocity. This may be effective in flushing the fine sediment through the gravel – the aim of the spate flow.

As with depth, the greatest differences between velocities produced under new flow regimes and those produced under the current regime appear to occur at the times of greatest discharge (minus any compensation release). However, this is limited to times of elevated flow and so, as discussed above, it can be concluded that the greater differences are a result of the elevated compensation flows and are not affected by the magnitude of the discharge. Like depth, velocity is most significantly affected during times of natural high flow and least affected during times of natural low flow.

Spatial variation

There is spatial variation in the predicted depth and velocity outputs for each suggested flow regime. Some cross-sections experienced a greater difference between depths and between velocities under the new regimes, while other sites experienced very little difference. For example, at site 16, the most upstream location, there was a 14% decrease and a 32% increase in depth during base and elevated flows (respectively), whereas at the downstream site 14, there was only a 6% decrease and a 21% increase. Conversely, site 14 experienced greater differences in velocity than site 16 (site 14 had a difference ranging from -20 to +22% and differences at site 16 ranged from -11 to +26%).

Table 6.11 shows the range of difference found between depths and velocities when depths/velocities for each suggested new compensation release regime were compared to the depths and velocities from the current compensation release regime. It shows that for the brown trout regime, most sites were dominated by changes to depth, with the exception of sites 14, 12 and seven. Sites four and three experienced similar changes to depth as to velocity, but for site four, change to velocity was marginally greater while at site three, change to depth was greater. More variation occurred with the grayling and Atlantic salmon regimes. There was a more uniform distribution of impact on depth and velocity compared to the brown trout regime, with seven sites having ranges of depth and velocity changes within 5% of each other (these were sites

16, 15, 12, 6, 4 and 2). For both the grayling and the Atlantic salmon regimes, all of the sites that had similar difference ranges were slightly dominated by change in depth. Three sites were dominated by changes to velocity (sites 14, 7 and 4). These sites were dominated by changes to velocity under all three flow regimes and the sites dominated by depth changes were the same for all three new flow regimes. In other words, the flow regimes had the same directional effect, but the magnitude changed according to magnitude of change in flow, with higher flows tending to cause changes to depth rather than velocity (i.e. the elevated flows in the brown trout regime).

Cross-section	Brown trout regime		Grayling regime		Atlantic salmon regime	
	Depth	Velocity	Depth	Velocity	Depth	Velocity
1	63.4	-28	22.4	-16.1	32.1	-15.6
2	57.4	29.68	14.9	10.68	21.3	17.2
3	39	35.8	11.1	9.4	16.1	14.6
4	35.5	40.8	10.2	14.4	13.1	21.8
5	49.1	3.4	15.4	-2.72	22	-2.38
6	28.6	12.3	7.5	5.1	11.7	6.37
7	32.4	41.8	9.7	12.3	13.2	18.7
8	43.7	9.81	11.9	5.4	17.1	6.2
9	57.7	13.6	16.5	3.3	24.3	4.8
10	31.5	18.4	10.5	-0.07	13.6	5.46
11	44.1	23.6	15.1	2.71	20.1	6.99
12	32.5	30.8	10.6	7.6	13.8	11.6
13	40.2	20.1	9.7	1.2	15.7	9.2
14	27.5	41.4	8.7	12	7.2	18.2
15	43.4	36.6	10.7	10	17.3	16
16	46.1	37	12.2	9.5	18.4	15.5

Table 6.11 Range of differences (%) between depth changes and between velocity changes caused by each flow regime for each site. A blue coloured cell denotes a greater range of differences for depths at a site, green denotes a greater range for velocities and orange is used for sites where the range of difference for depth and velocity were similar (within 5%). Site 1 is thought to be an anomaly because of the weir and therefore was not considered in this part of the analysis

6.5 Discussion

There are a number of flow characteristics which have repeatedly been observed in each of the different scenarios applied to HEC-RAS. The reasons for the trends will be explored here and their implications for habitat suitability will be addressed in the next Section.

6.5.1 Pre and post-impoundment effects

The HEC-RAS results suggest that the new flow regime has caused an increase in low flows (Q95) and a decrease in high flows (Q5), with little change to the magnitude of flows that occur for a large proportion of time (Q50), in the River Derwent. In other words, there has been a dampening of extremes and a more uniform, steady flow in the river. This supports conclusions drawn in Chapter Three and reflects the conclusions drawn in a number of other regulation-related investigations (e.g. Petts and Pratts, 1983, Isik *et al*., 2008). The increase in baseflow occurs because the reservoir allows water to be provided to the channel even in times of little rainfall and therefore drought scenarios are reduced. Furthermore, in high rainfall or snow melt effects, the influx of water from upstream tributaries and as runoff is 'buffered' when the water is caught and impounded by the reservoir.

The results provided by HEC-RAS suggest that changes to depths and velocities between pre- and post-impoundment periods are low (as demonstrated in Section 6.4). However, it must be noted that HEC-RAS, as used in this study, is underestimating predicted flow discharges. Therefore, the differences, and indeed the depths and velocities, could be greater than predicted, by the most downstream section that is Rowlands Gill. It is likely that flows at the most upstream sites are being predicted with relative accuracy but that this diminishes with distance downstream.

6.5.2 Application of new regimes

The differences between the depths produced and the velocities produced by each of the three species' flow regimes can be explained by the different level of elevated flow required by each species. The brown trout regime produces the most extreme changes and the grayling regime produces the least significant changes to flow depths and velocities (when compared to the current compensation release regime and when compared to other species' regimes) because the grayling regime requires the lowest elevated flow, within the annual allowance of water for release, more can be distributed across the remaining nine months. The scale of elevated flow also impacted upon the efficiency of conserving water for the baseflow period in the 11:1 months scenario. When the elevated flow was applied for only one month of the year, the change to baseflow depths and velocities was most prominent for the brown trout regime. This is because the elevated flow for brown trout is considerably higher than that for grayling or Atlantic salmon. Therefore, when two months of the elevated flow were reduced, a greater surplus was available from the brown trout regime for redistribution across the other 11 months (when compared to the grayling or Atlantic salmon regimes).

During the year-long flow simulation, differences in depth and velocity caused by the brown trout and Atlantic salmon regimes were more profound than those caused by grayling – this can be attributed to the scale of the elevated flows for brown trout and Atlantic salmon.

At low flows (Q95), depths and velocities produced by the new suggested flow regimes for each species are close to those produced pre-impoundment. The flow, in comparison to the current regime, has been reduced in order to accommodate the elevated flows and the pre-impoundment Q95 flows/depths/velocities are lower than the post-impoundment ones. Therefore, the depths and velocities for the new species' regimes at higher flows (Q50 and Q5) are *lower* than those found pre-impoundment because although an elevated flow has been included, impoundment has caused a significant decrease in the higher flows (Chapter Three). This effect is compounded by the drowning out of the impact of a compensation flow with increasing total discharge.

Low flow (Q95) scenarios also provided the conditions during which the change from a 9:3 month distribution to an 11:1 month distribution was most effective. At higher flows (Q50, Q5), there was little improvement in the depths and velocities when the elevated flow only lasted for one month. This is because at higher flows (which, downstream of the reservoir are controlled by inputs from tributaries and runoff) the compensation flow contributes a smaller proportion of the total flow and does not increase or decrease as a function of external sources of discharge.

It could be expected that the difference between the depths produced by two regimes would decrease with distance downstream as other inputs contribute to the hydrograph and deem the compensation release a less-significant contributor. However, there does not appear to be any spatial pattern in the differences in depth/velocity over time (i.e. difference between elevated depth or velocity for elevated flow is no more prominent in any one reach of the river). This may be due to the fact that HEC-RAS does not accrue much more water as it moves downstream, because, as can be seen in Figure 6.13, an increase in flow rate causes the effect of elevated and lowered flows to be drowned out: they are only effective at low and median flows, and most effective at low flows (Q95).

Spatial variation

There is clearly a difference in impact of each flow regime with respect to different sites along the river Derwent. The way in which water is conveyed through a reach or cross-section depends upon the channel form. For example, a wide channel would lend itself to a shallow, rapid flow, whilst a narrow channel would cause a deeper, slower flow. Roughness is also an important factor in conveyance and the greater the roughness value, the lower the velocity (and therefore the greater the depth, when there is conservation of mass) (Knighton, 1998, p 100). In order to explain the variation in impact of regimes and the different depth/velocity distributions for each cross-section, the geometry was considered. Table 6.12 shows the relationship between the range of depth/velocity differences (as outlined in Table 6.11 above) and a number of channel parameters at each cross-section. Although few of the correlations are significant at the 95% confidence level, the results indicate that both the area available as wetted perimeter and the roughness of the channel may have some influence on the range of differences between depths under various regimes, but little impact on velocity. It could be assumed, then, that the variations in velocity are a function of depth. The lower range of differences between velocities than between depths may also explain the weaker correlations between velocity and wetted perimeter/roughness.

Correlation parameters	Brown trout flow regime		Grayling flow regime		Atlantic salmon flow regime	
	Depth	Velocity	Depth	Velocity	Depth	Velocity
Maximum wetted perimeter	-0.5 (0.07)	-0.13 (0.65)	-0.49 (0.07)	-0.05 (0.85)	-0.4 (0.14)	-0.13 (0.64)
Manning's n	-0.32 (0.25)	-0.31 (0.25)	-0.17 (0.56)	-0.55 (0.03)	-0.13 (0.65)	-0.53 (0.04)
Channel width (m)	-0.24 (0.38)	0.06 (0.82)	-0.3 (0.28)	0.07 (0.82)	-0.22 (0.46)	0.03 (0.92)
Hydraulic radius (m)	0.14 (0.63)	0.20 (0.47)	0.18 (0.53)	0.20 (0.47)	0.20 (0.49)	0.15 (0.60)
Flow area (m^2)	-0.34 (0.22)	-0.04 (0.88)	-0.24 (0.39)	-0.1 (0.74)	-0.17 (0.55)	-0.15 (0.59)

Table 6.12 Correlation results between range of differences (%) for depth and velocity for three flow regimes, and channel parameters. Significance values (p) are shown in brackets

There appears to be little or no pattern between channel width, hydraulic radius or flow area and the range of variability experienced for any of the flow regimes. In other words, whether a cross-section has experienced a greater change to depth, or a greater change to velocity, is not affected by these parameters. As Table 6.13 shows, there may be a relationship between the slope of the energy grade line and the degree of change that occurs (although slope does not appear to favour one variable over another). All but one of the sites which have small differences between the amount of change to depth and the amount of change to velocity, have the lowest gradients.

Cross-section	Slope of the energy grade line (m/m)	Range of differences found from brown trout regime (%)		Range of differences found from grayling regime (%)		Range of differences found from Atlantic salmon regime (%)	
		Depth	Velocity	Depth	Velocity	Depth	Velocity
9	0.013021	57.7	13.6	16.5	3.3	24.3	4.8
10	0.012834	31.5	18.4	10.5	-0.07	13.6	5.46
14	0.012213	27.5	41.4	8.7	12	7.2	18.2
7	0.007889	32.4	41.8	9.7	12.3	13.2	18.7
13	0.007333	40.2	20.1	9.7	1.2	15.7	9.2
8	0.006839	43.7	9.81	11.9	5.4	17.1	6.2
15	0.006764	43.4	36.6	10.7	10	17.3	16
4	0.006733	35.5	40.8	10.2	14.4	13.1	21.8
5	0.004443	49.1	3.4	15.4	-2.72	22	-2.38
11	0.003905	44.1	23.6	15.1	2.71	20.1	6.99
2	0.003824	57.4	29.68	14.9	10.68	21.3	17.2
6	0.002323	28.6	12.3	7.5	5.1	11.7	6.37
16	0.001874	46.1	37	12.2	9.5	18.4	15.5
12	0.000859	32.5	30.8	10.6	7.6	13.8	11.6
3	0.000825	39	35.8	11.1	9.4	16.1	14.6
1	0.002327	63.4	-28	22.4	-16.1	32.1	-15.6

Table 6.13 Relationship between variable most affected and slope of energy gradient line. Blue cells denote most change occurred to depths, green cells denote most change occurred to velocity, orange cells are used when the amount of change between depth and velocity is similar (within 5%)

Although some sites experience a greater impact on velocity than on depth, the majority are predominantly affected by depth. When the new flow regime is at its most extreme (i.e. the brown trout regime), the effect on depth becomes greater than the effect on velocity. This suggests that velocity is affected by factors such as roughness but when discharge is increased, the effect of sensitive controls such as roughness are 'drowned out' and change to depth dominates.

The depths and velocities produced by HEC-RAS, even at the input boundary, are different to those entered into CRAB. Because CRAB uses depth and velocity inputs to determine the rate of compensation release required, it could be expected that when that output was entered into HEC-RAS under a steady flow, the depths and velocities would be recreated (at the input boundary). The reason that this does not occur is because HEC-RAS is a more sophisticated tool that allows for channel geometry within its calculations, a feature that CRAB lacks. Because CRAB is a simple, easily implemented design tool, there is allowance for the input of only one channel width. When studying a long stretch of river (in this case, 27 km), one width measurement cannot adequately represent the width of that river, not to mention the changing morphology of the area between the banks. The width of 11.55 m was used for the CRAB modelling. In reality, the widths for the cross-sections of the River Derwent used in HEC-RAS (comparatively, still a small sample of the river), range from 8.0 m to 23.26 m – a difference of 15.26 m. Furthermore, the extent to which the wetted perimeter could vary depends on the channel form, which is influenced by bed material and shape. The wetted perimeters of the cross-sections vary from 8.1 m to 18.3 m, and roughness coefficients, which affect the conveyance of water through a reach, range from 0.025 to 0.04. Consequently, although simple itself, HEC-RAS provides a much more detailed examination of flow through a changing channel and therefore the depths and velocities from HEC-RAS should be trusted over the results from CRAB: i.e. if the HEC-RAS depth and velocity outputs do not match the CRAB inputs, they are not necessarily inaccurate.

6.5.3 Implications for ecology

The implications of the HEC-RAS results for the ecology of the River Derwent will be considered in detail as part of Chapter Seven. However, the broader impacts will be examined briefly here. Table 6.14 reiterates the minimum depth and velocity requirements of each fish species, as detailed in Chapter Five (*minimum* values were used to allow for additional inputs to the system and to reduce the amount of water required for release. See Section 5.3.4 for details).

	Brown trout		Grayling		Atlantic salmon	
	Depth	Velocity	Depth	Velocity	Depth	Velocity
Spawning ('elevated flow')	0.26	0.21	0.1	0.3	0.17	0.2
Juvenile ('baseflow')	0.31	0.04	0.2	0.25	0.21	0.2

Table 6.14 Depth and velocity requirements for brown trout, grayling and Atlantic salmon, as used in the CRAB model to design new flow regimes. It should be noted that as spawning is the period of interest, if all of the depth/velocity requirements cannot be accommodated for the whole year, the spawning requirements should be prioritised – this may mean a reduction in flows for the rest of the year to levels much lower than those stated in this table

Pre- and post-impoundment

At all three flow magnitudes investigated (Q95, Q50 and Q5), when based on the steady simulation of flow though the River Derwent using HEC-RAS, the depths for all species have improved, with low flows being slightly elevated and high flows being somewhat reduced. At low flows, velocity appears to have increased post-impoundment and this is not necessarily in line with the velocity requirements stated above (which range from 0.04 to 0.3 m s^{-1}) but the change is slight, with an increase of only 0.05 m s^{-1} at most sites (not including cross-section 1). At higher flows, post-impoundment velocity has been reduced from velocities which were higher than preferred by the different species, to velocities which are more suitable for all species, but still higher than the requirements stated in Table 6.14.

Steady low, median and high flows

The new suggested flow regimes (9:3 month distribution) have shown that the elevated flows offer a period of elevated depth and velocity at each cross-section, with the brown trout regime providing the greatest depths and velocities (an increase of up to 55% for depth and 42% for velocity). In order to compensate for the elevated flows, the compensation release rate for the rest of the year must decrease and, according to the HEC-RAS outputs, this appears to have minimal effect on the depths and velocities, when compared to the current compensation release regime. Therefore, at this stage, it may be suggested that the elevated flows can be achieved with minimal disruption to flows for the rest of the year. For the grayling regime, even at low flows, the impact of an elevated flow was smaller, with increases in depth not exceeding 14% and in velocity, not exceeding 12% of the velocities produced under the current regime. Despite this, the depths and velocities produced under the grayling regime were still all greater than those required by any of the three species at elevated flows at every site (depths ranged from 0.12 to 0.51 m, spawning requirements range from 0.1 to 0.26 m).

Changing to an 11:1 month regime, in which elevated compensation flows remain at the same rate but for a shorter duration, appeared to be of little benefit when applied to the grayling and Atlantic salmon regimes as the elevated flows used in these regimes are lower than those used in the brown trout regime and therefore there is less water as surplus to distribute across the rest of the year. However, when applied to the brown trout regime, rest-of-year depths and velocities were increased by 14.3% and 10.8%, respectively (based on average for all sites, at low flow). This increases the baseflow depths to an average (across all sites) of 0.21 m – the depth requirements of the three fish species range from 0.2 to 0.31 m. Meanwhile, the elevated flow has an average depth of 0.29 m across all sites, which is greater than the depth required by any of the species during the spawning period (but more closely matches the depths required before the requirements were adjusted in Chapter Three). With this regime, the baseflow velocity averaged at 0.47 m s^{-1} and the elevated velocity at 0.55 m s^{-1}– both of which are higher than those required by any of the species at any life stage. However, it has been documented that it is the *combination* of a number of factors, usually (depth and one other) and the way in which they interact that determines the quality of habitat. The interaction of these flow conditions and the quality of habitat produced is the focus of Chapter Seven. The re-distribution of flows across the year became ineffective at high flow, particularly for the grayling and Atlantic salmon regimes. However, as the aim of the compensation redesign is to improve conditions at low flow, it proves to be beneficial when required.

Unsteady year long hydrograph

The brown trout flow regime, when applied to the 2002-2003 hydrograph produced the most extreme results. However, as found with the steady flow simulations, the difference between the decrease in depths/velocities in the baseflow period is small when compared to the grayling and Atlantic salmon flow regimes, whereas the difference in the increase in depths/velocities between the three flow regimes is somewhat larger. It can therefore be suggested that the brown trout flow regime is the most effective approach in elevating flow for a certain part of the year while causing minimum disruption to depths and velocities for the rest of the year. Again, this needs to be considered in more detail – with attention to the interactions of depth and velocity and the actual requirements of the three different species at different times of the year. This is the aim of the subsequent Chapter.

Spatial variation

The different depths and velocities produced at each site will render some sites more suitable habitat than others. For example, site 14 appears to have a suitable range of depths for most of the year, according to the HEC-RAS results (Figure 6.19), but has velocities which are much higher than those listed in Table 6.14, as can be seen in Figure 6.25. Again, this is an issue that can be assessed with the modelling technique to be used in Chapter Seven.

Spate flow

The spate flow has been shown to increase depths to 0.78 m (three times greater than those found on that day under current compensation flows) and velocities to 1.15 m s^{-1} (twice the velocity on that day under current compensation flows) and is consistent across all new flow regimes. The increase in velocity may increase shear stress and would therefore be fit for the purpose of flushing fine sediment through the gravel layer (Gilvear *et al*., 2002), the importance of which is addressed in Chapter Three. However, the rapid and intense increase in flow may prove detrimental to some organisms. As discussed in section 4.2, sudden increases in discharge can wash away small organisms such as macroinvertebrates (Crisp *et al*., 1995) and the velocities produced may be too severe for fish to resist. Conversely, fish (and to a degree, macroinvertebrates) require variability of flow as part of their life cycles (Hynes, 1970) and for the ecosystem as a whole they can act as 'reset mechanisms' (Poff and Ward, 1989). Given that the depth and velocity increases from the spate flow are not severe, the frequency is low and the duration short, the spate flow incorporated into this flow regime may be of benefit to the ecology of the River Derwent without causing catastrophic loss of life.

Time of elevated flow

The elevated (or spawning) flow periods, as they are currently incorporated into the new flow regimes, occur at the times of the year during which flow is naturally high (based on the flow record for the year 2002-2003 at Eddy's Bridge). This calls into question whether the elevated flows are actually necessary as the spawning times lie within times of natural higher flow. The depths and velocities produced under the new flow regimes in times of elevated flow are generally greater than the required depths and velocities, as found in Chapter Three and listed in Table 6.14. Therefore, the application of the elevated flows appear unnecessary. However, this will be investigated in more detail and with reference to specific sites in Chapter Seven.

6.6 Conclusion

Impoundment has caused an increase in flow depths and velocities at low flows, and an increase in depths and velocities at median and high flows, according to HEC-RAS results. Changing the compensation release regime will allow increases in depth and velocity for a certain period each year. The effect of this is most pronounced at low flows and when using the compensation release regime designed for the depth and velocity requirements of brown trout. The assessment of the year-long hydrograph presented an illustration of how flows may be affected across the year when the new compensation release regimes are applied to ‘actual flows’. In this case, the flow regimes designed for brown trout and Atlantic salmon appeared to be more effective at elevating depths and velocities than the regime for grayling. These changes will help to achieve the depth and velocity requirements of all three species, and will even take conditions above the optimum at times. Whether the new regimes will actually be beneficial to the ecology will be dependent upon the quality of habitat that is created as a result of the *combination* of depth and velocity. This requires a much more detailed investigation and will be addressed in Chapter Seven.

Chapter Seven

Fuzzy modelling of impacts of flow regimes on habitat

7.1 Introduction

Following the assessment of the change in flow regime using the hydrodynamic model in Chapter Six, the second part of the CRAM method utilises fuzzy rule-based modelling to determine how suitable the predicted flow depths and velocities would be for different fish species of different ages. Fuzzy modelling allows the suitability of a number of habitat feature combinations to be assessed by assigning them wholly, or partially to suitability groups. It allows for flexibility in suitability depending on the variables used and can incorporate use of expert knowledge as well as quantifiable data. The first part of the Chapter, Section 7.2, introduces habitat modelling and aims to justify the use of fuzzy rules within the context of habitat assessment. Section 7.3 describes the theory behind the model and the general methods of fuzzy modelling. It presents the fuzzy rules used for this specific study and describes the sensitivity analysis performed on the model. Results are presented in Section 7.4 and discussed in Section 7.5. Recommendations for redesign of the compensation release regime from Derwent Reservoir are also included in Section 7.5 and a summary of the Chapter is in Section 7.6.

7.2 Introduction to habitat modelling

There are a number of methods available to assess and to quantify ecological habitat suitability, each with benefits and limitations. Sections 7.2.2 to 7.2.4 will discuss the three key methods of habitat suitability assessment in order to place fuzzy rule-based modelling in context and justify the use of this method for this project.

7.2.1 Methods of assessing habitat quality

There are a number of methods for assessing the quality and abundance of instream habitat, each with its own merits and drawbacks. There are three main types of assessment:

- Habitat suitability indices/criteria linked to preference curves
- Multivariate analyses
- Fuzzy assessment

More traditional, preference-based methods (e.g. Bovee, 1986; Rubec *et al*., 1999; Brown *et al.,* 2000) have drawbacks, primarily in the inability to account for variable interaction in the assessment of habitat suitability. Preference rules were followed by multivariate analyses (e.g. Gore and Judy, 1981; Orth and Maughan, 1983; Jowett and Richardson, 1990) which allow a more comprehensive analysis of the overall role of variables (such as depth and velocity in a certain area of stream and their interactions). However, these methods are still subject to the uncertainties associated with habitat selection data (i.e. that an area is ‘suitable’ or ‘unsuitable’).

All habitat is potentially 'suitable' if an animal has no option but to live there for a certain amount of time, but not permanently. Thus, all habitat suitability indices are some degree greater than 0 (the numerical term assigned as 'unsuitable'), but how much greater than zero is not considered in multivariate analyses. Therefore, fuzzy rules have been applied to habitat suitability assessment in order to allow the user to account for ambiguity and include a previously un-mined data source – that of the environmental expert as unwritten knowledge.

7.2.2 Habitat suitability indices (preference rules)

Habitat Suitability indices (HSIs) can represent instream variable preferences for variables such as depth, velocity, substrate size, cover etc (Ahmadi-Nedushan, 2006). They are specific to a named species and can be applied to different life stages. Habitat suitability indices have been successfully used in the UK to assess the quality of habitat for different species in riverine environments and specifically to assess the impacts of changing flow regimes on habitat suitability (e.g. Maddock *et al.,* 2001). Bovee (1986) identified three types of HSI:

- Category I – professional judgement/literature
- Category II – habitat use by species observed within a stream, for the purpose of habitat studies
- Category III – habitat preference – deduced from observations taken from category II type investigations, combined with the availability of habitat combinations. Preference can be quantified as a ratio:
  percent utilisation : present availability (Ahmadi-Nedushan, 2006).

The overall quality and quantity of physical habitat is dependent on more than one variable and it is common practice to develop a *composite HSI,* which is intended to account for the cumulative effect of different variables. It can be calculated thus:

$$HSI = SI_1 \times SI_2 \times \ldots\ldots \times SI_n \qquad \textbf{Equation 7.1}$$

where: *HSI* = composite suitability index*, SI* = suitability of habitat variable 1, 2, … *n*. The suitability indices can be weighted according to the level of impact that each variable may have within a habitat (Leclerc, 2005).

The main tool used to determine habitat preferences in ecological modelling is the physical habitat simulation model (PHABSIM; Milhous *et al*., 1989), which is a component of the Instream Flow Incremental Methodology (IFIM; Bovee, 1982). PHABSIM consists of a hydraulic model and a habitat model (Beecher *et al*., 2002). The habitat model integrates the habitat quality for given depths, velocities, substrates and covers to give an index of microhabitat, the weighted usable area:

$$WUA = \Sigma_{i=1}^{n} Ai * Ci \qquad \textbf{Equation 7.2}$$

PHABSIM uses the Category III HSIs, as described above.

It is generally accepted that there are limitations to the HSI method, including those apparent in PHABSIM (e.g. Bourgeois *et al*., 1996; Guay *et al*., 2000; Schneider *et al*., 2002; Ahmadi-Nedushan, 2006). Primarily, simple HSIs assume that habitat variables (e.g. depth, velocity, substrate) can be treated separately and that there is no interaction between the variables in the determination of habitat quality (Leclerc, 2005, p 435). However, this is not a true representation of the habitat. It has been documented that simply multiplying the suitability of a number of habitat variables may produce errors (Vismara *et al*., 2001) because different variables may be dependent upon each other and have greater or lesser weighting in habitat quality than one another (Mould, 2006).

There are a number of other limitations. First, the model assumes that a large amount of poor habitat behaves in the same way as a small amount of excellent habitat. This may lead to a lack of correlation between the weighted usable area (WUA) and measured fish densities (Bourgeois *et al*., 1996). Second, equifinality can occur when a number of depth, velocity and substrate combinations produce the same WUA, but support different fish biomasses (Mathur *et al*., 1985). Third, differences in PHABSIM outputs may occur when the flows used in the model differ from the flows occurring at the time HSIs were developed (Bourgeois *et al*., 1996). Finally, more frequent flow conditions may artificially appear preferable if they are very abundant (Leclerc, 2005 p 435).

7.2.3 Multivariate analyses

In response to the limitations associated with assessment of habitat variables independently of each other (e.g. the assumption that there is no interaction between habitat variables), there has been a development of multivariate approaches, including combined preference functions, logistic regression, multivariate regression, ridge regression, principal component regression, generalised linear models, generalised additive models and neural networks (Schneider *et al*., 2002; Ahmadi-Nedushan *et al*., 2006). The merits and limitations of such approaches are summarised in Table 7.1. Lane *et al*., (2006) note that the advantage to probabilistic approaches (over preference curves) is that predictions from hydraulic models are considered simultaneously rather then independently, therefore considering their combined impacts. Leclerc (2005, p 346) notes that these *probabilistic,* multivariate approaches may offer better possibilities for model validation and transferability of methods from one river to another. For example, in regression techniques, independent variables can be standardised and logistic regression coefficients can be compared to identify which variables cause the greatest increase or decrease in the odds ratio. A merit of the probabilistic method is that it is similar to the HSI method and can produce a Probabilistic Habitat Index (PHI) which can predict the *probability* of the presence of individuals in certain habitats (Leclerc, 2005):

$$PHI(x, y, Q) = \frac{1}{1 + e^{-P(V,H,S)}}$$ **Equation 7.3**

where *P* is a polynomial composed of linear and/or quadratic terms that are functions of controlling abiotic variables *V, H, S*, etc where *V* is the velocity, *H* the depth and *S* the substrate grain size.

Jorde *et al*., (2001) highlight some of the limitations of multivariate techiniques as being the neglect of some of the possible combinations of physical parameters and lack of consideration given to spatial connectivity and networking of habitats. The method also assumes that knowledge is possessed to link discharge to velocity, depth and substrate size, and that the quantitative data that are held capture all the known or available information.

Approach	**Merits**	**Limitations**	**Applications**
Multiple linear regression	Straight forward to fit to observations and analyse abundance data. Widely available in most computer packages	Not appropriate when assumptions of normality, linear relationship and constant variance of errors are violated Estimated parameters are not stable when the variables are highly correlated	Vadas and Orth, 2001; Vismara et el., 2001; Yu and Lee, 2002
Logistic regression	Easy to use for the analysis of presence-absence data and is implemented in many software packages	Better adopted to presence-absence data, and hence they reduce the information available from continuous variables	Geist *et al*, 2000; Guay *et al*., 2000; Manel *et al*., 2001

Generalised linear models	Not limited to normally distributed variables. Offers flexibility to choose different types of distribution for responses	For small samples (less than 20), maximum-likelihood estimates are biased	Labonne *et al*., 2003
Generalised additive models	Considers non-linear relationships between response variables and environmental variables	Does not produce a parametric mathematical function	Lehman, 1998; Milner *et al*., 2001
Artificial neural networks	Ability to implicitly detect complex non-linear relationships between response variables and environmental variables	'Black-box' approach – contribution of the input variables in predicting species response is difficult to disentangle within the network (less straight forward than regression methods)	Gozlan, *et al*., 1999; Ibarra *et al*., 2003; Mastorillo *et al*., 1997
Ridge regression	Can handle highly correlated environmental variables (multicollinearity). Is beneficial when the amount of data is not large relative to number of variables	The estimates are biased	Not used yet
Principal component regression	Useful when environmental variables are numerous and the independent variables are highly correlated		Corbacho and Sanches, 2001; Jutila *et al*., 2001; Neumann and Wildman, 2002
Neural networks	Suited to analysis of non-linear relationships between species distribution and environmental variables. Can handle multivariate systems.	Do not provide users with a conventional mathematical function. Difficult to export a model.	Gozlan *et al*., 1999; Ibarra *et al*., 2003; Mastorillo *et al*., 1997

Table 7.1 Approaches to habitat assessment, after Ahmadi-Nedushan *et al*., 2006.

7.2.4 **Fuzzy modelling approach**

There exist limitations of the preference and multivariate methods, as outlined in Sections 7.2.2, 7.2.3 and Table 7.1. There are also limitations in environmental data (e.g. random variables, inaccurate data, estimations instead of precise measurements: Ahmadi-Nedushan *et al*., 2006). Therefore, it is prudent to be aware of the uncertainties with such modelling techniques and perhaps even to use them to the advantage of the investigation. Fuzzy modelling is a habitat assessment method by which the inherent ambiguities of environmental data and statistical tests can be accommodated. It addresses the main problem of preference curves (assumption that habitat controls interact independently) by producing an overall output that is based on a combination of the individual inputs of all the different parameters (e.g. a depth of 10 cm may be classed as poor habitat when velocity is 10 $m^3 s^{-1}$, but medium when velocity is 20 $m^3 s^{-1}$). It addresses the limitation of multivariate analyses (the degree to which a combination of parameters are suitable) by allowing a variable to belong to more than one suitability category (e.g. partially poor, partially medium suitability).

Fuzzy set theory, developed by Zadeh (1965), allows the processing of imprecise information by means of membership functions, in contrast to Boolean characteristic mappings (Zadeh, 1965, in Ahmadi-Nedushan *et al*., 2006). In a classic or 'crisp' data set, the characteristic mapping can take two values: it either belongs to the set, or it does not. In the fuzzy approach, an element can belong to a fuzzy set with its membership degree varying from one to zero (Adriaessens *et al*., 2004). Further to the applicability of the model to ambiguous data, the method also allows the knowledge of environmental experts, which is not always documented, to be incorporated into the analysis by assigning descriptions of a certain parameter to different values of that parameter. Fuzzy groups of habitat quality can be assigned titles such as 'low', 'medium' or 'good' for each combination of variables (Leclerc, 2005 p 437). The number of combinations is a function of the number of variables (e.g. depth, velocity) and the number of terms defining them (e.g. poor, medium, good) (Equation 7.4, Leclerc, 2005).

$$N = \prod_{j=1}^{J} k_j \qquad \textbf{Equation 7.4}$$

where J is the number of variables considered and k_j is the number of linguistic classes defining each of them.

The variables may belong to a number of categories, and overlapping may occur. Thus, a certain velocity may belong primarily to one category, but at some times can be part of an adjacent category (Figure 7.1). In this way the fuzziness allows the user to account both for the uncertainty in the scientists' understanding/data and the uncertainty of the requirements of fish (fish have preferred habitats and habitats which they will use when forced to (e.g. when there is pressure for space or flows are unusually high/low in their territorial area). The output of the fuzzy model is derived by calculating the degrees of fulfilment of all the rules in a whole rule system. These rules are then used for a 'defuzzification'. Thus, the result, which is still in the form of a fuzzy set, is transformed back into a standardised number to describe habitat quality (Jorde *et al*., 2001). Further description of how the method works is included in the explanation of the methods used, in Section 7.3.

The main tool for application of fuzzy modelling so far is CASIMIR (Computer Assisted Simulation Model for Instream Flow Requirements: Stuttgart Institution of Hydraulic Engineering, 1990) (Jorde *et al*., 2001).

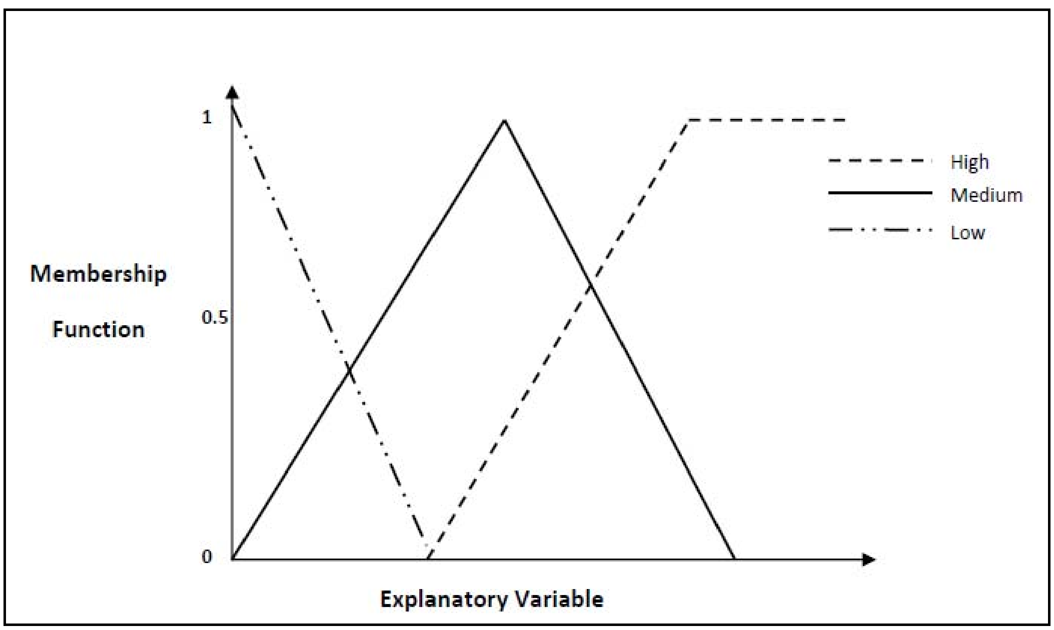

Figure 7.1 Fuzzy group interaction between habitat variable classes. Source: Leclerc, 2005 p 438

The methods used for determining fuzzy groups and their applications to habitat quality assessment are addressed in detail in Section 7.3.

There is strong and abundant evidence supporting the use of fuzzy rule-based modelling in the evaluation of habitat quality. First, the method allows the uncertainties of environmental data to be accommodated within the evaluation of habitat quality (Jorde *et al*., 2001; Van Broekhoven *et* *al*., 2006). Second, the experience and knowledge of environmental experts can be applied to the model and incorporated into the assessment of habitat quality (Jorde *et al*., 2001; Adriaenssens *et al*., 2004; Ahmadi-Nedushan, *et al*., 2006; Mouton *et al*., 2007). Mouton *et al*., (2007) note that use of expert knowledge may mean that the model can become transferable to other rivers with similar habitat characteristics. Third, by the nature of data input, calculation method and output, and its interpretability, the model lends itself to wider distribution and through its transparency is effective for use between stakeholders, managers, etc. who are unfamiliar with the scientific methods involved in habitat preference and multivariate techniques (Adriaenssens *et al*., 2004; Van Broekhoven *et al*., 2006; Mouton *et al*., 2007; Mouton *et al*., 2009). Fourth, the model considers multivariate effects but no independence of the input parameters is required (Ahmadi-Nedushan *et al*., 2006). This sets it above the preference and multivariate methods in terms of model sophistication. Finally, the fuzzy rule method allows the user to assess large numbers of combinations of physical parameters into habitat simulation tools and the addition of further parameters is easy, if required (Jorde *et al*., 2001).

7.2.5 Limitations

The application of the knowledge of individuals will naturally lead to inconsistencies, as noted by a number of authors (Acreman and Dunbar, 2004; Adriaenssens *et al*., 2004; Randin etal., 2006; Fitzpatrick *et al*., 2007). Mouton *et al*., (2009) also note that the formalisation of such data can be difficult and tedious. Van Broekhoven *et al*., (2006) note that the formulation of the fuzzy groups, being largely subjective, may lead to inconsistencies in analysis between different users. It may be difficult to convince river managers that such a subjective approach is scientifically sound. Some river managers may feel that there is too much subjectivity involved in aspects such as: choice of input variables, amount of overlap, formulation of fuzzy rules based on expert knowledge (Adriaenssens *et al*., 2004).

The interpretability of the model can become difficult if there are a large number of parameters, as the number of simulations required will increase with addition of each new parameter (Van Broekhoven *et al*., 2006; Mouton *et al*., 2008). The number of fuzzy rules is rapidly increasing as more parameters are considered (Ahmadi-Nedushan *et al*., 2006).

7.2.6 Applications

Although not widespread in the UK to date, there has been some application of fuzzy rule-based modelling to ecological investigations. Use is also growing in Europe and further afield. Schneider *et al*., (2002) note that the main application of fuzzy modelling is through the use of the Computer Aided SImulation Model for Instream Flow Requirements (CASIMIR) software and it has specific benefits in the setting of minimum flows in regulated rivers, with variation in compensation flows across the year to reflect times of elevated flow in fish spawning periods. Jorde *et al*., (2001) successfully applied the fuzzy-rule methodology (in CASIMIR) to rivers in Switzerland and found the results to more closely correlate with observed fish densities than results gained from preference functions. Mouton *et al*., (2007) took the fish habitat module of CASIMIR and coupled it with the hydraulic model HEC-RAS (as is carried out in this study) to define the suitability of certain flow conditions and management options (weir removal) for fish habitat in a Belgian river. Lane *et al*., (2006) and Mould *et al.,* (2007) built upon the simple 1D flow model and fuzzy rule methods by introducing an analysis of 2D and 3D hydrodynamic models for use with fuzzy rule-based assessment. The application was specific to the redesign of compensation flows in regulated rivers and the study highlighted the need to explore flow-biology interactions at the *within-reach* scale, especially when dealing with low flows (Lane *et al*., 2006).

The methods can be applied to other ecological groups besides fish: Mould (2006) conducted fuzzy rule-based analysis in an investigation into the impacts of changing reservoir compensation releases on macroinvertebrates (and fish also), in rivers in North Yorkshire, UK. Van Boekhoven *et al*., (2006) used fuzzy rules to identify areas of suitable habitat for macroinvertebrates in rivers of Central and Western Plains of Europe.

Further to water resource investigations, fuzzy rule-based modelling has been applied to water quality classifications, impact assessment of fish farming on benthos, threatened species classification, integrated environmental management and general sustainable development (Adriaenssens *et al*., 2004). CASIMIR can also be used in flood defence and watershed management studies to determine the effect of various flow regimes/conditions on habitat suitability (Schneider *et al*., 2002).

7.3 Methodology and fuzzy rules used

7.3.1 Fuzzy-rule theory

Fuzzy rules were developed and combined with hydraulic data (as created in Chapter Five) to create a ‘map’ of spatial distribution of habitat suitability along the River Derwent for different flow periods in the year. The analysis was applied to three fish species: brown trout, grayling and Atlantic salmon and the new compensation release regimes that were proposed in Chapter Five were tested on the species for which they were designed and for the other two species, to determine their overall impact. Fuzzy rule-based analysis was chosen because of its many attributes (discussed in Section 7.2.3) and for a number of factors which make its application practical and flexible. The method allows the combination of a number of data sources which may have a similar general result, but do not completely mirror each other (e.g. brown trout depth requirements, as observed in the field, will be limited to the areas *preferred* because that is where the fish was found, but will not account for areas where it is *possible* for the fish to live). It also enables the fuzzy groups that are used to be modified in relation to the advice of fisheries experts and with reference to a specific river. Furthermore, it is a transparent process which lends itself to the involvement of river users and managers.

The method closely follows that described by Lane *et al*., (2006) because of the similarities in the structure and aims of both investigations. As with Lane *et al*., (2006), depth and velocity were the variables considered in this analysis. This is because they are the primary variables that are believed to affect habitat quality and are the features of the river most significantly affected by impoundment and compensation flows. Substrate and cover are also considered important variables of fish habitat, but were not assessed here as they are assumed to remain relatively constant in response to changes in regulation.

Although there are some strong similarities between the methodology adopted in this investigation and that of Lane *et al.,* (2007), there are also differences which make this study original and new. First, the current study is time dependent and it considers flows during different life stages of fish and is applied to a whole year's worth of actual flow data, rather than being limited to flows of specified magnitude. Second, it is conducted at the tributary/river scale rather than the reach or sub-reach scale. This allows for a broader spatial analysis and considers the suitability of habitat for fish along the whole river rather than fine detailed habitat assessments. Finally, and as a result of point two, the investigation is conducted at the 1D rather than 2D scale. This is a necessity of investigating a long (27 km) stretch of river. At this scale, the detail of analysis involved in the application of a 2D model would be neither practical nor necessary.

Depth and velocity can both be interpreted as good, medium and poor. Habitat can be split into six classes: unsuitable, very poor, poor, good, very good and excellent (Lane *et al*., 2006). Fuzzy subsets were then defined for depth (D_i) and velocity (V_i) that defined the grade of membership of each predicted depth (*d*) or velocity (*v)* of each of the *i* (poor, medium or good) subsets (Lane *et al.,* 2006):

$$\begin{aligned}
D_p &= \{[d, \mu_{Dp}(d)]; d \in D, \mu_{Dp}(d) \in [0,1]\} \\
D_m &= \{[d, \mu_{Dm}(d)]; d \in D, \mu_{Dm}(d) \in [0,1]\} \\
D_g &= \{[d, \mu_{Dg}(d)]; d \in D, \mu_{Dg}(d) \in [0,1]\} \\
V_p &= \{[v, \mu_{Vp}(v)]; v \in V, \mu_{Vp}(v) \in [0,1]\} \\
V_m &= \{[v, \mu_{Vm}(v)]; v \in V, \mu_{Vm}(v) \in [0,1]\} \\
V_g &= \{[v, \mu_{Vg}(v)]; v \in V, \mu_{Vg}(v) \in [0,1]\}
\end{aligned}$$

Equation 7.5

where: *p* is poor, *m* is medium and g is good; and $\mu Li(l)$ is the grade of membership of the predicted value *l(d* or *v*) in *Li* (*Di* or *Vi*), which equals one for at least one value of *L* for each *i*. In this scheme, when 0< $\mu Li(l)$ <1, *l* has a partial membership of a certain group, and this is the sense in which the analysis is fuzzy, with *l* potentially being a partial member of more than one *Li*. Lane *et al*., (2006) then specified a fuzzy rule for habitat (*Hk*) based on two premises (for D and V):

If $Di \otimes Vj$ then H_k, for K values of *k* **Equation 7.6**

where *K* is the number of habitat classes*, I* is the subset of depth and *j* is the subset of velocity. In this case, *i* and *j* both = *3*, therefore there are 9 rules (possible combinations), and potentially nine values of *k*. In order to capture the fuzziness of the analysis, membership of *Di* and *Vj* is expressed as a grade which may vary between zero and one. Therefore, they used a product operation rule (Wang, 1994) to define the degree of fulfilment of a particular habitat class:

$$\mu_{Hk} = \mu_{Hk,Di(d)}\mu_{Hk,Vj(v)}$$ **Equation 7.7**

where μ_{Hk} is the degree of fulfilment of habitat class *k*, as defined by each possible combination of *Di* and *Vj* (from Equation 7.6), given the predicted values of *d* and *v*. The nine rules that come from Equation 7.6 could be used to provide nine habitat classes. However Lane *et al*., (2006) used a symmetrical habitat classification, weighting *D* and *V* equally in the determination of habitat suitability, as illustrated in Table 7.2.

		Velocity		
		Poor (presence rarely found)	**Medium** (presence sometimes found)	**Good** (presence often found)
Depth	**Poor** (presence rarely found)	**Unsuitable habitat** **0**	**Very poor habitat** **1**	**Poor habitat** **2**
	Medium (presence sometimes found)	**Very poor habitat** **1**	**Good habitat** **3**	**Very good habitat** **4**
	Good (presence often found)	**Poor habitat** **2**	**Very good habitat** **4**	**Excellent habitat** **5**

Table 7.2 Definition of habitat classes in relation to the rule set defined in Equation 7.5. (Source: Lane *et al*., 2006)

This could be made more sophisticated by changing the weightings to reflect the known importance of velocity and depth in contributing to a particular habitat class, possibly informed by field data or traditional habitat suitability analyses, or calibrated on to measured relationships between habitat and productivity for a specific reach or set of reaches. Lane *et al*., (2006) do not explore this in their paper and it is not explored in this thesis.

The analysis so far provides nine outcomes which indicate the degree of fulfilment of each rule. If there was no fuzziness in the system, then there would only be a single outcome. As the level of fuzziness increases, so the number of outcomes increases to the maximum of nine. In order to provide a single habitat suitability index they defuzzify the analysis to produce a single ‘crisp’ number. This can be done in a number of ways. Lane *et al*., (2006) weight the proportion of membership of the scores in Table 7.2: with perfect membership of excellent habitat scaled as 1.

7.3.2 Fuzzy rules used and simulations applied

The fuzzy rule-based modelling was based on habitat requirements for fish rather than macroinvertebrates, because these are much more well documented and have been the focus for the majority of the investigation (CRAB, HEC-RAS). Fuzzy rules were defined for the three species being investigated (brown trout, grayling and Atlantic salmon) and for three life stages (spawning, juvenile and adult). The habitat preferences were taken from observations documented in the academic literature (e.g. Raleigh *et al*., 1986; Heggennes and Ssaltveit, 1990; Sempeski and Gaudin, 1995; Heggennes, 1996; Moir *et al*., 1998; Degerman *et al*., 2000; Lucas and Bubb (2005) and a general rule was applied that depths/velocities that were frequently used were considered 'good'; sometimes used were of 'medium' quality and never used were 'poor'. Depths and velocities from a number of sources were considered but as there was a large degree of discrepancy between some of the findings, the rules formulated for this study were based on fewer sources which came from rivers that most closely reflected the characteristics of the River Derwent. The importance of using Habitat Suitability Indices that are appropriate to the river in question is highlighted by Maddock *et al.,* (2001) who found that curves developed from data from a Norwegian river and a UK river produced very different habitat preference results. The fuzzy rules used are collated in Tables 7.3 to 7.5 and can be displayed graphically as Figures 7.2 to 7.4.

	Poor	**Medium**	**Good**	**Source**
Spawning Depth (m)	>150	0-30	30-150	Raleigh, 1986
Spawning Velocity (ms)	>120	0-20, 60-120	20-60	Raleigh, 1986
Juvenile Depth (m)	<20, >100	20-30, 50-100	30-50	Heggennes and Saltveit, 1990
Juvenile Velocity (ms)	>80	35-80	0-35	Heggennes and Saltveit, 1990
Adult Depth (m)	<20, >100	20-40, 85-100	40-85	Heggennes, 1996
Adult Velocity (ms)	>50	25-50	0-25	Heggennes, 1996

Table 7.3 Fuzzy groups used for analysis of habitat suitability for brown trout

	Poor	**Medium**	**Good**	**Source**
Spawning Depth (m)	<20, >200	20-30, 80-200	30-80	Lucas and Bubb, 2005
Spawning Velocity (ms)	>100	0-10, 70-100	10 to 70	Lucas and Bubb, 2005
Juvenile Depth (m)	>40	0-10, 30-40	10 to 30	Sempeski & Gaudin 1995
Juvenile Velocity (ms)	>60	0-20, 40-60	20-40	Degerman *et al.,* 2000
Adult Depth (m)	<10, >175	10-25, 80-175	25-80	Lucas and Bubb
Adult Velocity (ms)	>120	0-10, 70-120	10 to 70	Lucas and Bubb

Table 7.4 Fuzzy groups used for analysis of habitat suitability for grayling

	Poor	**Medium**	**Good**	**Source**
Spawning Depth	<10, >50	10-15, 35-50	15-35	Moir *et al.*, 1998
Spawning Velocity	<35, >110	20-35, 80-110	35-80	Moir *et al.*, 1998
Juvenile Depth	<10, >110	10-20, 70-110	20-70	Heggennes and Saltveit, 1990
Juvenile Velocity	>140	0-10, 60-140	10 to 60	Heggennes and Saltveit, 1990
Adult Depth	<10, >110	10-20, 70-110	20-70	Heggennes and Saltveit, 1990
Adult Velocity	>140	0-10, 60-140	10 to 60	Heggennes and Saltveit, 1990

Table 7.5 Fuzzy groups used for analysis of habitat suitability for Atlantic salmon

(a) (b)

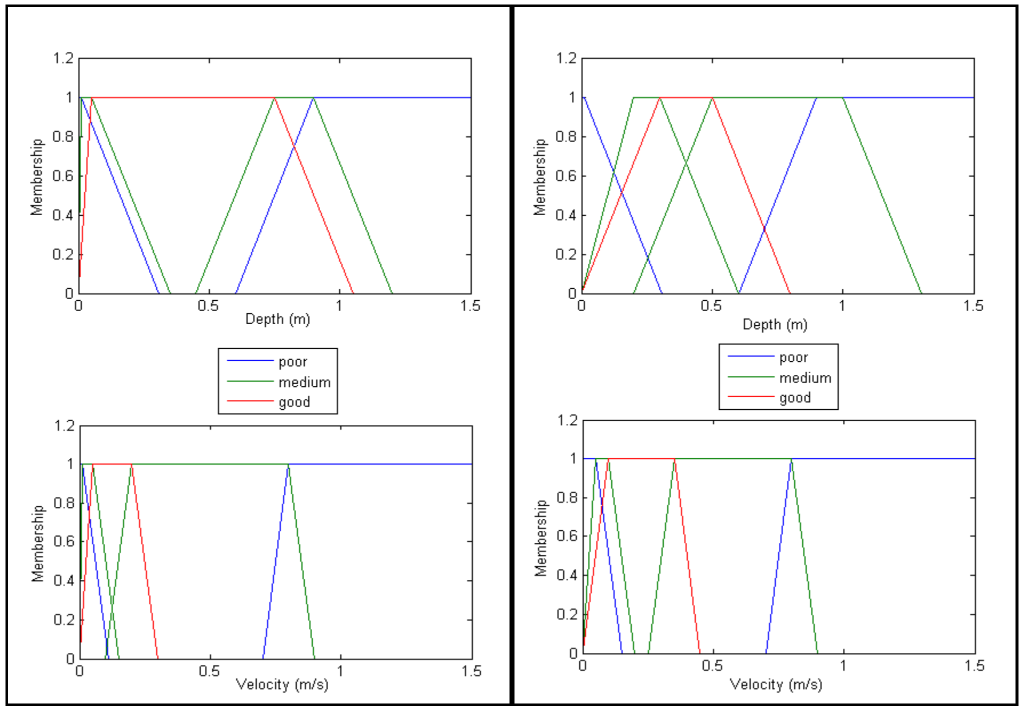

Figure 7.2 Graphical representation of the interaction of fuzzy groups for brown trout: (a) spawning and (b) juvenile

(a) (b)

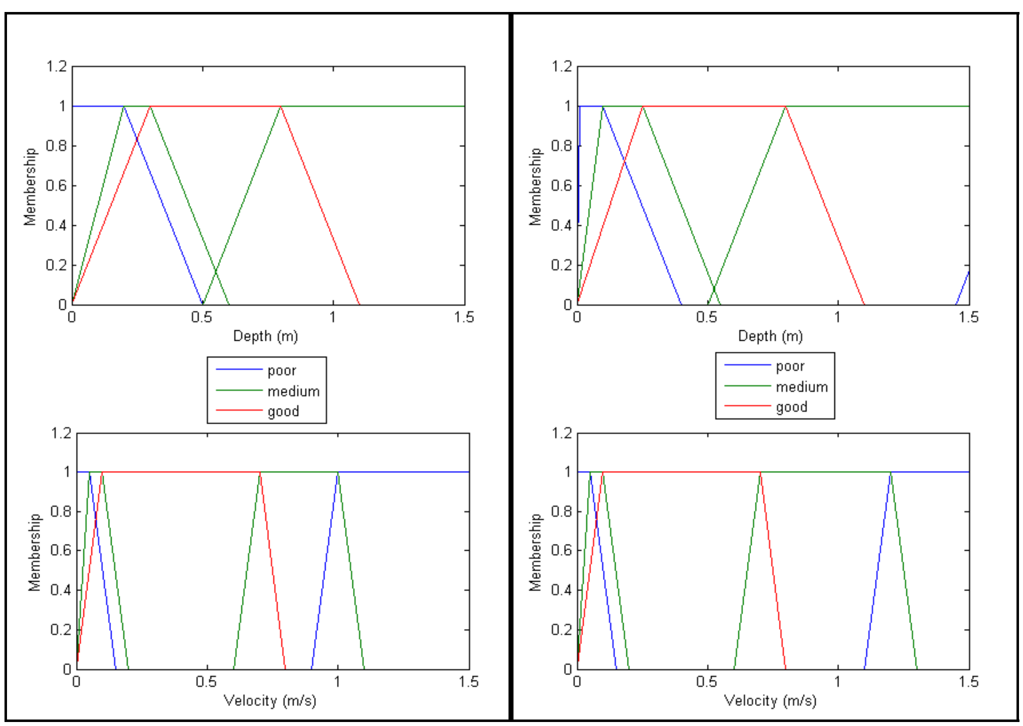

Figure 7.3 Graphical representation of the interaction of fuzzy groups for grayling: (a) spawning and (b) juvenile

(a) **(b)**

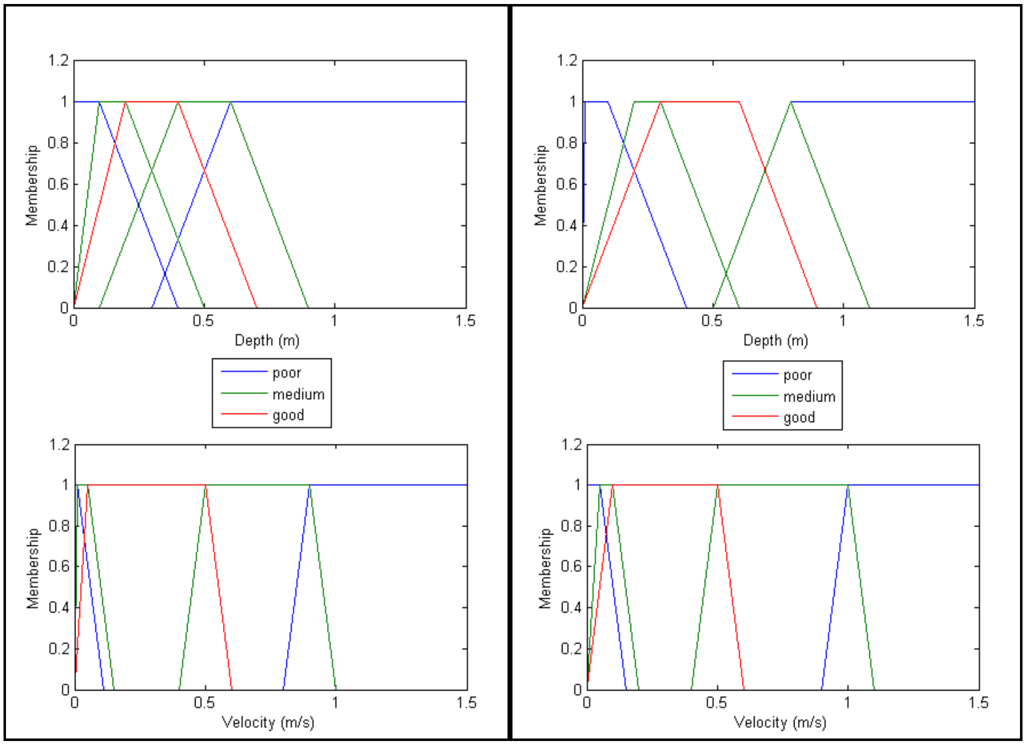

Figure 7.4 Graphical representation of the interaction of fuzzy groups for Atlantic salmon: (a) spawning and (b) juvenile

Because a number of different simulations and flow types were assessed using HEC-RAS (e.g. steady low, medium and high flows, unsteady annual flow based on 2002-2003 hydrograph), there were a large number of scenarios to be assessed. Therefore, as requirements for juvenile and adult fish were similar, these were consolidated to give one set of simulations, and another set of simulations was applied for spawning requirements of each species. The combination of flow scenarios and fish habitat requirements that were simulated are summarised in Tables 7.6 and 7.7.

Life stage fuzzy rules applied for →	Spawning			Juvenile		
Magnitude of flow applied to HEC-RAS →	Q95	Q50	Q5	Q95	Q50	Q5
Flow regime that was applied to HEC-RAS ↓						
Pre-impoundment						
Post-impoundment						
Regime suitable for **Brown trout** (9:3 months distribution)						
Regime suitable for **Grayling** (9:3)						
Regime suitable for **Atlantic salmon** (9:3)						
Regime suitable for **Brown trout** (11:1)						
Regime suitable for **Grayling** (11:1)						
Regime suitable for **Atlantic salmon** (11:1)						

Table 7.6 Combination of simulations applied to the fuzzy rule method of habitat assessment for this investigation – flows with only two levels – the baseflow and the elevated flow. This combination of simulations was repeated three times – for the habitat requirements for each of the three species studied (brown trout, grayling and Atlantic salmon, i.e. the fuzzy groups defined in Tables 7.3 to 7.5)

Fuzzy rules used for →	Brown trout		Grayling		Atlantic salmon	
HEC-RAS Results used from which flow regime ↓	Spawning	Juvenile	Spawning	Juvenile	Spawning	Juvenile
Current regime						
Regime suitable for **Brown trout**						
Regime suitable for **Grayling**						
Regime suitable for **Atlantic salmon**						

Table 7.7 Combination of simulations applied to the fuzzy-rule method of habitat assessment for this investigation – unsteady flow based on hydrology data for the water year 2002-2003. This combination of simulations was repeated three times – for the habitat requirements for each of the three species studied (brown trout, grayling and Atlantic salmon, i.e. the fuzzy groups defined in Tables 7.3 to 7.5)

7.3.3 Precision settings and sensitivity analysis

The precision values for depth and velocity are a measure of confidence in the preferences used (i.e. the slope of the line, or the degree to which the groups overlap) and can be altered within the model. This enables the user to allow for uncertainties in the hydrodynamic outputs or habitat preference data. The precision values in this case were set as 0.3 for depth and 0.1 for velocity. The sensitivity of the model to changes in precision values was assessed by applying the HEC-RAS results from a steady pre-impoundment low flow (Q95) simulation, based on brown trout spawning fuzzy groups. Results can be seen in Figure 7.5. The precision value for depth or velocity, which ranges from 0 to 1, controls the transition from 'no membership' to 'total membership' of a specified group (e.g. poor, medium, good). Therefore, HSI increases with depth or velocity precision value because groups become 'more fuzzy' – i.e. there is more overlap between the poor, medium and good groups, allowing a greater range of habitats to be considered suitable. As fuzziness increases, spatial patterns of HSI will be reduced (because it is 'easier' to belong to a membership group and therefore there will be less differentiation between sites). Therefore, spatial patterns of HSI should considered more important than the cumulative HSI for a larger area (because as the number of HSIs, (and therefore the overall fuzziness) accumulate, the HSI result will have increased uncertainty. There may be the requirement, then, for the HSI to be scaled by a fuzziness index – i.e. some quantification given to the fuzziness (and therefore reliability) of the HSI result.

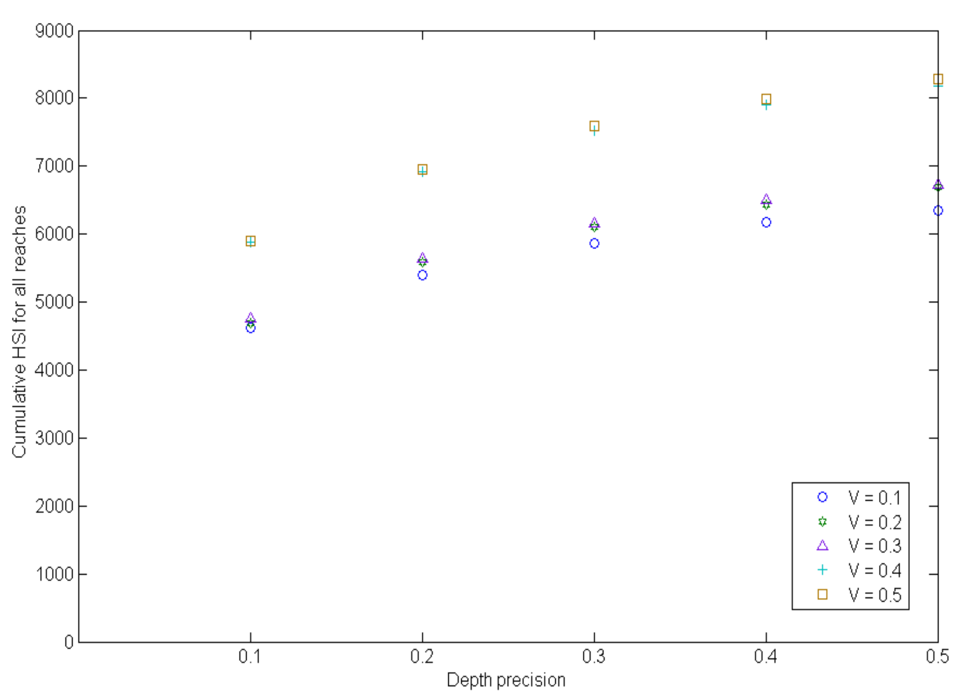

Figure 7.5 Sensitivity analysis results for a range of depth and velocity precision combinations

7.4 Results

Because of the large number of flow regime and species combinations that were assessed with the fuzzy rules for this Section, the results have been placed, in graphical form, in Appendix Three. The following Sections (7.4 and 7.5) describe and discuss the results with reference to Appendix Three.

7.4.1 Effects of impoundment

Brown Trout

The impacts of impoundment on habitat suitability for brown trout can be seen by comparing the pre- and post-impoundment graphs in Appendices Three (a) and Three (b) (pages 243 to 248). At low flows (Q95) and median flows (Q50), habitat quality for spawning brown trout appears to have deteriorated slightly in the downstream reaches of the river (e.g. at cross-sections 1-7, see Figure 6.2). Closer to the dam (e.g. cross-section sites 15-16), habitat appears to have improved following impoundment. At high flows (Q5), there is little change in habitat quality, other than the slight improvement in upstream reaches. The fuzzy model results suggest that there is a greater difference in habitat suitability between low and high flows pre-impoundment, most likely due to the greater difference between low and high flows experienced pre-impoundment than post-impoundment. For juvenile brown trout, post-impoundment low flow habitat quality is an improvement on the pre-impoundment habitat quality and the median flow appears to provide a more suitable habitat than low or high flows. Pre-impoundment, the low flow depths and velocities produced more suitable habitat indices than median or high flows, again, due to the greater low flow value that was experienced prior to impoundment. There is a greater improvement in habitat quality for median flows than low or high flows, between the pre- and post-impoundment results.

Grayling

The impacts of impoundment on habitat suitability for brown trout can be seen by comparing the pre- and post-impoundment graphs in Appendices Three (c) and Three (d) (pages249 to 254). There is a marked improvement post-impoundment in habitat suitability at high flows (Q5) for spawning grayling, at all sites, with some excellent quality habitat occurring at site 10. There is also an improvement at some sites (e.g. 8, 16), at median and low flows, but a deterioration at others (e.g. 7, 14, 15). Changing the impoundment of the river has caused different effects in different reaches. There has been a less significant change in habitat suitability for juvenile grayling between pre- and post-impoundment flows, although there is still a clear improvement post-impoundment at high flows for most sites. At median flows, there is again an improvement at some sites and deterioration at others, with sites 7 and 8 being notable examples of increased habitat quality and site 15 suffering a decrease. Low flows post-impoundment appear to cause an improvement in the upstream reaches and a slight deterioration in habitat suitability in the lower reaches. On the whole, habitat quality for grayling in the River Derwent appears to be medium to poor, with some isolated areas of very good habitat quality.

Atlantic Salmon

The impacts of impoundment on habitat suitability for brown trout can be seen by comparing the pre- and post-impoundment graphs in Appendices Three (e) and Three (f) (pages 255 to 260). Spawning habitat for Atlantic salmon at low and median flows has improved or remained unchanged following impoundment and there has been an improvement in habitat quality at high flows. At low and median flows habitat quality across different sites ranges from poor to very good, while all sites at high flows have low habitat qualities. The same trend occurred for juvenile Atlantic salmon at high and median flows, as for spawning Atlantic salmon. At low flows, habitat quality appears to improve in the upstream and downstream reaches and deteriorate in the mid-reaches. Habitat quality for juvenile Atlantic salmon at high flows is generally poor both pre- and post-impoundment, and at median and low flows is mostly medium to poor, with a few exceptions experiencing good conditions.

7.4.2 Effects of new flow regimes – steady analysis

The effect of the new flow regimes, on each species, (as suggested by CRAB (Chapter Four), based on requirements of selected species), are summarised in Tables 7.8 to 7.10 and are illustrated in Appendices Three (a) and Three (b) (brown trout), Three (c) and Three (d) (grayling) and Three (e) and Three (f) (Atlantic salmon), (pages 243 to 260). Because low flows are the issue being addressed through this project and will be the most affected by change in flow regime (see Chapters Three and Six), it is the impact of the new flow regimes on Q95 flows that have been focused upon in the Tables. However, where there are any notable impacts on the median and high flows, these have also been stated.

	Brown trout: Spawning		Brown trout: Juvenile		Effect of each regime (as detailed in first column), on juveniles, when the 11:1 distribution is used instead of the 9:3 distribution
Modified flow regime:	**Description of habitat created**	**Comparison to quality of habitat provided by current flow regime**	**Description of habitat created**	**Comparison to quality of habitat provided by current flow regime**	
Brown trout: based on 9:3 month distribution of flows	Poor at all sites	No improvement	Poor at most sites, medium at some (2,5,6,10,12)	Deterioration or no change at sites 1-6, 10-16 improvement at sites 7,8,9	Slight improvement, especially at site 6 – the site with the highest HSI.
Grayling: based on 9:3 month distribution of flows	Poor. Medium at site 5.	No change	Poor to medium. Optimum location is site 6.	Slight improvement at the sites with the most suitable habitat.	Slight improvement at site 6, slight deterioration at sites 7, 8 and 9.
Atlantic salmon: based on 9:3 month distribution of flows	Poor. Medium at site 5.	Slight deterioration at site 3. No change otherwise.	Poor to medium. Optimum locations are sites 10 and 11.	General deterioration except at sites 8 and 9.	Slight improvement at most sites. No change at site 16.

Table 7.8 Effect of new suggested flow regimes on habitat quality for brown trout. Information in the column titled 'Effect of 11:1 regime on juveniles' refers to the difference between the habitat suitability produced under the 11:1 month regime and the 9:3 month regime (not between the 11:1 month regime and the current flow regime. Results for this refer to juveniles only as when the 11:1 regime is applied, spawning release rate remains the same as in the 9:3 regime, but occurs for a shorter period. Therefore, there will be no change to the depths and velocities produced in the spawning period. Interpretations are based on data displayed in Appendices Three (a) and (b) (pp 234 - 248)

	Grayling: Spawning		**Grayling: Juvenile**		
Flow regime:	**Description of habitat created**	**Comparison to quality of habitat provided by current flow regime**	**Description of habitat created**	**Comparison to quality of habitat provided by current flow regime**	**Effect of 11:1 regime on juveniles**
Brown trout: based on 9:3 month distribution of flows	Mostly medium to good spawning habitat, best is at sites 8 and 9. Deterioration of habitat for some sites at high flows (Q5).	Improvement at most sties, deterioration at sites 3, 11 and 14.	Mostly medium. Poor at sites 14, 16.	Little change. Slight improvement at site 5. Deterioration at site 16.	Improvement at sites 3, 8, 11, 16.
Grayling: based on 9:3 month distribution of flows	Mostly medium to good. Poor at site 3.	Deterioration at the sites with highest HSIs (3 and 8), otherwise little change.	Ranges from poor to excellent, with most sites as poor/ok. Improvement can also be seen at median flows.	Little change except for improvement at site 3.	Improvement at site 8, deterioration at site 3, otherwise no change.
Atlantic salmon: based on 9:3 month distribution of flows	Ranges from poor to good. Optimum location is site 8.	Improvement at site 16, deterioration at sites 8 and 16, otherwise little change.	Poor to excellent. Optimum sites are 3 and 8.	Improvement at site 3, deterioration at site 8. Otherwise little difference.	Deterioration at site 3, improvement at site 8, otherwise little change.

Table 7.9 Effect of new suggested flow regimes on habitat quality for grayling. Information in the column titled 'Effect of 11:1 regime on juveniles' refers to the difference between the habitat suitability produced under the 11:1 month regime and the 9:3 month regime (not between the 11:1 month regime and the current flow regime. Results for this refer to juveniles only as when the 11:1 regime is applied, spawning release rate remains the same as in the 9:3 regime, but occurs for a shorter period. Therefore, there will be no change to the depths and velocities produced in the spawning period. Interpretations are based on data displayed in Appendices Three (c) and (d) (pp 249 - 254)

	Atlantic salmon: Spawning		**Atlantic salmon: Juvenile**		
Flow regime:	**Description of habitat created**	**Comparison to quality of habitat provided by current flow regime**	**Description of habitat created**	**Comparison to quality of habitat provided by current flow regime**	**Effect of 11:1 regime on juveniles**
Brown trout: based on 9:3 month distribution of flows	Ranges from poor to good. Sites 2 and 10 are optimum.	Deterioration at sites 2, 3 and 4. Others are unchanged. General deterioration of suitability median and high flows.	Ranges from poor to good. Mostly poor. Site 9 is good.	Deterioration at sites 4, 12 and 13. Improvement at sites 3, 7 and 9. General deterioration of suitability median and high flows.	Improvement at sites 3 and 13. Deterioration at sites 4 and 9.Closely matches HSIs for current regime.
Grayling: based on 9:3 month distribution of flows	Ranges from poor to very good. Best sites are 4 and 12. For median flows, higher HSIs occur with the new Grayling flow regime than the current compensation release regime.	Improvement at sites 2, 10, 12 and 15. Deterioration at sites 4, 13.	Ranges from poor to good, mostly medium.	Improvement at site 13, other wise little change. For median flows, higher HSIs occur at current compensation rate.	No impact.
Atlantic salmon: based on 9:3 month distribution of flows	Ranges from poor to excellent. Mostly medium/poor.	Improvement at sites 2, 10, 12, and 15. Deterioration at sites 4 and 13.	Ranges from poor to very good. Mostly medium.	Deterioration at site 4, otherwise little change.	No impact.

Table 7.10 Effect of new suggested flow regimes on habitat quality for Atlantic salmon. Information in the column titled 'Effect of 11:1 regime on juveniles' refers to the difference between the habitat suitability produced under the 11:1 month regime and the 9:3 month regime (not between the 11:1 month regime and the current flow regime. Results for this refer to juveniles only as when the 11:1 regime is applied, spawning release rate remains the same as in the 9:3 regime, but occurs for a shorter period. Therefore, there will be no change to the depths and velocities produced in the spawning period. Interpretations are based on data displayed in Appendices Three (e) and (f) (pp 255 - 260)

Tables 7.11 to 7.13 show the fuzzy analysis in terms of the impact of a particular flow regime on all of the species studied. This allows a view of whether a regime is generally beneficial or not. It is evident that under the 9:3 month distribution, the flow regime designed for brown trout is generally not beneficial and is even detrimental in cases. There is one exception, which is the impact that it has on spawning grayling with an improvement in habitat suitability at most sites. Application of the 11:1 month regime designed for brown trout, causes an improvement in the results that were gained from the 9:3 month distribution. The flow regime designed for grayling actually improves habitat quality for brown trout and Atlantic salmon but not for grayling. The 11:1 regime provides a small improvement on the 9:3 month regime. The Atlantic salmon regime appears to be beneficial to grayling but less so to brown trout and Atlantic salmon. All of the flow regimes cause both improvements and deteriorations for all species, regardless of the overall impact. This suggests that there are other controls, at the local scale, which dominate habitat suitability.

	Spawning		Juvenile		
Impact on each species:	Description of habitat created	Comparison to quality of habitat provided by current flow regime	Description of habitat created	Comparison to quality of habitat provided by current flow regime	Effect of 11:1 regime on juveniles
Impact on brown trout	Poor at all sites	No improvement	Poor at most sites, medium at some (2,5,6,10,12)	Deterioration or no change at sites 1-6, 10-16 improvement at sites 7,8,9	Slight improvement, especially at site 6 – the site with the highest HSI.
Impact on grayling	Mostly medium to good spawning habitat, best is at sites 8 and 9. Deterioration of habitat for some sites at high flows (Q5).	Improvement at most sties, deterioration at sites 3, 11 and 14.	Mostly medium. Poor at sites 14, 16.	Little change. Slight improvement at site 5. Deterioration at site 16.	Improvement at sites 3, 8, 11, 16.
Impact on Atlantic salmon	Ranges from poor to good. Sites 2 and 10 are optimum.	Deterioration at sites 2, 3 and 4. Others are unchanged. General deterioration of suitability median and high flows.	Ranges from poor to good. Mostly poor. Site 9 is good.	Deterioration at sites 4, 12 and 13. Improvement at sites 3, 7 and 9. General deterioration of suitability median and high flows.	Improvement at sites 3 and 13. Deterioration at sites 4 and 9.Closely matches HSIs for current regime.

Table 7.11 Impact of the new compensation release regime designed for brown trout, on each of the three species being studied (brown trout, grayling and Atlantic salmon)

	Spawning		Juvenile		
Impact on each species:	**Description of habitat created**	**Comparison to quality of habitat provided by current flow regime**	**Description of habitat created**	**Comparison to quality of habitat provided by current flow regime**	**Effect of 11:1 regime on juveniles**
Impact on brown trout	Poor. Medium at site 5.	No change	Poor to medium. Optimum location is site 6.	Slight improvement at the sites with the most suitable habitat.	Slight improvement at site 6, slight deterioration at sites 7, 8 and 9.
Impact on grayling	Mostly medium to good. Poor at site 3.	Deterioration at the sites with highest HSIs (3 and 8), otherwise little change.	Ranges from poor to excellent, with most sites as poor/ok. Improvement can also be seen at median flows.	Little change except for improvement at site 3.	Improvement at site 8, deterioration at site 3, otherwise no change.
Impact on Atlantic salmon	Ranges from poor to very good. Best sites are 4 and 12. For median flows, higher HSIs occur with the new Grayling flow regime than the current compensation release regime.	Improvement at sites 2, 10, 12 and 15. Deterioration at sites 4, 13.	Ranges from poor to good, mostly medium.	Improvement at site 13, other wise little change. For median flows, higher HSIs occur at current compensation rate.	No impact.

Table 7.12 Impact of the new compensation release regime designed for grayling, on each of the three species being studied (brown trout, grayling and Atlantic salmon)

	Spawning		Juvenile		
Impact on each species:	**Description of habitat created**	**Comparison to quality of habitat provided by current flow regime**	**Description of habitat created**	**Comparison to quality of habitat provided by current flow regime**	**Effect of 11:1 regime on juveniles**
Impact on brown trout	Poor. Medium at site 5.	Slight deterioration at site 3. No change otherwise.	Poor to medium. Optimum locations are sites 10 and 11.	General deterioration except at sites 8 and 9.	Slight improvement at most sites. No change at site 16.
Impact on grayling	Ranges from poor to good. Optimum location is site 8.	Improvement at site 16, deterioration at sites 8 and 16, otherwise little change.	Poor to excellent. Optimum sites are 3 and 8.	Improvement at site 3, deterioration at site 8. Otherwise little difference.	Deterioration at site 3, improvement at site 8, otherwise little change.
Impact on Atlantic salmon	Ranges from poor to excellent. Mostly medium/poor.	Improvement at sites 2, 10, 12, and 15. Deterioration at sites 4 and 13.	Ranges from poor to very good. Mostly medium.	Deterioration at site 4, otherwise little change.	No impact.

Table 7.13 Impact of the new compensation release regime designed for Atlantic salmon, on each of the three species being studied (brown trout, grayling and Atlantic salmon)

7.4.3 Effects of new flow regimes – unsteady annual hydrograph

The results above are based on hypothetical, steady flows. The annual hydrograph assessment will give a more realistic view of the quality of habitat produced by different compensation release regimes.

Brown Trout

The HSIs predicted for brown trout under the new flow regimes, based on an annual hydrograph are displayed in Appendix Three (g) (page 261). None of the new suggested flow regimes appeared to be able to improve the poor habitat suitability offered by the current compensation release regime for spawning brown trout. Because of the high discharges required in order to meet the depth and velocity requirements of brown trout, the compensation from Derwent Reservoir, even when combined with the baseflow input from runoff and tributaries, appears insufficient when the HEC-RAS outputs of depth and velocity are applied to the fuzzy modelling.

One cross-section is marginally more suitable for brown trout than the others – this is section 12. Conditions are slightly more favourable for juvenile brown trout at some sites. Sites 3, 6, 7, 12, and 15 provide medium to good habitat and sites 11 and 16 provided some excellent habitat between November and March. Overall, the most effective new flow regime for juvenile brown trout was the brown trout flow regime, as it provided the greatest amount of high HSI scores for the most sites (although when considering individual sites, the grayling and Atlantic salmon regimes both offered a higher HSI for one site). However, there was very little improvement on the HSI values produced by the current compensation release regime. Although the brown trout regime appears not to benefit brown trout, its impact on the other species should be considered before it is dismissed as ineffective. The brown trout regime offers no improvement to habitat suitability for Atlantic salmon. There is a limited amount of habitat improvement at site 12, for spawning grayling under the brown trout regime, but this is for a short period of the year and to the detriment of habitat quality at a number of other sites.

Grayling

The HSIs predicted for grayling under the new flow regimes, based on an annual hydrograph are displayed in Appendix Three (h) (page 262). The most effective new flow regime for spawning grayling was that designed for Atlantic salmon, providing the greatest amount of good habitat in the spring months. The most appropriate site is site 9. Sites 4 and 8 also provided some good quality habitat in the spawning period. All three regimes offer some improvement on the quality of habitat provided by the current compensation release regime. The best habitat is offered for juvenile grayling also through the Atlantic salmon regime (when applied in the autumn months). However, there is very little difference between the amount of good habitat offered by the Atlantic salmon regime and the amount offered by the grayling regime. Overall, the habitat provided in the River Derwent is more suitable for grayling than for brown trout.

Atlantic Salmon

The HSIs predicted for Atlantic salmon under the new flow regimes, based on an annual hydrograph are displayed in Appendix Three (i) (page 263). There is a great deal of variation in the quality of habitat offered to spawning and juvenile Atlantic salmon across the year and between sites. Some sites offer poor habitat for the whole year and others offer very good to excellent habitat throughout the year. The optimum sites for spawning are 2, 4, 7, 13 and 15 and the highest HSI scores are produced by the grayling compensation release regime. There is little difference between the quality of habitat produced by any of the new flow regimes and the current regime. Contrary to this, the quality of habitat created through the grayling regime for juvenile salmon is far greater than the quality produced by either the brown trout, Atlantic salmon or current regimes. Optimum sites include 2, 4, 7, 13 and 15.

What is noticeable is that for all three species and for most sites, when viewed by site, habitat is either suitable or not, regardless of regime. A site does not switch from suitable to unsuitable or vice versa within the year. Although sites are generally 'suitable' or 'unsuitable' for the whole year, there is annual temporal variation in the degree of 'suitability' or 'unsuitability' within the year. The current study has been unable to assess all of the factors controlling habitat suitability, but they should be considered important in any assessment of habitat suitability.

7.5 Discussion and recommendations

Channel impoundment has had a notable effect on the flow regime and on the ecology of the River Derwent. New compensation release regimes have been designed for a number of fish species. In this Chapter, the newly designed flow regimes were tested in order to assess the impact that they would have on the quality of fish habitat. The modelled flow results have been applied to fish habitat preferences in order to determine the effectiveness of implementing a new compensation release regime. This Section discusses the effectiveness, limitations and applications of the fuzzy rule-based modelling method.

7.5.1 Trends found in habitat suitability predictions

There is a variation in the direction of impact of impoundment on habitat quality. For all species and all flow scales (Q95, Q50 and Q5), there are some cross-sections which improve in habitat quality and some which deteriorate. This is relative to the original suitability of a cross-section. Because suitability is not linearly related to discharge, depth or velocity, it will increase to a certain degree with increase in any of these variables and will then decrease again. The purpose of applying categories of poor and medium habitat quality to both ends of the 'good' category is to account for this non-linear relationship and produce decreases in habitat quality as well as increases as parameters are altered.

7.5.2 Evaluation of the fuzzy model

This model only assessed the impacts of depth and velocity and their combined effect on habitat quality. However, there are other variables which impact upon habitat suitability and on the effect of depth and velocity also (e.g. geometry, Lane *et al.*, 2006). Substrate is an important control on habitat suitability but was not included in this assessment because of time constraints and lack of available data. Substrate is used directly by fish as a form of shelter from fast flows (Hynes, 1970). Therefore, a certain velocity may be more or less suitable depending on the presence of boulders and the stream bed topography. Substrate also affects flow structure within the channel. Lane *et al*., (2006) note that the kind of habitat that is produced by a given bed geometry will change as a function of discharge and stage. It is possible that depth and velocity can be considered as 'surrogates' for some of the other variables that impact habitat quality. For example, faster, deeper flows cause lower temperatures and faster flows may cause lower fine sediment concentrations (Lane *et al*., 2006). Therefore, some of the impacts of these other variables are already represented by different depth and velocity combinations. However, determining which other variables are represented, and to what extent may be a complicated and uncertain process and assumptions of representation must be made with caution. Further to substrate, variables which should also be considered as important controls of habitat quality include water temperature, fine sediment content, dissolved oxygen content and cover (Mouton *et al*., 2007). However, it should be noted that increasing the number of variables used within a single model may make the results more fuzzy, more difficult to interpret (Mouton *et al*., 2008) and lead to greater uncertainty in the actual habitat suitability predicted by the model. Regional adaptation of different species to local conditions should also be considered as one species in certain areas may be more tolerant of certain depths/velocities than the same species in another area, if that variable changes spatially (Mouton *et al*., 2007).

Model resolution

The issue with applying the fuzzy rule methodology at a 1D scale is that it does not well represent the scale of the habitat in which fish (and more so macroinvertebrates) live. Variability in depth and velocity occurs both horizontally and laterally within a cross-section, as well as downstream. These changes and their interactions are important features of habitat suitability but are not represented within a 1D model. Lane *et al*., (2006) note that a 1D approach tends to emphasize variability in section rather than downstream and that, because of the lack of process representation in 1D (even 2D models), a 3D approach may be more appropriate (although this is not without its limitations – taking resolution to a scale that is even finer than the habitat in which the organisms live). Organisms will occupy different areas of habitat depending on what they are doing (e.g. feeding in fast flows, resting in slower flows, or seeking refuge behind boulders or in pools) (Lane *et al*., 2006). This lends support to a 3D approach which is able to represent all of these processes within a model.

Representativeness of study areas

A major assumption in this study is that the cross-sections studied are representative of the habitat in that area of the channel. Because one of the criteria used to select study sites was accessibility, this inherently implies that disturbance of the habitat at that location is possible. Because fish observations were not taken for this study, it cannot be concluded that the sites identified are actually used by each of the fish species (although it is known that brown trout and grayling inhabit the river, there is no information on which parts they prefer). The results may be used hypothetically to suggest expected usable areas, based on predicted suitability. A key component in any modelling study is validation (Jorde *et al*., 2001). Although in fuzzy modelling this is not always carried out (Van Broekhoven *et al*., 2006), comparison of predicted suitable habitat areas to observed fish habitat use would be an effective method of testing the accuracy of the model and more weight could be given to results and suggestions, had the model been validated.

7.5.3 Implications of fuzzy model results for fish species

Effects of impoundment

Channel impoundment appears to have provided improved habitat quality at low flows for all three species, by elevating the low flow value of Q95. The same has occurred at high flows (Q5), through the lowering of the highest flows. This is contradictory to some of the theories presented in 4.2.4 (e.g. those of Hynes, 1970; Jowett, 1990; Minshall, 1988) that fish require extremes of flow for different life stages. However, it must be considered that the variability in flows is required with high and low flow *events* whereas the habitat suitability index is based on the assumption of that combination of depth and velocity occurring all of the time. Furthermore, the variability in flow (particularly high flow) is required for aspects such as the flushing of fine sediment, a process which does not *provide* 'habitat' for the fish, but conditions the habitat already available over a longer timescale. The fuzzy modelling used in this assessment is not sophisticated enough to account for the effects of a certain depth/velocity combination on different timescales.

Effects of new flow regimes – steady state low, median and high flows

When assessing the impact of the new proposed flow regimes based on steady low, median and high flows, different regimes appear to benefit different species, as illustrated by the results in Tables 7.11 to 7.13. It is expected that this is a result of the different requirements of each species in terms of magnitude of depth and velocity and in terms of time of year that different flow rates are required. Table 7.14 summarises the optimum regimes for each species.

	Brown trout		Grayling		Atlantic salmon	
	Spawning	**Juvenile**	**Spawning**	**Juvenile**	**Spawning**	**Juvenile**
Optimum new flow regime	None – no improvement on the already poor habitat.	Brown trout (11:1 month distribution) regime.	Brown trout (9:3) (but grayling regime is also good).	Brown trout 11:1 regime.	Atlantic salmon and grayling regimes have similar impacts.	Brown trout 11:1, but none are particularly effective.

Table 7.14 Optimum type of new flow regime for each species, based on steady low, median and high flows

The brown trout 11:1 month flow regime proved to be the most effective in improving habitat quality for juvenile fish of all species. It is likely that this is because the brown trout depth and velocity requirements demand the highest elevated flow while the 11:1 month regime ensures that flows for the rest of the year are not significantly reduced. In fact, according to Table 6.10, the juvenile depth and velocity requirements for juveniles are mostly higher than those for spawning fish, meaning that the extra flow offered outside of the spawning period by the 11:1 month regime is essential. Currently, CRAB can only accommodate the redistribution of flows on the 9:3 month scale. The results presented in Tables 7.8 to 7.13 suggest that the ability to control the distribution of high and low releases throughout the year would be beneficial in attaining an optimum flow regime. For spawning fish, the response to the different flow regimes was more varied. The fuzzy rules used in the model require higher depth and velocity values than those input to the CRAB model (because of contradictions in case studies). Therefore, the compensation release rate for spawning brown trout produces much lower depths and velocities than would be required to provide good quality habitat to spawning brown trout, when applied to the fuzzy model.

Effects of new flow regimes – annual hydrograph

None of the new suggested release regimes offer benefits to spawning brown trout. In fact, the current compensation release regimes provide marginally more suitable habitat than any of the new regimes. For juvenile brown trout, the new compensation release regime based on the brown trout requirements was the most effective at providing good habitat across a number of sites. The grayling and Atlantic salmon release regimes offered very good habitat in concentrated areas, although what they achieved in these areas was no improvement on the HSIs produced by the current compensation release regime. Therefore, based on the annual hydrograph data, it is suggested that, if a change is to be made, the brown trout compensation release regime would be the best choice for juvenile brown trout. It would make more of the river accessible to brown trout for longer periods of time, compared to the current situation, but this would be to the detriment of the very good quality of habitat currently being achieved at sites 11 and 16.

The optimum new flow regime for spawning grayling (i.e. spring) appeared to be the brown trout compensation release regime. This provided the highest HSI values between February and April, although there was very little change from the HSIs produced under the current regime, suggesting that a change may cause an unnecessary change to the flow regime of the river. For juvenile grayling, the compensation release regime based on grayling requirements produced the greatest amount of good to very good quality habitat. Again, the improvement on current conditions was negligible.

The Atlantic salmon regime was the optimum for spawning Atlantic salmon, of all of the new regimes considered. For juvenile Atlantic salmon, the Atlantic salmon regime was best in terms of good quality habitat, but provided very little change to the HSI scores produced under the current release regime. Therefore, a change in compensation release regime would have little benefit for juveniles.

The dominance of some sites over others as 'suitable habitats' suggests that the channel morphology may be a more significant factor in determining habitat suitability than flow rate. This may be a result of sediment size composition or the control that the shape of the cross-section has on depth and velocity. The location of suitable sites may also be largely controlled by factors external to the channel such as level of cover or oxygenation of water. The temporal variability in degree of habitat suitability indicates that the natural fluctuations in flow within the channel (a result of inputs from unregulated tributaries and surface runoff) are a more dominant control on the depths and velocities experienced than the release rates from the reservoir.

Considering the improvements made by the new compensation release regimes, the virtues of changing the compensation release regime seem to be few. Each of the species is mostly benefited by the release regime based on its own requirements, but to a negligible degree. Therefore, the question of whether a change in compensation release regime can be justified is raised. Changing the compensation release regime would require time, money and the monitoring efforts of the Environment Agency in order to ensure no detrimental changes to ecology were caused. Furthermore, as Chapter Four illustrated, the current flow regime appears to be more suitable for macroinvertebrate populations than the pre-impoundment regime. Changing the compensation release regime may have untold effects on both macroinvertebrate populations themselves and a knock-on effect on fish as macroinvertebrates are an important food source for fish.

One point that should be noted is that of the accuracy of the HEC-RAS outputs. As discussed in Chapter Six, the hydrodynamic model has not been fully verified and underestimates the discharge at the output boundary. This means that the depths and velocities are also likely to have been underestimated, with increasing error with distance downstream. Consequently, the predictions for the upstream reaches may be more reliable than those in the downstream section. A method of addressing this problem would be to calculate the percent error between the predicted and observed output hydrographs from HEC-RAS and amending the depth and velocity outputs accordingly. However, it would be imprudent to assume that the changes in depth and velocity, with increase in discharge, would be linear. As discussed in Section 7.5.3, the interaction of depth and velocity varies with discharge and the bed layer and banks would also have reduced impact on the flow with increasing discharge. It is possible, then, that such an approach would introduce further error to the model and would be unrepresentative of the processes taking place.

7.6 Conclusion

A fuzzy rule-based method was chosen to assess impact of flow regime on habitat quality because it allows the user to apply ambiguous knowledge to a model, a range of habitat combinations can be considered, it considers the combined effects of depth and velocity, it lends itself to the extension to stakeholder and manager application and it can be transferred to rivers for which there are no habitat preference data. Its application has shown that there are a range of depth and velocity values and combinations that may be considered as suitable habitat for the three fish species investigated. Each combination of habitat conditions has a varying degree of suitability and this is possibly more strongly controlled by geomorphology and cross-sectional shape than subtle changes in flow depth and velocity. Changing the compensation release regime will have varied effects on the suitability of habitat for each fish species with some periods of the year becoming more suitable and some less, depending on the pre-existing conditions. The three different flow regimes produce slightly different effects, based on the temporal pattern and the magnitude of the change. The overall benefit of changing the compensation release regime from Derwent Reservoir remains questionable. Chapter Eight will draw together the findings of the whole project and place the findings of the current Chapter in context.

Chapter Eight

Conclusions

8.1 Impact of regulation regimes on the ecohydraulic characteristics of the River Derwent

This thesis set out to assess the impacts of a current regulation regime and the predicted impacts of newly designed regulation regimes on habitat suitability for fish and macroinvertebrates in the River Derwent, Northumberland, with a focus on the effects of scale and long-term trends in such studies. The objectives were achieved through a number of statistical analysis and modelling processes including CRAB (a descriptive tool for the redesign of compensation release regimes) and CRAM (which consists of hydrodynamic and fuzzy components). The hydrodynamic model (HEC-RAS) was used to assess the impact of various compensation release regimes on hydrological characteristics such as water depth and velocity and the fuzzy rule-based modelling was used to assess the suitability of the predicted depth and velocity combinations for the provision of habitat in the River Derwent. The findings specific to the River Derwent are summarised below:

- Decrease in variability of flow and frequency of low and high flows since impoundment. This is supported by (un-validated) HEC-RAS results
- Increase in richness and diversity of macroinvertebrate populations since impoundment
- Brown trout and grayling populate the River Derwent and have done so for many decades
- New compensation release regimes, based on species requirements can be designed for the reservoir, as illustrated in Figure 8.1
- Low flows are most significantly affected by changes to flow regime
- The new compensation release regime designed for brown trout caused the biggest change in flows during the spawning period (September to November)
- Reducing the time of the spawning flow to one month had a limited impact on depths and velocities.
- The brown trout and Atlantic salmon regimes had a more profound impact on flow magnitude than the grayling regime
- The spate flow caused an intense, short increase in depths and velocities
- Analysis of a number of depth and velocity combinations using fuzzy rule-based modelling suggested that the new flow regimes had limited impact on habitat suitability
- Geomorphology is thought to have a large impact on habitat suitability
- If changes were to be made to the Derwent compensation release regime, the optimum regimes are as outlined in Table 8.1

- Changes to current flow regimes may have untold impacts upon macroinvertebrate and fish populations and should not be conducted without a more thorough investigation into potential impacts

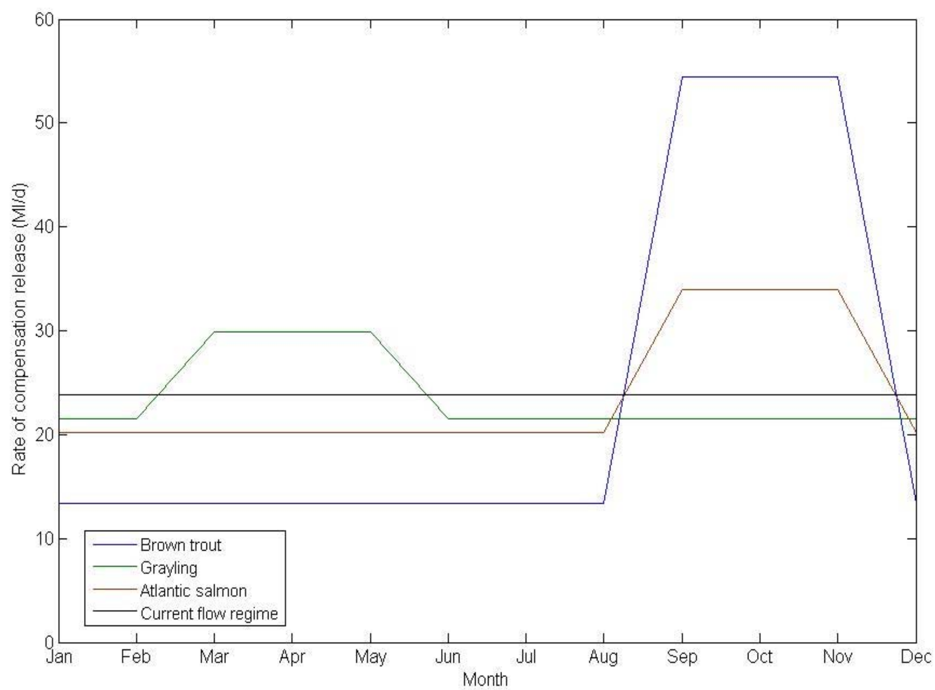

Figure 8.1 New suggested flow regimes based on habitat requirements of three fish species (brown trout, grayling, Atlantic salmon)

	Brown trout		**Grayling**		**Atlantic salmon**	
	Spawning	**Juvenile**	**Spawning**	**Juvenile**	**Spawning**	**Juvenile**
Optimum new flow regime	No improvement on the already poor habitat.	Brown trout (11:1 month distribution) regime.	Brown trout (9:3) (but grayling regime is also good).	Brown trout 11:1 regime.	Atlantic salmon and grayling regimes have similar impacts.	Brown trout 11:1, but none are particularly effective.

Table 8.1 Most appropriate flow regime for brown trout, grayling and Atlantic salmon, based on the depth and velocity requirements of each species. The 9:3 and 11:1 ratios refer the proportional distribution of flow in the designed flow regime. Newly designed regimes were first set to deliver a spawning flow for three months of the year and a flow suitable for juveniles during the other nine months. These distributions were then shifted to deliver spawning flow for only one months of the year and juvenile suitable flow for the other 11

8.2 Core findings

There are a number of core findings which can be drawn from this investigation:

1. In accordance with literature concerning flow regulation and ecology (e.g. Armitage, 1988; Maddock *et al*., 2001), impoundment has been found in the long-term (in this case a period of over fifty years) to cause a change in macroinvertebrate populations. As suggested by the disturbance hypotheses of Connell (1978) and Huston (1994), it appears that the long-term stabilising impacts of impoundment on a naturally flashy river may cause an increase in macroinvertebrate richness and diversity.

2. Long-term impacts on hydrology have also been identified and they include the dampening of the extremes of the long-term hydrograph and reduced variability of flow, as previously reported by a number of authors: Petts and Pratts, (1983); Ibañez *et al.,* (1995); Isik *et al.,* 2008.

3. The investigation highlighted the need to develop an optimised compensation release regime which accommodates the flow requirements of a number of species. As suggested by a number of authors (e.g. Heggenes and Saltveit, 1990; Summers *et al.,* 1996*;* Armstrong *et al.,* 2003), the magnitude and timing of flow depth and velocity requirements vary between species (e.g. grayling require a spawning flow in the spring). Therefore, careful assessment of the impacts of any new flow regime must be conducted for all of the key species that may be affected. This can be done through fuzzy model analysis by applying different flow regimes to the habitat requirements of a number of species and determining the regime which provides the greatest amount of good habitat for the greatest number of species.

4. In accordance with the findings of Lane *et al.,* (2006) and Mould *et al.,* (2007), the investigation has shown that, if the impacts of a new compensation release regime can be modelled accurately, then the suitability of combined habitat controls (e.g. depth and velocity) can be assessed through the application of fuzzy rule methodology. In contrast to the work of Lane *et al.,* (2006) and Mould *et al*., (2007), this study has demonstrated the application of fuzzy modelling at a river/catchment scale, showing that an assessment of spatial variation in habitat quality over distances of kilometres is possible, rather than a 2D assessment which provides detailed information on the habitat suitability of micro-topography at the reach scale. Information on habitat quality at these different scales will provide information on habitat suitability for different fish behaviours at different scales (e.g. migration at the river scale and choice of refugia at the reach scale). The two approaches could be combined to gain a comprehensive assessment of habitat for the whole river and for different behavioural features of fish.

8.3 Recommendations for future work and applications

This thesis has provided a number of valuable conclusions related to the application of studies concerning the impacts of compensation release regimes on habitat. First, it is suggested that regulated flows should not be unreservedly considered as negative controls on a river and its fauna. Second, when planning the implementation of such an investigation on other catchments, there are number of factors that should be considered. These are based around the suitability of the catchment for analysis:

- If the catchment has pre-impoundment data for ecology and/or hydrology, this should be analysed carefully to determine whether there has been an impact and, if there has, whether it needs to be mitigated.
- Hydrology and ecology data for a similar but unregulated catchment would enable the user to place the study river's state in context of other similar catchments and also to identify the possible effects of impoundment.
- There should be hydrological, ecological (preferably fish and macroinvertebrate) and morphological data available for the assessment. It will be needed for every step of the modelling process including calibration and validation. Furthermore, it will be necessary to monitor the impacts of any changes made as a result of the investigation, to ensure that there are no detrimental impacts of the changes.

Third, if the process were to be applied to another catchment, the steps followed in this case appear to be an effective and efficient route to the redesign of new compensation release regimes. Finally, change should not be implemented for change's sake and should be applied only when there is a definite need for improvement within the catchment and when the changes will not adversely affect the wider community (e.g. water use for other purposes such as drinking water supply). The implications of the project should be assessed strategically and with consideration for the whole catchment.

Chapter Eight has highlighted the findings of this study and offered suggestions for the application of the emerging methodology. It has been shown that the impact of impoundment on the River Derwent has had an impact on flows by dampening the extreme aspects of the hydrograph and on ecology with an increase in macroinvertebrate richness and diversity. The alteration of the compensation release regime (within the limits of the Derwent Water Order, 1957), appears to have a minimal impact on depth and velocity, particularly at any flow greater than baseflow. The newly designed compensation release regimes provide limited benefits for the fish of the river and therefore it has been concluded that no changes should be made to the compensation release regime unless there is deterioration of the ecology. The methodology is appropriate for application to other catchments where suitable data exist and may be a useful tool in the ever increasing task of sustainably balancing the needs of all water users.

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APPENDIX ONE - Revised BMWP scores used in calculation of BMWP and ASPT scores for macroinvertebrate analysis in Chapter 4. Source: http://www.cies.staffs.ac.uk/bmwptabl.htm 9/11/2009)

Common Name	Family	Original BMWP Score	Revised BMWP Score	Habitat Specific Scores		
				Riffles	Riffle/Pools	Pools
Flatworms	Planariidae	5	4.2	4.5	4.1	3.7
	Dendrocoelidae	5	3.1	2.3	4.1	3.1
Snails	Neritidae	6	7.5	6.7	8.1	9.3
	Viviparidae	6	6.3	2.1	4.7	7.1
	Valvatidae	3	2.8	2.5	2.5	3.2
	Hydrobiidae	3	3.9	4.1	3.9	3.7
	Lymnaeidae	3	3.0	3.2	3.1	2.8
	Physidae	3	1.8	0.9	1.5	2.8
	Planorbidae	3	2.9	2.6	2.9	3.1
Limpets and Mussels	Ancylidae	6	5.6	5.5	5.5	6.2
	Unionidae	6	5.2	4.7	4.8	5.5
	Sphaeriidae	3	3.6	3.7	3.7	3.4
Worms	Oligochaeta	1	3.5	3.9	3.2	2.5
Leeches	Piscicolidae	4	5.0	4.5	5.4	5.2
	Glossiphoniidae	3	3.1	3.0	3.3	2.9
	Hirudididae	3	0.0	0.3	-0.3	
	Erpobdellidae	3	2.8	2.8	2.8	2.6
Crustaceans	Asellidae	3	2.1	1.5	2.4	2.7
	Corophiidae	6	6.1	5.4	5.1	6.5
	Gammaridae	6	4.5	4.7	4.3	4.3
	Astacidae	8	9.0	8.8	9.0	11.2
Mayflies	Siphlonuridae	10	11.0	11.0		
	Baetidae	4	5.3	5.5	4.8	5.1
	Heptageniidae	10	9.8	9.7	10.7	13.0
	Leptophlebiidae	10	8.9	8.7	8.9	9.9
	Ephemerellidae	10	7.7	7.6	8.1	9.3
	Potamanthidae	10	7.6	7.6		
	Ephemeridae	10	9.3	9.0	9.2	11.0
	Caenidae	7	7.1	7.2	7.3	6.4

Stoneflies	Taeniopterygidae	10	10.8	10.7	12.1	
	Nemouridae	7	9.1	9.2	8.5	8.8
	Leuctridae	10	9.9	9.8	10.4	11.2
	Capniidae	10	10.0	10.1		
	Perlodidae	10	10.7	10.8	10.7	10.9
	Perlidae	10	12.5	12.5	12.2	
	Chloroperlidae	10	12.4	12.5	12.1	
Damselflies	Platycnemidae	6	5.1	3.6	5.4	5.7
	Coenagriidae	6	3.5	2.6	3.3	3.8
	Lestidae	8	5.4			5.4
	Calopterygidae	8	6.4	6.0	6.1	7.6
Dragonflies	Gomphidae	8				
	Cordulegasteridae	8	8.6	9.5	6.5	7.6
	Aeshnidae	8	6.1	7.0	6.9	5.7
	Corduliidae	8				
	Libellulidae	8	5.0			5.0
Bugs	Mesoveliidae *	5	4.7	4.9	4.0	5.1
	Hydrometridae	5	5.3	5.0	6.2	4.9
	Gerridae	5	4.7	4.5	5.0	4.7
	Nepidae	5	4.3	4.1	4.2	4.5
	Naucoridae	5	4.3			4.3
	Aphelocheiridae	10	8.9	8.4	9.5	11.7
	Notonectidae	5	3.8	1.8	3.4	4.4
	Pleidae	5	3.9			3.9
	Corixidae	5	3.7	3.6	3.5	3.9
Beetles	Haliplidae	5	4.0	3.7	4.2	4.3
	Hygrobiidae	5	2.6	5.6	-0.8	2.6
	Dytiscidae	5	4.8	5.2	4.3	4.2
	Gyrinidae	5	7.8	8.1	7.4	6.8
	Hydrophilidae	5	5.1	5.5	4.5	3.9
	Clambidae	5				
	Scirtidae	5	6.5	6.9	6.2	5.8
	Dryopidae	5	6.5	6.5		
	Elmidae	5	6.4	6.5	6.1	6.5

	Chrysomelidae *	5	4.2	4.9	1.1	4.1
	Curculionidae *	5	4.0	4.7	3.1	2.9
Alderflies	Sialidae	4	4.5	4.7	4.7	4.3
Caddisflies	Rhyacophilidae	7	8.3	8.2	8.6	9.6
	Philopotamidae	8	10.6	10.7	9.8	
	Polycentropidae	7	8.6	8.6	8.4	8.7
	Psychomyiidae	8	6.9	6.4	7.4	8.0
	Hydropsychidae	5	6.6	6.6	6.5	7.2
	Hydroptilidae	6	6.7	6.7	6.8	6.5
	Phryganeidae	10	7.0	6.6	5.4	8.0
	Limnephilidae	7	6.9	7.1	6.5	6.6
	Molannidae	10	8.9	7.8	8.1	10.0
	Beraeidae	10	9.0	8.3	7.8	10.0
	Odontoceridae	10	10.9	10.8	11.4	11.7
	Leptoceridae	10	7.8	7.8	7.7	8.1
	Goeridae	10	9.9	9.8	9.6	12.4
	Lepidostomatidae	10	10.4	10.3	10.7	11.6
	Brachycentridae	10	9.4	9.3	9.7	11.0
	Sericostomatidae	10	9.2	9.1	9.3	10.3
True flies	Tipulidae	5	5.5	5.6	5.0	5.1
	Chironomidae	2	3.7	4.1	3.4	2.8
	Simuliidae	5	5.8	5.9	5.1	5.5
Notes	* These families are now excluded from the list used for the calculation of the score. A blank indicates that there were insufficient records for the calculations. The Revised BMWP Scores are based on the analysis of frequency of occurrence of the families recorded in approximately 17,000 samples. The Habitat Specific Scores are based on the following substrate compositions: Riffles: >= 70% boulders and pebbles Pool: >= 70% sand and silt Riffle/Pool: the remainder					

APPENDIX TWO - Flow LIFE Scores assigned to each family for macroinvertebrate analysis in Chapter 4. Source: Extence *et al*., (1999)

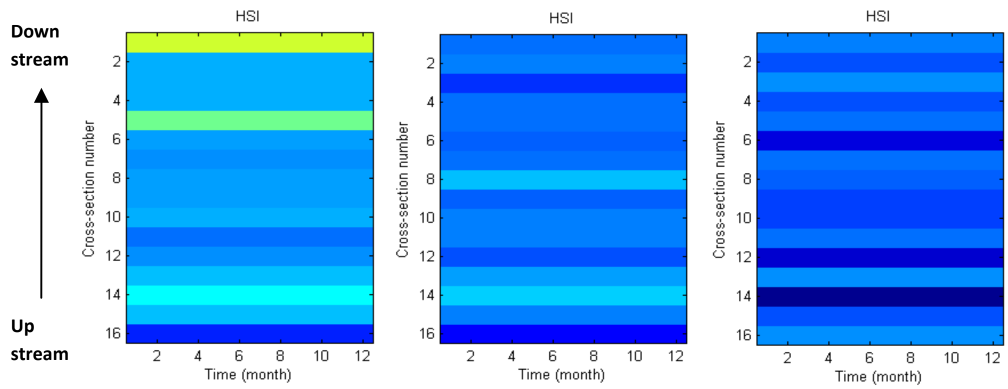
Tricladida					
Planariidae[a]	IV[b]	Dugesiidae[a]	IV	Dendrocoelidae	IV
Gastropoda					
Neritidae	II	Viviparidae	III	Valvatidae	IV
Hydrobiidae[a]	IV[b]	Bithyniidae[a]	IV	Lymnaeidae	IV[b]
Physidae	IV[b]	Planorbidae	IV	Ancylidae[a]	II
Acroloxidae[a]	IV				
Bivalvia					
Margaritiferidae	II	Unionidae	IV[b]	Sphaeriidae	IV[b]
Dreissenidae	IV				
Hirudinea					
Piscicolidae	II	Glossiphoniidae	IV	Hirudidae	IV
Erpobdellidae	IV				
Araneae					
Agelinidae	V				
Anostraca					
Chirocephalidae	VI				
Notostraca					
Triopsidae	VI				
Malacostraca					
Mysidae	V	Asellidae	IV	Corophidae	III
Gammaridae[a]	II	Crangonycitidae[a]	IV	Talitridae	VI
Astacidae	II				
Ephemeroptera					
Siphlonuridae	IV[b]	Baetidae	II[b]	Heptageniidae	I[b]
Leptophlebiidae	II[b]	Ephemerellidae	II	Potamanthidae	III
Ephemeridae	II[b]	Caenidae	IV[b]		
Plecoptera					
Taeniopterigidae	II[b]	Nemouridae	IV[b]	Leuctridae	II[b]
Capniidae	I[b]	Perlodidae	I	Perlidae	I
Chloroperlidae	I				
Odonata					
Platycnemididae	IV	Coenagriidae	IV	Lestidae	IV
Agriidae	III[b]	Gomphidae	II	Cordulegasteridae	II
Aeshnidae	IV	Corduliidae	IV[b]	Libellulidae	IV[b]
Hemiptera					
Mesovelidae	V	Hebridae	IV[b]	Hydrometridae	IV
Veliidae	IV[b]	Gerridae	IV	Nepidae	V
Naucoridae	IV	Aphelocheiridae	II	Notonectidae	IV
Pleidae	IV	Corixidae	IV		
Coleoptera					
Haliplidae	IV[b]	Hygrobiidae	V	Noteridae	IV[b]
Dytiscidae	IV[b]	Gyrinidae	IV[b]	Hydrophilidae	IV[b]
Hydraenidae	IV[b]	Scirtidae	IV[b]	Elmidae	II[b]
Megaloptera					
Sialidae	IV[b]				
Neuroptera					
Osmylidae	II	Sisyridae	IV[b]		
Trichoptera					
Rhyacophilidae[a]	I	Glossosomatidae[a]	II[b]	Philopotamidae	I
Polycentropodidae	IV[b]	Psychomyiidae[a]	II[b]	Ecnomidae[a]	III
Hydropsychidae	II	Hydroptilidae	IV[b]	Phryganeidae	IV
Limnephilidae	IV[b]	Molannidae	IV	Beraeidae	II
Odontoceridae	I	Leptoceridae	IV[b]	Goeridae	I
Lepidostomatidae	II	Brachycentridae	II	Sericostomatidae	II
Diptera					
Tipulidae	IV[b]	Ptychopteridae	II	Chaoboridae	V
Culicidae	V	Simuliidae	II	Syrphidae	V

[a] Historical data may include combination of both families, or separate families (use first family of pair in cases where both family names used, e.g. Gammaridae/Crangonycitidae = II).
[b] Families containing species/genera with variable flow requirements.

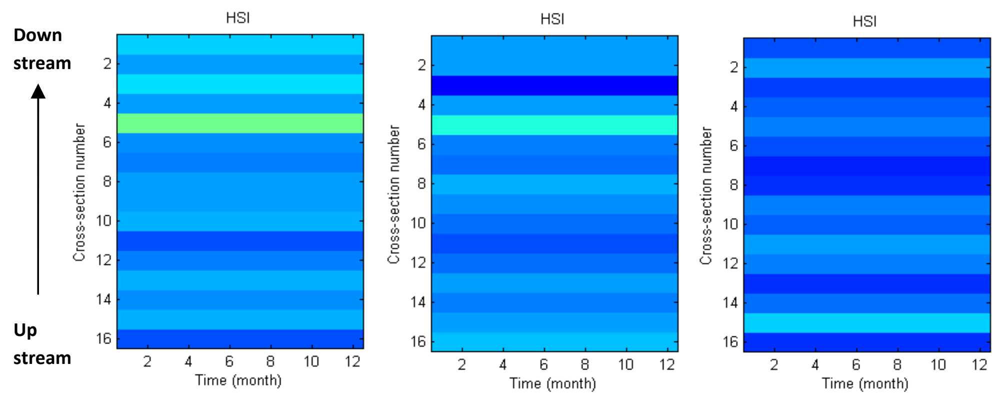
APPENDIX THREE (a) Fuzzy rule-based modelling results – Effect on spawning brown trout

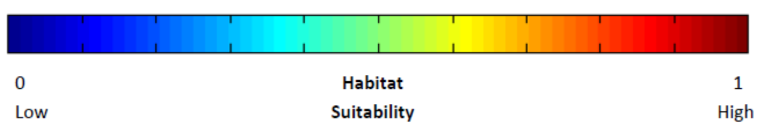
	Q95	Q50	Q5

Pre-impoundment flow regime

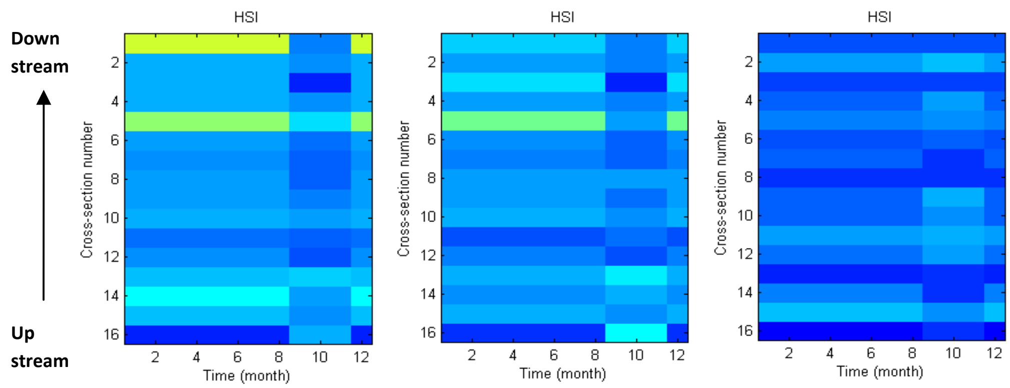


Post-impoundment flow regime

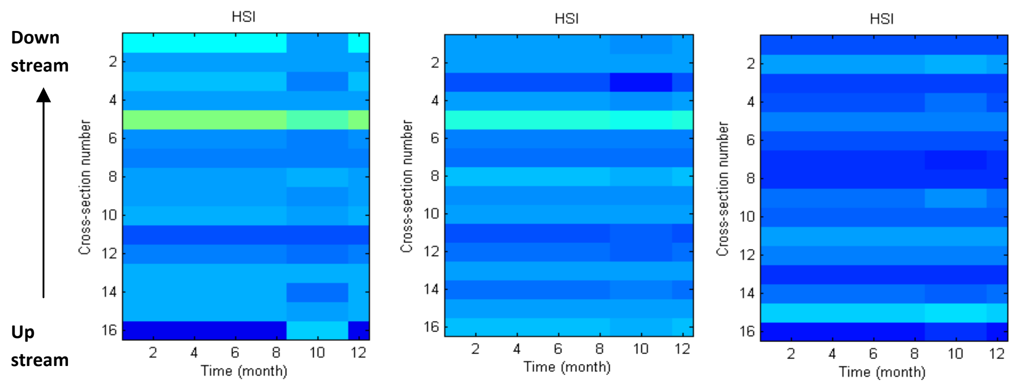




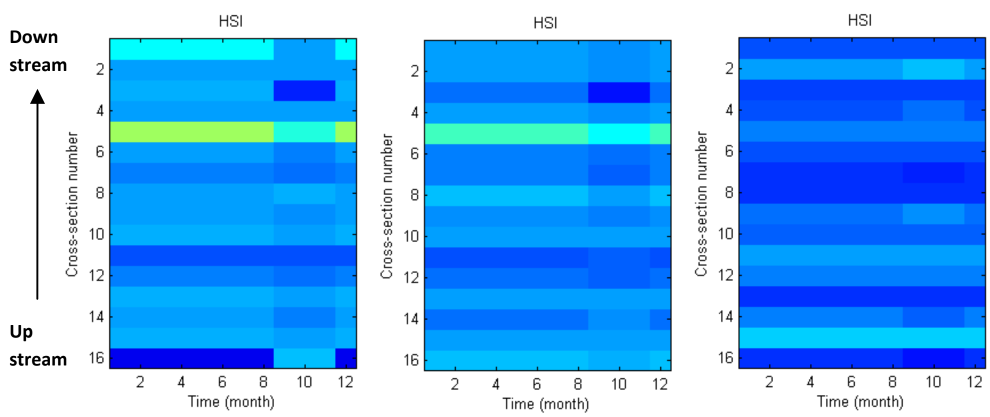
Brown trout 9:3 month flow regime

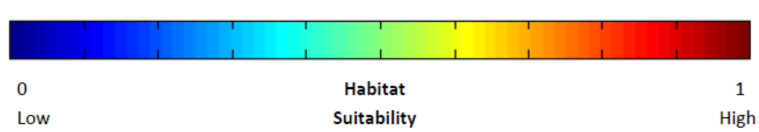


Grayling 9:3 month flow regime

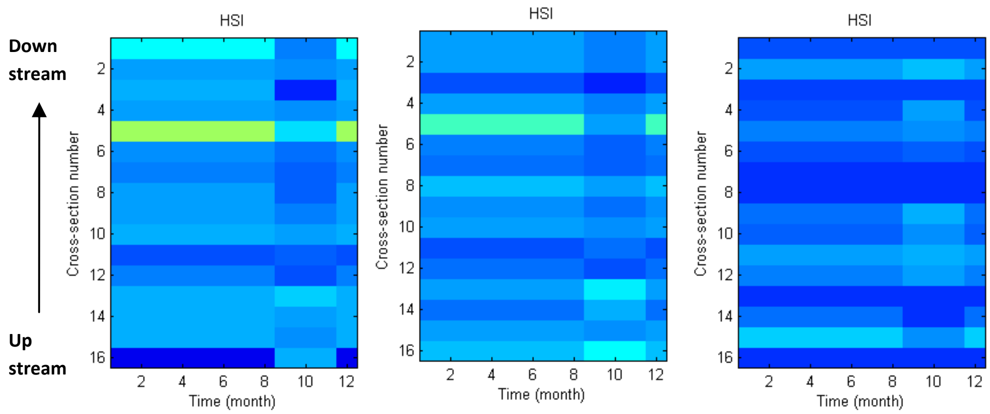


Atlantic salmon 9:3 month flow regime

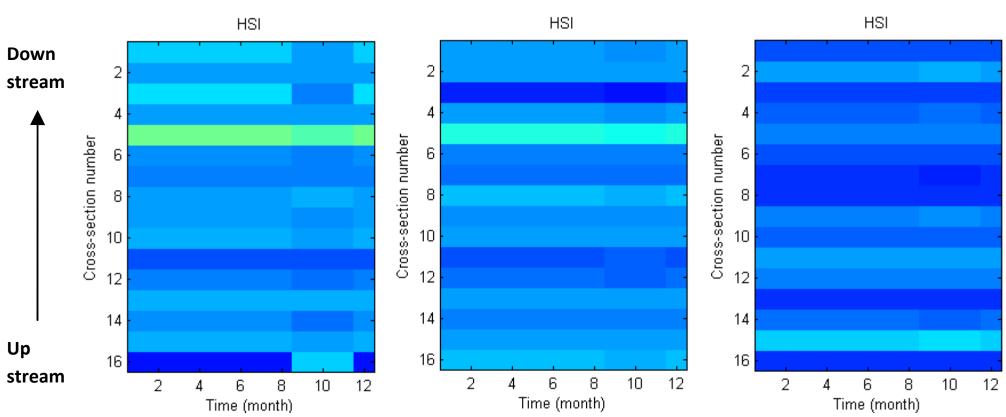




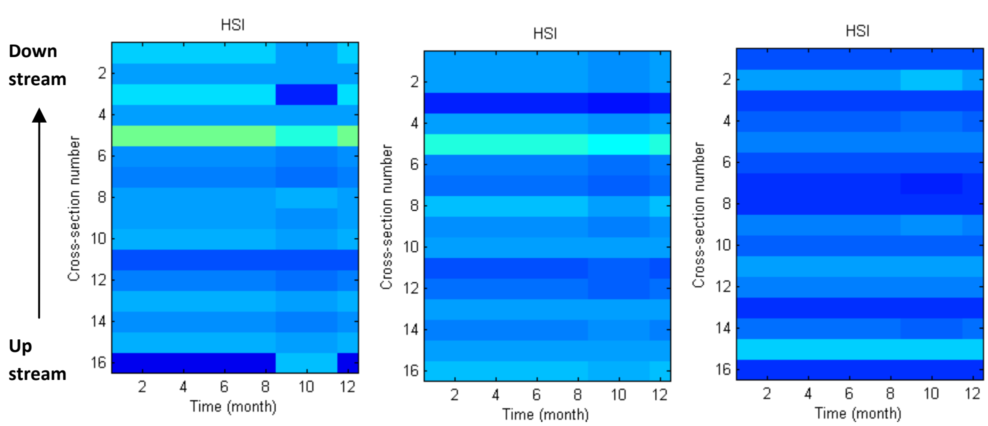
Brown trout 11:1 month flow regime

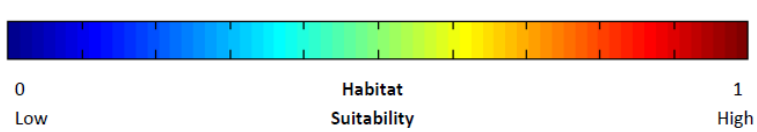


Grayling 11:1 month flow regime



Atlantic salmon 11:1 month flow regime

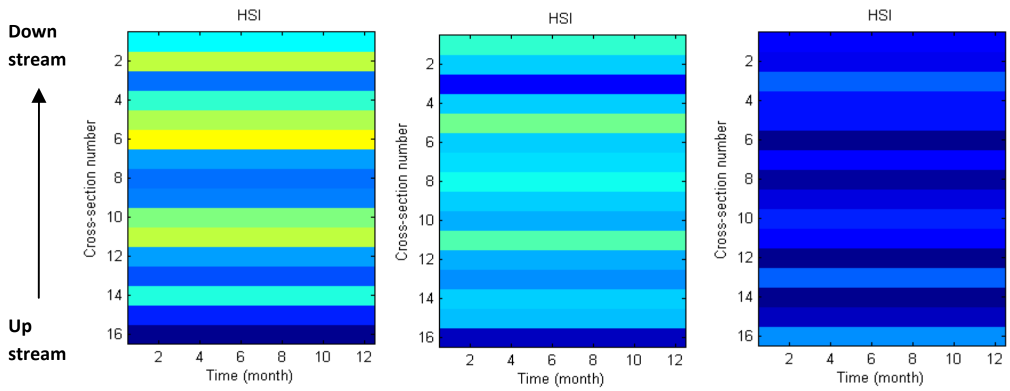




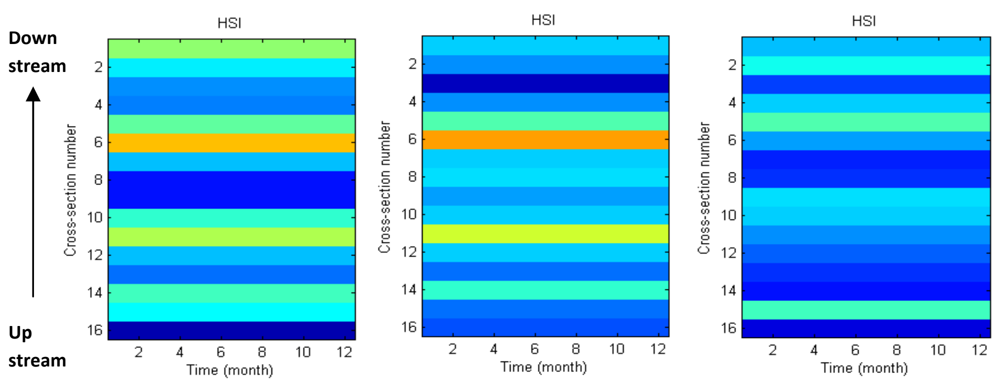
APPENDIX THREE (b) Fuzzy rule-based modelling results – Effect on juvenile brown trout

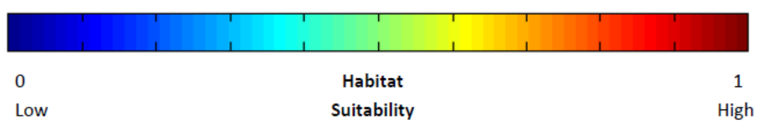
Q95 **Q50** **Q5**

Pre-impoundment flow regime

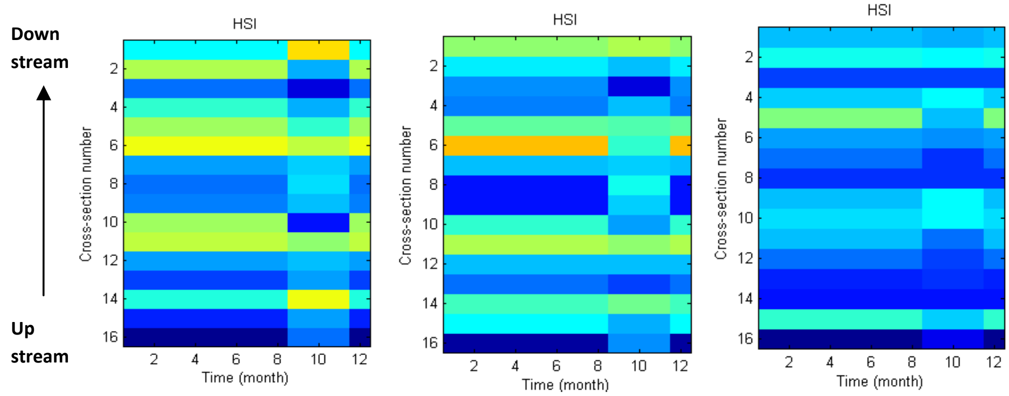


Post-impoundment flow regime

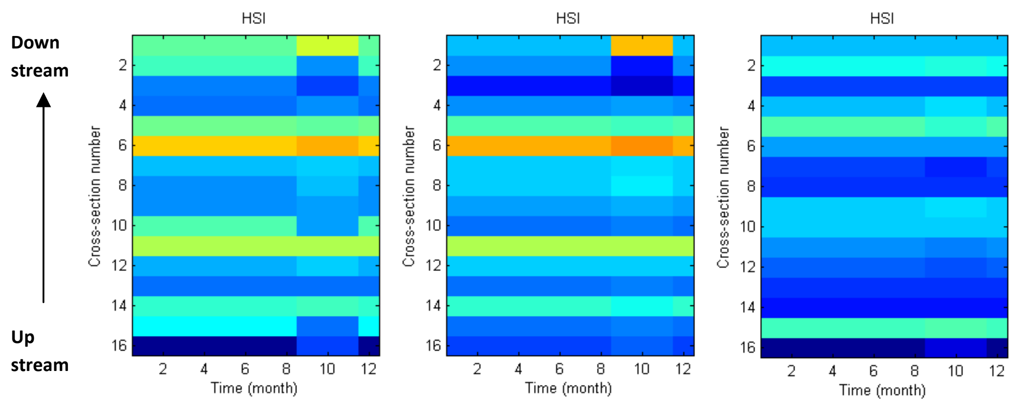




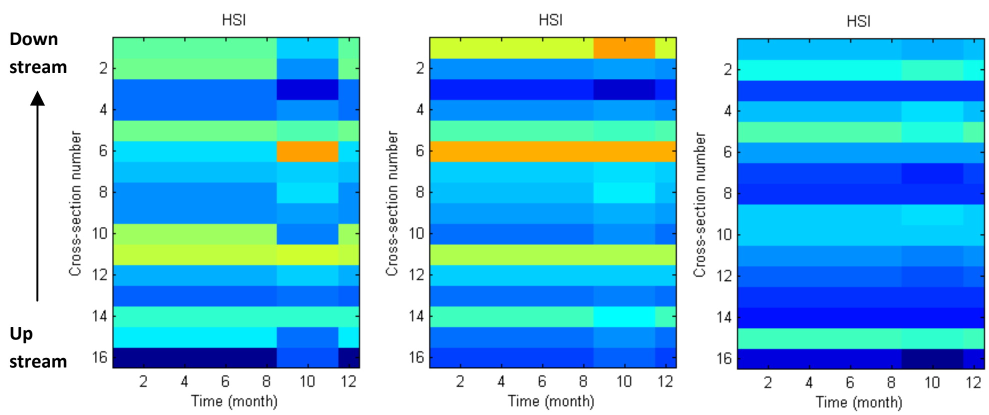
Brown trout 9:3 month flow regime

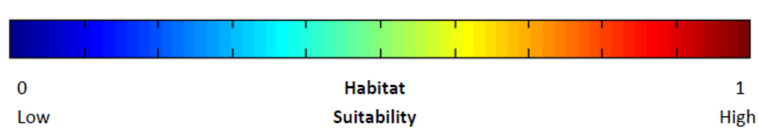


Grayling 9:3 month flow regime

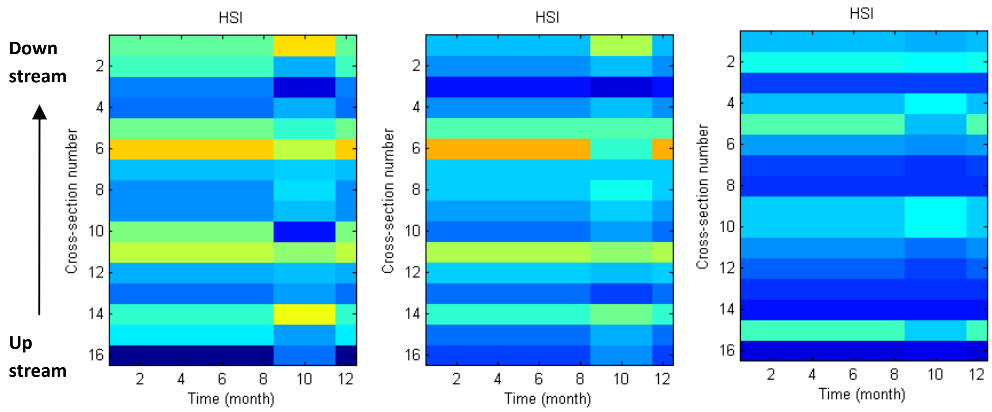


Atlantic salmon 9:3 month flow regime

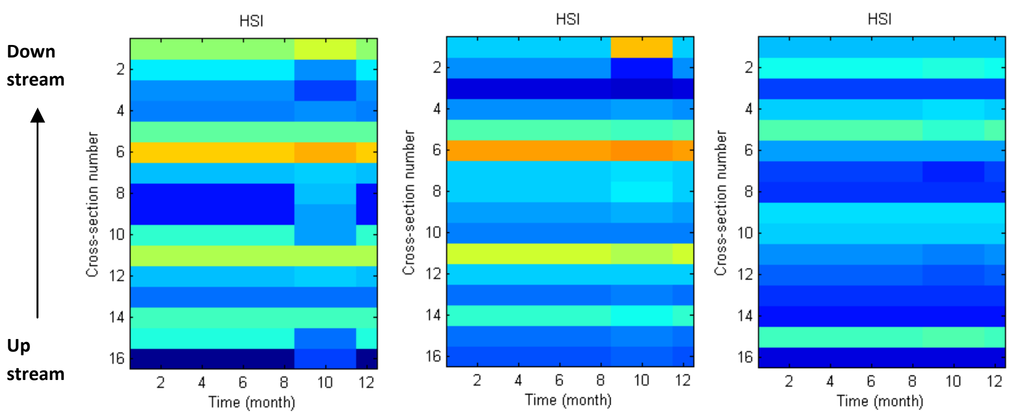




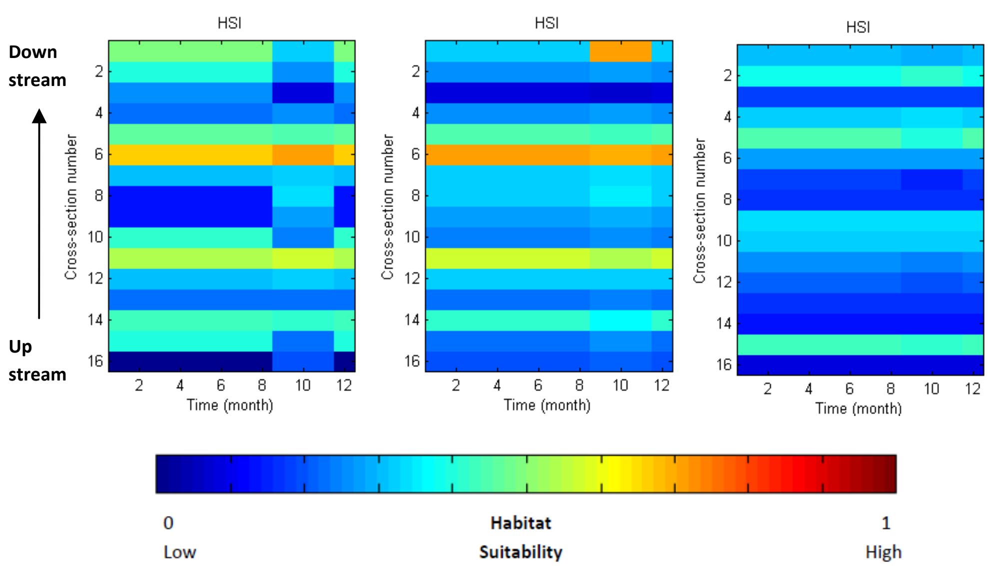
Brown trout 11:1 month flow regime



Grayling 11:1 month flow regime



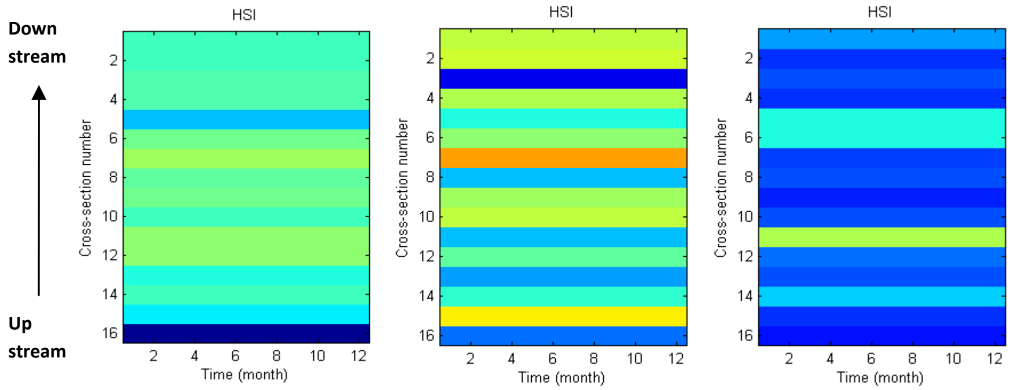
Atlantic salmon 11:1 month flow regime



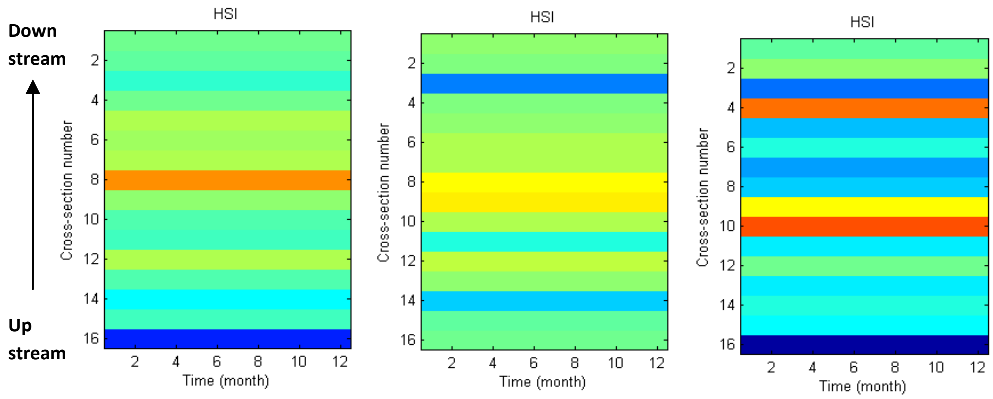
APPENDIX THREE (c) Fuzzy rule-based modelling results – Effect on spawning grayling

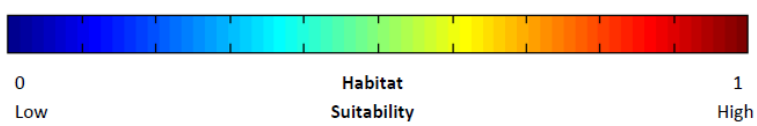
	Q95	Q50	Q5

Pre-impoundment flow regime

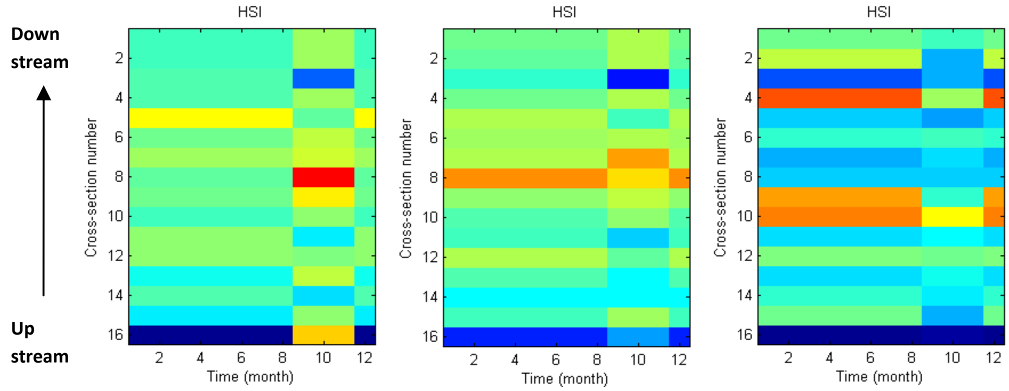


Post-impoundment flow regime

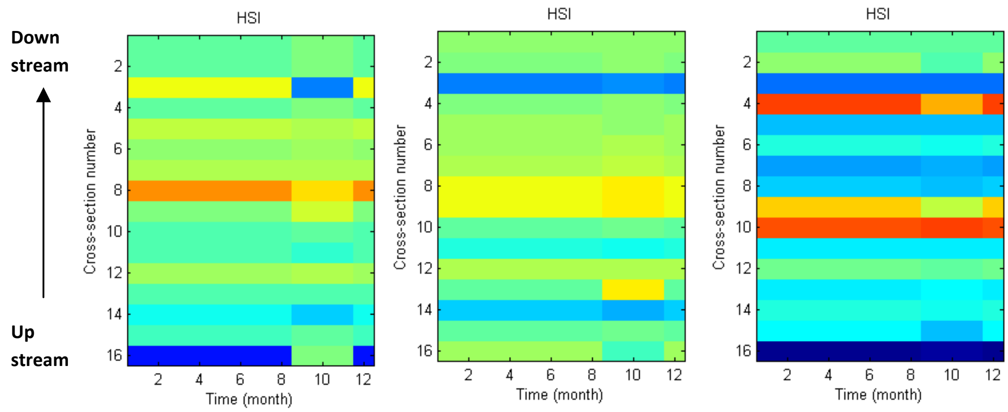




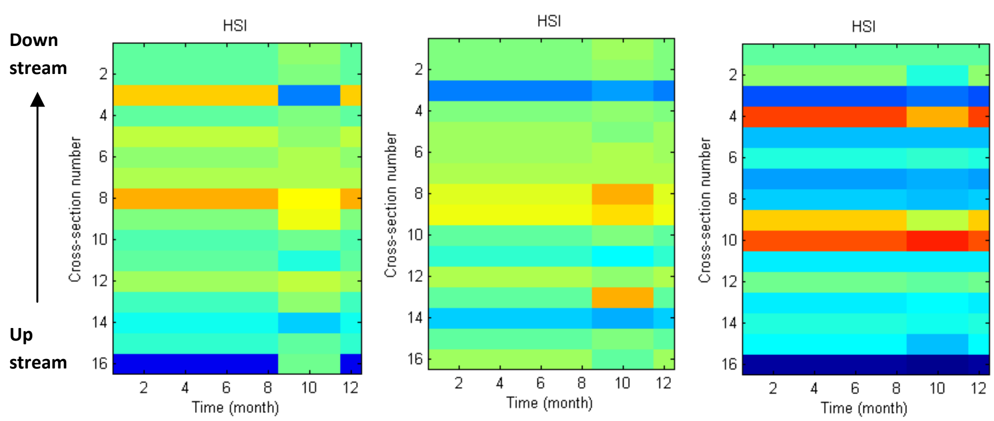
Brown trout 9:3 month flow regime

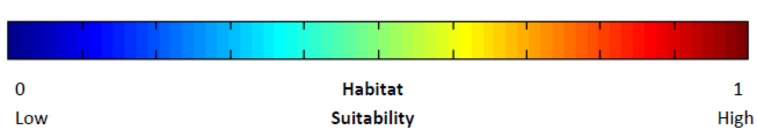


Grayling 9:3 month flow regime

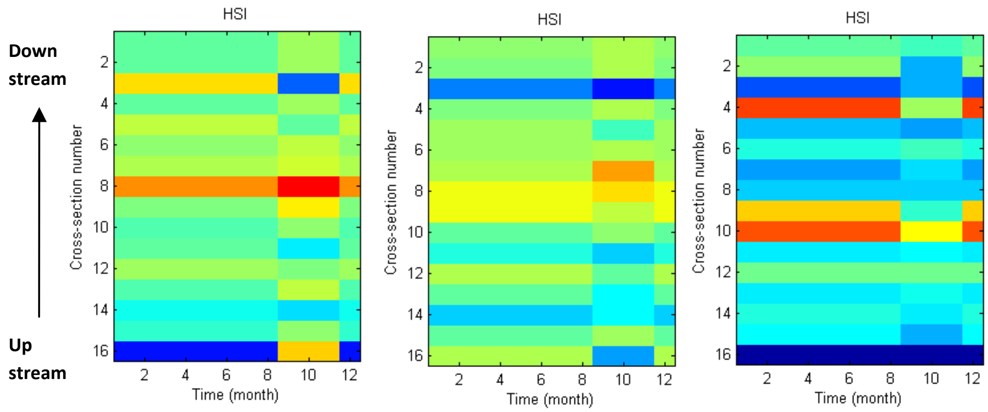


Atlantic salmon 9:3 month flow regime

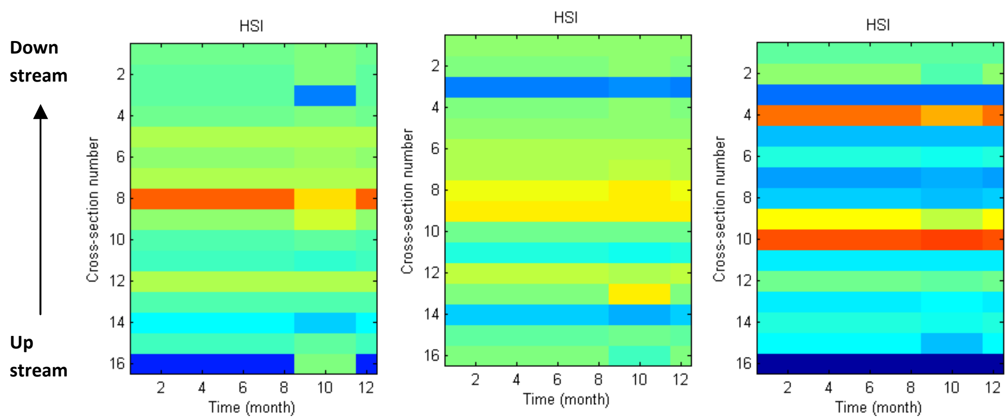




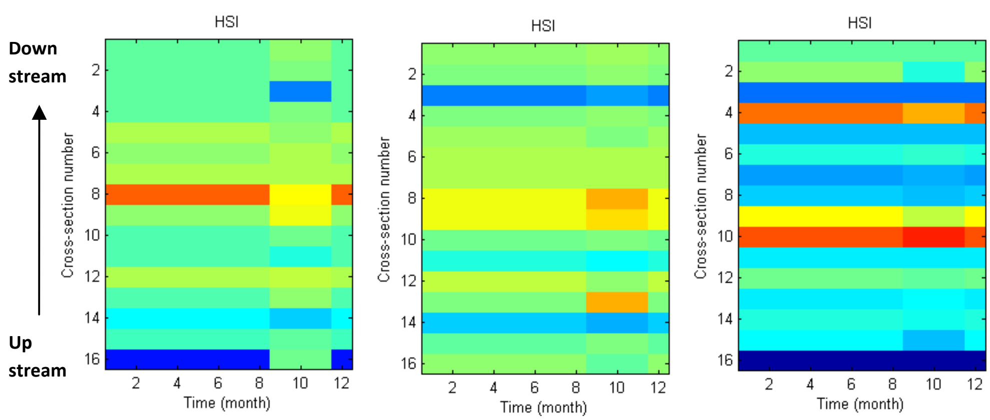
Brown trout 11:1 month flow regime

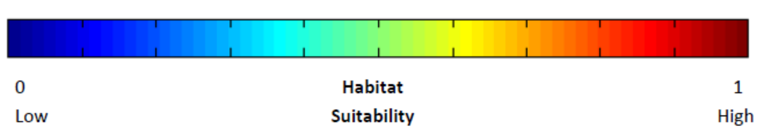


Grayling 11:1 month flow regime



Atlantic salmon 11:1 month flow regime

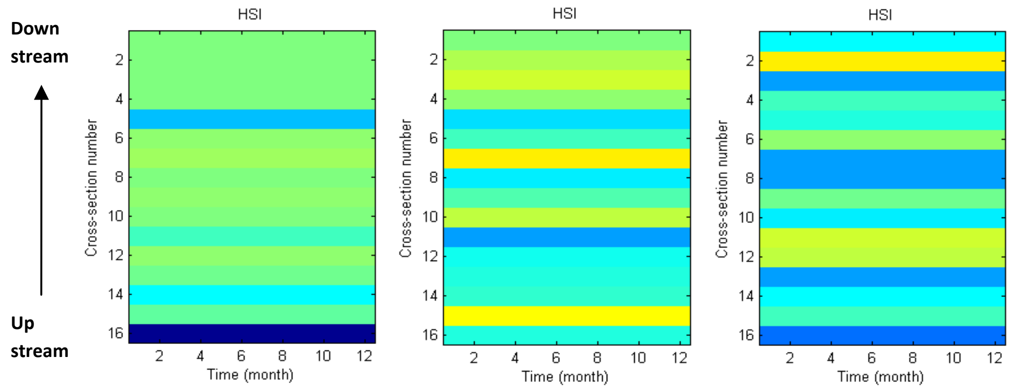




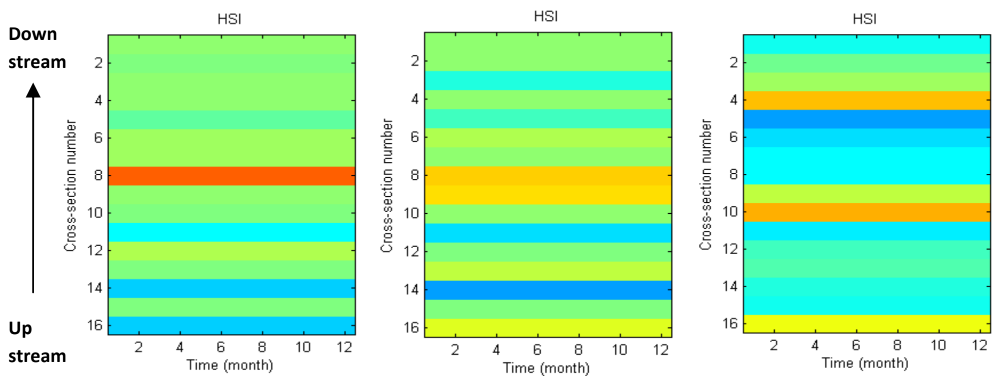
APPENDIX THREE (d) Fuzzy rule-based modelling results – Effect on juvenile grayling

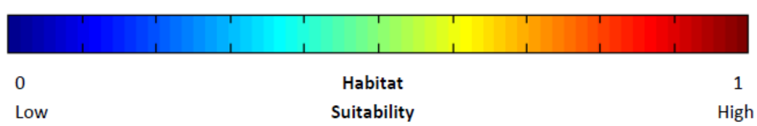
Q95 **Q50** **Q5**

Pre-impoundment flow regime

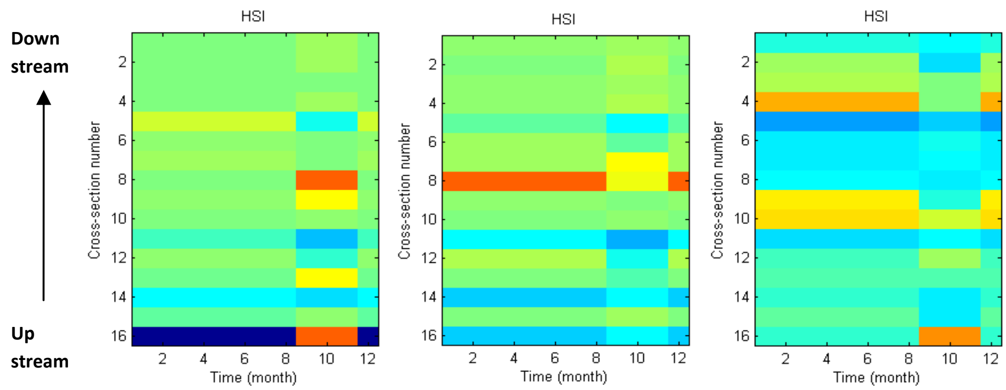


Post-impoundment flow regime

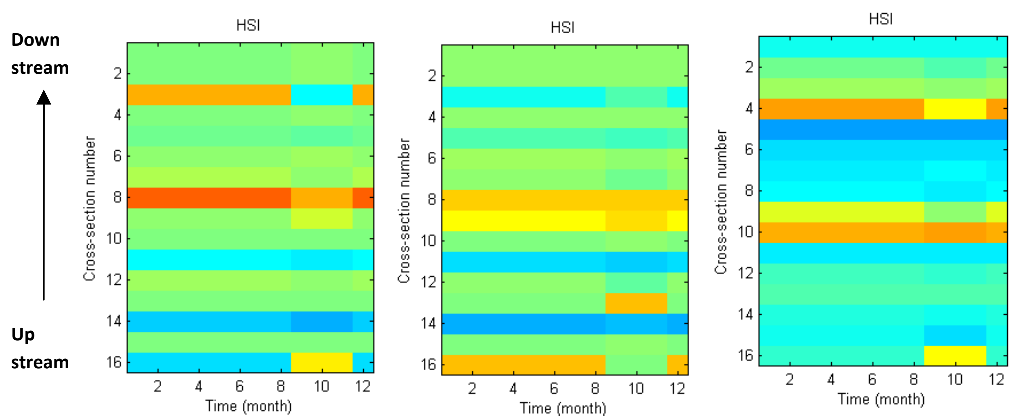




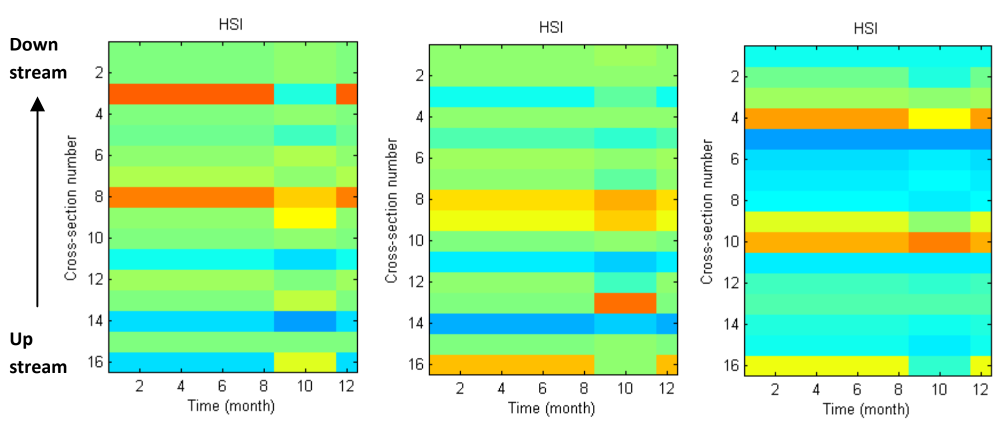
Brown trout 9:3 month flow regime

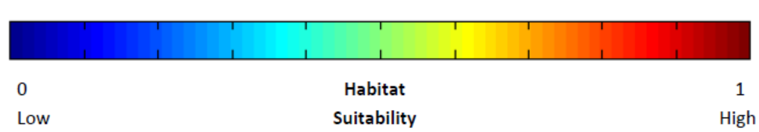


Grayling 9:3 month flow regime

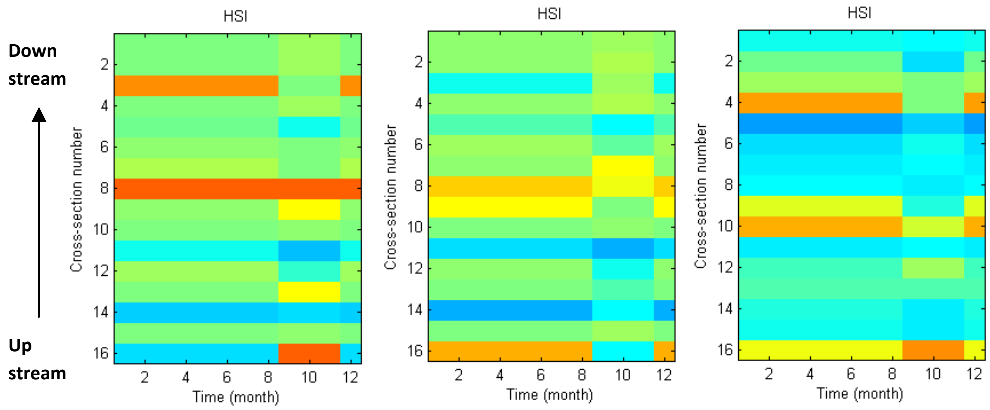


Atlantic salmon 9:3 month flow regime

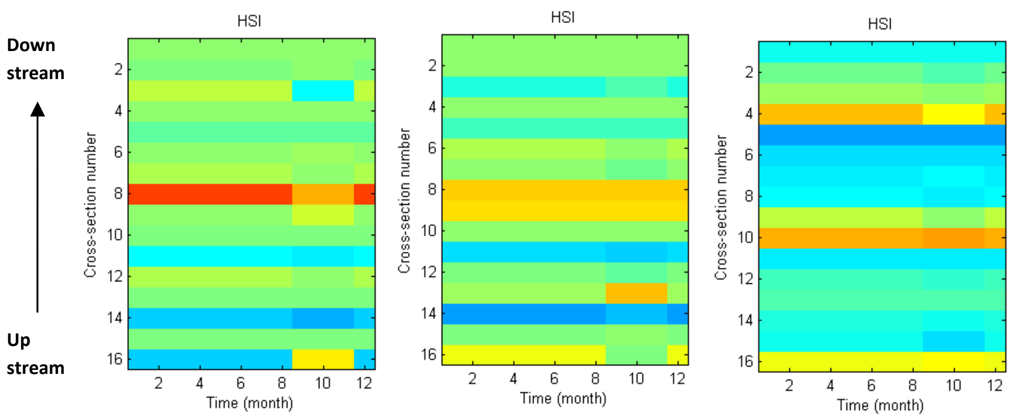




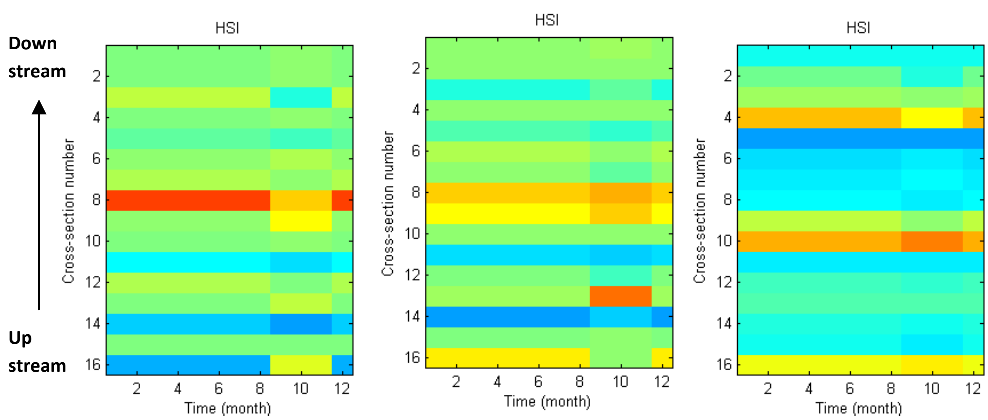
Brown trout 11:1 month flow regime

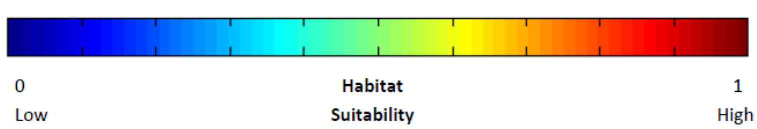


Grayling 11:1 month flow regime



Atlantic salmon 11:1 month flow regime

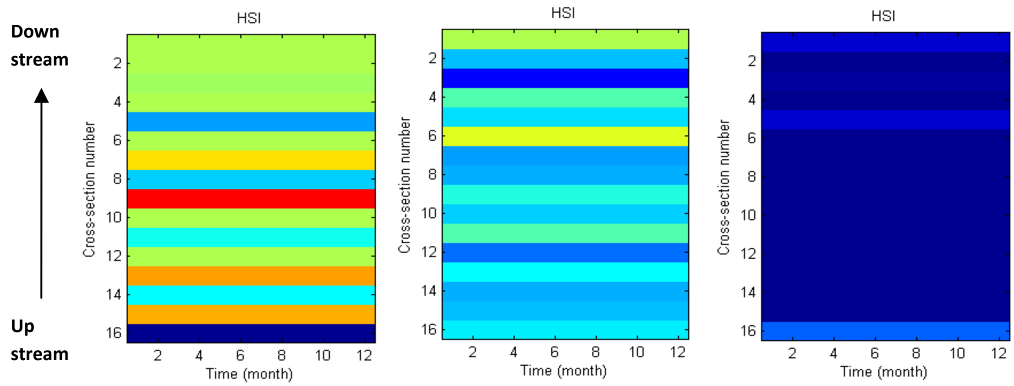




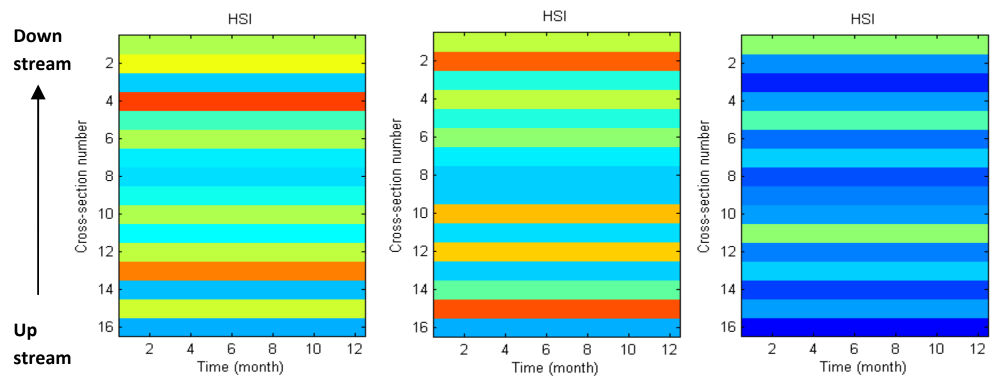
APPENDIX THREE (e) Fuzzy rule-based modelling results – Effect on spawning Atlantic salmon

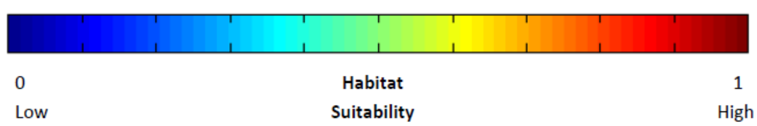
Q95 **Q50** **Q5**

Pre-impoundment flow regime

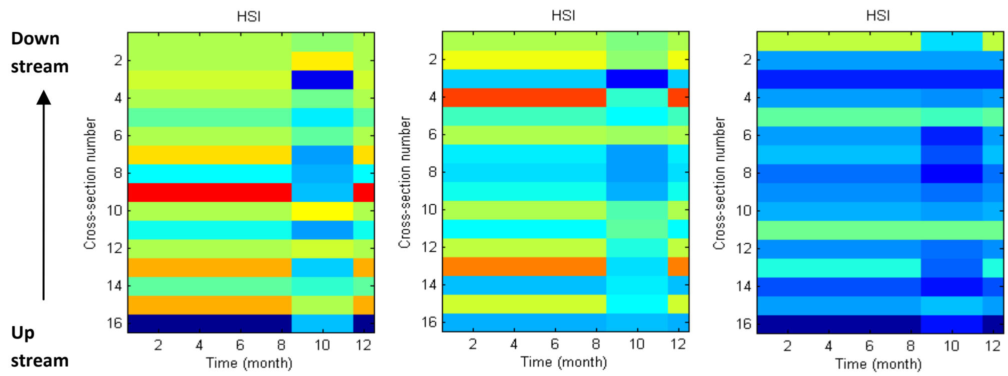


Post-impoundment flow regime

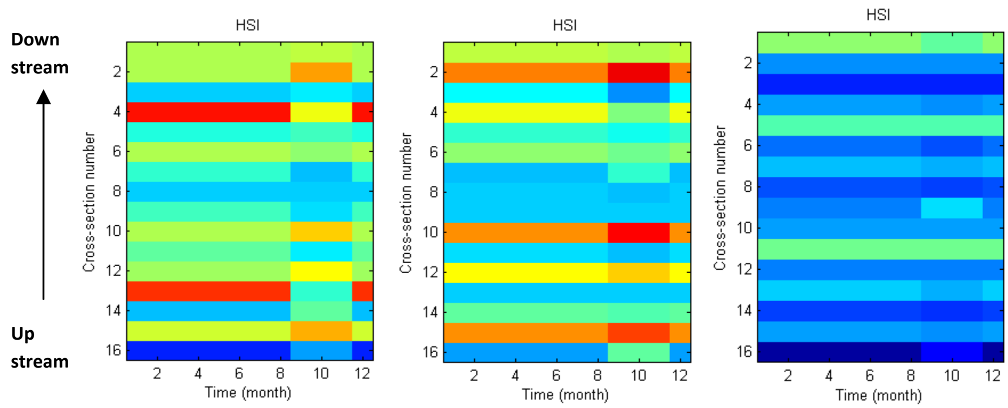




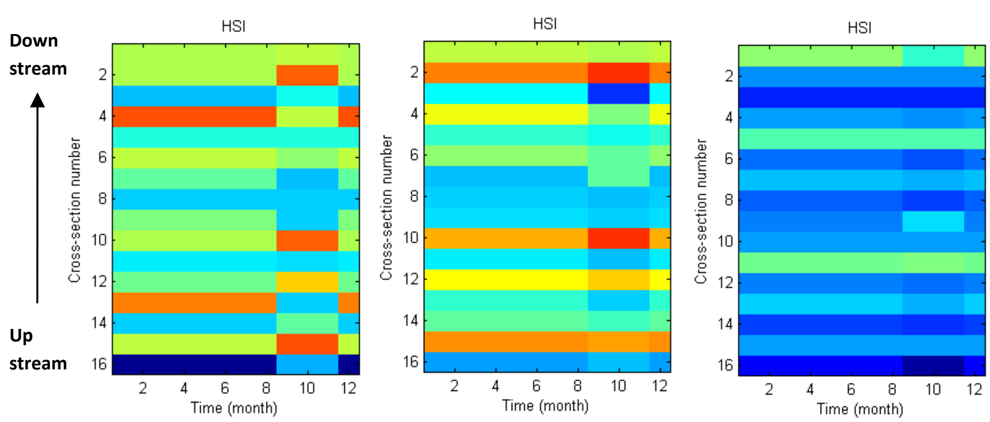
Brown trout 9:3 month flow regime

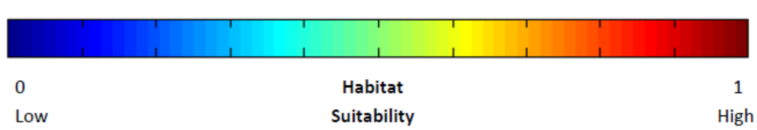


Grayling 9:3 month flow regime

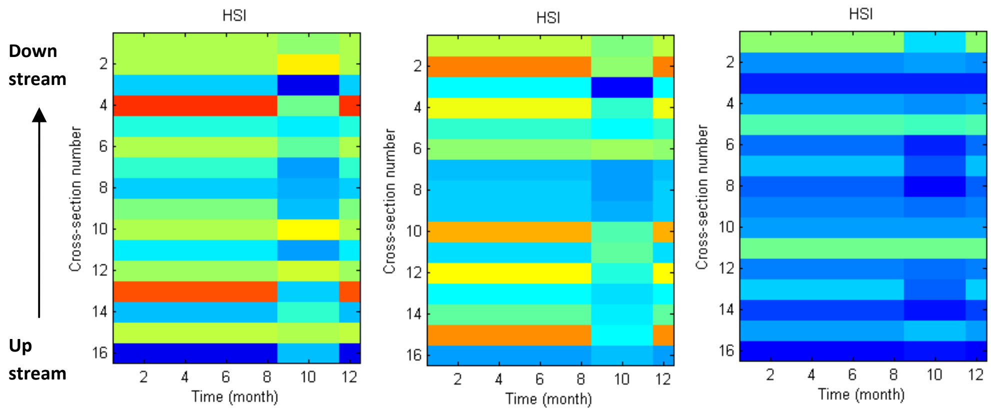


Atlantic salmon 9:3 month flow regime

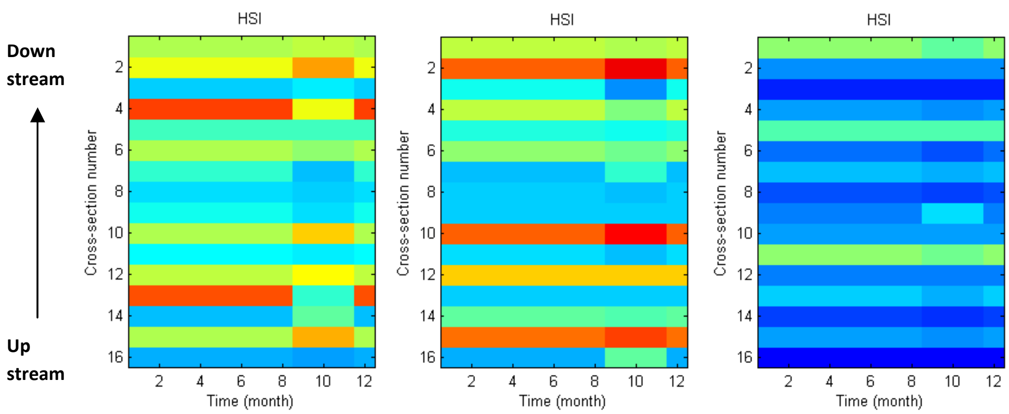




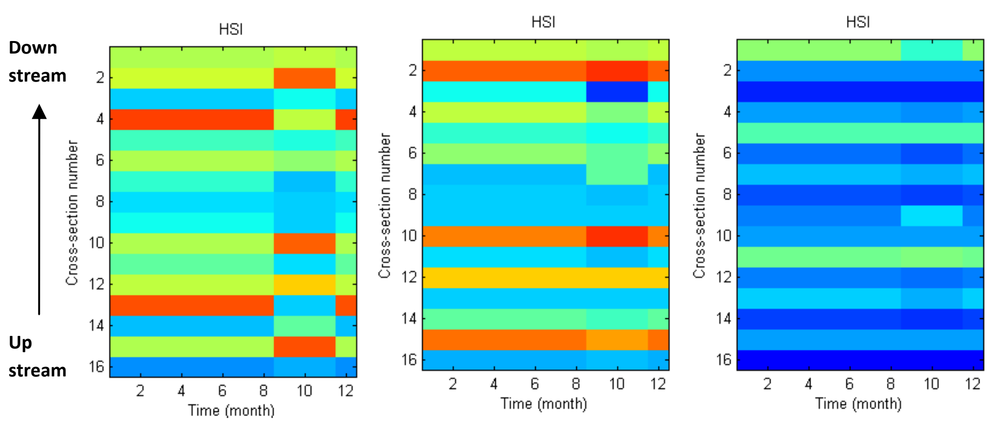
Brown trout 11:1 month flow regime

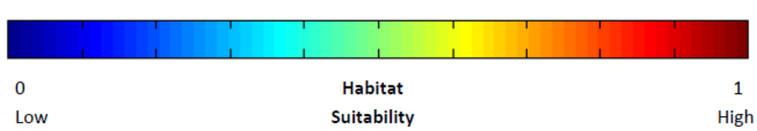


Grayling 11:1 month flow regime



Atlantic salmon 11:1 month flow regime

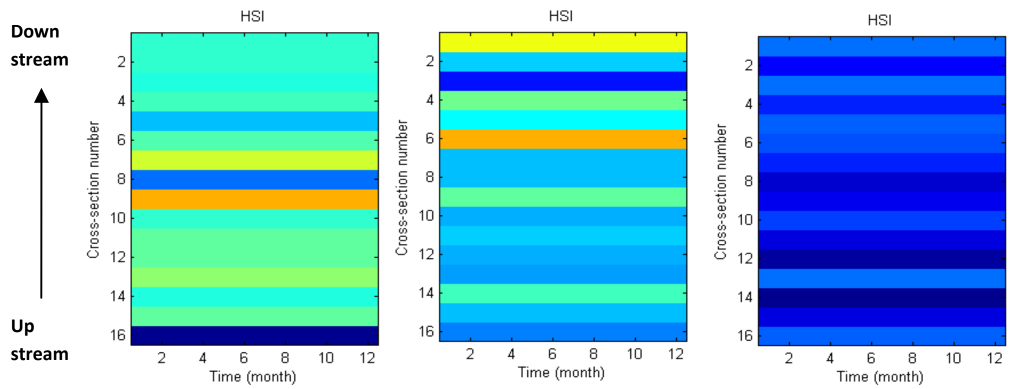




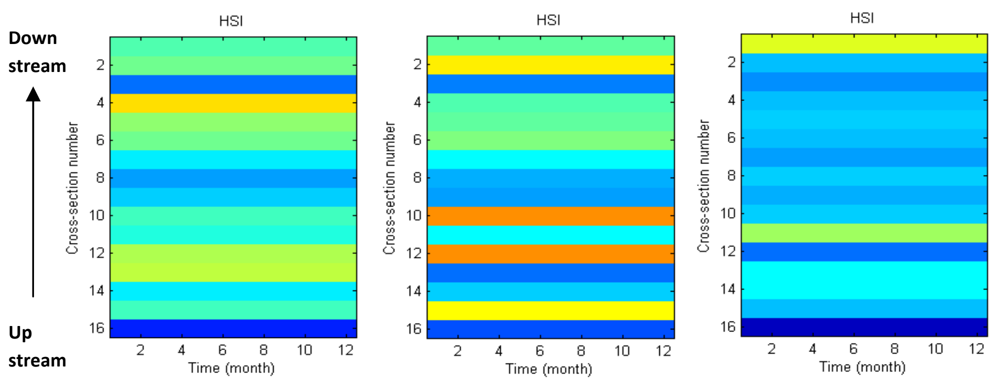
APPENDIX THREE (f) Fuzzy rule-based modelling results – Effect on juvenile Atlantic salmon

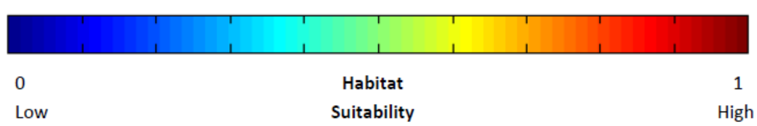
	Q95	Q50	Q5

Pre-impoundment flow regime

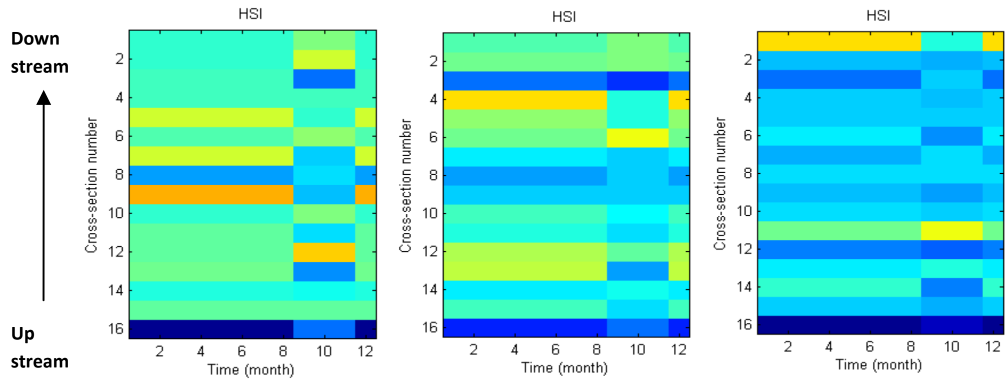


Post-impoundment flow regime

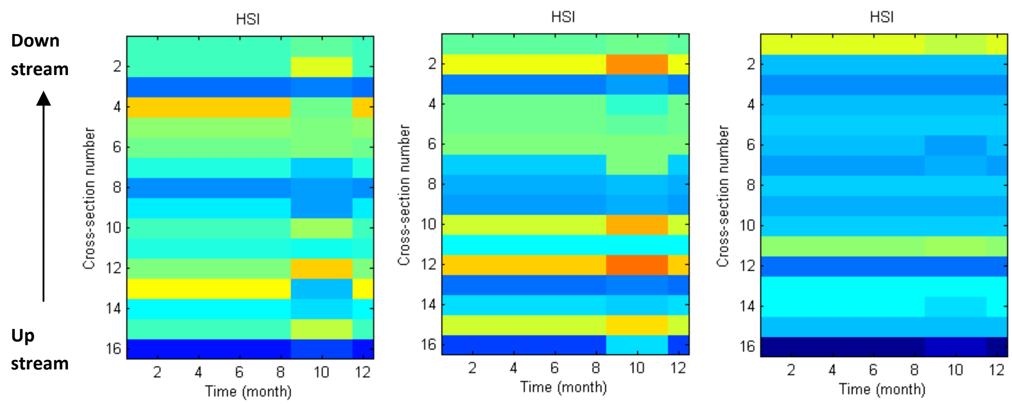




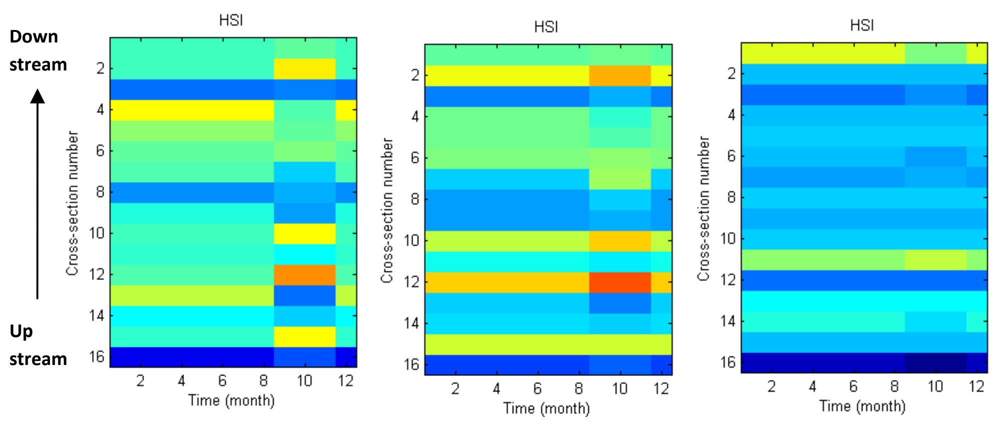
Brown trout 9:3 month flow regime

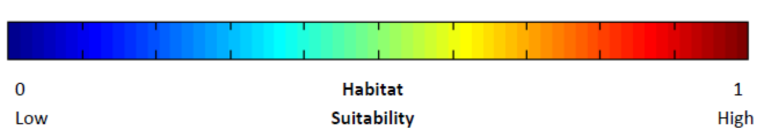


Grayling 9:3 month flow regime

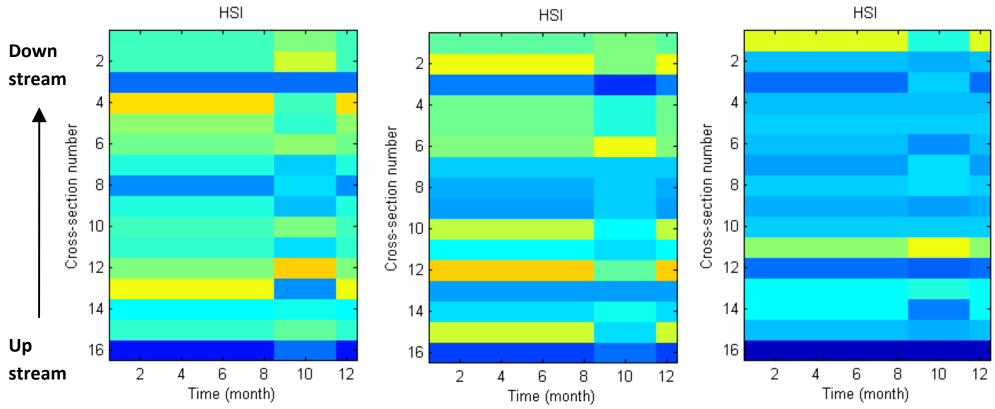


Atlantic salmon 9:3 month flow regime

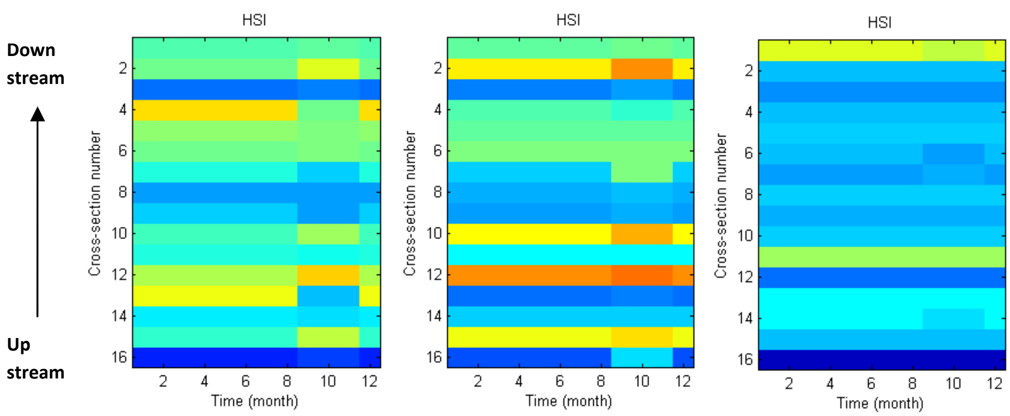




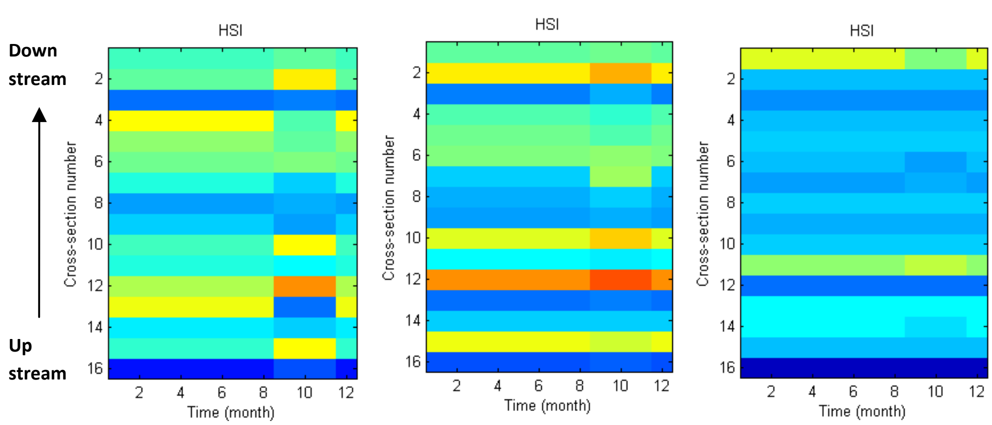
Brown trout 11:1 month flow regime

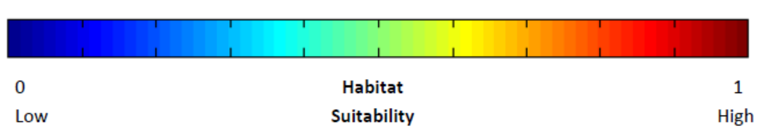


Grayling 11:1 month flow regime

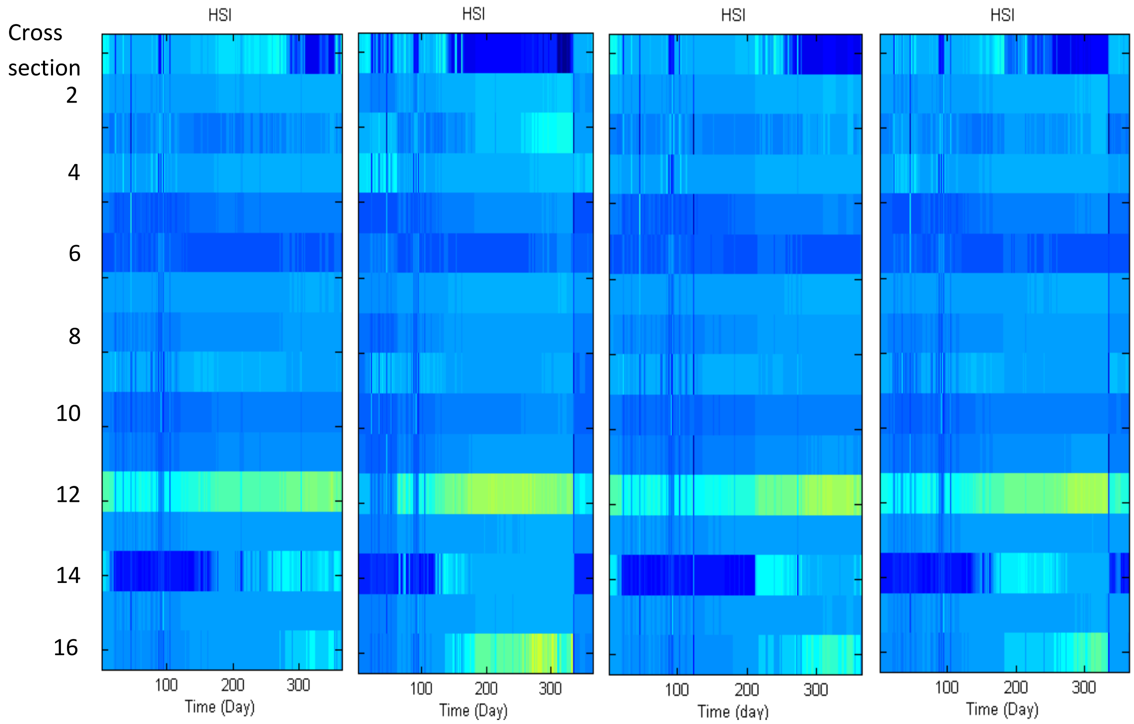


Atlantic salmon 11:1 month flow regime



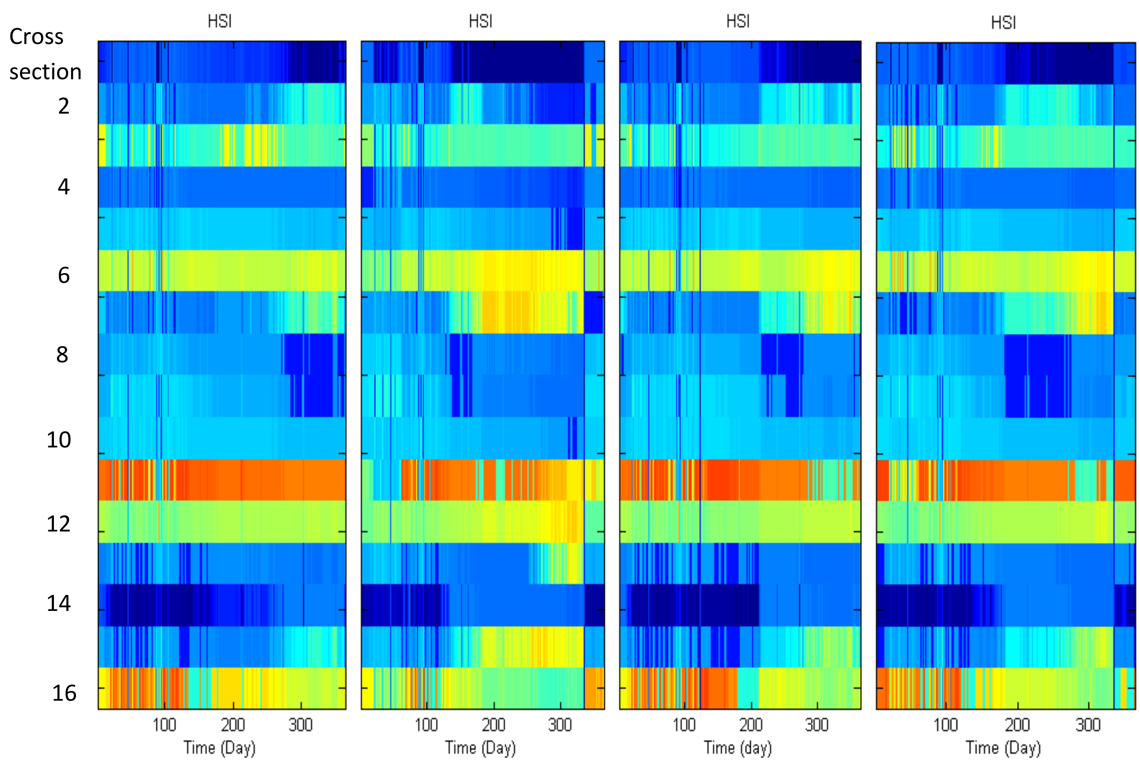


APPENDIX THREE (g) Fuzzy rule-based modelling results for annual hydrograph
Effect on spawning brown trout (cross section 1 = downstream, cross-section 16 = upstream):

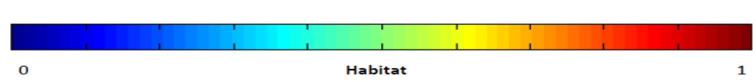


Current regime *Brown trout regime* *Grayling regime* *Atlantic salmon regime*

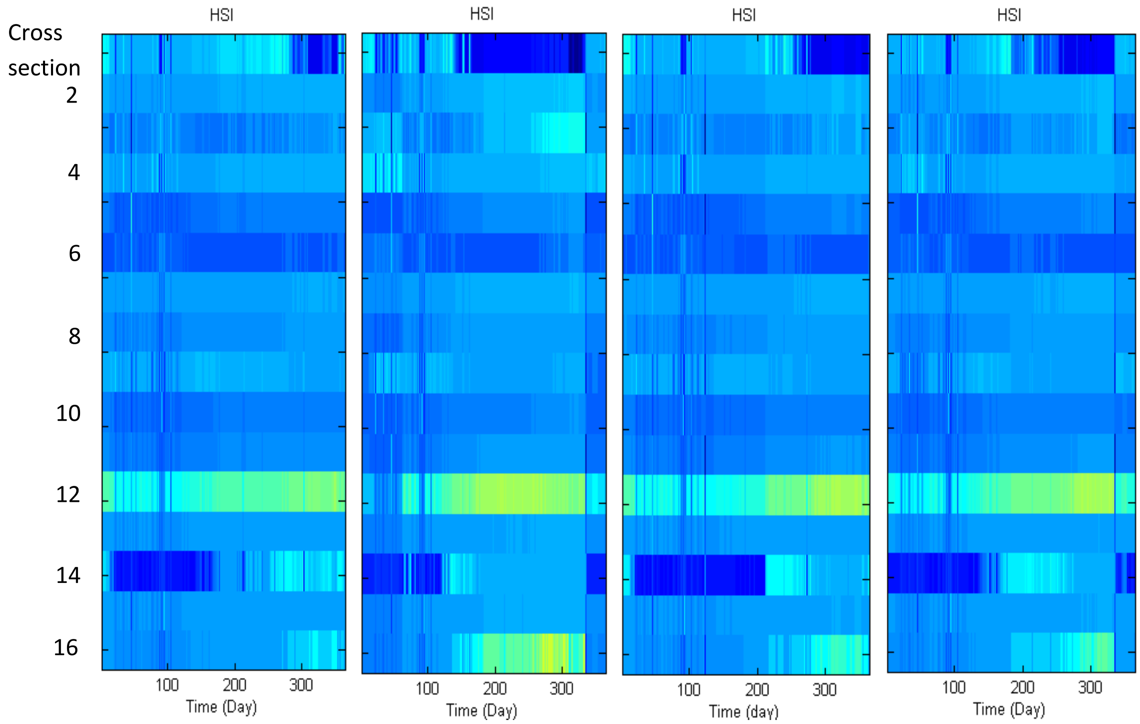
Effect on juvenile brown trout:



Current regime *Brown trout regime* *Grayling regime* *Atlantic salmon regime*

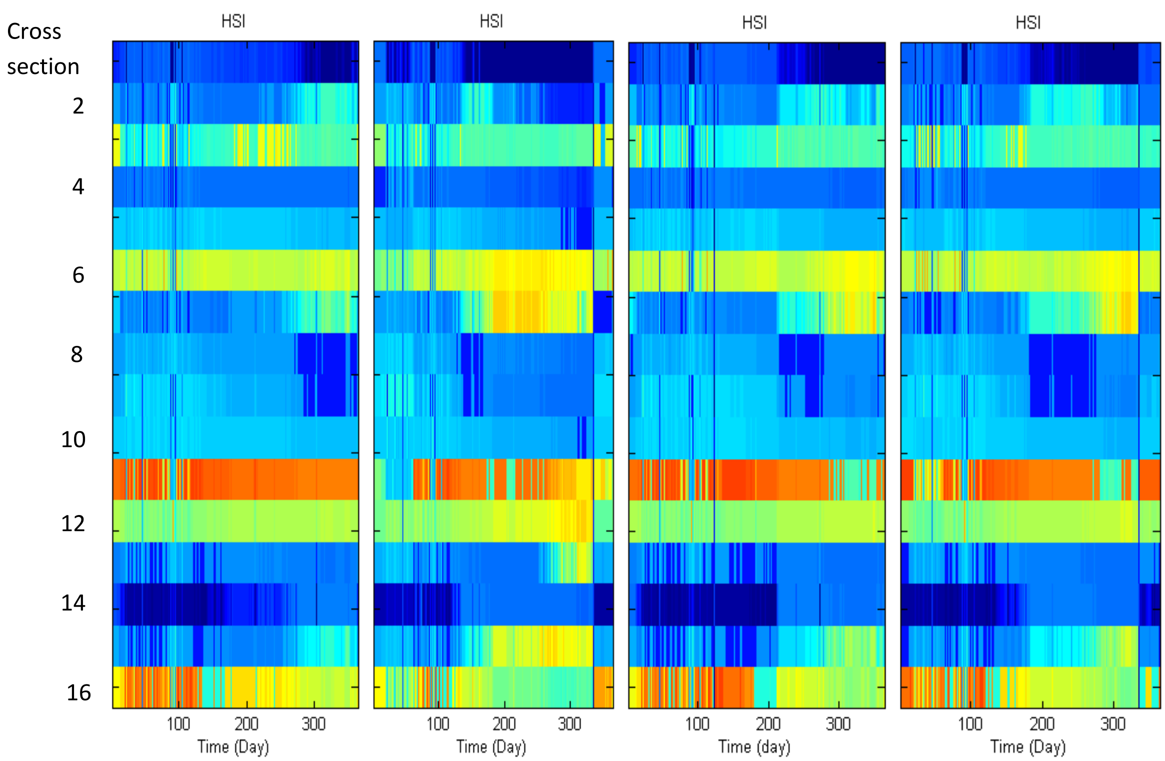


APPENDIX THREE (h) Fuzzy rule-based modelling results for annual hydrograph
Effect on spawning brown trout (cross section 1 = downstream, cross-section 16 = upstream):

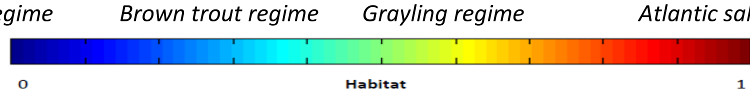


Current regime *Brown trout regime* *Grayling regime* *Atlantic salmon regime*

Effect on juvenile brown trout:

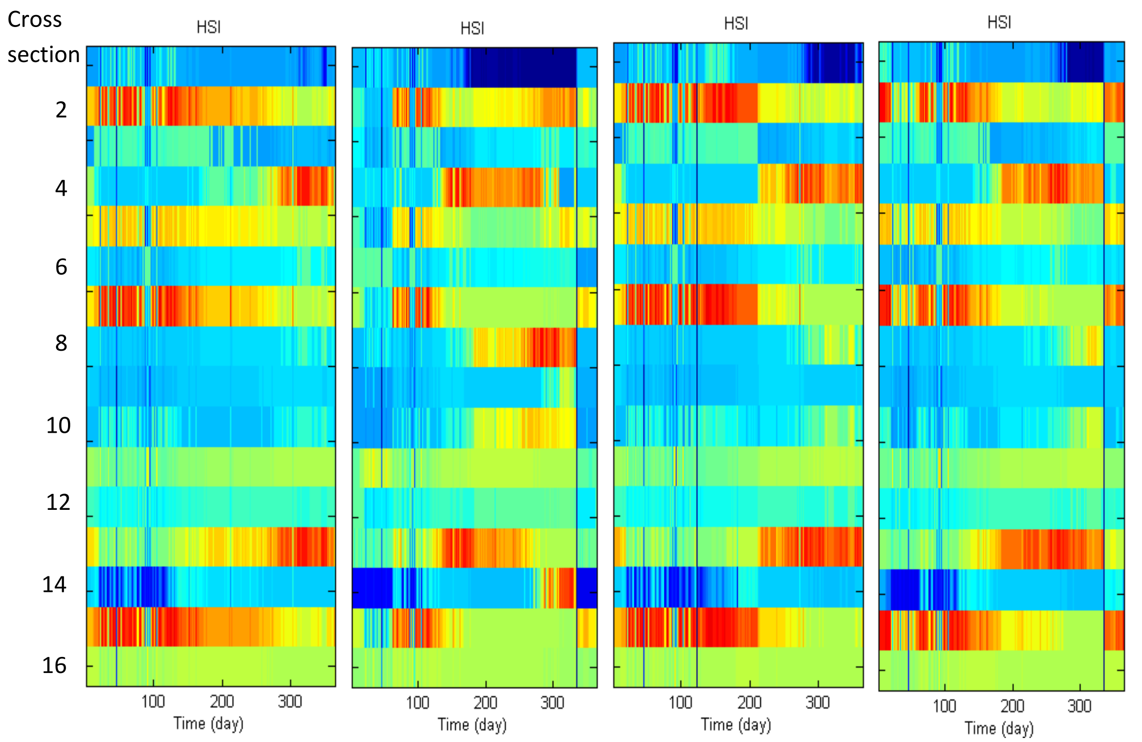


Current regime *Brown trout regime* *Grayling regime* *Atlantic salmon regime*



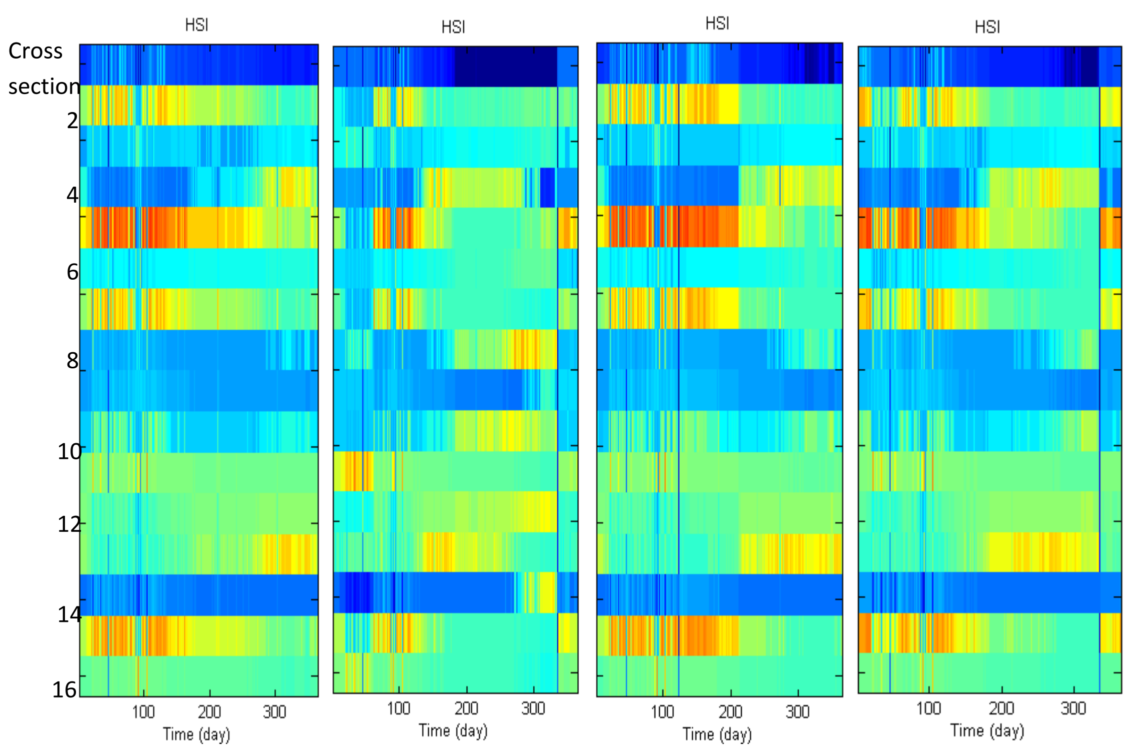
APPENDIX THREE (i) Fuzzy rule-based modelling results for annual hydrograph

Effect on spawning Atlantic salmon (cross section 1 = downstream, cross-section 16 = upstream):



Current regime *Brown trout regime* *Grayling regime* *Atlantic salmon regime*

Effect on juvenile Atlantic salmon:



Current regime *Brown trout regime* *Grayling regime* *Atlantic salmon regime*